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Antibiotic residues in aquatic environment: ecological disruption, and biotechnological solutions for environmental safety

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World Health Organization reported on antibiotic resistance trends, antibiotic residue is one of the emerging pharmaceutical microcontaminants entering the aquatic ecosystem from various point and non-point sources, due to their continuous usage in human and veterinary medicine, and agriculture. The lack of proper wastewater treatment process gives rise to the contaminants entering the drinking and ground water. The constant input of the trace levels of residues over a period leads to long-term effects on both aquatic and terrestrial life due to their bioactivity, which will lead to the development and the spread of the Antibiotic Resistant- Bacteria (ARB) and Antibiotic Resistance Genes (ARGs) that leads to the disruption of the microbial communities. Due to their wide occurrence various treatment methods have been studied for their complete removal from the aquatic system. Numerous papers have been published on the occurrence of the antibiotic residue levels in various compartments of the aquatic system. This study aims to provide an insight on the recent occurrence trends (2004–2025) in various water bodies globally. The study briefly introduces the sources of various antibiotics followed by a compilation of occurrence of antibiotics in the aquatic system around the world in the past 2 decades. The study revealed that Quinolones, Macrolides, Tetracycline and Sulfonamides are the most abundantly found antibiotics in the aquatic system; with concentrations ranging from few ng/L to hundreds of μg/L.

KEYWORDS
antibiotic pollution, antimicrobial resistance, aquatic environment, environmental toxicology, wastewater contamination

1 Introduction

Antibiotics are being used in medicine and agriculture for a long period of time to treat against microbial infections and promote growth (McEwen and Fedroka-Cray, 2002). Antibiotics vary in metabolism; some are excreted largely unchanged (e.g., sulfonamides), while others undergo partial metabolism before excretion (Szymańska et al., 2019). Literature studies (De Liguoro et al., 2003; Winckler and Grafe, 2001) have shown that some antibiotics are excreted rapidly and more than 50% of the antibiotics were recovered in their active form. They are found in surface water (Li et al., 2021; Ding et al., 2017), Ground water (Kamatham et al., 2024; Murata et al., 2011), drinking water (Voigt et al., 2020), animal and anthropogenic wastewater (Jiang et al., 2013; Zhang et al., 2013; Lübbert et al., 2017). This has led to the antibiotics constantly entering different aquatic compartments at various ranging concentrations from ng/L to μg/L which depends on the usage pattern of the particular region (McArdell et al., 2003).

While the European Union (EU) banned the usage of antibiotics in feed as growth promoters, they are used widely as drugs and feed additives in various countries for prophylactic, metaphylactic, and therapeutic purposes (CEC, 1998a; CEC, 1998b; Hamscher et al., 2003). The entering of the antibiotics in the aquatic system, even in low concentration levels over a long period of time leads to the promotion of antibiotic-resistant bacteria due to mutation, which affects the human health (Martínez, 2008; Dantas et al., 2008). Among the various antibiotic classes, sulfonamides, tetracyclines, quinolones and macrolides are the most prevalently used antibiotics. The occurrence of antibiotics in the aquatic bodies is influenced by various factors like spatial, temporal, seasonal and flow conditions, etc. Antibiotic resistance is closely linked to several UN Sustainable Development Goals, particularly SDG 3 (health), SDG 6 (clean water), SDG 2 (food security), SDG 12 (responsible use), SDG 14 (aquatic life), and SDG 17 (global partnerships). Unlike previous reviews that mainly focused on isolated reports of antibiotic occurrence or specific environmental compartments, the present review provides a comprehensive and updated global synthesis of antibiotic contamination trends in aquatic environments from 2004 to 2025. The review integrates spatial, temporal, and seasonal distribution patterns of major antibiotic classes across wastewater, surface water, groundwater, and drinking water systems, while critically examining their ecological and public health implications. Antibiotic resistance is closely linked to several UN Sustainable Development Goals, particularly SDG 3 (health), SDG 6 (clean water), SDG 2 (food security), SDG 12 (responsible use), SDG 14 (aquatic life), and SDG 17 (global partnerships).

The World Health Organization released a major global report on antibiotic resistance trends in October 2025. It draws on data from over 23 million laboratory-confirmed bacterial infections reported by 104 countries, offering the most comprehensive surveillance data to date. One in six bacterial infections worldwide in 2023 were resistant to standard antibiotics. Resistance has increased significantly between 2018 and 2023 across more than 40% of antibiotic-pathogen combinations monitored by WHO. Resistance prevalence is highest in the South-East Asian and Eastern Mediterranean regions, where about one in three infections were resistant (Tosas Auguet et al., 2025).

Though antibiotics like Streptomycin and penicillin occur in nature (McManus et al., 2002; Bennett and Chung, 2001), the discharge of antibiotics in the aquatic environment due to anthropogenic activity is of primary concern. Agricultural and sewage run-off from various point and non-point sources acts as the major sources for antibiotics in the aquatic environment. They are not eliminated by the Wastewater Treatment Plants (WWTPs) due to their ability to resist degradation of different treatment processes and readily soluble nature leading to their wide spread along running water (Feng et al., 2013). The pseudo-persistent nature of the antibiotics is mainly determined by their chemical structure and various other factors.

The occurrence of antibiotics in aquatic environments has been widely studied, yet much of the existing literature is fragmented, often limited to specific regions, antibiotic classes, or individual environmental compartments. This review is necessary to provide a consolidated and updated overview (2004–2025) of the global distribution, sources, and concentrations of antibiotics across multiple aquatic systems, which is currently lacking in the field. Unlike earlier reviews that primarily catalogued detection data, this work synthesizes occurrence trends in relation to usage patterns, seasonal and spatial variations, and highlights antibiotic classes that frequently exceed ecotoxicologically relevant thresholds. By comparing reported concentrations with potential risks to aquatic ecosystems and human exposure pathways, this review fills a critical gap between raw monitoring data and its implications for environmental toxicology and public health. Researchers and practitioners can use the insights presented here not only to identify high-risk antibiotic classes and priority regions for monitoring but also to design laboratory studies and wastewater treatment strategies that target the most persistent and impactful contaminants.

2 Sources of antibiotics

The antibiotics administered in the human body undergoes Phase I and Phase II reaction to form readily excreted water-soluble polar metabolites (Sukul and Spiteller, 2007). The main concern faced is that though the drug is metabolized in the body, mainly by the liver, a part of the initial administered dose is excreted via the renal (urine) and biliary (faeces) either in its original form or as metabolites, or even as conjugates of glucuronic or sulphuric acid at around 10%–90% due to incomplete absorption or metabolism (Iglesias et al., 2013; Gros et al., 2010; Dinh et al., 2017). Antibiotics enter aquatic environments through extensive anthropogenic activities, including human medicine, veterinary applications, agriculture, and aquaculture. Although some antibiotics, such as streptomycin produced by *Streptomyces griseus*, occur naturally and are widely used for plant disease management (Waksman, 1953; McManus et al., 2002), their excessive and persistent use has become a major environmental concern. Every year, large quantities of antibiotic residues are discharged into water bodies through wastewater, runoff, and industrial effluents, leading to their accumulation in different aquatic compartments (Michael et al., 2013; Ji et al., 2012). This continuous contamination contributes to aquatic toxicity, ecosystem disruption, and the spread of antimicrobial resistance (AMR), posing significant risks to both environmental and human health. The incomplete treatment of the WWTPs leads to partial elimination and the residues are found in wastewaters, surface waters like streams, lakes, rivers, and in groundwater (Moldovan, 2006; Li et al., 2012; Lin et al., 2015; Brown et al., 2006) stated that the contribution of the antibiotics from the municipal wastewater is almost equal with the contribution from the hospital wastewater. The entry of the veterinary antibiotics into the aquatic environment can be attributed to their use in growth and treatment process followed in the feedlots added with the inappropriate disposal of used containers, unused medicine livestock feed and drug manufacturing (Bártíková et al., 2016; Watanabe et al., 2010; Brown et al., 2006; Beni et al., 2020) also

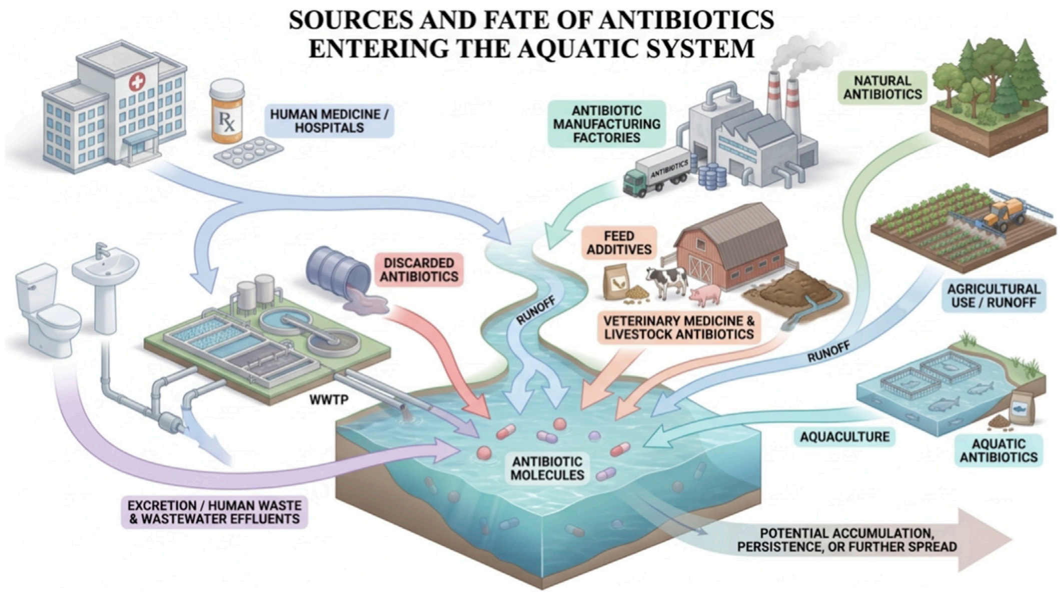


FIGURE 1
Sources and Fate of antibiotics entering the aquatic system (Image designed using Figure Labs).

stated that the hide processing facilities contribute to the presence of sulfonamides as the azo dye gets converted to sulfonamide compounds. In the study conducted by Watkinson et al. (2009), the presence of ionophores which are solely used in agriculture, is explained by the runoff from the meat processing plants and the abattoirs discharge. The compounds also tend to accumulate in the soils, sediments and tissues due to their highly adsorbable nature, and based on the physiochemical properties of both the soil and the antibiotics they are either retained in the soil like tetracycline and fluoroquinolones or are transported by runoff like sulfonamides (Burkhardt and Stamm, 2007; Thiele-Bruhn et al., 2004; Tolls, 2001). McEachran et al. (2015) also found that some antibiotics like Tylosin and Monensin were also transported via air and deposited in the soil and water samples (Figure 1).

3 Occurrence of antibiotics

The occurrence of antibiotics in various aquatic environment has been conducted in several countries over the years and an overview of the same was performed. The study from the literature showed that the concentration of antibiotics in the aquatic environment ranged from <1 ng/L in the surface water to millions of ng/L in the municipal wastewater. Various factors are known to affect the concentration of antibiotics in various aquatic environment like the usage pattern, consumption pattern, removal efficiency by the WWTPs, biotic and abiotic degradation, excretion rate, flow of water, the point and non-point origin sources, along with spatial and temporal conditions (Golet et al., 2003).

Tong et al. (2014) stated that the difference in the concentration can be due to the variations in dosage level followed during different breeding seasons along with their biogeochemical behavior seen in various aquatic phases. Antibiotic residue levels are usually seen less during the monsoon season due to the dilution of the water along with the flow of water, while increased concentrations are seen during winters which were at least twice than summers which can be due to the low removal efficiency and increased consumption which was with Karthikeyan and Meyer (2006) showing increased occurrence of trimethoprim and sulfamethoxazole in the WWTPs due to the two being commonly prescribed in the treatment of bacterial sinusitis (Mutiyar and Mittal, 2014; McArdell et al., 2003; Gao L. et al., 2012) also added that the increased concentration during winters to be due to the transfer of the antibiotics from the aqueous to the solid phase because of the reduced consumption of water. The low antibiotic residue seen during summer can be explained because of the enhanced bacterial activity and photodegradation due to increased temperature (Dinh et al., 2017) and because of the greater water flow seen during the summers (Murata et al., 2011). Rainwater plays a major role in the varied distribution and concentration of the antibiotics in different aquatic environments. The increased flow due to rainfall dilutes the antibiotic residues, but has the potential to lower the removal efficiency because of reduced Hydraulic Retention Time (HRT) in the WWTPs, concentrating them (Osorio et al., 2012; Sui et al., 2011; Luo et al., 2014) also stated that the rainfall can increase the antibiotic residue due to combined sewer overflow. The increased concentration of antibiotics during the monsoon season can also be attributed to the overloading of the WWTPs by the storm infiltration which in turn reduces the detention time leading to incomplete treatment (Prabhasankar et al., 2016).

Spatial distribution of the antibiotics varies with various factors. For example, Sulfonamides were absent in deep water sampling sites which creates an anoxic surrounding and at increased distance from the discharge point in a study conducted by Beni et al. (2020), which is explained by increased degradable behavior of sulfonamides in anoxic conditions and longer flow path (Bai et al., 2019). The concentration of the antibiotics has been shown to decrease with the increase of depth and distance (Aust et al., 2008, Watkinson et al., 2009; Dinh et al., 2017) showed that the concentration of the antibiotics decreased about 20%–55% from the point of discharge to further.

Li et al. (2012) stated that anthropogenic activity played a major role in the distribution of the antibiotics, with low levels found in the middle of the lakes with minimal disturbances while high concentrations were found near lakeshores and densely populated scenic spots with increased activity. This was also supported by McArdell et al. (2003), who showed that the antibiotic concentrations were higher on the WWTPs due to the increased commuters and hospital patients in a day wise study. He also attributed the increased concentration of erythromycin in the WWTP that receives from the international airport to the varied antibiotic consumption pattern followed in different countries.

Some antibiotics are to found in the wastewater effluents but not in the influents. For example, Ceftazidime, a cephalosporin antibiotic, was not detected in the influent sample but was found in the effluent sample. Their absence is explained by the cleaving of the metabolite to produce the original molecule during the treatment process (Kamatham et al., 2024; Prabhasankar et al., 2016). Similarly, concentrations sulfasalazine, sulfathiazole and Tylosin were higher in wastewater effluents than the influents (Watkinson et al., 2009).

3.1 Aminoglycosides

Aminoglycosides are extremely polar and basic in nature due to their amino sugar groups. Their presence in the aquatic environment is mostly a result of veterinary medicine than human consumption due to their toxic potential and adverse effects. In the aminoglycoside class studied, only amikacin was found in the groundwater, which is due to the incomplete removal by WWTPs. They existed in a range from 2.003 ng/L (Amikacin) in groundwater samples (Kamatham et al., 2024) to as high as 16,400 ng/L (Neomycin) in Wastewater influent (Tahrani et al., 2016).

3.2 Amphenicols

They are not frequently detected in the aquatic environment because of their high lipid solubility, basic characteristic, low consumption in human and veterinary medicine, with Chloramphenicol usage in meat producing animals being banned in the EU because of their carcinogenicity, and their association with the aplastic anemia and grey baby syndrome (Johnson, 2003). This is due to the reduced glucuronidation in the liver and reduced renal elimination of the drug. In a study conducted by da Silva et al. (2011), chloramphenicol was found in high concentrations in the suspended solids and was either below the limits of quantification of absent, indicating their high tendency to bind with the suspended solids. Amphenicols were seen in range from 11.58 ng/L in groundwater samples (Florfenicol, Li et al., 2021) to 18,400 ng/L (Florfenicol, Tahrani et al., 2016) in surface water. The concentrations of the amphenicols in the wastewater influents (3,300 ng/L; chloramphenicol and florfenicol, Tahrani et al., 2016) and surface water were higher than the hospital wastewater (36 ng/L; Chloramphenicol, Verlicchi et al., 2012), indicating their non usage in human medicine with usage in households.

3.3 Diaminopyrimidines

In this antibiotic class, only trimethoprim was found in almost all compartments of aquatic environment with high detection frequency and concentrations. This is mostly used in combination with sulfamethoxazole (co-trimoxazole) since the 1960s especially against the Gram-negative bacterial infections (Goldberg and Bishara, 2012) leading to their co-occurrence. They are found in concentrations as low as 5.3 ng/L in surface water (Kim et al., 2007) to 22,400 ng/L in Hospital wastewater (Lien et al., 2016). High concentrations were detected in the hospital wastewater and dairy wastewater indicating their intensive use in the human and veterinary medicine. Trimethroprim showed resistance to biodegradation and their removal through the WWTPs was reported to be negligible (Tamtam et al., 2008; Lindberg et al., 2005).

3.4 Imidazole

Despite their reduced consumption in both human and animal medicine due to their carcinogenicity and mutagenicity, the presence of imidazole antibiotics at various concentrations were observed in the aquatic environment. The antibiotic that was most detected in this group is metronidazole, a moderately lipophilic weak base, where 20% of the dose is excreted in their original form (Lau et al., 1992; Lamp et al., 1999). The concentrations of imidazole ranged from 1.79 ng/L (Albendazole; Santos et al., 2013) to 763,610.90 ng/L (Albendazole; Faleye et al., 2019) in the wastewater influents. Metronidazole concentrations ranged from 23 ng/L (Bisognin et al., 2021) to 164,264.40 ng/L (Faleye et al., 2019) in wastewater influents.

3.5 Lincosamides

Lincosamides enter the aquatic environment through various pathways like release of treated animal wastewater, excretion, liquid manure, intensive agriculture. They are not commonly detected in the aquatic environment, with high concentrations seen in surface water which can be attributed to their high sorption potential and limited solubility in the water (Mehrtens et al., 2021). They are found in concentrations ranging from 3.6 ng/L (Lincomycin; Brown et al., 2015) in wastewater effluent to 21,100 ng/L (Lincomycin; Boxall et al., 2006) in surface waters. Clindamycin was also observed in drinking water in Germany at 340 ng/L (Voigt et al., 2020).

3.6 Macrolides and glycopeptides

Compared to other antibiotic classes, macrolides are generally detected at lower concentrations than quinolones; however, exceptionally high peaks (up to 27,700 ng/L) have been reported. This pattern is attributed to their relatively low stability and strong sorption tendency to sludge through cation exchange, since under typical wastewater conditions macrolides are positively charged due to protonation of the dimethyl amino group, while sludge surfaces carry a negative charge (Michael et al., 2013). Macrolides are detected in quantity as low as 0.9 ng/L (Josthromycin; Li et al., 2012) in surface water to 27,700 ng/L (clarithromycin; Lübbert et al., 2017) in surface water. High detection frequency was seen in the hospital wastewater with concentrations ranging from 12 ng/L

(Josthromycin) to 14,000 ng/L (clarithromycin (Verlicchi et al., 2012). They were also detected in drinking and groundwater samples at maximum concentrations of 600 ng/L (clarithromycin; Voigt et al., 2020) and 54.8 ng/L (erythromycin-H2O; Lin et al., 2015), which poses a critical threat of direct exposure for human consumption.

In glycopeptides, only vancomycin was observed in the wastewater influent and surface samples at concentrations of 41 ng/L and 11.69 ng/L, respectively in Italy (Zuccato et al., 2010), which is because of their rare usage pattern seen only during life threatening Gram-positive bacterial infections, especially against methicillin-resistant *Staphylococcus aureus*.

3.7 Quinolones

Quinolones are primarily used in human medicine, but recently they are also used in veterinary medicine and aquaculture. They are found in almost all compartments of the aquatic environment due to the high consumption and hydrophilic nature and fat solubility. The low concentration of fluoroquinolones in the surface water can be due to photolytic degradation (Belden et al., 2007). Little to no changes are observed in the Fluoroquinolones during the transport to the WWTPs, with 1/3 of it removed during the mechanical treatment which is due to removal of suspended solids, and majority removed during the biological treatment process (Golet et al., 2003). They are seen in concentrations ranging from 1.6 ng/L in groundwater samples (Ciprofloxacin; Lin et al., 2015) to 332,155,000 ng/L (Ciprofloxacin; Zafar et al., 2021) in municipal wastewater. The overview of the studies showed that, of the quinolones, the highest detection frequency was seen as Ciprofloxacin > Ofloxacin > Norfloxacin > Enrofloxacin. The highest concentrations were observed in hospital and municipal wastewater indicating their usage pattern in human medicine. Though they are present in high concentrations in the wastewater influents, in the current study they were not observed in drinking water, which can be due to the >85% removal within the WWTPs.

3.8 Sulfonamides

These are widely used in the feedlots of livestock derived from p-amino-benzene-sulfonamide. They are found in concentrations as low as 0.02 ng/L (Sulfathiazole; Murata et al., 2011) in surface water to 16,009,000 ng/L (sulfamethoxazole; Zafar et al., 2021) in municipal wastewater. The high levels of sulfamethoxazole in this study were observed in swine, hospital and dairy wastewater, indicating their high usage in livestock agriculture. Murata et al. (2011), in his study, stated that the concentration of sulfamethazine was high in sites of pig farm wastewater and Sulfamethoxazole being high in livestock wastewater indicating their usage pattern in animal husbandry. Of the sulfonamides, sulfamethoxazole was observed the most followed by Sulfamethazine > sulfathiazole > sulfadiazine. Sulfamethoxazole is found to be detected in all compartments of the aquatic environment including drinking water and groundwater at concentrations 400 ng/L and 0.34 ng/L, respectively (Voigt et al., 2020; Murata et al., 2011).

3.9 Tetracyclines

Tetracyclines are seen throughout the year in WWTPs except during early winter samples (Karthikeyan and Meyer, 2006). Though the usage of tetracyclines is high in both medicine and agriculture purposes, they are detected in low concentrations in the aquatic environment. This is due to their potential to bind to Calcium and similar divalent cations forming stable complex making them bound to suspended matter and sediments (Hirsch et al., 1999; Bisognin et al., 2021). Tetracyclines are seen in concentrations ranging from 1.82 ng/L (Oxytetracycline; Zuccato et al., 2010) to 48,000 ng/L (Tetracycline; Karthikeyan and Meyer, 2006) in surface water and wastewater influent, respectively. Concentrations above 1,000 ng/L are seen mostly in the Swine, cattle and hospital wastewater. In the samples collected from fish ponds near pearl culture farm and pig farms, tetracyclines are found in high concentrations (Tong et al., 2014). Table 1 presents the chemical structure and name associated with the antibiotics.

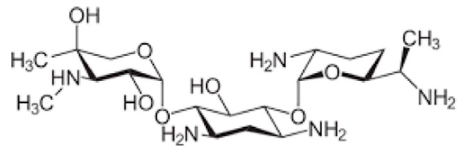
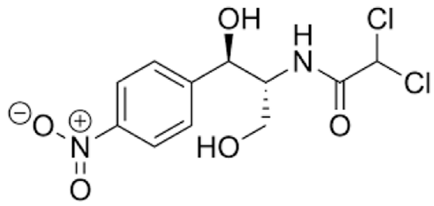
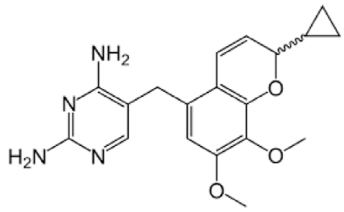
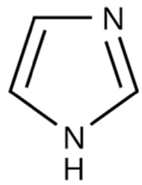
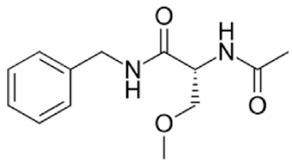
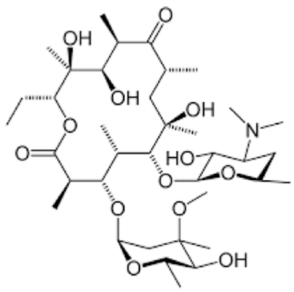
3.10 β-lactams (cephalosporins, carbapenem)

They include a wide range of antibiotics which contain at least one beta-lactam ring. Despite their high consumption the concentration and persistence are expected to be low due to their high chemical instability making them unstable in the environmental conditions. They are known to be removed through hydrolytic cleavage and their further mineralization to CO_2 and water by biotic (biological and enzymatic) and abiotic (chemical, thermal, presence of metal ions like Zn, Cu, Cd, pH, oxidizing and reducing agents, nucleophiles, polar solvents like water and alcohol) with degradation occurring over several weeks in the surface waters which can be attributed to their persistence in the aquatic environment (Watkinson et al., 2009; Mitchell et al., 2014; Deshpande et al., 2004). The concentrations of β-lactams range from 2.414 ng/L (Tazobactum; Kamatham et al., 2024) to 32,570,000 ng/L (Ampicillin; Zafar et al., 2021) in ground water and municipal wastewater respectively. The high concentration of β-lactams are seen in the municipal wastewater which indicate their high usage in households. Amoxycillin and ampicillin were the frequently detected β-lactam antibiotics in the study conducted, with highest concentrations seen at 6,940 ng/L for amoxycillin in wastewater influents.

Due to their wider activity, less toxicity and better stability than the penicillin lactams, the usage patterns of cephalosporin have increased in both human and veterinary medicine, with their usage being second or third-line therapy. Also due to their low Kow values they will hardly be eliminated through biotransformation and soil sorption and are not readily biodegradable (Ribeiro et al., 2018). Of this class, cephalexin was the most frequently detected antibiotic in various compartments of the aquatic environment ranging from 20.5 ng/L (Jiang et al., 2013) to 64,000 ng/L (Watkinson et al., 2009) in the surface water and wastewater influents, respectively. Cephalosporins were found in the aquatic environments at 2.739 ng/L (Ceftazidime; Kamatham et al., 2024) to 64,000 ng/L in cephalexin.

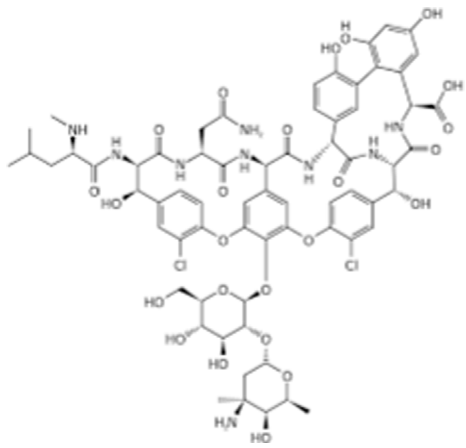
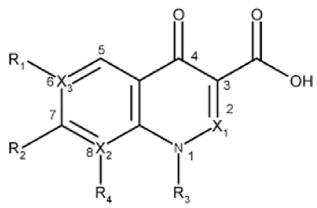
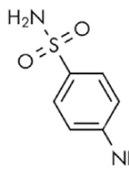
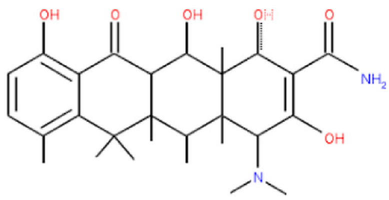
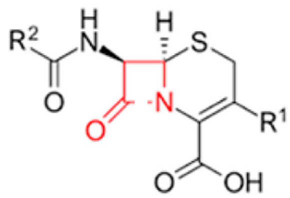
Imipenem, a carbapenem antibiotic, is less stable than penicillin and cephalosporin, do not chelate with the metal ions which further

TABLE 1 Chemical structure and name associated with the antibiotics.

S. No	Name	Chemical structure
1	Aminoglycosides	
2	Amphenicols	
3	Diaminopyrimidines	
4	Imidazoles	
5	Lacosamides	
6	Macrolides	

(*Continued*)

TABLE 1 Continued

S. No	Name	Chemical structure
7	Glycopeptides	
8	Quinolones	
9	Sulfonamides	
10	Tetracyclines	
11	β- lactams	

accelerates their decomposition (Deshpande et al., 2004). Only Imipenem was detected in the aquatic environment at low concentrations of 3.251 ng/L in ground water and 7.242 ng/L in wastewater influent (Kamatham et al., 2024).

3.11 Other antibiotics

Bacitracin, a polypeptide antibiotic was detected only in the ground water and wastewater influent samples from India (Kamatham et al., 2024).

Ethionamide, a thioamide antibiotic was detected at a concentration of 194.7 ng/L only in the wastewater influent sample of South Africa (Faleye et al., 2019).

In the ionophore class, only Monensin and Salinomycin were detected in the study at maximum concentrations of 3,007,800 ng/L (Zhang et al., 2013) and 30 ng/L (Watkinson et al., 2009) respectively.

Levamisole, a Imidathiazole antibiotic was found in concentrations of 24 ng/L and 182 ng/L in wastewater influent and hospital wastewater of Portugal respectively (Santos et al., 2013).

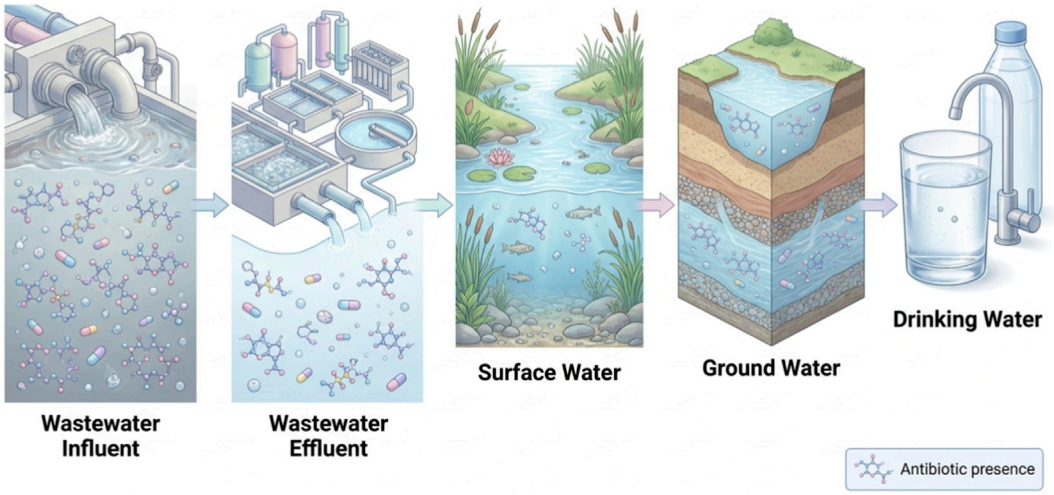


FIGURE 2
Occurrence of antibiotics in different aquatic system (Image designed using Figure Labs).

Linezolid, an oxazolidinone antibiotic and two triazoles, namely, Voriconazole and Flucanazole were detected only in the water samples of India at maximum concentrations of 13,600 ng/L, 2,500,000 ng/L, 236,950,000 ng/L, respectively (Lübbert et al., 2017).

Nifuroxazide, a nitrofuran antibiotic was observed only in the hospital wastewater and wastewater influents of Italy (Verlicchi et al., 2012).

Tiamulin, a pleuromutilin antibiotic was found in cattle and swine wastewater at 455 ng/L and 366 ng/L respectively (Zhang et al., 2013).

4 Occurrence of antibiotics in the aquatic environment

4.1 Wastewater influents

The wastewater influents showed considerably higher concentration of antibiotics than other aquatic compartments as they receive from various point origin sources, with values ranging from 1.79 ng/L (Santos et al., 2013) to 763,610 ng/L (Faleye et al., 2019) seen in albendazole. Out of 204 occurrences of antibiotics over various classes, the most frequently detected antibiotic class, they are ranked as Quinolones (47) > Macrolides (41) > Sulfonamides (37) > Tetracyclines (17) > β-Lactams and Diaminopyrimidines (10). Supplementary Table 2 (given in supplementary data file) shows the concentrations of various antibiotics in wastewater influent. Based on the concentrations, the antibiotic class are ranked as Imidazole (763,610.90 ng/L; Albendazole, Santos et al., 2013) > Quinolone (501,575.50; Ciprofloxacin, Faleye et al., 2019) > β-lactams (263,300 ng/L; Ampicillin, Mutiyar and Mittal, 2014) > Tetracycline (48,000; Tetracycline, Karthikeyan and Meyer, 2006) > Aminoglycoside (16,400 ng/L; Neomycin, Tahrani et al., 2016).

The antibiotics found in some of the sites were in accordance with the usage pattern found. For example, Tong et al. (2009) showed that the water which is received from swine farms had high concentrations of fluoroquinolones, tetracycline and macrolides while those near the fish ponds showed high number of Fluoroquinolones only. The concentrations were seen in range from 10 ng/L of penicillin V (Watkinson et al., 2009) to 111,100 ng/L of Ofloxacin (Lien et al., 2016) in the hospital wastewater. In municipal wastewater the concentrations ranged from 1 ng/L of Erythromycin and roxithromycin (Terzić et al., 2008) to 332,155,000 ng/L of Ciprofloxacin (Zafar et al., 2021). Figure 2 presents the concentrations of various antibiotics in Wastewater influent.

In swine wastewater, the concentrations occurred at ranges of 34.88 ng/L (Sulfamethoxazole) to 51,250 ng/L in Sulfamethazine (Jiang et al., 2013). The concentrations occurred at 61 ng/L to 19,000 ng/L of lincomycin and sulfamethoxazole, respectively in dairy wastewater (Watanabe et al., 2010). On the other hand, in cattle wastewater, the concentrations ranged from 40 ng/L to 3,007,800 ng/L in Tylosin and Monensin, respectively (Zhang et al., 2013).

The average high concentrations were seen in the municipal wastewater making them the primary source of antibiotic pollution in the surface waters. This was followed by hospital wastewaters and livestock wastewater. In the wastewater of human activity, namely, the municipal and hospital wastewater, of the 118 occurrences, the frequently detected antibiotic class was Quinolones (34) followed by Macrolides (21) > Sulfonamides (19) > Tetracyclines and imidazole (7). On the other hand, out of 48 occurrences, in wastewater of animal activity, Sulfonamides were detected more frequently which was followed by Tetracyclines (11) and Macrolides (5) (Table 5).

The compiled occurrence data demonstrated a clear geographic variation in antibiotic contamination studies, with a disproportionately high number of monitoring reports originating from Asian and European countries such as China, India, Italy, Portugal, and the Czech Republic, whereas comparatively fewer studies were reported from Africa and South America. The dataset also showed that regions with dense population, intensive agricultural activity, rapid urbanization, and inadequate wastewater treatment infrastructure generally reported higher antibiotic concentrations. Exceptionally elevated levels of

TABLE 2 Concentrations of various antibiotics in Wastewater effluent.

Antibiotic class	Name of the antibiotic	Reported concentration (ng/L)	Country	References
Aminoglycoside	Amikacin	1,800	Tunisia	Tahrani et al. (2016)
Aminoglycoside	Gentamycin	600	Tunisia	Tahrani et al. (2016)
Aminoglycoside	Kanamycin B	5,400	Tunisia	Tahrani et al. (2016)
Aminoglycoside	Neomycin	400	Tunisia	Tahrani et al. (2016)
Aminoglycoside	Paromycin	1,300	Tunisia	Tahrani et al. (2016)
Aminoglycoside	Sisomycin	1,000	Tunisia	Tahrani et al. (2016)
Aminoglycoside	Streptomycin	1,200	Tunisia	Tahrani et al. (2016)
Aminoglycosides	Streptomycin	323 ng/L	Curitiba, Southern Brazil	Nogueira et al. (2025)
Amphenicol	Chloramphenicol	60	Lake Victoria Basin, Kenya	Kimosop et al. (2016)
Amphenicol	Chloramphenicol	1,100	Tunisia	Tahrani et al. (2016)
Amphenicol	Florfenicol	800	Tunisia	Tahrani et al. (2016)
Ionophores	Monensin	1.2	Nebraska, United States	Brown et al. (2015)
Lincosamides	Lincomycin	3.6	Nebraska, United States	Brown et al. (2015)
Macrolides	Clarithromycin	536	Canada	Miao et al. (2004)
Macrolides	Erythromycin	294	South Korea	Kim et al. (2007)
Macrolides	Erythromycin-H2O	838	Canada	Miao et al. (2004)
Macrolides	Roxithromycin	18	Canada	Miao et al. (2004)
Macrolides	Tylosin	3,400	South–East Queensland, Australia	Watkinson et al. (2009)
Others	Trimethroprim	188	South Korea	Kim et al. (2007)
Oxazolidinones	Linezolid	13,600	Hyderabad, India	Lübbert et al. (2017)
Quinolones	Ciprofloxacin	400	Canada	Miao et al. (2004)
Quinolones	Ciprofloxacin	160	Lake Victoria Basin, Kenya	Kimosop et al. (2016)
Quinolones	Moxifloxacin	710	Hyderabad, India	Lübbert et al. (2017)
Quinolones	Norfloxacin	112	Canada	Miao et al. (2004)
Quinolones	Ofloxacin	506	Canada	Miao et al. (2004)
Sulfonamides	Sulfacetamide	151	Canada	Miao et al. (2004)
Sulfonamides	Sulfadiazine	19	Canada	Miao et al. (2004)
Sulfonamides	Sulfadimethoxine	1.2	Nebraska, United States	Brown et al. (2015)
Sulfonamides	Sulfamerazine	0.14	Nebraska, United States	Brown et al. (2015)
Sulfonamides	Sulfamethazine	363	Canada	Miao et al. (2004)
Sulfonamides	Sulfamethazine	6.8	Nebraska, United States	Brown et al. (2015)
Sulfonamides	Sulfamethazole	0.4	Nebraska, United States	Brown et al. (2015)
Sulfonamides	Sulfamethoxazole	871	Canada	Miao et al. (2004)
Sulfonamides	Sulfamethoxazole	80	Lake Victoria Basin, Kenya	Kimosop et al. (2016)
Sulfonamides	Sulfamethoxazole	49.9	Nebraska, United States	Brown et al. (2015)
Sulfonamides	Sulfamethoxazole	407	South Korea	Kim et al. (2007)
Sulfonamides	Sulfapyridine	228	Canada	Miao et al. (2004)
Sulfonamides	Sulfasalazine	150	South–East Queensland, Australia	Watkinson et al. (2009)

(*Continued*)

TABLE 2 Continued

Antibiotic class	Name of the antibiotic	Reported concentration (ng/L)	Country	References
Sulfonamides	Sulfathiazole	600	South–East Queensland, Australia	Watkinson et al. (2009)
Sulfonamides	Sulfisoxazole	34	Canada	Miao et al. (2004)
Tetracyclines	Doxycycline	46	Canada	Miao et al. (2004)
Tetracyclines	Tetracycline	977	Canada	Miao et al. (2004)
Triazole	Flucanazole	48,311,000	Hyderabad, India	Lübbert et al. (2017)
Triazole	Voriconazole	324,900	Hyderabad, India	Lübbert et al. (2017)
β- lactams	Amoxycillin	80	Lake Victoria Basin, Kenya	Kimosop et al. (2016)
β- lactams	Ampicillin	360	Lake Victoria Basin, Kenya	Kimosop et al. (2016)
β- lactams	Carbapenems	358 ng/L	Curitiba, Southern Brazil	Nogueira et al. (2025)

ciprofloxacin, metronidazole, sulfamethoxazole, and albendazole were observed in samples from South Africa and India, while widespread detection of quinolones, sulfonamides, and macrolides was consistently reported across China, Europe, Australia, and the United States. This uneven geographic distribution of available data indicates potential monitoring bias and highlights the lack of comprehensive surveillance studies in several developing regions, particularly in low-income countries where antibiotic misuse and untreated wastewater discharge may be significant contributors to environmental contamination. Therefore, the compiled tables emphasize the urgent need for globally harmonized monitoring programs and region-specific risk assessment strategies to better understand the environmental distribution and ecological implications of antibiotic residues in aquatic systems.

4.2 Wastewater effluents

Except for a few rare cases, most antibiotic concentrations in treated effluents were lower than those in influent water due to the multiple treatment processes involved. Among the 47 reported occurrences across different antibiotic classes, Sulfonamides were the most frequently detected (16 occurrences), followed by Macrolides (11 occurrences) and Aminoglycosides (7 occurrences). Table 2 shows the concentrations of various antibiotics in Wastewater effluents. Based on the concentration, the antibiotic class with the maximum concentration was Triazole (48,311,000; Flucanazole) followed by Oxazolidinones (13,600; Linezolid, Lübbert et al., 2017) > Aminoglycoside (5,400; Kanamycin B, Tahrani et al., 2016) > Macrolides (3,400; Tylosin, Watkinson et al., 2009).

4.3 Surface water

Antibiotics have been widely detected in the surface waters, given their occurrence in the Wastewaters, which are drained in the rivers (Mutiyar and Mittal, 2014). The study shows that the pattern of occurrence is almost on par with the study conducted. The detection frequency of the antibiotic class in the wastewater are in near consistency with the antibiotic class from the surface waters. Figure 3 and Supplementary Table 4 shows the concentrations of various antibiotics in the surface water. Antibiotics in the surface water ranged from 0.02 ng/L (Sulfathiazole; Murata et al., 2011) to 40,100 ng/L (Ciprofloxacin; Lübbert et al., 2017). The occurrence of the antibiotic class in the surface water, out of the 181 occurrences, the top 5 antibiotic class detected followed rank of Sulfonamides (56) > Macrolides (38) > Quinolones (33) > Tetracyclines (19) > Diaminopyrimidines (7). On the other hand, based on the concentrations present in the surface waters, the top 5 antibiotic class follows the rank of, Quinolones (40,100 ng/L; Ciprofloxacin) > Macrolides (27,700 ng/L; Clarithromycin) > Lincosamides (21,100 ng/L; Lincomycin) > Amphenicol (18,400 ng/L; Florfenicol) > Triazole (13,100 ng/L; Flucanazole) (Lübbert et al., 2017; Tahrani et al., 2016; Boxall et al., 2006).

4.4 Ground water and drinking water

The distribution of antibiotics is highly affected by the seasonal variations with increased quantity seen during spring and winter and reduced during summer (Tong et al., 2014). The low concentration of antibiotics in the groundwater can be due to the increased depth (Holm et al., 1995), increased rainfall leading to dilution of the antibiotics. In the 37 groundwater samples the most frequently detected antibiotic class were Sulfonamides (10) and Quinolones (6). In terms of highest concentration antibiotic class ranked as Sulfonamides (3,600 ng/L; Sulfamethazine, Watanabe et al., 2010) > Lincosamides (1,900 ng/L; Lincomycin, Watanabe et al., 2010) > Quinolones (317 ng/L; Flumequine, Lin et al., 2015). Table 3 shows the concentration of various antibiotics in drinking and ground water.

There have been limited studies conducted in the occurrence of the antibiotics in the drinking water which may be due to the concentrations mostly being below the levels of detection. While the concentration present cannot cause direct harm, they help in the spread and multiplication of ARBs. Only one sampling site with drinking water was considered in this study with macrolides and sulfonamides being detected more comparatively with the same being true for concentration (Voigt et al., 2020). The concentrations in the drinking water ranged from 7 ng/L (Roxithromycin) to 600 ng/L (Clarithromycin).

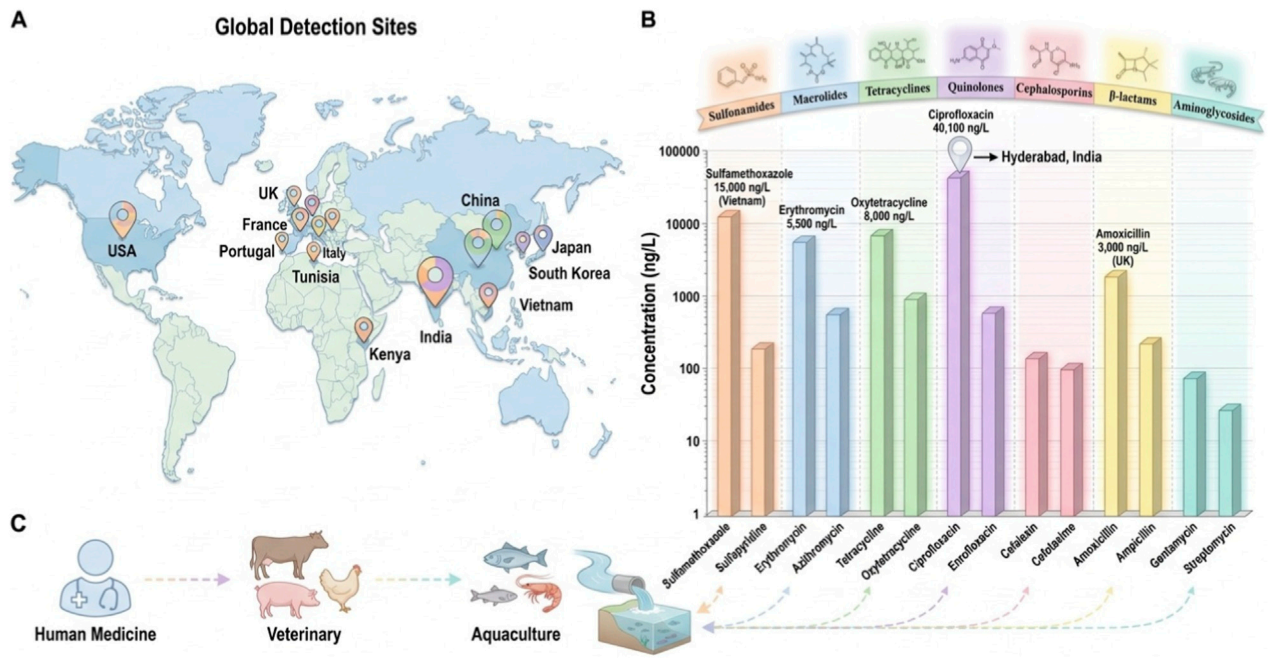


FIGURE 3
(A) Global detection sites of antibiotic contamination across aquatic systems. **(B)** Environmental concentrations of major antibiotic classes detected in different regions worldwide. **(C)** Major pathways of antibiotic entry into aquatic environments from human medicine, veterinary use, and aquaculture (Image designed using Figure Labs).

4.5 Point sources

The examination of the wastewater that are collected from various point sources helps in the identification of the antibiotic usage pattern among various regions and the source of the antibiotics. Due to their medicinal properties, antibiotics have become an invaluable substance for the sustaining of life (Lindberg et al., 2004). Generally, no distinction is usually made between the wastewater effluent and the effluent from any point source (Verlicchi et al., 2012). Hospital effluents are one of the major sources of antibiotics and the ARBs in the aquatic environment which can be due to excretion or disposal (Lien et al., 2016). Table 3 shows the concentration of various antibiotics that are present in the hospital wastewater effluents of the commonly detected antibiotic class in the hospital effluents, Quinolones (21) were detected the highest followed by Macrolides (15) and Sulfonamides (9). The high frequency of the quinolones indicates their common usage in the human medicine. Similarly, the concentration of Ofloxacin (111,100 ng/L; Lien et al., 2016) and Ciprofloxacin (101,000 ng/L; Lindberg et al., 2004) were the highest both of which were Quinolones (Supplementary Table 6).

Antibiotics in aquatic environments exhibit pseudo-persistence due to their continuous discharge from wastewater, hospitals, agriculture, and industrial sources, resulting in long-term environmental exposure despite degradation processes. Moreover, antibiotics commonly occur as mixtures, which may produce additive or synergistic toxic effects on aquatic organisms and promote the spread of antibiotic-resistant bacteria (ARB) and antibiotic resistance genes (ARGs). During wastewater treatment and environmental degradation, antibiotics can also form transformation products that may retain biological activity and toxicity, thereby posing additional ecological and public health concerns.

Various studies have shown the presence of antibiotics in the municipal wastewater effluents that collects sewage from a wide range of urbanization sources including hospitals, human consumption, agricultural run-off, livestock and poultry run-offs etc. This leads to the reason for some of the highest concentrations of antibiotics seen in the municipal wastewater effluents. Table 4 shows the concentration of the various antibiotics that are found in the municipal wastewater at few regions in the global level. The results obtained indicated that Quinolones (13) were the most detected antibiotic class in the municipal wastewater followed by sulfonamides (10). This can be interpreted as the incoming sewage from the hospital wastewater and the livestock feedlot run-offs. While only limited number of studies have been conducted exclusively on the municipal wastewater, the studies considered here showed higher levels of antibiotic concentrations in them. For example, Zafar et al., 2021 detected high concentrations of antibiotics in the Rawalpindi and Islamabad regions of Pakistan, namely, 332,155, 000 ng/L Ciprofloxacin, 32,570,000 ng/L Ampicillin, 16,009,000 ng/L Sulfamethoxazole, 6,638,000 ng/L Levofloxacin, etc. Similarly, Lübbert et al. (2017) also detected high levels of antibiotic concentration in the Hyderabad regions of India.

Another major source of antibiotic that originate from the point source is the run-offs from the livestocks and animal agriculture which can be due to veterinary medicine or feedlot. One of the major concern in the increased usage of antibiotics is that the bacteria present in the guts of the animals are subjected to develop antibiotic resistance (Zhang et al., 2013). These either directly enter the surface water or gets improperly treated in the WWTPs which is followed by

TABLE 3 Concentrations of various antibiotics in drinking and ground water.

Antibiotic class	Name of the antibiotic	Type of water source	Reported concentration (ng/L)	Country	References
Aminoglycoside	Amikacin	Ground water	2.003	Southern region of India	Kamatham et al. (2024)
Carbapenem	Imipenem	Ground water	3.251	Southern region of India	Kamatham et al. (2024)
Cephalosporin	Cefaperazone	Ground water	3.419	Southern region of India	Kamatham et al. (2024)
Imidazole	Dimetridazole	Ground water	1.8	Taipei and Hsinchu, Taiwan	Lin et al. (2015)
Imidazole	Metronidazole	Ground water	35.6	Taipei and Hsinchu, Taiwan	Lin et al. (2015)
Lincosamides	Clindamycin	Drinking water	340	Germany	Voigt et al. (2020)
Lincosamides	Clindamycin	Ground water	54.971	Southern region of India	Kamatham et al. (2024)
Lincosamides	Lincomycin	Ground water	1,900	San Joaquin Valley, United States	Watanabe et al. (2010)
Lincosamides	Lincomycin	Ground water	48.4	Taipei and Hsinchu, Taiwan	Lin et al. (2015)
Macrolides	Roxithromycin	Drinking water	7	Germany	Voigt et al. (2020)
Macrolides	Erythromycin-H2O	Drinking water	170	Germany	Voigt et al. (2020)
Macrolides	Azithromycin	Drinking water	550	Germany	Voigt et al. (2020)
Macrolides	Clarithromycin	Drinking water	600	Germany	Voigt et al. (2020)
Macrolides	Erythromycin	Ground water	6.697	Southern region of India	Kamatham et al. (2024)
Macrolides	Clarithromycin	Ground water	12.5	Taipei and Hsinchu, Taiwan	Lin et al. (2015)
Macrolides	Erythromycin-H2O	Ground water	54.8	Taipei and Hsinchu, Taiwan	Lin et al. (2015)
Others	Trimethroprim	Drinking water	390	Germany	Voigt et al. (2020)
Others	Trimethroprim	Ground water	17.8	Taipei and Hsinchu, Taiwan	Lin et al. (2015)
Polypeptide	Bacitracin	Ground water	6.567	Southern region of India	Kamatham et al. (2024)
Quinolones	Ciprofloxacin	Ground water	1.6	Taipei and Hsinchu, Taiwan	Lin et al. (2015)
Quinolones	Flumequine	Ground water	317	Taipei and Hsinchu, Taiwan	Lin et al. (2015)
Quinolones	Nalidixic acid	Ground water	16.4	Taipei and Hsinchu, Taiwan	Lin et al. (2015)
Quinolones	Norfloxacin	Ground water	9.3	Taipei and Hsinchu, Taiwan	Lin et al. (2015)
Quinolones	Ofloxacin	Ground water	11.8	Taipei and Hsinchu, Taiwan	Lin et al. (2015)

(*Continued*)

TABLE 3 Continued

Antibiotic class	Name of the antibiotic	Type of water source	Reported concentration (ng/L)	Country	References
Quinolones	Pipemidic acid	Ground water	10.2	Taipei and Hsinchu, Taiwan	Lin et al. (2015)
Sulfonamides	Sulfabactum	Ground water	3.251	Southern region of India	Kamatham et al. (2024)
Sulfonamides	Sulfadiazine	Ground water	14.4	Taipei and Hsinchu, Taiwan	Lin et al. (2015)
Sulfonamides	Sulfadimethoxine	Ground water	4.3	Taipei and Hsinchu, Taiwan	Lin et al. (2015)
Sulfonamides	Sulfadoxine	Drinking water	40	Germany	Voigt et al. (2020)
Sulfonamides	Sulfamethazine	Ground water	3,600	San Joaquin Valley, United States	Watanabe et al. (2010)
Sulfonamides	Sulfamethazine	Ground water	28.9	Taipei and Hsinchu, Taiwan	Lin et al. (2015)
Sulfonamides	Sulfamethoxazole	Drinking water	400	Germany	Voigt et al. (2020)
Sulfonamides	Sulfamethoxazole	Ground water	0.34	Nationwide survey	Murata et al. (2011)
Sulfonamides	Sulfamethoxazole	Ground water	1,820	Taipei and Hsinchu, Taiwan	Lin et al. (2015)
Sulfonamides	Sulfamethoxine	Ground water	130	San Joaquin Valley, United States	Watanabe et al. (2010)
Sulfonamides	Sulfamonomethoxine	Ground water	1.8	Taipei and Hsinchu, Taiwan	Lin et al. (2015)
Sulfonamides	Sulfathiazole	Ground water	3	Taipei and Hsinchu, Taiwan	Lin et al. (2015)
β- lactams	Piperacillin	Ground water	2.424	Southern region of India	Kamatham et al. (2024)
β- lactams	Tazobactum	Ground water	2.414	Southern region of India	Kamatham et al. (2024)

their entry to the surface water or the ground water (Campagnolo et al., 2002; Watanabe et al., 2010). Table 4 shows the concentrations of antibiotics at various point sources if livestock. The results obtained revealed that in all three types of livestock, Sulfonamides were the most used type of antibiotic class, followed by Tetracyclines and macrolides. The highest concentration was noted by Zhang et al. (2013) in the Cattle wastewater (3,007,800 ng/L; Monensin).

5 Fate and transformation of antibiotics in aquatic systems

5.1 Physicochemical processes

5.1.1 Hydrolysis

Hydrolysis is a significant abiotic degradation process, particularly for β-lactam antibiotics, macrolides, and other amide, ester, or carbamate-containing compounds. For example, amoxicillin easily degrades into amoxicillin penicilloic acid, diketopiperazine, and other products because of the instability of its β-lactam ring (Zhang et al., 2008). Likewise, penicillin is hydrolyzed to penicilloic acid by β-lactamase enzymes (Bavumiragira et al., 2022). Hydrolysis rate is strongly influenced by temperature and environmental pH. According to Hektoen et al. (1995), half-lives of cefalotin, cefoxitin, and ampicillin dropped from 5.3, 9.3, and 27 days at 25 °C and neutral pH to 0.067, 0.11, and 1.1 days at 60 °C, respectively. Sulfonamides are hydrolyzed in acidic conditions to produce sulfanilic acid (Tolls, 2001), and fluoroquinolones are more stable against hydrolysis under natural aquatic conditions (Luo et al., 2011).

5.1.2 Photodegradation

Photodegradation is a prevailing abiotic removal mechanism in surface waters, taking place by direct photolysis or indirect reactions via reactive oxygen species. Fluoroquinolones, sulfonamides, and

TABLE 4 Concentrations of various antibiotics in the municipal wastewater.

Antibiotic class	Name of the antibiotic	Reported concentration (ng/L)	Region/Country	References
Cephalosporin	Cefalexin	83	Serbia	Petrović et al. (2014)
Macrolides	Azithromycin	2	Western Balkan region	Terzić et al. (2008)
Macrolides	Clarithromycin	13,500	Hyderabad, India	Lübbert et al. (2017)
Macrolides	Erythromycin	1	Western Balkan region	Terzić et al. (2008)
Macrolides	Erythromycin-H2O	39	Mekong Delta, Vietnam	Managaki et al. (2007)
Macrolides	Josthromycin	8	Western Balkan region	Terzić et al. (2008)
Macrolides	Roxithromycin	1	Western Balkan region	Terzić et al. (2008)
Others	Trimethroprim	46	Mekong Delta, Vietnam	Managaki et al. (2007)
Others	Trimethroprim	259	Serbia	Petrović et al. (2014)
Others	Trimethroprim	2,550	Western Balkan region	Terzić et al. (2008)
Quinolones	Ciprofloxacin	19,400	Hyderabad, India	Lübbert et al. (2017)
Quinolones	Ciprofloxacin	3,700	Kristianstad, Sweden	Zorita et al. (2009)
Quinolones	Ciprofloxacin	3.32E+08	Rawalpindi and Islamabad, Pakistan	Zafar et al. (2021)
Quinolones	Ciprofloxacin	278	Serbia	Petrović et al. (2014)
Quinolones	Ciprofloxacin	7	Western Balkan region	Terzić et al. (2008)
Quinolones	Enrofloxacin	2	Western Balkan region	Terzić et al. (2008)
Quinolones	Levofloxacin	6,638,000	Rawalpindi and Islamabad, Pakistan	Zafar et al. (2021)
Quinolones	Moxifloxacin	694,100	Hyderabad, India	Lübbert et al. (2017)
Quinolones	Norfloxacin	3	Western Balkan region	Terzić et al. (2008)
Quinolones	Ofloxacin	16.7	Kristianstad, Sweden	Zorita et al. (2009)
Quinolones	Ofloxacin	2,558,000	Rawalpindi and Islamabad, Pakistan	Zafar et al. (2021)
Quinolones	Ofloxacin	220	Serbia	Petrović et al. (2014)
Quinolones	Ofloxacin	42	Western Balkan region	Terzić et al. (2008)
Sulfonamides	Sulfadiazine	132	Western Balkan region	Terzić et al. (2008)
Sulfonamides	Sulfamerazine	20	Western Balkan region	Terzić et al. (2008)
Sulfonamides	Sulfamethazine	251	Mekong Delta, Vietnam	Managaki et al. (2007)
Sulfonamides	Sulfamethazine	186	Western Balkan region	Terzić et al. (2008)
Sulfonamides	Sulfamethoxazole	360	Mekong Delta, Vietnam	Managaki et al. (2007)
Sulfonamides	Sulfamethoxazole	16,009,000	Rawalpindi and Islamabad, Pakistan	Zafar et al. (2021)
Sulfonamides	Sulfamethoxazole	432	Serbia	Petrović et al. (2014)
Sulfonamides	Sulfamethoxazole	11,600	Western Balkan region	Terzić et al. (2008)
Sulfonamides	Sulfapyridine	931	Western Balkan region	Terzić et al. (2008)
Sulfonamides	Sulfathiazole	4	Western Balkan region	Terzić et al. (2008)
Tetracyclines	Doxycycline	14,900	Hyderabad, India	Lübbert et al. (2017)
Triazole	Voriconazole	2,500,000	Hyderabad, India	Lübbert et al. (2017)
Triazole	Flucanazole	2.37E+08	Hyderabad, India	Lübbert et al. (2017)
β- lactams	Ampicillin	29,100	Hyderabad, India	Lübbert et al. (2017)
β- lactams	Ampicillin	32,570,000	Rawalpindi and Islamabad, Pakistan	Zafar et al. (2021)

TABLE 5 Concentrations of various antibiotics in animal farm wastewaters.

Antibiotic class	Name of the antibiotic	Type of water source	Reported concentration (ng/L)	Country	References
Ionophores	Monensin	Swine wastewater	895	Nebraska, United States	Zhang et al. (2013)
Lincosamide	Lincomycin	Swine wastewater	517	Nebraska, United States	Zhang et al. (2013)
Macrolides	Erythromycin	Swine wastewater	1,592	Nebraska, United States	Zhang et al. (2013)
Macrolides	Tylosin	Swine wastewater	4,913	Nebraska, United States	Zhang et al. (2013)
Macrolides	Virginiamycin	Swine wastewater	877	Nebraska, United States	Zhang et al. (2013)
Others	Trimethroprim	Swine wastewater	84	Mekong Delta, Vietnam	Managaki et al. (2007)
Pleuromutilin	Tiamulin	Swine wastewater	3,636	Nebraska, United States	Zhang et al. (2013)
Quinolones	Ciprofloxacin	Swine wastewater	325	Jiulongjiang River, China	Jiang et al. (2013)
Quinolones	Enrofloxacin	Swine wastewater	270.75	Jiulongjiang River, China	Jiang et al. (2013)
Quinolones	Norfloxacin	Swine wastewater	107.25	Jiulongjiang River, China	Jiang et al. (2013)
Quinolones	Ofloxacin	Swine wastewater	268	Jiulongjiang River, China	Jiang et al. (2013)
Sulfonamides	Sulfachloropyridazine	Swine wastewater	443	Nebraska, United States	Zhang et al. (2013)
Sulfonamides	Sulfadiazine	Swine wastewater	1,382.5	Jiulongjiang River, China	Jiang et al. (2013)
Sulfonamides	Sulfamerazine	Swine wastewater	548	Nebraska, United States	Zhang et al. (2013)
Sulfonamides	Sulfamethazine	Swine wastewater	19,153	Mekong Delta, Vietnam	Managaki et al. (2007)
Sulfonamides	Sulfamethazine	Swine wastewater	574	Nebraska, United States	Zhang et al. (2013)
Sulfonamides	Sulfamethazine	Swine wastewater	51,250	Jiulongjiang River, China	Jiang et al. (2013)
Sulfonamides	Sulfamethizole	Swine wastewater	490	Nebraska, United States	Zhang et al. (2013)
Sulfonamides	Sulfamethoxazole	Swine wastewater	34.88	Jiulongjiang River, China	Jiang et al. (2013)
Sulfonamides	Sulfamethoxazole	Swine wastewater	227	Mekong Delta, Vietnam	Managaki et al. (2007)
Sulfonamides	Sulfamethoxazole	Swine wastewater	342	Nebraska, United States	Zhang et al. (2013)
Sulfonamides	Sulfamethoxine	Swine wastewater	548	Nebraska, United States	Zhang et al. (2013)
Sulfonamides	Sulfathiazole	Swine wastewater	540	Nebraska, United States	Zhang et al. (2013)
Tetracyclines	Chlortetracycline	Swine wastewater	5,375	Jiulongjiang River, China	Jiang et al. (2013)
Tetracyclines	Chlortetracycline	Swine wastewater	2,740	Nebraska, United States	Zhang et al. (2013)
Tetracyclines	Oxytetracycline	Swine wastewater	5,800	Jiulongjiang River, China	Jiang et al. (2013)
Tetracyclines	Oxytetracycline	Swine wastewater	3,138	Nebraska, United States	Zhang et al. (2013)
Tetracyclines	Tetracycline	Swine wastewater	3,437.5	Jiulongjiang River, China	Jiang et al. (2013)
Tetracyclines	Tetracycline	Swine wastewater	2,860	Nebraska, United States	Zhang et al. (2013)
Lincosamide	Lincomycin	Dairy wastewater	61	San Joaquin Valley, United States	Watanabe et al. (2010)
Others	Trimethroprim	Dairy wastewater	2,300	San Joaquin Valley, United States	Watanabe et al. (2010)
Sulfonamides	Sulfamethazine	Dairy wastewater	15,000	San Joaquin Valley, United States	Watanabe et al. (2010)

(*Continued*)

TABLE 5 Continued

Antibiotic class	Name of the antibiotic	Type of water source	Reported concentration (ng/L)	Country	References
Sulfonamides	Sulfamethoxazole	Dairy wastewater	19,000	San Joaquin Valley, United States	Watanabe et al. (2010)
Sulfonamides	Sulfamethoxine	Dairy wastewater	11,000	San Joaquin Valley, United States	Watanabe et al. (2010)
Tetracyclines	Oxytetracycline	Dairy wastewater	800	San Joaquin Valley, United States	Watanabe et al. (2010)
Tetracyclines	Tetracycline	Dairy wastewater	3,100	San Joaquin Valley, United States	Watanabe et al. (2010)
Ionophores	Monensin	Cattle wastewater	3,007,800	Nebraska, United States	Zhang et al. (2013)
Macrolides	Erythromycin	Cattle wastewater	1,266	Nebraska, United States	Zhang et al. (2013)
Macrolides	Tylosin	Cattle wastewater	40	Nebraska, United States	Zhang et al. (2013)
Pleuromutilin	Tiamulin	Cattle wastewater	455	Nebraska, United States	Zhang et al. (2013)
Sulfonamides	Sulfachloropyridazine	Cattle wastewater	431	Nebraska, United States	Zhang et al. (2013)
Sulfonamides	Sulfamethazine	Cattle wastewater	342	Nebraska, United States	Zhang et al. (2013)
Sulfonamides	Sulfamethizole	Cattle wastewater	452	Nebraska, United States	Zhang et al. (2013)
Sulfonamides	Sulfamethoxine	Cattle wastewater	586	Nebraska, United States	Zhang et al. (2013)
Sulfonamides	Sulfathiazole	Cattle wastewater	130	Nebraska, United States	Zhang et al. (2013)
Tetracyclines	Chlortetracycline	Cattle wastewater	2,005	Nebraska, United States	Zhang et al. (2013)
Tetracyclines	Oxytetracycline	Cattle wastewater	1,471	Nebraska, United States	Zhang et al. (2013)
Tetracyclines	Tetracycline	Cattle wastewater	184	Nebraska, United States	Zhang et al. (2013)

tetracyclines are especially prone to photolysis (Anderson et al., 2004; Batchu et al., 2014), while macrolides are broken down at a slower rate because of restricted chromophore absorption (Rizzo et al., 2013). Seasonal and hydrological variability affect these processes. Ehalt Macedo et al. (2025) emphasized that under low-flow conditions, persistence of antibiotics in rivers is greater, but photodegradation still adds to the decay. Additionally, certain transformation products exhibit residual antibiotic activity, which has risks of resistance development and ecotoxicity (Aus der Beek et al., 2016; Rizzo et al., 2013).

5.1.3 Sorption to sediments

Sorption exerts a significant impact on the fate of antibiotics in the environment, especially the ones with high affinities for sediments and organic matter. Tetracyclines, for instance, have high coefficients of sorption because they chelate with cations and adsorb onto clays and organic carbon and hence sediment accumulation is tremendous (Pan and Chu, 2017). Sulfonamides, however, stay predominantly dissolved and mobile (Kümmerer, 2009), and fluoroquinolones are intermediate in their behavior, found both in the dissolved and particulate-bound fractions (Carballa et al., 2004). Ehalt Macedo et al. (2025) highlighted that sorption is frequently confused with removal; in fact, antibiotics may remain in sediments and even transform into their parent compounds under optimal conditions.

5.2 Biological processes

5.2.1 Biodegradation by microorganisms

Microbial degradation has a key function in the elimination of antibiotics from aquatic environments. Biodegradation may take place by means of biotransformation, biodegradation (molecule cleavage), or mineralization into inorganic compounds (Kümmerer, 2009). Microorganisms such as bacteria, fungi, and algae make use of antibiotics as either a carbon or energy source, or metabolize them through enzymatic action (Grenni et al., 2018).

Bacteria like Proteobacteria, Actinobacteria, and Bacteroidetes, and fungi like white-rot fungi (*Trametes versicolor*, Pycnoporus) are reported to secrete ligninolytic enzymes (laccase, manganese peroxidase, lignin peroxidase) which help them break down a diverse group of antibiotics. For instance, white-rot fungi have been shown to degrade 58%–78% of tetracyclines in 15 days (Chen et al., 2022; Eibes et al., 2011).

Microbial-mediated transformation reactions in aquatic ecosystems may result in detoxification, mineralization, or even activation of more toxic metabolites. For example, under the influence of nitrate, Gammaproteobacteria degrade sulfamethoxazole to 4-nitro-sulfamethoxazole, a by-product more toxic than the parent compound (Baquero et al., 2008). Antibiotics can also be co-metabolized by resistant bacterial communities, e.g., ciprofloxacin degraded by resistant bacteria with a removal of 40% being seen over 104 days in a river system (Kümmerer, 2009).

Environmental factors like temperature, pH, dissolved oxygen, and organic carbon content play a significant role in affecting biodegradation. Sulfamethoxazole, for instance, has a half-life of 77 days at 4 °C in natural river water but only 16 days at 25 °C (Xu et al., 2011). Likewise, nitrifying bacteria exhibit poor antibiotic degradation with reduced temperatures (Knapp et al., 2008).

Wastewater stabilization ponds, which Bavumiragira et al. (2025) researched, indicate that microbial process activity is seasonal too. Increased summer temperature and sunlight enhance bacterial–algal symbiosis, thus promoting the rate of biodegradation. Winter season, on the other hand, suppresses microbial activity such that antibiotics like sulfamethoxazole and ofloxacin persist for a longer time. The research proved that biodegradation in stabilization ponds is highly influenced by algal–bacterial interactions and seasonal fluctuations in microbial metabolism.

5.3 Persistence and bioaccumulation potential

Antibiotics are usually designed to be biologically active and capable of resisting quick degradation, thus their persistence in water bodies is a cause for concern. Antibiotics are resistant to the common wastewater treatment strategies and they can last for weeks to months based on their chemistry and environmental factors (Kümmerer, 2009).

5.3.1 Persistence in aquatic environments

A number of antibiotic classes exhibit long half-lives in sediments and surface waters. Fluoroquinolones and tetracyclines are particularly recalcitrant due to their high affinity to sediments and poor biodegradability (Daghrir and Drogui, 2013). For instance, fluoroquinolones like norfloxacin and ciprofloxacin are stable for months in river sediments (Golet et al., 2003). In a similar vein, tetracyclines are highly stable when adsorbed on sediments, but this decreases their bioavailability while enhancing environmental persistence (Tolls, 2001).

Ehalt Macedo et al. (2025) pointed out that despite human metabolic transformation, certain compounds can be converted back to their parent compound in the environment (Bound and Voulvoulis, 2004). Further, sulfonamide and macrolide transformation products might have antimicrobial activity and toxicological relevance (Rizzo et al., 2013; Aus der Beek et al., 2016). In global modeling research, Macedo et al. illustrated that under low-flow conditions, antibiotic levels in rivers frequently exceed thresholds conducive to resistance selection, highlighting their persistence at ecologically significant concentrations.

5.3.2 Bioaccumulation potential

While antibiotics tend to be polar and hydrophilic, several classes (most notably fluoroquinolones, macrolides, and tetracyclines) have exhibited bioaccumulation within aquatic organisms. For example, ciprofloxacin and ofloxacin have been found in fish tissues, which is due to their moderate lipophilicity and bioaccumulating capacity towards biomolecules (Santos et al., 2010). Bavumiragira et al. (2025) noted that during summer seasons, ofloxacin and norfloxacin concentrations remained higher in pond water, not only remaining persistent but also exhibiting medium-to-high environmental risk potential based on risk quotient (RQ) calculations.

Antibiotics associated with sediment may even bioaccumulate directly in bottom dwellers, leading to trophic transfer hazards (Martinez, 2009). Bioaccumulation combined with persistence increases environmental impacts, as below-inhibitory concentrations of antibiotics impose constantly selective pressure on microbial populations, which gives rise to resistance gene spread (Baquero et al., 2008).

5.4 Transport and mobility

5.4.1 Leaching

Leaching describes the downward movement of antibiotics from surface soils or waters to groundwater. Leaching is affected by hydrophobicity of compounds, ionization state, and soil organic matter. Polar and mobile sulfonamides are susceptible to leaching into aquifers, whereas tetracyclines as hydrophobic compounds are less mobile owing to intense sorption (Tolls, 2001). Sulfonamide contamination of groundwater has been extensively reported, with risks of prolonged exposure of subsurface ecosystems (Kemper, 2008).

5.4.2 Runoff

The runoff of agricultural areas is one of the predominant routes of entry for antibiotics into aquatic systems, particularly in areas with heavy veterinary drug usage. Soil amended with manure tends to release fluoroquinolones, macrolides, and sulfonamides into surface water upon precipitation events (Chee-Sanford et al., 2009). Zhang et al. (2015), for instance, noted the transport of sulfamethoxazole and ciprofloxacin into river catchments after storm runoff. Ehalt Macedo et al. (2025) highlighted that these pulses account for much of global river antibiotic loading, with concentration levels usually higher than resistance selection thresholds following monsoon or snowmelt events.

5.4.3 Sediment deposition and resuspension

Sediments are both sinks and secondary sources of antibiotics. Highly sorbing antibiotics like tetracyclines and fluoroquinolones get accumulated in sediments (Carballa et al., 2004), and they can remain there for months (Golet et al., 2003). Flood resuspension and dredging may again introduce these chemicals into the water column (Kim and Carlson, 2007). Bavumiragira et al. (2025) showed that seasonal hydrodynamics control this cycling: under summer low-flow conditions, antibiotics are deposited in pond sediments, while increased flows in rainy seasons facilitate remobilization.

5.4.4 Modeling antibiotic transport

Hydrological and fate models increasingly involve processes like sorption–desorption, leaching, and runoff in estimating antibiotic mobility (Hörsing et al., 2011). Uncertainties are present regarding the mobility of transformation products, as several metabolites have distinct sorption or leaching characteristics compared to their parent molecules (Rizzo et al., 2013).

5.5 Effect of climate and hydrological processes on degradation

Antibiotic degradation in aquatic systems is significantly influenced by climatic and hydrological variability. Temperature, sunlight exposure, precipitation, and river flow dynamics affect both abiotic (hydrolysis, photolysis) and biotic (microbial activity) processes, controlling the persistence and transport of antibiotics in various environments.

5.5.1 Temperature effects

Temperature is a main driving force for antibiotic degradation rates. Increased temperatures facilitate hydrolysis and microbial activity, which result in increased degradation of drugs like sulfonamides and fluoroquinolones (Xu et al., 2011). On the other hand, biodegradation decelerates significantly at low temperatures, causing antibiotics to remain in water and sediment for longer durations (Knapp et al., 2008). Seasonal research in stabilization ponds indicates that summer conditions favor bacterial–algal interaction, enhanced antibiotic removal, while winter conditions favor slower degradation (Bavumiragira et al., 2025).

5.5.2 Hydrology and flow conditions

Water flow regimes, either high flow in flooding or low flow in droughts, directly influence antibiotic concentration and degradation. Low flows decrease dilution capacity, leading to increased and prolonged concentrations (Ehalt Macedo et al., 2025). Conversely, intense rainfall and runoff episodes intensify antibiotic loading from agricultural soil and wastewater effluent (Zhang et al., 2015). Turbulence of flow also affects sediment resuspension, which will resuspend sorbed antibiotics back into the water column (Kim and Carlson, 2007).

5.5.3 Sunlight and seasonal photodegradation

Sunlight exposure is essential for photodegradation, which depends on latitude, season, and depth of water. Antibiotics like fluoroquinolones and sulfonamides degrade quicker under summer conditions of high solar radiation (Batchu et al., 2014). Winter conditions of decreased sunlight and ice cover in cold climates considerably reduce photolysis rates, increasing antibiotic persistence (Anderson et al., 2004).

5.5.4 Climatic change implications

Long-term climatic change will enhance these dynamics. More frequent droughts have the potential to increase residence times of antibiotics in rivers, whereas heavy precipitation events have the potential to lead to punctuated antibiotic concentrations from runoff (Kemper, 2008; Aus der Beek et al., 2016). These regimes enhance risks of resistance selection and ecological exposure, and climatic and hydrological variability emerges as a key consideration for antibiotic fate modeling.

6 Risks and ecological implications

6.1 Antimicrobial resistance (AMR)

Antibiotic residues select for resistant microbes in water, driving the spread of Antibiotic Resistance Gene (ARGs). Even trace levels can favor resistant strains via horizontal gene transfer (Okoye et al., 2022). Studies have found diverse ARGs (e.g., mcr, carbapenemases) co-occurring with mobile elements in groundwater, indicating high dissemination risk. Figure 4 presents the environmental AMR is a global health crisis (~1.3 million deaths in 2019) that can circulate back to humans through water and food (Thakur et al., 2025; Liu et al., 2023).

6.2 Toxic effects on aquatic organisms

Chronic antibiotic pollution harms non-target aquatic life. Sub-lethal antibiotic levels induce cellular/genotoxic stress in algae, invertebrates and fish. Antibiotics also disrupt microbial metabolism (e.g., nitrification, denitrification), altering nutrient cycling in sediments and water. Moreover, antibiotics can synergize with other pollutants (e.g., heavy metals) to exacerbate toxicity and ecosystem stress (Ljubojević Pelić et al., 2024). Research conducted in Australia on guppies exposed to fluoxetine (Prozac) at concentrations as low as 31.5 ng/L revealed that male guppies developed reduced body condition, diminished sperm velocity, and increased gonopodium length. Coupled with reduced behavioral variability, these sub-lethal effects could inhibit mating success and adaptation in natural environments (Aich et al., 2025). A study highlights that freshwater biofilm exposed to sulfonamide antibiotics (sulfamethazine and sulfamethoxazole) at 500–5,000 ng/L experienced decreased cell viability, reduced species diversity, diatom deformities, and elevated mortality (Świacka et al., 2023). The antibiotic norfloxacin (NOR) at sub-lethal doses (e.g., ≤100 mg/L) reduced heartbeat rate, impaired swimming behavior, and lowered feeding efficiency in Daphnia magna, which is a keystone freshwater organism. This indicates potential disruption of phytoplankton–zooplankton interactions and broader ecosystem functionality (Pan et al., 2017).

6.3 Impact on ecosystem functions

Persistent antibiotics restructure microbial communities, undermining ecosystem services. By killing sensitive bacteria, antibiotic residues alter biodegradation and nutrient fluxes, reducing system resilience. Reduced microbial diversity (dysbiosis) can impair food webs and biogeochemical cycles, ultimately degrading water quality and habitat stability (Maghsodian et al., 2022). Antibiotic exposure in aquatic ecosystems can impair nutrient cycling, because microbial community alterations undermine key functions such as decomposition, nitrogen cycling, and primary productivity. For instance, biofilm community structure and microbial diversity

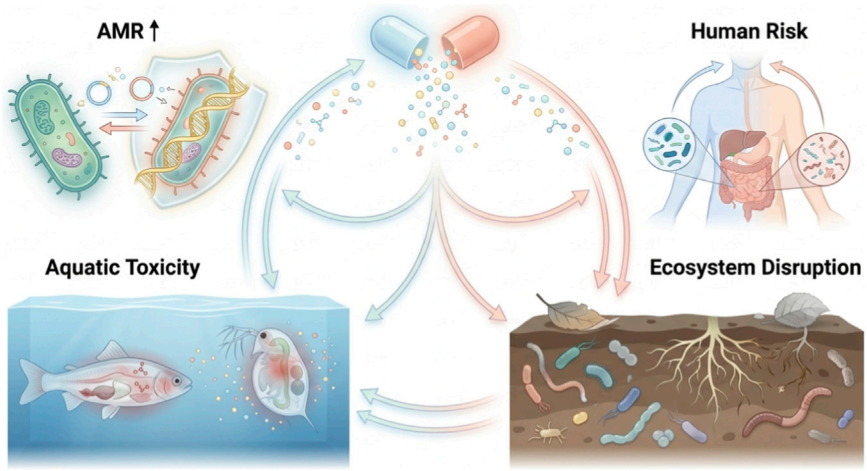


FIGURE 4
Environmental and health impacts of antibiotics, highlighting the development of antimicrobial resistance (AMR), aquatic toxicity, ecosystem disruption, and associated human health risks (Image designed using Figure Labs).

central to ecological resilience are altered upon exposure to antibiotic compounds (Świacka et al., 2023).

6.4 Human health implications

Aquatic antibiotics pose both indirect and direct human risks. Contaminated water used for drinking or irrigation can introduce antibiotic residues into crops and seafood, potentially causing organ toxicity (e.g., kidney/liver damage) or disrupting the human gut microbiome. More critically, polluted water serves as a reservoir for resistant pathogens (Akhter et al., 2024). Ingestion or contact with such water can transmit antibiotic-resistant infections to people. In sum, aquatic antibiotic pollution increases human exposure to drug-resistant bacteria and residual drug toxicity (Table 6). The persistence of antibiotic residues and ARGs in aquatic environments translates into contamination of drinking water sources and aquatic food chains (Watts et al., 2017). Humans consuming contaminated water or seafood are exposed to both residual antibiotics and AMR pathogens creating direct risks to public health. AMR's global impact is staggering. Nearly 5 million deaths were associated with resistant infections in 2019, of which 1.27 million were directly attributed to AMR. The WHO warns of a potential rise to 10 million deaths annually by 2050 if meaningful interventions are not implemented. The environmental spread of ARGs via resistant bacteria and mobile genetic elements magnifies this crisis, as waterborne ARGs can migrate into clinical settings and contribute to untreatable infections (Zhang et al., 2025).

Biotechnological approaches have emerged as sustainable and eco-friendly alternatives for the removal of antibiotic residues from aquatic environments due to their cost-effectiveness, reduced chemical demand, and minimal secondary pollution. Various microbial strains, including bacteria and fungi, are capable of biodegrading antibiotics through enzymatic transformation, mineralization, and co-metabolic pathways, thereby reducing their persistence and toxicity in water systems. Algal remediation and constructed wetlands have also shown considerable potential in removing antibiotics through bioaccumulation, adsorption, and phytodegradation mechanisms. Advanced systems such as membrane bioreactors (MBRs) and microbial fuel cells (MFCs) further enhance treatment efficiency by integrating biological degradation with filtration and bioelectrochemical processes. Additionally, biochar-assisted bioremediation improves microbial activity and adsorption capacity, facilitating the removal of recalcitrant antibiotic compounds. Despite their promising performance, factors such as operational stability, microbial adaptation, antibiotic concentration, and large-scale applicability remain major challenges. Nevertheless, these integrated biotechnological strategies represent promising tools for mitigating antibiotic pollution, limiting the dissemination of antibiotic resistance genes (ARGs), and supporting sustainable environmental safety.

7 Conclusion

This review provides a comprehensive synthesis of antibiotic occurrence in aquatic environments over the past 2 decades (2004–2024), covering 19 countries across both developed and developing regions. The findings highlight that quinolone, sulfonamides, macrolides, and tetracyclines are among the most frequently detected antibiotic classes, with concentrations ranging from a few ng/L to several μg/L depending on local usage patterns, wastewater treatment efficiency, and environmental conditions. Importantly, several reported concentrations exceed ecotoxicological thresholds, underscoring their potential role in the selection of antibiotic-resistant bacteria and genes, as well as risks to aquatic ecosystems. The review also emphasizes that occurrence patterns are highly variable across regions and seasons, reflecting differences in antibiotic consumption practices, treatment infrastructure, and environmental dynamics. While wastewater influents generally showed the highest concentrations, the detection of residues in surface water, groundwater, and even drinking water indicates a direct route for human and ecological exposure. This underlines the importance of continuous monitoring

TABLE 6 Risk category affected systems and ecological consequences.

Risk category	Affected organisms/ Systems	Mechanism of impact	Ecological consequences	Citation
ARG enrichment via microplastics + antibiotics	Carp (gut microbiome)	Co-exposure increases ARG abundance from tetracycline resistance genes	Elevated ARG spread and resistance potential in aquaculture	Zhang et al. (2025)
Amplified antibiotic toxicity via microplastics	Multiple aquatic species (meta-analysis)	MPs increase antibiotic uptake and toxicity (growth, immunity), but reduce reproductive toxicity	Altered population dynamics, impaired health	Huang et al. (2024)
Algal growth and nutrient removal shifts	Microalgae species	Antibiotic mixtures inhibit growth in some species; stimulate others via hormesis	Shifts in species dominance and altered nutrient cycling	Frascaroli et al. (2024)
Cyanobacteria physiological disruption	*Fremyella diplosiphon*	Antibiotics induce oxidative stress, membrane damage, pigment disruptions at high doses	Impaired photosynthesis, altered carbon fixation	Yalcin et al. (2022)
HGT-driven ARG proliferation without antibiotics	Shrimp aquaculture microbiome	Increase in ARGs and MGEs via plasmid-mediated transfer even in absence of antibiotic pressure	Rising resistance risks in food systems	Huang et al. (2024)
Reduced photosynthetic pigment content	Green algae/biofilms	Antibiotic exposure lowers pigment content and antioxidant enzyme activity	Compromised primary productivity	Hong et al. (2022)
Reproductive and behavioral impairment	Guppies	Fluoxetine exposure reduces sperm velocity, changes morphology, lowers behavioral plasticity	Decreased reproductive success, altered behavior	Aich and Polverino (2024)
Disrupted photosynthesis in diatoms	*Phaeodactylum tricornutum*	Alkylquinolone antibiotic inhibits photosynthetic efficiency	Reduced primary production in marine environments	Dow et al. (2020)
Complex antibiotic-MP co-toxicity	Freshwater micro-ecosystems	Antibiotic + MP combinations perturb submerged plant and epiphytic biofilm health	Ecosystem restoration challenges	Hong et al. (2022)
Fish oxidative stress and immune disruption	Various fish species	MPs provoke oxidative stress, gut microbiome changes; antibiotic co-presence intensifies effects	Reduced fitness and increased disease susceptibility	Dow et al. (2020)
Effects on photosynthetic light reactions	Model phototrophs	Four antibiotics inhibit growth, pigments, antioxidant enzymes	Potential declines in algae/diazotroph productivity	Dow et al. (2020)
Ecosystem-level imbalance via low-dose stressors	*Daphnia magna* and *Culex pipiens*	Ultra-low exposure disrupts interspecific competition	Long-term community shifts and altered species distributions	Schunck and Liess (2023)
Temperature-mediated ARG conjugation	Aquatic microbial communities	Higher temperatures enhance ARG conjugation rates	Accelerated AMR spread under warming climates	Frascaroli et al. (2024)
Microbial colonization on MPs	Marine environments	MPs serve as substrates for ARG-bearing biofilm formation	ARG reservoirs in aquatic systems	Schunck and Liess (2023)
Altered microalgal colorants and responses	*T. obliquus*, *A. protothecoides*	Antibiotic mixtures increase pigment production as ROS response	Variable stress adaptation, with potential ecosystem-level effects	Frascaroli et al. (2024)

and improved treatment technologies. By consolidating fragmented data, this review fills a critical gap between occurrence reports and their implications for environmental toxicology and public health. The evidence suggests that future research should prioritize the identification of high-risk antibiotic classes, establish standardized monitoring frameworks across countries, and link environmental levels to human exposure pathways. Strengthening global regulations and advancing wastewater treatment technologies remain essential steps in mitigating the persistence and spread of antibiotic residues in aquatic systems.

Author contributions

MSS: Methodology, Writing – original draft. Negaa: Writing – original draft. VS: Data curation, Formal Analysis, Writing – original draft. RK: Writing – review and editing. GA: Formal Analysis, Writing – review and editing. RS: Data curation, Formal Analysis, Writing – review and editing. BY: Writing – review and editing. KKA: Conceptualization, Methodology, Supervision, Writing – review and editing.

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Supplementary material

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