Water Science & Technology

Water Science & Technology Vol 85 No 6, 1800 doi: 10.2166/wst.2022.039

Occurrence and removal of 25 antibiotics during sewage treatment processes and potential risk analysis

Xinting Wang[a], Ning Huang[b], Jin Wang[b], Chunliu Lu[a], Guangying Li[a], Fang Li[b], Zaoshi Hu[b], Xiaoyu Bi[b], Lieshan Wu [a,*] and Yan Tian[b]

[a] School of Resources, Environment and Materials, Guangxi University, Nanning 530004, PR China
[b] Guangxi Zhuang Autonomous Region Ecological and Environmental Monitoring Centre, Nanning 530028, PR China
*Corresponding author. E-mail: wuls312@163.com

LW, 0000-0002-5973-432X

ABSTRACT

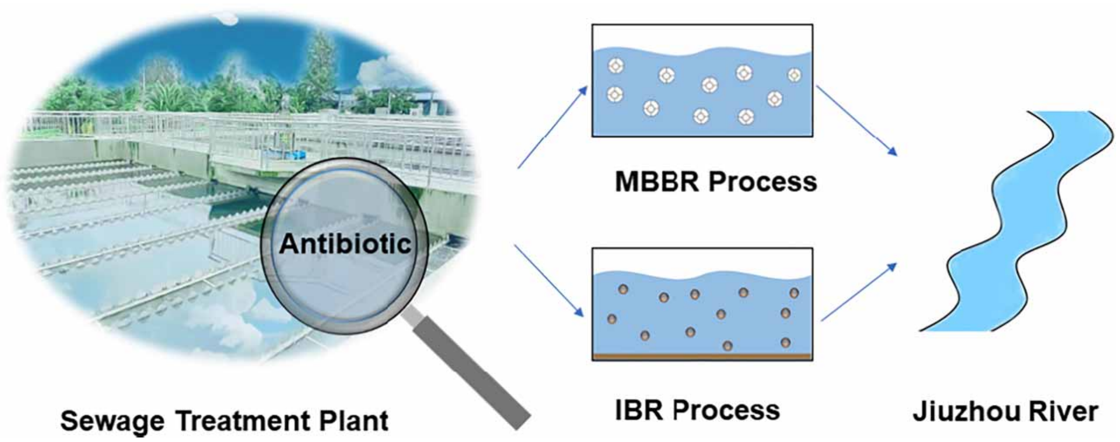
The occurrence and removal of 25 antibiotics, including ten quinolones (QNs), four macrolides (MLs), four tetracyclines (TCs) and seven sulfonamides (SNs), were analysed at two sewage treatment plants (STPs) with different treatment units in Guangxi Province, China. The results showed that 14 and 16 antibiotics were detected in the influent of the two STPs, with concentrations ranging from 13.7–4265.2 ng/L and 14.5–10761.7 ng/L, respectively. Among the antibiotics, TCs were the main type in the study area, accounting for more than 79% of the total concentration of all antibiotics. The antibiotic removal efficiencies of the different process units ranged from −56.73% to 100.0%. It was found that the SN removal efficiency of the multistage composite mobile bed membrane bioreactor (MBBR) process was better than that of the continuous-flow Intermission biological reactor (IBR) process, while the IBR process was better than the MBBR process in terms of removing TCs and MLs; however, there was no obvious difference in the QN removal efficiencies of these two processes. Redundancy analysis (RDA) showed a strong correlation between antibiotic concentration and chemical oxygen demand (COD). Risk assessments indicated that algae, followed by invertebrates and fish, were the most sensitive aquatic organisms to the detected antibiotics.

Key words: antibiotics, ecological risk, removal efficiency, sewage treatment plants

HIGHLIGHTS

- Tetracyclines are the main types of antibiotics in the study area.
- The removal efficiency of MBBR process for sulfonamides is better than that for IBR, but the opposite is true for tetracyclines.
- There is a strong correlation between the concentration of antibiotics and COD in wastewater in the study area.

GRAPHICAL ABSTRACT



1. INTRODUCTION

Antibiotics usually encompass a variety of compounds with antibacterial activity, including natural, synthetic and semisynthetic compounds (Sarmah *et al.* 2006). They can be used to treat human diseases (Qian-Qian *et al.* 2015), promote growth in animal husbandry, and improve feed efficiency (Dan *et al.* 2020). Therefore, in the past few decades, to meet human health needs and increase economic benefits, antibiotics have played a vital role in medical treatment (Zhi *et al.* 2020), animal husbandry (Gaballah *et al.* 2021) and agriculture (Parente *et al.* 2019). However, most organisms have incomplete absorption and metabolism of antibiotics (Sarmah *et al.* 2006). As a result, antibiotics and their metabolite residues can be directly or indirectly discharged into surface water or farmland irrigation water through livestock waste, threatening crop safety and human health (Shi *et al.* 2020).

With the gradual increase in the global demand for antibiotics (Klein *et al.* 2018), some scholars estimate that the global consumption of antibiotics is in the range of 100,000–200,000 tons (Cheng *et al.* 2014). Antibiotics continue to enter the environment in a variety of ways, making them a new organic pollutant that is ubiquitous in aquatic environments. STPs, as channels for domestic sewage discharge, are a more prominent issue in terms of concentrating antibiotic residues. Therefore, the occurrence, removal and transformation of antibiotics in STPs has become a popular issue in related research fields (Chen *et al.* 2019; Li *et al.* 2021; Wang *et al.* 2021).

Current STPs are mainly designed for conventional pollutants and do not take into account the removal of trace pollutants such as antibiotics. Therefore, most sewage treatment processes cannot effectively remove antibiotics from sewage and prevent them from entering water bodies (Yao *et al.* 2021). For example, in three STPs in Switzerland, the average clarithromycin removal efficiency was 81%, and the average tylosin removal efficiency was 76% (McArdell *et al.* 2003). The average erythromycin removal efficiency in two STPs in Hong Kong was 43%, and the average ciprofloxacin and aureomycin removal efficiencies were 66 and 83%, respectively (Li & Zhang 2011). As a result, antibiotic and metabolite residues can be detected when most treated sewage is discharged into the aquatic environment (Zhang *et al.* 2021), resulting in ecological risks to the surrounding environment of aquatic and terrestrial organisms to a certain extent (Morosini *et al.* 2020).

According to the data, there are studies related to antibiotics in STPs in the eastern and northern regions of China (Zhao *et al.* 2020; Bao *et al.* 2021) and fewer studies in the southern part of the country. The two STPs in this study are located in the Jiuzhou River basin in southern China. The aquaculture industry in this basin is developed, and sewage is directly discharged into the Jiuzhou River after treatment. Therefore, two STPs in the Jiuzhou River basin are studied. The occurrence and fate of this study are of great significance for understanding the risks and potential effects of antibiotics in the Jiuzhou River aquatic environment. This study involves the following aspects:

(1) Investigate the occurrence and distribution of antibiotics in the water phase of STP 1 and STP 2 sewage treatment;
(2) Compare the antibiotic removal effects of two different sewage treatment processes;
(3) Conduct a redundancy analysis of antibiotics in the water phase of two STPs, using water quality parameters;
(4) Carry out an ecological risk assessment of the typical antibiotics in STP 1 and STP 2.

The results can provide a reference for antibiotic pollution control and environmental hazard assessments in sewage treatment systems.

2. MATERIALS AND METHODS

2.1. Sample collection and preparation

During the treatment processes at the two STPs, sewage water samples were collected from different units. The STPs are all located on the Jiuzhou River in southern China. They mainly receive and treat local domestic sewage. The designed treatment flows are 500 m^3/d and 1,300 m^3/d, respectively. The first STP is a multistage composite mobile bed membrane bioreactor (MBBR) process, and the second is a continuous-flow transmission biological reactor (IBR) process. Water samples were collected from different treatment units in STP 1 and STP 2. The first STP consists of original influent, MBBR tank effluent, artificial wetland effluent (AWE) and final effluent; the second STP consists of original influent, IBR tank effluent, sedimentation tank effluent (STE) and final effluent. The specific sampling locations and sampling points are shown in Figure 1. The sampling plan was carried out in January 2021. During the sampling process, the STPs were in normal operation, and there was no rainfall projected for the near future. For each sampling, 1 L of each water sample was put into brown glass bottles

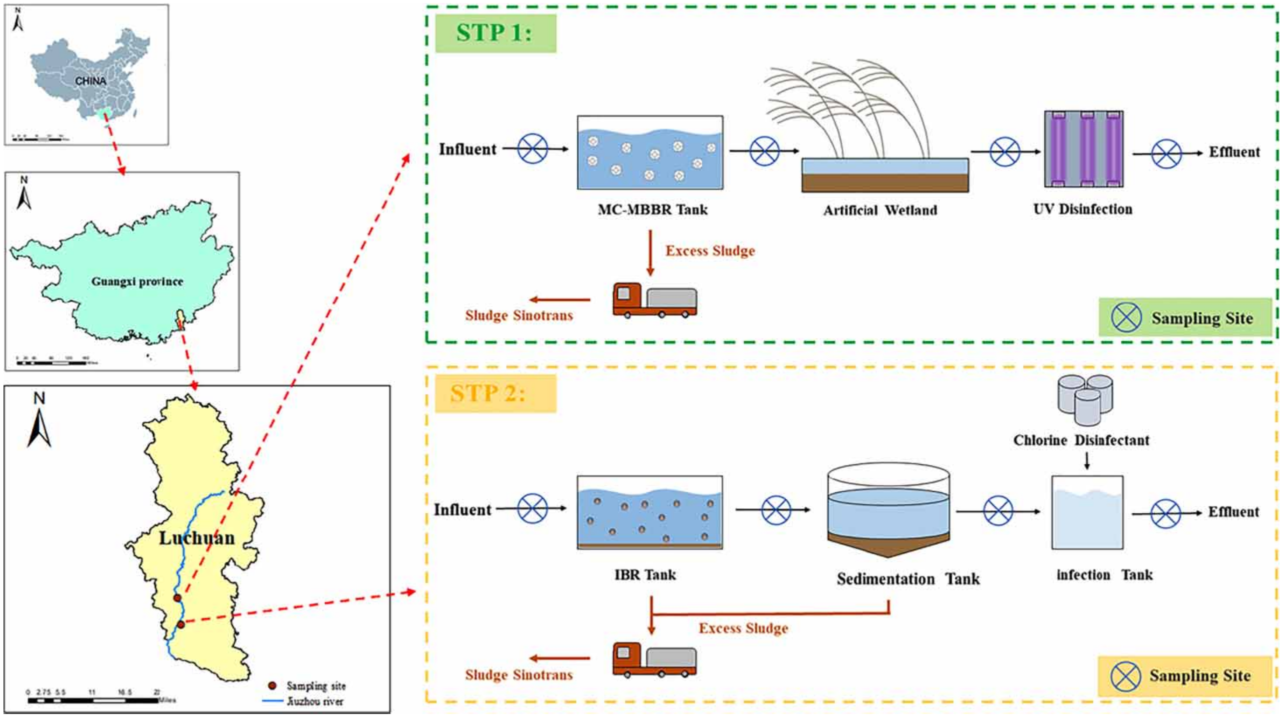


Figure 1 | Schematic diagram of location of STPs and sampling point of each treatment unit.

with three repetitions, and the samples were transported to the laboratory within 3 days for preprocessing and chemical analysis.

2.2. Chemicals and materials

The 25 target antibiotics were divided into four categories: (1) 10 kinds of quinolones (QNs): ciprofloxacin (CFX), danofloxacin (DNX), difloxacin (DFX), fleroxacin (FRX), marbofloxacin (MBX), norfloxacin (NFX), ofloxacin (OFX), oleandomycin (ODC), sarafloxacin (SFX), and sparfloxacin (SPX); (2) four kinds of macrolides (MLs): erythromycin (ERC), josamycin (JSC), kitasamycin (KSC), and tylosin (TLS); (3) four kinds of tetracyclines (TCs): chlortetracycline (CTC), doxycycline (DCC), oxytetracycline (OTC), and tetracycline (TCC); and (4) seven kinds of sulfonamides (SNs): sulfachlorpyridazine (SPZ), sulfadiazine (SDZ), sulfamethazine (SMZ), sulfamethazine (STZ), sulfamethoxazole (SXZ), sulfaquinoxaline (SQX), and sulfathiazole (SAZ).

All antibiotic analysis standards were purchased from Tianjin Alta Technology limited company. Detailed information and the chemical properties of the 25 antibiotics are listed in Table S1. Methanol (chromatographically pure) and acetonitrile (chromatographically pure) were purchased from Fisher Company in the United States, formic acid (chromatographically pure) was purchased from CNW Company in China, and other material information is listed in Table S2. Ultrapure water was used throughout the experiment, and methanol was used to dilute the antibiotic standard stock solution to prepare standard liquids of different concentrations.

2.3. Sample preparation and analysis

2.3.1. Pretreatment of the water samples

Based on the importance of the analysis step, water samples should be pretreated immediately when returned to the laboratory (Sharip *et al.* 2020). In the reference research, the preparation method of the water samples was modified (Pan *et al.* 2020), and the pretreatment steps were as follows: (1) Activation: use 10 mL methanol and 10 mL ultrapure water to carry out the solid phase extraction column activation so that the sample solution can smoothly flow through the solid phase extraction column; (2) Acidification: add H_2SO_4 or NaOH to 200 mL of the water sample to make the pH $= 3 \pm 0.1$; (3) Sample loading: transfer the adjusted water sample, pass it through the solid phase extraction column at a flow rate of 5 mL/min; (4) Rinse: rinse the solid phase extraction column with 10 mL ultrapure water to remove impurities; (5)

Dry: add nitrogen to dry the solid phase extraction column; (6) Elution: elute the solid phase extraction column with 3 mL formic acid/methanol mixture; (7) Collection: soak the solid phase extraction column with 3 mL formic acid/methanol mixture for 5 minutes, then use 9 mL formic acid/methanol mixture at a rate of 1 mL/min elute the sample and collect the eluate; (8) Constant volume: concentrate the above eluent to less than 1 mL in a nitrogen concentrator, dilute it to 1 mL in methanol, mix well, and place the water sample in a 2 mL brown sample flask through a 0.22 μm organic nylon filter, and preserve the water sample at −4 °C.

2.3.2. Analysis condition

A ZORBAX SB-C18 (2.1 × 100 mm, 3.5 μm) column and ultra-high performance liquid chromatography-tandem mass spectrometry (UPLC–MS/MS) were used to analyse the target antibiotics. Mobile phase A used for QNs and MLs was 0.5% formic acid, and mobile phase B was acetonitrile. The flow rate was 0.5 mL/min. The column temperature was 25 °C, and the injection volume was 5.0 μL. See Table S3 for the gradient elution procedure. The mass spectrometry conditions were the following: capillary voltage 4.0 kV, drying gas temperature 350 °C, and atomizing gas pressure 25 psi. The electrospray positive ion source (ESI+) multiple reaction detection (MRM) mode was used for mass spectrometry data acquisition. The mass spectrometric parameters are shown in Table S4. Mobile phase A used for TCs and SNs was 1% formic acid, and mobile phase B was acetonitrile. The flow rate was 0.3 mL/min. The column temperature was 30 °C, and the injection volume was 3.0 μL. The elution procedure is shown in Table S3, the mass spectrometry conditions were the same as those used for the QNs, and the mass spectrometer parameters are shown in Table S4.

2.4. Quality control and quality assurance

Using the external standard method, ultrapure water was used as a laboratory blank, and blank and label experiments were carried out on the target compound. According to all the steps of sample analysis, the blank experiment was repeated seven times, and the recovery rate and method detection limit were calculated. The results are shown in Table S5, none of the 25 target substances were detected in blank samples, LOQ was 6.9–21.6 ng/L, sample recovery was 60.2%–144.0%, and the detection curves showed good linear relationship ($R^2 > 0.9997$) at the 0.5–100 μg/L concentration of each antibiotic.

2.5. Risk assessment

In this study, risk quotients (RQs) were used to conduct ecological risk assessments of antibiotics. The RQs are the ratio of the measured environmental concentration (MEC) and the predicted no effect concentration (PNEC), as shown in (1). The calculation of PNEC is shown in (2), where LC_{50} is the semilethal concentration, EC_{50} is the half-maximal effective concentration, NOEC is the concentration of no observed effect; AF is the evaluation factor; when LC_{50} or EC_{50} is a chronic toxicity test, AF is taken as 100; and when LC_{50} or EC_{50} is an acute toxicity test, AF is taken as 1,000 (Dai *et al.* 2020):

$$\mathrm{RQs} = \frac{MEC}{PNEC} \tag{1}$$

$$\mathrm{PNEC} = \frac{LC_{50}}{AF} = \frac{EC_{50}}{AF} = \frac{NOEC}{AF} \tag{2}$$

In addition, the classic concentration additive model was used to evaluate the risk after mixing, and the formula is shown in (3):

$$\sum RQs_{\frac{MEC}{PNEC}} = \sum_{i=1}^{n} \frac{MEC_i}{PNEC_i} = \sum_{i=1}^{n} \frac{MEC_i}{EC_{50}\ algae, EC_{50}\ invertebrate, EC_{50}\ fish_i \times \left(\frac{1}{AF}\right)} \tag{3}$$

The ecological risk level was evaluated according to the RQ grading method proposed by other studies (Straub *et al.* 2019; He *et al.* 2020). When RQs < 0.1, there is a low ecological risk, and there is no need to take risk reduction measures at present; when 0.1 < RQs < 1.0, there is a medium ecological risk, indicating the need for further observation of the pollutant; when RQs > 1.0, there is a high ecological risk, indicating that the pollutant has ecological risk and corresponding measures should be taken to reduce the risk (Pan *et al.* 2020).

3. RESULTS AND DISCUSSION

3.1. Occurrence and concentrations of antibiotics

In the sewage collected from the different treatment units of STP 1 and STP 2, the occurrences and concentrations of 25 target antibiotics are shown in Table 1 and Figure 2. Fourteen and 16 antibiotics were detected in STP 1 and STP 2 sewage, respectively, and the detection rates were 56 and 64%. Among the antibiotics, TCs and QNs were the most frequently detected, followed by SNs, and the least frequently detected antibiotics were MLs. In addition, the concentrations of TCs and SNs in the raw water samples were much higher than those of QNs and MLs. This result may reflect the general use of TCs and SNs in the study area. This scenario is opposite that for the TCs and QNs frequently detected at the STP in Jinan (China) (Kun *et al.* 2021), but the situation is similar to that at the STPs in Guangdong (China) (Zhou *et al.* 2013). It may be that different regions have differing dependences on different classes of antibiotics (Phoon *et al.* 2020).

3.1.1. Occurrence and fate of antibiotics in STP 1

There were differences in the occurrences of antibiotics in the different treatment units. In the influent water, the concentrations of antibiotics detected in STP 1 were 13.7–4,265.2 ng/L, the concentrations of antibiotics detected in the MBBR tank were 15.0–2,773.6 ng/L, the concentrations of antibiotics detected in AWE were 12.0–693.3 ng/L, and the

Table 1 | Antibiotic concentrations were detected in STP 1 and STP 2 effluent collected from different treatment units

	STP 1				STP 2			
Compound name	**INF**	**MBBR**	**AWE**	**EFF**	**INF**	**IBR**	**STE**	**EFF**
CFX	35.8	19.9	12.0	10.7	14.5	15.8	12.0	11.3
DNX	38.4	43.6	34.1	28.1	39.2	36.2	35.8	19.4
DFX	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
FRX	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
MBX	13.7	15.0	12.3	8.2	14.6	11.8	11.5	8.1
NFX	124.1	121.4	60.2	36.4	129.9	140.2	81.9	39.3
OFX	59.3	46.7	23.8	16.8	54.7	55.4	42.9	16.4
ODC	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
SFX	26.4	28.7	26.4	26.2	26.3	26.1	26.0	21.3
SPX	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
TLS	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
KSC	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
JSC	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
ERC	277.3	434.6	432.8	269.2	620.7	421.2	400.9	142.7
CTC	739.5	308.8	88.6	9.8	1,349.0	605.9	100.4	9.9
DCC	452.4	433.7	414.6	18.6	1,200.0	657.4	450.8	17.9
OTC	4,265.2	2,773.6	693.3	15.5	10,761.7	6,215.5	1,791	17.2
TCC	3,120.3	1,219.0	227.7	16.6	7,193.0	2,867.9	729.4	17.0
SPZ	n.d.	n.d.	n.d.	n.d.	22.7	n.d.	n.d.	n.d.
SDZ	n.d.	n.d.	n.d.	n.d.	1,184.5	779.9	41.9	n.d.
SMZ	227.4	n.d.	n.d.	n.d.	190.4	160.5	n.d.	n.d.
STZ	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
SXZ	1,026.9	230.4	85.4	20.8	2,169.3	1,763.1	443.0	284.0
SQX	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
SAZ	344.5	n.d.	n.d.	n.d.	294.5	n.d.	n.d.	n.d.
∑Antibiotics	10,751.2	5,675.4	2,111.2	476.9	25,265.0	13,756.9	4,167.5	604.5

n.d.: Indicates below detection limit.

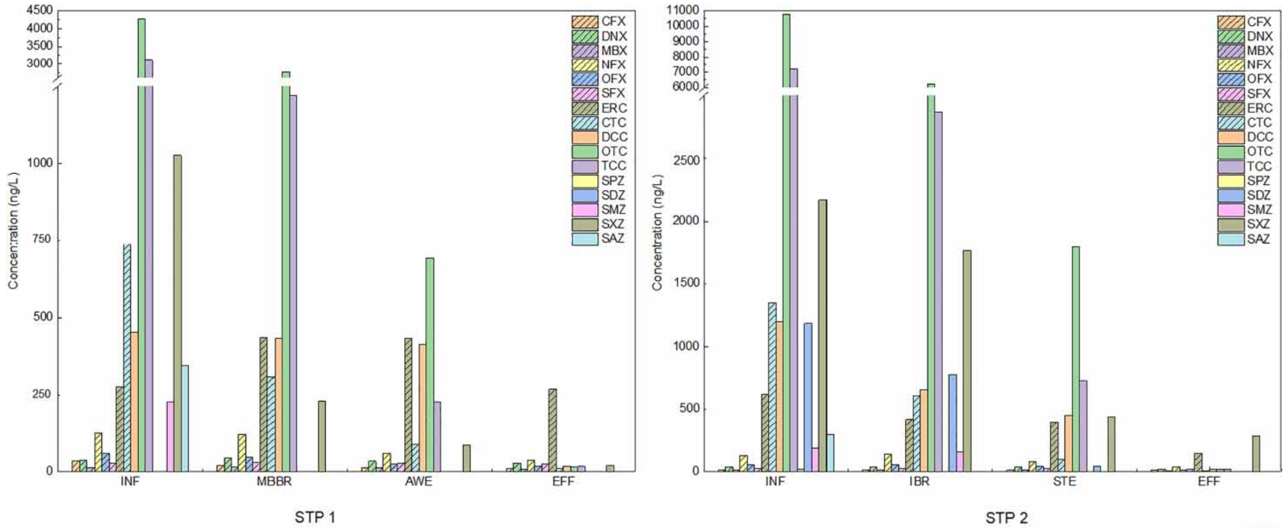


Figure 2 | Antibiotic concentrations were detected in STP 1 and STP 2 effluents collected from different treatment units.

concentrations of antibiotics detected in the effluent were 8.2–269.2 ng/L. In each processing unit, TCs and QNs were ubiquitous. It can be clearly seen in Figure 2 that OTC, TCC and SXZ mainly existed in the influent of STP 1. They are TCs and SNs, respectively. QNs and MLs accounted for 2.77% and 2.58% of the total influent concentration, respectively. This phenomenon may be caused by the inhibitory effect of TCs and SNs on both Gram-positive and Gram-negative bacteria (Qian-Qian *et al.* 2015). These antibiotics are mainly used to treat swine asthma, pasteurellosis, brucellosis, anthrax, and acute respiratory infections (Jinmei *et al.* 2021). However, SMZ is only used as an animal promoter in China (García-Galán *et al.* 2012) and should not be detected at high concentrations. Thus, it is possible that the raw water under investigation was affected by the migration and diffusion of antibiotics from nearby farms.

In the study area, QNs are detected at a high frequency, but they have low detected concentrations. Compared with other antibacterial drugs, QNs are widely used in human medicine, aquaculture and veterinary medicine for the treatment of bacterial diseases due to their wide antibacterial spectrum, strong antibacterial activity, lack of cross resistance and minimal toxic effects and side effects (Backhaus *et al.* 2000). However, QNs have a long half-life in water and do not easily degrade, so their detection rate in aquatic environments is high.

MLs are a typical antibiotic used by humans and animals. Only ERC was detected in STP 1, and the detected concentration was high and higher than that detected in a sewage plant in Hangzhou (China) (Bao *et al.* 2021). Studies have shown that ERC is an ML with a high detection rate in aquatic environments in China, but the detection rate of clarithromycin is higher than that of ERC in other developed countries (Göbel *et al.* 2005). This scenario indicates that there are certain differences in the uses of these antibiotics in different regions.

3.1.2. Occurrence and fate of antibiotics in STP 2

Similar to those in STP 1, the main classes of antibiotics in STP 2 were TCs and SNs, but there were differences in the contents and compositions of the four classes of antibiotics. In contrast to STP 1, in STP 2, the total detected concentration of antibiotics was 25,265.0 ng/L, which was more than twice that in STP 1. Compared with STP 1, STP 2 had concentrations of 12 antibiotics, in addition to CPX, SMZ, and SAZ, that were generally the same or higher. In addition, the ERC (620.7 ng/L), CTC (1,349.0 ng/L), DCC (1,200.0 ng/L), OTC (10,761.7 ng/L), TCC (7,193.0 ng/L), and SXZ (2,169.3 ng/L) detected in STP 2 were significantly higher than the ERC (277.3 ng/L), CTC (739.5 ng/L), DCC (452.4 ng/L), OTC (4,265.2 ng/L), TCC (3,120.3 ng/L), and SXZ (1,026.9 ng/L) detected in STP 1, and this result may be related to their processing scales and surrounding environments. Studies have shown that when TCs come into contact with activated sludge, they are easily adsorbed by the sludge in a short time and accumulate in the sludge. Most SNs have weak adsorption potential but good water solubility, while QNs and MLs are not easily degraded (Kun *et al.* 2021). Therefore, in the effluent, the detected concentrations of TCs were lower than those of the other three classes of antibiotics.

3.2. Different treatment units remove antibiotics

The removal efficiency of each treatment unit in the two STPs was calculated, as shown in Table 2 and Figure 3.

3.2.1. Removal efficiencies of the different treatment units in STP 1

As shown in Table 2, the antibiotic removal efficiency of the MBBR tank was −56.73% to 100%. In addition, the MBBR tank had a higher SN removal efficiency than the removal efficiencies of other three classes of antibiotics. Because TCs were the main antibiotics in STP 1 and the QN and ML removal efficiencies of this unit were not good, the total antibiotic removal efficiency was only 47.21% for the MBBR tank. It is worth noting that in the MBBR tank, negative removal of DNX, MBX, SFX, and ERC occurred, and their removal efficiencies were −13.54%, −9.49%, −8.71%, and −56.73%, respectively. Studies have shown that due to the low detection concentrations of QNs in winter, the QN removal efficiency fluctuates greatly, resulting in negative removal (Zhou *et al.* 2013). ERC, as an ML that is frequently detected in sewage, is frequently found to be negatively removed in different units throughout the STP process (Chang *et al.* 2010; Lin *et al.* 2018; Park *et al.* 2020). The phenomenon of negative removal may be caused by the following two reasons: first, the presence of other sewage water quality parameters, such as NH_4^+ and toxic compounds, may affect the degradability of antibiotics (Han *et al.* 2018). Second, although biofilm treatment technology is considered suitable for the treatment of biodegradable micropollutants (Petrovic *et al.* 2009), the scale of sludge flocculation is small. When the mass load discharged from the bioreactor is higher than that flowing into the bioreactor, part of the degraded or decomposed antibiotics may be reduced to the parent compound (Polesel *et al.* 2016), which produces antagonism. It is worth noting that the sulfonamide antibiotic removal efficiency of the MBBR tank is high. In addition to the SXZ removal efficiency of 77.56%, the SMZ and SAZ removal efficiency of the MBBR tank reached 100%. The reason for this result may be that in the biofilm system, the biological process may also involve highly active microorganisms with aerobic and anaerobic particles (Sabri *et al.* 2020; Kun *et al.* 2021). Particles increase the adsorption of antibiotic compounds in the MBBR tank.

When the four classes of target antibiotics passed through the AWE, except for that of CTC, OTC and TCC, the removal efficiency of other antibiotics ranged from 0.41% to 62.93%, and the total removal efficiency was 62.80%. Some studies have shown that the occurrence of redox is the main factor affecting the removal of antibiotics in an AWE because in the cold

Table 2 | Removal efficiency between treatment units in two STPs

	STP 1				STP 2			
Compound name	**MBBR**	**AWE**	**EFF**	**Total removal**	**IBR**	**STE**	**EFF**	**Total removal**
CFX	44.41%	39.70%	10.83%	70.11%	− 8.97%	24.05%	5.83%	22.07%
DNX	− 13.54%	21.79%	17.60%	26.82%	7.65%	1.10%	45.81%	50.51%
MBX	− 9.49%	18.00%	33.33%	40.15%	19.18%	2.54%	29.57%	44.52%
NFX	2.18%	50.41%	39.53%	70.67%	− 7.93%	41.58%	52.01%	69.75%
OFX	21.25%	49.04%	29.41%	71.67%	− 1.28%	22.56%	61.77%	70.02%
SFX	− 8.71%	8.01%	0.76%	0.76%	0.76%	0.38%	18.08%	19.01%
ERC	− 56.73%	0.41%	37.80%	2.92%	32.14%	4.82%	64.41%	77.01%
CTC	58.24%	71.23%	88.94%	98.67%	55.09%	83.43%	90.14%	99.27%
DCC	4.13%	4.40%	95.51%	95.89%	45.22%	31.43%	96.03%	98.51%
OTC	34.97%	75.00%	97.76%	99.64%	42.24%	71.18%	99.04%	99.84%
TCC	60.93%	81.32%	92.71%	99.47%	60.13%	74.57%	97.67%	99.76%
SPZ	–	–	–	–	100%	–	–	100%
SDZ	–	–	–	–	34.16%	94.63%	100%	100%
SMZ	100%	–	–	100%	15.70%	100%	–	100%
SXZ	77.56%	62.93%	75.64%	97.98%	18.72%	74.87%	35.89%	86.91%
SAZ	100%	–	–	100%	100%	–	–	100%
∑Antibiotics	47.21%	62.80%	77.41%	95.56%	45.55%	69.71%	85.49%	97.61%

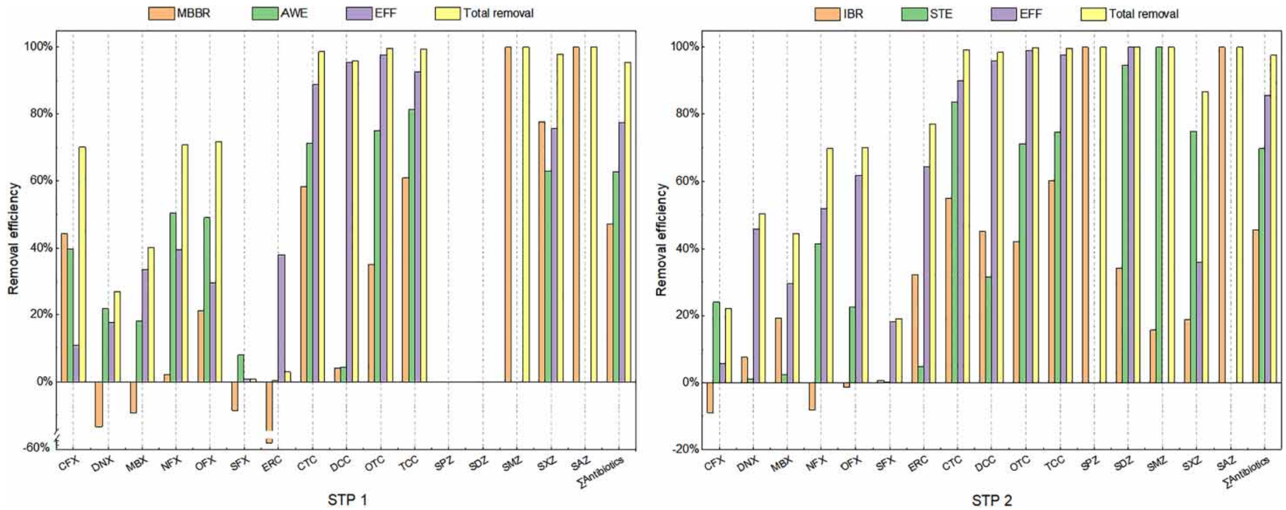


Figure 3 | The antibiotic removal efficiency of each processing unit.

winter, the plants existing in an AWE cause the anaerobic process of antibiotics through radial oxygen loss (Dan *et al.* 2020). Therefore, the overall antibiotic removal efficiency of the AWE was not high. After UV disinfection, the concentration of the antibiotics in STP 1 also decreased significantly, indicating that UV disinfection can effectively remove antibiotics in water, a result that is consistent with those of previous studies (Phoon *et al.* 2020). After the treatment, the concentrations of 16 antibiotics were in the range of n.d. ~269.2 ng/L, and the concentrations of the target antibiotics all decreased.

3.2.2. Removal efficiencies of the different treatment units in STP 2

When passing through the IBR tank, except for the SPZ and SAZ removal efficiencies, which were 100%, the removal efficiencies of the remaining 14 antibiotics were −8.97% to 60.13%, and the total antibiotic removal efficiency was 45.55%. In the IBR tank, CFX, NFX, and OFX had a negative removal trend, and the removal efficiency of other antibiotics was not high in this unit. Studies have shown that antibiotics can be adsorbed in a short period of time when they are in contact with activated sludge in IBR tanks (Kun *et al.* 2021). However, when antibiotics accumulate over a long-term cycle in this unit, high concentrations of antibiotics will inhibit the activity of microorganisms in activated sludge (Yang *et al.* 2020; Bao *et al.* 2021). This scenario leads to a low removal efficiency or even negative removal in the IBR tank (Hou *et al.* 2019).

The QN and ML concentrations in the sedimentation tank did not decrease significantly, which was consistent with the results of previous reports (Zhang *et al.* 2021) and may be related to the solid–liquid distribution coefficient (Tang *et al.* 2019). Chlorination is the last stage of STP 2. In this stage, the concentrations of all antibiotics decreased to different degrees compared with those in water, and the removal rate was 19.01%–99.84%. Although chlorinated disinfectants can react with dissolved organics such as antibiotics in water to produce a removal effect, they will produce a wide range of disinfection byproducts (Feng *et al.* 2019). Therefore, chlorination is not recommended for high-concentration antibiotic sewage.

3.2.3. Comparison of wastewater treatment units

Figure 3 shows that there were significant differences in the CFX and ERC removal efficiencies between the two STPs, and the removal efficiencies of the other QNs were 40.15%–70.67%. Studies have shown that the main QN removal mechanism is sludge adsorption. During the adsorption process, metal ions (such as Ca^{2+} and Mg^{2+}) promote the combination of carbonyl and carboxyl groups in QNs; the chelates formed are stable, so they are not easy to remove (El-Kommos *et al.* 2003).

In STP 1 and STP 2, the ERC removal efficiencies were 2.92% and 77.01%, respectively, with a large gap between the two processes. Studies have shown that ERC has strong hydrophobicity, and it is more likely to be removed by sludge adsorption (Park *et al.* 2020). Therefore, the ERC removal efficiency of the MBBR process was lower than that of the IBR process. It is worth noting that the detected concentrations of ERC in the effluent of the two STPs were 269.2 ng/L and 142.7 ng/L, respectively, which were relatively high antibiotic concentrations and similar to the high ERC concentrations detected in the STPs in South China and the United States (Yang & Carlson 2004; Zhou *et al.* 2013). This result also reflects that the ERC removal effect cannot be effectively removed during various processes, which has a great influence.

Comparing the two STPs, STP 1 had better SN removal efficiency than STP 2, mainly because of the biodegradation in STP 1 and STP 2. The TC removal efficiency of s the final effluent was more than 99%, which indicates that the removal efficiency of TC antibiotic water treatment is very high; this result showed that all units had a good removal effect, possibly because TC antibiotics are greatly affected by pH (Jiao *et al.* 2008). Under different pH conditions, the molecular structures of TCs have different forms of polar functional groups. With increasing pH value, the degradation ability of TCs is enhanced (Dai *et al.* 2020).

3.3. Correlation between antibiotics and environmental parameters

As mentioned above, changes in antibiotics in various treatment processes may be correlated with various environmental parameters, so the potential relationship between antibiotic concentration and environmental parameters (relevant parameters are shown in Table S6) was analysed through multiple factors.

The pH, water temperature (WT), chemical oxygen demand (COD), and ammonia nitrogen (NH_3-N) of the sewage in each treatment unit were measured, and Canoco5 software was used to directly analyse the relationship between the target antibiotic and the physical and chemical parameters of the water. According to detrended correspondence analysis (DCA), the gradient length was 2.0, which was less than 3. Therefore, the redundant analysis (RDA) model was used for analysis (Wang *et al.* 2017; Zhao *et al.* 2020). The analysis results are shown in Figure 4.

Although COD and WT are environmental factors, there had strong correlations with antibiotics. COD and most antibiotics had trends in the same direction and thus were positively correlated. However, WT was negatively correlated with antibiotics because its trend was not in the same direction.

Table S7 shows that the p values of COD, WT, pH, and NH_3-N were 0.042, 0.06, 0.152, and 0.91, respectively. These results indicate that the antibiotics in the study area were significantly related to COD, and there was no correlation with WT, pH, or NH_3-N. The reason for this phenomenon may be that the main antibiotics in the study area are TCs, and the COD removal process in each treatment unit is similar to that of TCs. Studies have shown that all organic matter is composed of C and H, so most organic matter can be chemically oxidized or oxidized by microorganisms in wastewater. Organic matter needs to

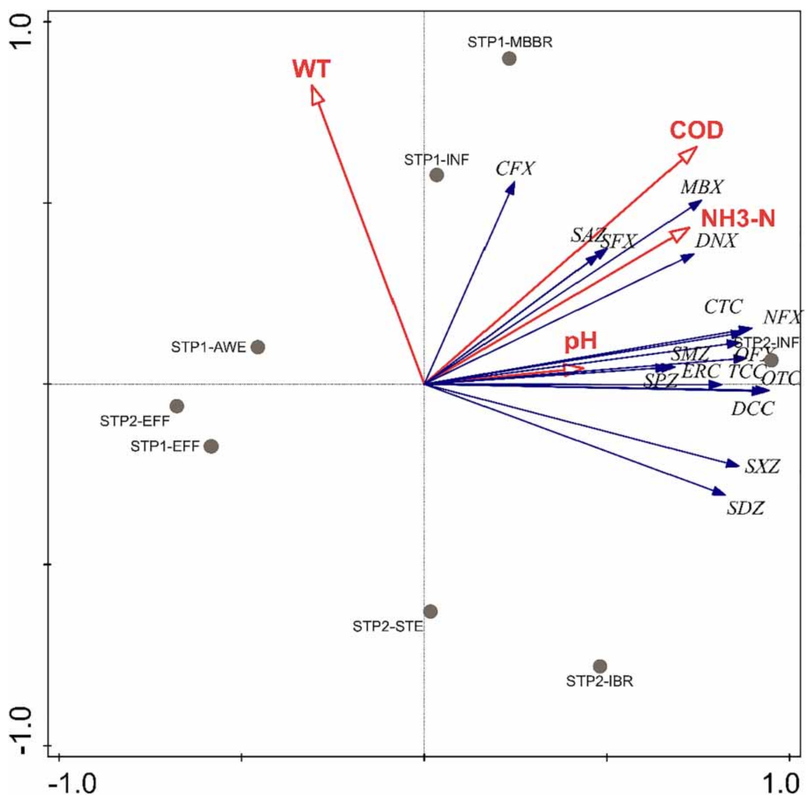


Figure 4 | RDA ordination diagram of antibiotics and major drivers.

consume oxygen during the chemical oxidation process. Therefore, the more antibiotics there are in wastewater, the higher the oxygen consumption; the two are positively correlated (Fu *et al.* 2020). The influence of many environmental factors results in this being an extremely complex water environment and system. Therefore, the relationship between traditional pollutants and antibiotics needs further exploration.

3.4. Ecological risks

The toxicity levels of antibiotics can be divided into acute toxicity and chronic toxicity (Marina *et al.* 2005). The former mainly occurs in green algae, bacteria and small daphnia, while chronic toxicity occurs mainly in large daphnia and fish, which is related to the trophic level at which antibiotics are located (Li *et al.* 2020). Under normal conditions, green algae, bacteria and small daphnia are at a low trophic level, while large daphnia and fish are at a high trophic level (Liu *et al.* 2015). Species at different trophic levels have different enrichment effects on pollutants, which makes the sensitivity of pollutants different, and this result is consistent with those of previous studies (Morosini *et al.* 2020). Discharging effluent containing antibiotics into the Kyushu River may adversely affect aquatic organisms in the aquatic environment and even human health (Wu *et al.* 2021). Therefore, an ecological risk assessment of the discharged effluent is of practical significance.

Current studies have shown that the toxic effects of TCCs on aquatic animals in aquatic environments rarely result in acutely toxic effects, but long-term exposure to antibiotics in the environment may be chronically toxic to nontarget organs (Dai *et al.* 2020). Antibiotics such as CFX, ERC, CTC, OTC, and TCC cause high ecological risks to some regions. For example, studies have found that in comparison to other antibiotics, OFX and CFX pose a higher risk to algae (Zhi *et al.* 2020). In ecosystems, different trophic levels have different sensitivities to target pollutants. For example, some scholars have studied the ecological risks to fish, invertebrates, and algae in rivers in the karst region of southwestern China (Huang *et al.* 2019), and the RQs of fish, invertebrates, and algae were 0–0.0009, 0–0.3031, and 0–14.6857, respectively, in this area. Therefore, different nutrient levels should be classified and discussed in an ecological risk assessment.

The RQs of antibiotics are shown in Table S8 and Figure 5. Except for the RQs of fish and invertebrates that show a medium risk in ERC, other target antibiotics pose a low risk. However, for the RQs of algae, CFX and ERC presented high risk levels, and OFX and TCC presented medium risk levels. Although individual concentrations of most target antibiotics were low, the RQs were in the range of 0–0.1 and showed a low risk. However, when multiple antibiotics coexist, the risk values of STP 1 and STP 2 to fish, invertebrates and algae change from low risk values to medium risk and high

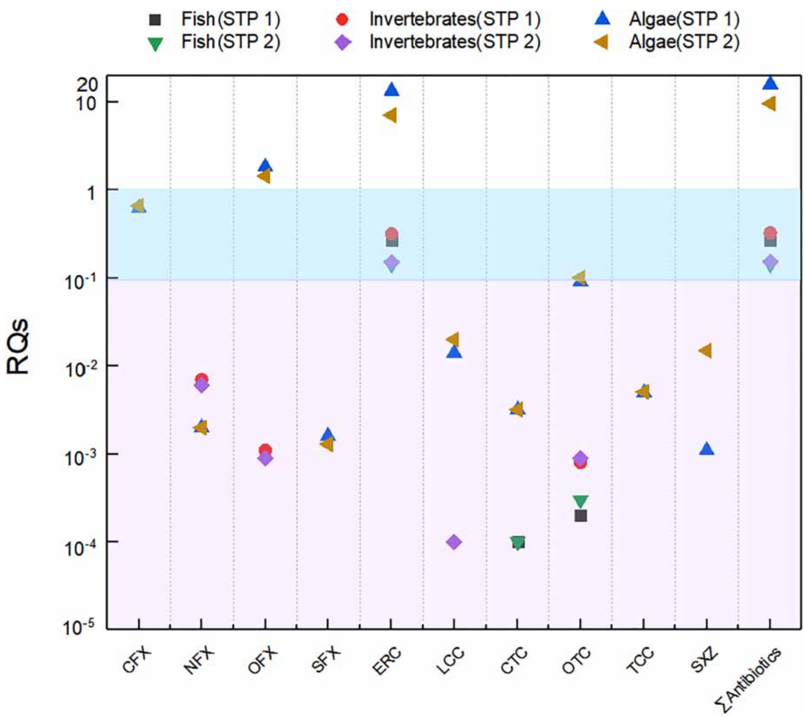


Figure 5 | RQs of antibiotics detected in the final effluent.

risk values. This result shows that, in comparison to the risk of other antibiotics, the risk of ERC in effluent in the study area is more prominent, and ERC has been included in the candidate list of drinking water pollutants by the United States Environmental Protection Agency in 2010 (Li & Zhang 2011); thus, attention should be given to the problem of ERC concentration residues. The results show that algae are the most sensitive species in the aquatic environment, followed by invertebrates, while fish sensitivity to antibiotics is generally not significant. Therefore, algae were the most sensitive aquatic organisms to the detected antibiotics, followed by invertebrates and fish, which was consistent with the results of previous reports (Suli *et al.* 2020).

Although TCs do not pose as high of a risk as other organic pollutants (such as polycyclic aromatic hydrocarbons), the residues of tetracycline antibiotics may promote the emergence of resistant bacteria (Agerso *et al.* 2006; Xiang *et al.* 2016); for example, soil bacteria with resistance to TCCs will develop, and bacterial activity will decrease after a period of time (Luo *et al.* 2010). In the past 70 years, soil antibiotic resistance genes have increased overall. It was also reported that 11 ARGs have been detected in the drinking water source of the Huangpu River (Jiang *et al.* 2013). Therefore, it is necessary to combine RQs with resistance genes.

4. CONCLUSIONS

This study investigated the occurrence and removal of 25 typical antibiotics at two STPs in Guangxi Province, China. The results showed that the detected concentrations of antibiotics were different in the different treatment units. Overall, in each treatment unit, in comparison to those of the other antibiotics, the detected concentrations of TCs accounted for the highest proportion, and it was the main antibiotic present. The SN removal efficiency of the MBBR process was better than that of the IBR process, but the TC and ML removal efficiencies of the IBR process were better than those of the MBBR process, which was based on the chemical structures, properties and environmental conditions of the sewage treatment plant. However, this result originates from a limited study site, so more investigations are needed to understand the fate of antibiotic occurrence and removal in STPs. In addition, the ecological risk assessment results indicated that ERC and SXZ may have toxic effects on aquatic organisms in this area. Risk mitigation measures therefore need to be developed. In addition to the management and elimination of antibiotics that occur in the environment, it is more important to encourage appropriate use of antibiotics and wastewater management to reduce the environmental risks caused by antibiotics from this source.

ACKNOWLEDGEMENTS

This work was supported by the Science and Technology Research and Development Program of Guangxi Zhuang Autonomous Region of China (Files no. AB21196039, no. AB21196039).

DECLARATION OF COMPETING INTERESTS

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

DATA AVAILABILITY STATEMENT

All relevant data are included in the paper or its Supplementary Information.

REFERENCES

Agerso, Y., Pedersen, A. G. & Aarestrup, F. M. 2006 Identification of Tn5397-like and Tn916-like transposons and diversity of the tetracycline resistance gene tet(M) in enterococci from humans, pigs and poultry. *Journal of Antimicrobial Chemotherapy* **57** (5), 832–839.

Backhaus, T., Scholze, M. & Grimme, L. H. 2000 The single substance and mixture toxicity of quinolones to the bioluminescent bacterium *Vibrio fischeri*. *Aquatic Toxicology* **49** (1), 49–61.

Bao, Y., Li, F., Chen, L., Mu, Q., Huang, B. & Wen, D. 2021 Fate of antibiotics in engineered wastewater systems and receiving water environment: a case study on the coast of Hangzhou Bay, China. *Science of the Total Environment* **769**, 144642.

Chang, X., Meyer, M. T., Liu, X., Zhao, Q., Chen, H., Chen, J. A., Qiu, Z., Yang, L., Cao, J. & Shu, W. 2010 Determination of antibiotics in sewage from hospitals, nursery and slaughter house, sewage treatment plant and source water in Chongqing region of Three Gorge Reservoir in China. *Environmental Pollution* **158** (5), 1444–1450.

Chen, C. X., Aris, A., Yong, E. L. & Noor, Z. Z. 2019 Evaluation of the occurrence of antibiotics at different treatment stages of decentralised and conventional sewage treatment plants. *International Journal of Environmental Science and Technology* 1–16.

Cheng, D., Liu, X., Wang, L., Gong, W., Liu, G., Fu, W. & Chen, M. 2014 Seasonal variation and sediment–water exchange of antibiotics in a shallower large lake in North China. *Science of the Total Environment* **476–477**, 266–275.

Dai, Y., Liu, M., Li, J., Yang, S., Sun, Y., Sun, Q., Wang, W., Lu, L., Zhang, K., Xu, J., Zheng, W., Hu, Z., Yang, Y., Gao, Y. & Liu, Z. 2020 A review on pollution situation and treatment methods of tetracycline in groundwater. *Separation Science and Technology* **55** (5), 1005–1021.

Dan, A., Zhang, X., Dai, Y., Chen, C. & Yang, Y. 2020 Occurrence and removal of quinolone, tetracycline, and macrolide antibiotics from urban wastewater in constructed wetlands. *Journal of Cleaner Production* **252**, 119677.

El-Kommos, M. E., Saleh, G. A., El-Gizawi, S. M. & Abou-Elwafa, M. A. J. T. 2003 Spectrofluorometric determination of certain quinolone antibacterials using metal chelation. *Talanta* **60** (5), 1033–1050.

Feng, Y. X., Guo, Q. Z. & Shao, B. 2019 Cytotoxic comparison of macrolide antibiotics and their chlorinated disinfection byproduct mixtures. *Ecotoxicology and Environmental Safety* **182**, 109415.

Fu, J., Zhang, Q., Huang, B., Fan, N. & Jin, R. 2020 A review on anammox process for the treatment of antibiotic-containing wastewater: linking effects with corresponding mechanisms. *Frontiers of Environmental Science & Engineering* **15** (1), 1.

Gaballah, M. S., Guo, J., Sun, H., Aboagye, D., Sobhi, M., Muhmood, A. & Dong, R. 2021 A review targeting veterinary antibiotics removal from livestock manure management systems and future outlook. *Bioresource Technology* **333**, 125069.

García-Galán, M. J., Blanco, S. G., Roldán, R. L., Díaz-Cruz, S. & Barceló, D. 2012 Ecotoxicity evaluation and removal of sulfonamides and their acetylated metabolites during conventional wastewater treatment. *Science of the Total Environment* **437**, 403–412.

Göbel, A., Thomsen, A., McArdell, C. S., Joss, A. & Giger, W. 2005 Occurrence and sorption behavior of sulfonamides, macrolides, and trimethoprim in activated sludge treatment. *Environmental Science & Technology* **39** (11), 3981–3989.

Han, T. N., Martin, R. & Yew-Hoong, G. K. 2018 Occurrence and fate of emerging contaminants in municipal sewage treatment plants from different geographical regions-a review. *Water Research* **133**, 182–207.

He, K., Borthwick, A. G., Lin, Y. C., Li, Y. N., Fu, J., Wong, Y. J. & Liu, W. 2020 Sale-based estimation of pharmaceutical concentrations and associated environmental risk in the Japanese wastewater system. *Environment International* **139**, 105690.

Hou, J., Chen, Z., Gao, J., Xie, Y., Li, L., Qin, S., Wang, Q., Mao, D. & Luo, Y. 2019 Simultaneous removal of antibiotics and antibiotic resistance genes from pharmaceutical wastewater using the combinations of up-flow anaerobic sludge bed, anoxic-oxic tank, and advanced oxidation technologies. *Water Research* **159**, 511–520.

Huang, F., Zou, S., Deng, D., Lang, H. & Liu, F. 2019 Antibiotics in a typical karst river system in China: spatiotemporal variation and environmental risks. *Science of the Total Environment* **650**, 1348–1355.

Jiang, L., Hu, X., Xu, T., Zhang, H., Sheng, D. & Yin, D. 2013 Prevalence of antibiotic resistance genes and their relationship with antibiotics in the Huangpu River and the drinking water sources, Shanghai, China. *Science of the Total Environment* **458**, 267–272.

Jiao, S., Zheng, S., Yin, D., Wang, L. & Chen, L. 2008 Aqueous oxytetracycline degradation and the toxicity change of degradation compounds in photoirradiation process. *Journal of Environmental Sciences* **20** (007), 806–813.

Jinmei, L., Ali, P. I., Guilong, Z. & Zhisheng, Y. 2021 Acceleration of emergence of *E. coli* antibiotic resistance in a simulated sublethal concentration of copper and tetracycline co-contaminated environment. *AMB Express* **11** (1) 14.

Klein, E. Y., Van Boeckel, T. P., Martinez, E. M., Pant, S., Gandra, S., Levin, S. A., Goossens, H. & Laxminarayan, R. 2018 Global increase and geographic convergence in antibiotic consumption between 2000 and 2015. *Proceedings of the National Academy of Sciences of the United States of America* **115** (15), E3463–E3470.

Kun, W., Tao, Z., Zhaoxin, S., Menghao, C. & Haichao, W. 2021 Antibiotic residues in wastewaters from sewage treatment plants and pharmaceutical industries: occurrence, removal and environmental impacts. *Science of the Total Environment* **788**, 147811.

Li, B. & Zhang, T. 2011 Mass flows and removal of antibiotics in two municipal sewage treatment plants. *Chemosphere* **83** (9), 1284–1289.

Li, F., Chen, L., Bao, Y., Zheng, Y., Huang, B., Mu, Q., Feng, C. & Wen, D. 2020 Identification of the priority antibiotics based on their detection frequency, concentration, and ecological risk in urbanized coastal water. *Science of the Total Environment* **747**, 141275.

Li, L. T., He, J. X., Gan, Z. W. & Yang, P. 2021 Occurrence and fate of antibiotics and heavy metals in sewage treatment plants and risk assessment of reclaimed water in Chengdu, China. *Chemosphere* **747**, 129730.

Lin, H., Li, H., Chen, L., Li, L., Yin, L., Lee, H. & Yang, Z. 2018 Mass loading and emission of thirty-seven pharmaceuticals in a typical municipal sewage treatment plant in Hunan Province, Southern China. *Ecotoxicology and Environmental Safety* **147**, 530–536.

Liu, J. C., Lu, G. H., Xie, Z. X., Zhang, Z. H., Li, S. & Yan, Z. H. 2015 Occurrence, bioaccumulation and risk assessment of lipophilic pharmaceutically active compounds in the downstream rivers of sewage treatment plants. *Science of the Total Environment* **511**, 54–62.

Luo, Y., Mao, D., Rysz, M., Zhou, Q., Zhang, H., Xu, L & Alvarez, P. 2010 Trends in antibiotic resistance genes occurrence in the Haihe River, China. *Environmental Science & Technology* **44** (19), 7220–7225.

Marina, I., Margherita, L., Angela, N., Luigia, P. & Alfredo, P. 2005 Toxic and genotoxic evaluation of six antibiotics on non-target organisms. *Science of the Total Environment.* **346** (1–3), 87–98.

McArdell, C. S., Molnar, E., Suter, M. J. & Giger, W. 2003 Occurrence and fate of macrolide antibiotics in sewage treatment plants and in the Glatt Valley watershed, Switzerland. *Environmental Science & Technology* **37** (24), 5479–5486.

Morosini, C., Poste, E., Mostachetti, M. & Torretta, V. 2020 Pharmaceuticals in water cycle: a review on risk assessment and wastewater and sludge treatment. *Environmental Engineering and Management Journal* **19** (8), 1339–1378.

Pan, C., Bao, Y. & Xu, B. 2020 Seasonal variation of antibiotics in surface water of Pudong New Area of Shanghai, China and the occurrence in typical wastewater sources. *Chemosphere* **239**, 124816.

Parente, C. E. T., Azeredo, A., Vollu, R. E., Zonta, E., Azevedo-Silva, C. E., Brito, E. M. S., Seldin, L., Torres, J. P. M., Meire, R. & Malm, O. 2019 Fluoroquinolones in agricultural soils: multi-temporal variation and risks in Rio de Janeiro upland region. *Chemosphere* **219**, 409–417.

Park, J., Kim, C., Hong, Y., Lee, W., Chung, H., Jeong, D.-H. & Kim, H. 2020 Distribution and removal of pharmaceuticals in liquid and solid phases in the unit processes of sewage treatment plants. *International Journal of Environmental Research and Public Health* **17** (3), 687.

Petrovic, M., Alda, M. D., Diaz-Cruz, S., Postigo, C., Radjenovic, J., Gros, M. & Barcelo, D. 2009 Fate and removal of pharmaceuticals and illicit drugs in conventional and membrane bioreactor sewage treatment plants and by riverbank filtration. *Philosophical Transactions* **367** (1904), 3979–4003.

Phoon, B. L., Ong, C. C., Saheed, M. S. M., Show, P.-L., Chang, J.-S., Ling, T. C., Lam, S.-S. & Juan, J.-C. 2020 Conventional and emerging technologies for removal of antibiotics from wastewater. *Journal of Hazardous Materials* **400**, 122961.

Polesel, F., Andersen, H. R., Trapp, S. & Plosz, B. G. 2016 Removal of antibiotics in biological wastewater treatment systems-a critical assessment using the activated sludge modeling framework for xenobiotics (ASM-X). *Environmental Science & Technology* **50** (19), 10316–10334.

Qian-Qian, Z., Guang-Guo, Y., Chang-Gui, P., You-Sheng, L. & Jian-Liang, Z. 2015 Comprehensive evaluation of antibiotics emission and fate in the river basins of China: source analysis, multimedia modeling, and linkage to bacterial resistance. *Environmental Science & Technology* **49** (11), 6772–6782.

Sabri, N. A., van Holst, S., Schmitt, H., van der Zaan, B. M., Gerritsen, H. W., Rijnaarts, H. H. M. & Langenhoff, A. A. M. 2020 Fate of antibiotics and antibiotic resistance genes during conventional and additional treatment technologies in sewage treatment plants. *Science of the Total Environment* **741**, 140199.

Sarmah, A. K., Meyer, M. T. & Boxall, A. B. A. 2006 A global perspective on the use, sales, exposure pathways, occurrence, fate and effects of veterinary antibiotics (VAs) in the environment. *Chemosphere* **65** (5), 725–759.

Sharip, Z., Abd Razak, S. B., Noordin, N. & Yusoff, F. M. 2020 Application of an effective microorganism product as a cyanobacterial control and water quality improvement measure in Putrajaya Lake. *Malaysia. Earth Systems and Environment* **4** (1), 213–223.

Shi, Y., Liu, J., Zhuo, L., Yan, X., Cai, F., Luo, W., Ren, M., Liu, Q. & Yu, Y. 2020 Antibiotics in wastewater from multiple sources and surface water of the Yangtze River in Chongqing in China. *Environmental Monitoring and Assessment* **192** (3), 159.

Straub, J. O., Caldwell, D. J., Davidson, T., D'Aco, V., Kappler, K., Robinson, P. F., Simon-Hettich, B. & Tell, J. 2019 Environmental risk assessment of metformin and its transformation product guanylurea. *I. Environmental Fate. Chemosphere* **216**, 855–865.

Suli, Z., Shizhou, S., Jing, Z., Gongyao, D. & Keqiang, Z. 2020 Systematic analysis of occurrence, density and ecological risks of 45 veterinary antibiotics: focused on family livestock farms in Erhai Lake basin, Yunnan, China. *Journal of Environmental Pollution.* **267**, 115539.

Tang, J. P., Wang, S., Fan, J. J., Long, S. X., Wang, L., Tang, C., Tan, N. F. & Yang, Y. 2019 Predicting distribution coefficients for antibiotics in a river water-sediment using quantitative models based on their spatiotemporal variations. *Science of the Total Environment* **655**, 1301–1310.

Wang, Z., Du, Y., Yang, C., Liu, X., Zhang, J., Li, E. & Wang, X. 2017 Occurrence and ecological hazard assessment of selected antibiotics in the surface waters in and around Lake Honghu, China. *Science of the Total Environment* **609**, 1423–1432.

Wang, K., Zhuang, T., Su, Z. X., Chi, M. H. & Wang, H. C. 2021 Antibiotic residues in wastewaters from sewage treatment plants and pharmaceutical industries: occurrence, removal and environmental impacts. *Science of the Total Environment* **788**, 147811.

Wu, Q., Pan, C.-G., Wang, Y.-H., Xiao, S.-K. & Yu, K.-F. 2021 Antibiotics in a subtropical food web from the Beibu Gulf, South China: occurrence, bioaccumulation and trophic transfer. *Science of the Total Environment* **751**, 141718.

Xiang, L., Wu, X.-L., Jiang, Y.-N., Yan, Q.-Y., Li, Y.-W., Huang, X.-P., Cai, Q.-Y. & Mo, C.-H. 2016 Occurrence and risk assessment of tetracycline antibiotics in soil from organic vegetable farms in a subtropical city, south China. *Environmental Science and Pollution Research* **23** (14), 13984–13995.

Yang, S. & Carlson, K. H. 2004 Solid-phase extraction-high-performance liquid chromatography-ion trap mass spectrometry for analysis of trace concentrations of macrolide antibiotics in natural and waste water matrices. *Journal of Chromatography A* **1038** (1–2), 141–155.

Yang, C.-W., Liu, C. & Chang, B.-V. 2020 Biodegradation of amoxicillin, tetracyclines and sulfonamides in wastewater sludge. *Water* **12** (8), 2147.

Yao, L., Chen, Z. Y., Dou, W. Y., Yao, Z. K., Duan, X. C., Chen, Z. F., Zhang, L. J., Zhao, J. L. & Ying, G. G. 2021 Occurrence, removal and mass loads of antiviral drugs in seven sewage treatment plants with various treatment processes. *Water Research* **207**, 117803.

Zhang, J.-N., Chen, J., Yang, L., Zhang, M., Yao, L., Liu, Y.-S., Zhao, J.-S., Zhang, B. & Ying, G.-G. 2021 Occurrence and fate of androg ens, progestogens and glucocorticoids in two swine farms with integrated wastewater treatment systems. *Water Research* **192**, 116836.

Zhao, B., Liu, Y., Dong, W., Li, M., Lu, S., Wang, W. & Fan, Y. 2020 Occurrence and fate of ten sulfonamide antibiotics in typical sewage treatment plants in the City of Jinan of Northeastern China. *Desalination and Water Treatment* **206**, 340–348.

Zhi, S. L., Shen, S. Z., Zhou, J., Ding, G. Y. & Zhang, K. Q. 2020 Systematic analysis of occurrence, density and ecological risks of 45 veterinary antibiotics: focused on family livestock farms in Erhai Lake basin, Yunnan, China. *Environmental Pollution* **267**, 115539.

Zhou, L.-J., Ying, G.-G., Liu, S., Zhao, J.-L., Yang, B., Chen, Z.-F. & Lai, H.-J. 2013 Occurrence and fate of eleven classes of antibiotics in two typical sewage treatment plants in South China. *Science of the Total Environment* **452**, 365–376.

First received 3 November 2021; accepted in revised form 20 January 2022. Available online 17 February 2022