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Abstract

Metrafenone (MF), as a new type of benzophenone fungicide, has been widely used in agriculture and is persistent in the environment. Understanding its photochemical fate is essential for the comprehensive evaluation of its ecological risk. In the present work, we reported a detailed study on the photochemical transformation of MF in aqueous solution under irradiation (at $\lambda > 290$ nm using a high pressure mercury lamp). MF was relatively photo-reactive showing a low polychromatic quantum yield of photolysis (1.06×10^{-4}) counterbalanced by a significant light absorption above 290 nm. A series of photoproducts were identified by high resolution mass spectrometry (HR-MS) analysis, and three different pathways, including oxidation of the methyl group, debromination and replacement of bromine by hydroxyl group were proposed. Among them, debromination was identified as the dominating process that could be achieved *via* homolytic C-Br bond cleavage from singlet and triplet MF, as confirmed by laser flash photolysis (LFP) experiments and density functional theory (DFT) calculations. Toxicity assessment revealed that photochemical degradation reduced the ecotoxicity of MF efficiently. Nitrate ions and humic acid promoted the MF photolysis, while bicarbonate exhibited no effect. Results obtained in this work would increase our understanding on the environmental fate of MF in sunlit surface waters.

Keywords: Metrafenone; Photolysis; Debromination; Homolytic cleavage; Bond dissociation energy.

1. Introduction

Fungicides are widely applied to control fungal pathogens on growing crops to improve yields (Cintia et al., 2021; Lin et al., 2020; Miller et al., 2021; Nermine et al., 2021; Patrick et al., 2021; Regueiro et al., 2015). Metrafenone (MF) is a new type of

benzophenone derivative fungicide, which is a brand-new active ingredient in the field of phytochemical protection (structure in Fig. 1). Studies have demonstrated that MF is effective in reducing spore germination, preventing appressorium formation, and inhibiting powdery mildew infestation (Franck et al., 2000; Krystina et al., 2006). Hence, it has been widely used to prevent and treat powdery mildew in crops such as wheat and grapes. However, repeated application of fungicides over a longtime period would inevitably lead to their accumulation in the environment. These toxic compounds can leach out of the soil, and contaminate surface water and groundwater, inducing potential health risks to residents (Freya and Jane, 2004; Pimmata et al., 2013). In agricultural production, MF is normally pre-applied and there is a tendency to increase the amount applied as microbial resistance increases, resulting in its persistence in the environment (Kunova et al., 2016; Leroux et al., 2013). Residual MF has been widely detected in grapes, juices, and wines after its application to vines, which eventually affect the human health (Briz-Cid et al., 2014; Cabras and Angioni, 2000; Noelia et al., 2018; Noguerol et al., 2010). Regueiro et al. (2015) once evaluated the toxicity of MF on cortical neurons in mice, they reported that MF induces depolarization of mitochondrial membranes, reduces cellular activity, and exhibits toxic effects after 7 days of exposure (Regueiro et al., 2015).

Fig. 1.

In order to accurately assess the ecological risks of fungicides, it is essential to investigate their transformation and fate in the environment (Baćmaga et al., 2019). Photochemical transformation is among the most important processes that affect the

fate of organic compounds in surface water (Ge et al., 2009; Yong et al., 2009). Direct photolysis of fungicides is caused by direct absorption of photons, while indirect photolysis is triggered by the absorption of light by photosensitizers. Active components in water (such as dissolved organic matter and ions) can interact with organic pollutants through energy transfer of excited substances or through photoproduction of reactive transient species, which affects the persistence and fate of fungicides in the environment (Avetta et al., 2014; Fréneau et al., 2019; Jennifer et al., 2019; Liu et al., 2016). As shown in Fig. 1, the molecular structure of MF contains chromophores such as benzene rings and carbonyl groups which can absorb ultraviolet or visible light, indicating that direct photolysis of MF is likely upon irradiation (Turro, 1978). López-Fernández et al. (2018) confirmed that photolysis is an effective route for the removal of MF from grape juice (López-Fernández et al., 2018). However, to our knowledge, specific studies on the photolytic fate of MF in natural surface waters is quite limited.

In the current study, the photolytic behavior of MF in aqueous solution is investigated systematically, the main degradation pathways and mechanisms are proposed. For this, steady-state irradiations are performed to measure the quantum yield of MF photolysis, the second order rate constant ($k_{^1O_2,MF}$) for the reaction between singlet oxygen ($^1O_2$) and MF and test the effect of pH and quenchers. In addition, laser flash photolysis (LFP) experiments are conducted to identify the transient species involved in the reaction and density functional theory (DFT) calculations are adopted to confirm the hypotheses. The and MF and are also

measured. The photodegradation products of MF are determined by high resolution mass spectrometry (HR-MS) and their toxicity is evaluated by Ecological Structure Activity Relationship (ECOSAR 2.0) software. Subsequently, the effects of water matrices including nitrate ions, bicarbonate, and humic acid on MF photodegradation are evaluated. Results obtained here will provide a scientific basis for a comprehensive understanding of the photochemical fate of MF in the aquatic environment.

2. Materials and methods

2.1. Materials

Metrafenone (MF, 99.7%) and humic acid (HA, 99%) were purchased from Sigma-Aldrich (St. Louis, MO, USA). Phosphoric acid with chromatographic purity and sorbic acid (99.8%) were obtained from Aladdin (Shanghai, China). Deuterium oxide (D_2O, 99.9%) was obtained from Energy Chemical (Shanghai, China). Isopropanol (IPA, 99.9%) was obtained from Thermo Fisher Scientific (Waltham, USA). Sodium nitrate (≥98.5%) and sodium bicarbonate (≥99.5%) were purchased from Sinopharm Chemical Reagent Co., Ltd and Richjoint, respectively. HPLC grade acetonitrile (ACN, 99%) was obtained from J&K Scientific Ltd (Shanghai, China). Other reagents were at least of analytical grade and used as received.

2.2. Photolysis experiments

2.2.1 Steady-state photolysis experiments

Steady-state photolysis experiments were conducted in a device equipped with a 500 W high pressure mercury lamp (Beijing Zhongjiao Jinyuan Technology Co., Ltd,

China) surrounded by a 290 nm cut-off filter as the light source. The mercury lamp was turned on preliminarily for 30 min. After stabilization, 25 mL solution to be irradiated containing target compounds (20 μM) were pipetted into a Pyrex glass reactor. During the irradiation process, Pyrex glass reactors were transferred to the water-cooled immersion well in the device to maintain the temperature at 20 °C. Dark controls were also performed under the same conditions. During the irradiated process, 0.5 mL aliquot of the solution to be irradiated was withdrawn at predetermined time intervals for analysis. To ensure the reliability of the data, all experiments were carried out at least in duplicates, and the error bars in the figures represent the standard deviation (SD).

2.2.2 Laser flash photolysis experiments

Laser flash photolysis (LFP) experiments were carried out using a Quanta Ray LAB-150–10 Nd:YAG laser at an excitation wavelength of 266 nm to investigate the absorbance and disappearance of the transient species in ACN solutions of MF. Solutions were purged with purified N_2 prior to the experiments to evaluate the effect of O_2 (triplet quencher) on the lifetime of transient species. Different time windows were used to fit the decay of the transient species of MF.

2.3. Analytical methods

The concentration of MF was measured using a high performance liquid chromatograph (HPLC, Waters) equipped with a separation module (Waters 2695) and a photodiode array detector (Waters 2998). Detailed operation conditions are provided in Text. S1.

The transformation products were identified by a high resolution mass spectrometry (HR-MS), which was coupled to a HPLC Waters Alliance system for chemical separation. The details of HR-MS analysis are given in Text. S2.

The bromine ion concentration was measured by ion chromatograph and the details are provided in Text. S3.

The details for electron paramagnetic resonance spectroscopy analysis (EPR) are illustrated in Text. S4.

2.4 Quantum yield

The polychromatic quantum yields of MF (Φ_{MF}) direct photolysis in aqueous solutions were determined using *p*-nitroanisole (PNA)/pyridine (pyr) as an actinometer according to Eq. (1).

$$\Phi_{MF} = \frac{k_{MF}}{k_{PNA}} \times \frac{\sum_{\lambda}(L_{\lambda}\varepsilon_{\lambda,MF})}{\sum_{\lambda}(L_{\lambda}\varepsilon_{\lambda,PNA})} \times \Phi_{PNA} \quad (1)$$

where k_{MF} and k_{PNA} were the observed photolysis rate constant of MF and PNA, respectively. ε_{λ} (mol^{-1} cm^{-1}) represented the molar absorptivity and L_{λ} (cm^{-2} s^{-1}) was the incident light intensity. $\sum_{\lambda}(L_{\lambda}\varepsilon_{\lambda,MF})$ and $\sum_{\lambda}(L_{\lambda}\varepsilon_{\lambda,PNA})$ were the light absorption for MF and actinometer, respectively. Φ_{PNA} was the quantum yield of PNA and was calculated according to Φ_{PNA} = 0.29[pyr] + 0.00029 (Laszakovits et al., 2016). The ε_{λ} values of aqueous MF and PNA solutions were measured using a UV-Vis spectrophotometer and specific pathlength quartz cells.

2.5. DFT calculations

In this study all the density functional theory (DFT) calculations were conducted by using Gaussian 16 package with hybrid density functional B3LYP method. The

details for DFT calculation were illustrated in Text. S6.

2.6. Ecotoxicity assessment

The aquatic toxicity data was an essential judgment basis for environmental hazard classification and ecological risk assessment of chemicals (Cash, 1998). In the present study, the ecotoxicity of MF and photoproducts was evaluated by the Ecological Structure Activity Relationship (ECOSAR 2.0). Three classical aquatic organisms (green algae, daphnia and fish) were assessed for acute toxicity (half-lethal and half-effect concentration (LC50, EC50)) and chronic toxicity values (Chv) (Wang et al., 2019).

3. Results and discussion

3.1 Direct photolysis of MF

The UV-Vis absorption spectrum of aqueous MF exhibits a broad absorption band ranging from 260 to 350 nm (λ_{max} = 292 nm, ε_{292} = 7750 mol^{-1} cm^{-1}) thus well overlapping with the emission spectrum of the mercury lamp (Fig. 2) and of the solar spectrum, implying that direct photodegradation can potentially take place for MF in natural conditions. The absorption band of MF is mainly due to the presence of chromophores such as benzene and ketone groups that are capable of absorbing UV-Vis light. In the present study, the absorption band corresponds to the low energy $\pi \rightarrow \pi^*$ excitation due to the presence of the benzene chromophore rather than the ketone chromophore ($n \rightarrow \pi^*$ excitation) (Turro, 1978). This could be further confirmed by TD-DFT calculations, as shown in Fig. S1, the broad band ranging from 260 nm to 350 nm is attributed to the S1 to S5 absorptions, among which the S4 state

has the largest oscillator strength (Table. S1). We thus analyzed the molecular orbital (MO) transition contributions of the S4 state. The two main transitions are the highest occupied molecular orbital (HOMO)−2 to the lowest unoccupied molecular orbital (LUMO) and HOMO−1 to LUMO. Based on the isosurfaces of HOMO−2 (π), HOMO−1 (π), and LUMO (π*) (Fig. S2), the excitation can be as assigned to a π→π* transition. In addition, solvatochromic experiments by using ACN as a solvent could also demonstrate this phenomenon of absorption band distribution (Fig. 2). Molecules that undergo a π→π* transition are believed to have excited states more strongly polarized than the ground state. Therefore, a strongly polar solvent (such as water) has a greater stabilizing effect on the excited state and can reduce the energy difference between the excited state and the ground state. Therefore, as polarity of the solvent decreased with the solvent substitution of H_2O by ACN, the absorption was hypochromatically shifted, which implied that the energy required from the ground state to the excited state became larger, demonstrating that the MF molecule underwent a π→π* transition (Da et al., 2001). The n→π* excitation was not observed (or was weak) because of the poor spatial overlap of the n and π* orbitals, resulting in a weak vibronic intensity, which should theoretically be at 350 nm (Lim and Jack, 1966).

Fig. 2.

Dark control experiments showed no loss of MF, therefore, hydrolytic or microbial degradations were not further considered in this study. Upon irradiation, a significant MF decay was observed, implying that photolysis limited the persistence

of MF in aqueous solutions. The decay of MF could be fitted by a pseudo-first-order kinetics, and the observed rate constant (k_{obs}) was calculated by linear regression of logarithmic concentration values ($\ln([MF]_t / [MF]_0)$) determined as a function of time (Eq. (2)). Half-life ($t_{1/2}$) was also calculated according to Eq. (3). In the present work, k_{obs} was determined to be 0.056 ± 0.003 h^{-1} under simulated sunlight, corresponding to a half-life ($t_{1/2}$) of 12.4 ± 0.6 h. These values were comparable with that reported values (0.072 ± 0.005 h^{-1} (k_{obs}) and 9.6 ± 0.7 h ($t_{1/2}$)) (López-Fernández et al., 2018). European Food Safety Authority (EFSA) concluded that the half-life of MF in the dark aerobic water sediment system (biological degradation) is 8.5-9.2 days (EFSA., 2006). It clearly shows that MF is difficult to biodegrade, therefore, we can conclude that photolysis plays an important role in the elimination of MF in the aquatic environment.

$$\ln([MF]_t/[MF]_0) = -k_{obs}t \quad (2)$$

$$t_{1/2} = \ln 2 / k_{obs} \quad (3)$$

Quantum yields are important parameters in photochemistry. The quantum of a given process or of a global reaction corresponds to the ratio of photons triggering a photochemical process or the reaction to the total number of absorbed photons (Rene, 2002) and therefore gives the efficiency of absorbed photons to achieved the process or the reaction. Here, the polychromatic quantum yield of MF photolysis (Φ_{MF}) at 20 μM was determined to be 1.06×10^{-4}, which is higher than the quantum yields of photolysis of other benzophenone derivatives that do not contain halogene atoms such as benzophenone-3 (1.51×10^{-5}) and 4-hydroxy-benzophenone (7.91×10^{-6}) (Li et al.,

2016). This result suggests that the presence of a Br atom in MF may favor the photodegradation.

The potential impact of pH solution on MF photolysis was also tested. Our results indicated that the rate of photodegradation was barely affected by the solution pH between 4 and 10 (Fig. S4). This phenomenon is reasonable based on the pH-independent absorption spectra of MF (Fig. S5) and on the absence of dissociable functional groups in the molecular structure of MF.

3.2 Identification of potential reactive intermediates

The triplet excited state of carbonyls has attracted a wide interest as a special oxidant in photochemical reactions. Upon irradiation, MF molecules were likely to absorb photons to make an energy level transition from the ground state to the singlet excited state (^{1}MF*), followed by an intersystem crossing (ISC) process to generate the triplet excited state (^{3}MF* containing two electrons with the same spin direction and unpaired) (Eq. (4)) (Nicholas et al., 2012).

The presence of a bromine atom in MF molecule is expected to enhance the spin-orbit coupling due to the heavy halogen effect, thus favoring the ISC process. Therefore, the generation of ^{3}MF* and its involvement in the photolysis process is likely. Sorbic acid (a triplet excited state quencher) was added to the MF solution to test the involvement of ^{3}MF* (Zhou et al., 2013). As shown in Fig. 3, the presence of sorbic acid (0.4 mM) partially inhibited the reaction, confirming the involvement of ^{3}MF* in the photo-transformation.

$$\mathrm{MF} \xrightarrow{h\nu} {}^1\mathrm{MF}^* \xrightarrow{ISC} {}^3\mathrm{MF}^* \quad (4)$$

It has been reported that the energy required to excite many aromatic compounds to their triplet excited states was greater than 250 kJ mol^{-1}, whereas the energy for $^1O_2$ formation was only 94 kJ mol^{-1} (Zepp et al., 1987). Therefore, under irradiation, energy transfer between the triplet excited state of organic compounds and dissolved oxygen may occur, resulting in the production of $^1O_2$. For example, the triplet excited state energy of benzophenone was as high as 288 kJ mol^{-1}, which was sufficient to transfer energy to dissolved oxygen and generate $^1O_2$ with a quantum yield of 0.34 (Turro et al., 2012). Since replacing H_2O with D_2O extends the lifetime of $^1O_2$ 13-fold (Rodgers and Snowden, 1982) a D_2O-H_2O mixture (50:50, v:v) was employed as a solvent to investigate whether $^1O_2$ was produced in the reaction and was responsible for MF oxidation. The reaction rate constant increased by a factor of 1.64 when the solvent was changed to the mixture of D_2O-H_2O (Fig. 3), indicating that $^1O_2$ was generated in the photolysis of MF by photosensitization. Therefore, these results are consistent with the formation of ^{3}MF* and the generation of $^1O_2$ by energy transfer to O_2 (Eq. (4) and (5)). Once formed, $^1O_2$ could react with MF (Eq (6)). We therefore further measured the bimolecular rate constant of this reaction, $k_{^1O_2,MF}$. The value found: 5.6×10^7 L mol^{-1} s^{-1} demonstrated the easy oxidation of MF by $^1O_2$. Furthermore, the irradiation of MF in the presence of isopropanol (0, 8 and 20 mM) used as an •OH radical scavenger ruled out the involvement of this radical in the photolysis of MF (Fig. S6).

$$^3MF^* + O_2 \rightarrow MF + {}^1O_2 \quad (5)$$

$$^1O_2 + MF \rightarrow \text{oxidation products} \quad (6)$$

Fig. 3.

In the case MF disappears through Eq (6), the rate of MF loss can be written:

$$-d[MF]/dt = Ia \times \Phi_{SO} \times (k_{^1O_2,MF}.[MF]/(k_{^1O_2,MF}.[MF]+ k_d) \quad (7)$$

Where Ia is the rate of light absorption by MF, Φ_{SO}, the quantum yield of $^1O_2$ formation and k_d, the rate of $^1O_2$ desactivation in water (2.5×10^5 s^{-1}, Rodgers and Snowden, 1982). Moreover, Φ_{MF} is equal to :

$$\Phi_{MF} = -d[MF]/dt/Ia = \Phi_{SO} \times (k_{^1O_2,MF}.[MF]/(k_{^1O_2,MF}.[MF]+ k_d) \quad -(8)$$

It means that Φ_{MF} depends on the [MF], in particular when $k_{^1O_2,MF}.[MF] << k_d$, that is the case at the chosen concentrations. To check this, we measured Φ_{MF} for several concentrations ranging from 20 to 5 µM. Φ_{MF} decreased from 1.06×10^{-4} for 20 µM to 0.52×10^{-4} for 5 µM, in accordance with a contribution of $^1O_2$ to the phototransformation of MF.

M. A. J. Rodgers, P. T. Snowden. Lifetime of ($^1O_2$) in Liquid Water As Determined by Time-Resolved Infrared Luminescence Measurements. J. Am. Chem. Soc. 1982, 104, 5541-5543.doi-org.ezproxy.uca.fr/10.1021/ja00384a070

3.3 Photoproducts identification

In this work, we systematically investigated intermediate photoproducts of MF *via* HR-MS analysis after 22 h irradiation. The major intermediate photoproducts are

289 listed in Table. S2. Bromine atom has two isotopes (relative atomic mass 78.9183 and
290 80.9163), based on this property, isotopic features were used as a diagnostic tool for
291 the identification of photoproducts containing bromine atoms. For instance, the parent
292 compound MF had a molecular ion cluster corresponding to m/z = 409.0634/411.0612
293 ($[M+H]^+$) with retention time of 4.91 min, suggesting that the molecular structure of
294 MF contained one bromine atom. Reviewing Table. S2, it can be seen that eight
295 photoproduct intermediates were detected. Specifically, TP1 with m/z =
296 423.0426/425.0404 ($[M+H]^+$) at retention time of 1.97/2.65 min was assigned as the
297 oxidation product in which a methyl group was oxidized into aldehyde. The m/z =
298 439.0375/441.0353 ($[M+H]^+$) molecular ion was labeled as the secondary product
299 (TP2), which was generated by oxidation of the methyl group to carboxyl with
300 retention time of 1.28/1.6 min. Based on the total ion currents (TICs), debrominated
301 product (TP3) was a major intermediate product generated during MF photolysis (Fig.
302 S7), which was eluted at 2.56 min with m/z = 331.1530. Furthermore,
303 debrominated-methyl oxidation products (TP4 and TP5) were identified based on m/z
304 = 345.1325 and 361.1273 ($[M+H]^+$) with retention time of 1.63/1.90 and 1.22/1.41
305 min, respectively. Notably, TP3, TP4 and TP5 were also observed in a previous study
306 (López-Fernández et al., 2018). Moreover, TP6 with m/z = 347.1480 ($[M+H]^+$) was
307 identified as the OH substitution product, which was another major debromination
308 product (Fig. S7). To the best of our knowledge, the formation of
309 debrominated-hydroxylated photoproduct in the photolysis of MF was reported for the
310 first time. TP7 and TP8 (m/z = 361.1273 and 377.1221, $[M+H]^+$) were assigned to the

methyl oxidized from of TP6.

To gain deeper insight into the debromination process, the evolution of releasing bromide was recorded during the photolysis of MF, and the yield of bromide was calculated according to Eq. (9) (Xu et al., 2020):

$$\text{Yield of bromide} = \frac{[Br^-]_t}{[MF]_0 - [MF]_t} \times 100\% \quad (9)$$

where $[Br^-]_t$ is the concentration of bromide ion detected at irradiation time *t*; $[MF]_0$ and $[MF]_t$ are the concentrations of MF at irradiation time zero and *t*, respectively. Based on the results of ion chromatography, we obtained an average bromine yield of 73±1% for MF direct photolysis (Fig. S8), which confirmed our conclusion that debromination is a major pathway in the direct photolysis of MF.

3.4 Photolysis pathways

Based on the detected intermediates, several photodegradation pathways of MF are proposed (Fig. 4). Pathway I corresponded to the oxidation of the β-methyl group, giving rise to a phenylacetaldehyde product (TP1), followed by a further oxidation process to convert the aldehyde group to a carboxyl group. Pathway II corresponded to the debromination process, leading to the generation of TP3. It was noteworthy that TP3 exhibited a relatively strong signal in the mass spectrum (Fig. S7), indicating that the yield of this product was relatively high during MF photolysis. Subsequently, TP3 could further undergo methyl oxidation and aldehyde oxidation to produce TP4 and TP5 sequentially. Such process has been previously reported by López-Fernández et al. (2018) in the photolysis process of MF. Pathway III corresponded to the substitution of bromine by a hydroxyl group to give TP6, then, the methyl group on

TP6 could sequentially be oxidized to form TP7 and TP8.

Furthermore, it has been well documented that for the photochemical transformation of carbonyl chemicals, Norrish type I and II reactions were proposed to take place through which fission of the carbon-carbon bond would occur at the α- and γ-positions, leading to the formation of bond cleavage or intramolecular cyclization products, respectively (Turro, 1978). However, in the present study, none of the above-mentioned products were identified by mass searching from the total ion chromatography (TIC) spectra; instead, the dominated photolysis pathway of MF was proposed as debromination process, which would be discussed in detail in the following section.

Fig. 4.

3.5 Photo-induced debromination

As mentioned, debrominated product (TP3) were found to be the major photoproduct of MF, indicating that C-Br bond dissociation was an important path of mechanistic explanation. This bond cleavage might take place either *via* a homolytic process to give a carbon-based benzoylbenzyl radical and bromine atom, or through a heterolytic process to generate a corresponding benzyl cation and bromide anion.

In order to gain a detailed insight into the photolysis mechanism of MF, LFP experiments were conducted to identify the potential transient intermediates involved in the photochemical process. Fig. 5 exhibited the time-dependent transient absorption spectrum of 0.25 mM MF in ACN after 266 nm laser excitation. As can be seen, a broad transient absorption band with maximum absorption around 345 nm was

obtained at the pulse end, and the presence of two different triplet scavengers, namely, oxygen and acrylamide were found to hardly affect its formation, indicating that this species could not be assigned to the triplet excited state of MF (Fig. S10). In accordance, the triplet excited states of benzophenone (BP) and some of its derivatives were reported to show maximum absorption around 525 nm (Ryuzi et al., 1997) (Fig. 5(d)). We looked after a signal in this wavelength range, but absorbances were very weak (Fig. 5(c)). On the other hand, the spectrum analogy of our transient (λ_{max} = 345 nm) with those of *m*-benzoylbenzyl radical (*m*-BBR, λ_{max} = 335 nm) and *p*-benzoylbenzyl radical (*p*-BBR, λ_{max} = 320 nm) (Yamaji et al., 2007) prompted us to assign it to the benzoylbenzyl radical of MF (MBR). Yamaji et al. (2007) reported that upon 266 nm excitation in LFP experiments, *m*-bromomethylbenzophenone (*m*-BMBP) would be excited to generate *m*-BBR, and this process could be achieved *via* an extremely fast ω-cleavage of both lowest singlet (S1) and triplet (T1) *m*-BMBP. In their case also, triplet *m*-BMBP could be hardly be detected by LFP (Yamaji et al., 2007). However, differently from m-BBR, MBR was found to be short-lived, undergoing a fast decay within 0.05 μs to give a relatively long-lived secondary intermediate (Fig. 5(a)). The shorter life-time of MBR might be attributed to the presence of methyl and methoxy substituents allowing fast intramolecular reactions to take place.

To further support the formation of MBR, DFT calculation were conducted to obtain the theoretical UV-Vis absorption of two potential intermediates, namely, MBR and the benzyl cation. The validation of this method was firstly checked by

calculating the absorption of parent MF. As seen, theoretical calculation exhibited a maximum absorption peak at 290 nm (Fig. S1), which was in a good agreement with the experimentally measured absorption peak at 292 nm (Fig. 2). In the case of MBR, calculation showed a maximum absorption peak around 350 nm (Fig. S11), matching well with the spectrum observed by LFP experiments (Fig. 5(a)). For the benzyl cation, the maximum absorption was calculated to be around 460 nm. Therefore, we have confirmed the homolytic cleavage of the C-Br bond of MF.

In addition, for the homolytic cleavage process, we further calculated the bond dissociation energy (BDE) of C-Br bond on the basis of $\triangle H$ derived from thermochemical parameters from the optimized MF, MBR and bromine atom at an ideal condition (298.15 K and 1 atm). A BDE value for C-Br bond cleavage of MF was estimated to be 67.9 kcal mol^{-1}, which was lower than that of the lowest singlet state (S1, 89.1 kcal mol^{-1}) and triplet state (T1, 71.2 kcal mol^{-1}). Based on the experimental and theoretical results, an energy diagram for the deactivation and bond dissociation processes of excited MF is depicted in Scheme.1. Similar to *m*-BMBP, the excited singlet state of MF produced an excited triplet state through intersystem crossing. Alternatively, S1 and T1 state could also undergo C–Br bond cleavage to produce MBR and bromine atom (Scheme. 1(a)). Bond dissociation from S1 was thermodynamically more favorable than from T1, and was proposed as the dominant pathway for C-Br bond cleavage.

Based on the products identification, DFT calculation and LFP experiments, the detailed debromination mechanism can be tentatively discussed. Once MBR was

generated, the formation of TP3 required access to hydrogen atoms. They cannot be supplied by water, while the methyl group on the ortho-position was likely to act as a good hydrogen donor. In this case MBR might undergo an extremely fast intramolecular rearrangement process, leading to the rapid disappearance of MBR and to the formation of the corresponding secondary transient species (Fig. 5(b)). The new produced intermediate can experience an intermolecular hydrogen extraction from another MF molecule, giving TP3, or undergo a series of further oxidation processes to generate the methyl oxidation products.

In conclusion, a detailed diagram of MF photolysis process is illustrated in Scheme. 1(b). Specifically, upon irradiation, MF would be excited to singlet state, which could either directly break the C-Br bond or undergo ISC process to give ^{3}MF*. Similarly, ^{3}MF* undergoes C-Br bond homolytic cleavage reaction to give rise to MBR and •Br. Meanwhile, ^{3}MF* can also transfer its energy to dissolved oxygen, resulting in the formation of $^{1}O_2$, which in turn leads to the further oxidation of MF.

Fig. 5.

Scheme.1.

3.6 Toxicity prediction during MF transformation

In the present study, the acute and chronic toxicity of MF and photoproducts to fish, daphnia and green algae was evaluated through the ECOSAR procedure (Wei et al., 2018). The toxicity was classified according to the Globally Harmonized System of Classification and Labelling of Chemicals (Liu et al., 2019).

The acute toxicity values of MF were determined to be 1.20 mg L^{-1}, 0.88 mg L^{-1} and 1.85 mg L^{-1} for fish, daphnia and green algae, respectively. These results showed that MF was toxic for aquatic species, especially with lower trophic organisms such as daphnia (< 1.0 mg L^{-1}). The chronic toxicity values (Chv) of MF were predicted to be 0.16 mg L^{-1}, 0.17 mg L^{-1} and 0.85 mg L^{-1} for fish, daphnia and green algae, respectively, suggesting that MF was more likely to have a chronic toxic effect on organisms. Simultaneously, the trend in chronic toxicity for the three different trophic levels was consistent with acute toxicity. In summary, MF was obviously toxic to aquatic species and can disrupt the ecological balance of the aquatic environment, thus the detection and control of MF in the environment should be given more attention.

Predicting the toxicities of intermediate products during MF transformation was also particularly important (Karci. and Akin., 2014). Therefore, we calculated the toxicity changes of the intermediate products in the three MF photolysis pathways for fish, Daphnia and green algae, and the results were shown in Fig. 6. In ECOSAR predictions, toxicity actually decreases if the toxicity parameter (e.g., Lethal Concentration 50 (LC50)) decreases by at least one order of magnitude. From this perspective, the acute toxicity of intermediate products showed a decreasing trend only for Daphnia. For fish, all the debromination products (TP3 and TP6) and the methyl aldolization products (TP1, TP4 and TP7) were in the same order of magnitude of acute toxicity as MF. However, the acute toxicity of the methyl carboxylation products (TP2, TP5 and TP8) was directly decreased by two orders of

magnitude. The acute toxicity variation for green algae was similar with fish, but the toxicity of both TP6 and TP7 in the OH-debromination pathway was one order of magnitude higher than MF, indicating that toxicity was enhanced. A little different from the acute toxicity, the chronic toxicity of both debromination pathways showed a continuous decreasing trend only for green algae (except TP1). For fish and daphnia, the chronic toxicity of TP1, TP3, TP4, TP6 and TP7 remained in the same order of magnitude as MF. Aldehyde groups are known to be toxic to organisms through chemical bonding with amino bearing biomolecules such as enzymes, proteins, condensates, nucleic acid derivatives etc. *via* Schiff bases (Slomkowski, 1998; Sokolsky et al., 2006). This might be responsible to higher toxicity of intermediate products containing aldehyde groups. Notably, further oxidation of the aldehyde products to the carboxylate products decreased the toxicity rapidly. All the above results indicated that the toxicity of some intermediate products of three photolysis pathways may maintain or even increase, but the toxicity of the final carboxyl-containing product is greatly reduced, photodegradation is still a method that can safely reduce the ecotoxicity of MF in aquatic environment.

Fig. 6.

3.7 Effects of natural water constituents

3.7.1 Effect of nitrate

Nitrate is one of the most abundant anions in the aquatic environment whose photochemical reactivity can affect the photochemical fate of the organic contaminants (Mack and Bolton, 1999; Zuo et al., 2006). In the present study, the

photodegradation experiments of MF were conducted in 25 mL aqueous solutions containing 20 μM substrates with different nitrate concentrations (0, 5, 20 and 50 mM) under irradiation for 8 h and the results are presented in Fig. 7(a).

The photolysis rate of MF was observed to be significantly accelerated as the nitrate concentration increased. The rate constants were 0.055, 0.080, 0.143 and 0.167 h^{-1} for nitrate concentrations of 0, 5, 20 and 50 mM, respectively. •OH generated by nitrate photolysis is one of the major ROS accounting for the attenuation of pollutants in natural waters (George et al., 1988; Monica et al., 2013). Eqs (9)-(13) show how solar irradiation of nitrate leads to the formation of $•NO_2$ and •OH directly and indirectly *via* its photoisomerization to peroxynitrite/peroxynitrous acid (Kotnik et al., 2016; Vione et al., 2009). In this work, the •OH generated by nitrate photolysis was identified by EPR analysis (Fig. S12(a)), thus was proposed to be responsible for the decay of MF.

$$NO_3^- \xrightarrow{h\nu} \bullet NO_2 + \bullet O^- \quad (9)$$

$$\bullet O^- + H_2O \rightarrow \bullet OH + OH^- \quad (10)$$

$$NO_3^- \xrightarrow{h\nu} ONOO^- \quad (11)$$

$$ONOO^- + H^+ \text{ € } HOONO \quad (12)$$

$$HOONO \rightarrow \bullet NO_2 + \bullet OH \quad (13)$$

3.7.2 Effect of bicarbonate

Bicarbonate is the major anion responsible for water alkalinity. Bicarbonate has been reported to play an important influence on the indirect photolysis of organic pollutants in natural waters (Huang and Scott, 2000). Experimental results show that

the presence of bicarbonate barely affected the photolysis process of 20 μM MF (Fig. 7(b)). The corresponding rate constants were found to be equal to 0.059, 0.064, 0.059 and 0.062 h^{-1} as the bicarbonate concentration increased from 0 to 20 mM. Bicarbonate can interact with •OH to form carbonate radicals (CO_3•$^-$) (Eq. (14)) (Huang and Scott, 2000), which is a strong electrophile that can rapidly oxidize phenolic compounds (Neta et al., 1988). However, our quenching experiments ruled out the involvement of •OH in the direct photolysis of MF.

$$HCO_3^- + \bullet OH \rightarrow CO_3\bullet^- + H_2O \quad (14)$$

In addition, the presence of bicarbonate can also affect the photochemical transformation of organic compounds *via* altering the solution pH (Ji et al., 2012). However, this effect is expected to be not important due to the pH-independent photolysis of MF (vide supra).

3.7.3 Effect of humic acid

Humic substances are ubiquitous in soils and surface waters which are formed by decomposition of living organisms. In the aquatic environment, they are able to compete with pollutants to absorb photons for photolysis as an inner filter (light screener) (Luciano et al., 2009; Monica et al., 2003). Meanwhile, as a photosensitizer, they are capable of affecting the persistence of organic pollutants in the environment through energy transfer or reaction with photochemically produced reactive intermediates (PPRI) (Manjun et al., 2006; Richard et al., 1997; Yong et al., 2009; Zepp et al., 1985). In view of this, the photolysis of 20 μM MF was conducted in the presence of humic acids (HA) at a concentration ranging from 0 and 20 mg L^{-1}, and

the results are shown in Fig. 7(c).

The kinetic rate constants were determined to be 0.056, 0.064, 0.094, and 0.139 h^{-1} at HA concentrations 0, 2, 5 and 20 mg L^{-1}, respectively. Compared with ultrapure water controls, the photodegradation rate of MF in the presence of HA was significantly higher. HA was a macromolecule with a complex structure, containing a large number of chromophores comprised of benzene ring, carboxyl, carbonyl, and methoxyl moieties (Laurentiis et al., 2013). The HA-enhanced photolysis of MF may occur either through energy transfer from the excited HA to the ground state MF, or *via* oxidation by PPRI, such as •OH and $^1O_2$, which promote the photolysis of MF (Rajib et al., 2011). The involvement of •OH and $^1O_2$ in the photolysis of MF in the presence of HA was confirmed by our EPR analyses, as illustrated in Fig. S12(b).

From this section, it is clear that natural water constituents such as nitrate and humic acid have significant impacts on the photodegradation rate of MF. Therefore, direct photolysis is not expected to be the main photodegradation pathway of MF in the environment, and further studies are required to better understand its photoreactivity.

Fig. 7.

4. Conclusions

In this study, the photodegradation of MF in aqueous solution under simulated solar irradiation was investigated, including photolysis kinetics, reactive intermediates, products identification, photolysis pathways and mechanisms. Under simulated sunlit, direct photolysis rate constant of MF in aqueous solution was 0.056 ± 0.003 h^{-1},

corresponding to a half-life of 12.4 ± 0.6 h, indicating that photochemical degradation was an important pathway for the removal of the fungicide MF in nature. Three different pathways were proposed based on HR-MS analysis, including methyl oxidation, debromination and replacement of Br by OH group. Among them debromination process was identified as the main path. Homolytic cleavage of the C-Br bond from both ^{1}MF* and ^{3}MF* is likely to take place. MF is also likely oxidized by $^{1}O_2$ produced by energy transfer from ^{3}MF* to O_2. NO_3^- and humic acid were found to enhance the decay of MF, while HCO_3^- exhibited no significant effect on MF photolysis. Toxicity assessment revealed that photochemical transformation was an efficient process to reduce the ecotoxicity of MF. This study provided important information for the understanding of photolytic behavior of MF in the aqueous environment.

Declaration of competing interest

There is no declaration of interest in this investigation. The authors declare no competing financial interest.

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