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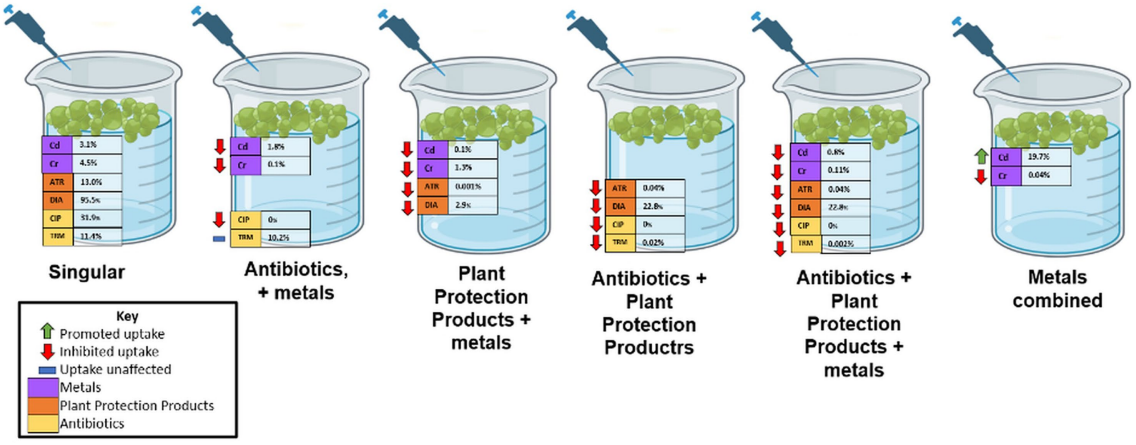
Phytoremediation of emerging and traditional contaminants in environmentally relevant co-exposure settings: a tool to reduce environmental hazards?

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Water bodies are routinely exposed to a mixture of contaminants via various anthropogenic exposure mechanisms. Phytoremediation offers a low-cost nature-based solution to remove contaminants from polluted water bodies. The ability of the aquatic macrophyte, *Lemna minor* to accumulate both organic [antibiotics, plant protection products (PPPs)] and inorganic contaminants (metals), individually and in co-contaminant mixtures, was evaluated to offer new insights into its potential for phytoremediation in realistic exposure scenarios. When exposed individually, diazinon accumulated to the greatest extent in *L. minor*, (95.51 ± 2.13%), followed by ciprofloxacin (31.91 ± 6.07%), atrazine (13.04 ± 3.44%), trimethoprim (11.43 ± 5.77%), chromium (4.5 ± 2.74%), and cadmium (3.08 ± 1.36%) These findings highlight that accumulation by *L. minor* is highly compound specific. Further results revealed that contaminant co-exposure significantly (p = 0.002–0.029) influenced the uptake of selected contaminants. The presence of chromium increased cadmium uptake by a factor of 6.34, while the combination of multiple contaminants (e.g., metals, PPPs, and antibiotics) reduced uptake. Changes in uptake capacity have potential knock-on effects for environmental risk assessments, including an assessment of the potential for antibiotics to contribute to the development of antibiotic resistance. For ciprofloxacin and diazinon, phytoremediation for 72 h was sufficient to reduce the concentration of a single antibiotic below concentrations predicted to result in the development of resistance (PNEC-R) or effects to *Daphnia magna*. In co-contaminant mixtures a risk was identified with RQ values being a factor of 17 greater than the toxicity threshold for *D. magna*. Our results support the need for a more comprehensive assessment of the impact of co-contaminant mixtures on *L. minor* uptake capacity to optimise phytoremediation strategies for improved water quality and reduced environmental risks.

KEYWORDS
antibiotics, co-contaminants, *Lemna minor*, metals, pesticides, plant protection products (PPPs), removal, uptake



GRAPHICAL ABSTRACT
Co-exposure treatments: Organics combined (atrazine, diazinon, ciprofloxacin, trimethoprim), antibiotics/PPPs and metals combined (either ciprofloxacin, trimethoprim, cadmium and chromium, or atrazine, diazinon, cadmium, and chromium), antibiotics PPPs, and metals (atrazine, diazinon, ciprofloxacin, trimethoprim, cadmium, chromium) (Supplementary Table 1).

1 Introduction

Organic and inorganic contaminants routinely enter freshwater compartments via the discharge of both treated and untreated wastewater effluent and to a lesser extent via runoff from agricultural fields receiving organic wastes or wastewater (Robinson et al., 2023; Wronski and Brooks, 2023). Typically, these contaminants are associated with poor removal rates within Wastewater Treatment Plants (WWTPs) (Gros et al., 2010). For example, a study conducted in France monitored 261 organic and 25 inorganic contaminants at 760 WWTPs and reported that 86 and 100% of the assessed organic and inorganic chemicals, respectively were identified in both the influent and effluent (Aemig et al., 2021). Furthermore, sewage overspills routinely result in the direct release of untreated wastewater into rivers (Kay et al., 2017). The continuous migration and persistence of many organic and inorganic contaminants results in their ubiquity in environmental aquatic systems with concentrations typically reported to range from 1 ng/L to 50 µg/L and 0.56 to 1338.99 µg/L, respectively (Sousa et al., 2018; Zhou et al., 2020).

Concurrently with increasing reports of chemical presence in global freshwater systems, significant declines in biodiversity are also being observed (Groh et al., 2022; Vorosmarty et al., 2010). Biodiversity is a complex system and is well known to be influenced by numerous factors acting in synergy and whilst it is difficult to specifically link loss of biodiversity to chemical pollution due to a multitude of other compounding factors recent reports have outlined that 62% of surface water bodies within the EU are failing to reach good chemical status for 'priority substances; and only 15% of UK rivers are achieving a good ecological status (Posthuma et al., 2020; The Rivers Trust, 2024). Furthermore, the United Nations Environment Programme (UNEP) and other organisations have specifically identified environmental pollution by anthropogenic chemicals as a major driver of biodiversity loss. For example, this has been explicitly recognized in the Post-2020 Global Biodiversity Framework discussions in which the current draft Target 7 addresses pollution and calls for the reduction of pesticide and nutrient inputs to the environment (Convention on Biological Diversity, 2022).

The presence of antibiotics, metals, and plant protection products (PPPs) in aquatic systems poses a distinct environmental and public health concern, as these substances can create conditions that promote the development and spread of antibiotic resistance (ABR). Antibiotics exert selective pressure on microbial communities, encouraging the survival and proliferation of resistant strains (Andersson and Hughes, 2014). Similarly, metals, such as heavy metals, can co-select for resistance by inducing stress responses or interacting with genetic mechanisms that confer cross-resistance to antibiotics (Singer et al., 2016). PPPs, including pesticides and herbicides, may also contribute by disrupting microbial ecosystems or enhancing horizontal gene transfer of resistance genes (Qiu et al., 2022). Together, these contaminants create pressures and amplify the risk of resistant pathogens emerging and persisting in aquatic environments.

Recent scientific advances linking human health and ecosystem effects to the presence of contaminants in the environment necessitates the need to clean up and remove these pollutants from freshwater systems. Considerable research focus has been on the development and use of advanced treatment technologies to treat wastewater such as activated carbon, ultraviolet (UV) irradiation or ozonation, and innovative advancements such as *Chlorella pyrenoidosa*—bioremediation, NF270-2540 membranes and reverse osmosis (Ahmed et al., 2017, Angeles et al., 2020). However, even though advanced treatment options may offer a suitable means of contaminant removal, they are associated with high running and implementation costs (Rizzo et al., 2019). The large-scale operation of advanced wastewater treatment plants is therefore not always possible due to limited resources (Garrone et al., 2018). As such, it has been estimated that over 80% of the world's sewage is discharged untreated into the environment (World Health Organization, 2021) where limited remediation technologies can be employed without disrupting ecological systems.

It is therefore imperative that options to treat contaminated wastewater and the receiving environment (e.g., rivers and subsequently groundwater and terrestrial compartments) include

treatment solutions that are sensitive to the needs of the population and available resources and infrastructure. Phytoremediation is one such potential solution, which utilises the ability of plants to take up and accumulate pollutants from the surrounding contaminated environment. Hyperaccumulating plants such as duckweed (*Lemna* spp.), water hyacinth (*Eichhornia crassipes*), and water lettuce (*Pistia stratiotes*) are well known for their ability to tolerate high levels of chemical pollution whilst having the capacity to take up contaminants via their roots and translocate these chemicals to above ground parts of the plant. Plants can act as environmental filters by removing harmful substances from water and soil, thereby improving environmental health (Chowdhury et al., 2023). To date, a considerable amount of research has focused on the capacity of duckweed (*Lemna* spp.) to remove pollutants in controlled laboratory settings. For example, Hanafiah et al. (2021) demonstrated high removal rates of heavy metals (As, Cd, Cu, Fe, Pb, Zn, Mg, and Hg) by *L. minor* (<100%) and Reinhold et al. (2010) observed that 47.5 and 55.6% of ibuprofen and fluoxetine were removed in 9 d, respectively, by *L. minor*. The freshwater aquatic macrophytes have also been demonstrated to take up and accumulate PPPs such as isoproturon and glyphosate from contaminated river water (25 and 8% removal, respectively, after 4 d of exposure) (Dosnon-Olette et al., 2011).

Whilst research has also demonstrated the capacity of aquatic macrophytes to remove pollutants *in situ* such as acid mine drainage sites near the Maden Copper deposit in Turkey (Sasmaz Kislioglu, 2023) where chemicals are present in complex co-contaminant mixtures, assessments typically focus on assessing the removal capacity of singular chemical or chemical groups. Neglecting to account for the presence of co-contaminants across a diverse range of chemical classes provides an incomplete picture of removal efficiency as competitive interactions with other pollutants may affect the ability of aquatic macrophytes to sequester contaminants. There is a clear need to explore how phytoremediation is affected by the presence of co-contaminants given the increasing reliance on more 'sustainable' wastewater treatment options, such as phytoremediation, which are and a solution to pressing global water quality challenges.

This study contributes novel perspectives on the ability of *L. minor* to accumulate chemicals from three distinct chemical classes in multiple co-contaminant combinations. Given the potential ecotoxicological implications and suggested contribution to the development of ABR, selected contaminants included metals (cadmium and chromium), antibiotics (ciprofloxacin and trimethoprim), and PPPs (atrazine and diazinon).

2 Materials and methods

2.1 Chemicals and reagents

All chemicals and stocks were of the highest possible purity (<97%) and purchased from sigma Aldrich and Toronto Research Chemicals. Methanol and acetonitrile were LCMS grade (<99% pure) and obtained from Fisher and SLS, respectively, Internal standards atrazine—D5 and diazinon—D10 and ciprofloxacin—D8 which were purchased from BioServ UK Limited and Cambridge Bioscience, respectively. Reagents/buffers (Citric acid, orthophosphate, Na_2EDTA, Hoagland's No.2 Basal Salt Mixture) were also of the highest available purity (<98%) and purchased from VWR, Fischer, and SLS.

2.2 Mesocosms and chemical exposure

Lemna minor were obtained from a local freshwater pond (West Yorkshire, UK) free from contaminant exposure after which *L. minor* were cultured in the laboratory in Hoagland's growth media under controlled conditions (12 h light, 70% humidity, 23 °C) for 2 weeks prior to use in the experiments. Hoagland No. 2 Basal Salt Mixture was prepared as per the manufacturers guidelines (1.6 grams per 1 L) (Hoagland and Arnon, 1938). Each treatment consisted of a 72 h exposure in a 250 mL glass beaker with 100 mL of Hoagland's growth media. Contaminants were added to the growth media individually (herein referred to as 'singular' exposures) to assess uptake capacity on a singular contaminant basis as well as in co-contaminant combinations as outlined in Supplementary Table 1. After dosing each beaker, 3 g (ww) of *L. minor* were transferred and the beakers kept under controlled conditions for the duration of the study, replicating the culturing conditions employed as outlined above. Each treatment was performed in triplicate with exposure concentrations based on reported environmental concentrations for each contaminant. These were 1.1 mg/L, 3.2 mg/L for the antibiotic's ciprofloxacin and trimethoprim, respectively and 10 mg/L and 0.15 mg/L for PPPs atrazine and diazinon, respectively (Table 1) (Umweltbundesamt, 2022; Manaia et al., 2024). Controls were also employed where *L. minor* were kept under similar exposure conditions but in the absence of contaminants.

2.3 Quantification of contaminant residues

2.3.1 Organics

After 72 h 3 g of *L. minor* from each treatment were removed from solution, rinsed in deionised water, gently dried with paper towel

TABLE 1 Physiochemical properties of the selected contaminants and the exposure concentrations.

Classification	Chemical	Molecular weight (g/mol)	pK_a (a), pK_a (b)	log K_{OW}	Solubility in water (mg/L) pH 7 25 °C	Exposure concentration (mg/L)
Antibiotics (organic)	Ciprofloxacin	331.34[1]	6.16 (a), 8.74 (b)[2]	0.28[3]	36,000[4]	1.1
	Trimethoprim	290.32[1]	7.12 (a)[5]	0.91[6]	400[7]	3.2
PPPs (organic)	Atrazine	215.68[1]	1.6 (b)[8]	2.61[9]	33[9]	10
	Diazinon	304.35[1]	2.6[8]	3.30[8]	40[10]	0.15
Metals (inorganic)	Cadmium	112.41[1]			Negligible	0.0015
	Chromium	151.99			Negligible	0.0015

[1]PubChem (2025), [2]Torniainen et al. (1996), [3]Takács-Novák et al. (1992), [4]McEvoy (2001), [5]Perrin (1965), [6]Hansch and Leo (1995), [7]Yalkowsky and Dannenfelser (1992), [8]MacBean (2008–2010), [9]Yalkowsky et al. (2010), [10]Sharom et al. (1980).

ready for freeze drying. The remaining dried *L. minor* was weighed (200–300 mg) and transferred to 15 mL falcon tube prior to extraction; internal standards were added (100 ng/L) followed by 6 mL of 0.1 M Na^2 EDTA McIlvaine buffer (pH 3). Samples were then vortexed for 20 s, sonicated for 10 min (23 °C), shaken at 200 rpm for 30 min, and then centrifuged at 3,500 rpm for 20 min. The supernatant was collected and stored at 3 °C until the extraction procedure was complete. The *L. minor* pellet was then re extracted using acetonitrile, and the combined supernatant was diluted with deionized water to achieve a final solvent concentration of 5%. OASIS HLB cartridges (500 mg) were conditioned using 6 mL of methanol followed by 6 mL of water. The entire extracted sample was then loaded at a rate of 5 mL/min, washed with 10 mL of deionized water and eluted with 3 mL of methanol. After taking the sample to complete dryness, it was reconstituted in 0.5 mL 10% methanol ready for analyses. Samples were stored at –20 °C before analysis.

The extraction method for organics was validated to 10 ng/g, 50 ng/g, and 100 ng/g (dw). To evaluate the extractive efficiency of the method parent compounds were spiked and left for 1 h prior to extraction. *L. minor* was extracted as described above, following the addition of 100 ng/L of the internal standard mixture. Recoveries were found to be consistent across the assessed concentrations and ranged from 10.21 to 66.79% (Supplementary Table 2). The low recovery of diazinon was expected as diazinon hydrophobic, nonpolar with a log K_{OW} of 3.3, it also has a low solubility in water however the use of internal standards enabled a correction to be applied which accounted for this.

2.3.2 Inorganics

For the metal analysis, 50 mg of *L. minor* was removed from each treatment beaker and freeze dried. The dried *L. minor* was then transferred to a microwave furnace, 5 mL of concentrated nitric acid was added. Samples underwent microwave assisted digestion, the system settings were as follows, 10-min ramp, 10-min hold, and 30-min cool. Sample extracts were evaporated to dryness and then diluted in 10 mL of deionized water. For both the extract and media 10 mL was then filtered through a 0.45 µm nylon filter and stored at 3 °C prior to analyses. Metal recovery from digestion was evaluated in plant matter (tomato and poplar leaves), the assessed concentration was 1 ppm, and blank matrix was used for accurate quantification.

2.4 Analyses

A Thermo Vanquish coupled with a Quantiva was utilised for the quantification of the organic residues in the extracted samples. A reversed phase method was utilized that was comprised of a HST3 acuity column (Waters C18 100 Å, 1.8 µm, 1.5 mm × 100 mm), the mobile phases consisted of 0.1% formic acid in ultra-pure deionized water and 0.1% formic acid in methanol. A 25 µL injection volume was selected and the column temperature was set to 30 °C and mobile phase B was applied in the following percentages 0–80% (0.5–8 min), 80–99% (8.5–9.5 min), 99% (9.5–11.5 min), 99–1% (11.5–12 min), and 1% (12–15 min). Mass spectrometer conditions were set as the following, ion transfer tube 350 °C, ionisation spray 5000 V, sheath gas 50.7, Aux gas 18, sweep gas 23, vaporizer temperature 370 °C. Please see Supplementary Table 3 for the further details [chemical specifics and Limits of Detection (LOD)]. Thermo Xcalibur (V 4.3) was used to process chromatographic peaks and for quantification. Metals (cadmium and chromium) were analysed using an ICP-OES (Thermo Scientific ICAP 7600 ICP-OES) following the manufacturers recommended configuration and parameters. In brief these were, 1.6 kW of plasma, gas flow 0.6 L/min, auxiliary gas 0.5 L/min, carrier gas flow 0.6 L/min. Quantification was achieved using the Certified Reference Material LGC6027, the wavelengths for the analysis were 228.802 nm and 357.869 nm for cadmium and chromium, respectively.

2.5 Statistical analysis

Minitab 18 V.21 was used for the statistical analyses. The Shapiro–Wilk test was utilized to test for normality within the datasets and Kruskal–Wallis followed by Dunns for pair-wise comparisons between the treatments of the chemicals. Statistically significant differences between the treatments were reported when $p \leq 0.05$.

3 Results and discussion

3.1 Singular uptake organics and inorganics

The accumulation of antibiotics, PPPs and metals on a single compound basis highlighted compound specific in the uptake capacity by *L. minor* between the contaminants assessed in this study. Diazinon was taken up to the greatest extent by *L. minor* with an average removal of 95.51 ± 2.13%, followed by ciprofloxacin (31.91 ± 6.06%), atrazine (13.04 ± 3.44%), trimethoprim (11.43 ± 5.77%), chromium (4.5 ± 2.74%) and, cadmium (3.08 ± 1.3%; see Figure 1).

Notably, the percent uptake of cadmium and chromium was lower than values previously reported in the literature. For example, Aslanzadeh et al. (2024) and Megateli et al. (2009) reported uptake of chromium and cadmium by *Lemna* species between 90% and 14–26% (24 h), respectively. Chromium uptake was found to be comparable, whereas cadmium removal was considerably lower than observed in our study. Differences in uptake are likely attributed towards the differences in the choice of *Lemna* species (*L. minor vs L. gibba)* and dosing concentration as exposure concentrations (0.001–0.1 mg/L and 1–100 mg/L) were between 20 and 4.54–8.56 times higher than used in our experimental set up. Nevertheless, the pattern of greater cadmium uptake over that of chromium mirrors previous findings where Amna et al. (2015) also observed a higher uptake potential of cadmium over chromium in *Linum usitatissimum* (Flax).

For the organic contaminants, atrazine and ciprofloxacin uptake by *L. minor* was consistent with previous studies. For example, removal percentages of 24–63 and 26.55% were reported at 5 µg/L and 1.5 µg/L (Brain et al., 2012), and 0.15–5.07% at 0.01–10 mg/L for atrazine ciprofloxacin, respectively (Kochi et al., 2023). The uptake of trimethoprim was 5.34 times lower than reported by Latrou et al. (2017) over a 24-time period. The uptake of diazinon by *Lemna* species has not been previously reported in the literature to enable a comparison to be made to findings observed in this study but similarly high levels of uptake have been observed in algae, with percent removal ranging from 48 to 64% (Khoei et al., 2018).

Inter-laboratory differences are expected to contribute to some of the variability in the measured uptake capacity of contaminants; the parameters likely driving this are linked to differences in experimental design (i.e., water properties, exposure concentration, species, and duration). This highlights the importance of considering exposure

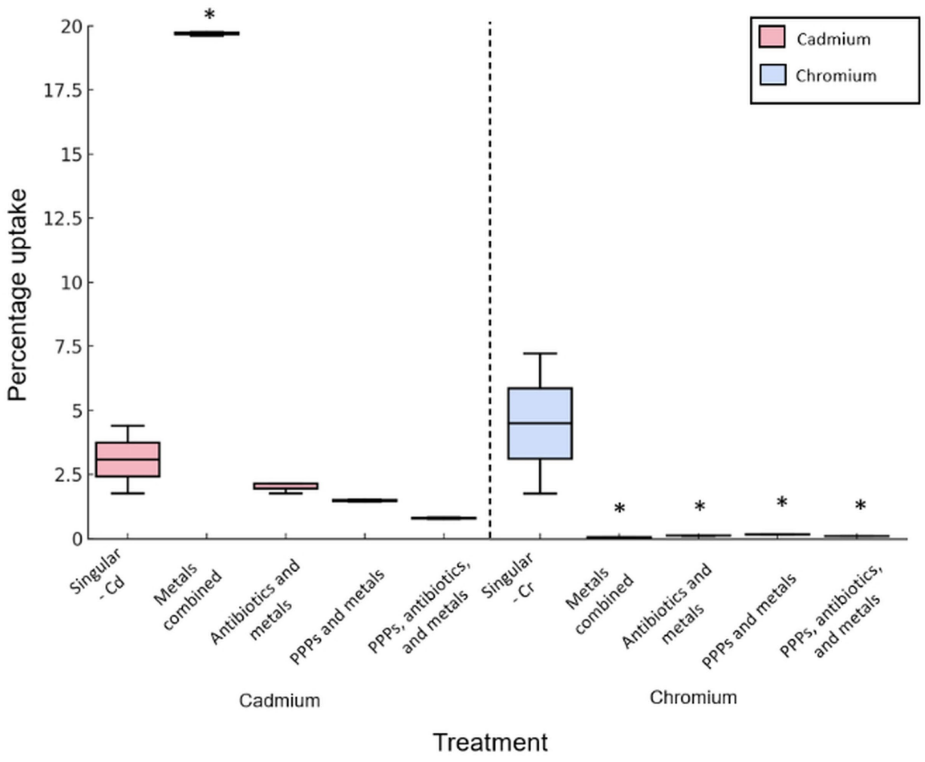


FIGURE 1
Percent uptake of metals into *L. minor* under differing treatments. * Denotes a statistically significant ($p < 0.05$) difference between a singular exposure and the co-contaminant mixtures. The box and whisker boxes represent the min–max, 2,575% quartiles, and the median.

concentration and experimental design when evaluating the uptake capacity of aquatic macrophytes across different studies as these factors appear to strongly influence accumulative potential. Nevertheless, our results show that *L. minor* can take up and accumulate a suite of chemically diverse contaminants which supports the use of aquatic macrophytes as a potential remediation measure which warrants further evaluation into how the presence of co-contaminants can influence this uptake capacity. Furthermore, our findings underscore the importance of understanding chemical specific uptake potential as significant differences were observed in the uptake capacity between chemicals belonging to the same chemical class (e.g., diazinon 95.51% and atrazine 13.04%).

3.2 Co-contaminant exposure combinations

3.2.1 Metals uptake

Co-contaminant exposure significantly influenced the uptake capacity of *L. minor* within just 72 h ($p < 0.05$) (Figures 1; Supplementary Table 4). For chromium, co-exposure with another metal or organic contaminant was found to statistically inhibit uptake (Figure 2) ($p \leq 0.0004$–0.035) (Supplementary Table 4). For example, the removal capacity of chromium by *L. minor* was reduced by, 83.0, 25.0, 18.5, and 27.7%, respectively, when chromium was in combination with antibiotics, PPPs or cadmium (Figure 1). This finding was similar to the pattern observed for cadmium although to a lesser extent as reductions in percent uptake of cadmium in the antibiotics—metals, and PPPs—metals co-contaminant combinations were not statistically significant ($p = 0.08$–0.36), however PPPs, antibiotics, and metals were found to be statistically significant ($p = 0.012$). In these scenarios, cadmium uptake was reduced by 9.7 and 1.4% when present in combination with antibiotics and metals and PPPs, respectively (Figure 1). Interestingly, cadmium uptake was however significantly inhibited ($p = 0.012$) in the all-contaminants scenario (antibiotics, PPPs, and metals), with uptake reduced by 1.9%, similarly to chromium uptake which was reduced by a factor of 27.7 when chromium was present in a combined metal-PPP-antibiotic exposure (Figure 1).

Interestingly for cadmium, co-exposure with chromium increased the uptake of cadmium by *L. minor* by a factor of 2.8 (6.3 < 17.5%) (Figure 1). An increase in uptake when in the presence of chromium is likely explained via the enhanced activity of antioxidant enzymes (superoxide dismutase and catalase) which are known to mitigate the oxidative stress induced by chromium, thus altering the transporter activity and cadmium uptake (Gill and Tuteja, 2010; Shanker et al., 2004). A similar effect was observed via Bah et al. (2011), where cadmium uptake in *Typha angustifolia* was increased by 26–38% when existing in synergy with chromium (Figure 2).

3.2.2 Organic contaminant uptake

Similarly to the results described above for the metals, co-exposure of atrazine, diazinon, and ciprofloxacin with metals or other organics significantly reduced the uptake capacity of the organic contaminants themselves ($p < 0.05$); Supplementary Table 4). This finding demonstrates that metals co-existing with organic contaminants (antibiotics and PPPs) results in reduced uptake of ciprofloxacin diazinon, and atrazine ($p \leq 0.05$) (Supplementary Table 4). For example, when organic chemicals existed singularly but also with metals uptake was reduced by a factor of 31.9, 72.7 and 13 for ciprofloxacin, diazinon, and atrazine, respectively. Antagonistic effects were evident in all of the mixture treatments for these compounds. For diazinon the

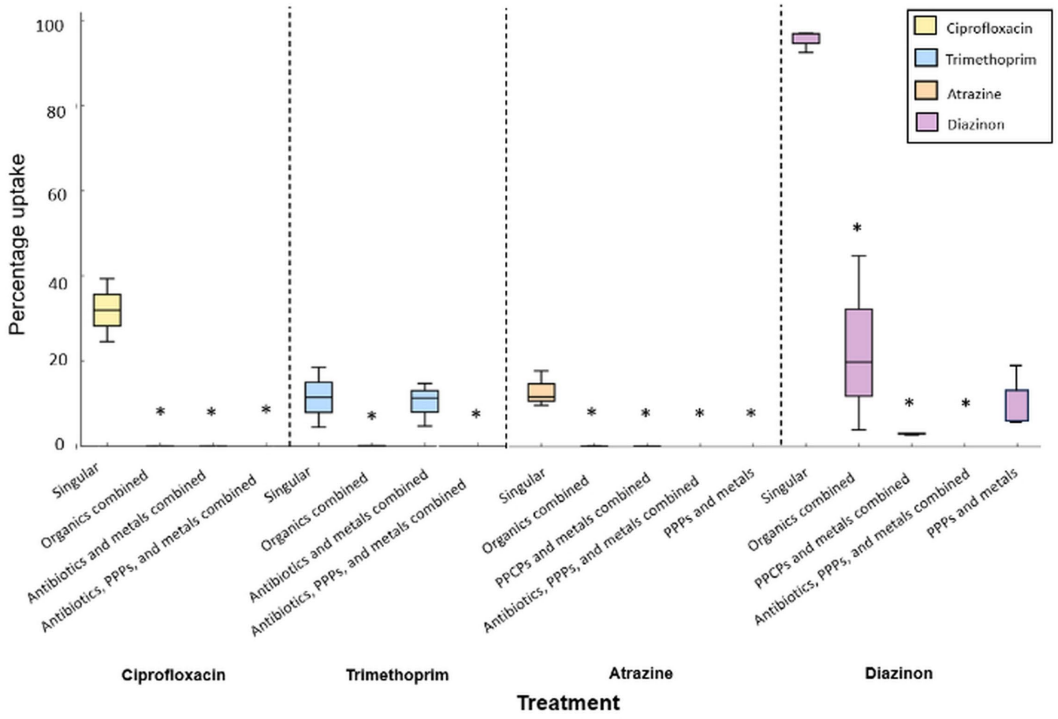


FIGURE 2
Percent uptake of organics into *L. minor* under differing treatments. * Denotes a statistically significant ($p < 0.05$) difference between a singular exposure and the co-contaminant mixtures. The box and whisker boxes represent the min–max, 25–75% quartiles, and the median.

combination which resulted in the greatest reduction in uptake was identified when diazinon was present in a mixture with antibiotics and metals (3% < 95.5%). For ciprofloxacin, all co-contaminant mixtures significantly reduced ciprofloxacin uptake by factor of 46 ($p = 0.003$) which was similar to the results from the atrazine exposure.

Ciprofloxacin is associated with a moderate acid pK_a and a moderate basic pK_a, (Table 1) limiting its capacity to pass the root cytosol (Trapp et al., 2023); these properties are likely why ciprofloxacin has been shown to accumulate in the roots of other macrophytes such as *E. crassipes* (Yan et al., 2021). It is likely the high uptake of ciprofloxacin observed in our study is associated with accumulation in the roots, but this cannot be confirmed as our analysis did not separate the *L. minor* root from the shoot. In recently published research Marques et al. (2024) evaluated the capability of *L. gibba* to remove ciprofloxacin (60 ng/L) in treated wastewater containing a mixture of contaminants. After 7 days, the authors reported a removal rate below that of this study (24 ± 8.1% < ~74.09%) which could be explained by differences in *Lemna* spp. used between the two studies.

In comparison to ciprofloxacin and atrazine, trimethoprim uptake was not inhibited in the antibiotics and metals scenario (0.9 μg/g (dw) v 1.2 μg/g (dw)) (antibiotics—metals); average trimethoprim concentrations in a singular exposure was 1.2 μg/g (dw). When trimethoprim existed in a co-contaminant scenario with antibiotics, and PPCPs this was reduced to 0.002 μg/g (dw), 0.4 μg/g (dw) in the antibiotics – metals (Supplementary Table 4). One potential explanation of this is that the metabolism of trimethoprim was increased when exposed with other organic chemicals which resulted in changes to stress-induced metabolic processes (Li et al., 2020). When trimethoprim was present with the other organics and metals the difference in uptake was not significant indicating that the presence of metals altered uptake mechanisms or did not trigger stress-induced metabolism. Regardless this finding reiterates the notion that uptake mechanisms are complex when considering numerous contaminants from differing classes.

In previous studies, synergistic effects have been reported where greater accumulation of formulated dimethomorph in *L. minor* was observed when exposed to copper simultaneously (Megateli et al., 2013). The authors suggested this could be related towards an altered cuticle structure or subsequent disruptions to plant defences which may have contributed to enhanced dimethomorph uptake. A similar phenomenon may have occurred in our study for trimethoprim, although further mechanistic studies are needed to confirm this.

4 Environmental relevance

Globally, our aquatic environments are increasingly contaminated with a diverse array of pollutants, including PPPs (i.e., herbicide and pesticide) used in agriculture (Silva et al., 2019), antibiotics stemming from human use and livestock operations (Kim et al., 2011; Almeida et al., 2014), and heavy metals released from industrial activities, mining, and urban runoff (Mishra et al., 2019). Individually, these contaminants pose significant risks to biodiversity, water quality, and human health. However, in the environment they are typically present as co-contaminant mixtures which can lead to altered responses, including the potential for synergistic effects (You et al., 2022). It is therefore important that environmental risk assessments and associated mitigation strategies such as phytoremediation account for this.

Although our experiment employed a short-term (72 h) laboratory exposure, this approach provides a controlled baseline to isolate co-contaminant interactions that are difficult to quantify in natural waters, where contaminant profiles fluctuate with hydrology, seasonality, microbial degradation, and plant growth dynamics. Numerous

field-based studies demonstrate that *Lemna* spp. can effectively reduce certain contaminants in natural or semi-natural systems; however, efficiency in real water bodies is highly variable and depends on factors such as water turnover, nutrient load, competition with other macrophytes, and environmental stressors. Our findings therefore offer mechanistic insight into the directionality and magnitude of co-contaminant effects, which is critical for interpreting, calibrating, and improving phytoremediation performance in natural settings.

4.1 Ecotoxicological implications

Our results clearly demonstrate that phytoremediation is a viable method for removing organic and inorganic contaminants from polluted water. However, the uptake capacity is significantly influenced by the co-exposure of multiple contaminants in the aqueous media. As a compound-specific effect was observed, and the specific combination of contaminants (i.e., metal-PPP vs. metal-PPP-antibiotic) strongly influenced the uptake capacity of *L. minor* this emphasises the need for tailored mixture assessments when evaluating phytoremediation potential. As co-contaminant exposure typically resulted in a reduced uptake capacity, failure to account for co-contaminant interactions may lead to significantly overestimated removal rates, ultimately compromising the success of planned remediation measures. For example, if the calculated removal rate for diazinon in a single compound exposure (95%) was applied to published surface water concentrations for diazinon (0.01–24.5 mg/L; Table 2), concentrations would be reduced to 0.005–1.2 mg/L after 72 h. However, as the uptake of diazinon by *L. minor* was reduced in the presence of atrazine, and metals, surface water concentrations calculated using removal rates from the co-contaminant scenario would only be reduced to 0.008–12.3 mg/L. Similar findings have also been identified for the antibiotic compounds ciprofloxacin and trimethoprim which confirms that under realistic exposure scenarios contaminant removal via phytoremediation is reduced (Table 3).

A reduced capacity for uptake by *L. minor* raises important concerns with respect to the true extent of environmental exposure and the efficiency of phytoremediation strategies. For instance, environmental residues of diazinon are well-documented for their harmful effects on non-target aquatic organisms (Table 2), prompting significant efforts to mitigate and remove diazinon from contaminated environments (Dehghani et al., 2019). Pham and Bui (2018), previously reported a *Daphnia magna* EC_{50} value of 2 µg/L for diazinon. Considering a surface concentration of 24.5 µg/L (Salinas—California, Zhang et al., 2012), removal via phytoremediation using *L. minor* in a single-exposure scenario could reduce mortality risk to *D. magna* (RQ 0.4). It is clear that reduced contaminant uptake has important implications for environmental risk assessments which warrant further exploration. However, it is important to note that the LC_{50} values used in this analysis (Anderson and Lydy, 2002; Pham and Bui, 2018; Husak et al., 2022) were calculated following exposure to a single compound. Therefore, a comprehensive evaluation of the impacts on reduced phytoremediation capacity would utilise an effect measurement also calculated under similar exposure conditions (i.e., co-contaminant mixtures).

4.2 Implications for the development of antibiotic resistance

It is not known to what extent environmental pollution from antibiotics contributes to the development of resistance, but encountered concentrations are frequently above concentrations predicted to select for resistance. Hence, measures are needed to manage risks (Ågerstrand et al., 2023) and phytoremediation has been discussed as one such potential to remove environmental divers of AMR including antibiotics, PPPs, metals (Chowdhury et al., 2023). The potential environmental implications of reduced phytoremediation efficacy are also therefore significant with respect to the potential implications for the development of antibiotic resistance. For example, if the protection goal is to reduce the levels

TABLE 2 Evaluation of co-contaminant impacts on phytoremediation capacity for atrazine and diazinon ecotoxicological risk evaluation.

Chemical	Treatment	Exposure concentration (µg/L)	Percent removal (*L. minor* (%)	LC_{50} in *Daphnia magna* (mg/L)	Predicted concentration following removal via *L. minor* (µg/L)	Risk quotient (RQ)
Atrazine	Singular	0.5^{A}, 2^{B}, 5^{B}, 15^{C}	13.04	0.022[1]	0.4^{A}, 2.0^{B}, 4.0^{C}, 10^{D}	0.02^{A}, 0.08^{B}, 0.2^{C}, 0.6^{D}
	PPPs + Antibiotics		0.04		0.5^{A}, 2.0^{B}, 5.0^{C}, 15^{D}	0.02^{A}, 0.09^{B}, 0.2^{C}, 0.7^{D}
	PPPs + Metals		0.053		0.05^{A}, 2^{B}, 5^{C}, 14^{D}	0.02^{A}, 0.09^{B}, 0.2^{C}, 0.7^{D}
	PPPs + Antibiotics + Metals		0.063		0.05^{A}, 2^{B}, 5^{C}, 10^{D}	0.02^{A}, 0.09^{B}, 0.2^{C}, 0.7^{D}
	No phytoremediation (*L. minor*)		0		0.05^{A}, 2^{B}, 5^{C}, 15^{D}	0.02^{A}, 0.09^{B}, 0.2^{C}, 0.7^{D}
Diazinon	Singular	10^{A}, 15^{B}, 1.90^{C}, $24{,}470^{D}$	95	0.002[2]	0.5^{A}, 0.75^{B}, 80^{C}, $1{,}200^{D}$	0.3^{A}, 0.4^{B}, $\mathbf{41.4}^{C}$, $\mathbf{679.7}^{D}$
	PPPs + Antibiotics		22.82		8^{A}, 10^{B}, $1{,}150^{C}$, $18{,}890^{D}$	$\mathbf{4.2}^{A}$, $\mathbf{6.4}^{B}$, $\mathbf{638.9}^{C}$, $\mathbf{10492.2}^{D}$
	PPPs + Metals		5.77		9^{A}, 14^{B}, $1{,}400^{C}$, $23{,}100^{D}$	$\mathbf{5.2}^{A}$, $\mathbf{7.9}^{B}$, $\mathbf{780}^{C}$, $\mathbf{12810.1}^{D}$
	PPPs + Antibiotics + Metals		29.22		7^{A}, 10^{B} $1{,}050^{C}$, $17{,}320^{D}$	$\mathbf{3.9}^{A}$, $\mathbf{5.9}^{B}$, $\mathbf{585.9}^{C}$, $\mathbf{9622.2}^{D}$
	No phytoremediation (*L. minor*)		0		10^{A}, 15^{B}, $1{,}490^{C}$, $24{,}470^{D}$	$\mathbf{6}^{A}$, $\mathbf{8.3}^{B}$, $\mathbf{827.8}^{C}$, $\mathbf{13594.4}^{D}$

[A][Acayaba et al. (2021) (atrazine), Dahmardeh Behrooz et al. (2021) (diazinon)], [B][Zhang et al. (2012) (atrazine), (study dose (diazinon)], [C][Della-Flora et al. (2018) (atrazine)], and [D][study dose (atrazine), Zhang et al. (2012) (diazinon)], assign the assessed MEC to the range of concentrations reported (Umweltbundesamt, 2022). LC_{50} data for *Daphnia magna* (24 h) (Pham and Bui, 2018; Husak et al., 2022), risk quotients were calculated by comparing the predicted concentration following *L. minor* concentrations to PNEC values. A bold unit demonstrates a risk was quantified (RQ<1).

of known ABR divers in the environment and the capacity of *L. minor* to accumulate contaminants is reduced in a co-contaminant exposure, this may result in an altered assessment of risk. Based on the findings presented in this study, a 72 h exposure to singular antibiotics (ciprofloxacin) at 0.077 mg/L which was identified as a 'low' environmentally relevant concentration (see Table 2) reduced measured environmental concentrations of the selected antibiotics in the media to concentrations lower than the predicted no effect concentration for resistance; PNEC-R (AMR Alliance, 2016) (Table 3). However, for ciprofloxacin, phytoremediation was not enough to reduce the exposure concentration (0.21 mg/L) below the PNEC-R when present in the all-contaminant combination (metals-antibiotics-PPPs) (Table 3).

Whilst it is evident that not employing phytoremediation would be considered a worst-case scenario, as contaminants are not actively removed from the exposure media, it is important to consider the implications of co-contaminant exposure on phytoremediation efficiency to enable an accurate assessment of risk. Furthermore, it is crucial to evaluate not only the efficiency of aquatic macrophytes in removing antibiotics but also other known drivers of AMR, such as metals and pesticides. Even partial removal of antibiotics, without considering the presence of metals like cadmium and chromium in the environment, could still promote the proliferation of resistance genes through enhanced co-selection and cross selection via horizontal gene transfer at lower concentrations (Peltier et al., 2010). This effect is likely to be exacerbated in the presence of pesticides, which are known to induce microbial stress.

5 Future considerations

Mixtures are inherently complex to manage and regulate; multiple potential combinations of chemicals result in thousands of potential exposure scenarios. Herein, we assessed the potential removal capacity of six chemicals, two antibiotics, two metals and two pesticides by *L. minor* to chemicals singularly and in co-contaminant combinations. While this study represents how environmental complexity can alter phytoremediation potential it is still not comprehensive enough to facilitate generalised conclusions across multiple contaminant classes. However, given the widespread occurrence of both traditional and emerging contaminants including organic and inorganic contaminants assessed in this study, it not feasible to experimentally determine the removal rates for every potential chemical combination. Therefore, we recommend that future evaluations of removal efficiencies should be conducted with a clear remediation goal in mind and be based on effect measurements (e.g., LC_{50}) which are also determined as part of a mixture exposure (Anderson and Lydy, 2002). For example, for antimicrobial resistance, attention should be given to the co-exposure of resistance drivers; for ecotoxicological effects, the presence of other toxic chemicals (e.g., those observed to effect key ecological indicators such as oxidative stress, or carcinogenicity) should be considered; and for bioaccumulation, focus should be on chemicals with a known high bioaccumulation capacity. Nevertheless, our findings support the need to standardise chemical mixture assessment (Kienzler et al., 2016), these could include prioritising chemicals from differing groups, or chemicals which contain an array of physiochemical properties and mode of actions. Such a selection would require careful planning but could be used in regulatory and research settings to provide a worst-case understanding of effects in mixtures, as the continuation of singular assessments is clearly not representative of environmental processes.

Further refinement of the experimental set up employed in this study is also required to increase the environmental relevance of these findings. For example, phytoremediation assessments in the laboratory are typically conducted in short timeframes. The life cycle of

TABLE 3 Evaluation of co-contaminant impacts on phytoremediation capacity for ciprofloxacin and trimethoprim ecotoxicological risk evaluation.

Chemical	Treatment	Exposure concentration (µg/L)	Percent removal (*L. minor* (%)	PNEC (µg/mL) antimicrobial resistance[1]	Predicted concentration following removal via *L. minor* (µg/L)
Ciprofloxacin	Singular	$2.5 \times 10^{-9(A)}$, $7.7 \times 10^{-2(B)}$, 1^{C}, $1{,}100^{D}$	31.91	0.06[1]	$1.7 \times 10^{-9(A)}$, 0.05^{B}, **0.98^{C}**, **750^{D}**
	Antibiotics + Metals		0.01		$2.5 \times 10^{-9(A)}$, **0.07^{B}**, **1^{C}**, **$1{,}100^{D}$**
	PPPs + Antibiotics		0.001		$2.5 \times 10^{-9(A)}$, **0.07^{C}**, **1^{B}**, **$1{,}100^{D}$**
	PPPs + Antibiotics + Metals		0.69		$2.5 \times 10^{-9(A)}$, **0.00007^{B}**, **1^{C}**, **$1{,}100^{D}$**
	No phytoremediation (*L. minor*)		0		$2.5 \times 10^{-9(A)}$, **0.00007^{B}**, **1^{C}**, **$1{,}100^{D}$**
Trimethoprim	Singular		11.43	0.5[1]	$1.8 \times 10^{-3(A)}$, $7.1 \times 10^{-2(B)}$, **1^{C}**, **$2{,}800^{D}$**
	PPPs + Antibiotics	$2 \times 10^{-3(A)}$, $8 \times 10^{-2(B)}$, 1.3^{C}, $3{,}200^{D}$	0.02		$2 \times 10^{-3(A)}$, $8 \times 10^{-2(B)}$, **1.3^{C}**, **$3{,}200^{D}$**
	Antibiotics + Metals		3.43		$1.9 \times 10^{-3(A)}$, $7.7 \times 10^{-2(B)}$ **0.12^{C}**, **$3{,}100^{D}$**
	Antibiotics + Metals PPPs		9.23		$1.93 \times 10^{-3(A)}$, $7.3 \times 10^{-2(B)}$, **1.8^{C}**, **$2{,}900^{D}$**
	No phytoremediation (*L. minor*)		0		$2 \times 10^{-3(A)}$, 0.08^{B}, **1.3^{C}**, **$3{,}200^{D}$**

[A][Gao et al. (2012) (Ciprofloxacin)], [B][Loos et al. (2012) (ciprofloxacin), (trimethoprim), Grabic et al. (2012) (trimethoprim)], [C][Biel-Maeso et al. (2018) (ciprofloxacin), Rodríguez-Gil et al. (2010) (trimethoprim)] and [D][study dose (ciprofloxacin), study dose (trimethoprim)], assign the assessed MEC to the range of concentrations (minimum, median, maximum, study dose) (Umweltbundesamt, 2022). PNEC values were obtained from the AMR Industry Alliance (2023)[1] (Tell et al., 2019; Giebułtowicz et al., 2020), risk quotients were calculated by comparing the predicted concentration following *L. minor* concentrations to PNEC values. A bold unit demonstrates $PEC_{SURFACE\ WATER} > PNEC_{R}$.

L. minor is around 7–10 days whereas our assessment was only 72 h. Therefore, key environmental processes which may occur following longer exposure such as degradation and in-plant metabolism may be overlooked. Furthermore, additional phytoremediation studies are needed to capture the natural variability of environmental systems by utilizing a range of environmental conditions such as temperature, water properties are well known to influence the speciation of contaminants and thus their fate in the environment. For example, the European Chemicals Agency (2017) guidance document for the risk assessment of pharmaceuticals underpins the requirement and procedure for the detailed risk assessment of a pharmaceutical or bioactive compound. Specifically, European Chemicals Agency (2022) details laboratory procedures and the requirement for tests to be performed in a range of water properties (such as pelagic pH) as these may influence the speciation of the chemical itself. As highlighted in Chowdhury et al. (2023) the use multiple macrophyte species (submerged, floating, and emergent plants) to remove contaminants from divergent chemical classes offers most promise in terms of maximising pollutant removal and this was not assessed in this current study (Chowdhury et al., 2023). Nevertheless, future work can build on the experimental platform created here to scale the number of compounds and aquatic macrophytes considered in a co-contaminant exposure to increase the representativeness of the results to better inform remediation efforts.

6 Conclusion

This study demonstrates the potential of *L. minor* as an effective phytoremediation tool for various organic and inorganic contaminants, such as diazinon > trimethoprim > atrazine > ciprofloxacin > cadmium >chromium. Our results show that the uptake of these contaminants differ significantly between contaminants, and the co-exposure of multiple contaminants can significantly alter removal efficiencies. Specifically, synergistic effects were observed in some cases, such as the increased uptake of cadmium in the presence of chromium, or reduced uptake of organic contaminants such as atrazine, diazinon, and ciprofloxacin in co-contaminant mixtures. Our findings highlight the need to account for these complex interactions between pollutants to avoid overestimating phytoremediation success. Furthermore, whilst *L. minor* proved effective in reducing concentrations of certain contaminants below antimicrobial predicted no-effect concentrations (PNECs) or ecotoxicological end points for *D. magna*, further refinement of phytoremediation practices is required to optimise their application in real-world scenarios. The findings emphasise that while phytoremediation holds promise, it requires careful consideration of environmental conditions, co-occurring contaminants, and appropriate study designs to ensure its practical application in mitigating water pollution and the associated risks.

Data availability statement

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

Author contributions

KC: Writing – original draft, Investigation, Data curation, Methodology, Formal analysis, Conceptualization. JN: Visualization, Methodology, Investigation, Formal analysis, Writing – original draft, Data curation. LC: Funding acquisition, Writing – review & editing, Conceptualization.

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Conflict of interest

The author(s) declared that this work was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Supplementary material

The Supplementary material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/frwa.2026.1812220/full#supplementary-material

References

Aemig, Q., Hélias, A., and Patureau, D. (2021). Impact assessment of a large panel of organic and inorganic micropollutants released by wastewater treatment plants at the scale of France. *Water Res.* 188:116524. doi: 10.1016/j.watres.2020.116524

Acayaba, R. D. A., de Albuquerque, A. F., Ribessi, R. L., Umbuzeiro, G. D. A., and Montagner, C. C. (2021). Occurrence of pesticides in waters from the largest sugar cane plantation region in the world. *Environ. Sci. Pollut. Res.* 28, 9824–9835. doi: 10.1007/s11356-020-11428-1

Ågerstrand, M., Josefsson, H., Wernersson, A. S., and Larsson, D. J. (2023). Opportunities to tackle antibiotic resistance development in the aquatic environment through the water framework directive. *Ambio* 52, 941–951. doi: 10.1007/s13280-022-01828-7

Ahmed, M. B., Zhou, J. L., Ngo, H. H., Guo, W., Thomaidis, N. S., and Xu, J. (2017). Progress in the biological and chemical treatment technologies for emerging contaminant removal from wastewater: a critical review. *J. Hazard. Mater.* 323, 274–298. doi: 10.1016/j.jhazmat.2016.04.045

Almeida, A., Duarte, S., Nunes, R., Rocha, H., Pena, A., and Meisel, L. (2014). Human and veterinary antibiotics used in Portugal—a ranking for ecosurveillance. *Toxics* 2, 188–225. doi: 10.3390/toxics2020188

Amna, Ali, N., Masood, S., Mukhtar, T., Kamran, M. A., Rafique, M., et al. (2015). Differential effects of cadmium and chromium on growth, photosynthetic activity, and metal uptake of *Linum usitatissimum* in association with *Glomus intraradices*. *Environ. Monit. Assess.* 187, 1–11. doi: 10.1007/s10661-015-4557-8

AMR Industry Alliance. (2023). AMR industry Alliance progress report 2023. Available online at: https://www.amrindustryalliance.org/wp-content/uploads/2023/02/AMR-Table-1-Update-20230222_corrected.pdf (Accessed February 13, 2026).

Anderson, T. D., and Lydy, M. J. (2002). Increased toxicity to invertebrates associated with a mixture of atrazine and organophosphate insecticides. *Environ. Toxicol. Chem.* 21, 1507–1514. doi: 10.1002/etc.5620210724

Andersson, D. I., and Hughes, D. (2014). Microbiological effects of sublethal levels of antibiotics. *Nat. Rev. Microbiol.* 12, 465–478. doi: 10.1038/nrmicro3270

Angeles, L. F., Mullen, R. A., Huang, I. J., Wilson, C., Khunjar, W., Sirotkin, H. I., et al. (2020). Assessing pharmaceutical removal and reduction in toxicity provided by advanced wastewater treatment systems. *Environ. Sci. Water Res. Technol.* 6, 62–77. doi: 10.1039/c9ew00559e

Aslanzadeh, M., Saboora, A., and Moradlou, O. (2024). Phytoremediation potential of duckweed (*Lemna minor* L.) for hexavalent chromium removal in synthetic wastewater: unveiling physiological response and defense mechanisms against excessive heavy metal uptake. *Int. J. Environ. Sci. Technol.* 21, 10155–10174. doi: 10.1007/s13762-024-05721-6

Bah, A. M., Dai, H., Zhao, J., Sun, H., Cao, F., Zhang, G., et al. (2011). Effects of cadmium, chromium and lead on growth, metal uptake and antioxidative capacity in *Typha angustifolia*. *Biol. Trace Elem. Res.* 142, 77–92. doi: 10.1007/s12011-010-8746-6

Biel-Maeso, M., Corada-Fernández, C., and Lara-Martín, P. A. (2018). Monitoring the occurrence of pharmaceuticals in soils irrigated with reclaimed wastewater. *Environ. Pollut.* 235, 312–321. doi: 10.1016/j.envpol.2017.12.085

Brain, R. A., Hosmer, A. J., Desjardins, D., Kendall, T. Z., Krueger, H. O., and Wall, S. B. (2012). Recovery of duckweed from time-varying exposure to atrazine. *Environ. Toxicol. Chem.* 31, 1121–1128. doi: 10.1002/etc.1806

Chowdhury, K. F., Hall, R. J., McNally, A., and Carter, L. J. (2023). Phytoremediation as a tool to remove drivers of antimicrobial resistance in the aquatic environment. *Rev. Environ. Contam. Toxicol.* 261:16. doi: 10.1007/s44169-023-00039-9

Convention on Biological Diversity (2022). The Post-2020 Global Biodiversity Framework: Goals and Targets for 2030. Available online at: https://www.cbd.int/conferences/post2020 (Accessed October 21, 2024).

Dahmardeh Behrooz, R., Esmaili-Sari, A., Urbaniak, M., and Chakraborty, P. (2021). Assessing diazinon pollution in the three major rivers flowing into the Caspian Sea (Iran). *Water* 13:335. doi: 10.3390/w13030335

Dehghani, M. H., Kamalian, S., Shayeghi, M., Yousefi, M., Heidarinejad, Z., Agarwal, S., et al. (2019). High-performance removal of diazinon pesticide from water using multi-walled carbon nanotubes. *Microchem. J.* 145, 486–491. doi: 10.1016/j.microc.2018.10.053

Della-Flora, A., Becker, R. W., Ferrão, M. F., Toci, A. T., Cordeiro, G. A., Boroski, M., et al. (2018). Fast, cheap and easy routine quantification method for atrazine and its transformation products in water matrixes using a DLLME-GC/MS method. *Anal. Methods* 10, 5447–5452. doi: 10.1039/c8ay02227e

Dosnon-Olette, R., Couderchet, M., Oturan, M. A., Oturan, N., and Eullaffroy, P. (2011). Potential use of *Lemna minor* for the phytoremediation of isoproturon and glyphosate. *Int. J. Phytoremediation* 13, 601–612. doi: 10.1080/15226514.2010.525549

European Chemicals Agency. (2017). Biocidal Products Regulation (BPR) Guidance - Volume IV: Part B+C. Available online at: https://echa.europa.eu/documents/10162/2324906/bpr_guidance_ra_vol_iv_part_b-c_en.pdf/e2622aea-0b93-493f-85a3-f9cb42be16ae?t=1513159347576 (Accessed February, 02, 2026).

European Chemicals Agency. (2022). Biocidal Products Regulation (BPR) Guidance - Volume IV: Part A. Available online at: https://echa.europa.eu/documents/10162/2324906/bpr_guidance_vol_iv_part_a_en.pdf/4a70aa9e-7491-7fc5-0734-6777ade10b02?t=1648525444087 (Accessed February 02, 2026).

Gao, L., Shi, Y., Li, W., Liu, J., and Cai, Y. (2012). Occurrence, distribution and bioaccumulation of antibiotics in the Haihe River in China. *J. Environ. Monit.* 14, 1247–1254. doi: 10.1016/j.marpolbul.2012.12.028

Garrone, P., Grilli, L., Groppi, A., and Marzano, R. (2018). Barriers and drivers in the adoption of advanced wastewater treatment technologies: a comparative analysis of Italian utilities. *J. Clean. Prod.* 171, S69–S78. doi: 10.1016/j.jclepro.2016.02.018

Giebułtowicz, J., Nałęcz-Jawecki, G., Harnisz, M., Kucharski, D., Korzeniewska, E., and Płaza, G. (2020). Environmental risk and risk of resistance selection due to antimicrobials' occurrence in two polish wastewater treatment plants and receiving surface water. *Molecules* 25:1470. doi: 10.3390/molecules25061470

Gill, S. S., and Tuteja, N. (2010). Reactive oxygen species and antioxidant machinery in abiotic stress tolerance in crop plants. *Plant Physiol. Biochem.* 48, 909–930. doi: 10.1016/j.plaphy.2010.08.016

Grabic, R., Fick, J., Lindberg, R. H., Fedorova, G., and Tysklind, M. (2012). Multi-residue method for trace level determination of pharmaceuticals in environmental samples using liquid chromatography coupled to triple quadrupole mass spectrometry. *Talanta* 100, 183–195. doi: 10.1016/j.talanta.2012.08.032

Gros, M., Petrović, M., Ginebreda, A., and Barceló, D. (2010). Removal of pharmaceuticals during wastewater treatment and environmental risk assessment using hazard indexes. *Environ. Int.* 36, 15–26. doi: 10.1016/j.envint.2009.09.002

Groh, K., Vom Berg, C., Schirmer, K., and Tlili, A. (2022). Anthropogenic chemicals as underestimated drivers of biodiversity loss: scientific and societal implications. *Environ. Sci. Technol.* 56, 707–710. doi: 10.1021/acs.est.1c08399

Hanafiah, M. M., Hasan, R. M., Nizam, N. U. M., and Aziz, N. H. A. (2021). Water quality of the Klang River, Selangor, Malaysia and heavy metal removal using phytoremediation. *Appl. Ecol. Environ. Res.* 19. doi: 10.15666/aeer/1905_36553677

Hansch, and Leo, (1995) Hansch, C., Leo, A., and Hoekman, D., 1995. Exploring QSAR: Hydrophobic, Electronic, and Steric Constants. Washington, DC: American Chemical Society, 48.

Hoagland, D. R., and Arnon, D. I. (1938). *'The Water Culture Method for Growing Plants Without Soil', California Agriculture Experimental Station Circular 347*. Berkeley, CA: University of California, College of Agriculture, Agricultural Experiment Station.

Husak, V., Strutynska, T., Burdyliuk, N., Pitukh, A., Bubalo, V., Falfushynska, H., et al. (2022). Low-toxic herbicides roundup and atrazine disturb free radical processes in Daphnia in environmentally relevant concentrations. *EXCLI J.* 21:595. doi: 10.17179/excli2022-4690

Kay, P., Hughes, S. R., Ault, J. R., Ashcroft, A. E., and Brown, L. E. (2017). Widespread, routine occurrence of pharmaceuticals in sewage effluent, combined sewer overflows and receiving waters. *Environ. Pollut.* 220, 1447–1455. doi: 10.1016/j.envpol.2016.10.087

Kienzler, A., Bopp, S. K., van der Linden, S., Berggren, E., and Worth, A. (2016). Regulatory assessment of chemical mixtures: requirements, current approaches and future perspectives. *Regul. Toxicol. Pharmacol.* 80, 321–334. doi: 10.1016/j.yrtph.2016.05.020

Kim, K. R., Owens, G., Kwon, S. I., So, K. H., Lee, D. B., and Ok, Y. S. (2011). Occurrence and environmental fate of veterinary antibiotics in the terrestrial environment. *Water Air Soil Pollut.* 214, 163–174. doi: 10.1007/s11270-010-0412-2

Kochi, L. Y., Kitamura, R. S. A., Rocha, C. S., Brito, J. C. M., Juneau, P., and Gomes, M. P. (2023). Synergistic removal of ciprofloxacin and sulfamethoxazole by *Lemna minor* and *Salvinia molesta* in mixed culture: implications for phytoremediation of antibiotic-contaminated water. *Water* 15:1899. doi: 10.3390/w15101899

Khoei, A. J., Joogh, N. J. G., Darvishi, P., and Rezaei, K. (2018). Application of physical and biological methods to remove heavy metal, arsenic and pesticides, malathion and diazinon from water. *Turk. J. Fish. Aquat. Sci.* 19, 21–28. doi: 10.4194/1303-2712-v19_1_03

Latrou, E. I., Gatidou, G., Damalas, D., Thomaidis, N. S., and Stasinakis, A. S. (2017). Fate of antimicrobials in duckweed *Lemna minor* wastewater treatment systems. *J. Hazard. Mater.* 330, 116–126. doi: 10.1016/j.jhazmat.2017.02.005

Li, Y., Lian, J., Wu, B., Zou, H., and Tan, S. K. (2020). Phytoremediation of pharmaceutical-contaminated wastewater: insights into rhizobacterial dynamics related to pollutant degradation mechanisms during plant life cycle. *Chemosphere* 253:126681. doi: 10.1016/j.chemosphere.2020.126681

Loos, R., Carvalho, R., Comero, S., António, D.C., Ghiani, M., Lettieri, T., et al. (2012). EU Wide Monitoring Survey on Waste Water Treatment Plant Effluents European Commission - Joint Research Centre (JRC), Institute for Environment and Sustainability (IES), Ispra. Available online at: http://ies.jrc.ec.europa.eu/ (Accessed February 02, 2026).

MacBean, C. (2008–2010). "Atrazine," in *The Pesticide Manual*. 15th ed. Clive Tomlin (Alton: British Crop Protection Council).

Manaia, C. M., Aga, D. S., Cytryn, E., Gaze, W. H., Graham, D. W., Guo, J., et al. (2024). The complex interplay between antibiotic resistance and pharmaceutical and personal care products in the environment. *Environ. Toxicol. Chem.* 43, 637–652. doi: 10.1002/etc.5555

Marques, R. Z., Oliveira, P. G. D., Barbato, M. L., Kitamura, R. S. A., Maranho, L. T., Brito, J. C. M., et al. (2024). Green solutions for antibiotic pollution: assessing the phytoremediation potential of aquatic macrophytes in wastewater treatment plants. *Environ. Pollut.* 357:124376. doi: 10.1016/j.envpol.2024.124376

ed. G. K. McEvoy (2001). *AHFS Drug Information 2001*. Bethesda, MD: American Society of Health-System Pharmacists, 739.

Megateli, S., Semsari, S., and Couderchet, M. (2009). Toxicity and removal of heavy metals (cadmium, copper, and zinc) by *Lemna gibba*. *Ecotoxicol. Environ. Saf.* 72, 1774–1780. doi: 10.1016/j.ecoenv.2009.05.004

Megateli, S., Dosnon-Olette, R., Trotel-Aziz, P., Geffard, A., Semsari, S., and Couderchet, M. (2013). Simultaneous effects of two fungicides (copper and dimethomorph) on their phytoremediation using *Lemna minor*. *Ecotoxicology* 22, 683–692. doi: 10.1007/s10646-013-1060-2

Mishra, S., Bharagava, R. N., More, N., Yadav, A., Zainith, S., Mani, S., et al. (2019). Heavy metal contamination: an alarming threat to environment and human health. *Environ. Biotechnol. Sustain. Future* 103–125. doi: 10.1007/978-981-10-7284-0_5

Peltier, E., Vincent, J., Finn, C., and Graham, D. W. (2010). Zinc-induced antibiotic resistance in activated sludge bioreactors. *Water Res.* 44, 3829–3836. doi: 10.1016/j.watres.2010.04.041

Pham, T. L., and Bui, H. M. (2018). Comparison of diazinon toxicity to temperate and tropical freshwater *Daphnia* species. *J. Chem.* 2018:9217815. doi: 10.1155/2018/9217815

Perrin, D. D. (1965). *Dissociation Constants of Organic Bases in Aqueous Solution*. London: Butterworths.

Posthuma, L., Zijp, M. C., De Zwart, D., de Van Meent, D., Globevnik, L., Koprivsek, M., et al. (2020). Chemical pollution imposes limitations to the ecological status of European surface waters. *Sci. Rep.* 10:14825. doi: 10.1038/s41598-020-71537-2

PubChem. (2025). PubChem Compound Summary National Center for Biotechnology Information. Available online at: https://pubchem.ncbi.nlm.nih.gov/ (Accessed February 02, 2026).

Qiu, D., Ke, M., Zhang, Q., Zhang, F., Lu, T., Sun, L., et al. (2022). Response of microbial antibiotic resistance to pesticides: an emerging health threat. *Sci. Total Environ.* 850:158057. doi: 10.1016/j.scitotenv.2022.158057

Reinhold, D., Vishwanathan, S., Park, J. J., Oh, D., and Saunders, F. M. (2010). Assessment of plant-driven removal of emerging organic pollutants by duckweed. *Chemosphere* 80, 687–692. doi: 10.1016/j.chemosphere.2010.05.045

Rizzo, L., Malato, S., Antakyali, D., Beretsou, V. G., Đolić, M. B., Gernjak, W., et al. (2019). Consolidated vs new advanced treatment methods for the removal of contaminants of emerging concern from urban wastewater. *Sci. Total Environ.* 655, 986–1008. doi: 10.1016/j.scitotenv.2018.11.265

Robinson, R. F., Mills, G. A., Gravell, A., Schumacher, M., and Fones, G. R. (2023). Occurrence of organic pollutants in the river Itchen and river test—two chalk streams in southern England, UK. *Environ. Sci. Pollut. Res.* 30, 17965–17983. doi: 10.1007/s11356-022-23476-w

Rodríguez-Gil, J. L., Catalá, M., Alonso, S. G., Maroto, R. R., Valcárcel, Y., Segura, Y., et al. (2010). Heterogeneous photo-Fenton treatment for the reduction of pharmaceutical contamination in Madrid rivers and ecotoxicological evaluation by a miniaturized fern spores bioassay. *Chemosphere* 80, 381–388. doi: 10.1016/j.chemosphere.2010.04.045

Sasmaz Kislioglu, M. (2023). Removal of ag, au, and as from acid mine water using *Lemna gibba* and *Lemna minor*—a performance analysis. *Water* 15:1293. doi: 10.3390/w15071293

Shanker, A. K., Cervantes, C., Loza-Tavera, H., and Avudainayagam, S. (2004). Chromium toxicity in plants. *Environ. Int.* 31, 739–753. doi: 10.1016/j.envint.2004.01.003

Sharom, M. S., Solomon, K. R., Higgs, T. R., and Sherratt, R. S. (1980). Persistence and metabolism of diazinon in aqueous environments. *Water Res.* 14, 1095–1100.

Silva, V., Mol, H. G., Zomer, P., Tienstra, M., Ritsema, C. J., and Geissen, V. (2019). Pesticide residues in European agricultural soils–a hidden reality unfolded. *Sci. Total Environ.* 653, 1532–1545. doi: 10.1016/j.scitotenv.2018.10.441

Singer, A. C., Shaw, H., Rhodes, V., and Hart, A. (2016). Review of antimicrobial resistance in the environment and its relevance to environmental regulators. *Front. Microbiol.* 7:1728. doi: 10.3389/fmicb.2016.01728

Sousa, J. C., Ribeiro, A. R., Barbosa, M. O., Pereira, M. F. R., and Silva, A. M. (2018). A review on environmental monitoring of water organic pollutants identified by EU guidelines. *J. Hazard. Mater.* 344, 146–162. doi: 10.1016/j.jhazmat.2017.09.058

Takács-Novák, K., Nemes-Kovacs, E., Szasz, G., and Noszal, B. (1992). Lipophilicity of antibacterial fluoroquinolones. *Int. J. Pharm.* 79, 89–96.

Tell, J., Caldwell, D. J., Häner, A., Hellstern, J., Hoeger, B., Journel, R., et al. (2019). Science-based targets for antibiotics in receiving waters from pharmaceutical manufacturing operations. *Integr. Environ. Assess. Manag.* 15, 312–319. doi: 10.1002/ieam.4141

Trapp, S., Shi, J., and Zeng, L. (2023). Generic model for plant uptake of ionizable pharmaceuticals and personal care products. *Environ. Toxicol. Chem.* 42, 793–804. doi: 10.1002/etc.5582

Torniainen, K., Tammilehto, S., and Ulvi, V. (1996). The effect of pH, buffer type and drug concentration on the photodegradation of ciprofloxacin. *Int. J. Pharm.* 132, 53–61. doi: 10.1016/0378-5173(95)04332-2

Umweltbundesamt. (2022). Pharmaceuticals in the Environment. Available online at: https://www.umweltbundesamt.de/en/database-pharmaceuticals-in-the-environment-0 (Accessed February 02, 2026).

The Rivers Trust. (2024). State of our Rivers Report 2024. Available online at: https://theriverstrust.org/rivers-report-2024 (Accessed March 7, 2024)

Vorosmarty, C. J., McIntyre, P. B., Gessner, M. O., Dudgeon, D., Prusevich, A., Green, P., et al. (2010). Global threats to human water security and river biodiversity. *Nature* 467, 555–561. doi: 10.1038/nature09440

World Health Organization. (2021). Global status report on water, sanitation and hygiene: increasing access for health and reducing inequalities. Available online at: https://www.who.int/publications/i/item/9789240031352

Wronski, A. R., and Brooks, B. W. (2023). Global occurrence and aquatic hazards of antipsychotics in sewage influents, effluent discharges and surface waters. *Environ. Pollut.* 320:121042. doi: 10.1016/j.envpol.2023.121042

Yalkowsky, S. H., He, Y., and Jain, P. (2010). *Handbook of Aqueous Solubility Data*. 2nd Edn Boca Raton, FL: CRC Press, 152.

Yalkowsky, S. H., and Dannenfelser, R. M. (1992). AQUASOL database of aqueous solubility, version 5 Tucson, AZ University of Arizona

Yan, Y., Deng, Y., Li, W., Du, W., Gu, Y., Li, J., et al. (2021). Phytoremediation of antibiotic-contaminated wastewater: insight into the comparison of ciprofloxacin absorption, migration, and transformation process at different growth stages of *E. crassipes*. *Chemosphere* 283:131192. doi: 10.1016/j.chemosphere.2021.131192

You, X., Li, H., Pan, B., You, M., and Sun, W. (2022). Interactions between antibiotics and heavy metals determine their combined toxicity to *Synechocystis* sp. *J. Hazard. Mater.* 424:127707. doi: 10.1016/j.jhazmat.2021.127707

Zhang, X., Starner, K., Goh, K. S., and Gill, S. (2012). Analysis of diazinon agricultural use in regions of frequent surface water detections in California, USA. *Bull. Environ. Contam. Toxicol.* 88, 333–337. doi: 10.1007/s00128-011-0488-4

Zhou, Q., Yang, N., Li, Y., Ren, B., Ding, X., Bian, H., et al. (2020). Total concentrations and sources of heavy metal pollution in global river and lake water bodies from 1972 to 2017. *Glob. Ecol. Conserv.* 22:e00925. doi: 10.1016/j.gecco.2020.e00925