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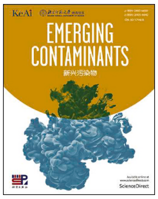

Exploring the influence of redox and temperature variations on the attenuation of pharmaceuticals in groundwater: Insights from parallel batch reactor experiments

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ABSTRACT

Pharmaceuticals are present in the environment from anthropogenic pollution sources, leading to groundwater contamination when reaching urban aquifers. Once in the subsurface, their fate depends on the hydrochemical processes and environmental conditions within the aquifer. In this context, this study investigates how different redox and temperature conditions affect the natural attenuation of pharmaceuticals in the subsurface. Batch experiments were conducted under oxic and suboxic (i.e., up to nitrate reduction) conditions and at two temperatures (25 °C and 35 °C). The controlled conditions achieved with parallel batch reactor systems allowed us a systematic investigation of the processes and factors involved in the fate of ten pharmaceuticals (atenolol, citalopram, climbazole, irbesartan, lamotrigine, sitagliptin, carbamazepine, metoprolol, trimethoprim, and venlafaxine), providing insights into the mechanisms governing their attenuation. The results showed that oxic conditions were highly effective in reducing pharmaceuticals concentrations, achieving up to 91 % attenuation for irbesartan, followed by citalopram (90 %), climbazole (77 %), sitagliptin (76 %) and metoprolol (75 %). Atenolol and climbazole were also attenuated regardless of redox conditions. High temperatures increased the total removal of citalopram, irbesartan, sitagliptin, and trimethoprim by 5–12 %, while slightly enhancing the sorption affinity of carbamazepine, irbesartan, and atenolol by 5 %. However, trimethoprim, carbamazepine, and lamotrigine were the most persistent compounds, with average removal rates of 6 %, 15 %, and 24 %, respectively. Overall, more than half of the targeted pharmaceuticals showed significant average removal (>60 %), highlighting the influence of the processes involved in groundwater on the natural attenuation of these compounds. Sorption seemed to be the primary process contributing to the target pharmaceuticals attenuation in oxic conditions, while biodegradation played a secondary role, particularly for atenolol and metoprolol. These findings contribute to improve our understanding of the behaviour of pharmaceuticals in aquatic environment and thus to improve management practices for better water quality.

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1. Introduction

The global population is expected to increase by nearly 2 billion people over the next 30 years, from the current 8 billion to 9.7 billion in 2050, and could peak at nearly 10.4 billion in the mid-2080s, resulting in increased water stress [1]. This population escalation raises concerns not only about freshwater availability but also about the exacerbation of global freshwater quality issues.

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In addition, it is expected that water scarcity will increase as a result of the climate change. In this context, urban groundwater, which accounts for more than a third of global water use, is considered an alternative source of drinking water to mitigate the impact of climate change [2–4] and the increasing demand on the freshwater availability. However, urban aquifers are exposed to the effects of urbanization, which might alter the physicochemical conditions of aquifers due to the contribution of significantly increasing temperatures of both the aquatic environment and the substrate [5]. Since temperature can influence shifts in redox zonation, these two factors are often considered to be interdependent [6,7]. Therefore, changes in temperature and redox conditions may ultimately affect water quality by influencing the microbial activity and, in turn, the fate and distribution of organic contaminants in aquatic ecosystems [8]. On top of this, shallow urban aquifers are vulnerable to various sources of pollution, such as land use, sewerage and septic system leakage, river seepage, hospital effluent and discharges from wastewater treatment plants that are not specifically designed to remove trace organic compounds [2,9]. As a result, a wide range of trace organic compounds including pharmaceuticals, may reach the aquatic environment, posing a significant threat to freshwater resources, the ecosystems that depend on them [1], and thereby reducing their quality for drinking water supply [3,10,11]. For this reason, there is growing concern about the presence of a wide range of pharmaceuticals and their transformation products (TPs) in groundwater across Europe [12–14].

Once pharmaceuticals enter groundwater, their behaviour becomes highly uncertain, as it depends on coupled hydrothermochemical processes occurring simultaneously in the subsurface [15]. While microbial degradation is believed to be the main mechanism for pharmaceutical attenuation in aquifers [16], the understanding of the process responsible for their natural removal (i.e., biodegradation, hydrolysis and sorption) [17] still is limited. Environmental conditions, such as organic matter composition, temperature, and redox conditions, significantly affect the removal of these substances [18]. Additionally, temperature can influence soil sorption characteristics by expanding the oxic zone [19], thereby impacting the removal of pharmaceuticals through changes in redox zonation [20]. Therefore, understanding the individual contributions of redox and temperature is crucial for determining the behaviour of pharmaceuticals in the subsurface [17].

Laboratory-scale investigations conducted to study the removal of pharmaceuticals have identified sorption and biodegradation as the main mechanisms involved in their attenuation [21,22]. However, these previous studies have limitations, including the narrow range of physicochemical properties of the selected compounds, making difficult the identification of trends and deeper understanding of pharmaceutical removal behaviour. In addition, several factors limit the extrapolation of the behaviour of these compounds to real-world aquifers with general characteristics, such as: the absence of different redox and temperature conditions, the high-dose of concentrations spiked in the reactors that commonly exceed environmental thresholds, the focus on specific compounds rather than a wide range of compounds (i.e., cocktail of compounds), as typically found in aquatic ecosystems, the lack of microbial communities adapted to the batch reactors, and the use of liquid and solid phases only from a specific aquifer.

The aim of this study is to evaluate the processes involved in the natural attenuation of ten selected pharmaceuticals (atenolol, citalopram, climbazole, irbesartan, lamotrigine, sitagliptin, carbamazepine, metoprolol, trimethoprim, and venlafaxine) using parallel batch reactor systems under different environmental conditions: redox (oxic and suboxic, up to nitrate reduction conditions) and temperature (at 25 and 35 °C). Additionally, this research seeks to identify the favoured natural removal mechanisms for these substances. We hypothesize that the attenuation of these substances will be higher under oxic conditions and at higher temperatures, while being reduced in abiotic controls due to the inhibition of microbial activity. Our study addresses a knowledge gap by comparing oxic and nitrate reducing conditions, and demonstrates the potential for some substances to undergo natural attenuation under nitrate reducing conditions.

2. Materials and methods

2.1. Selection of the target compounds

Urban groundwater is continuously polluted by a variety of pharmaceuticals, making it crucial to identify the most common compounds to understand their persistence in the environment. For this study, pharmaceuticals were selected based on two criteria: (i) their widespread use and (ii) their frequent detection in river-aquifer systems. To narrow down the list, this study focused on pharmaceuticals frequently detected in an urban aquifer located in the northeast of Barcelona, Spain [23,24]. The selected compounds were: metoprolol, atenolol, venlafaxine, carbamazepine, sitagliptin, citalopram, climbazole lamotrigine, trimethoprim, and irbesartan. These pharmaceuticals reached maximum concentration up to 990 ng/L for lamotrigine and some of them were close or exceeded the threshold of 100 ng/L (i.e., trimethoprim carbamazepine, metoprolol, and irbesartan). Some pharmaceuticals present low average concentrations in the aquifer but sometimes higher than its main recharge source, which is a polluted river receiving effluents from WWTPs (e.g. 0.6 vs. 240 ng/L for atenolol, 49.7 vs. 277.2 ng/L for venlafaxine; 0.05 vs. 406.2 ng/L for sitagliptin) [24]. Thus, understanding the natural attenuation of these pharmaceuticals is crucial to determine the potential uses of urban groundwater, such as the urban aquifers of Barcelona. Their physicochemical properties which may describe their environmental fate are listed in Table S1.

2.2. Experimental setup

A laboratory study was conducted to assess the behaviour of pharmaceuticals under different redox and temperature conditions. The study consisted in three batch experiments that were operated for 21 days. One batch experiment was conducted under oxic conditions at 25 °C, the second under oxic conditions at 35 °C, and the third under nitrate reducing conditions at 25 °C. These experiments allowed for a comparison of the effects of temperature and redox conditions on pharmaceutical attenuation. Each batch experiment was run in triplicate, with three reactors (3.5 L capacity each) and an additional abiotic control. Triplicate measurements were performed in this experiment to ensure reproducibility, reliability and accuracy of the results. This approach reduces any potential experimental or analytical error by analysing water samples from the three different reactors (i.e., triplicates). The bottles of the oxic experiment were sealed with three screw connectors with ports. The first port was connected to sampling tubes made of PTFE material with an internal diameter of 2 mm. The second port was connected to a one-way pressure valve that allows air to escape but prevents it from entering to avoid changes in the redox conditions. The third port was attached to an air compressor to maintain oxic conditions. In the case of the nitrate reducing conditions experiment, pure N_2 was bubbled in each reactor using a silicone tube at the beginning of the experiment to remove DO, and then the reactors were kept into a sealed chamber to ensure oxygen-depleted conditions (Fig. S1 of the Supplementary Material). Each reactor was filled with both solid and aqueous phase, maintaining a water-to-soil volumetric ratio of 85:15 to accurately simulate natural environmental conditions, ensuring consistent contact between contaminants and reactive surfaces, while enhancing the representativeness and relevance of our results. The aqueous phase consisted in a mixture of 80 % of synthetic water and 20 % of water collected from aquifers of Barcelona. The aquifers were selected according with the redox conditions of the experiment. Synthetic groundwater was prepared according with Smith et al. (2002) [25] and Bolster et al. (1999) [26] by dissolving six different salts in Milli-Q water ($MgSO_4{\cdot}7H_2O$, KNO_3, $NaHCO_3$, $CaCl_2$, $Ca(NO_3)_2{\cdot}4H_2O$, $CaSO_4{\cdot}2H_2O$) (see Text S1 and Table S2 of the Supplementary Material). Similarly, the solid phase consisted of a mixture of 80 % of fine silica sand of 0.1–0.3 mm (Sand TECHNICAL industrial washed - VWR Chemicals) and a 20 % of saturated soils collected from aquifers in Barcelona, chosen to agree the redox conditions of the respective experiments (i.e., soil for the oxic experiments was collected from a 15 m deep borehole excavated to undertake a construction in Barcelona, while soil for the nitrate reducing conditions experiment was obtained from the shallow aquifer of the Besòs river, where suboxic from up to nitrate reducing conditions prevail). The soil samples were homogenized to less than 2 mm by dry sieving. The abiotic control was used to assess whether biodegradation or sorption was a significant process in the attenuation for each pharmaceutical by comparing its concentrations with those of the batch reactors. Biological activity in the abiotic control was suppressed by adding 60 mg/L of $HgCl_2$.

Up to 7 mg/L of dissolved organic matter concentration (DOC) were added to the reactors to promote bacterial growth. DOC consisted of a mixture of lactate (easily biodegradable organic carbon; BDOC) and humic acid (refractory DOC). The mixing ratio between lactate and humic acid was varied depending of the simulated redox conditions since it is expected that oxic conditions occur when BDOC is limited because high availability of easily biodegradable DOC induces the removal of oxygen and the system to move towards the nitrate reducing zone. Thus, a mixture of 6:4 (wt/wt) of lactate and humic acid was used in the nitrate reducing experiment, while a mixture of 4:6 (wt/wt) was adopted for the oxic experiments (Li et al., 2014). Before spiking the contaminants, a twenty-five days acclimation phase was conducted (see Text S2 from the Supplementary Material). Seven mg/L of DOC were added at the beginning of the acclimation phase and just after spiking the contaminants. Pharmaceuticals were also introduced into the reactors (triplicates for each experiment and controls) after the acclimation phase by spiking a solution containing 3 μg/L of each selected compound.

Concerning the redox conditions, in the oxic experiments, the dissolved oxygen concentration was maintained at around 6 mg/L by bubbling air into the reactors twice a day for 10 min using an air compressor. The redox state of the nitrate reducing experiment was controlled through the experiment with the criteria of not exceeding 0.5 mg/L of O_2 and maintaining concentrations greater than 1 mg/L of NO_3^-. The monitoring of these water constituents showed a relatively rapid consumption of NO_3^- that required the addition of 20 mg/L of the salt $Ca(NO_3)_2{\cdot}4H_2O$ during the experiment. For temperature control, the experiments at 25 °C were carried out at the room temperature (measured with a sensor), while the experiment at 35 °C was developed inside an incubator.

2.3. Analytical methods

Pharmaceuticals were analysed using a Liquid Chromatography Tandem Mass Spectrometry analysis (LC-MSMS) [23]. For the analysis of target pharmaceuticals, 11 mL aliquots were fortified with a 15 μL isotopically labelled standard (IS) working solution. The final concentration of IS in the vial was 25 ng/mL. The samples were concentrated using a vacuum-assisted evaporation system (Syncore Analyst, BÜCHI Labortechnik AG, Switzerland) using graduated glass sample tubes with 0.3 mL residual volume. The residual volume was transferred to 2 mL amber glass LC vials and dried up after gentle nitrogen stream (React-Therm III Reacti-Vap III Heating Module PIERCE). The samples were reconstituted with 300 μL of H_2O: MeCN (95:5), sonicated, vortexed, and transferred to a conical glass insert for further analysis. At the end of the procedure, the samples were frozen at −20 °C. Waters ACQUITY UPLC system (Waters, Milford, MA) connected to a Waters XEVO TQ-S triple quadrupole mass spectrometer (Waters, Milford, MA) operating in positive electrospray ionization mode (ESI+) was used for the pharmaceutical analysis. A detailed description of the optimization of the HPLC-MS/MS conditions and data analysis has been provided in Text S3 of the Supplementary Material. Analytical reference standards (>99 % purity) and relative isotopically labelled compounds for internal standard calibration were purchased from Sigma-Aldrich (Merck, Darmstadt, Germany) and Toronto Research Chemicals (Toronto, ON, Canada), respectively. Depending on the solubility of each compound, individual standard stock solutions were prepared at a concentration of 1000 μg/mL in methanol (MeOH, ≥99.9 %) purchased from Merck (Darmstadt, Germany). For sample preparation, chromatographic separation, and the MS analysis, methanol (MeOH, ≥99.9 %), HPLC water, ammonium formate (HCO_2NH_4, ≥97.0 %, ACS grade), and formic acid (≥96.0 %, ACS grade) were supplied by Sigma-Aldrich (Merck, Darmstadt, Germany).

Dissolved oxygen was measured using the HACH HQd electrode. The Shimadzu TOC-VCPN total organic carbon analyser was used to quantify DOC, with samples filtered through 0.45 μm and acidified with HCl prior to analysis. The UV/VIS Spectrophotometer was used to measure ultraviolet absorbance (UVA) at 254 nm with a 1-cm quartz cell, also according to Standard Method 5910B. The ratio of UVA to DOC is known as the specific UV absorbance (SUVA). SUVA values, which measure UV light absorption per unit of DOC concentration, are commonly used to assess the quality of dissolved organic matter in water. Anions and cation samples were analysed by ionic chromatography (IC) and coupled plasma atomic emission spectrometry (ICP-AES), respectively.

2.4. Data analysis

2.4.1. Assessment of the removal pharmaceuticals

The attenuation of each compound in each batch reactor was calculated using Eq. (1), based on the initial concentration (C_0) (ng/L) and the concentration at the end of the incubation period (21 days) (C_f). The initial concentrations were calculated by determining the concentrations immediately after the solute application.

$$Attenuation\ (\%) = \frac{C_0 - C_f}{C_0} \times 100 \qquad \text{Eq. (1)}$$

The attenuation was divided into three categories: high (>80), moderate (30–80), and low (<30) [20].

Moreover, a first-order kinetic model was used to assess the dissipation of these compounds over time. The corresponding first-order rate law is typically used to describe the concentration decrease of a given compound with reaction time [27–29]. Eq. (2) represents the first-order rate law that has been fitted for pharmaceuticals, where C_t represents the concentration of pharmaceuticals at time t, C_0 represents the initial concentration of pharmaceuticals, and k is the first-order removal rate constant:

$$C_t = C_0 e^{-kt} \quad \text{Eq. (2)}$$

After 21 days of incubation, pharmaceuticals half-lives were calculated based on the attenuation rate obtained from the first-order reaction.

2.4.2. Statistical methods

According to the Shapiro-Wilk test, normality was not found for all the groups, neither equal variance checks with the Barlett test. Therefore, Kruskal-Wallis, a non-parametric test, was used to identify significant differences between temperature and redox on pharmaceuticals attenuation. Significance level (p) for the correlations and the comparisons was set at 5 %. The working hypotheses were: (i) there would be an increase on attenuation at higher temperatures, and (ii) the presence of oxygen in the system would increase the attenuation of the compounds. Additionally, Spearman's correlation coefficient (R^2) were calculated to evaluate relationship between the total attenuation and the physicochemical properties of the compounds such as Log K_{ow}, water solubility, molecular weight (Tables S2 and S3 of the Supplementary Material).

3. Results and discussion

The results from the 12 reactors (three triplicates and one control for each condition considered), run in parallel over 21 days, provided insights into how and to what extent 10 pharmaceuticals can be naturally attenuated under different environmental conditions. Unfortunately, microbial activity was not completely inhibited in the abiotic control of the nitrate reducing conditions, as DOC and NO_3^- concentrations sharply decreased from day 2–7 of the experiment (Fig. S2 of the Supplementary Material). This observation suggest that nitrate was reduced by the bacteria in the solid and/or liquid phase. Thus, the results of this control were not considered in the discussion (sections 3.2 to 3.5). In contrast, the inhibition of microbial activity was sustained in the abiotic controls of oxic conditions at 25 and 35 °C, as the concentrations of DO and NO_3^- were constant during the experiment. Concentration of DOC slightly varied from 7.64 to 8.48 mg/L at 25 °C and from 7.32 to 7.16 mg/L at 35 °C (Fig. S2 of the Supplementary Material).

3.1. Assessment of pharmaceuticals removal as a function of organic matter composition and physicochemical parameters

The relationship between the removal of the pharmaceuticals and their physicochemical properties evaluated using Spearman's correlation coefficient showed values lower than 0.6, indicating a moderate to weak correlation (Table S3). Therefore, there is not sufficient evidence to predict the environmental behaviour of pharmaceuticals based solely on their physicochemical properties, supporting the concept that attenuation is mainly compound-specific, as also stated by Jaeger et al. (2019) [30]. Furthermore, there is still limited understanding of the compound-specific co-metabolism and the bacterial communities required for the attenuation of pharmaceuticals [31].

The consumption of DOC under oxic conditions at different temperatures remained within the same order of magnitude (0.8 mg/L consumed by the end of our experiments in both redox scenarios), suggesting that DOC alone is not the key factor explaining pharmaceutical attenuation, as its turnover showed a similar trend across temperatures (Table 1). The slight accumulation of DOC at the end of the experiment under oxic conditions of 0.8 units could be related to microbial activity that was degrading more available forms of organic matter for further microbial metabolism [32]. In addition, the availability of nutrients, as we fed the system with BDOC at the beginning of the experiment, could lead to the accumulation of more recalcitrant forms of DOC over time [33]. This observation is also consistent with findings by Mueller et al. (2022) [8], who reported that seasonal differences in polar trace organic attenuation were not influenced by DOC, as DOC concentration remained similar between seasons. Although our experiments involved a moderate temperature variation of 10 °C, we can further confirm that this difference did not significantly impact DOC consumption. Therefore, as Mueller et al. (2022) [8] concluded, temperature is unlikely to be the main parameter responsible for variations in pharmaceuticals attenuation.

During the experiment, SUVA values under oxic conditions decreased by approximately 34 %, indicating a reduction in the aromaticity of the organic matter present in the system. This reduction suggests the breakdown of aromatic compounds and is consistent with the correlation between pharmaceutical removal and DOC trends, which is driven by biodegradation processes. Microbial activity reduces DOC while transforming pharmaceuticals to more polar metabolites, a process widely believed to enhance their attenuation efficiency [34]. Conversely, under nitrate reducing conditions, SUVA values increased by more than 100 % from their initial concentrations (Table 1). Since SUVA can be considered a proxy for aromaticity [35], the observed increase in SUVA values under nitrate reducing conditions suggests a preservation of the aromatic dissolved organic matter components. This may be related to anaerobic microbial processes involved under nitrate reducing conditions, which can selectively preserve or transform aromatic compounds [36,37]. Therefore, this conditions may influence both the composition and reactivity of dissolved organic matter, and ultimately affect the interaction with pharmaceuticals [38].

Abiotic controls showed higher DOC levels, as expected due to the inhibition of microbiological activity with $HgCl_2$ (Table 1). The SUVA absorbance results aligned with DOC trends, further supporting this relationship. In the nitrate reducing experiments, the observed DOC consumption trends indicated that the microbial communities were adapting to the reducing redox environment. The static nitrate reducing conditions seemed to enhance DOC consumption by anaerobic bacteria, suggesting that removal of pharmaceuticals through biodegradation is closely linked to DOC trends. This finding is consistent with previous studies, which highlighted that redox conditions and DOC trends are decisive for organic contaminants removal, especially in oxic zones where microbial activity drives high chemical reactivity [7]. Schaper et al. (2018) [7] also noted that BDOC can limit contaminant removal under oligotrophic conditions, while in eutrophic ones, high BDOC concentrations fuel microbial metabolism and contribute their cometabolic removal. Therefore, although it is generally assumed that oxic conditions combined with limited BDOC lead to higher attenuation due to adapted microbial communities, our results emphasize the role of anaerobic bacteria under nitrate reducing conditions in the degradation of organic compounds. Suggesting that, in natural aquifer systems, pharmaceuticals attenuation can occur not only in environments with abundant DOC but also under nitrate reducing conditions, where anaerobic bacteria promote biodegradation and cometabolic removal processes.

Finally, small differences in pH values were observed between oxic and nitrate reducing experiments. The slight changes in the pH values throughout the nitrate reducing experiment could be associated with the flushing of the reactors with nitrogen gas, observation that was reported by Burke et al. (2014) [6] with pH shifts ranging from 0.2 to 1.3 units. In the case of the oxic experiment, the pH changes could be explained by the addition of organic matter, which could shift the pH, as reported by Wenk

Table 1
Values of DOC, SUVA, pH and biogeochemical constituents in the batch reactors (average) and the abiotic control at the beginning and at the end of the experiment for the different physicochemical properties. NR = nitrate reducing.

Sample		Redox	T (°C)	DOC [mg L^{-1}]	SUVA [L mg^{-1} m]	F^- [mg L^{-1}]	Cl^- [mg L^{-1}]	NO_3^- [mg L^{-1}]	SO_4^{2-} [mg L^{-1}]	DO [mg L^{-1}]	pH
Initial	Batch reactors	Oxic	25 °C	2.65	4.96	N.D.	72.12	35.12	90.10	6.46	7.77
Final	Batch reactors	Oxic	25 °C	3.48	3.28	N.D.	71.56	35.04	90.55	7.67	8.09
Initial	Abiotic control	Oxic	25 °C	7.64	7.31	0.82	105.43	40.29	85.89	7.66	7.79
Final	Abiotic control	Oxic	25 °C	8.49	5.06	0.94	103.74	39.58	85.64	7.75	8.08
Initial	Batch reactors	Oxic	35 °C	2.90	6.14	N.D.	72.21	39.30	96.39	6.04	7.88
Final	Batch reactors	Oxic	35 °C	3.71	3.45	N.D.	70.00	38.25	95.06	6.93	8.01
Initial	Abiotic control	Oxic	35 °C	7.32	11.15	0.72	119.96	40.16	86.60	6.65	7.75
Final	Abiotic control	Oxic	35 °C	7.12	9.53	0.79	118.24	39.40	86.35	6.82	8.21
Initial	Batch reactors	NR	25 °C	7.02	1.30	N.D.	48.86	10.82	77.15	0.14	7.07
Final	Batch reactorss	NR	25 °C	3.29	2.82	N.D.	57.19	12.19	89.44	0.19	7.54
Initial	Abiotic control	NR	25 °C	10.35	1.44	0.21	86.65	24.74	89.86	0.48	7.15
Final	Abiotic control	NR	25 °C	4.11	15.49	N.D.	130.56	39.54	86.16	0.24	7.19

N.D. = not determined.

et al. (2021) [39]. However, given the small magnitude of these pH fluctuations and the lack of light exposure in our experiments, the effects of pH on contaminant transformation or microbial inhibition may be negligible.

3.2. Effect of redox conditions on the removal of pharmaceuticals

The Kruskal-Wallis test identified pharmaceuticals that exhibited significant differences in attenuation under different redox conditions, (*P-value* ≤ 0.05) (Table S4). For instance, atenolol, lamotrigine, sitagliptin, carbamazepine, and venlafaxine showed significant differences in removal efficiency between oxic and nitrate reducing conditions according to statistical analysis at 25 °C. The remaining compounds did not show significant differences (Table S4). Fig. 1 shows the normalised concentrations in the batch reactors and the abiotic control for climbazole, irbesartan, and sitagliptin, while the remaining results are shown in Fig. S3 of the Supplementary Material.

The total removal (%) of the batch reactors for the target pharmaceuticals are shown in Fig. 2. The removal of metoprolol was significantly influenced by oxygen availability, with up to 50 % of its concentration attenuated under nitrate reducing conditions (Fig. 2 and Fig. S4a of the Supplementary Material). This result aligns with the findings of [6,37–40], who reported similar percentages for metoprolol. These studies observed comparable attenuations under both oxic and nitrate reducing conditions for this compound. Reith et al. (2023) [41] reported strong attenuation of up to 75 % for metoprolol. Despite the low Log K_{ow}, Bertelkamp et al. (2014) [17] suggested that both biodegradation and irreversible sorption contribute to metoprolol removal. Rutere et al. (2021) [42] found that attenuation of metoprolol from the aqueous phase under both oxic and reducing conditions was associated with shifts in active microbial communities, highlighting its high biodegradation potential under contrasting redox conditions.

Sitagliptin, climbazole, citalopram, irbesartan and venlafaxine maybe attenuated under both oxic and nitrate reducing conditions (Figs. 1 and 2 and Fig. S3). Among the compounds studied by Jaeger et al. (2021) [40], sitagliptin, irbesartan and venlafaxine showed degradation trends with decreasing redox conditions. In particular, sitagliptin and venlafaxine were reported to have rapid degradation rates. In contrast, lamotrigine showed low attenuation under oxic and nitrate reducing conditions (Fig. 2 and Fig. S4a of the Supplementary Material).

Lamotrigine is one of the most persistent pharmaceuticals in the environment and in conventional water treatment processes [41], as well as carbamazepine [21,42–45]. Maeng et al. (2021) [46] found that the three fused rings of carbamazepine hinder its biodegradability, especially under oxic conditions. This study found that carbamazepine exhibited significant recalcitrance regardless of redox status (Fig. 2). This finding is supported by de Wilt et al. (2018) [47], who concluded that carbamazepine attenuation was low under aerobic and nitrate reducing conditions but also under sulphate reduction and methanogenic conditions, requiring additional treatment. For example, carbamazepine removals close to 60 % were reported using biofilters [48] and mixed selected cultures [49], and up to 90–95 % using advanced oxidation processes in WWTPs (i.e., ozonation and combination of ultraviolet light and hydrogen peroxide) [50].

3.3. Effect of temperature on the removal of pharmaceuticals

The role of the temperature on the natural attenuation of target pharmaceuticals was evaluated under oxic conditions at 25 and 35 °C (Fig. 2 and Fig. S5 of the Supplementary Material). Statistical analysis partially supported the initial hypothesis that increasing temperature would enhance microbial activity and improve the removal efficiency of the selected pharmaceuticals. In fact, the Kruskal-Wallis test showed statistically significant temperature dependence only for trimethoprim (Table S4). However, this hypothesis was better supported when comparing the results of the evolution of the concentrations of the batch reactors and the abiotic control (Fig. 1 and Fig. S3).

At both 25 °C and 35 °C, the attenuation of citalopram and irbesartan were equal or higher than 90 % (Fig. 2 and Fig. S5 of the Supplementary Material). The high sorption efficiency and extended residence time of these pharmaceuticals seems to be primary factor driving their attenuation [51,52]. However, differences in the sorption behaviour of citalopram and irbesartan have been reported. Citalopram may exhibit strong sorption due to electrostatic interactions with negatively charged surfaces, whereas the sorption of irbesartan may depend on its molecular configuration [51].

Despite these differences, the increase in temperature likely enhanced the overall microbial activity, contributing to slightly higher attenuation for both compounds. After analysing the effect of temperature on the overall removal efficiency and removal trend, it was found that citalopram, irbesartan, sitagliptin, and trimethoprim were slightly more attenuated at 35 °C, with an

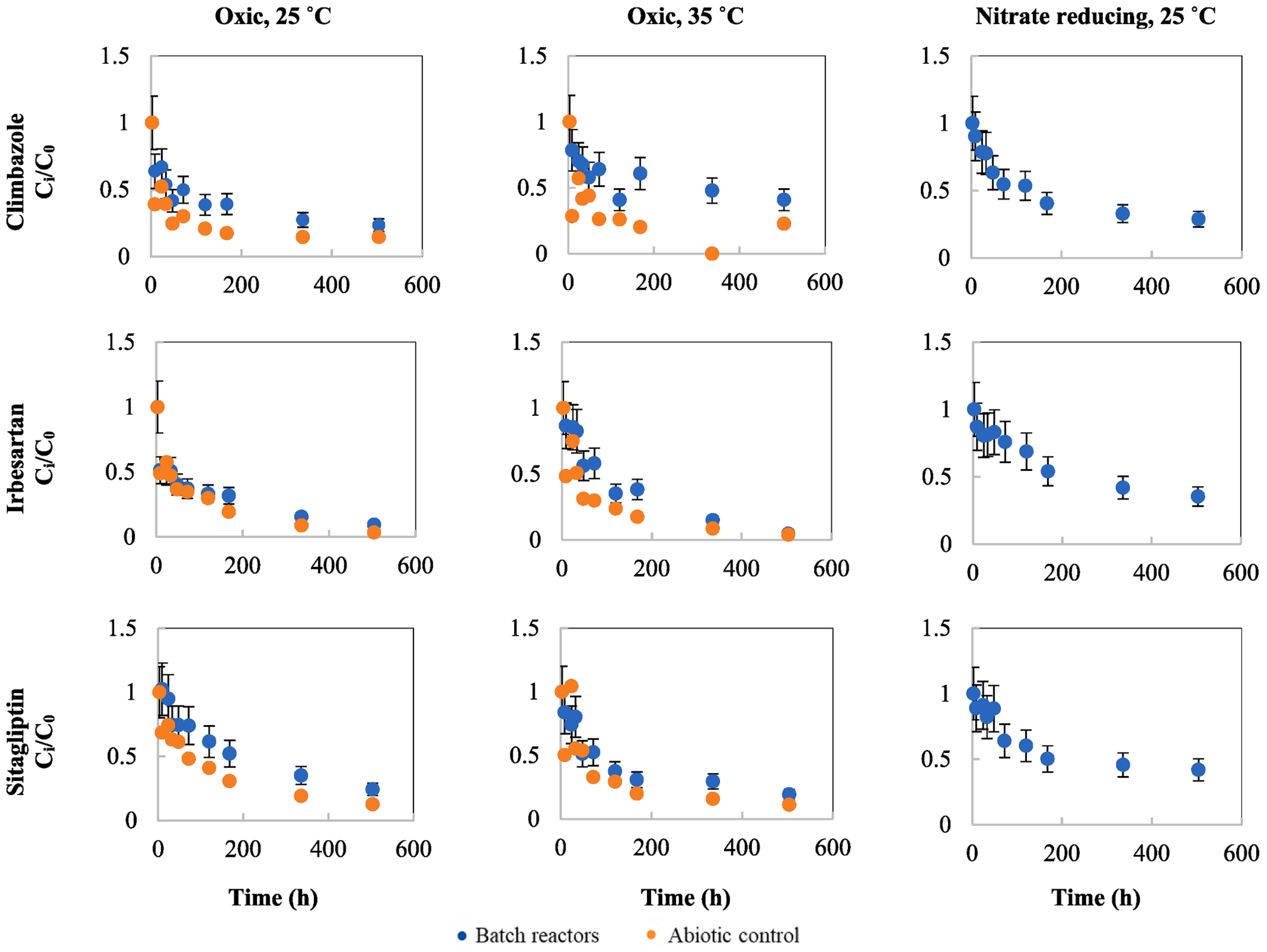

Fig. 1. Total Attenuation (average of the batch reactors, blue dotes) and sorption (abiotic control, orange dotes) for climbazole, irbesartan and sitagliptin at different redox conditions (oxic and nitrate reducing) and temperatures (25 and 35 °C). Note that the concentrations are normalised (C_t/C_0, where C_t is the concentration at each time step and C_0 is the initial concentration) for batch reactors and abiotic controls. NR= Nitrate reducing.

increase in attenuation (from 5 % to 12 %) (Fig. 2 and Fig. S5a). Similar to our results, Muñoz-Palazon et al. (2022) [53] observed improved removal rates for trimethoprim in batch experiments at the high temperature of 35 °C. Jaeger et al. (2021) [40] also reported a strong correlation between water temperature and sitagliptin attenuation rates, which is consistent with our findings. High temperatures likely increase kinetic energy, enhancing molecular movement and collisions with sorption sites, and boosting interactions between pharmaceuticals and sorbents, leading to more efficient binding and retention [4]. On the other hand, high temperatures stimulate microbial activity and enzyme production, facilitating the breakdown of pharmaceuticals through cometabolism, where microorganisms use one compound as an energy source to enhance the attenuation of others [4].

For citalopram, sorption was the main removal process at 25 °C and 35 °C (100 % and 97 %, respectively) (Fig. S5b of the supplementary material). In contrast, atenolol, climbazole, metoprolol, and venlafaxine showed lower removal values at 35 °C (Fig. 2). This could be attributed to either lower adsorption efficiency at higher temperatures or potential desorption over time. The increased energy at elevated temperatures may weaken the physical bonds between the compound and the sediment, leading to desorption. In addition, sorption has been reported to be temperature-dependent, but findings vary. For example, Mueller et al. (2022) [8] found that carbamazepine was better attenuated at low temperatures. Additionally, Wang et al. (2023) [54] have highlighted the complexity of temperature in pharmaceutical degradation, showing that elevated temperatures can reduce removal efficiency of compounds such as carbamazepine due to temperature sensitivity. This is consistent with the slight decrease in total attenuation observed for carbamazepine. However, atenolol, irbesartan, carbamazepine, and sitagliptin did not show a significant increase in sorption capacity at higher temperatures, with only minor improvements in their attenuation (from 2 to 5 %) (Fig. S5b of the Supplementary Material).

3.4. Total attenuation and role of the redox vs. temperature on the removal of pharmaceuticals

Table 2 summarizes the physicochemical conditions and the main processes controlling the attenuation of each pharmaceutical. Overall, seven pharmaceuticals (atenolol, citalopram, climbazole, irbesartan, sitagliptin, metoprolol and venlafaxine) showed moderate to high attenuation, while three (lamotrigine, carbamazepine and trimethoprim) were more persistent, with a total attenuation of less than 30 % (Fig. 2 and Table 2).

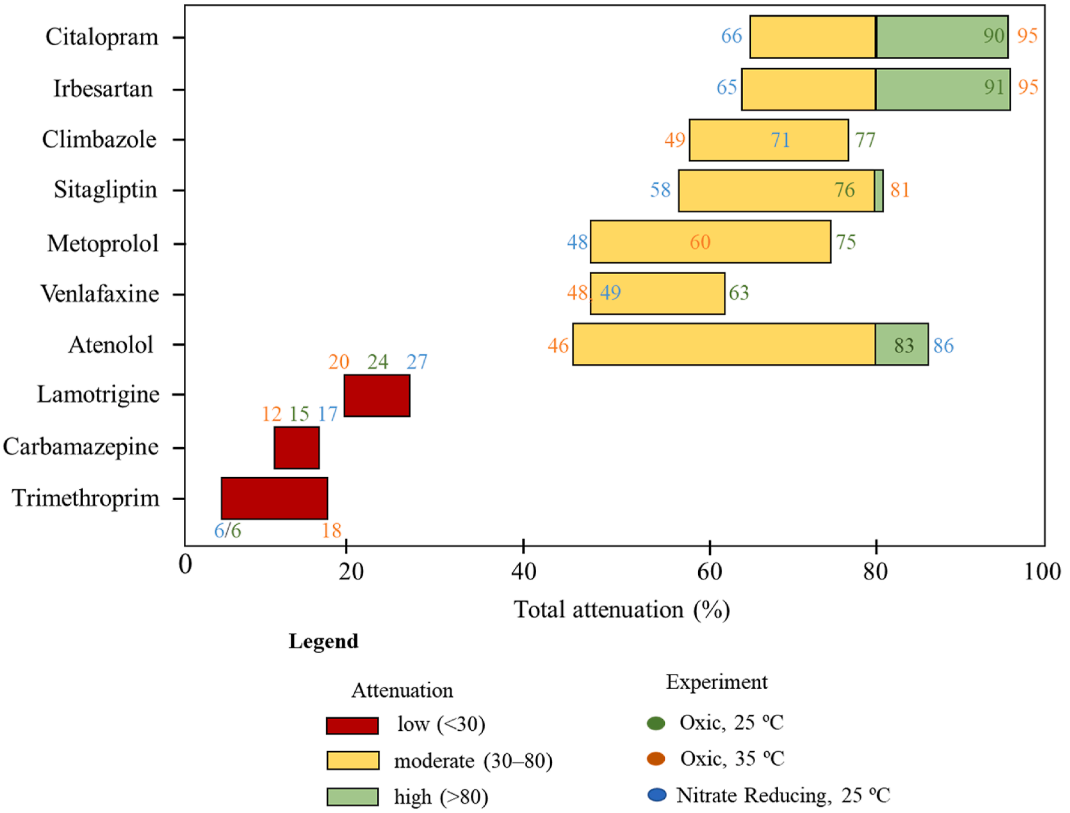

Fig. 2. Total attenuation (%) of selected pharmaceuticals in all the experiments classified by attenuation level: low (<30 %, red), moderate (30–80 %, yellow), and high (>80 %, green). Numerical labels indicate the total attenuation in oxic conditions at 25 (green) and 35 °C (orange) and nitrate reducing at 25 °C (blue).

Table 2
Physicochemical conditions and main processes that might control the attenuation of each pharmaceutical. S=Sorption, B = biodegradation, NR= Nitrate reducing, and * in oxic conditions.

Pharmaceutical	Enhancement of total attenuation		Main attenuation process*	Attenuation extent
	Oxic Conditions	Temperature		
Atenolol	No	No	S + B	Moderate (46 %, 35 °C) to high attenuation (83–86 %)
Citalopram	Yes	Yes	S	Moderate (66 %, NR) to high attenuation (90–95 %)
Climbazole	Yes	No	S	Moderate attenuation (59–77 %)
Irbesartan	Yes	Yes	S	Moderate (65 %, NR) to high attenuation (91–95 %)
Lamotrigine	No	No	B (oxic, 25 °C)	Low attenuation (20–27 %)
Sitagliptin	Yes	Yes	S	Moderate (58–76 %) to high attenuation (81 %, 35 °C)
Carbamazepine	No	No	S	Low attenuation (12–17 %)
Metoprolol	Yes	No	S + B	Moderate attenuation (48–75 %)
Trimethoprim	No	Yes	S	Low attenuation (6–18 %)
Venlafaxine	Yes	No	S	Moderate attenuation (48–63 %)

For most of the pharmaceuticals, the main attenuation process seemed to be adsorption under the conditions tested, with the exception of lamotrigine under oxic conditions at 25 °C. However, biodegradation seemed also to contribute to the attenuation of metoprolol and atenolol. Atenolol showed 30 % attenuation under oxic at 25 °C, while metoprolol resulted in 24 % and 23 % attenuation under oxic conditions at 25 °C and 35 °C, respectively (Table 2).

For atenolol, attenuation appeared to be independent of the redox conditions, with high attenuation observed under both oxic and nitrate reducing conditions at 25 °C (83 % and 86 %, respectively) (Fig. 2). Previous works have reported that atenolol and metoprolol can be biotransformed at laboratory scale under specific redox conditions. For instance, Barbieri et al. (2012) [55] pointed out that the presence of atenololic acid in the biotic but not in the abiotic batch experiment suggested that atenolol was biotransformed during the denitrifying experiment. In the same line, Rutere et al. (2021) [56] reported that metoprolol was transformed mainly to metoprolol acid in oxic microcosms, while metoprolol acid and α-hydroxymetoprolol were formed in anoxic microcosms. Irbesartan, citalopram and sitagliptin showed higher removal rates under oxic conditions (91 %, 90 % and 76 %, respectively) compared to nitrate reducing conditions (65 %, 66 % and 58 %, respectively), with their attenuation enhanced by temperature (increased by 4 %–5 %). Climbazole showed moderately high attenuation in both redox conditions (71 %–77 % in nitrate reducing and oxic conditions) and was unaffected by temperature. Metoprolol removal was significantly higher under oxic (75 %) compared to nitrate reducing conditions (48 %) and temperature did not increase its removal. Venlafaxine showed moderate removal in both redox conditions (49 %–63 %). Lastly, trimethoprim, carbamazepine, and lamotrigine were the most persistent compounds, showing low attenuation across all the physicochemical conditions considered (Fig. 2).

To sum up, oxic conditions enhanced the attenuation of pharmaceuticals in our set of batch experiments, although atenolol showed high attenuation even under nitrate reducing conditions. Oxic conditions primarily enhance pharmaceuticals attenuation through oxidation and biotransformation, and influence sorption mechanisms by modifying sorbent surface properties, altering electrostatic and hydrogen bonding interactions, and affecting sorption-desorption equilibria [57], highlighting the complexity of contaminant fate in oxygen-rich environments. Moreover, the preference for a specific redox state was not found to be entirely exclusive; compounds that degraded under oxic conditions also showed that they could be attenuated under more reducing conditions, as seen with pharmaceuticals such as climbazole, irbesartan and citalopram. Among the compounds significantly affected by oxygen availability, metoprolol showed remarkable removal rates of up to 76 % under oxic conditions, in line with findings from previous research [6,44,58–60]. Notably, a significant relationship between metoprolol degradation and water temperature has also been previously reported, with removal efficiency reaching 62 % at higher temperatures under oxic conditions [30]. Atenolol also showed high degradability under both redox conditions, which is consistent with earlier studies, indicating high biodegradation rates during groundwater recharge, regardless of redox conditions in the aquifer [61–64]. The persistence of carbamazepine and trimethoprim in the environment, due to their molecular structures, underscores the challenges associated with their degradation, especially under oxic conditions [10,21,44,45,65]. The potential remobilization of pharmaceuticals from sediments, caused by oxygen bumbling, as suggested by Jaeger et al. (2019) [30], further highlights the dynamic nature of the pharmaceutical transport and fate in aquatic systems. However, as Reith et al. (2023) [43] pointed out, the attenuation of organic contaminants is governed by the complex interplay of multiple biogeochemical factors rather than by any single parameter.

3.5. First-order constants of pharmaceuticals

The attenuation rate constants (*k*) for the target pharmaceuticals were evaluated using first-order rate law (Eq. (2)). The removal rate constants were determined experimentally, but some compounds, such as carbamazepine, lamotrigine and, trimethoprim, were excluded as they had regression coefficients (R^2) below 0.70, having poor fit to the first-order model. These pharmaceuticals did not follow the first-order model as they are persistent, with a total attenuation ranging from 6 % to 27 % (Table 2).

Table 3 summarizes the rate constants, half-life ($t_{1/2}$) and R^2 for the target pharmaceuticals using first-order rate law (Eq. (2)). As an example, Fig. 3 shows the fit of first-order model for irbesartan, metoprolol and venlafaxine across all the experiments.

Table 3
First-order rate constant (k) and half-lives ($t_{1/2}$) for the target pharmaceuticals.

Pharmaceuticals	Redox	T (°C)	k (h^{-1})	$t_{1/2}$ (h)	R^2
Atenolol	Oxic	25	0.0034	203.9	0.95
	Oxic	35	0.001	693.1	0.77
	Nitrate reducing	25	0.0038	182.4	0.98
Citalopram	Oxic	25	0.0038	182.4	0.82
	Oxic	35	0.0049	141.5	0.86
	Nitrate reducing	25	0.0027	256.7	0.76
Climbazole	Oxic	25	0.0024	288.8	0.81
	Oxic	35	0.0013	533.2	0.75
	Nitrate reducing	25	0.0024	288.8	0.84
Irbesartan	Oxic	25	0.0038	182.4	0.9
	Oxic	35	0.0058	119.5	0.98
	Nitrate reducing	25	0.002	346.6	0.94
Sitagliptin	Oxic	25	0.0027	256.7	0.97
	Oxic	35	0.0029	239.0	0.81
	Nitrate reducing	25	0.0017	407.7	0.8
Metoprolol	Oxic	25	0.0027	256.7	0.98
	Oxic	35	0.0016	433.2	0.93
	Nitrate reducing	25	0.0012	577.6	0.84
Venlafaxine	Oxic	25	0.0019	364.8	0.92
	Oxic	35	0.0011	630.1	0.88
	Nitrate reducing	25	0.0012	577.6	0.84

The rate constants (*k*) were converted to half-lives to determine the residence time of the pharmaceuticals removed from the batch reactors (Fig. 3). Half-live values ranged from 119.5 h for irbesartan to 693.1 h for atenolol, both at 35 °C. Temperature increased the removal of citalopram and irbesartan, with half-lives decreasing from 182.5 vs. 141.5 h and 182.4 to 119.5 h at 25 and 35 °C, respectively. Similarly, sitagliptin slightly increase its removal at high temperature (Table 3). Citalopram, irbesartan, sitagliptin, metoprolol, and venlafaxine presented short half-lives values under oxic conditions, whereas atenolol appeared to remove fast in nitrate reducing conditions (182.4 vs. 203 h) (Table 3). Climbazole is equally attenuated under both oxic and nitrate reducing conditions, ($t_{1/2}$ equal to 288.8 h) (Table 3).

Previous studies have reported the half-life values for the target pharmaceuticals, especially in oxic conditions (Table S5). Overall, the values of half-lives evaluated in this research were lower than those reported in the literature, although some cases fell within the reported ranges. Kinetic values are highly dependent on the specific compound and environmental conditions involved in the experiments. For example, the values of half-life for atenolol under oxic conditions, ranged from 55.2 to 2904 h [66–68]. Kodešová et al. (2020) [69] investigated the degradation of citalopram and irbesartan in soil in oxic conditions, reporting half-lives ranging from 2364 to 4932 h and from 501.6 to 669.6 h for irbesartan, respectively. Degradation of the venlafaxine in water has been studied by Rúa-Gómez and Püttmann (2013) [70] reporting of 2736 h in oxic conditions. In nitrate reducing conditions, half-life values of metoprolol ranged from 36 to 6398 h [67,71].

4. Conclusions and future prospects

The attenuation of selected pharmaceuticals was assessed using batch experiments under oxic and nitrate reducing conditions at different temperatures (25 °C and 35 °C). Laboratory experiments revealed that 7 out of 10 pharmaceuticals exhibited moderate to high attenuation, ranging from 46 % for atenolol at 35 °C to 95 % for citalopram and irbesartan at 35 °C. These results highlight the positive influence of the physicochemical properties of the groundwater on the natural attenuation of these compounds. In contrast, lamotrigine, carbamazepine and trimethoprim showed low attenuation under all conditions tested, with trimethoprim showing particularly low removal under oxic conditions (6 %). Sorption seems to be the main process contributing to the attenuation of most pharmaceuticals in oxic conditions, while biodegradation played a secondary role for a few pharmaceuticals, such as metoprolol, atenolol and lamotrigine. To further investigate the role of degradation in the target pharmaceuticals, it is of paramount importance to determine the bacterial communities present in the soil and water used in the experiments as well as to include an extra reactor simulating the biotic control. The former

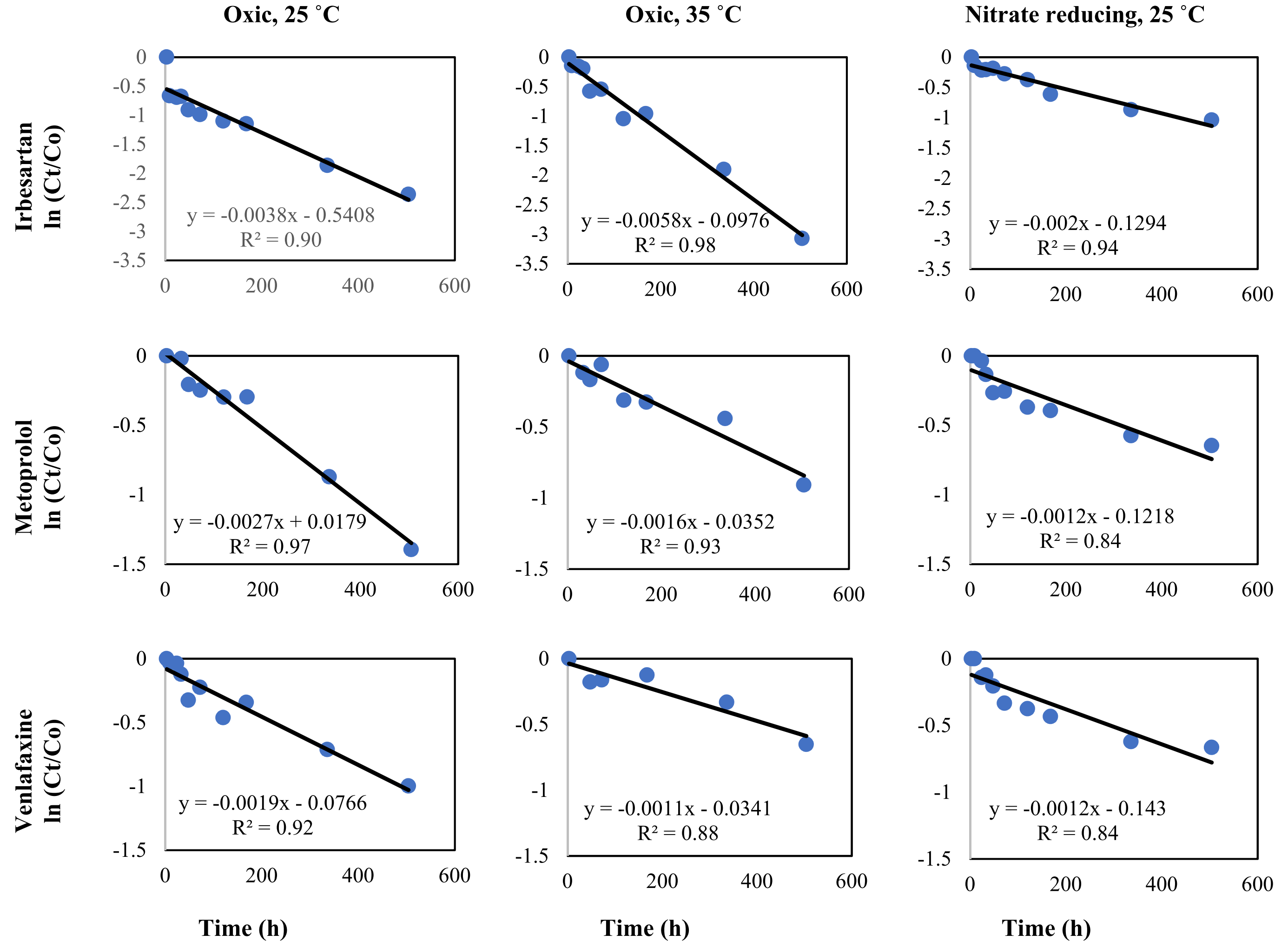

Fig. 3. First-order kinetics fitting the removal of irbesartan, metoprolol and venlafaxine at different redox conditions (oxic and nitrate reducing) and temperatures (25 and 35 °C).

will allow to identify the bacteria involve in redox processes and the latter will allow investigating the degradation of pharmaceuticals as an isolated process.

Considering all the physiochemical conditions, atenolol, citalopram, irbesartan and venlafaxine achieved attenuation of more than 55 %. The attenuation of citalopram, irbesartan and sitagliptin, was enhanced under oxic conditions and at higher temperatures, as that of trimethoprim at 35 °C. Similarly, climbazole, metoprolol and venlafaxine were more effectively attenuated under oxic conditions than under nitrate reducing ones. In contrast, atenolol, lamotrigine and carbamazepine showed better removal under nitrate reducing conditions.

In summary, this research highlights the significant influence of groundwater physicochemical properties on the natural attenuation of pharmaceuticals. While our findings make a valuable contribution to the scientific understanding of pharmaceutical behaviour at the laboratory scale, the challenge of transferring these results to field studies remains. This emphasizes the need for careful interpretation of laboratory results in natural environments. Moreover, the continuous addition of nutrients and the diverse microbial community in natural environments underline the complexity of field studies, where isolating individual effects is inherently challenging. Finally, TPs of the target pharmaceuticals were out of the scope of this research but they could also contribute to the removal of the parent compounds. Consequently, future research should focus on integrating laboratory experiments with field measurements to bridge the gap between controlled conditions and real-world environments considering also TPs. Such an approach is essential to improve groundwater contamination mitigation strategies and enhancing groundwater quality management.

CRediT authorship contribution statement

Alejandra Villa: Writing – original draft, Visualization, Methodology, Investigation, Data curation, Conceptualization. **Nafiseh Salehi Siavashani:** Writing – original draft, Data curation, Visualization. **Estanislao Pujades-Garnes:** Writing – original draft, Supervision, Methodology, Funding acquisition, Investigation, Conceptualization. **Nicola Montemurro:** Data curation, Methodology. **Sandra Pérez:** Writing – original draft, Methodology. **Jan Willem Foppen:** Investigation, Conceptualization. **Marc Teixidó:** Methodology, Investigation, Conceptualization. **Anna Jurado:** Writing – original draft, Supervision, Investigation, Funding acquisition, Conceptualization.

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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.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.emcon.2025.100537.

References

[1] J. Lukač Reberski, J. Terzić, L.D. Maurice, D.J. Lapworth, Emerging organic contaminants in karst groundwater: a global level assessment, J. Hydrol. 604 (2022) 127242, https://doi.org/10.1016/j.jhydrol.2021.127242.

[2] D.J. Lapworth, B. Lopez, V. Laabs, R. Kozel, R. Wolter, R. Ward, E. Vargas Amelin, T. Besien, J. Claessens, F. Delloye, E. Ferretti, J. Grath, Developing a groundwater watch list for substances of emerging concern: a European perspective, Environ. Res. Lett. 14 (2019) 035004, https://doi.org/10.1088/1748-9326/aaf4d7.

[3] I. Tubau, E. Vázquez-Suñé, J. Carrera, S. González, M. Petrovic, M.J. López de Alda, D. Barceló, Occurrence and fate of alkylphenol polyethoxylate degradation products and linear alkylbenzene sulfonate surfactants in urban ground water: Barcelona case study, J. Hydrol. 383 (2010) 102–110, https://doi.org/10.1016/j.jhydrol.2009.11.030.

[4] Y. Xu, X. Yu, B. Xu, D. Peng, X. Guo, Sorption of pharmaceuticals and personal care products on soil and soil components: influencing factors and mechanisms, Sci. Total Environ. 753 (2021) 141891, https://doi.org/10.1016/j.scitotenv.2020.141891.

[5] S.A. Benz, D.J. Irvine, G.C. Rau, P. Bayer, K. Menberg, P. Blum, R.C. Jamieson, C. Griebler, B.L. Kurylyk, Global groundwater warming due to climate change, Nat. Geosci. 17 (2024) 545–551, https://doi.org/10.1038/s41561-024-01453-x.

[6] V. Burke, J. Greskowiak, T. Asmuß, R. Bremermann, T. Taute, G. Massmann, Temperature dependent redox zonation and attenuation of wastewater-derived organic micropollutants in the hyporheic zone, Sci. Total Environ. 482–483 (2014) 53–61, https://doi.org/10.1016/j.scitotenv.2014.02.098.

[7] J.L. Schaper, W. Seher, G. Nützmann, A. Putschew, M. Jekel, J. Lewandowski, The fate of polar trace organic compounds in the hyporheic zone, Water Res. 140 (2018) 158–166, https://doi.org/10.1016/j.watres.2018.04.040.

[8] B.M. Mueller, H. Schulz, A. Höhne, A. Putschew, J. Lewandowski, Seasonal differences in the attenuation of polar trace organics in the hyporheic zone of an urban stream, Water Resour. Res. 58 (2022) e2021WR031272, https://doi.org/10.1029/2021WR031272.

[9] N.M. Burri, R. Weatherl, C. Moeck, M. Schirmer, A review of threats to groundwater quality in the anthropocene, Sci. Total Environ. 684 (2019) 136–154, https://doi.org/10.1016/j.scitotenv.2019.05.236.

[10] A. Villa, M.G. Ospina, M.A.C. Bolaños, T.V. Medeiros Do Nascimento, J. W. Foppen, Groundwater contamination by hospital effluent with a focus on carbamazepine and sulfamethoxazole, in: N.A. Khan, V. Vambol, S. Vambol, N. Mozaffari, N. Mozaffari (Eds.), Hosp. Wastewater Treat., IWA Publishing, 2022, pp. 101–132, https://doi.org/10.2166/9781789062625_0101.

[11] U. Khan, J. Nicell, Human health relevance of pharmaceutically active compounds in drinking water, AAPS J. 17 (2015) 558–585, https://doi.org/10.1208/s12248-015-9729-5.

[12] S.Y. Bunting, D.J. Lapworth, E.J. Crane, J. Grima-Olmedo, A. Koroša, A. Kuczyńska, N. Mali, L. Rosenqvist, M.E. Van Vliet, A. Togola, B. Lopez, Emerging organic compounds in European groundwater, Environ. Pollut. 269 (2021) 115945, https://doi.org/10.1016/j.envpol.2020.115945.

[13] A. Jurado, E. Pujades, M. Walther, M.S. Diaz-Cruz, Occurrence, fate, and risk of the organic pollutants of the surface water watch List in European groundwaters: a review, Environ. Chem. Lett. (2022), https://doi.org/10.1007/s10311-022-01441-w.

[14] C. Postigo, D. Barceló, Synthetic organic compounds and their transformation products in groundwater: occurrence, fate and mitigation, Sci. Total Environ. 503–504 (2015) 32–47, https://doi.org/10.1016/j.scitotenv.2014.06.019.

[15] O. Nikolenko, E. Pujades, M. Teixidó, C. Sáez, A. Jurado, Contaminants of emerging concern in the urban aquifers of Barcelona: do they hamper the use of groundwater? Chemosphere 341 (2023) 140023 https://doi.org/10.1016/j.chemosphere.2023.140023.

[16] J. Greskowiak, E. Hamann, V. Burke, G. Massmann, The uncertainty of biodegradation rate constants of emerging organic compounds in soil and groundwater – a compilation of literature values for 82 substances, Water Res. 126 (2017) 122–133, https://doi.org/10.1016/j.watres.2017.09.017.

[17] C. Bertelkamp, J. Reungoat, E.R. Cornelissen, N. Singhal, J. Reynisson, A.J. Cabo, J.P. Van Der Hoek, A.R.D. Verliefde, Sorption and biodegradation of organic micropollutants during river bank filtration: a laboratory column study, Water Res. 52 (2014) 231–241, https://doi.org/10.1016/j.watres.2013.10.068.

[18] A. Abdelrady, S. Sharma, A. Sefelnasr, A. Abogbal, M. Kennedy, Investigating the impact of temperature and organic matter on the removal of selected organic micropollutants during bank filtration: a batch study, J. Environ. Chem. Eng. 7 (2019) 102904, https://doi.org/10.1016/j.jece.2019.102904.

[19] K.L. Vaughan, M.C. Rabenhorst, B.A. Needelman, Saturation and temperature effects on the development of reducing conditions in soils, Soil Sci. Soc. Am. J. 73 (2009) 663–667, https://doi.org/10.2136/sssaj2007.0346.

[20] A. Jurado, E. Vázquez-Suñé, E. Pujades, Urban groundwater contamination by non-steroidal anti-inflammatory drugs, Water 13 (2021) 720, https://doi.org/10.3390/w13050720.

[21] R.M. Baena-Nogueras, E. González-Mazo, P.A. Lara-Martín, Degradation kinetics of pharmaceuticals and personal care products in surface waters: photolysis vs biodegradation, Sci. Total Environ. 590–591 (2017) 643–654, https://doi.org/10.1016/j.scitotenv.2017.03.015.

[22] A. Kiecak, L. Sassine, M. Boy-Roura, M. Elsner, J. Mas-Pla, C. Le Gal La Salle, C. Stumpp, Sorption properties and behaviour at laboratory scale of selected pharmaceuticals using batch experiments, J. Contam. Hydrol. 225 (2019) 103500, https://doi.org/10.1016/j.jconhyd.2019.103500.

[23] F. Labad, A. Ginebreda, R. Criollo, E. Vázquez-Suñé, S. Pérez, A. Jurado, Occurrence, data-based modelling, and risk assessment of emerging contaminants in an alluvial aquifer polluted by river recharge, Environ. Pollut. 316 (2023) 120504, https://doi.org/10.1016/j.envpol.2022.120504.

[24] O. Nikolenko, F. Labad, E. Pujades, L. Scheiber, S. Pérez, A. Ginebreda, A. Jurado, Combination of multivariate data analysis and mixing modelling to assess tracer potential of contaminants of emerging concern in aquifers, Environ. Pollut. 341 (2024) 123020, https://doi.org/10.1016/j.envpol.2023.123020.

[25] E.J. Smith, W. Davison, J. Hamilton-Taylor, Methods for preparing synthetic freshwaters, Water Res. 36 (2002) 1286–1296, https://doi.org/10.1016/S0043-1354(01)00341-4.

[26] C.H. Bolster, A.L. Mills, G.M. Hornberger, J.S. Herman, Spatial distribution of deposited bacteria following Miscible Displacement Experiments in intact cores, Water Resour. Res. 35 (1999) 1797–1807, https://doi.org/10.1029/1999WR900031.

[27] S. Alam, A. Borthakur, S. Ravi, M. Gebremichael, S.K. Mohanty, Managed aquifer recharge implementation criteria to achieve water sustainability, Sci. Total Environ. 768 (2021) 144992, https://doi.org/10.1016/j.scitotenv.2021.144992.

[28] C.A.J. Appelo, D. Postma, Geochemistry, Groundwater and Pollution, 0 ed., CRC Press, 2004 https://doi.org/10.1201/9781439833544.

[29] L. Yin, R. Ma, B. Wang, H. Yuan, G. Yu, The degradation and persistence of five pharmaceuticals in an artificial climate incubator during a one year period, RSC Adv. 7 (2017) 8280–8287, https://doi.org/10.1039/C6RA28351A.

[30] A. Jaeger, M. Posselt, A. Betterle, J. Schaper, J. Mechelke, C. Coll, J. Lewandowski, Spatial and temporal variability in attenuation of polar organic micropollutants in an urban lowland stream, Environ. Sci. Technol. 53 (2019) 2383–2395, https://doi.org/10.1021/acs.est.8b05488.

[31] A. Jaeger, C. Coll, M. Posselt, J. Mechelke, C. Rutere, A. Betterle, M. Raza, A. Mehrtens, K. Meinikmann, A. Portmann, T. Singh, P.J. Blaen, S. Krause, M. A. Horn, J. Hollender, J.P. Benskin, A. Sobek, J. Lewandowski, Using recirculating flumes and a response surface model to investigate the role of hyporheic exchange and bacterial diversity on micropollutant half-lives, Environ. Sci. Process. Impacts 21 (2019) 2093–2108, https://doi.org/10.1039/C9EM00327D.

[32] Y. Li, Z. Chen, J. Chen, M.J. Castellano, C. Ye, N. Zhang, Y. Miao, H. Zheng, J. Li, W. Ding, Oxygen availability regulates the quality of soil dissolved organic matter by mediating microbial metabolism and iron oxidation, Glob. Change Biol. 28 (2022) 7410–7427, https://doi.org/10.1111/gcb.16445.

[33] J.B. Cotner, N.J. Anderson, C. Osburn, Accumulation of recalcitrant dissolved organic matter in aerobic aquatic systems, Limnol. Oceanogr. Lett. 7 (2022) 401–409, https://doi.org/10.1002/lol2.10265.

[34] Koleta Majewska Kamińska, Anna Skwierawska, Natalia Łukasik, Katarzyna Kozłowska-Tylingo, Removal of pharmaceutically active compounds in sequencing batch reactor, J. Environ. Sci. Eng. A (2015) 4, https://doi.org/10.17265/2162-5298/2015.09.005.

[35] J.L. Weishaar, G.R. Aiken, B.A. Bergamaschi, M.S. Fram, R. Fujii, K. Mopper, Evaluation of specific ultraviolet absorbance as an indicator of the chemical composition and reactivity of dissolved organic carbon, Environ. Sci. Technol. 37 (2003) 4702–4708, https://doi.org/10.1021/es030360x.

[36] A. Paul, C. Dziallas, E. Zwirnmann, E.T. Gjessing, H.-P. Grossart, UV irradiation of natural organic matter (NOM): impact on organic carbon and bacteria, Aquat. Sci. 74 (2012) 443–454, https://doi.org/10.1007/s00027-011-0239-y.

[37] W. Yu, H. Yang, J. Chen, P. Liao, J. Wu, L. Jiang, W. Guo, Molecular insights into the microbial degradation of sediment-derived DOM in a macrophyte-dominated lake under aerobic and hypoxic conditions, Sci. Total Environ. 916 (2024) 170257, https://doi.org/10.1016/j.scitotenv.2024.170257.

[38] S. Bagnis, M.F. Fitzsimons, J. Snape, A. Tappin, S. Comber, Processes of distribution of pharmaceuticals in surface freshwaters: implications for risk assessment, Environ. Chem. Lett. 16 (2018) 1193–1216, https://doi.org/10.1007/s10311-018-0742-7.

[39] J. Wenk, C. Graf, M. Aeschbacher, M. Sander, S. Canonica, Effect of solution pH on the dual role of dissolved organic matter in sensitized pollutant photooxidation, Environ. Sci. Technol. 55 (2021) 15110–15122, https://doi.org/10.1021/acs.est.1c03301.

[40] A. Jaeger, M. Posselt, J.L. Schaper, A. Betterle, C. Rutere, C. Coll, J. Mechelke, M. Raza, K. Meinikmann, A. Portmann, P.J. Blaen, M.A. Horn, S. Krause, J. Lewandowski, Transformation of organic micropollutants along hyporheic flow in bedforms of river-simulating flumes, Sci. Rep. 11 (2021) 13034, https://doi.org/10.1038/s41598-021-91519-2.

[41] O.S. Keen, I. Ferrer, E. Michael Thurman, K.G. Linden, Degradation pathways of lamotrigine under advanced treatment by direct UV photolysis, hydroxyl radicals, and ozone, Chemosphere 117 (2014) 316–323, https://doi.org/10.1016/j.chemosphere.2014.07.085.

[42] P. Manyepa, K.M. Gani, M. Seyam, I. Banoo, B. Genthe, S. Kumari, F. Bux, Removal and risk assessment of emerging contaminants and heavy metals in a wastewater reuse process producing drinkable water for human consumption, Chemosphere 361 (2024) 142396, https://doi.org/10.1016/j.chemosphere.2024.142396.

[43] C.J. Reith, S. Spahr, A. Putschew, J. Lewandowski, Attenuation of trace organic compounds along hyporheic flow paths in a lowland sandbed stream, J. Hydrol. 624 (2023) 129905, https://doi.org/10.1016/j.jhydrol.2023.129905.

[44] N. Schmidt, D. Page, A. Tiehm, Biodegradation of pharmaceuticals and endocrine disruptors with oxygen, nitrate, manganese (IV), iron (III) and sulfate as electron acceptors, J. Contam. Hydrol. 203 (2017) 62–69, https://doi.org/10.1016/j.jconhyd.2017.06.007.

[45] Y. Zhang, S.-U. Geißen, C. Gal, Carbamazepine and diclofenac: removal in wastewater treatment plants and occurrence in water bodies, Chemosphere 73 (2008) 1151–1161, https://doi.org/10.1016/j.chemosphere.2008.07.086.

[46] S.K. Maeng, S.K. Sharma, K. Lekkerkerker-Teunissen, G.L. Amy, Occurrence and fate of bulk organic matter and pharmaceutically active compounds in managed aquifer recharge: a review, Water Res. 45 (2011) 3015–3033, https://doi.org/10.1016/j.watres.2011.02.017.

[47] A. De Wilt, Y. He, N. Sutton, A. Langenhoff, H. Rijnaarts, Sorption and biodegradation of six pharmaceutically active compounds under four different redox conditions, Chemosphere 193 (2018) 811–819, https://doi.org/10.1016/j.chemosphere.2017.11.084.

[48] P. Biswas, B.P. Vellanki, Application of biofilters developed under different influent characteristics in attenuating emerging contaminants from wastewater, Environ. Sci. Water Res. Technol. 9 (2023) 2581–2594, https://doi.org/10.1039/D3EW00201B.

[49] H. Ha, B. Mahanty, S. Yoon, C.-G. Kim, Degradation of the long-resistant pharmaceutical compounds carbamazepine and diatrizoate using mixed microbial culture, J. Environ. Sci. Health Part A 51 (2016) 467–471, https://doi.org/10.1080/10934529.2015.1128712.

[50] Y.-Y. Lee, C. Fan, F. Haque, Hybrid combination of advanced oxidation and biological processes for the micropollutant removal of carbamazepine, npj Clean Water 5 (2022) 60, https://doi.org/10.1038/s41545-022-00203-z.

[51] A. Klement, R. Kodešová, M. Bauerová, O. Golovko, M. Kočárek, M. Fér, O. Koba, A. Nikodem, R. Grabic, Sorption of citalopram, irbesartan and fexofenadine in soils: estimation of sorption coefficients from soil properties, Chemosphere 195 (2018) 615–623, https://doi.org/10.1016/j.chemosphere.2017.12.098.

[52] J. Kwon, K.L. Armbrust, Degradation of citalopram by simulated sunlight, Environ. Toxicol. Chem. 24 (2005) 1618–1623, https://doi.org/10.1897/04-522R.1.

[53] B. Muñoz-Palazon, A. Rosa-Masegosa, R. Vilchez-Vargas, A. Link, S. Gorrasi, J. Gonzalez-Lopez, A. Gonzalez-Martinez, Biological removal processes in aerobic granular sludge for treating synthetic hospital wastewater: effect of temperature, J. Water Process Eng. 47 (2022) 102691, https://doi.org/10.1016/j.jwpe.2022.102691.

[54] Q. Wang, W. Zheng, Y. Wang, T. Zhang, Z. Zhou, Z. Wu, Insights into effects of operating temperature on the removal of pharmaceuticals/pesticides/synthetic organic compounds by membrane bioreactor process, Environ. Pollut. 335 (2023) 122145, https://doi.org/10.1016/j.envpol.2023.122145.

[55] M. Barbieri, T. Licha, K. Nödler, J. Carrera, C. Ayora, X. Sanchez-Vila, Fate of β-blockers in aquifer material under nitrate reducing conditions: batch experiments, Chemosphere 89 (2012) 1272–1277, https://doi.org/10.1016/j.chemosphere.2012.05.019.

[56] C. Rutere, M. Posselt, A. Ho, M.A. Horn, Biodegradation of metoprolol in oxic and anoxic hyporheic zone sediments: unexpected effects on microbial communities, Appl. Microbiol. Biotechnol. 105 (2021) 6103–6115, https://doi.org/10.1007/s00253-021-11466-w.

[57] J.O. Ighalo, P.-S. Yap, K.O. Iwuozor, C.O. Aniagor, T. Liu, K. Dulta, F.U. Iwuchukwu, S. Rangabhashiyam, Adsorption of persistent organic pollutants (POPs) from the aqueous environment by nano-adsorbents: a review, Environ. Res. 212 (2022) 113123, https://doi.org/10.1016/j.envres.2022.113123.

[58] C. Bertelkamp, A.R.D. Verliefde, K. Schoutteten, L. Vanhaecke, J. Vanden Bussche, N. Singhal, J.P. Van Der Hoek, The effect of redox conditions and adaptation time on organic micropollutant removal during river bank filtration: a laboratory-scale column study, Sci. Total Environ. 544 (2016) 309–318, https://doi.org/10.1016/j.scitotenv.2015.11.035.

[59] W. Liu, N.B. Sutton, H.H.M. Rijnaarts, A.A.M. Langenhoff, Anoxic conditions are beneficial for abiotic diclofenac removal from water with manganese oxide (MnO2), Environ. Sci. Pollut. Res. 25 (2018) 10141–10147, https://doi.org/10.1007/s11356-018-1569-2.

[60] D. Oberleitner, W. Schulz, A. Bergmann, C. Achten, Impact of seasonality, redox conditions, travel distances and initial concentrations on micropollutant removal during riverbank filtration at four sites, Chemosphere 250 (2020) 126255, https://doi.org/10.1016/j.chemosphere.2020.126255.

[61] S. Huntscha, H.P. Singer, C.S. McArdell, C.E. Frank, J. Hollender, Multiresidue analysis of 88 polar organic micropollutants in ground, surface and wastewater using online mixed-bed multilayer solid-phase extraction coupled to high performance liquid chromatography–tandem mass spectrometry, J. Chromatogr. A 1268 (2012) 74–83, https://doi.org/10.1016/j.chroma.2012.10.032.

[62] R. López-Serna, A. Jurado, E. Vázquez-Suñé, J. Carrera, M. Petrović, D. Barceló, Occurrence of 95 pharmaceuticals and transformation products in urban groundwaters underlying the metropolis of Barcelona, Spain, Environ. Pollut. 174 (2013) 305–315, https://doi.org/10.1016/j.envpol.2012.11.022.

[63] K. Markiewicz, A. Białk-Bielińska, P. Łukaszewicz, P. Stepnowski, J. Dołżonek, Insight into the sorption of 5-fluorouracil and methotrexate onto soil–pH, ionic strength, and Co-contaminant influence, Molecules 26 (2021) 1674, https://doi.org/10.3390/molecules26061674.

[64] J. Regnery, A.D. Wing, M. Alidina, J.E. Drewes, Biotransformation of trace organic chemicals during groundwater recharge: how useful are first-order rate constants? J. Contam. Hydrol. 179 (2015) 65–75, https://doi.org/10.1016/j.jconhyd.2015.05.008.

[65] A. Jurado, R. López-Serna, E. Vázquez-Suné, J. Carrera, E. Pujades, M. Petrovic, D. Barceló, Occurrence of carbamazepine and five metabolites in an urban aquifer, Chemosphere 115 (2014) 47–53, https://doi.org/10.1016/j.chemosphere.2014.01.014.

[66] M. Ramil, T. El Aref, G. Fink, M. Scheurer, T.A. Ternes, Fate of beta blockers in aquatic-sediment systems: sorption and biotransformation, Environ. Sci. Technol. 44 (2010) 962–970, https://doi.org/10.1021/es9027452.

[67] V. Burke, D. Richter, U. Hass, U. Duennbier, J. Greskowiak, G. Massmann, Redox-dependent removal of 27 organic trace pollutants: compilation of results from tank aeration experiments, Environ. Earth Sci. 71 (2014) 3685–3695, https://doi.org/10.1007/s12665-013-2762-8.

[68] H. Yamamoto, Y. Nakamura, S. Moriguchi, Y. Nakamura, Y. Honda, I. Tamura, Y. Hirata, A. Hayashi, J. Sekizawa, Persistence and partitioning of eight selected pharmaceuticals in the aquatic environment: laboratory photolysis, biodegradation, and sorption experiments, Water Res. 43 (2009) 351–362, https://doi.org/10.1016/j.watres.2008.10.039.

[69] R. Kodešová, A. Chroňáková, K. Grabicová, M. Kočárek, Z. Schmidtová, Z. Frková, A. Vojs Staňová, A. Nikodem, A. Klement, M. Fér, R. Grabic, How microbial community composition, sorption and simultaneous application of six pharmaceuticals affect their dissipation in soils, Sci. Total Environ. 746 (2020) 141134, https://doi.org/10.1016/j.scitotenv.2020.141134.

[70] P.C. Rúa-Gómez, W. Püttmann, Degradation of lidocaine, tramadol, venlafaxine and the metabolites O-desmethyltramadol and O-desmethylvenlafaxine in surface waters, Chemosphere 90 (2013) 1952–1959, https://doi.org/10.1016/j.chemosphere.2012.10.039.

[71] E. Hamann, P.J. Stuyfzand, J. Greskowiak, H. Timmer, G. Massmann, The fate of organic micropollutants during long-term/long-distance river bank filtration, Sci. Total Environ. 545–546 (2016) 629–640, https://doi.org/10.1016/j.scitotenv.2015.12.057.