REVIEW ARTICLE OPEN

Proclivities for prevalence and treatment of antibiotics in the ambient water: a review

Chandrashekhar Bhagat^{1,4}, Manish Kumar^{2,4}✉, Vinay Kumar Tyagi³ and Pranab Kumar Mohapatra¹

In the initial two decades of the twenty-first century, antibiotic (AB) resistance in human pathogens has emerged as a major challenge for water, sanitation, and public health. Considering these challenges, we critically reviewed AB-related studies with particular emphasis on their (i) patterns of consumption, (ii) pathway prevalences and environmental implications in ambient waters, and (iii) benefits and limitations of existing AB removal/purging techniques. We found that lifestyle, land use, urbanization, the ease of availability, and the tendency of the medical practitioners to recommend ABs are the key factors governing the AB use pathway and enrichment in the environment. In the developing world, the most prevalent group of ABs is quinolone, whereas in the developed world, older-generation AB groups are most prevalent. Further, enormous variability in the consumption of ABs around the globe is explicitly highlighted in this study. Ciprofloxacin has been reported in the highest concentration among all the ABs with 28–31 mg L⁻¹ in the raw wastewater of the Indian subcontinent. We found that adsorption may be one of the most efficient AB removal techniques, and NaOH-activated carbon prepared from Macadamia nut shells is the most effective adsorbent identified to date. The literature showed that the Langmuir isotherm and pseudo-second-order kinetic model explain the AB adsorption mechanism most effectively. The future challenge lies in developing advanced protocols and markers to prioritize the strategy and simulate the ecotoxicities of the individual and a mixture of ABs.

npj Clean Water (2020)3:42; <https://doi.org/10.1038/s41545-020-00087-x>

INTRODUCTION

In recent centuries, our environment has become a dumping ground of natural and synthetic chemicals, such as pharmaceuticals and personal care products (PPCPs), and active pharmaceutical ingredients or pharmaceutically active compounds^{1–3}. Among various PPCPs, antibiotics (ABs) are becoming ubiquitous in the environment and pose serious human and environmental health concerns^{1,2}. ABs are classified into various types based on their chemical structure and mechanisms of action (i.e., aminoglycosides, penicillins/beta-lactams, quinolones, tetracyclines, macrolides, lincosamides, and sulfonamides). Moreover, they can also be grouped into two classes based on their spectrum known as narrow-spectrum ABs (i.e., narrow-spectrum penicillins) and broad-spectrum ABs (i.e., broad-spectrum penicillins). A summary of their basic structure, chemical formulas, molecular weight, and trade names is provided in Table 1. Growing demands and extensive application of ABs in various sectors lead to their massive annual production and consumption (131,000 tons used only for animal production globally in 2013) across the globe (Table 1). Recently, AB prevalence in the environment is being noticed and causing concern due to its wide applications in farming, aquaculture, human, and animal illnesses^{2–4}, animal husbandry, and livestock storage for prophylactic, therapeutic, and growth promoter agents^{5,6}. Considering that there are only limited studies in emerging economies, we have explicitly mentioned India's percentage share of ABs in the global consumption of ABs (Table 1).

Due to the extensive application and wide-spectrum use of ABs in recent decades, AB residues are now frequently being detected in raw or treated wastewater (WW)^{7–9}, hospital effluents^{2,10,11}, surface water^{2,12–14}, groundwater^{15,16}, and soil sediments in the

natural systems^{6,17}. The range of concentration is from ng L⁻¹ (norfloxacin, 11.1–964 ng L⁻¹)⁸ to mg L⁻¹ (ciprofloxacin, 28–31 mg L⁻¹)¹⁸. Owing to the reality that ABs can cause severe damage to the aquatic and human life even in trace concentrations, their buildup in the environment and way up to the trophic levels has been recognized as one of the gravest environmental issues to policymakers and industry personnel worldwide¹⁶. Due to tedious analytical procedures for the finding of AB concentration, only a few studies in the emerging economies, such as India, Pakistan, Bangladesh, and Sri Lanka, have reported¹⁹ in the removal methods^{7,20}, and the impact of ABs on aquatic matrices^{3,4}. Thus, it is imperative to compare the prevalence of ABs in various parts of the world and from different aqueous matrices (surface water, groundwater, and WW).

At present, the common consensus is that municipal WW treatment plants (WWTPs) are not always efficient²¹ enough to entirely remove PPCPs from the WW^{1,22} and thus act as the provenance of pharmaceuticals (specifically ABs) into the environment²³. There is the need of the hour to develop more efficient purging techniques, to meet the goal of WW reuse, and to meet the criteria for safe discharge into the ambient water environment²⁴. Several treatment processes have been used for AB removal, including ozonation, Fenton oxidation (dark light)²⁵, photo-Fenton oxidation^{26–29}, semiconductor photocatalysis (SP) using undoped/doped titania (TiO₂)³⁰, photocatalysis^{24,31,32}, Fenton oxidation combined with reverse osmosis (two-stage RO)³³, and electrochemical oxidation³⁴. Adsorption-based techniques using activated carbon³⁵ or biochar^{36–38}, or ecological treatment processes like constructed wetlands (CWs) and stabilization ponds^{1,8,39} have also been tried and tested for their efficiency on the removal of AB. Therefore, it is very pertinent to have a

¹Discipline of Civil Engineering, Indian Institute of Technology, Gandhinagar 382355 Gujarat, India. ²Discipline of Earth Sciences, Indian Institute of Technology, Gandhinagar 382355 Gujarat, India. ³Department of Civil Engineering, Indian Institute of Technology, Roorkee 247667 Uttarakhand, India. ⁴These authors contributed equally: Chandrashekhar Bhagat, Manish Kumar. ✉email: manish.env@gmail.com

Table 1. Enumeration of types and trade names of antibiotics along with their common uses and corresponding side effects with the % consumption of each antibiotic type in India to the world's consumption in year 2015.

Types and trade names	Structure	General information, uses, and side effect
Aminoglycosides: Garamycin, Genoptic, TOBI, Tobradex, Tobrex, Amikin, Humatin, Amikin, Arikayce, Kantrex, Nebcin, Neo-Fradin, Zemdri		Amino sugars combined through glycoside linkage to a hexose nucleus. Formula: $C_{21}H_{26}N_4O_{12}$; mol. wt.: 462 g mol ⁻¹ Uses: urinary tract infection, septicemia, severe pelvic inflammatory disease, ocular infections Side effect: chronic kidney disease, paralysis
Beta-lactams: Amoxil, Augmentin, Unasyn, Dynapen, Bactocill, PC Pen VK		Structurally glycosylated tetracyclines Uses: to treat breast cancer, anticancer Side effect: cardiac toxicity, glycemia, hyperglycemia
Carbapenems: Doribax, Merrem, Invanz, Primaxin IM, Primaxin IV		Analogous to the penicillins (sulfur substituted by carbon). Formula: $C_{21}H_{19}NO_4$; mol. wt.: 349.386 g mol ⁻¹ Uses: intra-abdominal, gynecological, joint/skin infections Side effect: hiccups, blood dyscrasias, leucopenia
Cephalosporins: Duricef, Velosef, Kefzol, Keflex, Ancef, Biocef, Panixine, Maxipine		Own a cephem nucleus to which two side series are allied: esterifies carbamate group (R1) and nucleus (R3) Uses: skin and soft tissue; CNS, sexually transmitted diseases Side effect: allergic reactions, fever, anaphylaxis
Penicillins: Dycill, Unipen, Bactocill, Bicillin L-A, Penicillin VK		Thiazolidine ring are linked to a beta-lactam ring. Formula: $C_{16}H_{17}N_2NaO_4S$; mol. wt.: 356.372 g mol ⁻¹ Uses: infections caused by bacteria Side effect: nausea, vomiting, diarrhea, abdominal pain
Glycopeptides: Vancocin-HCL, Vancocin, Firvanq, Orbactiv, Dalvance, Vibativ, Vancocin-HCL Pulvules		This structure tranquil for carbohydrate moieties (glycans) covalently joined to adjacent chains of amino acid Uses: <i>Clostridium difficile</i> -associated diarrhea, endocarditis, enterocolitis, severe skin, and skin structure infections Side effect: abdominal pain, back pain, diarrhea, edema
Lincosamides: Pirlimycin, Lincomycin, Cleocin, Cleocin T, Evoclin, Clindacin, Clindangel		Family of lincomycin and glycosides, a pyrrolidine ring linked through an amide bond to pyranose moiety. Formula: $C_{36}H_{67}ClN_4O_{11}S_2$; mol. wt.: 831.519 g mol ⁻¹ Uses: microflora, Gram positive, protozoal infections Side effect: diarrhea, nausea, heartburn, headache
Quinolones: Maxaquin, Floxin, Tequin, Noroxin, Avelox, Cipro I.V., Levaquin, Proquin XR, Factive		Two fused rings of a carboxylic acid and ketone group. If R4 F is attached, then it is known as a fluoroquinolone compound Uses: STDs, urinary, abdominal, respiratory tract infections Side effect: diarrhea and abdominal pain, insomnia, dizziness
Sulfonamides: Bactrim, Bactrim DS, Septra, Azulfidine, Eryzole, Gantrisin, Co-trimoxazole, Sulfatrim		They are differentiated by the sulfonyl group linked to amine group. Formula: $C_7H_8F_7NO_3S$; mol. wt.: 319.194 g mol ⁻¹ Uses: diabetes, pain and inflammation, high blood pressure Side effect: skin rash, itching, headache, tiredness, nausea
Tetracyclines: Declomycin, Adoxa, Dorxy, Xerava, Dynacin, Minocin, Solodyn, Nuzyz, Panmycin, Sumycin		Octa-hydro naphthacene ring skeleton, consisting of four fused Uses: syphilis, gonorrhea, endocervical, rectal infections Side effect: hepatotoxicity, liver failure, black hairy tongue
Chloramphenicol: Achlor, Aciphenicol, Acronicol, Brymcentin, Cadimycetine, Chemocetin, Chorampheniol, Decol, Ecol, Economycetin		Consists of nitrobenzene moiety linked to propanol and amino group. Formula: $C_{11}H_{12}Cl_2N_2O_5$; mol. wt.: 323 g mol ⁻¹ Uses: serious bacterial infections, infections of eyes and ears Side effect: Gray syndrome in babies, anemia, dark colored urine, difficulty in breathing, headache, nausea, and vomiting
Mitomycin: Almito, Mitocin, Mitodus, Mitomycin-C, Oncocin		Formula: $C_{15}H_{18}N_4O_5$; mol. wt.: 334.332 g mol ⁻¹ Uses: stomach cancer, pancreas cancer

Table 1 continued

Types and trade names	Structure	General information, uses, and side effect
Trimethoprim: Antrima, Aubril, Bactrim, Moly KID, Sepmax, Septram, Superistol		Side effect: uremic syndrome, fatigue pulmonary, cystitis Diaminopyrimidine and structural analogs of pteridine moiety of folic acid. Formula: $C_{14}H_{18}N_4O_3$; mol. wt.: 290 g mol^{-1} Uses: viral infection, bacterial, and urinary tract infection
Norfloxacin: Acinor, Almet-T Ambenil-D, Bacigyl-N, Dakson-M		Side effect: diarrhea, vomiting, stomach upset, and loss of appetite First generation and core structure is same as fluoroquinolone. Formula: $C_{16}H_{18}FN_3O_3$; mol. wt.: 319 g mol^{-1} Uses: bacterial infection, viral infection
Ciprofloxacin: Adiflox, Alcip, Baycip, Bekaycin, Ceplox, Cipro XR		Side effect: nausea, abdominal pain, indigestion, constipation Second generation of fluoroquinolone. Formula: $C_{17}H_{18}FN_3O_3$; mol. wt.: $331.347 \text{ g mol}^{-1}$ Uses: pneumonia, bacterial respiratory infection
Cetirizine: Aceper, Alcoryl Bioder Plus, Carcet FC, Dexphen		Side effect: diarrhea, dizziness, abdominal pain, vomiting Second-generation histamine. Formula: $C_{21}H_{25}ClN_2O_3$; mol. wt.: 388 g mol^{-1} Uses: allergy symptom, runny nose, itching eyes/nose Side effect: excessive tiredness, dry mouth, stomach pain

comparative understanding of the pros and cons of each available technique, along with the best efficiency achieved with it.

Overall, the first two decades of the twenty-first century have been the decades of water, health, and sanitation, where AB resistance in human pathogens has become a critical health and social issue. It has led to numerous research articles on the occurrence, fate, transport, and removal of ABs. However, a single review article that may kickstart with types, classes, use, and side effects of ABs, and then attempts at providing the recent proclivities of prevalence, pathways, and purging of ABs in ambient water, is of immense importance. Thus, under the light of the above discussions, this review provides a comprehensive account of the consumption pattern and reported occurrences in the environmental matrices to build a better understanding of the removal efficiencies of the existing techniques with a particular emphasis on India. The objective is threefold, namely (i) the pattern of consumption, prevalences in ambient waters, pathways, and environmental implications of various ABs, (ii) benefits and apprehensions of existing AB removal/purging techniques, and (iii) enumeration of the most frequently used removal techniques and their efficiency, kinetics, and isotherms associated with AB removal. We intend to report the decadal change in production and consumption of AB along with reported prevalence in the various aqueous matrices in the selected part of the world, and then provide the best adsorbent, most fit isotherm, and removal kinetics reported for AB purging through extensive discussion on various advance removal techniques and processes. This review tries to cover the overall cycle of AB contaminants for both the general and specialized audience working in this field.

PATTERN OF CONSUMPTION AND PREVALENCE OF CONTAMINATION

AB consumption pattern

AB consumption is increasing on a daily basis due to increasing population, affordability, lifestyle, and availability. Figure 1a–d shows the spatial and temporal variation in AB consumption of selected countries from the developed world (Australia, Japan, Sweden, United Kingdom, and the United States) and the

developing world (Bangladesh, China, India, Pakistan, and Sri Lanka) (the year 2000–2015). Given the reported data, it is not surprising to see the drastic increases of ABs used from 131,000 tons in 2013 and are expected to be 200,000 tons by 2030⁴⁰. The top five AB-consuming countries were India (2623 tons in 2013, predicted to increase by 84% by 2030), China (78,200 tons in 2013, predicted to surge by 59% by 2030), United States (9476 tons in 2013, 22% increase by 2030), Brazil (6448 tons in 2013, 41% increase by 2030), and Spain (2202 tons in 2013, 6% increase by 2030)⁴⁰. The percentage share of different AB consumption in India is shown in Table 1. Penicillin is the most consumed AB followed by tetracycline, quinolones, microlides, trimethoprim, and aminoglycosides (Fig. 2). Consumption of ABs in the selected developed countries is almost 1.5–3 times higher than developing countries from the developing world. The total use of penicillin and trimethoprim in the year 2000 (Fig. 1a) in the selected countries from the developed and developing world were 12,384 and 3374, and 1705 and 1350 daily dose (DD) per 1000 people, respectively. No distinct difference in AB consumption pattern could be observed between selected countries from the developed and developing world (Fig. 1). Apart from the increased consumption of tetracycline and macrolides in India from the year 2000 to 2015 (Fig. 1a–d), no specific AB consumption pattern seems obvious.

Prevalence of AB contamination

Table 2 and Supplementary Table 1 summarizes the AB contaminants along with their concentrations in various environmental spheres. Generally, AB concentrations are reported in mg L^{-1} in pharmaceutical effluents and hospital effluents^{11,18}, in $\mu\text{g L}^{-1}$ in municipal effluents⁷, and ng L^{-1} in freshwater (surface and groundwater) and seawater. These broad-range concentrations of ABs are found universally applicable with good spatial and temporal change around the globe⁴¹. Figure 2 depicts the concentration of ABs reported in India, China, and the United States, with the highest reported AB concentration in the Indian subcontinent, followed by China and the United States. Among all ABs, ciprofloxacin showed the highest concentration of $31,000,000^{18} \text{ ng L}^{-1}$ in the ambient waters, which is indicative of its vast abundance and usage.

Fig. 2 Maximum concentration of antibiotics reported in the surface water bodies. Maximum concentration of antibiotics reported in the surface water bodies of India, China, and the United States that ranged between ng L^{-1} and $\mu\text{g L}^{-1}$. Indian aquatic environment seems to have the highest contamination, while Ciprofloxacin is the most commonly found antibiotics.

Interestingly, due to the varied types and uses of ABs, several of them were not common in various countries, making their comparison challenging. The concentration of ABs reported in the WW in the past two decades is shown in Table 2 and Supplementary Table 1. In India, the highest level of ABs reported is of ciprofloxacin ($28,000,000\text{--}31,000,000 \text{ ng L}^{-1}$), followed by losartan ($2,400,000\text{--}2,500,000 \text{ ng L}^{-1}$) and cetirizine ($1,300,000\text{--}1,400,000 \text{ ng L}^{-1}$)¹⁸. Another Indian study showed high levels of ampicillin ($104,200 \pm 98,110 \text{ ng L}^{-1}$) in the WW⁴¹. It was observed that the AB concentration is higher in WW than other environmental aqueous matrices, that is, surface (river) water and groundwater and hospital effluents. Several studies reported the AB traces in the river water systems. Mutiyar and Mittal⁴¹ studied the occurrence of ABs in the Yamuna river, India. They reported the high concentrations of ampicillin ($13,800 \text{ ng L}^{-1}$), ciprofloxacin (1400 ng L^{-1}), gatifloxacin (480 ng L^{-1}), sparfloxacin (2100 ng L^{-1}), and cefuroxime (1700 ng L^{-1}) in river

water. These concentrations of ABs are roughly a thousand times lower than the values observed in WWTP effluents (Table 2 and Supplementary Table 1)¹⁸. Varying levels of AB concentrations have been reported in water in Indian rivers: amphetamine (984 ng L^{-1}) in Cooum river, Chennai⁴²; sulfamethoxazole ($900\text{--}1600 \text{ ng L}^{-1}$) in Kaveri river⁴²; ciprofloxacin ($10\text{--}130 \text{ ng L}^{-1}$) and sulfamethoxazole ($50\text{--}120 \text{ ng L}^{-1}$) in the Ganges River⁴³; ciprofloxacin ($7447\text{--}5,015,000 \text{ ng L}^{-1}$), enrofloxacin ($2262\text{--}181,609 \text{ ng L}^{-1}$), norfloxacin ($16,148\text{--}251,137 \text{ ng L}^{-1}$), and pefloxacin ($741\text{--}38,335 \text{ ng L}^{-1}$) in river Mousi⁴⁴. Concentrations of carbamazepine ($3\text{--}75 \text{ ng L}^{-1}$), theophylline ($609\text{--}2939 \text{ ng L}^{-1}$), and acetaminophen ($2156\text{--}5967 \text{ ng L}^{-1}$) were found in the Brahmaputra river, and its tributary Bharalu, Assam, India².

Overall, the concentration range of ABs found in Indian rivers was much higher than that in China. Figure 3 shows the variation in AB concentrations reported worldwide. Table 3 shows the concentration of ABs found in different environmental spheres of India, China, the United States, and the rest of the world, indicating a significant deviation in reported AB concentrations ($2000\text{--}7000 \text{ ng L}^{-1}$) ranging from few ng L^{-1} (3.24 ng L^{-1}) to thousands of $\mu\text{g L}^{-1}$ ($31,000 \mu\text{g L}^{-1}$). No correlation was found between the concentration of the ABs in different matrices within the same country.

Impending environmental consequences of AB prevalence

The occurrence of high concentrations of ABs in the aquatic ecosystem and the resulting AB-resistant genes (ARGs) have recently been reported as a serious risk to the environment and humanoid health³². The presence of ABs in the environment has severe effects on aquatic and earthly organisms^{45–47}. ABs alter the metabolic activities and community composition in microbes⁴⁸, leading to environmental pollution by AB-resistant bacteria (ARB) and ARGs^{2,9,49–51}. Even in low concentrations, they pose a grave environmental problem due to the high endurance and ecotoxicity, influence the aquatic species, and have severe effects on the ecosystem⁵². AB contamination effect includes the development of direct toxicity to microorganisms (flora and fauna) and probable

Table 2. Summary of concentration range of each antibiotics in various ambient waters, that is, surface water, river, hospital effluents, wastewater influent, and effluents.

Antibiotics	Concentration of antibiotics in ng L ⁻¹				
	Environmental matrices				
	Surface water (lake and ponds)	River	WWTPs		Hospital effluent
			Influent	Effluent	
Amoxicillin	0–200 × 10 ³	NA	0–6940	0–62.5	90–900
Ampicillin	NA	0–137 × 10 ²	23.5–263.3	0–12.68	NA
Penicillins	1.8–5.9	NA	20–138 × 10 ²	NA	NA
Sparfloxacin	NA	0–2090	0–632 × 10 ²	NA	NA
Cloxacillin	NA	NA	0–320	NA	NA
Ciprofloxacin	14–501.5 × 10 ⁴	0–250 × 10 ⁴	0.82–454 × 10 ⁴	0–310 × 10 ⁵	0–150 × 10 ²
Enrofloxacin	0–181.6 × 10 ³	NA	10–270	0–100 × 10 ²	60–100
Gatifloxacin	NA	0–42	0–840	0–1.22	
Lomefloxacin	35–102.6 × 10 ²	NA	NA	0–300 × 10 ³	0–1162
Levofloxacin	414	NA	86,700 ⁷	0–300 × 10 ³	NA
Norfloxacin	30–251 × 10 ³	0.87–6800	11.1–182 × 10 ²	85–420 × 10 ³	90–1620
Ofloxacin	0–488	0.43–117 × 10 ²	15.7–7900	0–160 × 10 ³	0–2340
Chlortetracycline	0–600	NA	0.14–420	NA	NA
Doxycycline	NA	NA	100–109 × 10 ²	NA	NA
Oxytetracycline	110–4200	NA	0.2–470 × 10 ²	NA	NA
Tetracycline	110–4200	31–255 × 10 ²	0–480 × 10 ²	NA	0–80
Clindamycin	NA	NA	5.16–1870	6.96–952	NA
Lincomycin	3.13–248.9	NA	15.20–730	3.92–430	NA
Lincosamides	60	NA	60–80	NA	6–170 ⁵
Erythromycin	1.41–15.9	NA	12–4740	0–6300	NA
Roxithromycin	NA	NA	870–1500	NA	NA
Sulfadimethoxine	35–60	NA	0.2–350	NA	NA
Sulfamethazine	20–120	NA	1.1–1500	NA	NA
Sulfathiazole	NA	NA	300–350	NA	NA
Sulfamethoxazole	0–2000	3.1–137 × 10 ²	0.4–2260	0–1100	100–300
Trimethoprim	3–710	17–4000	1.93–4010	1–2080	NA
Metoprolol	NA	NA	NA	800–950 × 10 ³	NA
Cefuroxime	NA	0–1700	0–9500	0.22	NA
Losartan	NA	NA	NA	240–250 × 10 ⁴	NA
Miconazole	NA	NA	23.4–1410	8.92–1024	NA
Cetirizine	NA	NA	(130–140) × 10 ⁴	NA	NA
Cephalexin	NA	NA	1020–5640	NA	NA

NA not available.

risks to humanoid health through the intake of polluted non-grail fauna⁵³.

The high concentration of ABs in the environment could act as bacteriostatic and bactericidal agents for the microbes. The bacteriostatic proctors reduce the proliferation of bacterial cells, whereas bactericidals eradicate the bacterial cells⁵⁴. AB encroachment enhances the proliferation and enzymic activities of the bacterial biome (biotic community), which alters the ecological functions (i.e., biomass production and transformation of nutrients)^{55,56}. These effects lead to changes in the transformations of nitrogen species, methanogenesis, sulfate reductions, nutrient cycles, and degradation of organic matters⁵⁷. Some of the bacteria can develop resistance against the ABs, as they may have been exposed already to such types of molecules in the environment⁵⁸. As summarized in Supplementary Table 2, most of the ABs affect bacterial growth and microbial biomass abundance, which may

decrease or increase depending on the concentration of ABs^{59–61}. The occurrence of ciprofloxacin in water may reduce the respiration or catabolic activity of microbes⁶¹. Nitrification and denitrification are also affected due to the presence of some ABs like sulfadiazine⁶² and sulfamethoxazole⁶³. The occurrence of AB residues in the environment alters the natural cycles and imbalance of the whole ecological system.

Sources and pathways of ABs

Since the discovery of ABs, their demand and production have grown exponentially worldwide, and consequently are discharged continuously into the environment as metabolites, deterioration products, or both by varied origins of input^{12,41}, as shown in Fig. 4. The source of ABs is primarily the human/animal excretion, as almost 80–90% of consumed ABs are discharged in consistent

Fig. 3 Worldwide variation of antibiotic concentrations. Worldwide variation in antibiotic concentrations exhibiting a huge variation in the reported antibiotic concentration with maximum observed deviation for cefuroxime ($2915.4 \pm 4447.9 \text{ ng L}^{-1}$), followed by cephalixin ($2800.1 \pm 3959.6 \text{ ng L}^{-1}$).

Table 3. Statistical summary of the mean and standard deviation in reported maximum and minimum values for India, China, the United States, and the world.

Countries	Environmental matrices									
	Surface water		River		WW influent		WW effluent		Hospital effluent	
	Min	Max	Min	Max	Min	Max	Min	Max	Min	Max
India	0.05 ± 0.06	1.09 ± 2.19	0.03 ± 0.06	4.10 ± 4.70	54.20 ± 265.35	61.28 ± 282.60	1485 ± 5946.95	165.05 ± 6580.40	NA	NA
China	NA	NA	1 ± 0.002	2.71 ± 6.06	0.09 ± 0.3	1150 ± 2080	1 ± 2	0.58 ± 0.420	NA	NA
United States	0	0.10 ± 0^a	NA	NA	NA	18.2 ± 0^a	NA	NA	NA	NA
Worldwide	NA	NA	0.4 ± 1.2	36.40 ± 290.400	29.60 ± 124.30	182.50 ± 133	273.50 ± 4043	0.53 ± 3.71	0.32 ± 0.79	1.9 ± 4.62

All units are given in ng L^{-1} .
 NA not available.
^aOnly one sample.

form and metabolites, because of the partial absorption by organisms in human and animal bodies⁶⁴. AB contamination spreads through WWTPs, runoff from agricultural fields, anthropogenic effluents, animal houses, slaughterhouses, and runoff from landfill³⁴. Recently, WWTPs are considered as the primary point source of AB contamination in the aquatic environment^{23,65,66}. However, unmanaged disposal of expired ABs from factories, hospitals, medicals, and residential locality contributes significantly⁶⁷ to sewerage and WWTP systems. ABs used as fertilizers or pesticides may pollute the soil and subsequently surface water and groundwater through runoff, infiltration, or percolation after rainfall and from landfill sites^{3,4}.

The effluent containing AB residues is generated from the residential areas (residential and hospitals), industries, slaughterhouses, animal houses, and dairy farms, and finally reaches the WWTPs. The sludge generated from the WWTPs also contains AB residues and is disposed of at landfill sites or used as a soil conditioner in the agricultural lands. Thus, WWTPs are well-known point sources, and agricultural fields are a line source of AB contamination. Wastes (liquid/solid) from pharmaceutical units or incidental spills during production and transportation can also be considered primary point sources of AB contamination in the environment^{68,69}. Also, the direct discharge from aquaculture (fisheries), poultry production is an important pathway of AB contamination. Overall, AB contamination in the environment occurs in a closed-loop manner termed as an “AB contaminant

cycle” that includes food chain routes and bioaccumulation and biomagnification through crop irrigation by WWTP effluents or contaminated soil, trophic levels, and aquaculture habitat (fish and crabs). Figure 4 also depicts how ABs enter into different ecosystems and finally reach the origin in a closed-loop manner.

AB PURGING TECHNIQUES AND PROCESSES

AB purging techniques

Table 4 summarizes the various techniques used for AB degradation and removal. Adsorption, SP, and ozonation techniques are the most studied techniques for AB removal. As far as the removal processes are concerned, both nonbiotic processes like sorption, hydrolysis, photolysis, oxidation, reduction, liquid extraction, and membrane techniques, and biotic processes are equally tried and tested for the purging of ABs. It has been reported that most of the removal techniques depend on factors, such as the composition of the matrix, temperature, pH of the system, and concentration of contaminants. In general, it has been reported that the conventional WWTPs fail to remove the highly polar contaminants like ABs^{2,3,21}, which emphasizes the need to develop elimination techniques for such contaminants^{3,4}. We present an overview of different removal techniques used for eliminating ABs in the past two decades.

Fig. 4 Pathways of antibiotics in different environmental matrices. Pathways of antibiotics in different environmental matrices where wastewater treatment plants being the most critical point source and agricultural fields and landfill sites are nonpoint sources for antibiotics.

In situ techniques (CWs)

CWs are categorized under in situ removal techniques that include vegetations like *Phragmites australis* and *Typha latifolia*¹ with a natural tendency to improve water quality by the cycling of nutrients and capturing contaminants from WW. In the past couple of years, researchers have been trying to find out wetlands' efficiency to treat the AB-contaminated water^{1,31,35}. WW treatment using CW could emerge as an economical and promising substitute to the traditional WWTPs for eliminating or reducing the level of nitrogen, phosphorus, pharmaceuticals (ABs), and biological oxygen demand (BOD) in treated water¹. Earlier published research reported that CWs are very efficient in reducing the concentration of ABs and ARGs. CW could achieve relatively similar or high elimination rates of the ABs than conventional mechanized WWTPs. The finding revealed that the most frequently detected ABs in the municipal WW might be effectively reduced up to 78–100% through the CW treatment systems. The fundamental mechanisms of AB removal in CW are sorption and biodegradation⁷⁰. Finding the percentage contribution of both the mechanisms remains an open question for researchers. The ABs' removal efficiency of CW varies from 10.3% (sulfadiazine)⁷⁰ to 100% (ciprofloxacin)¹ (Table 4). CW could be a promising solution for AB removal from the municipal WW⁷⁰, but can serve relatively small localities or towns⁷¹.

Ex situ techniques (conventional treatment)

Conventional treatments are categorized as ex situ techniques because WW was brought to the treatment plant from the actual site and treated under various stages of filtration, coagulation–flocculation, sedimentation, and biological (activated sludge process (ASP) and trickling filters) treatment^{8,23,72,73}. Various WW treatment processes are being tested for AB removal across the globe (Table 4). These processes work effectively for WW having low initial concentrations of contaminants. Mutiyyar and Mittal⁴¹ reported that AB removal efficiency of the conventional ASP-based WWTP varies from 99.37% (sparfloxacin) to 55.47% (gatifloxacin). In China, Leung et al.²³ reported that the AB removal efficiency of WWTP (ASP) varies from 70–80% (cephalexin) to 3–5% (ofloxacin). Sometimes, there is also an increase in the concentration of ABs. Prabhasankar et al.²¹ found that the removal efficiency varied from –23.07% (erythromycin: influent = 26 ng L⁻¹, effluent = 32 ng L⁻¹) to 87% (ampicillin).

The negative removal efficiencies of ABs can be explained by one of the following reasons. First, most of the ABs found that the WW samples can be excreted from the human body through the urine and feces as a mixture of stable parent compounds and conjugates of glucuronic acid^{74–76}. Thus, these conjugates may undergo enzymatic cleavage to form their parent compounds throughout biological WW treatment processes, resulting in increased concentrations in treated effluents (appeared as negative removal efficiency). Second, ABs excreted through the urine and feces might be fenced in fecal particles in the WW and slowly released throughout the treatment processes, hence increasing the AB concentration in the treated effluent⁷⁷ (i.e., appeared as negative removal efficiency). Third, negative removal efficiencies of ABs in the WWTPs could be due to sampling errors.

However, there is no guaranteed removal of ABs that are found in conventional WWTPs⁷⁷. Several researchers studied the performance of physicochemical methods for AB removal and reported that maximum removal of 30% was achieved⁷⁸. The AB removal efficiency of physical and chemical treatments depends on various physicochemical factors like effluent pH, temperature, initial concentration, and types of ABs, and operational factors of treatment system like sludge retention time and hydraulic retention time. This dependency makes removal efficiency highly variable and is a shortcoming of the conventional techniques^{79–81}. Due to the low efficiencies and shortcomings of these methodologies, they are not considered feasible for AB-contaminated effluent treatment.

REMOVAL PROCESSES AND MECHANISMS

Adsorption

Adsorption techniques are broadly practiced around the globe to treat AB-contaminated liquid waste. There are a few mechanisms involved in adsorption processes for removing the AB from the WW using adsorbents; the mechanisms are ion exchange, pore filling, and pi–pi electron interaction⁸², the electrostatic mechanism⁸³, surface complexation, and hydrogen bonding. Removal using the ion-exchange mechanism depends upon the pollutant size and surface functional group (SFG) of the adsorbent used⁸⁴. The pore structure of the adsorbent facilitates a AB contaminant adsorption through the pore-filling mechanism⁸⁵. The adsorption efficiency is dependent on the physicochemical properties of the adsorbent as well as the medium characteristics. The characteristics of the adsorbent include surface area, porosity, and aperture opening (diameter)⁸⁶, and so on. The liquid-phase characteristics are pH, temperature, type of contaminants and concentration, and organic matter.

The removal process (Table 5) is highly influenced by pH because it unswervingly affects the physicochemical properties and the adsorbent's surface activity. Sorption reaction is temperature dependent because it is an exothermic reaction. At the same time, organic matter also influences the removal efficiency due to its direct competition with the targeted molecules (AB residues) for adsorptive sites. The nature of the adsorbent highly influences the rate of adsorption and its capacity. Several studies focused on increasing the adsorption efficiency by improving the SFG by using thermal or chemical preactivation.

Granular activated carbon (GAC) exhibited promising removal capacity with limitations about their cost and difficult synthesis procedures⁸⁷. Nowadays, finding a substitute for GAC is a growing and attractive research area for the researcher. Low-cost adsorbents produced from agricultural and industrial waste, are used in the removal of ABs from WW, for example, activated carbon produced from Jeriva⁸³, Alfalfa hays and pine wood^{88–91}, sawdust⁹², rice straw and swine manure⁹³, macadamia nut shells⁹⁴, dried duckweed³⁷, *Trapa natans* husk⁹, *Albizia lebeck* seed pods⁹³,

Table 4. Tally of various removal techniques applied for the removal of different antibiotics and their corresponding efficiency.

Removal techniques	Types of antibiotics	Concentration (ng L ⁻¹)			Remarks	
		I	E	R (%)		
Conventional treatment ^{8,21,22,41,119}	Amoxicillin	0–172.5	0–62.5	49.7	Only 34% of water is reused in Delhi (India). High variation in concentration in the same STP. Low F/M ratio impart good settling in STP	
	Ampicillin	9	1	87	Ampicillin is highly stable in river water relatively than M-Q water. The concentration of AB increase in monsoon as compared to another season. Temperature plays an important role in adsorption and usually is inversely proportional to removal rates. Removal efficiency depends on retention time, temperature, inflow, quality of influent, and quantity of influent	
	Trimethoprim	238	133	44		
	Sulfamethoxazole	509	1015	0		
	Chloramphenicol	3	3	0		
	Erythromycin	26	32	0		
	Naproxen	219	28	8		
	Ampicillin	104.2	12.68	87.83	The efficiency of removal for ABs varies with plants, their operations, locations, and environmental factors. Conventional treatment did not remove all influent concentrations with an ASP. Antibiotics contamination probably has toxic effects on aquatic organisms	
	Ciprofloxacin	20.06	8	60.11		
	Gatifloxacin	2.74	1.22	55.47		
	Sparfloxacin	22.49	0.14	99.37		
	Cefuroxime	3.42	0.22	93.56		
	Chlortetracycline	0.14–30	NA	70–95	ASP is not capable to remove all targeted ABs. The dominant mechanism in ASP adsorption and biodegradation. It is a more efficient process than the stabilizing pond and micropower biofilm method. Due to desorption, there might be a chance to increase some AB concentration	
	Sulfamethoxazole	0.4–61.1	NA	40–80		
	Roxithromycin	4.3–780	NA	98–100		
Constructed wetlands ^{1,8,14,70}	Ofloxacin	15–5411	NA	95–100	Conventional treatment like ASP is not capable of handling higher concentrations of ABs. PST efficiency is limited to 0–29%	
	Ofloxacin	147–7900	NA	3–5		
	Erythromycin	243–4740	NA	5–20		
	Cephalexin	1–5,640	NA	70–80		
	Clarithromycin	400	NA	59	Constructed wetland is efficient and economical for the removal of antibiotics compared to conventional plants. Antibiotic treatment using constructed wetland do not have a substantial consequence of bacterial diversity and evenness. ARG found frequently. Average removal efficiency does not fall below 59%. Consequently, a necessity to search alternative efficient and low-cost techniques that supplement conventional methods of treatment	
	Clindamycin	100	NA	74		
	Doxycycline	100	NA	100		
	Erythromycin	2000	NA	96		
	Oxytetracycline	400	NA	97		
	Sulfamethoxazole	1000	NA	77		
	Tetracycline	1000	NA	99		
	Trimethoprim	1000	NA	61		
	Fluoroquinolones	193	NA	100	Integrated constructed wetlands (ICWs) are a treatment for AB contamination useful in rural areas. Removal efficiency is from 10–100%. The AB loading rate is found in 3479 and 199 µg day ⁻¹ . Anaerobic degradation predominant mechanism for biodegradation of antibiotics. ICWs consist of floating macrophytes, emerging microphytes, and stabilization ponds	
	Lincosamides	60.7	NA	78		
	Sulfacetamide	3.04	NA	23.8		
Membrane techniques ^{117,118,179,180}	Sulfadiazine	2.40	NA	10.3		
	Sulfamethazine	54.1	NA	95.1		
	Trimethoprim	1.93	NA	100		
	Sulfamonomethox	50.8	NA	100		
	Erythromycin, Lincomycin, floxacilin Sulfamerazine, Novobiocin	6.03–11600	NA	17.9–98.5	Three HLR and four substrate (Oyster shell, Zeolite, Medical stone, ceramic) use to optimize the operation, 20 cm day ⁻¹ HLR and zeolite are best for removal. Biodegradation and adsorption are the dominant mechanisms in the removal process. Removal of COD, TN, and ammonia nitrogen increases with an increase in HLR in zeolite CWLs	
	Chlortetracycline	0.14–30	NA	55–85	CW work efficiently in summer as compared to winter. Desorption is a chance to increase the concentration of ABs. Degradation and plant uptakes might be a removal pathway. Hydraulic retention time and temperature are the key factors that affect the removal process	
	Sulfamethoxazole	0.4–61.1	NA	40–80		
	Roxithromycin	4.3–780	NA	80–90		
	Ofloxacin	15–5411	NA	60–100		
	Tetracycline	NA	NA	98.9	Hybrid carbon membranes (GO and AO) have a 15 µm thickness and is more effective than AC, GO, and carbon nanotube. Adsorption capacity is 449 mg g ⁻¹ and the surface area is 414 m ² g ⁻¹	
	Sulfamethoxazole, Trimethoprim, Norfloxacin, Roxithromycin	NA	NA	>99%	Forward osmosis with electrochemical oxidation (FOWEO) is used for the removal of antibiotics. The FOWEO is efficient than the forwarding the reverse osmosis technique. There is no significant effect of the mass flux rate on removal efficiency. The orientation of the membrane has influenced on the removal	
	Sulfamethoxazole, Flumequine	NA	NA	>99%	FM is efficient than UFM. Adsorption is the main mechanism in UF. An increase in operating pressure rate of permeation is increase, but the problem of fouling is increases. UF is more sensitive to fouling as compared to NF. External fouling is predominant in NFM. pH has negatively influenced on UFM and positive on NFM	
	Spectinomycin	NA	NA	100	An increase in the pH leads to an increase in rate of second-order reaction. Rapid degradation observed around neutral pH. Spectinomycin is degraded in 10 s	
	Amoxicillin	NA	NA	90.	Almost 90% of elimination is happening within 4 min and 18% mineralization after 20 min. There is a small percentage of mineralization observed even for a long duration of treatment	

Table 4 continued

Removal techniques	Types of antibiotics	Concentration (ng L ⁻¹)			Remarks
		I	E	R (%)	
	Ceftriaxone	(250–140) × 10 ⁷	NA	95	An increase in pH from 3 to 7 leads to an increase the COD removal. No advantage of the addition of H ₂ O ₂ . Biodegradability increased in terms of the BOD ₅ /COD ratio. After 60 min, 95% of degradation of COD is achieved
	Penicillin	830 × 10 ⁶	NA	83	An increase in pH leads to an increase in degradation. Mostly first-order kinetics followed by both processes. Biodegradability ratio, BOD ₅ /COD, increased by a factor of 6 and 23 for ozonation and pre-ozonation, for 20 min
	Ampicillin, Azithromycin, Erythromycin, Ofloxacin, rimethoprim, Tetracycline	100 × 10 ³	NA	>99	The efficiency of this process is dependent on ozone dose and hydraulic retention time (HRT). Inactivation of <i>Escherichia coli</i> takes place at HRT = 40 min and dose = 0.25 g DOC ⁻¹ . Maximum efficiency was achieved at HRT = 40 and 60 min. Forty-minute HRT is reported to be optimum. An increase in oxidant concentration improves the efficiency of the treatment process. Antibiotics are efficiently removed by ozone from SBR
	Sulfonamides	200 × 10 ³	NA	85	DOM also influence the efficiency of treatment. HRT and sludge retention time (SRT) are 10 and 45 days, respectively. At ozone dose, 13.3 mg L ⁻¹ min ⁻¹ with a COD concentration of 190 mg L ⁻¹ , 96% removal takes place in 5 min
	Tetracyclines	NA	NA	95	Achieved the highest removal at 75 mg L ⁻¹ ozone dose. Degradation products found to be water soluble and acidic in nature. Increasing the ozone dose leads to increasing the removal up to a certain dose after that saturation is achieved
	Macrolide	50–1000	NA	17–92	Ozonation efficiently removed the sulfonamides, but BAC increasing the sulfonamides resistance bacteria. BAC alone can remove ABs up to 83–98%. Optimum contact time 20 min with pH range was 7.1–7.9
	Amoxicillin	130–540	NA	95–99	Combined sponge MBR and ozonation used for the removal ABs from the hospital effluent. The main mechanism of removal is direct ozonation and reaction with hydroxyl radicals. Sponge MBR removes the total nitrogen as nitrification and denitrification, which is takes place simultaneously
	Doxycycline	351–4640	NA	88–96	
	Ciprofloxacin	130–1110	NA	89–91	
	Sulphadiazine	100–6200	NA	84–92	
	Sulfamethoxazole	265–3820	NA	76–90	
		101–4400	NA	65–67	
	Sulfachlorpyridazine	200 × 10 ⁵	NA	100	Nitrogen-doped reduces graphene oxide (N-RGO) metal-free is used as catalyst. Peroxymonosulfate (PMS) had excellent oxidation efficiency towards sulfonamide antibiotics. PMS and (N-RGO + PMS) systems remediate the aquatic environment efficiently in 10 min for 100% removal of ABs
		500 × 10 ⁵	NA	42–92	Seventy percent of TiO ₂ and 30% of calcite catalyst is the optimal combination for better removal. Ninety percent of removal takes place in the presence of UV and 82% in the presence of solar irradiation. Surface adsorption and formation of hydroxyl radicals influence degradation efficiency Tetracycline AB species are dominant in the 6–7.5 pH range. The efficiency of removal decreased with initial concentration increases
		250 × 10 ⁵	NA	82.92	Magnetic graphene oxide cerium-doped TiO ₂ hybrid catalyst enhanced degradation. Adsorption capacity of GO is more than bare Ce-TiO ₂ . MGO enhances adsorption tetracycline. For maximum removal, the optimum mass ratio of Ce-doped TiO ₂ /MGO is 10%
		(25–300) × 10 ⁵	NA	100	TiO ₂ (Degussa P-25) is the most efficient catalyst among all catalysts. Degradation of amoxicillin increased with the increase in doses of catalyst up to 100–250 mg L ⁻¹ . Transformation accelerated in acidic conditions
	Amoxicillin	104 × 10 ⁶	NA	100	ZnO very efficient catalyst for degradation. Complete degradation achieved with a dose of ZnO is 0.5 g L ⁻¹ , irradiation time is 180 min, and pH is 11. Degradation is increasing with pH and ZnO dose
	Ampicillin	105 × 10 ⁶	NA	100	
	Cloxacillin	103 × 10 ⁶	NA	100	
	Tetracycline	200 × 10 ⁵	NA	94.91	Visible-light-driven catalysts mostly uses in the treatment. Catalyst AgI (20% by wt)/BiVO ₄ promising catalyst. AgI loading remarkably promoted the photocatalytic activity of BiVO ₄ . Initial concentration decreased efficiency
	Oxolinic acid	200 × 10 ⁵	NA	76	The photocatalytic process is able to transform the target ABs into more oxidized by-products. Catalyst used is TiO ₂ (Degussa P-25). An increase in catalyst load improved removal efficiency. Maximum removal at pH (7.5) and 1 g L ⁻¹ of TiO ₂ dose. Follows the Langmuir isotherm model
		300 × 10 ⁴	NA	93	MAC with TiO ₂ showed good performance when coupling with ultraviolet light and ultrasonic light. Antibiotic concentration up to 10–20 mg L ⁻¹ degraded completely in 180 min
		13.6 μM	NA	100	At low- or medium-pressure UV lamps with or without hydrogen peroxide, degradation is expected to better. Degradation under polychromatic UV irradiation in the absence of H ₂ O ₂ is likely due to hydroxyl radical formation via nitrate photolysis. Doxycycline, Doxycycline, G
		22.5 μM	NA	80–85	
		4.67 μM	NA	90–95	
		29.9 μM	NA	65–70	

Semiconductor photocatalysis^{22,24,30–32,84–188}

Table 4 continued

Table 4 continued					
Removal techniques	Types of antibiotics	Concentration (ng L ⁻¹)			Remarks
		<i>I</i>	<i>E</i>	<i>R</i> (%)	
Oxidation ^{1,19,52,187}	Ciprofloxacin	6.04 μM	NA	30–35	Penicillin G, and Ciprofloxacin are degraded directly by photolysis at 254 nm low-pressure UV lamp Efficiency will increase with the increase in H ₂ O ₂ dose. No O and HO ₂ radical produce during the oxidation process
	Trimethoprim	68.9 μM	NA	100	
	Tetracyclines	400 × 10 ⁴	NA	Up to 97	
	Penicillin G	NA	NA	100	Gamma-radiation was effective. Mineralization rate was much lower than the degradation rate in the AOP process. Fe ²⁺ ions improved the degradation
	Erythromycin	13.6 μM	NA	NA	Oxidation was done using UV/H ₂ O ₂ and shows the promising degradation technique. An anti-bacterial-active product, which is observed in 254 nm UV did not form or rapidly
	Doxycycline	22.5 μM	NA	NA	despoiled when exposed to the polychromatic UV. Extra care should be taken before and
	Clindamycin	4.67 μM	NA	NA	after the oxidation, as Penicillin G is directly reacted with H ₂ O ₂
	Penicillin G	29.9 μM	NA	NA	
	Ciprofloxacin	6.04 μM	NA	NA	
	Trimethoprim	68.9 μM	NA	NA	
Actual values of concentrations of influent and effluents for which certain removal efficiencies were obtained are also provided along with major findings of that particular study. / influent concentration, <i>E</i> effluent concentration, <i>R</i> removal efficiency, NA not available.					

hazelnut shell^{36,37,94,95}, coal fly ash⁹⁶, and decaffeinated tea and coffee waste^{88,94,97,98}. The properties of these adsorbents and their removal efficiencies are summarized in Table 5.

In some studies, activated sewage sludge^{38,99} and clay minerals (i.e., montmorillonite, rectorite, illite¹⁰⁰, and bentonite¹⁰¹) have been used as adsorbents for removal of ABs (refer to Table 5). The chemically modified nanocomposites are used as adsorbents and reported in various studies are nano-hydroxyapatite³⁵, multiwalled carbon nanotube-modified MIL-53(Fe)^{102–104}, and magnetic nanocomposite¹⁰⁵ (chitosan, diphenylurea, formaldehyde, and magnetic nanoparticle MnFe₂O₄). Carvalho et al.⁸³ worked on eliminating ciprofloxacin using an adsorbent, produced from Jeriva (agricultural waste). The surface area and maximum adsorbent capacity of the adsorbent reported are 1435 m² g⁻¹ and 335.8 mg g⁻¹, respectively. Pour-etal and Sadegh¹⁰⁶ used activated carbon nanoparticles, made up of vine wood for the treatment of penicillin G and tetracycline AB-contaminated water/WW. The maximum adsorbent capacity for penicillin G and tetracycline ABs reported is 8.41 and 1.98 mg g⁻¹ and with removal efficiencies of 38.98–81.26% and 57.69–88.17%, respectively, at 400 mg L⁻¹ optimum dose. Jang et al.⁸⁸ worked on removing tetracycline using *Pinus taeda*-derived biochar (SA: 959.9 m² g⁻¹) and reported 274.8 mg g⁻¹ adsorbent capacity and with an excellent efficacy of removal. Jang et al.⁸⁹ investigated the elimination of sulfamethoxazole using pine wood biochar and found that at the 100 mg L⁻¹ doses, maximum removal takes place, and the maximum adsorbent capacity reported was 397.29 mg g⁻¹. The adsorption mechanism followed the Freundlich isotherm model and Elovich kinetic model. Table 5 summarizes all the studies related to different adsorbents. Most research articles have reported that adsorption processes are a promising tool for the treatment of AB-contaminated water/WW.

Oxidation

Advance oxidation treatment processes (AOTPs) result from the production of free hydroxyl and oxy radicals. These radicals are highly reactive and have an oxidation potential of 2.8 V, which is higher than ordinary or classical oxidants, making them highly effective in treating the AB-contaminated WW¹⁰⁷. There are some oxidizing agents often used to produce hydroxyl radicals like O₃, H₂O₂, and so on mixed with semiconductor catalysts/ultraviolet radiation/metal catalysts. In this process, it is expected that AB contaminants get oxidized and reduced into less harmful products. It might be possible that the by-products produced after oxidation are more harmful than the parental compounds.

Ozonation

Ozone is known as an effective oxidant due to its high oxidation potential ($E^0 = 2.07$ V). Ozone has high redox potential than classical oxidants like H₂O₂ (hydrogen peroxides), ClO₂ (chlorine dioxides), Cl₂ gas, and hypochlorites. It can react directly or indirectly with AB contaminants. The AB degradation mechanism primarily has direct oxidation rather than indirect oxidation via OH[•]¹⁰⁸ (radicals). It is necessary to have a C=C (carbon–carbon) or aromatic bonds with the N (nitrogen), O (oxygen), or S (molecules of sulfur) to carry out the direct oxidation using O₃ because it reacts only with the nucleophilic molecules. If this condition is not fulfilled, then the degradation of O₃ in WW to generate an OH[•] (hydroxyl radicals, it has unpaired electron) takes place through another mechanism.

Table 5. The various (40 studies) adsorbent used for the removal of a specific group of antibiotics along with the corresponding surface area, adsorption capacities with specific experimental doses, removal efficiencies, and kinetics and isotherms models.

Adsorbent	Dose and adsorbent properties			Efficiency	Isotherm model		Kinetic model	
	SA (m ² g ⁻¹)	Dose (mg L ⁻¹)	Capacity (mg g ⁻¹)		L	F Others	P-2nd order	Others
Ciprofloxacin antibiotics								
¹⁷³ Polysaccharide-based sodium alginate/k-carrageenan double-network gel beads	NA	250	291.6@ 5.1 pH	55–60%	✓	✓ x	✓	x
⁸⁴ Activated carbon produced from Jerivá	1435	600	335.8	93.43%	x x	Liu	x	Avrami
⁹⁵ ACs from <i>Albizia lebeck</i> seed pods by KOH chemical activation	1824.88	750	108.74 @ 30 °C; 118.05 @ 40 °C; 131.14 @ 50 °C	96.12%	✓	x	✓	x
¹⁰² Montmorillonite, rectorite, and illite	715, 363	100 × 10 ²	395, 135, and 33	95%	✓	x x	✓	x
³⁶ Nano-hydroxyapatite	NA	NA	10.7	47.3%	✓	Diffusion	✓	x
⁹⁸ Modified coal fly ash	17.5	4000	1.547	NA	✓	x x	✓	x
³⁶ Nano-hydroxyapatite [Ca ₁₀ (PO ₄) ₆ (OH) ₂]	NA	20,000	1.2715	47.3%	✓	✓ x	✓	x
³⁷ Hazelnut shell activated carbon	1247.2	1500	73.64	92.67%	✓	x x	✓	x
Norfloxacin								
⁹⁵ ACs from <i>Albizia lebeck</i> seed pods by KOH chemical activation	1824.88	1824.88	166.99 @ 30 °C; 160.90 @ 40 °C; 153.99 @ 50 °C	98.13%	✓	x x	✓	x
¹⁷⁴ <i>Trapa natans</i> husk activated carbon	NA	NA	1	90.7%	x	Tempkin	✓	x
³⁶ Nano-hydroxyapatite	NA	NA	9.2	51.6%	✓	✓ x	✓	x
¹⁷⁴ Aquatic plant residue <i>Trapa natans</i> husk	1274	70	2	90.7%	x	Tempkin	✓	x
³⁶ Nano-hydroxyapatite [Ca ₁₀ (PO ₄) ₆ (OH) ₂]	NA	20,000	1.4862	51.6%	✓	✓ x	✓	x
Penicillin G								
¹⁰¹ Dried <i>Rhizopus arrhizus</i>	NA	NA	459; 330; 375	78.3%; 61.1%; 64.4%	✓	x Redlich-Peterson	✓	Saturation type
Dried activated sludge								
³⁸ Dried duckweed (<i>Lemna minor</i>)	32.5	4000	36.1	94.6%	✓	x x	✓	x
¹⁷⁵ Decaffeinated tea waste	NA–	8000–400 × 10 ²	46.53	39.3%	x	✓ x	x	P-1st order
¹⁰⁸ Activated carbon nanoparticle from vine wood	13.397	400	8.41	39–81%	✓	x x	✓	x
Tetracycline								
⁸¹ Graphene oxide	NA	0.181	313	71.4%	✓	x Temkin	✓	x
⁹³ NaOH-activated carbon prepared from macadamia nut shells. 3 NaOH:1 CHAR	1524	25	455.83	100% @ pH = 3	x x	Temkin	x	Elovich
³⁹ Ferric activated sludge-based adsorbent from biological sludge	126.86	100	87.87	NA	x	✓ x	✓	x
^{104,105} Multiwalled carbon nanotube-modified MIL-53(Fe)	60.17	200	364.37	NA	✓	✓ x	✓	Elovich
⁸⁹ Biochar derived from Alfalfa hays and activated by NaOH	796.50	100	302.37	NA	x x	x x	x	x
⁹¹ Iron and zinc-doped sawdust biochar	NA	15,000	89.87	91.7%	✓	x x	✓	x
⁹² H ₃ PO ₄ modified biochar derived from rice straw and swine manure	372.21	330	167.5	NA	✓	✓ x	✓	x
³⁹ Ferric activated sludge-based adsorbent	319.04	330	160.3	NA	✓	✓ x	✓	x
⁸⁹ <i>Pinus taeda</i> -derived activated biochar	126.86	2000	87.87	NA	x	✓ x	✓	x
¹⁷⁶ Waste expanded polystyrene	959.9	100	274.8	NA	x	✓ x	x	x
	1041.64	200	621.12	NA	✓	x x	✓	Elovich

Table 5 continued

Adsorbent	Dose and adsorbent properties			Efficiency	Isotherm model			Kinetic model	
					L	F	Others	P-2nd order	Others
	SA (m ² g ⁻¹)	Dose (mg L ⁻¹)	Capacity (mg g ⁻¹)						
⁹⁵ Magnetic nanocomposite (chitosan, diphenylurea, formaldehyde, and MnFe ₂ O ₄)	442.6	33.33	168.24	NA	✓	x	x	✓	x
¹⁰⁸ ACs nanoparticle from vine wood	13.397	400	1.98	57.69–88.17%	✓	x	x	✓	x
Sulfadiazine									
¹⁷⁷ Expanded graphite	NA	100	16.586	80%	✓	x	Tempkin	x	✓
Enrofloxacin									
¹⁷⁸ Natural zeolite	NA	10,000	19.24	NA	✓	x	x	✓	x
¹⁷⁹ Clinoptilolite Natural zeolite	NA	10,000	2.430	NA	✓	✓	x	✓	x
Amoxicillin									
¹⁰³ AC, bentonite	1092.95 91.627	NA	189.551 47.3715	94.67% 88.01%	✓	✓	x	x	✓
¹⁰⁸ ACs nanoparticle from vine wood	13.397	400	2.96	88–94%	✓	x	x	✓	x
Oxytetracycline									
⁸¹ Graphene oxide	NA	0.181	212.31	71.4%	✓	x	Tempkin	✓	x
¹⁰⁴ Multiwalled carbon nanotube-modified MIL-53(Fe)	60.17	200	325.59	NA	x	x	x	x	x
Doxycycline									
⁸¹ Graphene oxide	NA	0.181	398.40	71.4%	✓	x	Tempkin	✓	x
Chlortetracycline									
¹⁰⁶ Multiwalled carbon nanotube-modified MIL-53(Fe)	60.17	200	180.68	NA	✓	x	x	✓	x
Chloramphenicol									
¹⁸⁰ <i>Typha orientalis</i> via phosphoric acid activation	794.8	600	137.1	65.6%	x	✓	x	✓	x
Sulfamethoxazole									
⁹⁰ ACs from pine wood	959.9	100	397.29	NA	x	✓	x	Elovich	
Cephalexin									
¹⁰⁸ ACs nanoparticle from vine wood	13.397	100	7.08	43.52–76.0	✓	x	x	✓	x

SA surface area, L Langmuir, F Freundlich, P-1st pseudo first order, P-2nd pseudo-second order, NA not available.

The reaction is given below:

As per reactions (i) and (ii), the degradation is pH sensitive, and hence we can accelerate the initiation of degradation by increasing the pH of the medium

The rate of the above reaction (vii) is fast, and it helps to minimize the oxidation capacity of the matrix, as it requires low dissolved organic carbon and alkalinity, and improves the efficiency of the treatment process. The AOTPs are very effective when the variability in inflow and AB composition in WW is high. In normal circumstances, these processes may be uneconomical due to the costly treatment units, maintenance, and energy requirement. There is a limitation related to the mass transformation in the oxidation process with O_3 ; if the mass transformation is inadequate, it is also one of the major disadvantages. It can negatively affect the removal efficiency of the process and, hence, result in increasing the operating costs. The performance of this process is susceptible to some factors like the concentration of solids, the presence of microorganism/pathogens, CO_3 (carbonate), HCO_3 (bicarbonate), HRT, ozone dose¹⁰⁹, pH, and temperature^{110,111}.

Iakovides et al.¹⁰⁹ have studied the removal of ABs by ozonation. They suggested that treatment optimization is necessary because excess ozone creates problems for useful microorganisms in treated effluents. Ozonation is highly dependent on the ozone dose and reaction time. A rise in the oxidant level leads to a multiplication of AB degradation rate in the treatment process. Ben et al.¹¹² integrated the sequencing batch reactor (SBR) with ozonation and achieved the removal of ABs (tetracycline, microlide, and sulfonamides) by 80–100% efficiency. Alsager et al.¹¹³ studied AB removal (amoxicillin, doxycycline, ciprofloxacin, and sulfadiazine) from water and dairy milk, and the efficiency was 95%. Ozonation with an adequately optimized dosage is a favorable technique for removal (>99% efficiency) of AB contaminants¹⁰⁹. Nowadays, ozonation is coupled with classical or advanced treatment processes to achieve the effective removal of ABs. Balcioglu and Ötke¹¹⁴ reported that hybrid ozone– H_2O_2 treatment has higher AB removal efficiency than ozone alone. Li et al.¹¹⁵ combined ozonation with biologically activated carbon filtration and the sponge membrane bioreactor with ozonation to remove the different AB contaminants from water.

Photocatalysis

SP techniques have gained significant interest due to being cost-effective, ecofriendly, and known to be a sustainable technology with negligible waste production¹¹⁶. SP requires three components to carry out the degradation of the ABs using an oxidative mechanism, a catalyst with photosensitive (e.g., TiO_2 or inorganic semiconductors), a photon energy source, and an efficient oxidizing agent^{22,24,30}. This fundamental principle includes the stimulation of a semiconductor (generally used TiO_2 because of its immense stability, efficient performance, cost-effectiveness, and ease of availability) by artificial means. The semiconductors are

characterized by their valence and bandgap, and conduction bands. The mechanisms involved in this process are as follows:

Reaction (viii) shows the loss of an electron through direct oxidation or photocatalysis. Reaction (ix) shows indirect oxidation. The mechanism shown in reaction (ix) is as fast as reaction (viii). All the reported studies show that for the photomineralization, the presence of H_2O and molecular oxygen is necessary. The electron holes shown in Eq. (viii) have high oxidation potential to produce $\text{OH}^\bullet$ (hydroxyl radicals) from the H_2O molecules/ OH^- adsorbed over the surface of semiconductors:

The generated e^- (electrons) can decrease dissolved oxygen and produce $\text{O}_2^{\bullet-}$ that is transformed into H_2O_2 (refer reactions 13–15):

H_2O_2 similarly acts like an electron (e^-) receptor producing additional $\text{HO}^\bullet$ ions:

Earlier studies reported that decomposition of ABs not only occurs due to $\text{OH}^\bullet$ via a similar process but also other species of radicals that are derived from oxygen itself. The efficiency of the entire AB removal mechanism is also influenced by pH, temperature, types of catalyst, and its dose, the intensity of radiation, and the composition of WW. Belhouchet et al.²⁴ investigated the applicability of SP techniques to remove the residue of tetracycline and found a wide range of removal efficiency 41.9–90.6% (at an initial concentration of 50 mg L^{-1}). Cao et al.²² also studied the removal of tetracycline by employing SP techniques and used a hybrid catalyst, which is made up of magnetic graphene oxide cerium-doped TiO_2 to accelerate the decomposition of the contaminant. Dimitrakopoulou et al.³⁰ investigated the efficiency of SP techniques for the removal of amoxicillin ABs and used different doses of TiO_2 catalyst where it was found that 250 mg L^{-1} was the optimum dose for maximum removal. Elimination of tetracycline was also attempted and inferred by several studies^{30,39}. The SP technique is a favorable technology for the removal of ABs at a laboratory scale from several decades²⁰. The SP technique is not feasible for industries because it is hard to pass radiation through liquid waste, which carries fine colloidal particles in suspensions, and challenging to eliminate the used catalyst after treatment.

Membrane techniques

Membrane techniques come under the physical treatment process. Water is passed through a semipermeable membrane with the help of mechanical forces (pressure is applied). This technique does not degrade the ABs, but retains them over the surface or transfers them into a new phase. The pores in the membranes vary from 0.001 to $0.02 \mu\text{m}$. Several membrane techniques are used to treat the WW, such as RO (forward, reverse, and backward), ultrafiltration, nanofiltration, and microfiltration. The RO and nanofiltration techniques as a tertiary

Fig. 5 Predominance of antibiotic removal techniques. Predominance of removal techniques applied for antibiotic removal exhibiting adsorption-based technique being the most effective and economical for the purging of antibiotics from water.

treatment step are very effective in treating the AB-contaminated water¹¹⁷, but the energy requirement is too high. Other associated problems include clogging due to mudball formation, fouling of the membrane, operation low mass flux, and high capital expenditure and operating expenditure. Liu et al.¹¹⁸ studied the performance of a hybrid carbon membrane (graphene oxide and activated carbon) for removing tetracycline ABs from water and reported 98.9% removal with 414 m² g⁻¹ of surface area. They concluded that the hybrid membrane process is more effective than a single membrane. Acero et al.¹¹⁷ worked on the treatment of flumequine and sulfamethoxazole using nanofiltration and ultrafiltration. They achieved >70% AB removal efficiency using nanofiltration, which was higher than ultrafiltration.

Procuring prospective

Many techniques are employed to remove ABs from WW to prevent hazardous impacts on human health and the environment. Figure 5 gives an idea about the published research on the removal of ABs and highlights the most studied ABs. Based on this review, it can be concluded that tetracyclines and quinolones are the most studied AB classes. According to AB data (refer to Fig. 1), penicillin and tetracycline are the most consumed ABs in the selected countries from both the developed and developing world. In developing countries, penicillin and quinolones are found in the environment more frequently^{18,21,41,44,119,120}. The published research proves that sorption techniques for the removal of ABs are promising and economical as they use various waste materials to produce low-cost adsorbents. These processes are capable of treating low as well as high concentrations of AB-rich WW. The sludge can be treated by incineration. The NaOH-activated carbon made from macadamia nut shell¹²¹ is reported to be the most efficient (100% removal of tetracycline) adsorbent. The minerals like montmorillonite, rectorite, and illite also have excellent AB removal efficiency (95%)¹⁰⁰. It can be concluded from exiting literature that there is a need to check the feasibility of the developed biochar treatment process and to have an emphasis on the pilot-scale model.

This review gives a holistic idea about the AB consumption, sources of AB contaminants, contaminant cycle, effects, and removal techniques. It also highlights the following key issue: there is a significant literature gap existing in the understanding of fate, persistence, and transportation of by-products after the treatment of ABs. An advanced protocol must be developed to conduct the ecotoxicity test and evaluate its adverse effects on the environment. There is a need for developing predictive models to forecast the fate, persistence, pathways, and behavior of ABs in the aqueous systems.

DISCUSSION

In the past two decades, the occurrence and fate of AB contaminants in the environment have drawn exclusive attention of the scientific community and policymakers. The residues of ABs found in the different environmental matrices lie in a wide range of concentrations (ng L⁻¹ to mg L⁻¹). The concentration in the aquatic environment is increasing day by day since the application of ABs is extensive in humans as well as in veterinary sectors. ABs are enduring and have high resistance to decomposition, but are deposited in different environmental matrices. AB residues are harmful to humans, aquatic life, and the ecosystem even at low levels (i.e., at very low concentration). Due to this, several techniques for AB removal come into the picture to resolve the problem of AB contamination effectively.

Treatment units used in conventional water and WWTPs fail to remove the AB contaminants, and there is a dire need to develop new efficient and economical techniques. In recent years, SPs and ozonation techniques are being widely used and tested. Ozonation techniques have an advantage that they can effectively work even though there is fluctuation in the flow rate and composition of WW. However, the disadvantages include high cost and energy requirements. Many studies have shown that adsorption techniques are an excellent alternative to oxidation techniques. Adsorption is the most efficient for the removal of ABs (>50% efficiency). The most efficient (100% removal of tetracycline) adsorbent reported in the literature is NaOH-activated carbon, made from macadamia nut shell¹²¹, followed by KOH-activated carbon prepared from lebeck seed pod (98.13%) removal^{92,122–178}. Minerals like montmorillonite, rectorite, and illite also have excellent AB removal efficiency (95%)¹⁰⁰. Most of the studies have shown that sorption processes followed the Langmuir model, which revealed that the single topmost layer effectively adsorbs the ABs on the surface of the adsorbent. The literature review infers that most of the adsorption kinetic studies followed the pseudo-second-order model. It has been reported that ABs adsorbed on the surface of adsorbents depend on chemisorption. In contrast, the elimination from a suspension is due to physicochemical exchange between solid and liquid phases.

There is a gap in understanding the fate, persistence, and transportation of by-products after the treatment of ABs, which can be toxic and harmful to the environment. The concentration of ABs at less than the detectable limit in the water environment does not guarantee the complete removal of ABs and their by-products. Likewise, the AB contaminants sequester differently in the environment and show distinct ecotoxicological effects that cannot be simulated in a lab-scale experiment. Hence, there is a need to develop an advanced protocol to conduct the ecotoxicity test and evaluate its adverse effects on the environment with a suitable feasibility study.

Plenty of ABs occur in the aquatic environment, but are not reported due to the gaps in their scientific understanding and advancement of analytical equipment. Quantification of the concentrations of ABs through developing the advanced analytical equipment could help to realize the severity of the problem. Many laboratory-based removal techniques are found efficient, which can be extended to on-field applications. It would eventually help in the betterment of human health, along with other components of the ecosystem. Further, a combination of different treatment processes could be considered over a single treatment process or conventional methods to remove the ABs from the aqueous medium effectively. It is imperative to understand ABs' behavior, particularly their prevalence and persistence in the WWTPs, so that predictive models can be developed to forecast the source, fate, persistence, pathways, and behavior of ABs in the aqueous systems for their sound management. Research should also be taken into account for developing the prioritization strategy for

rational and advanced identification through various potential ecological markers of AB contamination.

DATA AVAILABILITY

All other relevant data are available from the corresponding authors on request.

Received: 10 April 2020; Accepted: 26 August 2020;

Published online: 16 October 2020

REFERENCES

- Berglund, B. et al. Efficient removal of antibiotics in surface-flow constructed wetlands, with no observed impact on ANTIBIOTIC resistance genes. *Sci. Tot. Environ.* **476**, 29–37 (2014).
- Kumar, M. et al. Concurrence of antibiotics resistant bacteria (ARB), viruses, pharmaceuticals and personal care products (PPCPs) in ambient waters of Guwahati, India: urban vulnerability and resilience perspective. *Sci. Tot. Environ.* **693**, 133–640 (2019).
- Kümmerer, K. Antibiotics in the aquatic environment—a review—part I. *Chemosphere* **75**, 417–434 (2009).
- Kümmerer, K. Antibiotics in the aquatic environment—a review—part II. *Chemosphere* **75**, 435–441 (2009).
- Chen, J. et al. Removal of antibiotics and antibiotic resistance genes in rural WW by an integrated constructed wetland. *Environ. Sci. Pol. Res.* **22**, 1794–1803 (2015).
- Kang, J., Zhou, L., Duan, X., Sun, H. & Wang, S. Catalytic degradation of antibiotics by metal-free catalysis over nitrogen-doped graphene. *Cat. Today* <https://doi.org/10.1016/j.cattod.2018.12.002> (2018).
- Mohapatra, S., Huang, C. H., Mukherji, S. & Padhye, L. P. Occurrence and fate of pharmaceuticals in WWTPs in India and comparison with a similar study in the United States. *Chemosphere* **159**, 526–535 (2016).
- Dong, H., Yuan, X., Wang, W. & Qiang, Z. Occurrence and removal of antibiotics in ecological and conventional WW treatment processes: a field study. *J. Environ. Manag.* **178**, 11–19 (2016).
- Ram, B. & Kumar, M. Correlation appraisal of ANTIBIOTIC resistance with fecal, metal and microplastic contamination in a tropical Indian river, lakes and sewage. *npj Clean Water* **3**, 1–12 (2020).
- Diwan, V. et al. Detection of antibiotics in hospital effluents in India. *Curr. Sci.* **97**, 1752–1755 (2009).
- Watkinson, A. J., Murby, E. J., Kolpin, D. W. & Costanzo, S. D. The occurrence of antibiotics in an urban watershed: from WW to drinking water. *Sci. Total Environ.* **407**, 2711–2723 (2009).
- Mahmood, A. R., Al-Haideri, H. H. & Hassan, F. M. Detection of antibiotics in drinking water treatment plants in Baghdad City. *Iraq. Adv. Public Health* **2019**, 7851354 (2019).
- Diwan, V. et al. Seasonal variations in water-quality, antibiotic residues, resistant bacteria and antibiotic resistance genes of *Escherichia coli* isolates from water and sediments of the Kshipra River in Central India. *Int. J. Environ. Res. Public Health* **15**, 1281–1312 (2018).
- Diwan, V. et al. A three-year follow-up study of antibiotic and metal residues, antibiotic resistance and resistance genes, focusing on Kshipra—a river associated with holy religious mass-bathing in India: protocol paper. *Int. J. Environ. Res. Public Health* **14**, 574–596 (2017).
- Kolpin, D. W. et al. Pharmaceuticals, hormones, and other organic WW contaminants in US streams, 1999–2000: a national reconnaissance. *Environ. Sci. Technol.* **36**, 1202–1211 (2002).
- Karthikeyan, K. G. & Meyer, M. T. Occurrence of antibiotics in WW treatment facilities in Wisconsin, USA. *Sci. Tot. Environ.* **361**, 196–207 (2006).
- Diwan, V., Lundborg, C. S. & Tamhankar, A. J. Seasonal and temporal variation in release of antibiotics in hospital wastewater: estimation using continuous and grab sampling. *PLoS ONE* **7**, 68715–68733 (2013).
- Larsson, D. J., de Pedro, C. & Paxeus, N. Effluent from drug manufactures contains extremely high levels of pharmaceuticals. *J. Hazard Mater.* **148**, 751–755 (2007).
- Balakrishna, K. et al. A review of the occurrence of pharmaceuticals and personal care products in Indian water bodies. *Ecotoxicol. Environ. Saf.* **137**, 113–120 (2017).
- Homem, V. & Santos, L. Degradation and removal methods of antibiotics from aqueous matrices—a review. *J. Environ. Manag.* **92**, 2304–2347 (2011).
- Prabhasankar, V. P. et al. Removal rates of antibiotics in four sewage treatment plants in South India. *Environ. Sci. Pol. Res.* **23**, 8679–8685 (2016).
- Cao, M. et al. Visible light activated photocatalytic degradation of tetracycline by a magnetically separable composite photocatalyst: graphene oxide/magnetite/cerium-doped titania. *J. Colloid Interface Sci.* **467**, 129–139 (2016).
- Leung, H. W. et al. Distribution, fate and risk assessment of antibiotics in sewage treatment plants in Hong Kong, South China. *Environ. Int.* **42**, 1–9 (2012).
- Belhouichet, N., Hamdi, B., Chenchouni, H. & Bessekhouad, Y. Photocatalytic degradation of tetracycline antibiotics using new calcite/titania nanocomposites. *J. Photochem. Photobiol. A* **372**, 196–205 (2019).
- Ay, F. & Kargi, F. Advanced oxidation of amoxicillin by Fenton's reagent treatment. *J. Hazard Mater.* **179**, 622–627 (2010).
- Mavronikola, C. et al. Mineralisation of the antibiotic amoxicillin in pure and surface waters by artificial UVA-and sunlight-induced Fenton oxidation. *J. Chem. Technol. Biotechnol.* **84**, 1211–1217 (2009).
- Trovó, A. G., Melo, S. A. S. & Nogueira, R. F. P. Photodegradation of the pharmaceuticals amoxicillin, bezafibrate and paracetamol by the photo-Fenton process—application to sewage treatment plant effluent. *J. Photochem. Photobiol. A* **198**, 215–220 (2008).
- Trovo, A. G. et al. Degradation of the antibiotic amoxicillin by photo-Fenton process—chemical and toxicological assessment. *Water Res.* **45**, 1394–1402 (2011).
- Trovó, A. G. et al. Degradation of sulfamethoxazole in water by solar photo-Fenton. Chemical and toxicological evaluation. *Water Res.* **43**, 3922–3931 (2009).
- Dimitrakopoulou, D. et al. Degradation, mineralization and antibiotic inactivation of amoxicillin by UV-A/TiO₂ photocatalysis. *J. Environ. Manag.* **98**, 168–174 (2012).
- Chen, F. et al. Enhanced photocatalytic degradation of tetracycline by AgI/BiVO₄ hetero junction under visible-light irradiation: mineralization efficiency and mechanism. *ACS Appl. Mater. Inter.* **8**, 32887–32900 (2016).
- Kakavandi, B., Bahari, N., Kalantary, R. R. & Fard, E. D. Enhanced sono-photocatalysis of tetracycline ANTIBIOTIC using TiO₂ decorated on magnetic activated carbon (MAC@T) coupled with US and UV: a new hybrid system. *Ultrason. Sonochem.* **55**, 75–85 (2019).
- Zhang, G., Ji, S. & Xi, B. Feasibility study of treatment of amoxicillin wastewater with a combination of extraction, Fenton oxidation and reverse osmosis. *Desalination* **196**, 32–42 (2006).
- Weichgrebe, D. et al. Electro-chemical oxidation of drug residues in water by the example of tetracycline, gentamicin and aspirin. *Water Sci. Technol.* **49**, 201–206 (2004).
- Chen, Y. et al. Adsorptive removal and adsorption kinetics of fluoroquinolone by nano-hydroxyapatite. *PLoS ONE* **10**, 0145–0167 (2015).
- Balarak, D., Mostafapour, F. K. & Azarpira, H. Adsorption kinetics and equilibrium of ciprofloxacin from aqueous solutions using *Corylus avellana* (Hazel nut) activated carbon. *J. Pharm. Res. Int.* **13**, 1–14 (2016).
- Balarak, D., Mostafapour, F. K. & Joghataei, A. Experimental and kinetic studies on penicillin G adsorption by *Lemna minor*. *J. Pharm. Res. Int.* **9**, 1–10 (2016).
- Yang, X., Xu, G., Yu, H. & Zhang, Z. Preparation of ferric-activated sludge-based adsorbent from biological sludge for tetracycline removal. *Bioresour. Technol.* **211**, 566–573 (2016).
- Chen, J. et al. Removal of antibiotics and antibiotic resistance genes from domestic sewage by constructed wetlands: effect of flow configuration and plant species. *Sci. Tot. Environ.* **571**, 974–982 (2016).
- Klein, E. Y. et al. Global increase and geographic convergence in antibiotic consumption between 2000 and 2015. *Proc. Natl. Acad. Sci. USA* **115**, E3463–E3470 (2018).
- Mutiya, P. K. & Mittal, A. K. Occurrences and fate of an antibiotics amoxicillin in extended aeration-based sewage treatment plant in Delhi, India: a case study of emerging pollutant. *Desalin. Water Treat.* **51**, 6158–6164 (2013).
- Iyanee, F. S. et al. Occurrence of antibiotics in river water: a case study of Vrisha bhavathi River near Bangalore, India. In *33rd Inter. Symp. on Halog. Pers. Org. Pol., DIOXIN 25–30* (Non-profit organization founded by the International Advisory Board of the International Symposium on Halogenated Persistent Organic Pollutants (POPs), 2013).
- Sharma, B. M. et al. Health and ecological risk assessment of emerging contaminants (pharmaceuticals, personal care products, and artificial sweeteners) in surface and groundwater (drinking water) in the Ganges River Basin, India. *Sci. Tot. Environ.* **646**, 1459–1467 (2019).
- Gothwal, R. & Thatikonda, S. Role of environmental pollution in prevalence of antibiotic resistant bacteria in aquatic environment of river: case of Musi river, South India. *Water Environ. J.* **31**, 456–462 (2015).
- Costanzo, S. D., Murby, J. & Bates, J. Ecosystem response to antibiotics entering the aquatic environment. *Mar. Pollut. Bull.* **51**, 218–223 (2005).
- Kotzerke, A. et al. Alterations in soil microbial activity and N-transformation processes due to sulfadiazine loads in pig-manure. *Environ. Pollut.* **153**, 315–322 (2008).

47. Liu, H., Zhang, G., Liu, C. Q., Li, L. & Xiang, M. The occurrence of chloramphenicol and tetracyclines in municipal sewage and the Nanming River, Guiyang City, China. *J. Environ. Monit.* **11**, 1199–1205 (2009).
48. Underwood, J. C. et al. Effects of the antimicrobial sulfamethoxazole on groundwater bacterial enrichment. *Environ. Sci. Technol.* **45**, 3096–3101 (2011).
49. Su, H. C. et al. Persistence of antibiotic resistance genes and bacterial community changes in drinking water treatment system: from drinking water source to tap water. *Sci. Tot. Environ.* **616**, 453–461 (2018).
50. Tao, C. W. et al. Evaluation of five antibiotic resistance genes in wastewater treatment systems of swine farms by real-time PCR. *Sci. Tot. Environ.* **496**, 116–121 (2014).
51. Pruden, A., Pei, R., Storteboom, H. & Carlson, K. H. Antibiotic resistance genes as emerging contaminants: studies in northern Colorado. *Environ. Sci. Technol.* **40**, 7445–7450 (2006).
52. Chen, Y. Y. et al. Aqueous tetracycline degradation by H₂O₂ alone: removal and transformation pathway. *Chem. Eng. J.* **307**, 15–23 (2017).
53. Rigos, G., Nengas, I., Alexis, M. & Troisi, G. M. Potential drug (oxytetracycline and oxolinic acid) pollution from Mediterranean sparid fish farms. *Aquat. Toxicol.* **69**, 281–288 (2004).
54. French, G. L. Bactericidal agents in the treatment of MRSA infections—the potential role of daptomycin. *J. Antimicrob. Chemother.* **58**, 1107–1117 (2006).
55. Pallecchi, L., Bartoloni, A., Paradisi, F. & Rossolini, G. M. Antibiotics resistance in the absence of antimicrobial use: mechanisms and implications. *Expert Rev. Anti Infect. Ther.* **6**, 725–732 (2008).
56. Pauwels, B. & Verstraete, W. The treatment of hospital WW: an appraisal. *J. Water Health* **4**, 405–416 (2006).
57. Roose-Amsaleg, C. & Laverman, A. M. Do antibiotics have environmental side-effects? Impact of synthetic antibiotics on biogeochemical processes. *Environ. Sci. Pollut. Res.* **23**, 4000–4012 (2016).
58. Grenni, P., Ancona, V. & Caracciolo, A. B. Ecological effects of antibiotics on natural ecosystems: a review. *Microchem. J.* **136**, 25–39 (2018).
59. Hou, L. et al. Effects of sulfamethazine on denitrification and the associated N₂O release in estuarine and coastal sediments. *Environ. Sci. Technol.* **49**, 326–333 (2014).
60. Awad, Y. M. et al. Sulphamethazine in poultry manure changes carbon and nitrogen mineralisation in soils. *Chem. Ecol.* **32**, 899–918 (2016).
61. Heuer, H. et al. Fate of sulfadiazine administered to pigs and its quantitative effect on the dynamics of bacterial resistance genes in manure and manured soil. *Soil Biol. Biochem.* **40**, 1892–1900 (2008).
62. Tomlinson, T. G., Boon, A. G. & Trotman, C. N. A. Inhibition of nitrification in the activated sludge process of sewage disposal. *J. Appl. Bacteriol.* **29**, 266–291 (1966).
63. Xu, Y. et al. The combined effect of sulfadiazine and copper on soil microbial activity and community structure. *Ecotoxicol. Environ. Saf.* **134**, 43–52 (2016).
64. Bautitz, I. R. & Nogueira, R. F. P. Degradation of tetracycline by photo-Fenton process—solar irradiation and matrix effects. *J. Photochem. Photobiol. A* **187**, 33–39 (2007).
65. Kim, S. & Aga, D. S. Potential ecological and human health impacts of antibiotics and antibiotic-resistant bacteria from wastewater treatment plants. *J. Toxicol. Environ. Health B* **10**, 559–573 (2007).
66. Massey, L. B. et al. Antibiotic fate and transport in three effluent-dominated Ozark streams. *Ecol. Eng.* **36**, 930–938 (2010).
67. Calamari, D. et al. Strategic survey of therapeutic drugs in the rivers Po and Lambro in northern Italy. *Environ. Sci. Technol.* **37**, 1241–1248 (2003).
68. Mompelat, S., Le Bot, B. & Thomas, O. Occurrence and fate of pharmaceutical products and by-products, from resource to drinking water. *Environ. Int.* **35**, 803–814 (2009).
69. Diaz-Cruz, M. S., de Alda, M. J. L. & Barcelo, D. Environmental behavior and analysis of veterinary and human drugs in soils, sediments and sludge. *TrAC Trends Anal. Chem.* **22**, 340–351 (2003).
70. Chen, C. et al. Occurrence of antibiotics and antibiotic resistances in soils from wastewater irrigation areas in Beijing and Tianjin, China. *Environ. Pollut.* **193**, 94–101 (2014).
71. Shao, L. et al. Embodied energy assessment for ecological wastewater treatment by a constructed wetland. *Ecol. Model.* **252**, 63–71 (2013).
72. Subedi, B. et al. Mass loading and removal of pharmaceuticals and personal care products, including psychoactive and illicit drugs and artificial sweeteners, in five sewage treatment plants in India. *J. Environ. Chem. Eng.* **3**, 2882–2891 (2015).
73. Arkan, O. A. Degradation and metabolization of chlortetracycline during the anaerobic digestion of manure from medicated calves. *J. Hazard Mater.* **158**, 485–490 (2008).
74. Jelic, A. et al. Occurrence, partition and removal of pharmaceuticals in sewage water and sludge during wastewater treatment. *Water Res.* **45**, 1165–1176 (2011).
75. Osorio, V. et al. Simultaneous determination of diclofenac, its human metabolites and microbial nitration/nitrosation transformation products in wastewaters by liquid chromatography/quadrupole-linear ion trap mass spectrometry. *J. Chromatogr. A* **1347**, 63–71 (2014).
76. Zonja, B., Pérez, S. & Barceló, D. Human metabolite lamotrigine-N-2-glucuronide is the principal source of lamotrigine-derived compounds in wastewater treatment plants and surface water. *Environ. Sci. Technol.* **50**, 154–164 (2016).
77. Göbel, A., McArdell, C. S., Joss, A., Siegrist, H. & Giger, W. Fate of sulfonamides, macrolides, and trimethoprim in different wastewater treatment technologies. *Sci. Tot. Environ.* **372**, 361–371 (2007).
78. Vieno, N., Tuhkanen, T. & Kronberg, L. Elimination of pharmaceuticals in sewage treatment plants in Finland. *Water Res.* **41**, 1001–1012 (2007).
79. Gao, L., Shi, Y., Li, W., Liu, J. & Cai, Y. Occurrence, distribution and bioaccumulation of antibiotics in the Haihe River in China. *J. Environ. Monit.* **14**, 1247–1254 (2012).
80. Gao, Y. et al. Adsorption and removal of tetracycline antibiotics from aqueous solution by graphene oxide. *J. Colloid Interface Sci.* **368**, 540–546 (2012).
81. Cizmas, L., Sharma, V. K., Gray, C. M. & McDonald, T. J. Pharmaceuticals and personal care products in waters: occurrence, toxicity, and risk. *Environ. Chem. Lett.* **13**, 381–394 (2015).
82. Hu, Y. et al. Occurrence and reduction of antibiotic resistance genes in conventional and advanced drinking water treatment processes. *Sci. Tot. Environ.* **669**, 777–784 (2019).
83. de Oliveira Carvalho, C. et al. Kinetic, equilibrium, and thermodynamic studies on the adsorption of ciprofloxacin by activated carbon produced from *Jerivá* (*Syagrus romanzoffiana*). *Environ. Sci. Pollut. Res.* **26**, 4690–4702 (2019).
84. Premarathna, K. S. D. et al. Biochar-based engineered composites for sorptive decontamination of water: a review. *Chem. Eng. J.* **372**, 536–550 (2019).
85. Ahmad, M. et al. Trichloroethylene adsorption by pine needle biochars produced at various pyrolysis temperatures. *Biores. Technol.* **143**, 615–622 (2013).
86. Estevinho, B. N., Martins, I., Ratola, N., Alves, A. & Santos, L. Removal of 2, 4-dichlorophenol and pentachlorophenol from waters by sorption using coal fly ash from a Portuguese thermal power plant. *J. Hazard Mater.* **143**, 535–540 (2007).
87. Crisafulli, R. Removal of some polycyclic aromatic hydrocarbons from petrochemical wastewater using low-cost adsorbents of natural origin. *Biores. Technol.* **99**, 4515–4519 (2008).
88. Jang, H. M., Yoo, S., Choi, Y. K., Park, S. & Kan, E. Adsorption isotherm, kinetic modeling and mechanism of tetracycline on *Pinus taeda*-derived activated biochar. *Biores. Technol.* **259**, 24–31 (2018).
89. Jang, H. M. et al. Engineered biochar from pine wood: characterization and potential application for removal of sulfamethoxazole in water. *Environ. Eng. Res.* **24**, 608–617 (2018).
90. Zhou, Y. et al. Modification of biochar derived from sawdust and its application in removal of tetracycline and copper from aqueous solution: adsorption mechanism and modelling. *Biores. Technol.* **245**, 266–273 (2017).
91. Chen, T. et al. Sorption of tetracycline on H₃PO₄ modified biochar derived from rice straw and swine manure. *Biores. Technol.* **267**, 431–437 (2018).
92. Xie, H., Liu, W., Zhang, J., Zhang, C. & Ren, L. Sorption of norfloxacin from aqueous solutions by activated carbon developed from *Trapa natans* husk. *Sci. China Chem.* **54**, 835–843 (2011).
93. Ahmed, M. J. & Theydan, S. K. Fluoroquinolones ANTIBIOTICS adsorption onto microporous activated carbon from lignocellulosic biomass by microwave pyrolysis. *J. Taiwan Inst. Chem. Eng.* **45**, 219–226 (2014).
94. Pehlivan, E. & Altun, T. Biosorption of chromium (VI) ion from aqueous solutions using walnut, hazelnut and almond shell. *J. Hazard Mater.* **155**, 378–384 (2008).
95. Fan, H. T., Shi, L. Q., Shen, H., Chen, X. & Xie, K. P. Equilibrium, isotherm, kinetic and thermodynamic studies for removal of tetracycline antibiotics by adsorption onto hazelnut shell derived activated carbons from aqueous media. *RSC Adv.* **6**, 109983–109991 (2016).
96. Zhang, C. L., Qiao, G. L., Zhao, F. & Wang, Y. Thermodynamic and kinetic parameters of ciprofloxacin adsorption onto modified coal fly ash from aqueous solution. *J. Mol. Liq.* **163**, 53–56 (2011).
97. Paredes-Laverde, M., Silva-Agredo, J. & Torres-Palma, R. A. Removal of norfloxacin in deionized, municipal water and urine using rice (*Oryza sativa*) and coffee (*Coffea arabica*) husk wastes as natural adsorbents. *J. Environ. Manag.* **213**, 98–108 (2018).
98. Ahsan, M. A. et al. Adsorptive removal of methylene blue, tetracycline and Cr (VI) from water using sulfonated tea waste. *Environ. Technol. Innov.* **11**, 23–40 (2018).
99. Aksu, Z. & Tunç, Ö. Application of biosorption for penicillin G removal: comparison with activated carbon. *Proc. Biochem.* **40**, 831–847 (2005).
100. Wang, C. J., Li, Z. & Jiang, W. T. Adsorption of ciprofloxacin on 2:1 dioctahedral clay minerals. *Appl. Clay Sci.* **53**, 723–728 (2011).
101. Putra, E. K., Pranowo, R., Sunarso, J., Indraswati, N. & Ismadji, S. Performance of activated carbon and bentonite for adsorption of amoxicillin from WW: mechanisms, isotherms and kinetics. *Water Res.* **43**, 2419–2430 (2009).

102. Xiong, W. et al. Adsorption of tetracycline ANTIBIOTICS from aqueous solutions on nanocomposite multi-walled carbon nanotube functionalized MIL-53 (Fe) as new adsorbent. *Sci. Tot. Environ.* **627**, 235–244 (2018).
103. Xiong, W. et al. Ni-doped MIL-53 (Fe) nanoparticles for optimized doxycycline removal by using response surface methodology from aqueous solution. *Chemosphere* **232**, 186–194 (2019).
104. Xiong, W. et al. Multi-walled carbon nanotube/amino-functionalized MIL-53 (Fe) composites: remarkable adsorptive removal of antibiotics from aqueous solutions. *Chemosphere* **210**, 1061–1069 (2018).
105. Shi, S., Fan, Y. & Huang, Y. Facile low temperature hydrothermal synthesis of magnetic mesoporous carbon nanocomposite for adsorption removal of ciprofloxacin antibiotics. *Ind. Eng. Chem. Res.* **52**, 2604–2612 (2013).
106. Pouretedal, H. R. & Sadegh, N. Effective removal of amoxicillin, cephalixin, tetracycline and penicillin G from aqueous solutions using activated carbon nanoparticles prepared from vine wood. *J. Water Proc. Eng.* **1**, 64–73 (2014).
107. Klavarioti, M., Mantzavinos, D. & Kassinos, D. Removal of residual pharmaceuticals from aqueous systems by advanced oxidation processes. *Environ. Int.* **35**, 402–417 (2009).
108. Ben, W., Qiang, Z., Pan, X. & Chen, M. Removal of veterinary antibiotics from sequencing batch reactor (SBR) pretreated swine WW by Fenton's reagent. *Water Res.* **43**, 4392–4402 (2009).
109. Iakovides, I. C. et al. Continuous ozonation of urban WW: removal of antibiotics, antibiotics-resistant *Escherichia coli* and ANTIBIOTIC resistance genes and phytotoxicity. *Water Res.* **159**, 333–347 (2019).
110. Von Gunten, U. Ozonation of drinking water: Part I. Oxidation kinetics and product formation. *Water Res.* **37**, 1443–1467 (2003).
111. Von Gunten, U. Ozonation of drinking water: Part II. Disinfection and by-product formation in presence of bromide, iodide or chlorine. *Water Res.* **37**, 1469–1487 (2003).
112. Ben, W., Qiang, Z., Pan, X. & Nie, Y. Degradation of veterinary antibiotics by ozone in swine WW pretreated with sequencing batch reactor. *J. Environ. Eng.* **138**, 272–277 (2011).
113. Alsager, O. A., Alnajrani, M. N., Abuelizz, H. A. & Aldaghmani, I. A. Removal of antibiotics from water and waste milk by ozonation: kinetics, by-products, and antimicrobial activity. *Ecotoxicol. Environ. Saf.* **158**, 114–122 (2018).
114. Balcioglu, I. A. & Ötger, M. Treatment of pharmaceutical wastewater containing antibiotics by O₃ and O₃/H₂O₂ processes. *Chemosphere* **50**, 85–95 (2003).
115. Li, G., Ben, W., Ye, H., Zhang, D. & Qiang, Z. Performance of ozonation and biological activated carbon in eliminating sulfonamides and sulfonamide-resistant bacteria: a pilot-scale study. *Chem. Eng. J.* **341**, 327–334 (2018).
116. Das, R. Application photocatalysis for treatment of industrial waste water—a short review. *Open Access Lib. J.* **60**, 69–76 (2014).
117. Acero, J. L., Benitez, F. J., Real, F. J. & Teva, F. Removal of emerging contaminants from secondary effluents by micellar-enhanced ultrafiltration. *Sep. Purif. Technol.* **181**, 123–131 (2017).
118. Liu, M. K. et al. Effective removal of tetracycline ANTIBIOTICS from water using hybrid carbon membranes. *Sci. Rep.* **7**, 43717 (2017).
119. Mutiyar, P. K. & Mittal, A. K. Risk assessment of ANTIBIOTIC residues in different water matrices in India: key issues and challenges. *Environ. Sci. Pollut. Res.* **21**, 7723–7736 (2014).
120. Gothwal, R. & Shashidhar, T. Antibiotic pollution in the environment: a review. *Clean Soil Air Water* **43**, 479–489 (2015).
121. Martins, A. C. et al. Removal of tetracycline by NaOH-activated carbon produced from macadamia nut shells: kinetic and equilibrium studies. *Chem. Eng. J.* **260**, 291–299 (2015).
122. Ahmed, M. J. Adsorption of quinolone, tetracycline, and penicillin antibiotics from aqueous solution using activated carbons. *Environ. Toxicol. Pharmacol.* **50**, 1–10 (2017).
123. Matsuo, H., Sakamoto, H., Arizono, K. & Shinohara, R. Behavior of pharmaceuticals in waste water treatment plant in Japan. *Bull. Environ. Contam. Toxicol.* **87**, 31–35 (2011).
124. Watkinson, A. J., Murby, E. J. & Costanzo, S. D. Removal of antibiotics in conventional and advanced WW treatment: implications for environmental discharge and WW recycling. *Water Res.* **41**, 4164–4176 (2007).
125. Muñoz, I. et al. Bridging levels of pharmaceuticals in river water with biological community structure in the Llobregat river basin (northeast Spain). *Environ. Toxicol. Chem.* **28**, 2706–2714 (2009).
126. Karthikeyan, K. G. & Meyer, M. T. Occurrence of antibiotics in wastewater treatment facilities in Wisconsin, USA. *Sci. Tot. Environ.* **361**, 196–207 (2006).
127. Archana, G., Dhodapkar, R. & Kumar, A. Ecotoxicological risk assessment and seasonal variation of some pharmaceuticals and personal care products in the sewage treatment plant and surface water bodies (lakes). *Environ. Monit. Assess.* **189**, 446–465 (2017).
128. Chen, Y., Zhang, H., Luo, Y. & Song, J. Occurrence and dissipation of veterinary antibiotics in two typical swine wastewater treatment systems in east China. *Environ. Monit. Assess.* **184**, 2205–2217 (2012).
129. Kolpin, D. W., Skopek, M., Meyer, M. T., Furlong, E. T. & Zaugg, S. D. Urban contribution of pharmaceuticals and other organic WW contaminants to streams during differing flow conditions. *Sci. Tot. Environ.* **328**, 119–130 (2004).
130. Subedi, B., Balakrishna, K., Joshua, D. I. & Kannan, K. Mass loading and removal of pharmaceuticals and personal care products including psychoactive, anti-hypertensive, and antibiotics in two sewage treatment plants in southern India. *Chemosphere* **167**, 429–437 (2015).
131. Wang, Q. J. et al. Determination of four fluoroquinolone antibiotics in tap water in Guangzhou and Macao. *Environ. Pollut.* **158**, 2350–2358 (2010).
132. Valverde, R. S., Pérez, I. S., Franceschelli, F., Galera, M. M. & García, M. G. Determination of photoirradiated tetracyclines in water by high-performance liquid chromatography with chemiluminescence detection based reaction of rhodamine B with cerium (IV). *J. Chromatogr. A* **1167**, 85–94 (2007).
133. Akiba, M. et al. Impact of wastewater from different sources on the prevalence of antimicrobial-resistant *Escherichia coli* in sewage treatment plants in South India. *Ecotoxicol. Environ. Saf.* **115**, 203–208 (2015).
134. Osorio, V., Pérez, S., Ginebreda, A. & Barceló, D. Pharmaceuticals on a sewage impacted section of a Mediterranean River (Llobregat River, NE Spain) and their relationship with hydrological conditions. *Environ. Sci. Pollut. Res.* **19**, 1013–1025 (2012).
135. Hartig, C., Storm, T. & Jekel, M. Detection and identification of sulphonamide drugs in municipal waste water by liquid chromatography coupled with electrospray ionisation tandem mass spectrometry. *J. Chromatogr. A* **854**, 163–173 (1999).
136. Hirsch, R., Ternes, T., Haberer, K. & Kratz, K. L. Occurrence of antibiotics in the aquatic environment. *Sci. Tot. Environ.* **225**, 109–118 (1999).
137. Singh, K. P., Rai, P., Singh, A. K., Verma, P. & Gupta, S. Occurrence of pharmaceuticals in urban wastewater of north Indian cities and risk assessment. *Environ. Monit. Assess.* **186**, 6663–6682 (2014).
138. Mulroy, A. Monitoring and analysis of water and wastes. *Water Environ. Technol.* **13**, 32–36 (2001).
139. Miao, X. S., Bishay, F., Chen, M. & Metcalfe, C. D. Occurrence of antimicrobials in the final effluents of WW treatment plants in Canada. *Environ. Sci. Technol.* **38**, 3533–3541 (2004).
140. Fick, J. et al. Contamination of surface, ground, and drinking water from pharmaceutical production. *Environ. Toxicol. Chem.* **28**, 2522–2527 (2009).
141. Jha, R. R., Singh, N., Kumari, R. & Patel, D. K. Ultrasound-assisted emulsification microextraction based on a solidified floating organic droplet for the rapid determination of 19 antibiotics as environmental pollutants in hospital drainage and Gomti river water. *J. Sep. Sci.* **40**, 2694–2702 (2017).
142. Chen, H., Jing, L., Teng, Y. & Wang, J. Characterization of antibiotics in a large-scale river system of China: occurrence pattern, spatiotemporal distribution and environmental risks. *Sci. Tot. Environ.* **618**, 409–418 (2018).
143. Zhang, R. et al. Occurrence, distribution and ecological risks of fluoroquinolone antibiotics in the Dongjiang River and the Beijiang River, Pearl River Delta, South China. *Bull. Environ. Contam. Toxicol.* **99**, 46–53 (2017).
144. Yang, J. F. et al. Simultaneous determination of four classes of antibiotics in sediments of the Pearl Rivers using RRLC-MS/MS. *Sci. Tot. Environ.* **408**, 3424–3432 (2010).
145. Li, W., Gao, L., Shi, Y., Liu, J. & Cai, Y. Occurrence, distribution and risks of antibiotics in urban surface water in Beijing, China. *Environ. Sci. Proc. Impacts* **17**, 1611–1619 (2015).
146. Jiang, Y. et al. Distribution and ecological risk of antibiotics in a typical effluent-receiving river (Wangyang River) in north China. *Chemosphere* **112**, 267–274 (2014).
147. Mirzaei, R., Mesdaghinia, A., Hoseini, S. S. & Yunesian, M. Antibiotics in urban WW and rivers of Tehran, Iran: consumption, mass load, occurrence, and ecological risk. *Chemosphere* **221**, 55–66 (2019).
148. Deng, W. J., Li, N. & Ying, G. G. Antibiotic distribution, risk assessment, and microbial diversity in river water and sediment in Hong Kong. *Environ. Geochem. Health* **40**, 2191–2203 (2018).
149. Mokh, S., El Khatib, M., Koubar, M., Daher, Z. & Al Iskandarani, M. Innovative SPE-LC-MS/MS technique for the assessment of 63 pharmaceuticals and the detection of ANTIBIOTIC-resistant-bacteria: a case study natural water sources in Lebanon. *Sci. Tot. Environ.* **609**, 830–841 (2017).
150. Khan, G. A., Berglund, B., Khan, K. M., Lindgren, P. E. & Fick, J. Occurrence and abundance of antibiotics and resistance genes in rivers, canal and near drug formulation facilities—a study in Pakistan. *PLoS ONE* **8**, 62712 (2013).
151. Xu, Y. et al. Spatiotemporal profile of tetracycline and sulfonamide and their resistance on a catchment scale. *Environ. Pollut.* **241**, 1098–1105 (2018).
152. Wang, W., Wang, H., Zhang, W., Liang, H. & Gao, D. Occurrence, distribution, and risk assessment of antibiotics in the Songhua River in China. *Environ. Sci. Pollut. Res.* **24**, 19282–19292 (2017).
153. Wu, M. et al. Distribution, fate, and risk assessment of antibiotics in five WW treatment plants in Shanghai, China. *Environ. Sci. Pollut. Res.* **23**, 18055–18063 (2016).

154. Wu, M. H. et al. Occurrence, fate and interrelation of selected ANTIBIOTICS in sewage treatment plants and their receiving surface water. *Ecotoxicol. Environ. Saf.* **132**, 132–139 (2016).
155. Azanu, D., Styryshave, B., Darko, G., Weisser, J. J. & Abaidoo, R. C. Occurrence and risk assessment of antibiotics in water and lettuce in Ghana. *Sci. Tot. Environ.* **622**, 293–305 (2018).
156. Ngumba, E., Gachanja, A. & Tuhkanen, T. Occurrence of selected antibiotics and antiretroviral drugs in Nairobi River Basin. *Kenya Sci. Tot. Environ.* **539**, 206–213 (2016).
157. Hsu, J. T. et al. Prevalence of sulfonamide-resistant bacteria, resistance genes and integron-associated horizontal gene transfer in natural water bodies and soils adjacent to a swine feedlot in northern Taiwan. *J. Hazard Mater.* **277**, 34–43 (2014).
158. Suzuki, S., Ogo, M., Koike, T., Takada, H. & Newman, B. Sulfonamide and tetracycline resistance genes in total-and culturable-bacterial assemblages in South African aquatic environments. *Front. Microbiol.* **6**, 796–820 (2015).
159. Hammesfahr, U., Bierl, R. & Thiele-Bruhn, S. Combined effects of the antibiotics sulfadiazine and liquid manure on the soil microbial-community structure and functions. *J. Plant Nutr. Soil Sci.* **174**, 614–623 (2011).
160. Hammesfahr, U., Heuer, H., Manzke, B., Smalla, K. & Thiele-Bruhn, S. Impact of the ANTIBIOTIC sulfadiazine and pig manure on the microbial community structure in agricultural soils. *Soil Biol. Biochem.* **40**, 1583–1591 (2008).
161. Cui, H. et al. Influence of ciprofloxacin on microbial community structure and function in soils. *Biol. Fertil. Soils* **50**, 939–947 (2014).
162. Demoling, L. A., Bååth, E., Greve, G., Wouterse, M. & Schmitt, H. Effects of sulfamethoxazole on soil microbial communities after adding substrate. *Soil Biol. Biochem.* **41**, 840–848 (2009).
163. Binh, C. T. T. et al. Short-term effects of amoxicillin on bacterial communities in manured soil. *FEMS Microbiol. Ecol.* **62**, 290–302 (2007).
164. Li, D., Qi, R., Yang, M., Zhang, Y. & Yu, T. Bacterial community characteristics under long-term ANTIBIOTIC selection pressures. *Water Res.* **45**, 6063–6073 (2011).
165. Laverman, A. M. et al. Exposure to vancomycin causes a shift in the microbial community structure without affecting nitrate reduction rates in river sediments. *Environ. Sci. Pollut. Res.* **22**, 13702–13709 (2015).
166. Čermák, L. et al. Bacterial communities of two contrasting soils reacted differently to lincomycin treatment. *Appl. Soil Ecol.* **40**, 348–358 (2008).
167. Cordova-Kreylos, A. L. & Scow, K. M. Effects of ciprofloxacin on salt marsh sediment microbial communities. *ISME J.* **1**, 585–601 (2007).
168. Näslund, J., Hedman, J. E. & Agestrand, C. Effects of the antibiotic ciprofloxacin on the bacterial community structure and degradation of pyrene in marine sediment. *Aquat. Toxicol.* **90**, 223–227 (2008).
169. Müller, A. K., Westergaard, K., Christensen, S. & Sørensen, S. J. The diversity and function of soil microbial communities exposed to different disturbances. *Microb. Ecol.* **44**, 49–58 (2002).
170. Sengeløv, G. et al. Bacterial ANTIBIOTIC resistance levels in Danish farmland as a result of treatment with pig manure slurry. *Environ. Int.* **28**, 587–595 (2003).
171. Weber, K. P. et al. Effect of gold nanoparticles and ciprofloxacin on microbial catabolism: a community-based approach. *Environ. Toxicol. Chem.* **33**, 44–51 (2014).
172. Li, L., Zhao, J., Sun, Y., Yu, F. & Ma, J. Ionically cross-linked sodium alginate/k-carrageenan double-network gel beads with low-swelling, enhanced mechanical properties, and excellent adsorption performance. *Chem. Eng. J.* **372**, 1091–1103 (2019).
173. Gharbani, P., Mehrizad, A. & Jafarpour, I. Adsorption of penicillin by decaffeinated tea waste. *Pol. J. Chem. Technol.* **17**, 95–99 (2015).
174. Hu, A. et al. A green and low-cost strategy to synthesis of tunANTIBIOTIC pore sizes porous organic polymers derived from waste-expanded polystyrene for highly efficient removal of organic contaminants. *Chem. Eng. J.* **370**, 251–261 (2019).
175. Zhang, L., Wang, Y., Jin, S., Lu, Q. & Ji, J. Adsorption isotherm, kinetic and mechanism of expanded graphite for sulfadiazine antibiotics removal from aqueous solutions. *Environ. Technol.* **38**, 2629–2638 (2017).
176. Ötör, H. M. & Akmehtmet-Balcioglu, I. Adsorption and degradation of enrofloxacin, a veterinary ANTIBIOTIC on natural zeolite. *J. Hazard Mater.* **122**, 251–258 (2005).
177. De Velasco-Maldonado, P. S., Hernández-Montoya, V., Montes-Morán, M. A., Vázquez, N. A. R. & Pérez-Cruz, M. A. Surface modification of a natural zeolite by treatment with cold oxygen plasma: characterization and application in water treatment. *Appl. Surf. Sci.* **434**, 1193–1199 (2018).
178. Li, Y., Zhang, J. & Liu, H. Removal of chloramphenicol from aqueous solution using low-cost activated carbon prepared from *Typha orientalis*. *Water* **10**, 351–370 (2018).
179. Liu, P., Zhang, H., Feng, Y., Shen, C. & Yang, F. Integrating electro-chemical oxidation into forward osmosis process for removal of trace antibiotics in WW. *J. Hazard Mater.* **296**, 248–255 (2015).
180. Qiang, Z., Adams, C. & Surampalli, R. Determination of ozonation rate constants for lincomycin and spectinomycin. *Ozone Sci. Eng.* **26**, 525–537 (2004).
181. Andreozzi, R. et al. Lincomycin solar photodegradation, algal toxicity and removal from wastewaters by means of ozonation. *Water Res.* **40**, 630–638 (2006).
182. Arslan-Alaton, I. & Caglayan, A. E. Toxicity and biodegradability assessment of raw and ozonated procaine penicillin G formulation effluent. *Ecotoxicol. Environ. Saf.* **63**, 131–140 (2006).
183. Bui, X. T., Chen, S. S., Nguyen, P. D., Nguyen, T. T. & Nguyen, T. B. Hospital wastewater treatment by sponge membrane bioreactor coupled with ozonation process. *Chemosphere* **230**, 377–383 (2019).
184. Elmolla, E. S. & Chaudhuri, M. Degradation of amoxicillin, ampicillin and cloxacillin antibiotics in aqueous solution by the UV/ZnO photocatalytic process. *J. Hazard Mater.* **173**, 445–449 (2010).
185. Elmolla, E. S. & Chaudhuri, M. Photocatalytic degradation of amoxicillin, ampicillin and cloxacillin antibiotics in aqueous solution using UV/TiO₂ and UV/H₂O₂/TiO₂ photocatalysis. *Desalination* **252**, 46–52 (2010).
186. Giraldo, A. L. et al. Degradation of the antibiotic oxolinic acid by photocatalysis with TiO₂ in suspension. *Water Res.* **44**, 5158–5167 (2010).
187. Keen, O. S. & Linden, K. G. Degradation of antibiotic activity during UV/H₂O₂ advanced oxidation and photolysis in WW effluent. *Environ. Sci. Technol.* **47**, 13020–13030 (2013).
188. Chu, L., Zhuang, S. & Wang, J. Degradation kinetics and mechanism of penicillin G in aqueous matrices by ionizing radiation. *Radiat. Phys. Chem.* **145**, 34–38 (2018).

ACKNOWLEDGEMENTS

This work is supported by the UKIERI fund of DST. Support from IIT Gandhinagar and WET lab is also acknowledged. We highly appreciate the efforts made by Dr. Tom Van der Voorn, University of Osnabruck and Dr. Santanu Mukherjee, IIT Gandhinagar to review this paper and give valuable comments to improve the quality of the paper. We highly acknowledge the valuable contribution of the anonymous reviewers for improving the quality of this review article.

AUTHOR CONTRIBUTIONS

C.B.: writing the original draft and editing; M.K.: conceptualization, supervision, and editing; V.K.T.: reviewing and editing; P.K.M.: reviewing and editing.

COMPETING INTERESTS

The authors declare no competing interests.

ADDITIONAL INFORMATION

Supplementary information is available for this paper at <https://doi.org/10.1038/s41545-020-00087-x>.

Correspondence and requests for materials should be addressed to M.K.

Reprints and permission information is available at <http://www.nature.com/reprints>

Publisher's note Springer Nature remains neutral with regard to jurisdictional claims in published maps and institutional affiliations.

Open Access This article is licensed under a Creative Commons Attribution 4.0 International License, which permits use, sharing, adaptation, distribution and reproduction in any medium or format, as long as you give appropriate credit to the original author(s) and the source, provide a link to the Creative Commons license, and indicate if changes were made. The images or other third party material in this article are included in the article's Creative Commons license, unless indicated otherwise in a credit line to the material. If material is not included in the article's Creative Commons license and your intended use is not permitted by statutory regulation or exceeds the permitted use, you will need to obtain permission directly from the copyright holder. To view a copy of this license, visit <http://creativecommons.org/licenses/by/4.0/>.

© The Author(s) 2020