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Review

Parabens transformation products in water and their (eco) toxicological implications

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ABSTRACT

Parabens are used as preservatives in personal care products and many other products of daily use. Their removal in wastewater treatment plants (WWTP) is inefficient, allowing their spread into the environment. A great diversity of biological and chemical transformations occurs in these water systems leading to the formation of parabens transformation products (PTP). The presence and persistence of PTP in water sources are primarily affected by the type of tertiary wastewater treatment employed and the extent to which paraben-containing products are used. Despite laboratory evidence of PTP formation, their detection in real-world water systems remains limited. Few reports described their presence in real WWTP, rivers, swimming pools, and even drinking water (DW), occasionally at concentrations surpassing their parent counterparts. Among them, *para*-hydroxybenzoic acid (PHBA) is the most frequently detected, reaching concentrations up to 200 μg/L in WWTP influents, followed by hydroxylated parabens (1 μg/L), as well as chlorinated and brominated parabens (<0.1 μg/L). The toxicological implications of these PTP raise concerns, considering the health risks associated with parabens such as their potential endocrine disruption and toxicity. This study provides a comprehensive analysis of PTP formation during water treatment processes and its prevalence in water sources in real and laboratory conditions. Their environmental and public health impacts are also addressed, highlighting the need to invest in the monitoring of PTP in water systems using integrated water management approaches and promote community education to reduce the use of these compounds, safeguarding environmental and human health.

1. Introduction

The detection of more than 700 disinfection by-products (DBP) in treated water has raised awareness of their threat to the environment and public health [1–3]. DBP, formed during water disinfection processes, result from the interaction of disinfectants with organic matter, anthropogenic contaminants, bromide, or iodide [1]. Among various anthropogenic contaminants, parabens are commonly used as preservatives in personal care products – PCP (488 to 8200 μg/g) [4], pharmaceuticals (2689 μg/g) [5], and in food products (1113 μg/g) [6]. The most common parabens include methylparaben (MP), ethylparaben (EP), propylparaben (PP), *iso*-propylparaben (iPP), butylparaben (BP), *iso*-butylparaben, and benzylparaben (BzP) [7]. Parabens, due to their wide use, are constantly released into the environment, especially into water sources [8]. The presence of parabens has been reported worldwide including in Africa [9], America [10], European Union [11], Asia [12], and Oceania [13].

These compounds have been found in wastewater treatment plants (WWTP) and surface water at concentrations exceeding 100 μg/L, and even in drinking water (DW) at concentrations up to 6 μg/L [8].

To ensure low environmental impact and a safe release into the environment, wastewater undergoes different treatment stages in a WWTP (initially, a primary treatment followed by a biological treatment/secondary treatment and, eventually a tertiary treatment). The tertiary treatment can include additional biological processes for nutrient removal and/or disinfection to treat recalcitrant chemicals or inactivate biological contaminants [3]. A disinfection step can be performed through chlorination, chloramination, ozonation, or ultraviolet (UV) irradiation [3]. However, these conventional strategies are

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Nomenclature

2,4,5-triOH-MP	tri-hydroxylated methylparaben
2-OH-3,5-diCl-PP	mono-hydroxylated-di-chlorinated propylparaben
3,5-diBr-BzP	di-brominated benzylparaben
3,5-diBr-BP	di-brominated butylparaben
3,5-diBr-EP	di-brominated ethylparaben
3,5-diBr-iBP	di-brominated *iso*-butylparaben
3,5-diBr-iPP	di-brominated *iso*-propylparaben
3,5-diBr-MP	di-brominated methylparaben
3,5-diBr-MP	di-brominated methylparaben
3,5-diBr-PP	di-brominated propylparaben
3,5-diCl-BP	di-chlorinated butylparaben
3,5-diCl-BzP	di-chlorinated benzylparaben
3,5-diCl-EP	di-chlorinated ethylparaben
3,5-diCl-MP	di-chlorinated methylparaben
3,5-diCl-PP	di-chlorinated propylparaben
3,5-diOH-MP	di-hydroxylated methylparaben
3-Br-BP	mono-brominated butylparaben
3-Br-EP	mono-brominated ethylparaben
3-Br-iBP	mono-brominated *iso*-butylparaben
3-Br-MP	mono-brominated methylparaben
3-Br-PP	mono-brominated propylparaben
3-Cl-BP	mono-chlorinated butylparaben
3-Cl-BzP	mono-chlorinated benzylparaben
3-Cl-EP	mono-chlorinated ethylparaben
3-Cl-iBP	mono-chlorinated *iso*-butylparaben
3-Cl-MP	mono-chlorinated methylparaben
3-Cl-PP	mono-chlorinated propylparaben
3-OH-EP	mono-hydroxylated Ethylparaben
3-OH-MP	mono-hydroxylated methylparaben
AhR	Aryl hydrocarbon receptors
AOPs	Advanced oxidative processes
ARG	antibiotic resistance genes
BA	Benzoic acid
BP	Butylparaben
BzP	Benzylparaben
C-DBP	Carbonaceous DBP
ChV	Chronic toxicity value
DBP	Disinfection by-products
DHBA	di-hydroxybenzoic acid
DW	Drinking water
DWDS	Drinking water distribution systems
DWTP	Drinking water treatment plants
EC$_{50}$	Effective concentration to inhibit 50 % of the population
ECOSARs	ecological structure – activity relationships
EP	Ethylparaben
EPA	Environmental Protection Agency
EPS	Extracellular polymeric substances
ERα	Human estrogen receptor α
FDA	Food and Drug Administration
GR	Glucocorticoid receptor
GRAS	Generally recognized as safe
HAA	Haloacetic acids
HAN	Haloacetonitrile
HEK	Human embryonic kidney
HPLC	High-performance liquid chromatography
HQ	Hazard quotient
iBP	*iso*-butylparaben
IC$_{50}$	Half maximal inhibitory concentration
iPP	*iso*-propylparaben
k$_{app}$	Apparent second-order rate constant
LC	Liquid chromatography
LC$_{50}$	Lethal concentration to kill 50 % of the population
LOEC	Lowest-observed-effect concentration
MEC	Measured environmental concentrations
MP	Methylparaben
MS	Mass spectrometry
N-DBP	Nitrogenous DBP
NOEC	No-observed-effect concentration
PCP	Personal care products
PHBA	*para*-hydroxybenzoic acid
PhP	Phenylparaben
PNECs	Predicted no-effect concentrations
PP	Propylparaben
PTP	Parabens transformation products
PtP	Pentylparaben
REC$_{25}$	Concentration of the compound needed to achieve 25 % of the maximal response to β-naphthoflavon
REC$_{60}$	Concentration of the compound needed to achieve 60 % of the maximum inhibitory activity of 4-hydroxytamoxifen
THM	Trihalomethanes
TR	Thyroid receptors
US	United States
UV	Ultraviolet irradiation
WHO	World Health Organization
WWTP	Wastewater treatment plants

inefficient for removing parabens [14], making the formation of parabens transformation products (PTP) unavoidable [15]. The formation of PTP across a WWTP begins in the pretreatment step where the transformations that occur are expected to be the same as those in the environment: photodegradation, potentially biodegradation, and even hydrolysis with low probability [16]. During biological treatments, microorganisms can biotransform parabens into a diversity of metabolites [3]. An additional disinfection step, where chlorine is typically used, causes electrophilic aromatic substitution and the consequent formation of halogenated parabens [17]. At this stage, the formation of other DBP resulting from the interaction with organic and inorganic precursors in water may also occur, leading to further interaction between DBP and PTP [3]. It is important to consider that DBP are a broad category of compounds formed during the water treatment process, and PTP are specific by-products that arise from the transformation of parabens during the water treatment process.

Nevertheless, the conversion of parabens into other transformation products can emerge in household drains, owing to the widespread use of PCP-containing parabens [18] or during swimming pool water disinfection, facilitating their entrance into WWTP [19]. In WWTP, wastewater suffers diverse treatments to remove the presence of these contaminants. However, these treatments do not remove 100 % of PTP or their parent parabens, and WWTP effluents often contain residual concentrations of these contaminants [20]. Consequently, when these WWTP effluents are discharged into surface waters (rivers, lakes), PTP and their parent parabens contaminate these water sources [21]. Furthermore, the presence of PTP in surface waters may also be due to human discharges, such as sewage overflows, industrial wastewater, runoff from agricultural areas or natural parabens photolysis and microbial degradation [21]. Additionally, PTP have already been detected in human fluids due to parabens metabolization in the human body or due to direct exposure to PTP, resulting in the excretion of these compounds through the urine with consequent discharge into sewage [22].

Consequently, these compounds can enter groundwater through percolation from surface waters or agricultural activities [23]. Contaminated water does enter drinking water treatment plants (DWTP) to be effectively treated and safely delivered to DW consumers. Even after water treatment in WWTP further disinfection treatment of treated effluent water in DWTP occurs and favours the interaction between the remaining parabens and disinfectants, contributing to the formation of additional PTP, and/or even further transformation of the existing ones [1].

Some researchers have studied the formation of chlorinated, brominated, and hydroxylated parabens in aquatic environments [2,3]. However, using controlled laboratory conditions, without considering the complexity of environmental samples and ecological interactions [2]. Indeed, research is mostly focused on the presence of parabens in water sources, ignoring that parabens can undergo natural or induced chemical transformations during water treatment, leading to the formation of PTP [21].

The highest environmental concentrations of PTP have been reported in WWTP influent. For example, in the United States (US), some authors found concentrations of PHBA up to 293000 ng/L in a specific WWTP, being the average of the determination of this compound in the referred WWTP of 5280 ng/L [24]. Hydroxylated and chlorinated parabens are typically present at lower levels in these systems and their concentrations vary across regions. In India, the maximum concentration of mono-hydroxylated MP (3-OH-MP) found in WWTP influents was 1050 ng/L [25], while in Spain, di-chlorinated MP (3,5-diCl-MP) reached a maximum concentration of 90 ng/L [26]. Surface waters also exhibit varying concentrations of chlorinated and brominated parabens. A Malaysian river contained a maximum concentration of 13400 ng/L of chlorinated parabens [12], while a Japanese river contained 110 ng/L of brominated parabens [27]. Moreover, the presence of chlorinated parabens is frequently reported in swimming pool waters ranging between 0.64 and 1122 ng/L, due to common swimming pool disinfection practices and the introduction of parabens by swimmers [19]. Regarding the presence of PTP in DW, there is only one study reporting the presence of di-chlorinated parabens (3,5-diCl-MP and 3,5-diCl-EP) and PHBA in Texas, at concentrations ranging from 62 to 711 ng/L [21].

The generation of these transformation products and their presence along various water sources may impact the ecosystems as these products are known to be toxic to aquatic organisms including green algae, fish, and *Daphnia* [28]. Although there are no consensual opinions, PTP (in particular halogenated parabens) may have a higher potential risk to humans than the corresponding parent parabens, due to their higher toxicity [29]. Seeing these impacts and considering the interconnection between different water cycles and treatment stages, it is important to adopt a One-Water approach to manage water resources and prioritize the presence of both parabens and their transformation products, avoiding potential risks resulting from their inefficient management [30].

This review is pioneer in providing a comprehensive analysis of the scientific evidence of the presence of PTP in real water systems, aiming to understand the primary exposure pathways and anticipate their possible chemical transformations. By exploring the occurrence and fate of these products in water sources along with recent findings on their human toxicity and ecotoxicity, this review highlights potential environmental and health risks that have been underrepresented in previous research. Understanding the mechanisms involved in PTP synthesis, their recalcitrance, and their environmental and public health threats is of relevance for informed decision- and policy-making, and improved risk assessment in the context of water quality protection.

2. Water treatment and formation of PTP

Research on PTP began in 2006, revealing that even free chlorine levels in tap water (0.5 mg/L) can rapidly generate chlorinated PTP within a few minutes [31]. Indeed, parabens can be converted into a high number of transformation products along WWTP, depending on the type of water disinfection treatments employed [16]. Furthermore, due to the widespread use of disinfectants in daily routine, the natural presence of DBP in aquatic environments is also increasing, and parabens can also react with existing DBP, generating new PTP [32].

The initial stage of wastewater treatment generally involves the use of clarifiers to remove suspended particles and solids from water (e.g. coagulation, sedimentation) [3]. Although these water treatment processes are not designed to remove emerging contaminants (including parabens) from water, the removal of hydrophobic parabens may occur with efficiencies of up to 30 % [14]. During this stage, the transformation products that are expected to form are those typically generated in the environment through natural degradation pathways such as photodegradation (exposure to natural light), hydrolysis under basic conditions or microbial interactions [3].

Fig. 1 illustrates the potential transformation products of parabens occurring along a typical WWTP. Photodegradation involves the addition of hydroxyl radicals ($OH^{\bullet}$) to the aromatic ring of parabens, and the removal of hydrogens, resulting in the transformation of parabens into hydroxylated compounds (Fig. 2A) [33]. These hydroxyl radicals ($OH^{\bullet}$) are previously generated from the photolysis (when exposed to ultraviolet irradiation or other forms of high-energy radiation) of molecules commonly present in water such as hydrogen peroxide (Eq. (1)), nitrate (Eq. (2)), and nitrite ions (Eq. (3)) [34]:

$$H_2\,O_2 \rightarrow 2OH^{\bullet} \tag{1}$$

$$NO_3^- \rightarrow NO_2 + O^{\bullet};\ O^{\bullet} + H_2\,O \rightarrow 2\ OH^{\bullet} \tag{2}$$

$$NO_2^- \rightarrow NO + O^{\bullet};\ O^{\bullet} + H_2\,O \rightarrow 2\ OH^{\bullet} \tag{3}$$

On the other hand, biodegradation plays a crucial role in converting parabens into PHBA and the corresponding alcohol through the hydrolysis of the ester bond by esterases, with further transformation into phenol via decarboxylases (with the removal of the carboxyl group from PHBA) (Fig. 2B and C) [35]. Although PHBA is considered the major degradation product shared by original parabens, other degradation products of parabens (in particular of MP), such as oxalic, propanedioic, and fumaric acids can also occur during this first stage of water treatment [36].

During secondary water treatment, the remaining parabens and PHBA may undergo hydroxylation due to biotransformation, in the aerobic process [16]. This transformation depends on the presence of microorganisms and environmental factors like temperature, pH, and dissolved oxygen levels. In aerobic conditions, the transformation of parabens is potentiated by bacteria. In this case, parabens are transformed into PHBA via esterase activity (Fig. 2B) and subsequently can be converted to phenol (Fig. 2C) or benzoic acid (Fig. 2D) via decarboxylase or dehydroxylase, respectively. However, these PTP do not occur under anaerobic conditions [37].

The most common tertiary water treatment is disinfection including chlorination, bromination, ozonation, ultrafiltration, and/or UV irradiation, which lead to the formation of chlorinated, brominated, and hydroxylated parabens [3]. However, the formation of PTP during ultrafiltration is relatively uncommon [16]. The process of formation of these compounds for each disinfection treatment will be described in sections 3.1 to 3.4.

Considering the wide range of PTP formed within WWTP, it becomes necessary to identify which of these transformation products are being formed at specific stages of the water treatment. The identification of these PTP comprises the use of the analytical methods used for the original parabens, with high-performance liquid chromatography (HPLC) being the most commonly applied for the identification of chlorinated [38] and hydroxylated parabens [25]. In general, higher retention times are expected for halogenated parabens since more hydrophobic compounds often correspond to longer retention times [38]. On the other hand, lower retention times are expected for PHBA, dihydroxybenzoic acid (DHBA), and mono-hydroxylated parabens in relation to the parent parabens [39].

Techniques for PTP identification and quantification are presented in

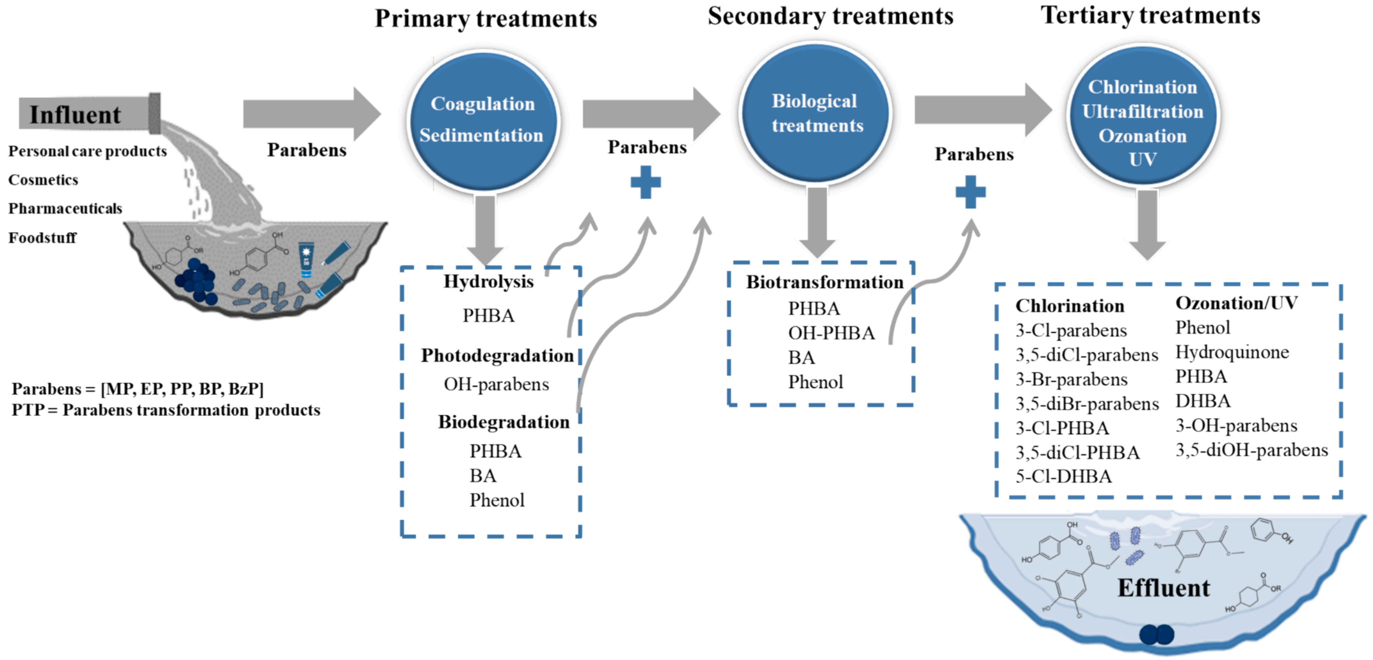

Fig. 1. Predictive formation of parabens transformation products (PTP) along WWTP using different water disinfection treatments. Legend: 3,5-diBr – di-brominated; 3,5-diCl – di-chlorinated; 3,5-diOH – di-hydroxylated; 3-Br – mono-brominated; 3-Cl – mono-chlorinated; 3-OH – mono-hydroxylated; BA – benzoic acid; BP – butylparaben; BzP – benzylparaben; DHBA – dihydroxybenzoic acid; EP – ethylparaben; MP – methylparaben; PHBA – *para*-hydroxybenzoic acid; PP – propylparaben. Created with BioRender.com.

Table S1 and Fig. S1 (Supplemental Information A). These analytical methods can provide accurate measurements and expand our comprehension of paraben degradation pathways under realistic conditions, helping to correctly identify PTP.

2.1. *Chlorination and bromination*

Chlorination is the most commonly used disinfection strategy in WWTP and typically involves the addition of 1–15 mg/L of chlorine for 30 min [3]. Therefore, the most commonly detected PTP in these water systems are chlorinated parabens which are mainly promoted by the facile reaction between the phenolic hydroxyl group of parabens and hypochlorous acid (HOCl) present in water [40]. In addition, chlorination of parabens can also occur naturally in rivers contaminated with residual concentrations of chlorine [41].

The primary chlorination mechanism of parabens involves the addition of chlorine to the aromatic ring through electrophilic aromatic substitution (replacement of a hydrogen atom on the benzene ring by chlorine cation at the positions ortho or para to the hydroxyl group), with the possibility of a second addition, and, in exceptional cases, even a third addition [42]. This results in the formation of mono- and di-chlorinated parabens, and mono- and di-chlorinated forms of PHBA (Fig. 1). The chemical reactions illustrating this phenomenon are presented in Fig. 2E and F.

The chlorination of parabens appears to follow an apparent second-order rate constant (k_{app}) with values of $9.65 \times 10^{-3}\ M^{-0.614} \cdot s^{-1}$, $1.77 \times 10^{-2}\ M^{-1.019} \cdot s^{-1}$, $2.98 \times 10 - 2\ M^{-0.851} \cdot s^{-1}$, and $1.76 \times 10^{-2}\ M^{-0.860} \cdot s^{-1}$ for MP, EP, PP, and BP, respectively, at pH 7.0 [41]. However, this process is also influenced by the pH and temperature, being less favourable for higher pH levels but more favourable with increasing temperatures [41]. Moreover, chlorination of parabens is also slower in the presence of ammonium and humic acids [41].

In the natural environment, chlorinated parabens exhibit significantly longer half-lives, exceeding a week [26], compared to the few hours observed by the original parabens, which readily biodegrade under aerobic conditions [7]. Indeed, PTP, such as PHBA, and dichlorinated parabens tend not to degrade further in the absence of biological activity, thus increasing their presence in water sources [3]. This may make their removal from water difficult in comparison to their parent parabens, highlighting their increased persistence.

On the other hand, bromine, existing naturally in water as bromide ions (Br^-), can be introduced into water from anthropogenic sources or may be used as a disinfectant. Consequently, the formation of hypobromous acid (HOBr) from free bromide (Eq. (4)) leads to the bromination of parabens and PHBA also through electrophilic aromatic substitution as in the chlorination mechanism (Fig. 2E and F) [3].

$$Br^- + HOCl \rightarrow HOBr + Cl^- \tag{4}$$

Brominated parabens may undergo further reactions, including ipso substitution, which involves the exchange of Br^+ with Cl^+ [43]. During ipso substitution, a chlorine cation can replace a bromine cation, resulting in a chlorinated paraben and a bromonium ion. Psoras et al. [43] reported that chlorination of mono-brominated EP (3-Br-EP) yielded 3,5-diCl-EP and di-brominated EP (3,5-diBr-EP) in just 2 min. These authors also reported for the first time the formation of polybrominated trihalomethanes (THM) resulting from the ipso substitution of chlorine in (mono)brominated aromatic compounds (such as brominated PTP) and further incorporation of bromine in THM precursors present in water (Supplemental Information B – Fig. S2). By definition, THM are chemical compounds in which three of the four hydrogen atoms of methane (CH_4) are replaced by halogen atoms. This is of utmost concern since THM have been linked to health diseases such as bladder and colon cancer, asthma, and reproductive dysfunction [43].

2.2. *Ozonation and PTP*

Ozonation has a great effect on the removal of parabens as ozone (O_3) is one of the most highly reactive oxidants [44]. During ozonation, parabens transform by direct hydroxylation of the aromatic ring or alkyl chain, leading to the formation of hydroxylated parabens (Fig. 3A) [33]. Direct oxidation by O_3 primarily adds hydroxyl groups to the aromatic ring as this process requires double bonds. In contrast, oxidation by $OH^\bullet$

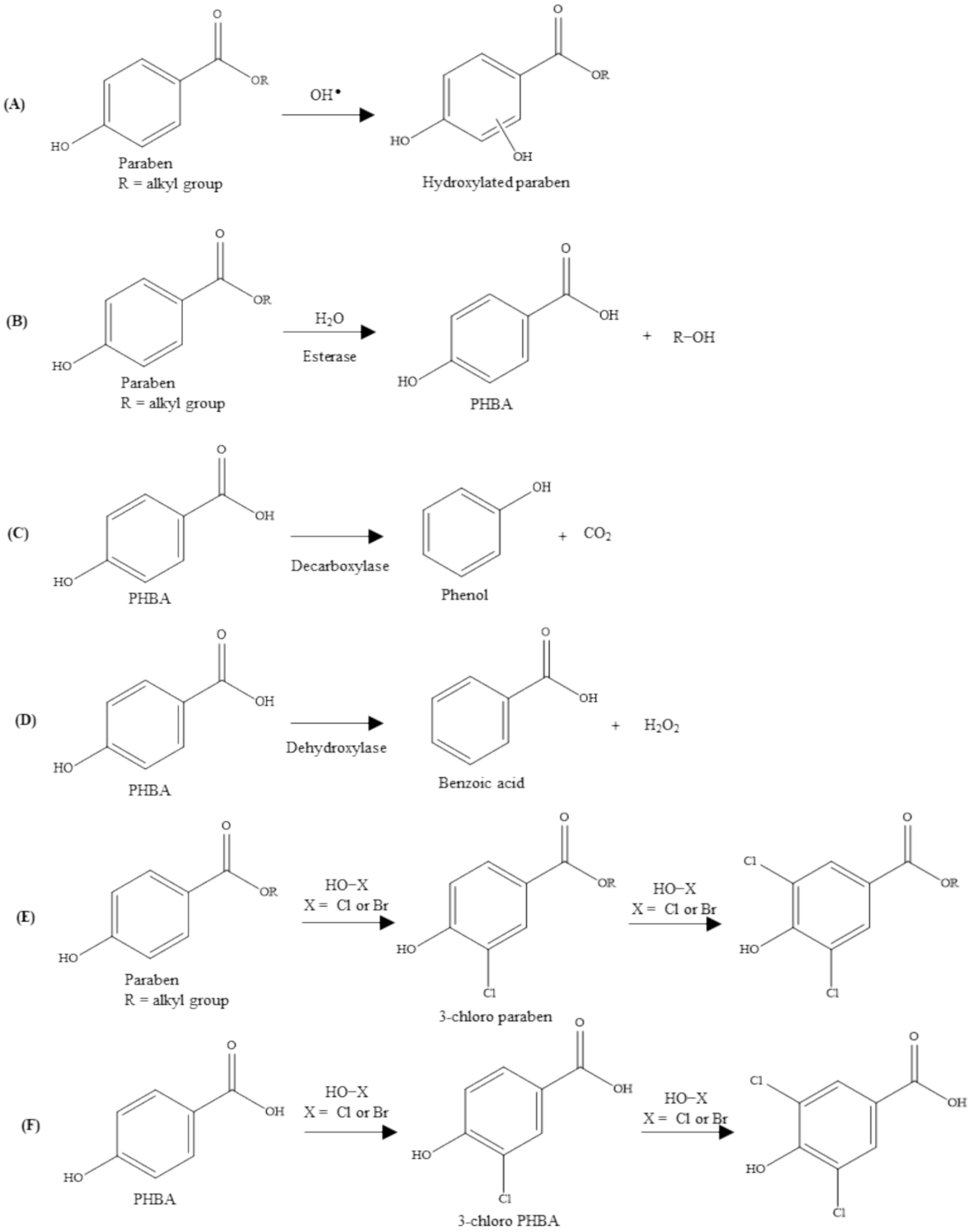

Fig. 2. Parabens and PHBA transformation occurring along typical WWTP through photodegradation (A), biodegradation (B, C, and D), chlorination, and bromination (E and F).

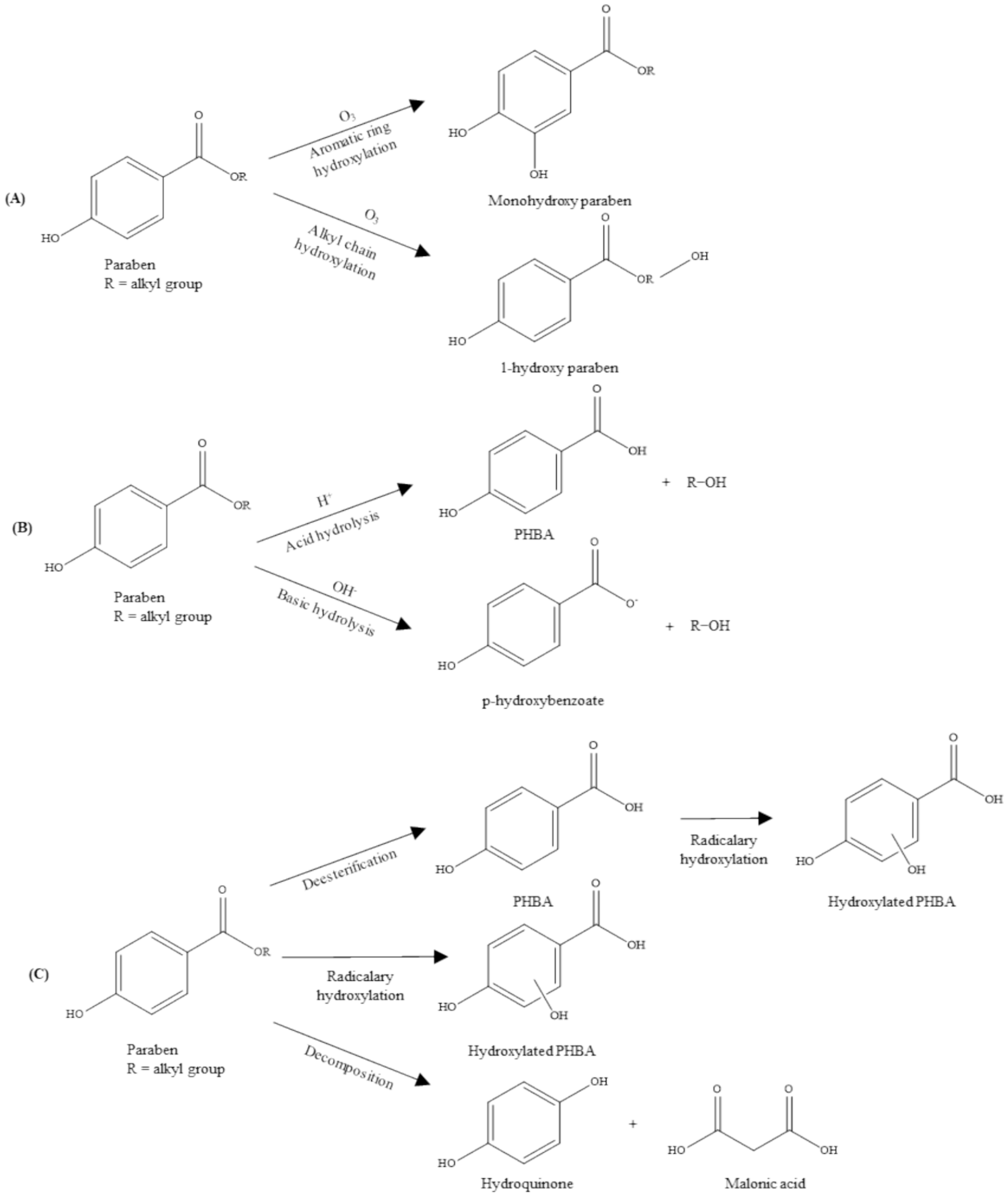

Fig. 3. Parabens transformation occurring during ozonation and UV disinfection (A – direct hydroxylation of the aromatic ring or alkyl chain [33]; B – acid and base hydrolysis [45]; C – de-esterification, radical hydroxylation, and decomposition [51]).

can introduce functional groups into the side chains of parabens [3] (Fig. 3A). Furthermore, parabens can undergo both acid and base hydrolysis during this process (Fig. 3B). Acid hydrolysis occurs at pH values below 4, while basic hydrolysis occurs at pH values above 10 [45]. Additionally, O_3 can also react with water and form hydrogen peroxide (H_2O_2), which subsequently decomposes into $OH^\bullet$, further interacting with parabens [46]. The reaction rates of oxidation of parabens by $OH^\bullet$ are higher than those for oxidation by O_3 [47].

Tay et al. [47] reported the detection of mono-, di-, and tri-hydroxylated parabens, and major breakdown products of parabens such as hydroquinone (which is a harmful by-product consisting of a benzene ring with two hydroxyl groups attached to the 1 and 4 positions) and PHBA, during ozonation under laboratory conditions. In another study, Gomes et al. [48] examined the use of volcanic rocks to enhance the removal of parabens by ozonation, resulting in the total degradation of parabens and the formation of PHBA, DHBA, and hydroquinone as paraben transformation products. The ozonation of parabens is affected by the physicochemical characteristics of the aqueous matrix since it affects O_3 decomposition and consequently the formation of $OH^\bullet$ [49]. Michael-Kordatou et al. [49] assessed the ozonation of EP in different aqueous matrices (ultrapure water, bottled water, synthetic humic acid solution, and a secondary wastewater effluent) at laboratory conditions. They reported the degradation rates for EP in different matrices in descending order: ultrapure water > bottled water > wastewater effluent > humic acid solution [49]. This occurs because aqueous matrices richer in carbonate species inhibit O_3 decay to $OH^\bullet$, hence promoting ozonation through direct oxidation [49]. However, while EP hydroxylated transformation products showed preservation of the aromatic ring of the original paraben, structural changes in the ester chain were observed [49].

During ozonation, it is also important to consider the possible formation of bromate (BrO_3^-), particularly in the presence of water containing naturally residual concentrations of bromide as described in the previous section. This can lead to the subsequent formation of brominated parabens, potentially impacting water safety [49]. Due to the frequency of this phenomenon, the US Environmental Protection Agency (US EPA) has already established the maximum value of 10 μg/L allowed for BrO_3^- in DW [50].

2.3. UV disinfection and PTP

UV disinfection generally uses UVC irradiation (200 to 280 nm) that encompasses the absorption wavelength of parabens (254 nm), allowing their direct excitation, which is a key factor for their degradation [3]. UV disinfection generates $OH^\bullet$, which can either oxidize parabens, often leading to the addition of an OH group or hydrolyze them to form PHBA (as represented in Fig. 3A and B) [47]. A recent study examined the degradation process of parabens (MP, EP, and BP) with UV irradiation and identified three main mechanisms: de-esterification, radical hydroxylation, and decomposition [51]. The chemical reactions illustrating this transformation are presented in Fig. 3C [51].

UV disinfection also facilitates the photodegradation of parabens that can be potentiated in the presence of chlorine [52] or even NH_2Cl [3]. Moreover, this disinfection strategy is often combined with ultrafiltration or the use of catalysts in advanced oxidation processes (AOPs), to promote $OH^\bullet$ formation and subsequently, the formation of hydroxylated parabens [53].

2.4. Advanced oxidation processes and PTP

AOPs are highly effective water treatment methods used to remove persistent organic pollutants, such as parabens. AOPs utilize powerful oxidizing agents, such as $OH^\bullet$ radicals to break down these complex chemical structures into harmless by-products [54]. Therefore, these processes may also lead to the formation of PTP [55]. In general, AOPs for parabens removal include can include the following methods: photolysis; photocatalysis; ozone-based technologies; Fenton oxidation; persulfate, peroxymonosulfate and H_2O_2 oxidation; electrochemical, and ultrasound technologies [54]. Lincho et al [54] have recently explained these technologies in detail. In this section, some examples of PTP formation resulting from AOPs in water treatment are presented.

The degradation of MP by UV/persulfate treatment led to the production of seven PTP: 3-OH-MP; 2,5-diOH-MP; 2,4,5-triOH-MP, PHBA, DHBA, BA, and phenol [55]. This process relies on the formation of reactive species, namely hydroxyl ($OH^\bullet$) and sulfate ($SO_4^{\bullet-}$) radicals resulting in the hydroxylation of the aromatic ring of MP or the hydrolysis of ester chain (as described for ozonation – section 2.2) [55]. The $SO_4^{\bullet-}$ radicals are more selective and can react easily with the aromatic ring through electron transfer reactions [55], whereas non-selective $OH^\bullet$ radicals can react with the aromatic ring or the aliphatic chain via either hydrogen abstraction or addition reactions [28,33]. The degradation of PP throughout activated persulfate was also studied, indicating that the hydroxylation of the aromatic ring of PP mainly takes place in the *ortho* position relative to the phenolic OH group [56].

Similar findings were also observed by researchers during the photodegradation of parabens (MP, EP, PP, and BP) using Fe(III)-citrate complexes [57] and during EP photodegradation using simulated solar irradiation and Ag_3PO_4, as photocatalyst [58].

Another study reported the degradation of a parabens mixture (EP, MP, PP) in ultrapure water and wastewater, at laboratory conditions, using UV-C/peroxymonosulfate[59]. The first pathway reported included hydroxylation and dealkylation reactions. Hydroxylation lead to the formation of dihydroxybenzoic acid esters from all parabens through the electrophilic attack of $OH^\bullet$ and direct electron transfer from parabens molecules to $SO_4^{\bullet-}$ with subsequent hydrolysis of parabens [59]. Further hydroxylation formed tri- and tetra-hydroxybenzoic acid esters, whereas dealkylation resulted in PHBA, DHBA, and BA [59]. Then, dimerization reactions may occur with an electron transfer that lead to the formation of radicals which can react via C-C coupling. Subsequent prolonged $OH^\bullet$ and/or $SO_4^{\bullet-}$ oxidation leading to the cleavage of the aromatic rings of PTP may also result in the formation of other PTP. Therefore, the pathways of parabens degradation through UV-C/peroxymonosulfate is know so far to include hydroxylation and dealkylation, radical coupling, and cleavage of the aromatic rings [59]. A similar EP degradation route was also reported using heat-activated persulfate oxidation [60]. Circa of 13 PTP from EP were identified suggesting that EP degradation occurred through: i) hydroxylation; ii) dealkylation, and; iii) oligomerization reactions. In this case, EP degradation was lower in the secondary treated wastewater due to the presence of natural organic matter and inorganic salts that compete with EP for the reactive oxygen species [60].

The degradation of MP in aqueous solutions using heterogeneous photocatalysis with TiO_2 and H_2O_2 was found to be mainly attributed to $OH^\bullet$ electrophilic attack, leading to the formation of hydroxylated MP [61]. Simulated solar radiation in the presence of N-doped TiO_2 catalysts for EP degradation also revealed the formation of PTP, with $OH^\bullet$ reactions appearing to prevail during the initial steps, as evidenced by the rapid formation of hydroxylated and dealkylated intermediates (3-OH-MP, PHBA, DHBA, phenol) [62]. Organic acids such as formic, oxalic, acetic, malonic, and succinic acids were also formed from EP degradation [62].

The electrochemical oxidation over a boron-doped diamond anode of EP was also evaluated involving hydroxylation and demethylation reactions, suggesting the role of electrogenerated $OH^\bullet$ in the process [63]. The oxidation of EP using this technique with Na_2SO_4 as the supporting electrolyte resulted in the formation of 21 PTP, including chlorinated and non-chlorinated dimers and trimers [63]. Interestingly, degradation in wastewater was faster than in ultrapure water, which may be explained by the presence of inorganic ions (e.g. Cl^-, SO_4^{2-}) that may lead to the indirect formation of reactive species [63].

3. Occurrence and quantification of PTP in real water sources

The occurrence and persistence of parabens and their transformation products in aquatic environments strongly depend on the degree of use of these products by the population [12]. Policies governing their use and strategies employed for wastewater management are also factors influencing the persistence of these products [12]. Most studies reporting removal rates of parabens and quantification of transformation products have been conducted in laboratory settings, often under specific and controlled environmental conditions. However, these controlled conditions do not fully represent the variable environmental factors found in nature, such as temperature fluctuations, pH variations, microbial activity, and the presence of other chemicals. Nonetheless, there are few studies monitoring the presence of PTP along real water sources, which may be explained by the associated limitations of sampling and analytical procedures in these real-world conditions as well as the challenges imposed by the dynamic and complexity of natural ecosystems [2].

Studies assessing the presence of PTP along real WWTP, in surface water and DW are discussed in the next section, as well as the presence of these pollutants in swimming pools, highlighting their potential health risks for swimmers.

3.1. Occurrence of PTP in WWTP

In WWTP, often considered the major reservoir of parabens, the occurrence of PTP is expected to be more pronounced in relation to other water sources [12]. However, to date, only eight studies have reported the formation of PTP along real WWTP. These studies involved wastewater facilities located in New York (US) [16,24], Texas (US) [21], Persian Gulf [64], China [65], Taiwan [12], India [25], and Spain [26]. The identification and respective concentrations of these PTP are presented in Table 1.

The most widely found PTP was PHBA [64]. This is already expected since PHBA is the most common degradation product of general parabens and is also present naturally in the environment [64]. This product

Table 1
Concentrations of parabens transformation products (PTP) found in WWTP influents and effluents.

Location	PTP	Concentration (WWTP influent)	Concentration (WWTP effluent)	Units	Reference
Plant A, New York, US	PHBA	1460–41100	555–5530	ng/L	[24]
	DHBA	220–1960	202–3170		
	3-OH-MP	12.7–346	21.90–478.0		
	3-OH-EP	0.36–340	0.3600–156.0		
Plant B, New York, US	PHBA	3890–293000	633.0–4040		
	DHBA	43.8–2270	107.0–427.0		
	3-OH-MP	28.7–102	9.010–84.50		
	3-OH-EP	0.36–176	0.3600–39.60		
Sludge, New York, US	PHBA	1450–3820	N	ng/g dry wt	
	DHBA	72.3–294			
	3-OH-MP	5.55–20.0			
	3-OH-EP	0.36–0.55			
Plant 1, Texas, US	PHBA	$C_{ave} = 27.9$	$C_{ave} = 26.2$	ng/L	[21]
	DHBA	$C_{ave} = 7.10$	$C_{ave} = 16.9$		
	3-OH-MP	$C_{ave} = 91.0$	$C_{ave} = 46.3$		
	3-OH-EP	$C_{ave} = 1.15$	$C_{ave} = 0.20$		
	3-Cl-MP	$C_{ave} = 0.110$	$C_{ave} = 0.0600$		
	3-Cl-EP	$C_{ave} = 0.260$	$C_{ave} = 0.160$		
	3,5-diCl-MP	$C_{ave} = 0.660$	$C_{ave} = 14.5$		
	3,5-diCl-EP	$C_{ave} = 0.260$	$C_{ave} = 5.70$		
Plant 2, Texas, US	PHBA	$C_{ave} = 26.2$	$C_{ave} = 16.6$		
	DHBA	$C_{ave} = 18.8$	$C_{ave} = 13.6$		
	3-OH-MP	$C_{ave} = 64.8$	$C_{ave} = 23.1$		
	3-OH-EP	$C_{ave} = 0.770$	$C_{ave} = 0.320$		
	3-Cl-MP	$C_{ave} = 0.104$	$C_{ave} = 0.0650$		
	3-Cl-EP	$C_{ave} = 0.138$	$C_{ave} = 0.0760$		
	3,5-diCl-MP	$C_{ave} = 0.375$	$C_{ave} = 1.19$		
	3,5-diCl-EP	$C_{ave} = 0.404$	$C_{ave} = 1.51$		
Hospital and municipal, Persian Gulf	BA	$C_{ave} = 10.000$	$C_{ave} = 10.000$		[64]
	PHBA	4443–58193	2121–55673		
Taiwan	3,5-diCl-MP	$C_{max} = 152$	N		[12]
	3-Cl-MP	$C_{max} = 5.40$	N		
	3,5-diCl-PP	$C_{max} = 161$	$C_{max} = 10.1$		
	3-Cl-PP	$C_{max} = 0.400$	N		
	3,5-diCl-MP	$C_{max} = 152$	N		
	3-Cl-BP	$C_{max} = 1.88$	N		
China	BA	$C_{max} = 32000$	44.4	mg/d/1000 people	[65]
	DHBA	$C_{max} = 3190.0$	12.9		
	3-OH-MP	$C_{max} = 37.000$	0.320		
	3-OH-EP	$C_{max} = 32.600$	0.420		
	PHBA	$C_{ave} = 590.00$	$C_{ave} = 123$	ng/L	[16]
	3,5-diCl-MP	$C_{ave} = 22.900$	$C_{ave} = 11.7$		
	3,5-diCl-EP	$C_{ave} = 30.000$	$C_{ave} = 19.3$		
India	BA	487–2360.0	N		[25]
	PHBA	92–31500	N		
	3-OH-MP	3.1–1050.0	N		
	3-OH-EP	1.4–392.00	N		
Spain	3-Cl-MP	$C_{max} = 61$	$C_{max} = 6.9$		[26]
	3,5-diCl-MP	$C_{max} = 90$	$C_{max} = 12$		

3,5-diCl-EP – di-chlorinated ethylparaben; 3,5-diCl-MP – di-chlorinated methylparaben; 3,5-diCl-PP – di-chlorinated propylparaben; 3-Cl-BP – mono-chlorinated butylparaben; 3-Cl-EP – mono-chlorinated ethylparaben; 3-Cl-MP – mono-chlorinated methylparaben; 3-Cl-PP – mono-chlorinated propylparaben; 3-OH-EP – mono-hydroxylated ethylparaben; 3-OH-MP – mono-hydroxylated methylparaben; BA – benzoic acid; DHBA – di-hydroxybenzoic acid; PHBA – *para*-hydroxybenzoic acid; C_{ave} – average concentration; C_{max} – maximum concentration; N – not quantified; PTP – parabens transformation products; WWTP – wastewater treatment plant.

Note: The data selection was obtained through an advanced search in PubMed and SCOPUS databases, by searching articles using specific keywords “Parabens transformation products AND Occurrence AND Wastewater”. The output results were 13 and 9 papers in SCOPUS and PubMed, respectively. Duplicated papers were not considered for analysis and for real water sources only 8 papers were considered for PTP occurrence in WWTP.

has reached maximum concentrations of 293000 ng/L (median concentration of 5280 ng/L) followed by DHBA with 2270 ng/L (median concentration of 720 ng/L), both found in a WWTP influent in New York [24]. Another study found that while the concentration of PHBA decreased downstream of a WWTP effluent, the concentration of DHBA remained stable [10]. This stability in DHBA concentrations may be attributed to the oxidation of PHBA to DHBA, which can help to explain the decrease in PHBA levels [10]. However, no consistent trend is observed in the most commonly detected halogenated paraben [66]. Besides that, among the different hydroxylated PTP, 3-OH-MP appears to be the most frequently detected and, in many cases, occurs at the highest concentrations, followed by 3,5-diCl-MP. For example, 3-OH-MP was detected at the highest concentration of 1050 ng/L in an Indian influent WWTP [25] and 3,5-diCl-MP was found at the maximum concentration of 152 ng/L in an influent of WWTP in Taiwan [12]. This is not surprising as MP is the most widely used paraben in the world [20].

Most of the studies analyzing the presence of PTP in WWTP refer to the US (Table 1), where significantly elevated concentrations have been detected. However, the study of the presence of these compounds is still geographically limited. Besides that, when comparing the concentrations of transformation products found in both WWTP influents and effluents, an overall decrease in their concentration from influents to effluents was observed, suggesting the removal of these compounds along WWTP processes. For instance, González-Mariño et al. [26] reported a 10-fold decrease of mono-chlorinated MP (3-Cl-MP) and 3,5-diCl-MP from the influent to the effluent of a Spanish WWTP. However, Wang and Kannan (2016) found an exception, with an increase in 3-OH-MP concentration from 346 to 478 ng/L, from WWTP influent to effluent. This may be explained by the hydroxylation of MP expressed by a higher concentration of 3-OH-MP compared to MP [24]. Similar trends were observed for mono-hydroxylated EP (3-OH-EP) and its respective parent paraben (EP) [24].

Penrose and Cobb [21] studied PTP formation along a municipal WWTP in Texas using different water disinfection treatments (*i.e.*, chlorination and UV disinfection). Although different treatments were used, the concentrations of PTP detected in both plants were similar [21]. As expected, PHBA was the most detected transformation product (ranging from 16.5 to 27.9 ng/L), followed by DHBA (ranging from 7.1 to 18.5 ng/L). Mono-hydroxylated species, namely 3-OH-MP and 3-OH-EP, were detected in more than 50 % of samples from both plants, with concentrations ranging from 0.2 to 1.2 ng/L (Table 1) [21]. However, the formation of 3-Cl-MP was favoured over mono-chlorinated EP (3-Cl-EP), being the first detected in more than 80 % of the effluent samples while the second one was only detected in 20 % of the effluent samples [21]. These results may be indicative of the higher MP pollution compared to EP. Furthermore, di-chlorinated parabens (3,5-diCl-MP and 3,5-diCl-EP) were found at higher concentrations than their respective mono-chlorinated parabens (3-Cl-MP and 3-Cl-EP), indicating successive chlorination of parabens [21]. That study also analyzed the impact of seasons on PTP concentrations, suggesting increased use of products-containing parabens during summer and spring, resulting in higher concentrations of sourced and transformation products detected [21].

In general, PTP and their intermediates are detected in WWTP influents and effluents at higher concentrations than their respective parent parabens [24,25,65]. Wang and Kannan [24] evaluated the presence of both parent parabens and PTP along two different WWTP located in New York. They found a higher concentration of PTP (5460 – 10000 ng/L) than those reported for original parabens (73 – 158 ng/L) in WWTP influents [24]. This tendency was also reported by Ma et al. [65] in China, with average concentrations of PTP of 6130 mg/d/1000 people and average concentrations of 283 mg/d/1000 people for original parabens in wastewater influents. In India, the same occurred with higher values reported for the sum of PTP (4110 – 34600 ng/L) than for original parabens (131 – 920 ng/L) in WWTP influents [25]. This reflects the high occurrence of PTP in wastewater samples originating from homes and industries, indicating the wide use of products containing parabens and their natural hydrolysis, photodegradation and biodegradation. Moreover, these products may also be excreted by humans and some of them, such as PHBA and DHBA, may also occur naturally in the environment [22]. For instance, PHBA can be found in plants, green tea, grape, and berry plants [65] and DHBA can be detected in onions and garlic [67].

In WWTP effluents, the higher concentrations of PTP found compared to parent parabens are explained by the fast biodegradation of parabens in aquatic environments due to their short environmental half-lives (10 to 35 h) [25]. Moreover, along WWTP treatments, parabens are transformed into PTP (as described in section 3), explaining the decrease of parabens concentration in these systems. The subsequent removal and/or degradation of PTP is more difficult to occur leading to their presence in WWTP effluents. This is also translated into higher removal efficiencies for parabens (89.6–99.9 %) than those for their transformation products (25.9 – 90.6 %) [24]. However, the “removal” of parabens is mainly a transformation of these compounds into their transformation products. Wang and Kannan [24] reported concentrations ranging from 1.96 to 5.57 ng/L for parent parabens and 2060 to 2550 ng/L for PTP in final WWTP effluents. Another study reported lower removal rates (33.9 – 40.7 %) for di-chlorinated-parabens (3,5-diCl-MP and 3,5-diCl-EP) compared to parent parabens – MP and EP (99.8 %) when using ultrafiltration and ozonation disinfection [16]. Karthikraj et al. [25] also reported lower removal efficiencies (28–76 %) for PHBA, DHBA, 3-OH-MP, and 3-OH-EP than that reported for the parent parabens (80–100 %) in five different WWTP. Consequently, in these five WWTP effluents, the total concentration of all PTP detected was much higher (2560 – 3800 ng/L) than that reported for their parent parabens (16 – 67 ng/L) [25]. Different chlorinated parabens such as 3-Cl-MP, mono-chlorinated PP (3-Cl-PP), mono-chlorinated BP (3-Cl-BP), 3,5-diCl-MP, and di-chlorinated PP (3,5-diCl-PP) were detected at concentrations of 152, 5.4, 161, 0.4, 1.88 ng/L, respectively, in a WWTP influent near Taichung River [12]. However, in the respective WWTP effluent, only 3,5-diCl-PP at 10.1 ng/L was detected [12]. Indeed, the relative concentration of 3,5-diCl-PP in water after chlorination was 1.49 times higher than that before chlorination [12]. Therefore, chlorination increased the concentration of 3,5-diCl-PP. The other PTP seem to be removed along the treatments applied in this WWTP [12].

In the European Union, the presence of three halogenated-MP (3-Cl-MP, 3,5-diCl-MP, and di-brominated MP – 3,5-diBr-MP) was only studied in Spain, and only 3-Cl-MP and 3,5-diCl-MP were detected at concentrations ranging from 0.01 to 0.1 mg/L [26]. Considering this, more studies evaluating the presence of these products along WWTP will be beneficial to deeply study the pathways of parabens transformation and to mitigate their presence.

3.2. Occurrence of PTP in surface waters

PTP can be released into surface waters through the direct discharge of WWTP effluents containing these pollutants and may be potentiated by the interaction between these pollutants and the residual concentrations of chlorine present in surface waters [10]. Exemplifying, although Chen et al. [12] only detected 3,5-diCl-PP on a WWTP effluent at 10.1 ng/L, an increase in the concentration of 3,5-diCl-PP to 11.7 ng/L and the generation of 3,5-diCl-MP and 3-Cl-MP at 1.80 and 0.90 ng/L, respectively, were observed in the Taichung River [12]. This corroborates the direct discharge of parabens or chlorinated parabens from WWTP effluents or other untreated sources into rivers, which subsequently react with residual concentrations of chlorine present in the river, forming chlorinated parabens [10].

To our knowledge, only seven studies are reporting the presence of PTP in surface waters, including rivers and seawater (Table 2). These studies were performed in the Brazos River (Texas – US) [3], Persian Gulf [64], Taichung River (Taiwan) and Malaysia [12], Beijing (China) [68], Mudun River (China) [66], Kitakami River (Japan) [27], and Shizuoka River (Japan) [18]. Among different PTP, PHBA remains the

Table 2

Concentrations of parabens transformation products (PTP) found in surface and tap water.

	Location	PTP	Concentration	Units	Reference
Surface water	Brazos River, Texas, US	3-Cl-MP	$C_{max} = 0.818$	ng/L	[21]
		3-Cl-EP	$C_{max} = 0.780$		
		3,5-diCl-MP	$C_{max} = 1.53$		
		3,5-diCl-EP	$C_{max} = 0.510$		
		PHBA	$C_{max} = 10.3$		
		DHBA	$C_{max} = 7.96$		
	Persian Gulf, Asia	PHBA	68.2–104.8		[64]
	Taichung River, Taiwan	3,5-diCl-MP	$C_{max} = 1.800$		[12]
		3-Cl-MP	$C_{max} = 0.9000$		
		3,5-diCl-PP	$C_{max} = 11.70$		
	Kota Kinabalu, Malaysia	3,5-diCl-MP	$C_{max} = 13400$		
		3-Cl-MP	$C_{max} = 267.00$		
		3,5-diCl-PP	$C_{max} = 1430.0$		
		3-Cl-PP	$C_{max} = 2480.0$		
		3-Cl-BP	$C_{max} = 1380.0$		
		3,5-diCl-BP	$C_{max} = 67.200$		
	Beijing, China	PHBA	$C_{ave} = 239.0$ $C_{max} = 1625$		[68]
		3-Cl-MP	$C_{max} = 5.050$		
		3,5-diCl-MP	$C_{ave} = 13.00$ $C_{max} = 80.00$		
		3-Cl-EP	$C_{ave} = 0.2000$ $C_{max} = 18.40$		
		3,5-diCl-EP	$C_{ave} = 31.00$ $C_{max} = 128.0$		
	Mudun River, China	∑ PHBA, DHBA, 3-OH-MP, 3-OH-EP, and BA	20200–29700		[66]
	Road runoff, China		$C_{max} = 4710.0$		
	Kitakami River, Japan	3-Br-MP	$C_{max} = 19.0$		[27]
		3-Br-EP	$C_{max} = 33.0$		
		3,5-diBr-MP	$C_{max} = 52.0$		
		3,5-diBr-EP	$C_{max} = 11.0$		
		3,5-diBr-PP	$C_{max} = 50.0$		
		3,5-diBr-BP	$C_{max} = 110$		
		3,5-diBr-BzP	$C_{max} = 49.0$		
	Shizuoka River, Japan	3,5-diCl-MP	$C_{max} = 6.10$		[18]
		3,5-diCl-PP	14.0–28.0		
Tap water	Texas, US	PHBA	$C_{ave} = 0.399$		[21]
		3,5-diCl-MP	$C_{ave} = 0.171$		
		3,5-diCl-EP	$C_{ave} = 0.0676$		

3,5-diBr-BP – di-brominated butylparaben; 3,5-diBr-BzP – di-brominated benzylparaben; 3,5-diBr-EP – di-brominated ethylparaben; 3,5-diBr-MP – di-brominated methylparaben; 3,5-diBr-PP – di-brominated propylparaben; 3,5-diCl-BP – di-chlorinated butylparaben; 3,5-diCl-EP – di-chlorinated ethylparaben; 3,5-diCl-MP – di-chlorinated methylparaben; 3,5-diCl-PP – di-chlorinated propylparaben; 3-Br-EP – mono-brominated ethylparaben; 3-Br-MP – mono-brominated methylparaben; 3-Cl-BP – mono-chlorinated butylparaben; 3-Cl-EP – mono-chlorinated ethylparaben; 3-Cl-MP – mono-chlorinated methylparaben; 3-Cl-PP – mono-chlorinated propylparaben; 3-OH-EP – mono-hydroxylated ethylparaben; 3-OH-MP – mono-hydroxylated methylparaben; BA – benzoic acid; C_{ave} – average concentration; C_{max} – maximum concentration; DHBA – di-hydroxybenzoic acid; PHBA – *para*-hydroxybenzoic acid; PTP – parabens transformation products.

Note: The data selection was obtained through an advanced search in PubMed and SCOPUS databases, by searching articles using specific keywords "Parabens transformation products AND Occurrence AND Surface Water". The output results were 6 and 3 papers in SCOPUS and PubMed, respectively. With the specific keywords "Parabens transformation products AND Occurrence AND Drinking Water", the output results were 1 and 0 papers in SCOPUS and PubMed, respectively. Duplicated papers were not considered for analysis and for real water sources only 7 and 1 papers were considered for PTP occurrence in surface water and DW, respectively.

most detected in surface waters worldwide, with maximum values of 1625 ng/L in the Beijing River [68], followed by 104.8 ng/L in the seawater of the Persian Gulf [64].

The presence of chlorinated parabens in rivers was first detected in 2012 in the Shizuoka River in Japan [18]. They found that 3,5-diCl-MP was present at a maximum concentration of 6.1 ng/L, while 3,5-diCl-PP reached 28 ng/L [18]. In general, 3,5-diCl-MP was observed at higher concentrations than 3-Cl-MP (Table 2), indicating its greater stability and resistance to environmental degradation [10]. This trend was also observed in the case of chlorinated-EP, with 3-Cl-EP and 3,5-diCl-EP reaching maximum concentrations of 18.4 and 128 ng/L, respectively [68]. Interestingly, Chen et al. [12] reported lower concentrations of 3,5-diCl-PP and di-chlorinated BP (3,5-diCl-BP) than their respective mono-chlorinated parabens in surface water in Kota Kinabalu, Malaysia (Table 2). In Chinese surface waters, mono and di-chlorinated parabens were commonly detected, with 3,5-diCl-EP reaching maximum concentrations of 128 ng/L in Beijing [68], and total transformation products reaching 100 times higher concentrations (29700 ng/L) in the Mudun river [66].

Among the studies evaluating the presence of PTP in surface waters, Gouukon et al. [27] was the only one reporting the presence of brominated parabens in the Kitakami River in Japan. These PTP includes mono-brominated MP (3-Br-MP), 3-Br-EP, di-brominated MP (3,5-diBr-MP), 3,5-diBr-EP, di-brominated PP (3,5-diBr-PP), di-brominated BP (3,5-diBr-BP), and di-brominated BzP (3,5-diBr-BzP) [27]. The PTP were detected in more than 25 % of the samples analyzed, with concentrations ranging from 11 to 110 ng/L for 3,5-diBr-EP and 3,5-diBr-BP, respectively (Table 2) [27]. Besides brominated parabens are generally detected at higher concentrations than their parent parabens in surface waters [27], chlorinated parabens and other PTP are commonly found at lower concentrations than their respective parent parabens in the rivers [10].

It is important to consider that the photodegradation of parabens in real environmental surface waters is greatly influenced by environmental characteristics such as the presence of organic matter, the presence of salts and ions, and turbidity [57]. Exemplifying, in deionized water the MP degradation in the presence of Fe (III)-citrate complexes resulted in the formation of diverse PTP such as hydroquinone, nonanedioic acid, succinic acid, glutaric acid, adipic acid, malonic acid, and heptanedioic acid. However, in real environmental surface waters (Liangzi and Donghu lakes, Changjiang river, and seawater), the MP photodegradation was significantly hindered.

The photodegradation of MP was inhibited in the seawater and river water, followed by the lake waters due to the coexisting ions (Mg^{2+} and $Ca^{2+)}$ and dissolved organic matter that scavenge $OH^{\bullet}$ radicals [57]. Moreover, the halide ions (Cl^{-}, Br^{-}, and I^{-}) were found to scavenge $OH^{\bullet}$ radicals inhibiting the degradation of MP [57].

Besides that, the intensity of sunlight and the respective light adsorption of natural waters also influence the photodegradation and photolysis of parabens [57]. Clearer water may lead to higher light absorption, facilitating parabens photodegradation and, consequently the formation of hydroxylated parabens.

3.3. Occurrence of PTP in DW

As discussed in section 2.1, the persistence of PTP from water chlorination appears to be higher when compared to other forms of parabens. However, limited data are available concerning the presence and fate of these compounds in DW, including tap water. There is only one recent study in Texas reporting the occurrence of chlorinated parabens in tap water (Table 2). In that study, 3,5-diCl-MP and 3,5-diCl-EP were quantified at 0.17 and 0.07 ng/L, respectively. Additionally, PHBA was detected at 0.4 ng/L [21]. Previous studies assessed the occurrence of PTP in tap water under controlled laboratory conditions. Canosa et al. [31] were the first to demonstrate that even low concentrations of free chlorine (0.5 mg/L) in tap water can transform parabens into chlorinated by-products. Similarly, Terasaki et al. [18] reported the formation of chlorinated derivatives from parabens in tap water, with a notable increase in the concentration of 3-Cl-PP during the first 2.5 min of the experiment, followed by a rapid decrease, resulting in the presence of 3,5-diCl-PP due to further chlorination. Furthermore, Alvarez-Rivera et al [52] reported the presence of mono and di-brominated parabens in tap water samples. The presence of these products in tap water supports the hypothesis that chlorinated parabens may arise in DWTPs and along chlorinated drinking water distribution systems (DWDS) when trace concentrations of parabens react with residual concentrations of chlorine, leading in some cases, to the presence of chlorinated PTP in treated water. The scarcity of studies investigating the presence of PTP in DW underscores the underexplored nature of this research topic. However, considering that these transformation products can potentially pose risks to human health and significantly impact environmental ecosystems to a greater extent than the original parabens, it is relevant to prioritize and further investigate their occurrence, routes, and (eco) toxicological effects.

3.4. *Occurrence of PTP in swimming pools*

Current practices for the disinfection of swimming pool water are crucial to prevent the spread of infectious microorganisms. This emphasis on microbial safety is essential, due to the high demand for swimming activities, whether for leisure or sports. Unlike DW, which relies primarily on natural organic matter as the main organic precursor of DBP, swimming pool water introduces human-sourced organic compounds. These can include body fluids and parabens-containing PCP, such as sunscreen. Their presence is known to favours the formation of PTP in swimming pool waters (Table 3) [69].

The presence of chlorinated parabens and their parent parabens in swimming pools was first reported by Terasaki and Makino [70] in Japan. Notably, di-chlorinated-iPP (3,5-diCl-iPP) was found at a maximum concentration of 25 ng/L, while 3,5-diCl-MP and mono-chlorinated-BzP (3-Cl-BzP) at concentrations below the limit of quantification [70]. In China, chlorinated parabens were detected in swimming pools, with di-chlorinated parabens (3,5-diCl-MP and 3,5-diCl-EP) showing higher concentrations than mono-chlorinated variants (3-Cl-MP and 3-Cl-EP) [19]. Specifically, 3,5-diCl-MP and 3,5-diCl-EP were measured at 36.2 and 24.2 ng/L, while 3-Cl-MP and 3-Cl-EP were found at 7.49 and 0.64 ng/L, respectively [19].

Table 3
Concentrations (ng/L) of parabens transformation products (PTP) found in swimming pools.

Location	PTP	Concentration	Reference
Beijing, China	3-Cl-MP	$C_{ave} = 1.490$ $C_{max} = 7.490$	[19]
	3,5-diCl-MP	$C_{ave} = 4.870$ $C_{max} = 36.20$	
	3-Cl-EP	$C_{ave} = 0.02000$ $C_{max} = 0.6400$	
	3,5-diCl-EP	$C_{ave} = 11.10$ $C_{max} = 24.20$	
	PHBA	$C_{ave} = 261.0$ $C_{max} = 1122$	
Japan	3,5-diCl-iPP	<LOD – 25.00	[70]

C_{ave} – average concentration; PHBA – *para*-hydroxybenzoic acid; 3-Cl-MP – mono-chlorinated methylparaben; 3-Cl-EP – mono-chlorinated ethylparaben; C_{max} – maximum concentration; <LOD – below the limit of detection; 3,5-diCl-EP – di-chlorinated ethylparaben; 3,5-diCl-MP – di-chlorinated methylparaben; 3,5-diCl-iPP – di-chlorinated *iso*-propylparaben; PTP – parabens transformation products.
Note: The data selection was obtained through an advanced search in PubMed and SCOPUS databases, by searching articles using specific keywords "Parabens transformation products AND Occurrence AND Swimming pools". The output results were 2 and 0 papers in SCOPUS and PubMed, respectively. Duplicated papers were not considered for analysis and for real water sources only 2 papers were considered for PTP occurrence in swimming pools water.

Unfortunately, many countries lack specific regulatory limits for DBP in swimming pool waters, especially for PTP. Regulatory agencies have traditionally focused on regulating DBP such as THM, haloacetic acids (HAA), and haloacetonitrile (HAN) in swimming pools [69]. Among them, only THM has been regulated in European countries, with Germany setting a maximum limit of 20 µg/L and Belgium, France, the United Kingdom, and Finland specifying a maximum limit of 100 µg/L, in swimming pool waters [71].

4. Techniques to mitigate PTP formation during water treatment

Although several techniques have been adopted for the removal of parabens present in wastewater, the complete removal of these pollutants from water sources is still challenging, making it even more difficult to optimize the removal of their metabolites and respective PTP. Physical removal technologies including adsorption with activated carbon and membrane filtration are not efficient, and AOPs emerged as the most efficient leading to the degradation of parabens, even resulting in the formation of PTP as described in section 2.4 [72]. However, these PTP are thought to be less harmful to human health and the environment [59,73].

There are few studies evaluating the removal of PTP along water treatment plants. Indeed, the research community is more interested in removing parent parabens along water treatment. Some studies reported comparisons of removal efficiencies between parabens and their PTP (as presented in section 3.1) along conventional water treatment and suggested low removal efficiency regarding PTP [24]. However, with AOPs, this % of PTP removal can increase as described by Li et al. [16] who reported that chlorinated parabens were removed from 33.9 to 40.7 % and 59.2 to 82.8 % through conventional and advanced treatment processes (ultrafiltration and ozonation), respectively [16]. In the same study, ultrafiltration membranes were able to remove 9.9 and 3.1 % of 3,5-diCl-MP and 3,5-diCl-EP, respectively [16]. Due to the small pore size of ultrafiltration membranes (<100 nm), they can effectively exclude PTP, which are expected to be larger than those of parent parabens. Following this, PTP generated during ozonation of parabens were partially removed (PHBA – 92.4 %; 3,5-diCl-MP – 82.8 %; and 3,5-diCl-EP – 59.2 %) [16]. This may be explained by the fact that the by-products generated during the ozonation are more resistant to further ozone oxidation or cleavage of the aromatic ring than the parent compounds [16]. Di-chlorinated PTP also revealed strong sorption onto sludge particles, which decreased the efficiency of the process for O_3 oxidation [16]. This reinforces that the presence of organic matter reduced the degradation of parabens and PTP.

More recently, Huo et al. [74] studied the effect of pH on UV/H_2O_2 treatment halogenated parabens removal from water. Briefly explaining this method, when H_2O_2 is irradiated by UV it absorbs energy to break the O–O bond and produce $OH^\bullet$. At the same time, H_2O_2 is also formed again, reacting with $OH^\bullet$ to form $HO_2^\bullet$. Subsequently, $HO_2^\bullet$ reacts with H_2O_2, $OH^\bullet$, and $HO_2^\bullet$ to form H_2O and O_2 [74]. They verified that the removal of mixed species was more difficult than single species [74]. Moreover, they found that di-halogenated parabens are more easily removed than mono-halogenated parabens and consequently parent parabens at neutral and weakly alkaline pH using this technique [74].

The combination of ozonation and UV/H_2O_2 treatments was also proven to be effective in removing parabens and reducing the formation of oxidized transformation products in DW by other studies [15,75,76]. Indeed, the presence of H_2O_2 reduced the time required for the total degradation of parabens [15]. However, the degradation reaction of the mixture of parabens (commonly presented in natural water sources) was found to be slower in comparison to the degradation of individual parabens [15].

Lee et al. [75] reported the complete elimination of MP and its chlorinated and brominated PTP (3-Cl-MP, 3,5-diCl-MP, 3-Br-MP, 3,5-diBr-MP, and 3-Br-5-Cl-MP) from a wastewater effluent by applying ozonation (with a specific dose of > 0.26 $gO_3/gDOC$) and UV_{254}/H_2O_2. However, UV_{254}/chlorine treatment is a more cost-effective and feasible option compared to UV_{254}/H_2O_2, discouraging the use of the second option [76].

The photodegradation of PTP through ultraviolet absorption using simulated sunlight was also considered a great strategy to degrade PTP in water [77] resulting in photoproducts with negligible estrogenic activity. Chlorinated (3-Cl-MP, 3,5-diCl-MP, 3-Cl-EP, 3,5-diCl-EP) and brominated (3-Br-MP, 3,5-diBr-MP, and 3-Br-EP) parabens were exposed to this treatment and the results suggested that brominated parabens exhibited higher degradation efficiency than chlorinated parabens, and mono-halogenated parabens had higher degradation than these di-halogenated [77]. Recently, Antonopoulou [73] found that the combination of UV-C with the strong oxidant persulfate (PS), namely the UV-C/PS process was effective for the simultaneous degradation of MP, PP, and PHBA in different aqueous matrices (wastewater, surface water, and drinking water). More interesting, was the fact that the PTP formed during this process did not show significant ecotoxicity in relation to microalgae species (*Scenedesmus rubescens* and *Dunaliella tertiolecta*). The same author also reported the UV-C/peroxymonosulfate process (UV-C/PMS) as an efficient strategy to degrade parabens mixtures reducing the toxicity from the wastewater [59]. These results, highlight UV-C/PS and UV-C/PMS as good strategies to eliminate parabens from wastewater and avoid the formation of harmful PTP [59,73].

Curiously, periphyton, a microbial aggregate mainly composed of microalgae and bacteria, has shown efficient removal (80 %) of chlorinated parabens (3-Cl-MP and 3,5-diCl-MP) and from 20 to 100 % for original parabens [38]. Previous studies have already reported the removal of parent parabens using microalgae achieving removal efficiencies greater than 30 % [78]. These biological and natural treatments may hold promise in mitigating the presence of these compounds in water [38].

5. Toxicological implications of PTP

Parabens have been associated with health concerns, such as endocrine disruption, breast tumours, and DNA damage, increasing awareness about the use and consumption of products containing these compounds [51]. Lincho et al. [79] have compiled studies reporting the endocrine-disrupting effects of parabens, mimicking estrogen activity but also antiandrogenic and antagonist activity. Pereira et al. [8] also reviewed the impact of parabens on human health suggesting other health complications such as reproductive disorders, transgenerational effects, perturbation of the central nervous system, among others.

Like parabens that have been found in human tissues and fluids [80], PTP such as PHBA, DHBA, OH-MP, and OH-EP have also been detected in human blood and urine [22,65]. These PTP were detected at median concentrations of 167, 115, 11.3, and 0.24 ng/L, respectively, in urine samples of Chinese university students [22]. Median concentrations of 2.88, 1.67, 1.06, and 0.19 ng/L were also detected in blood samples for PHBA, DHBA, OH-MP, and OH-EP, respectively [22]. Moreover, PHBA was also detected at 29.9 ng/L in blood samples of adult Indian females [81]. The higher concentrations of PHBA and DHBA compared to other PTP are related to their metabolization not only from parabens but also from other precursors and natural sources [22].

In general, DBP are known to be hazardous compounds that have adverse effects on both animal and human health [1]. However, scientific data on the toxicity of PTP is scarce [82]. Therefore, considering the health concerns associated with parabens, their transformation products are also expected to threaten human and animal health [8]. The toxicity of parabens increases with their lipophilicity, and chain length [83]. Indeed, the halogenation of parabens appears to increase endocrine-disruptive activity [29]. Therefore, chlorinated and brominated parabens may be more toxic than their parent and hydroxylated counterparts. This makes sense since more hydrophobic compounds (halogenated parabens) tend to have higher bioaccumulation potential, whereas hydroxylated parabens are more hydrophilic and more easily metabolized [3]. Consequently, chlorination and bromination of water will result in the formation of more harmful PTP in comparison to the use of UV irradiation, ozonation, and other AOPs.

The increase in endocrine-disruptive activity from parabens and PTP exposure may be explained by the increased activation of thyroid receptors (TR) and aryl hydrocarbon receptors (AhR) reflected by the higher affinity to these receptors [82]. This may lead to hormonal dysregulation, changes in gene expression, and metabolic disturbances. Therefore, these compounds can have multiple impacts on hormonal and regulatory systems [82]. Studies revealing the endocrine disruptive activity of PTP are listed in Supplemental Information C (Table S2). Álvarez et al. [51] conducted a cytotoxicity study of MP, EP, and BP transformation products by evaluating their effects on a cultured human embryonic kidney cell line (HEK-293) and found no cytotoxicity. However, a recent study reported that halogenated parabens led to higher toxicity with the addition of a second chlorine both in relation to human and fish cell lines. Moreover, PHBA was shown to be more toxic to fish hepatocytes than human hepatocytes by 100-fold. These discrepancies in cytotoxicity suggest tissue-dependency of halogenated parabens [84]. A recent study by Weiss et al. [85] evaluating the impact of halogenated parabens and their respective parent parabens on AhR using *in vitro* studies reported neither the parent (PP and BzP) nor the synthesized chlorinated parabens (3-Cl-PP and 3-Cl-BzP) AhR activity. In the same study, only 3-Br-PP and 3-Br-BzP slightly increased the AhR activity by 1.5-fold and 1.4-fold, respectively, while 3,5-diBr-PP and 3,5-diBr-BzP were devoid of agonistic activity (Table S2) [85]. On the other hand, both chlorination and bromination enhanced the antagonistic activities of BzP, as evidenced by the decrease in half maximal inhibitory concentrations (IC_{50}) from 41.66 μM (BzP) to 17.49 μM (3-Cl-BzP) and 8.90 μM (3-Br-BzP) [85]. The same trend was reported for PP ($IC_{50} = 57$ μM) and their halogenated compounds (3-Cl-PP and 3-Br-PP) with IC_{50} of 21 and 16 μM, respectively [85].

Another study suggested that halogenated compounds cause AhR activation in relation to their parent parabens due to the presence of a halogen being mandatory for AhR activation [86]. Indeed, Gouukon et al. [27] have reported the highest activity on AhR (agonistic activity) for 3-Br-BP with an EC_{50} of 3.9 nM in relation to the parent paraben (BP). Most studies (Table S2) suggest that brominated parabens surpass the AhR binding affinities of their chlorinated counterparts, increasing their endocrine disruptive activity. This corroborates Jakopin [29] who used *in silico* predictions (Endocrine Disruptome software and Virtual-ToxLab™) and found mono-brominated PP (3-Br-PP) and mono-brominated BP (3-Br-BP) as the most potent endocrine disruptors with calculated affinities for AhR of 428 nM and 254 nM, respectively (Table S2) [29]. Interestingly, although mono-halogenated parabens exhibit higher AhR affinity compared to the parent parabens, this affinity decreases for di-halogenated (Table S2). This is explained since the hydrolysis of the ester group eliminates the AhR activities [82].

Regarding the impact of halogenated parabens on glucocorticoid receptors (GR), Jakopin [29] also identified di-brominated-BzP (3,5-diBr-BzP) as the most potent endocrine disruptor, with an IC_{50} of 331 nM for glucocorticoid receptor (GR) revealing stronger affinities of brominated parabens for GR than chlorinated ones (Table S2) [29]. Furthermore, di-halogenated products from iPP (3,5-diCl-iPP, IC_{50} 2.56 μM; 3,5-diBr-iPP, IC_{50} 6.29 μmM), iBP (3,5-diCl-iBP, IC_{50} 3.78 μM; 3,5-diBr-iBP, IC_{50} 5.15 μM) and BzP (3,5-diCl-BzP, IC_{50} 1.94 μM; 3,5-diBr-BzP, IC_{50} 331 nM) are predicted to have stronger affinities for the GR than their mono-halogenated counterparts (Table S2) [29].

On the other hand, mono-halogenated parabens appear to have a higher affinity towards TRα than their di-halogenated counterparts, with the opposite trend observed for TRβ [29]. Overall, mono-chlorinated BzP (3-Cl-BzP) was proposed as the most potent agonist on TRα and TRβ, with calculated IC_{50} values of 424 nM and 342 nM, respectively [29].

In yeast two-hybrid assays using *Saccharomyces cerevisiae* that incorporated human ERα, Terasaki et al. [87] found that chlorinated parabens exhibited less estrogen agonistic activity compared to parent parabens. Chlorination masked the apparent estrogen agonistic activity of most parabens, leading to an increase in their antagonistic activity [88]. This neutralization of estrogen agonistic activity of halogenated parabens can be attributed to the halogen atoms at the *ortho* position of phenolic rings, which inhibit the interaction of the phenolic OH group with the estrogen receptor [88].

The estrogenic activity of PTP was also assessed using a yeast two-hybrid assay (as described previously) by Wan et al. [77] who reported negligible estrogenic activities for the photoproducts of both chlorinated and brominated parabens. The estrogenic activities and EC_{50} were 253.0, 553.8, 58.7, and 204.6 mM for 3-Cl-MP, 3-Br-MP, 3-Cl-EP, and 3-Br-EP, respectively [77]. Comparing these values, the chlorinated derivatives exhibited greater estrogenic activity than the corresponding brominated derivatives [77].

Using a similar two-hybrid assay, it was further determined that brominated alkyl parabens (3,5-diBr-MP, 3-Br-EP, 3,5-diBr-EP, 3-Br-PP, 3,5-diBr-PP, di-brominated iPP – 3,5-diBr-iPP, 3-Br-BP, 3,5-diBr-BP, mono-brominated iBP – 3-Br-iBP, di-brominated iBP – 3,5-diBr-iBP), exhibited antagonistic activity towards human ERα (Table S2) [89]. The intensity of this antagonistic activity increases as the number of bromine substituents increases [89]. The antagonistic activity was evaluated based on REC_{60}, which represents the concentration of the compound needed to achieve 60 % of the maximum inhibitory activity of 4-hydroxytamoxifen (commonly used as a positive control for antagonist activity) [89]. In that study, REC_{60} values for brominated parabens ranged from 430 to 10000 nM, with 3,5-diBr-BP demonstrating the most significant antagonistic activity [89].

Niu et al. [90] also found through computational simulation that a hydroxylated PTP from EP through photolysis has higher potential health risks than the parent compound, affecting many body systems, including the blood, cardiovascular and gastrointestinal systems as well as kidney, and liver. PTP can create more hydrogen bonds with lower energy binding with human estrogen receptor α (ERα) concerning the parent EP, suggesting increased estrogenic activity [90]. Indeed, a recent study reported increased estrogenic activity for two BzP transformation products (BzP-*o*-phenol and BzP-m-phenol) resulting from photodegradation concerning the parent paraben (BzP) [91]. This was explained by their lower IC_{50} values of 0.26 and 0.50 μM, respectively, in comparison to BzP (6.42 μM) [91]. These PTP also showed lower free energies with the parent paraben demonstrating higher binding affinities toward Erα [91]. Gao et al. [92] used computational analysis through the Advanced Chemistry Development platform and observed negative impacts on the gastrointestinal system and liver from the exposure to 3-OH-MP.

The photodegradation of EP in water led to the formation of PHBA, 3-OH-EP, and an EP oligomer [93]. Although PHBA was an inactive endocrine disruptor, the EP oligomer revealed increased estrogenic activity and decreased EC_{50} values in comparison to the parent EP [93]. Conversely, 3-OH-EP showed reduced estrogenic activity ($EC_{50} = 2.32 \times 10^{-4}$ M) relative to the parent EP ($EC_{50} = 1.35 \times 10^{-5}$ M) [93]. Additionally, some PTP resulting from parabens hydrolysis, such as PHBA, phenols, and hydroquinone, were found to reduce anti-androgenic activity when tested using rat liver enzymes [94].

Although there is still a large knowledge gap about the toxicity and health effects of PTP, the negative impact from the ingestion of polluted water with these compounds should not be disregarded.

6. Ecotoxicology of PTP

PTP are usually less biodegradable than their parent compounds [10], which leads to their higher environmental stability, and thus increased bioaccumulation potential [78]. By definition, the bioaccumulation factor (BAF) of an environmental contaminant is defined by the ratio between the chemical concentration in a given organism (ng/kg) and the total chemical concentration in the water phase (ng/L) [95]. Among the different PTP, OH-MP and OH-EP have an effective bioaccumulation effect in marine organisms (log BAF>3.7), whereas PHBA is considered to have bioaccumulated potential (3.3 < log BAF<3.7) [95]. Conversely, the parent parabens (*i.e.*, MP, EP, PP, BP, and BzP) showed no potential for bioaccumulation (log BAF<3.3). This makes sense since more polar compounds may have a higher ability to interact with biological tissues and aqueous solutions. Moreover, the bioaccumulated potential of PHBA may also be influenced by the fact that original parabens could be degraded by organisms leading to PHBA as a product [3]. However, another study focused on subtropical marine food web suggested higher bioaccumulation of MP in comparison to PHBA, based on the metabolization of PHBA along the food web [96].

The power of PTP bioaccumulation may cause potential risks to exposed organisms [28].

The hazard quotient (HQ) is commonly assessed to understand the ecological risks posed by these environmental pollutants [16]. HQ is defined as the ratio between the measured environmental concentrations (MEC) of these compounds on organisms and the predicted no-effect concentrations (PNEC) [16]. A HQ value of less than 1 indicates low risk to the environment. However, other parameters can also be used, particularly the acute toxicity commonly expressed by effective concentration (EC_{50}) or lethal concentration (LC_{50}) [97]. Moreover, chronic toxicity values are described by no-observed-effect concentration (NOEC) and lowest-observed-effect concentration (LOEC) [97]. ChV-defined chronic toxicity values will also be used in this section to characterize the ecotoxicity of PTP [28]. Studies evaluating the ecotoxicity of PTP towards aquatic organisms are presented in Table 4.

Overall, halogenated parabens are more toxic to aquatic microorganisms than their parent parabens. For example, the expected PNEC for MP against aquatic invertebrates was reported to be 18700 ng/L, whereas for 3,5-diCl-MP was 16000 ng/L [16]. Terasaki et al. [87] also demonstrated that the toxicity values of 3,5-diCl-MP and 3,5-diCl-EP for *Daphnia magna* increased 3.9- and 2.8-fold, respectively, in comparison to their parent compounds, accounting for lower EC_{50} values of PTP (Table 4). Contradictory results were observed against *Ceriodaphnia dubia* by Terasaki et al. [98]. In their research, the mortality rate and reproduction inhibition of *C. dubia* increased with increasing hydrophobicity of parabens and decreased with the degree of parabens chlorination [98]. Acute toxicity values for BzP and 3,5-diCl-BzP were 0.30 and 3.1 mg/L, respectively [98].

Hydroxylated parabens have shown varied levels of toxicity to different organisms [28]. The Environmental Protection Agency (EPA) defines compounds as very toxic, toxic, harmful, and non-harmful for LC_{50} or EC_{50} values < 1; between 1 and 10; 10 and 100; and > 100 mg/L, respectively [99]. For ChV values < 0.1; between 0.1 and 1; 1 and 10; and > 10 mg/L, compounds were also defined as very toxic, toxic, harmful, and non-harmful, respectively [99]. Considering both ChV and EC_{50} values, 3,5-diOH-MP was considered non-harmful for green algae (ChV and EC_{50} > 100.0 mg/L), toxic to *Daphnia* (LC_{50} = 1.17 mg/L, ChV=0.23 mg/L) and harmful to fish (LC_{50} = 10.91 mg/L, ChV=2.69 mg/L) [28]. However, 3-OH-MP was found to be toxic to green algae (EC_{50} = 8.51 mg/L, ChV=0.96 mg/L); harmful to fish (LC_{50} = 42.56 mg/L, ChV=9.83 mg/L) and non-harmful for *Daphnia* (ChV and EC_{50} > 100.0 mg/L) [28]. Different results were obtained by Gao et al. [33], who observed increasing toxicity for hydroxylated parabens to green algae but less toxicity to *Daphnia* and fish when using ecological structure – activity relationships (ECOSARs) described by EPA [100]. Specifically, 3-OH-EP showed 1.4 and 3 times higher acute and chronic

Table 4
Ecotoxicity of parabens transformation products (PTP) towards aquatic organisms.

PTP	Ecotoxicity	Reference
3,5-diCl-MP	Acute toxicity:	[87]
	– *V. fisheri*: $EC_{50} = 3$ ng/L	
	– *D. magna*: $EC_{50} = 16$ ng/L	
	– Bacteria: HQ=1.9–6.4 × 10^{-3}	[16]
	– Invertebrate: HQ=1.2–3.5 × 10^{-3}	
	Chronic toxicity:	[98]
	– *C. dubia*: $EC_{50} = 10$ ng/L	
3,5-diCl-EP	Acute toxicity:	[87]
	– *V. fisheri*: $EC_{50} = 3.6$ ng/L	
	– *D. magna:* $EC_{50} = 13$ ng/L	
	– Bacteria: HQ=1.9–6.4 × 10^{-3}	[16]
	– Invertebrate: HQ=1.2–3.5 × 10^{-3}	
3,5-diCl-BzP	– *C. dubia*: NOEC<0.63 mg/L; $EC_{50} = 3.1$ mg/L	[98]
3,5-diOH-MP	– Green algae: $EC_{50} = 2069.34$ mg/L, ChV=140.74 mg/L	[28]
	– *Daphnia*: $LC_{50} = 1.17$ mg/L, ChV=0.23 mg/L	
	– Fish: $LC_{50} = 10.91$ mg/L, ChV=2.69 mg/L	
3-OH-MP	– Green algae: $EC_{50} = 8.51$ mg/L, ChV=0.96 mg/L	
	– *Daphnia*: $LC_{50} = 241.80$ mg/L, ChV=209.48 mg/L	
	– Fish: $LC_{50} = 42.56$ mg/L, ChV=9.83 mg/L	
3-OH-EP	– Green algae: $EC_{50} = 6.29$ mg/L, ChV=0.77 mg/L	[33]
	– *Daphnia:* $LC_{50} = 128.41$ mg/L, ChV=92.41 mg/L	
	– Fish: $LC_{50} = 23.51$ mg/L, ChV=4.96 mg/L	
3-OH-PP	– Green algae: $EC_{50} = 4.62$ mg/L, ChV=0.61 mg/L	
	– *Daphnia:* $LC_{50} = 67.79$ mg/L, ChV=40.52 mg/L	
	– Fish: $LC_{50} = 12.91$ mg/L, ChV=2.49 mg/L	

3,5-diCl-BzP – di-chlorinated benzylparaben; 3,5-diCl-EP – di-chlorinated ethylparaben; 3,5-diCl-MP – di-chlorinated methylparaben; 3,5-diOH-MP – di-hydroxylated methylparaben; 3-OH-EP – mono-hydroxylated ethylparaben; 3-OH-MP – mono-hydroxylated methylparaben; 3-OH-PP – mono-hydroxylated propylparaben; *C. dubia* – *Ceriodaphnia dubia*; ChV – chronic toxicity value; *D. magna* – *Daphnia magna*; EC_{50} – effective concentration to inhibit 50% of the population; HQ – risk quotient; LC_{50} – Lethal concentration to kill 50% of the population; NOEC – no-observed-effect concentration; PTP – parabens transformation products; *V. fisheri* – *Vibrio fisheri.*

toxicity than EP, respectively [33]. Similar results were obtained by Qiu et al. [44] through the assessment of the ecotoxicity of parabens and their transformation products from ozonation using ECOSAR. Curiously, in that study, after multiple O_3-addition reactions, the acute toxicity of PTP decreased compared to the original compounds (MP and EP) [44]. Besides that, it is important to note that wastewater ozonation can lead to the formation of persistent oxidation products, which have been shown to significantly affect the root and shoot growth of plant species [101]. Li et al. [102] studied the oxidation of MP and PHBA by manganese dioxide (MnO_2) and iodine (I_2) and found that both aromatic transformation products were more toxic than the parent MP and PHBA. This increase in toxicity may be attributed to the reduction in the hydrophilicity of polymeric and iodinated products concerning MP [102].

7. Research needs and opportunities

In the context of mitigating the environmental formation of PTP and monitoring their presence along water management plants, it is imperative to align these strategies with the overarching principles of the One Water approach [30]. Recognizing that all water has value, regardless of its source, emphasizes the interconnected nature of water systems and the need for comprehensive management practices [30]. It is important to note that water management is cyclic: water is used, enters WWTP, returns to the environment (surface water), undergoes treatment again for DW, and then returns to the sewage system to re-enter WWTP.

Due to the existence of few techniques able to remove PTP, particularly AOPs, allied to the higher difficulty in removing these compounds in comparison to their original parabens [24], it is crucial to sensitize the community to the need to reduce the use of products containing recalcitrant contaminants. This will reduce their load in sewage and WWTP, emphasizing the importance of source control and minimizing the introduction of these contaminants into the water cycle.

Moreover, the prioritization of water disinfection treatments should be carefully planned to minimize the formation of harmful PTP. Therefore, ozonation and UV-based disinfection combined with other AOPs should be integrated into water treatment plants, avoiding halogen-based disinfection such as chlorination and bromination [16]. The use of AOPs must be prioritized since the PTP generated are less harmful than those resulting from halogenated-based disinfection strategies [54].

Another issue that should be addressed is the optimization of techniques to monitor and identify the presence of these PTP in real time along water treatment plants. Nowadays, the analytical methods for identifying and quantifying PTP are the same as those used for original parabens and may not be specific to the different types of PTP [25,38]. This will allow a realistic perception of the most PTP targets to be eliminated and further provide the study of specific toxicity, eliminating the research gap about toxicities and health effects that may prevent potential public health issues. Mitigation strategies to avoid PTP formation in water should consider the entire water cycle, developing integrated solutions that prioritize sustainability, efficiency, and protection of water resources for current and future generations. This involves a careful balance between pathogen control and PTP prevention.

8. Conclusions

The presence of PTP along WWTP is unavoidable, being the most often detected PHBA and hydroxylated, chlorinated, and brominated parabens. Furthermore, PTP have been found in surface waters, groundwater, and treated water from DWTP. Studies reporting the presence of these products in real water treatment plants are very scarce and were mostly performed in the US and China. Moreover, the presence of these pollutants of concern has been emphasized in swimming pools, where the simultaneous presence of parabens and chlorine is constant, leading to their interaction and generation of PTP. In general, PTP are found in water systems at higher concentrations than their respective parent parabens, increasing their persistence in the environment. In addition, PTP appear to have critical ecotoxicological and toxicological effects, revealing potential risks for aquatic organisms (*i.e.* bacteria, algae, fish, plants, etc.). Overall, addressing the challenges posed by PTP requires ongoing research considering the One Water approach, promoting community awareness, and integrating sustainable and efficient mitigation strategies to avoid their presence.

CRediT authorship contribution statement

Ana Rita Pereira: Conceptualization, Methodology, Investigation, Data curation, Writing – original draft. **Inês B. Gomes:** Conceptualization, Writing – review & editing, Supervision. **Mourad Harir:** Writing – review & editing, Validation, Methodology, Formal analysis, Conceptualization. **Lúcia Santos:** Writing – review & editing, Visualization, Validation, Supervision, Methodology, Formal analysis, Data curation, Conceptualization. **Manuel Simões:** Writing – review & editing, Visualization, Validation, Supervision, Software, Resources, Project administration, Methodology, Funding acquisition, Formal analysis, Data curation, Conceptualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.cej.2024.155129.

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