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# Phototransformation of the fungicide tebuconazole, and its predicted fate in sunlit surface freshwaters

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#### 4 Luca Carena, Andrea Scozzaro, Monica Romagnoli, Marco Pazzi, Luca Martone,

5 Claudio Minero, Marco Minella, Davide Vione\*

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7 Dipartimento di Chimica, Università degli Studi di Torino, Via Pietro Giuria 5, 10125 Torino,
8 Italy.

9 \* Corresponding author. *davide.vione@unito.it* 

10

# 11 Abstract

12 The fungicide tebuconazole (TBCZ) is expected to undergo negligible direct photolysis in surface 13 freshwaters, but it can be degraded by indirect photochemistry. TBCZ mainly reacts with hydroxyl 14 radicals and, to a lesser extent, with the triplet states of chromophoric dissolved organic matter 15 (<sup>3</sup>CDOM\*). Indirect photochemistry is strongly affected by environmental conditions, and TBCZ 16 lifetimes of about one week are expected in sunlit surface waters under favourable circumstances 17 (shallow waters with low concentrations of dissolved organic carbon, DOC, during summer). In 18 these cases the time trend would follow pseudo-first order kinetics (mono-exponential decay). 19 Under less favourable conditions, photoinduced degradation would span over a few or several 20 months, and TBCZ phototransformation would depart from an exponential trend because of 21 changing sunlight irradiance. The TBCZ phototransformation products should be less toxic than 22 their parent compound, thereby providing potential for photodegradation to decrease the 23 environmental impact of TBCZ. Hydroxylation is a major TBCZ transformation route, due to either •OH attack, or one-electron oxidation sensitised by <sup>3</sup>CDOM\*, followed by reaction of the oxidised 24 25 transient with oxygen and water.

26

Keywords: tebuconazole; surface-water photochemistry; direct photolysis; dissolved organic
matter; transformation pathways.

29

# 30 Introduction

31

32 Tebuconazole((RS)-1-p-chloro-phenyl-4,4-dimethyl-3-(1H-1,2,4-triazol-1-ylmethyl)pentan-3-ol),

hereinafter TBCZ, is a broad-spectrum triazole fungicide that is widely used for the protection of
orchards, vegetables, cereals, and lawns (Miyauchi et al., 2005; Clausen and Yang, 2007; Lyu et al.,
2018). It is used as both seed dressing and foliar spray, and it acts on fungi by blocking the
biosynthesis of sterols. TBCZ inhibits the demethylation process that takes place on the cellular
membrane of these pathogenic organisms (Paul et al., 2008; Fera et al., 2009).

38 TBCZ has moderate to low toxicity towards mammals, but it is peculiarly toxic to aquatic life forms 39 (EFSA, 2008). Moreover, it is classified as type C human carcinogen (possibly carcinogenic to 40 humans) (Želonková et al., 2019), and has potential to cause endocrine-disrupting effects (Yu et al., 41 2013). The persistence of TCBZ in soil is highly variable (Muñoz-Leoz et al., 2011), and TBCZ 42 might also be resistant to biodegradation in the aquatic environment (Boethling et al., 2004). TBCZ has been detected in surface waters at ng  $L^{-1}$  levels, and even in wastewater at some tens ng  $L^{-1}$ , 43 44 although there are arguably additional routes than wastewater for TBCZ to enter the water 45 environment (most notably, runoff from soil as well as urban areas) (Kahle et al., 2008).

Past studies have excluded direct photolysis, as a significant transformation route of TBCZ in surface waters, due to insignificant absorption of sunlight (Coody, 1987). The direct photolysis is the transformation of a compound triggered by photon absorption (Yan and Song, 2014) but, although it can be an important process for some substances, it is not the only phototransformation pathway for a contaminant in surface waters. Indirect photochemistry is another potentially important route, in which sunlight is absorbed by naturally occurring compounds called 52 photosensitisers (Remucal, 2014). The main photosensitisers in surface waters are nitrate, nitrite, 53 and the chromophoric dissolved organic matter (CDOM) (Dong and Rosario-Ortiz, 2012; Mostafa 54 and Rosario-Ortiz, 2013; Vione and Scozzaro, 2019). Radiation absorption by photosensitisers triggers the generation of photochemically produced reactive intermediates (PPRIs), such as the 55 hydroxyl radicals (•OH), singlet oxygen (<sup>1</sup>O<sub>2</sub>), and CDOM triplet states (<sup>3</sup>CDOM\*). PPRIs can 56 57 react with contaminants. and cause their degradation (Vione et al., 2014). Moreover, the oxidation of  $HCO_3^{-}/CO_3^{2-}$  by •OH, and of  $CO_3^{2-}$  by <sup>3</sup>CDOM\* produces the carbonate radical,  $CO_3^{\bullet-}$ , as a 58 further PPRI (Canonica et al., 2005; Yan et al., 2019). The different PPRIs reach relatively low 59 steady-state concentrations in surface waters  $(10^{-18} - 10^{-13} \text{ M})$  (Rosario-Ortiz and Canonica, 2016), 60 61 due to the budget between photogeneration and scavenging/quenching (Vione et al., 2014). Indeed, 62 •OH is generated by irradiation of nitrate, nitrite, and CDOM, and it is scavenged by DOM (dissolved organic matter, not necessarily chromophoric), inorganic carbon ( $HCO_3^{-}/CO_3^{2-}$ ), as well 63 64 as bromide in saltwater and seawater (Gligorovski et al., 2015). DOM is also the main scavenger for CO<sub>3</sub><sup>•-</sup> (Yan et al., 2019), but it is unable to quench significant fractions of <sup>3</sup>CDOM\* or <sup>1</sup>O<sub>2</sub>, unless 65 the values of the water dissolved organic carbon (DOC) are very high (Cory et al., 2009; Wenk et 66 67 al., 2013). Therefore, the main <sup>3</sup>CDOM\* sink is dissolved O<sub>2</sub>, and the reaction between <sup>3</sup>CDOM\* and O<sub>2</sub> is the only significant <sup>1</sup>O<sub>2</sub> source (~50% yield) (McNeill and Canonica, 2016). Finally, <sup>1</sup>O<sub>2</sub> 68 is mainly quenched by collision with water (Wilkinson and Brummer, 1981). 69

Although DOM is usually unable to significantly scavenge <sup>3</sup>CDOM\*, its antioxidant (mostly phenolic) moieties can inhibit the photodegradation of some contaminants, which have initially been oxidized by <sup>3</sup>CDOM\* and/or CO<sub>3</sub><sup>--</sup> (Canonica and Laubscher, 2008; Carena et al., 2022). Indeed, <sup>3</sup>CDOM\* and CO<sub>3</sub><sup>--</sup> degrade contaminants by electron or H-atom abstraction and, depending on reduction potential and reaction kinetics, the transient species resulting from initial pollutant oxidation may be back-reduced to the starting compound by the phenolic moieties contained in DOM (Wenk et al., 2011; Wenk and Canonica, 2012; Hao et al, 2020; Carena et al.,
2022).

78 The reaction between PPRIs and contaminants is usually a minor pathway for PPRI dissipation, but 79 it can have a key role in contaminant fate (Remucal, 2014; Vione et al., 2014; Baena-Nogueras et 80 al., 2017). Indeed, indirect photochemistry adds to the direct photolysis in photodegradation of 81 sunlight-absorbing contaminants, and it is the only possible phototransformation route for 82 compounds that do not absorb sunlight. It is actually very rare for contaminants not to undergo 83 indirect photodegradation (most compounds should react at least with the very strong oxidant 'OH; 84 Buxton et al., 1988). However, the kinetics and environmental importance of the indirect 85 photochemistry process(es) have to be assessed on a case-by-case basis.

86 To this purpose, the main goal of this work is the assessment of the overall photodegradation 87 kinetics and pathways of TBCZ, with a particular emphasis on indirect photochemistry. The 88 assessment is carried out by a combination of photodegradation experiments and photochemical 89 modelling. The latter is important, on the one side, to understand which photodegradation lifetimes 90 can be expected for TBCZ in different environmental conditions; on the other hand, modelling 91 gives insight into the peculiar photochemical pathways that are more important for TBCZ in the 92 environment, and that may produce key transformation products. By so doing, product 93 identification can be focused on the main expected phototransformation routes (Gornik et al., 2021).

#### 95 Materials and methods

96

## 97 Chemicals and solvents

98 Chemicals used in this study were of analytical grade. Solvents and eluents were of gradient-grade 99 purity for liquid chromatography. A detailed list that specifies purity grade and supplier for each 100 reagent is provided in **Text S1 (SM)**. Ultra-pure water was of Milli-Q quality (18.5 M $\Omega$  cm, TOC < 101 2 ppb).

102

#### 103 Irradiation experiments

104 Different irradiation set-ups and different solutions were used for the study of TBCZ 105 photoreactivity. Direct photolysis was studied by irradiating TBCZ alone under a UVB lamp 106 (Philips narrow band TL 20W/01, emission maximum at 313 nm). The same lamp was used to 107 study the reaction kinetics between TBCZ and •OH, by irradiating TBCZ together with  $H_2O_2$  as 108 •OH source. The •OH scavenger 2-propanol was added at different concentration values, to achieve 109 competition kinetics with TBCZ, and determine the second-order reaction rate constant between 110 TBCZ and •OH.

The reaction between TBCZ and CO<sub>3</sub><sup>•-</sup> was studied by UVB irradiation of mixtures of TBCZ + 111 NaNO<sub>3</sub> + NaHCO<sub>3</sub>, TBCZ + NaNO<sub>3</sub> + phosphate buffer (at the same total concentration and pH as 112 the NaHCO<sub>3</sub>-containing solutions), and TBCZ + NaHCO<sub>3</sub> (blank experiment, to check for the direct 113 114 photolysis of TBCZ). The initial concentration values were: 20 µM TBCZ; 10 mM NaNO<sub>3</sub> (when present), as well as 0, 5, or 11 mM NaHCO<sub>3</sub> (or phosphate buffer). The rationale here is that 115 116 irradiated NaNO<sub>3</sub> yields •OH, which in the presence of NaHCO<sub>3</sub> oxidises HCO<sub>3</sub><sup>-/</sup>CO<sub>3</sub><sup>2-</sup> to CO<sub>3</sub><sup>--</sup>. The comparison of TBCZ degradation rates, in the three different systems, indicates whether or not 117 the reaction with  $CO_3^{\bullet-}$  is significant (Vione et al., 2009). The choice of nitrate instead of  $H_2O_2$  in 118 119 this series of experiments is motivated by the need of exploiting a solvent-cage effect, which involves the geminate species produced by nitrate photolysis (Vione et al., 2011, and see theSupplementary Material for further details).

The triplet photosensitiser 4-carboxybenzophenone (CBBP, in its deprotonated anionic form) was used as CDOM proxy, due to similar reactivity between the triplet states <sup>3</sup>CBBP\* and <sup>3</sup>CDOM\* (Avetta et al., 2016). Irradiation of mixtures of TBCZ and CBBP was carried out under a UVA black lamp (Philips TL-D 18 W, with emission maximum at 369 nm), under which conditions CBBP can be selectively excited.

127 The reaction between TBCZ and  ${}^{1}O_{2}$  was studied by using the dye Rose Bengal as  ${}^{1}O_{2}$  source, upon 128 irradiation under a yellow lamp (TL D 18W/16 Yellow). Excited Rose Bengal in its triplet state 129 transfers energy to ground-state O<sub>2</sub>, to produce  ${}^{1}O_{2}$  (DeRosa and Crutchley, 2002).

Chemical actinometry with 2-nitrobenzaldehyde (Carena et al., 2019; Galbavy et al., 2010; Willett
and Hites, 2000) was used, together with the lamp emission spectra (taken with an Ocean Optics
USB 2000 CCD Spectrophotometer) to determine the spectral photon flux density occurring in the
irradiated solutions. The emission spectra thus obtained for the three lamps are reported in Figure
S1 (SM).

During irradiation, the aqueous solutions (5 or 20 mL, pH ranging from 6 to 7 except for the NaNO<sub>3</sub>/HCO<sub>3</sub><sup>-</sup> (CO<sub>3</sub><sup>•-</sup>) experiments, where it reached up to 8.5) were introduced into cylindrical Pyrex glass cells, tightly closed with a screw cap, and magnetically stirred. Pyrex glass has a cut-off wavelength around 290 nm, thus it transmits >50% radiation above 300 nm (see **Figure S2 (SM)**). Anyway, radiation absorption by Pyrex was taken into account in actinometry experiments.

140

#### 141 Monitoring of TBCZ photodegradation

After the scheduled irradiation time, each cell was withdrawn from the lamp, and its contents were
analysed by high-performance liquid chromatography coupled with diode array detection (HPLCDAD), to quantify residual TBCZ. The used instrument was a VWR-Hitachi Chromaster, equipped
with 5260 autosampler (60 μL injection volume), 5160 quaternary pump, and 5430 DAD detector.

146 The column was a Merck LiChroCART RP-18 cartridge (125 mm  $\times$  4 mm  $\times$  5  $\mu$ m). Isocratic 147 elution was carried out with a 75:25 mixture of methanol and acidified water (H<sub>3</sub>PO<sub>4</sub>, pH 2.8). The 148 eluent flow rate was 1 mL min<sup>-1</sup>, which under the reported conditions produced a TBCZ retention 149 time of 5.4 min. The detection wavelength was 221 nm.

The TBCZ time trends followed pseudo-first order kinetics, and could be fitted with the equation  $C_{\text{TBCZ}} = C^{\circ}_{\text{TBCZ}} e^{-k t}$ , where  $C_{\text{TBCZ}}$  is the concentration of TBCZ at the time t,  $C^{\circ}_{\text{TBCZ}}$  the initial TBCZ concentration, and k its pseudo-first order photodegradation rate constant. The initial rate of TBCZ photodegradation was  $R_{\text{TBCZ}} = k C^{\circ}_{\text{TBCZ}}$ .

154

#### 155 Identification of TBCZ phototransformation products

TBCZ phototransformation products were identified by high-performance liquid 156 The 157 chromatography, coupled with diode array detection and mass spectrometry (HPLC-DAD-MS). A Thermo Scientific Accela chromatograph was used, equipped with Accela autosampler (20 µL 158 159 injection volume), a quaternary Accela pump, and DAD Accela PDA detector. The column used was a Phenomenex Synergi Polar-RP 80A cartridge (150 mm  $\times$  2 mm  $\times$  4  $\mu$ m), which was 160 161 maintained at 30 °C. The mass spectrometer was a Thermo LCQ Fleet (ion-trap) Mass 162 Spectrometer, equipped with electrospray ionisation (ESI). Elution was carried out in binary 163 gradient mode. The mobile phase consisted of formic acid 0.1% (v/v; eluent A), and methanol (eluent B). The following gradient program (time, %A/%B) was applied: 0 min, 95/5; 20 min, 164 0/100; 30 min, 0/100; 30.1 min, 95/5; 35 min, 95/5. The flow rate was 0.2 mL min<sup>-1</sup>. The DAD 165 166 detection wavelength was 221 nm.

167 The initial Full Scan analysis was carried out in positive ion mode. It gave the characteristic ions, 168 on which the successive MS<sup>n</sup> analyses were carried out. The collision energy was chosen each time 169 between 25 or 35 eV, and the different values used for the identification of each relevant ion are 170 reported in **Table S1 (SM)**.

#### 172 Modelling of TBCZ phototransformation in surface freshwaters

173 TBCZ photodegradation kinetics and pathways in surface freshwaters were modelled with the 174 APEX software (Bodrato and Vione, 2014), which has proven its suitability to assess the 175 photochemical fate of contaminants in environmental aqueous systems (Avetta et al., 2016). APEX requires as input data both the chemical and photochemical features of a water body (i.e., 176 177 concentration of photosensitisers and reactive species scavengers, water absorption spectrum and 178 water depth, of which reasonable values were assumed), and the photoreactivity parameters of the 179 considered contaminant (UV-visible absorption spectrum, direct photolysis quantum yield, and second-order reaction rate constants with •OH, CO<sub>3</sub>•-, <sup>1</sup>O<sub>2</sub>, and <sup>3</sup>CDOM\*). The output values of 180 181 photodegradation kinetics (pseudo-first order photodegradation rate constants, k, and half-life times, 182  $t_{1/2}$ ) are average values over a water column of given depth (Bodrato and Vione, 2014). The main 183 goal of the irradiation experiments, carried out in this work, was to provide the contaminant input 184 data for APEX (TBCZ quantum yield for direct photolysis, as well as second-order reaction rate 185 constants with PPRIs; see Table S2 (SM)), not to directly simulate environmental photochemistry conditions. 186

The APEX software was initially run supposing a clear-sky scenario, corresponding to July, 15<sup>th</sup>, at 187 188 45°N latitude. In this way, the time unit obtained as output corresponds to a 24-h day, which takes 189 the day-night cycle into account. Seasonal variability of photodegradation was obtained by means 190 of the APEX Season function (Bodrato and Vione, 2014), which returns first-order 191 photodegradation rate constants in different months of the year. The latter approach has interesting 192 implications for the time trend of a contaminant. Actually, quite complete (~95%) contaminant 193 photodegradation can be achieved in one month, if the first-order lifetime is around a week  $(t_{1/2} \sim 7)$ days, which corresponds to  $k \sim 0.1 \text{ day}^{-1}$ ). In case of longer lifetimes, photodegradation continues in 194 195 the following month(s), when the pollutant experiences different conditions of sunlight irradiance. 196 This issue decreases the accuracy of a description, based on a single (and monthly-based) first-order 197 photodegradation rate constant. It has been shown recently (Vione, 2021) that the resulting time trend can be well approximated by a series of first-order tracts, each lasting for one month: in fact, in a month's time the irradiance conditions do not vary much, under the hypothesis of consistent fine weather. The approximated trend is described by the following equation (Vione, 2021):

201 
$$C_t / C_o = \prod_{m=0}^M e^{-k_m \,\Delta t_m}$$
 (1)

202 where m = 0 is the initial month of pollutant emission, m a generic month, M the total number of months included in the simulation,  $k_m$  (day<sup>-1</sup> units, as returned by *APEX-season*) the pseudo-first 203 204 order degradation rate constant of the contaminant in the month m, and  $\Delta t_{\rm m} = 28, 30$ , or 31 days, 205 depending on the month. Moreover,  $C_0$  is the initial contaminant concentration (m = 0), and  $C_t$  is the 206 concentration at the end of the month M. In these simulations, it was assumed that water chemistry 207 and depth did not vary over time. The rationale for this assumption is that seasonal variations of the 208 irradiance play a much more important role in the photodegradation of contaminants, than the 209 typical, corresponding variations in water chemistry and depth (Vione, 2021).

The ECOSAR software (Mayo-Bean et al., 2012) was finally used, to assess the acute and chronic toxicity of TBCZ, and of its detected phototransformation products. By means of a quantitative structure-activity relationship approach, ECOSAR can predict the acute and chronic toxicity (LC50, EC50, chronic values ChV) for freshwater organisms such as fish, daphnid (crustaceans), and algae. Concerning accuracy, a compound can be said to be more toxic than another when the predicted values differ by at least an order of magnitude (Mayo-Bean et al., 2012).

#### 217 **Results and Discussion**

218

#### 219 Direct photolysis of TBCZ

Solutions (20 mL) containing 5  $\mu$ M TBCZ were irradiated for up to 70 h under the UVB lamp. The choice of this lamp was motivated by the fact that TBCZ absorbs sunlight mostly in the UVB region, and with very low absorption coefficients (around or below 1 M<sup>-1</sup> cm<sup>-1</sup> at  $\lambda$  > 300 nm, see **Figure 1**). While previous reports have indicated that TBCZ does not absorb sunlight (Coody, 1987), our results suggest that sunlight absorption by TBCZ would be, if not totally negligible, at least extremely low.





Figure 1. Left Y-axis: absorption spectrum of TBCZ (molar absorption coefficient,  $\varepsilon_{\text{TBCZ}}(\lambda)$ ). Right y-axis: emission spectrum of the Philips TL 20W/01 UVB lamp (spectral photon flux density,  $p^{\circ}(\lambda)$ ).

230

The initial degradation rate of TBCZ in these conditions was  $R_{\text{TBCZ}} = (3.46\pm0.40) \times 10^{-12} \text{ M s}^{-1}$ (mean ± standard error of experiments run in duplicate). The fraction of TBCZ degraded after 70 h continuous irradiation was as low as 17%, which indeed suggests a limited importance of the direct photolysis process. At the same time, negligible degradation was detected in dark experiments where 5  $\mu$ M TBCZ was placed under the same lamp, using the same irradiation cells wrapped in aluminium foil, to attain comparable temperature and stirring conditions as for the irradiation experiments.

By excluding dark processes such as hydrolysis, in agreement with previous reports (Wiche and Bogdoll, 2007), the degradation of TBCZ observed under irradiation could be attributed to direct photolysis alone. The direct photolysis quantum yield of TBCZ can be calculated as the ratio between  $R_{\text{TBCZ}}$  and the photon flux absorbed by TBCZ, as follows (Braslavsky, 2007):

242 
$$\Phi_{TBCZ} = \frac{R_{TBCZ}}{\int_{\lambda} p^{\circ}(\lambda) [1 - 10^{-\varepsilon_{TBCZ}(\lambda) b \ C^{\circ}_{TBCZ}}] d\lambda} = 0.22 \pm 0.03$$
(2)

where  $R_{TBCZ}$  is the initial degradation rate of TBCZ by direct photolysis,  $\varepsilon_{TBCZ}$  its molar absorption coefficient, b = 1.6 cm the optical path length of radiation in the solution inside the irradiation cells, and  $C^{\circ}_{TBCZ} = 5 \ \mu M$  (TBCZ initial concentration). The values of  $p^{\circ}(\lambda)$  (lamp spectral photon flux density) are those reported in **Figure 1**, which also shows the values of  $\varepsilon_{TBCZ}(\lambda)$ . Integration was carried out between 300 and 330 nm, where the spectra of TBCZ and the lamp overlap.

248

#### 249 *Reaction between TBCZ and •OH*

The reaction rate constant between TBCZ and •OH was determined by means of competition kinetics with 2-propanol (2-Prop), using  $H_2O_2$  under UVB irradiation as the photochemical source of •OH (Zellner et al., 1990). These irradiation conditions are known to trigger some direct photolysis of TBCZ (*vide supra*), thus the reaction system can be outlined as follows (Buxton et al., 1988; Gligorovski et al., 2015):

$$H_2O_2 + hv \rightarrow 2 \bullet OH \tag{3}$$

256 •OH + TBCZ  $\rightarrow$  Products  $[k_{\cdot_{OH+TBCZ}}]$  (4)

257 
$${}^{\bullet}\text{OH} + \text{H}_2\text{O}_2 \rightarrow \text{H}_2\text{O} + \text{HO}_2{}^{\bullet} \qquad [k_{{}^{\bullet}\text{OH} + \text{H}_2\text{O}_2} = 2.7 \times 10^7 \text{ M}^{-1} \text{ s}^{-1}]$$
 (5)

258 •OH + 2-Prop 
$$\rightarrow$$
 Products'  $[k_{\cdot_{OH+2-Prop}} = 1.9 \times 10^9 \text{ M}^{-1} \text{ s}^{-1}]$  (6)

259 
$$TBCZ + hv \rightarrow Products'' (direct photolysis)$$
 (7)

260 Upon application of the steady-state approximation to  $^{\circ}$ OH, one gets the following expression for 261 the initial degradation rate of TBCZ,  $R_{\text{TBCZ}}$  (see **Figure 2** for the experimental results):

262 
$$R_{\text{TBCZ}} = \frac{R_{f, \bullet \text{OH}} k_{\bullet \text{OH+TBCZ}} C_{\text{TBCZ}}^{\circ}}{k_{\bullet \text{OH+TBCZ}} C_{\text{TBCZ}}^{\circ} + k_{\bullet \text{OH+H}_2 O_2} C_{\text{H}_2 O_2} + k_{\bullet \text{OH+2-Prop}} C_{2-\text{Prop}}} + R_{\text{res.}}$$
(8)

where  $R_{f,OH}$  is the formation rate of OH by H<sub>2</sub>O<sub>2</sub> photolysis (reaction 3),  $k_{OH+TBCZ}$ ,  $k_{OH+H_2O_2}$ , and  $k_{OH+2-Prop}$  the second-order reaction rate constants between OH and TBCZ, H<sub>2</sub>O<sub>2</sub>, and 2-propanol, respectively (reactions 4-6),  $C_{TBCZ}^{o} = 20 \,\mu\text{M}$  the initial concentration of TBCZ,  $C_{H_2O_2} = 5 \,\text{mM}$  that of H<sub>2</sub>O<sub>2</sub>, and  $C_{2-Prop}$  the initial concentration of 2-propanol.





268

Figure 2. Trend of the TBCZ initial photodegradation rate ( $R_{\text{TBCZ}}$ ) as a function of the concentration of 2-propanol ( $C_{2-\text{Prop}}$ ), used as •OH scavenger. The dashed curve represents data fit with Eq. 8, the solid curves are 95% confidence limits of the fit.

273  $R_{\text{res.}}$  is a residual rate value, accounting for possible side reactions of TBCZ (for instance, its direct 274 photolysis and/or reactions with radicals produced from oxidation of 2-Prop by •OH) (Gornik et al., 275 2021). The trend of  $R_{\text{TBCZ}}$  vs.  $C_{2\text{-Prop}}$  is reported in **Figure 2**. The numerical fit of the experimental 276 data with **Eq. (8)** yielded  $k_{\cdot_{\text{OH+TBCZ}}} = (1.2\pm0.3) \times 10^{10} \text{ M}^{-1} \text{ s}^{-1} (\mu \pm \sigma)$ , and  $R_{\text{f}, \cdot_{\text{OH}}} = (1.6\pm0.1) \times 10^{-8} \text{ M}$ 277 s<sup>-1</sup>.

278

# 279 Triplet-sensitised degradation of TBCZ (<sup>3</sup>CBBP\* as <sup>3</sup>CDOM\* proxy)

280 To assess the potential for TBCZ to be degraded by reaction with <sup>3</sup>CDOM\*, here CBBP was used 281 as proxy of triplet-sensitisation by CDOM, following an already established (Minella et al., 2018) 282 and validated (Carena et al., 2019) experimental protocol. Irradiation of TBCZ and CBBP under a UVA lamp triggers the reaction between TBCZ and the CBBP triplet state, <sup>3</sup>CBBP\*, while at the 283 284 same time minimising the direct photolysis of TBCZ. The problem with CBBP is that it also yields 285 <sup>1</sup>O<sub>2</sub> under irradiation, although with very-well known kinetics (Minella et al., 2018). Therefore, the degradation of TBCZ by photogenerated <sup>1</sup>O<sub>2</sub> is to be taken into account. The overall reaction 286 287 system involving CBBP + TBCZ under UVA irradiation can be schematised as follows (ISC: intersystem crossing;  $S_{\Delta} = 0.46$ : yield of <sup>1</sup>O<sub>2</sub> upon reaction between <sup>3</sup>CBBP\* and O<sub>2</sub>;  $k_{3_{CBBP*+TBCZ}}$ : 288 second-order reaction rate constant between <sup>3</sup>CBBP\* and TBCZ;  $k_{1_{O,+TBCZ}}$ : second-order reaction 289 rate constant between  ${}^{1}O_{2}$  and TBCZ;  $k_{d}$ : first-order deactivation constant of  ${}^{1}O_{2}$  in aqueous 290 291 solution) (Wilkinson and Brummer, 1981; Minella et al., 2018):

292 
$$CBBP + hv \rightarrow {}^{1}CBBP^{*} \xrightarrow{ISC} {}^{3}CBBP^{*}$$
 (9)

$$293 \qquad {}^{3}\text{CBBP}^{*} \xrightarrow{Internal \ conversion} \rightarrow \text{CBBP}$$
(10)

- 294  ${}^{3}CBBP^{*} + O_{2} \rightarrow CBBP + {}^{1}O_{2}$  [ $S_{\Delta}$ ] (11)
- $^{3}CBBP^{*} + O_{2} \rightarrow CBBP + O_{2}$ (12)
- 296  ${}^{3}CBBP^{*} + TBCZ \rightarrow Products \qquad [k_{{}^{3}CBBP^{*}+TBCZ}]$  (13)

297 
$${}^{1}\text{O}_{2} + \text{TBCZ} \rightarrow \text{Products'} \qquad [k_{{}^{1}\text{O}_{2} + \text{TBCZ}}]$$
(14)

298 
$${}^{1}O_{2} \rightarrow O_{2}$$
  $[k_{d} = 2.5 \times 10^{5} \text{ s}^{-1}]$  (15)

Upon application of the steady-state approximation to  ${}^{3}CBBP*$  and  ${}^{1}O_{2}$ , one gets the following expression for the degradation rate of the substrate (Minella et al., 2018), TBCZ in the present case:

$$301 R_{\text{TBCZ}} = \Phi_{_{^{3}\text{CBBP}^{*}}} P_{a,\text{CBBP}} \left( \frac{k_{_{^{3}\text{CBBP}^{*}+\text{TBCZ}}} C^{\circ}_{\text{TBCZ}}}{k' + k_{_{^{3}\text{CBBP}^{*}+\text{TBCZ}}} C^{\circ}_{\text{TBCZ}}} + \frac{0.68 S_{\Delta} k' k_{_{^{1}\text{O}_{2}+\text{TBCZ}}} C^{\circ}_{\text{TBCZ}}}{(k' + k_{_{^{3}\text{CBBP}^{*}+\text{TBCZ}}} C^{\circ}_{\text{TBCZ}}) (k_{d} + k_{_{^{1}\text{O}_{2}+\text{TBCZ}}} C^{\circ}_{\text{TBCZ}})} \right) (16)$$

where  $\Phi_{_{3}_{CBBP*}} \sim 1$  is the quantum yield of <sup>3</sup>CBBP\* production by irradiated CBBP (Marciniak et 302 al., 1994),  $k' = 6 \times 10^5 \text{ s}^{-1}$  is the first-order rate constant of <sup>3</sup>CBBP\* deactivation in aerated solution, 303 by internal conversion and reaction with O<sub>2</sub> (Minella et al., 2018),  $P_{a,CBBP} = 1.2 \times 10^{-8}$  Einstein L<sup>-1</sup> 304 s<sup>-1</sup> is the photon flux absorbed by 67  $\mu$ M CBBP, and 0.68 is the fraction of <sup>3</sup>CBBP\* that gets 305 inactivated by dissolved oxygen (Minella et al., 2018). Note that the fraction of <sup>3</sup>CBBP\* that is 306 inactivated by internal conversion is 0.32, while the fraction of <sup>3</sup>CBBP\* that is inactivated by 307 reaction with TBCZ is negligible, if  $C_{\text{TBCZ}}^{\circ}$  is sufficiently low. If  $C_{\text{TBCZ}}^{\circ}$  is low enough, one should 308 get a linear trend of  $R_{\text{TBCZ}}$  vs.  $C^{\circ}_{\text{TBCZ}}$  ( $R_{\text{TBCZ}} = m \times C^{\circ}_{\text{TBCZ}}$ ). Calculation of the line slope m (Minella 309 et al., 2018) allows for the determination of the second-order rate constant of the reaction between 310 <sup>3</sup>CBBP\* and TBCZ,  $k_{3_{CBBP*+TBCZ}}$ : 311

312 
$$k_{_{3}_{CBBP^{*}+TBCZ}} = k' \left( \frac{m}{P_{_{a,CBBP}}} - \frac{0.68 S_{\Delta} k_{_{1}_{O_{2}}+TBCZ}}{k_{_{d}}} \right)$$
(17)

The reactivity of TBCZ towards 
$${}^{1}O_{2}$$
 was found to be very low (**Text S2**), thus the term  $k_{{}^{1}O_{2}+TBCZ}$  in  
Eq. (17) can be neglected (*i.e.*,  $0.68 S_{\Delta} k_{{}^{1}O_{2}+TBCZ} (k_{d})^{-1} \ll m (P_{a,CBBP})^{-1}$ , from which one gets  
 $k_{{}^{3}CBBP^{*}+TBCZ} \sim k' m (P_{a,CBBP})^{-1}$ ). A linear trend of  $R_{TBCZ}$  vs.  $C_{TBCZ}^{o}$  was observed in the TBCZ  
concentration range from 5 to 20  $\mu$ M (see Figure S5 (SM)), from which it was possible to obtain  
 $k_{{}^{3}CBBP^{*}+TBCZ} = (2.45\pm0.10) \times 10^{8} \text{ M}^{-1} \text{ s}^{-1}$ .

With some substrates, the primary intermediates produced by triplet-sensitised oxidation can be reduced back to the parent compounds, by phenolic antioxidants (AOs) (Canonica and Laubscher, 2008). In particular, the AO moieties occurring in DOM play a major role in back-reduction. With phenol (PhOH) as model AO (Wenk and Canonica, 2012), TBCZ as substrate, and CBBP as CDOM proxy, the back-reduction process (**reaction 20**) would operate as follows, to inhibit the degradation of TBCZ that initially proceeds through **reactions (18,19**):

$$324 \qquad TBCZ + {}^{3}CBBP^{*} \rightarrow TBCZ^{\bullet^{+}} + CBBP^{\bullet^{-}}$$
(18)

$$325 \qquad \text{TBCZ}^{\bullet+} \to \text{Products} \tag{19}$$

 $326 \qquad TBCZ^{\bullet+} + PhOH \rightarrow TBCZ + PhO^{\bullet} + H^{+} \qquad (20)$ 

The addition of phenol, in the concentration range of 0-50  $\mu$ M, had a negligible effect on the degradation of TBCZ sensitised by <sup>3</sup>CBBP\*, as shown in **Figure S6 (SM)**. Phenol concentration was not increased above 50  $\mu$ M, to avoid scavenging of <sup>3</sup>CBBP\* by phenol itself (Vione et al., 2018), which would inhibit the degradation of TBCZ independently of back-reduction. Note that the AO effect of 50  $\mu$ M phenol would be approximately equivalent to that of 20 mg<sub>C</sub> L<sup>-1</sup> DOC in natural waters (Leresche et al., 2016). Therefore, it can be inferred that TBCZ would not undergo significant back reduction in environmental conditions.

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# 335 Predicted photodegradation of TBCZ in sunlit surface waters

Additional experiments suggested that TBCZ would not be degraded by the carbonate radical,  $CO_3^{\bullet-}$ , to a significant extent (see **Text S3 (SM)**). Therefore, TBCZ in natural surface freshwaters would mainly be degraded by •OH, <sup>3</sup>CDOM\*, and perhaps the direct photolysis. Our innovative approach consists in predicting the contribution of the different photoreaction pathways to the photodegradation of TBCZ, by means of the photochemical model contained in APEX, at the same time assessing the overall lifetime of this compound. By so doing, we surmise that the environmental photo-fate of TBCZ is affected by environmental factors such as seasonal light intensity, water depth, and DOM concentration in water, and are able to quantify the role of eachfactor.

Figure 3 reports an example of the modelled photodegradation kinetics and pathways of TBCZ, under summertime irradiation conditions. It is shown that reaction with •OH would be the main TBCZ photodegradation pathway, followed by  ${}^{3}$ CDOM\* especially at high DOC. In contrast, direct photolysis and reactions with  ${}^{1}O_{2}$  or CO<sub>3</sub>•- would be minor to negligible.

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Figure 3. Modelled first-order photodegradation rate constants of TBCZ ( $k_{degr}$ , left Y-axis), together with the corresponding half-life times (right Y-axis) ( $t_{1/2} = \ln 2 k_{degr}^{-1}$ ), as a function of the dissolved organic carbon (DOC). Assumed water conditions: depth d = 1 m,  $[NO_3^{-1}] = 10^{-4}$  M,  $[NO_2^{-1}] = 10^{-6}$  M,  $[HCO_3^{-1}] = 10^{-3}$  M,  $[CO_3^{2-1}] = 10^{-5}$  M. Days correspond to fair-weather 15 July at 45°N latitude. Calculations were carried out with the APEX software; the coloured areas of the graph highlight the importance of the different photodegradation pathways (d.p. = direct photolysis).

The resulting TBCZ lifetime would be in the range of days to weeks, varying depending on the environmental conditions. In particular, given the major role of •OH in TBCZ photodegradation, the process would be favoured in shallow, nitrate- and nitrite-rich environments (Zepp et al., 1987) with low DOC. For instance, with the same water chemistry and sunlight irradiance conditions used in **Figure 3**, but with water depth d = 5 m instead of 1 m, the lifetime ( $t_{1/2}$ ) of TBCZ would increase to 20-70 days (see **Figure S8 (SM)**).

The predicted prevalence of indirect over direct TBCZ photodegradation is consistent with the results of irradiation experiments, in which TBCZ was UVA-irradiated in both ultra-pure water and lake water. Indeed, TBCZ photodegradation was faster in lake water, as shown in **Figure 4**, which suggests an important role played by lake-water photosensitisers over the direct photolysis process, which is the only operational reaction pathway occurring in ultra-pure water.



372

**Figure 4.** TBCZ time trend upon UVA irradiation, in ( $\blacksquare$ ) ultra-pure water and ( $\bullet$ ) lake water (Lake Soprano, Western Italian Alps, 2100 m a.s.l.; water chemistry: 25 µM NO<sub>3</sub><sup>-</sup>, 1 µM NO<sub>2</sub><sup>-</sup>, 0.84 mgc L<sup>-1</sup> DOC, 1.1 mg<sub>C</sub> L<sup>-1</sup> inorganic carbon, pH 6.8). For comparison, the TBCZ time trend in lake water in the dark ( $\blacklozenge$ ) is also reported. UVA irradiance was 25 W m<sup>-2</sup>.

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379 month to complete in some environments. In these circumstances, the time trend can be described 380 by Eq. (1), as series of first-order degradation intervals where the rate constant  $k_{\rm m}$  changes from one 381 month to the following (Vione, 2021). Using APEX Season to derive  $k_m$ , and applying Eq. (1), one 382 gets the time trends shown in Figure 5a/b. In particular, Figure 5a was obtained by assuming the 383 most favourable conditions to photodegradation, among those modelled in Figure 3 (depth d = 1 m, DOC = 1 mg<sub>C</sub> L<sup>-1</sup>). Conversely, in the case of Figure 5b, the least favourable conditions among 384 385 those of Figure 3 were assumed (d = 1 m, DOC =  $10 \text{ mg}_{\text{C}} \text{ L}^{-1}$ ). 386 Interestingly, if environmental conditions allow TBCZ photodegradation to be relatively fast 387 (Figure 5a), extensive removal ( $\geq 95\%$ ) is obtained more quickly if initial emission takes place in 388 summer or spring compared to autumn or winter, as expected (the irradiance of sunlight is in fact 389 higher in spring/summer compared to autumn/winter). In contrast, for slower photodegradation 390 kinetics (Figure 5b), one has that  $\geq$  95% removal is obtained faster for emission in winter or spring, 391 compared to summer or autumn. Actually, in this case photodegradation spans over several months, 392 and the conditions of the initial month/season of emission play more limited role. Indeed, the 393 irradiance conditions of the following season(s) are comparatively more important. Figure 5b additionally suggests that, with d = 1 m and DOC = 10 mg<sub>C</sub> L<sup>-1</sup>, it would take from 6-7 months to 394 395 about 1 year for photodegradation to eliminate TBCZ from the water environment. 396

The results reported in Figure 3 suggest that TBCZ photodegradation might take more than one



Months from emission

Figure 5. Modelled time trends of TBCZ photodegradation (Eq. 1), as a function of the month of initial emission (m = 0, highlighted near each curve). (a) Depth d = 1 m, DOC = 1 mg<sub>C</sub> L<sup>-1</sup>; (b) d = 1 m, DOC = 10 mg<sub>C</sub> L<sup>-1</sup>. Other water conditions: [NO<sub>3</sub><sup>-</sup>] = 10<sup>-4</sup> M, [NO<sub>2</sub><sup>-</sup>] =  $10^{-6}$  M, [HCO<sub>3</sub><sup>-</sup>] =  $10^{-3}$  M, and [CO<sub>3</sub><sup>2-</sup>] =  $10^{-5}$  M. Consistent fine weather was hypothesised here.

## 407 Identification of TBCZ phototransformation products

In the direct photolysis of TBCZ, only the ion at m/z 308 ( $[M + H]^+$ ) was detected, i.e., parent TBCZ (see **Table 1** for its fragmentation). The fragment ion at m/z 290 resulted from the loss of a water molecule from the parent ion, while the fragment m/z 221 was obtained through the loss of 1,2,4-triazacyclo-2,4-pentadiene from m/z 290. Moreover, the loss of butyl gives the ion with m/z 165 (Calza et al., 2002) (see **Figure S9 (SM)**). In the MS<sup>3</sup> experiment, the ion at m/z 130 could be produced by loss of chlorine.

The main ions found in the degradation of TBCZ by 'OH are reported in **Table 1**, while the hypothesised structures of the related products are shown in **Table 2**. Identification was possible thanks to comparison with literature findings, referred to the heterogeneous photocatalytic ( $TiO_2 +$ hv) degradation of TBCZ (Calza et al., 2002). For the details about photoproducts identification by LC-MS<sup>n</sup>, the reader is referred to the work by Calza et al. (2002), and to the SM of the present work.

The only degradation products detected with <sup>3</sup>CBBP\* were two hydroxyderivatives, with ions having m/z 324 (see **Table 1**). Both ions showed loss of two water molecules (m/z 306 and 288), and the subsequent loss of triazole (m/z 237) in MS<sup>2</sup> experiments. The formation of TBCZ hydroxyderivatives upon reaction with <sup>3</sup>CBBP\* needs some explanation. The TBCZ alkyl chain could be oxidised by <sup>3</sup>CBBP\* via hydrogen transfer, giving a radical species that could add oxygen. Further reaction with water could then produce the hydroxyderivative(s), and the overall process (**reactions 21-24**) simulates **•**OH attack (De Laurentiis et al., 2012 & 2014):

427 
$$R-H + {}^{3}CBBP^{*} \rightarrow R^{\bullet} + CBBPH^{\bullet}$$
 (21)

$$428 \qquad \mathbf{R}^{\bullet} + \mathbf{O}_2 \to \mathbf{R} \text{-} \mathbf{OO}^{\bullet} \tag{22}$$

429 
$$R-OO^{\bullet} + H_2O \rightarrow R-OH + HO_2^{\bullet}$$
 (23)

$$430 \qquad CBBPH^{\bullet} + O_2 \rightarrow CBBP + HO_2^{\bullet} \qquad (24)$$

- 431 Tentative pathways leading to TBCZ hydroxylation, upon reaction with either •OH or <sup>3</sup>CDOM\*, are
- 432 reported in **Scheme 1**. Note the partial overlap between the •OH and <sup>3</sup>CDOM\* reactions.



433

434 Scheme 1. Tentative reaction pathways leading to the detected TBCZ hydroxyderivatives, 435 involving either •OH or <sup>3</sup>CDOM\*. The attack position of •OH on the aromatic ring of TBCZ is 436 highly uncertain (the shown position, in *meta* to chlorine, is just an arbitrary assumption).

437

438 Based on the structures of TBCZ and its identified products, ECOSAR calculations (Mayo-Bean et 439 al., 2012) provided insights into their potential acute (values of lethal concentration LC50 and 440 effective concentration EC50) and chronic (chronic value ChV) toxicity, towards freshwater 441 organisms. Calculation results are reported in Table 2, and they represent an order-of-magnitude 442 estimate of toxicity values (Mayo-Bean et al., 2012). On this basis, the obtained results suggest that 443 TBCZ phototransformation products have similar or lower toxicity compared to their parent compound. A decrease in toxicity is expected, especially after fragmentation of the aromatic ring. 444 445 Therefore, TBCZ photodegradation is likely to lead to detoxification.

Table 1. Main fragmentations in MS<sup>n</sup> of TBCZ and its degradation products, because of direct
 photolysis (48 hours of irradiation), •OH reaction (30 minutes of irradiation), and reaction with
 <sup>3</sup>CBBP\* (48 hours of irradiation).

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Photochemical conditions	MS <sup>2</sup> (m/z)	MS <sup>3</sup> (m/z)	Retention time (min)	Ions m/z, relative abundance in brackets (referred to the most intense ion)	
d.p.	308*	-	20.92	290 (24), 221 (23), 181 (24), 165 (100), 151 (22), 139 (15), 125 (55)	
	308**	165	20.64	130 (100)	
•OH	324	306	19.66	306 (100), 288(36), 237 (26), 181 (75), 167 (38), 141 (45), 109 (5)	
	324	306	18.86	306 (70), 237 (15), 196 (12), 181 (100), 167 (93), 155 (9), 141 (5)	
	324	306	16.00	322 (10), 305 (8), 288 (23), 280 (43), 263 (100), 237 (9), 181 (18)	
	322	304	20.64	304 (8), 286 (5), 235 (100), 179 (12), 139 (8)	
	278	260	16.57	260 (9), 242 (12), 191 (91), 178 (100), 173 (62), 145 (46), 109 (36), 107 (31), 95 (10)	
	272	236	12.53	180 (18), 167 (100), 162 (19), 137 (7), 111 (45), 83 (10), 70 (31)	
	224	206	14.33	206 (16), 178 (6), 137 (100), 109 (9), 107 (14), 70 (46)	
<sup>3</sup> CDOM* ( <sup>3</sup> CBBP*)	324	-	20.36	306 (6), 288 (100), 237(10), 207 (33), 177 (19), 168 (23), 151 (13), 139 (29)	
	324	-	19.90	306 (44), 288 (100), 237 (1), 168 (23)	

450 \* Summary of MS<sup>2</sup> experiment for TBCZ. \*\* Summary of MS<sup>3</sup> experiment for TBCZ.

**Table 2.** Acute (LC50/EC50) and chronic (ChV) toxicity values (mg  $L^{-1}$  units) of TBCZ and its detected phototransformation products, calculated by means of the ECOSAR software, on the basis of molecular structures. The m/z values refer to the [M+H]<sup>+</sup> molecular ions, which are discussed in the text (the reported structures show the neutral molecules M). The reported values are here the lowest, among the multiple classes that ECOSAR can identify for a given compound.

	Fish		Crustaceans		Algae	
	LC50	ChV	LC50	ChV	EC50	ChV
	2.4	0.02	3.5	0.19	0.9	0.5
но сі-сі- N N M/z 324b	4.7	0.04	3.9	0.4	0.4	0.7
m/z 324a/c	19.6	0.09	17.6	1.3	3.3	1.3
сі-Сі-Сі-Сі-Сі-Сі-Сі-Сі-Сі-Сі-Сі-Сі-Сі-Сі	6.1	0.1	4.6	0.6	1.4	1.6
он он N N m/z 272	1,090	1.9	377	45.5	40.9	7.7
о=с=сн м м m/z 224	56.9	37.4	233	6.8	12.3	42.2

# 447 Conclusions

448

449 The fungicide TBCZ does not undergo direct photolysis to a significant extent in environmental 450 conditions. From measured photoreactivity parameters (which are summarised in Table S2(SM)), it 451 can be inferred that the main TBCZ phototransformation processes in sunlit surface waters are the 452 reactions with •OH (especially) and <sup>3</sup>CDOM\*. In the latter case, no back-reduction of partially 453 oxidised TBCZ is expected to be operated by the phenolic antioxidant mojeties occurring in DOM. 454 Therefore, no inhibition of <sup>3</sup>CDOM\* degradation by DOM phenols should take place in high-DOC 455 waters. The photochemical lifetime of TBCZ in the natural environment would strongly depend on 456 the environmental conditions. The value of  $t_{1/2}$  could be as low as one week under favourable circumstances (shallow waters with low DOC during summertime, when degradation by •OH is 457 458 highly enhanced). Conversely, the inhibition of the 'OH process at high DOC ('OH is in fact 459 efficiently scavenged by DOM) is only partially offset by the enhancement of degradation by <sup>3</sup>CDOM\*. In these conditions, TBCZ degradation could span over several months, departing from 460 461 first-order kinetics, because of seasonally changing irradiance of sunlight. TBCZ elimination could 462 thus require 6-7 months, upon initial emission in winter or spring, and up to one year if initial 463 emission takes place in summer or autumn.

The phototransformation of TBCZ has potential to lower its environmental impact, when judging from the expected toxicity of the detected products. In particular, TBCZ photodegradation mainly proceeds by hydroxylation, followed by fragmentation of the aromatic ring.

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