Environ Monit Assess (2021) 193: 98
https://doi.org/10.1007/s10661-021-08887-5

# Contamination, distribution, and risk assessment of antibiotics in the urban surface water of the Pearl River in Guangzhou, South China

**Chang Wang · Yanping Zhao · Shuang Liu · Qin Xiao · Weixin Liang · Yumei Song**

Received: 20 August 2020 / Accepted: 17 January 2021 / Published online: 28 January 2021


**Abstract** To assess the impact of antibiotic pollution to the ecosystem in urban water, the occurrence, seasonal, and spatial distributions, potential sources, and ecological risks of 18 targeted antibiotics in urban river, Pearl River located in Guangzhou city, were investigated. Surface water samples were sampled from 24 sites in Guangzhou center of Pearl River during dry and wet seasons. The results indicated that the concentrations of antibiotic residues were at the nanogram per liter level, except sulfamethazine (SMD) (μg/L). Sulfonamides (SAs) were the dominant antibiotics, contributing 60.4–65.0% to the total antibiotics. The concentrations of SAs, fluoroquinolones (QUs), macrolides (MLs), tetracyclines (TCs), and lincosamides (LCs) were higher in dry season than those in wet season at most sampling sites, which possibly resulted from the dilution effect of heavy rainfall. The concentrations of the antibiotic residues in Guangzhou were comparable or higher than other urban rivers. The calculation on risk quotients indicated that erythromycin-$H_2O$ (ETM-$H_2O$) and tetracycline (TC) were of high risks. The source identification by the Pearson correlation analysis and principal component analysis-multiple linear regression (PCA-MLR) method suggested that municipal wastewater treatment plants were primary sources of antibiotics. These results would provide important information for the environmental protect.

**Keywords** Antibiotics · Ecological risk assessment · Urban River · Pearl River

C. Wang · Y. Zhao · S. Liu · W. Liang · Y. Song (✉)
Guangdong Provincial Key Laboratory of Emergency Test for Dangerous Chemicals, Guangdong Provincial Engineering Research Center for Online Monitoring of Water Pollution, Institute of Analysis, Guangdong Academy of Sciences (China National Analytical Center, Guangzhou), Guangzhou, China
e-mail: symei2011@163.com

C. Wang · Q. Xiao
School of Public Health, Sun Yat-Sen University, Guangzhou 510080, China

## Introduction

Antibiotics are extensively applied to cure diseases, enhance animal growth in many fields, such as livestock, aquaculture and agriculture (Rodriguez-Mozaz et al. 2015; Wang et al. 2013). Many antibiotics are poorly absorbed by the target organisms and the residues cannot be completely eliminated by most wastewater treatment plants (Feng et al. 2019; Liu et al. 2018a; Weng et al. 2020). At least 30% antibiotics enter aquatic ecosystems through agricultural activity, animal waste discharge, sewage effluent or surface runoff (Deng et al. 2016; Kümmerer 2009; Sarmah et al. 2006). Due to the high water solubility, hydrolytical stability, resistance to degradation, the residues can exist for a long time in the environment. China is the world's largest consumer and producer of antibiotics (Van Boeckel et al. 2014), and the residues are widely detected in Chinese aquatic environment, including surface water (Hu et al. 2018; Liu et al. 2018b; Tang et al. 2015), river water (He et al. 2018; Yu et al. 2019), seawater (Niu et al. 2016), municipal sewage (Xu et al. 2015),

groundwater (Chen et al. 2017; Li et al. 2018), and even drinking water (Li et al. 2018). Antibiotic residues have potential risks to both ecosystem and human health (Białk-Bielińska et al. 2011; Khetan and Collins 2007; Tang et al. 2015), by exerting toxicity to organisms at different trophic levels, causing a serious problem of antimicrobial resistance, and contributing to the widespread of antibiotic resistance genes (ARGs) (Reardon 2014; Yang et al. 2013).

The urban rivers serve as the main source of water supply for rural and urban users, irrigated agriculture, ecological landscaping and recreational activities. The water quality is greatly affected by the activities of human beings. Urban rivers have been often polluted by antibiotics and other pollutants, because they always receive massive wastes from rural and urban areas (Chen et al. 2019; Qiao et al. 2018). Several studies demonstrated that the urban rivers unfortunately became reservoirs of antibiotics and antimicrobial resistance (Chen et al. 2019; Marathe et al. 2017). Therefore, to protect the safety of ecosystem and human health, it is of great importance to investigate the occurrence and distribution of antibiotic pollutants in urban rivers.

Guangzhou is one of the most important economic and business centers in South China, with a population of more than 15 million people. Lots of industrial and domestic sewage, livestock, and poultry wastewater run to the urban rivers, Pearl River located in Guangzhou. Various antibiotics, ARGs, and antibiotic-resistant pathogens have been found in the Pearl River Estuary which is nearly located to Guangzhou (Chen et al. 2013b; Liang et al. 2013). However, the studies related pollution in urban rivers, Pearl River in Guangzhou area are few.

This study aimed to investigate the pollution status of the antibiotics in a stretch of Pearl River, a Guangzhou urban river. In this research, the concentrations of 18 target antibiotics, belonging to five major groups: fluoroquinolones (QUs), sulfonamides (SAs), macrolides (MLs), tetracyclines (TCs), and lincosamides (LCs), were detected by ultra-performance liquid chromatography-tandem mass spectrometry (UPLC-MS/MS) technique. Surface water was sampled by the diffusive gradients in thin films (DGT), a method that has been widely applied for collecting water samples with a great advantage of cost-effective time-integrated in situ continuous monitoring. The antibiotic sources were identified by correlation coefficient and principal component analysis (PCA)-based multiple linear regression (MLR) analyses. The ecological risks of antibiotics to aquatic organisms were evaluated based on environmental risk quotients (RQs).

## Materials and methods

### Chemicals and standards

Antibiotic standards of SAs consisting of trimethoprim (TMP), SMD, sulfachloropyridazine (SCP), sulfamethoxazole (SMX), sulfamonmethoxine (SDM), sulfadiazine (SDZ), sulfamerazine (SMR), sulfaquinoxaline (SQX) and sulfapyridine (SPD), and QUs consisting of ofloxacin (OFX), norfloxacin (NFX) and ciprofloxacin (CFX), TCs consisting of oxytetracycline (OTC) and TC, MLs consisting of clarithromycin (CLM), roxithromycin (ROM), and ETM-H2O, and LCs consisting of lincomycin (LIM)] were supplied by Dr. Ehrenstorfer (GmbH, Germany). The physicochemical characteristics of these 18 antibiotics are summarized in Table S1. The internal standards used for SAs, QUs, TCs, MLs, and LIM were sulfamonmethoxine-d3, sulfadiazine-d4, norfloxacin-d5, tetracyclin-d6, and roxithromycin-d7, respectively. Stock solutions (1000 mg/L) and working solutions (10 mg/L) of each tested compound and internal standards were prepared by diluting in methanol and then kept at − 20 °0 methanol and then kept at and internal standards were prepared by dirade.

### Sample collection

Surface water was sampled from 24 sites in urban rivers of Guangzhou, Pearl River (Fig. 1). Sampling sites were chosen to rationally reflect the water quality distribution statues of the river, including the sections that were greatly affected by domestic sewage, industrial wastewater, and aquaculture area. Sampling sites Z1–Z4 were located in the West channel of Guangzhou city; sites Z5–Z11 were located in the Front channel; sites Z12–Z24 were located in the Back channel. Sampling was conducted in August 2018 to represent wet season and December 2018 to represent dry season. All the samples were collected by o-DGTs (LSNC, 0.04 cm XAD gel, 0.076 cm agarose gel, DGT® Research Ltd., China) within 1 week as described previously (Chen et al. 2015). All samples were collected in triplicate from each site.

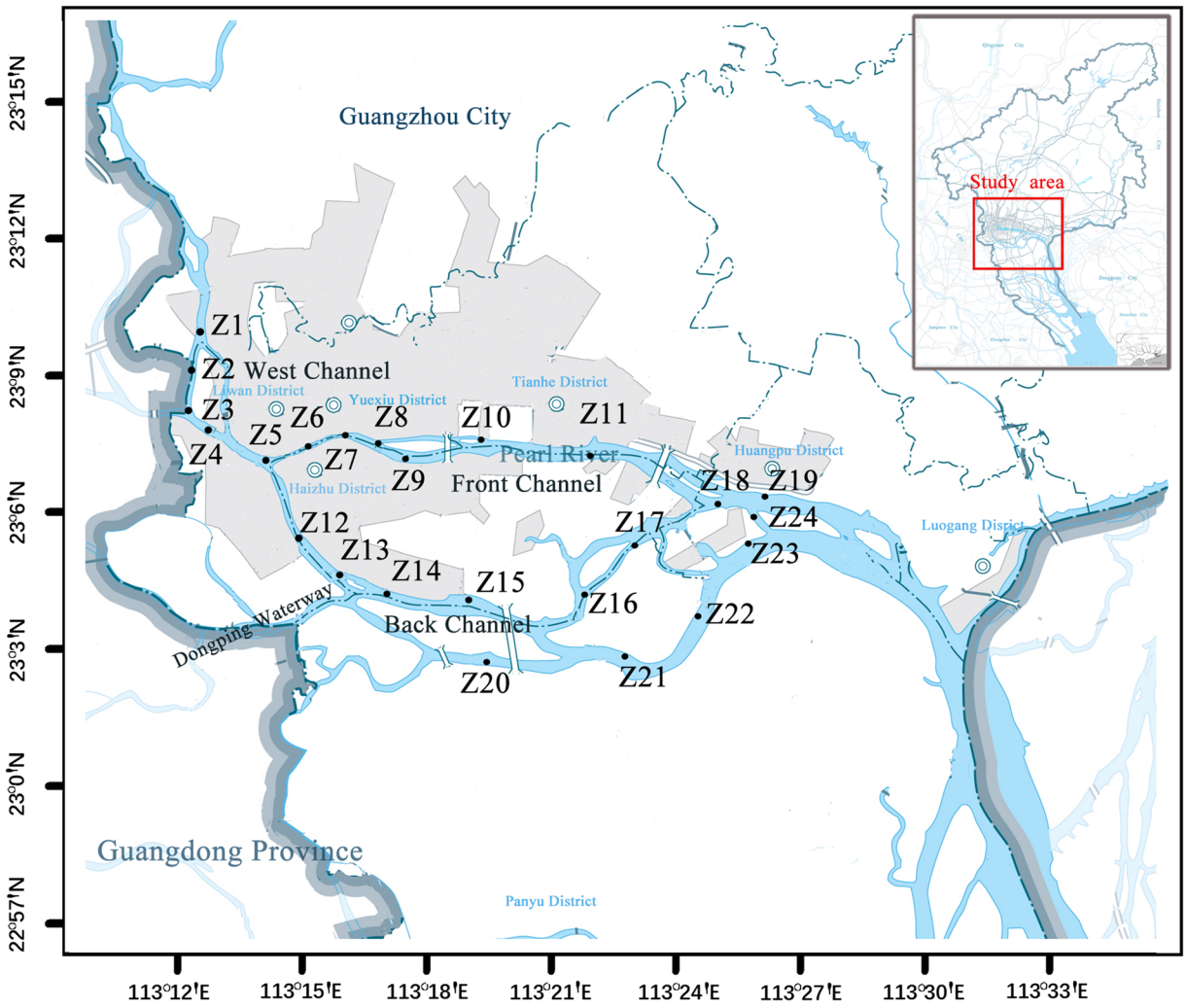


**Fig. 1** Sampling locations of surface water in the Pearl River

## Sample treatment and analyses

After disassembling o-DGTs, the binding gel was transferred into a 15-mL clean amber glass vial. Then, 5 mL of methanol was added and a 20-min extraction was performed in an ultrasonic bath, and this procedure was repeated twice. The combined extract was evaporated to dryness under nitrogen blow down, redissolved in 1 mL of methanol, and then filtered (0.22 μm) into a 2-mL amber vial. The final extracts were spiked with 10 ng of the internal standards before analyzing using UPLC-MS/MS.

The target antibiotics were analyzed by an Agilent 1290 UPLC-MS/MS system. Chromatographic analysis was conducted with the Agilent Poroshell 120 EC-C18 column (2.7 um, 4.6 × 50 mm) at 30 °C, and the injection volume was 10 μL. The conditions for mobile phase were shown in Table S1. UPLC-MS/MS analysis was performed in the multiple reaction monitoring mode using positive electrospray ionization source. The information related to precursor ion, product ion, and other MS/MS operating parameters was demonstrated in Table S1. Linearity was evaluated by constructing a calibration curve (range: 0.05–500 ng/mL) using the ratio of the peak area of the analyte and internal standard.

The concentration of antibiotics measured by o-DGT was calculated using Eq. (1), according to previous studies (Chen et al. 2013a; Chen et al. 2015).

$$C_{\mathrm{DGT}} = M\Delta g/\mathrm{Dt}A \quad (1)$$

where $M$, $\Delta g$, Dt, and $A$ were the mass (g) of antibiotics accumulated in the binding gel, sum of thickness of diffusive gel (0.076 cm), and filter membrane (0.014 cm), diffusion coefficient ($cm^2$/sec) of antibiotics in the gel (Table S1), deployment time (sec), and exposure area (3.14 $cm^2$), respectively.

## Quality control and quality assurance

Detection of antibiotics was carried out according to the quality control and quality assurance (QC/QA) procedures. Solvent and procedure blanks were simultaneously run in sequence to eliminate background noise and monitor system performance. As similar to previous section, the linearity was evaluated by constructing a calibration curve (range: 0.05–500 ng/mL) using the ratio of the peak area of the analyte and internal standard. The correlation coefficients of all calibration curves were greater than 0.99 (Table S1). To assess sensitivity and repeatability, the standard solutions were inserted into the sample sequence every 10 samples. If the variation in the peak area was greater than 15%, fresh standards were analyzed again, and the subsequent samples were quantified according to the new calibration curve. The limits of detection (LOD) and quantification (LOQ) were defined as the analyte concentrations corresponding to the signal-to-noise (S/N) ratios of 3:1 and 10:1, respectively. The LODs and LOQs of each antibiotic in surface water ranged from 0.01 to 2.15 ng/L and from 0.40 to 25.5 ng/L. The recoveries of the 18 antibiotics (40 and 200 ng/L) spiked into the surface water samples ($n$ = 3) ranged from 70.0 to 124.3% (Table S1). The extraction rate was > 95%. The relative standard deviation (RSD) of all the antibiotics was under 10%. Measurement of reagent blanks revealed that the laboratory glassware and analytical system were free of contamination.

## Ecological risk assessment

The potential of contaminants to cause adverse effects was determined according to the environmental risk assessment (ERA) risk (Lee et al. 2008). The values of RQs were estimated by using Equation (2):

$$\mathrm{RQ} = \frac{\mathrm{MEC}}{\mathrm{PNEC}} \tag{2}$$

where MEC and PNEC were the measured environmental concentration and predicted no-effect concentration of each contaminant, respectively. The values of PNEC were calculated by assessment factor method using the following Eq. (3):

$$\mathrm{PNEC} = L(E)C_{50}\ or\ L(N)\mathrm{OEC}/\mathrm{AF} \tag{3}$$

where NOEC, LOEC, $EC_{50}$, and $LC_{50}$ were the no-observed effect concentration, lowest observed effect concentration, median effective concentration, and lethal concentration, respectively. The values of $L$ ($N$) OEC and $L$ ($E$) $C_{50}$ were obtained from the published literature, and the lack of toxicity data was supplemented by EPA Ecosar software. If more than one value was available for the same species, the lowest value was used for the calculation to avoid risk hazards. Assessment factor (AF) was selected according to the evaluation factor in the European Union Technical Guidance Document (TGD) and the situation of biotoxicity. AF for acute toxicity was 1000. AF for chronic toxicity was determined according to the trophic level of aquatic organisms, with the first, second, and third trophic level was 100, 50, and 10, respectively.

In general, the levels of the risk were classified into three groups: (i) high risk (RQ > 1); (ii) medium risk ($0.1 \le \mathrm{RQ} \le 1$); and (iii) low risk (RQ < 0.1) (Hernando et al. 2006; Park and Choi 2008) (Hernando et al. 2006; Park and Choi 2008).

## PCA-MLR model

Source contribution analysis was carried out using PCA-based MLR analysis (Jiang et al. 2016; Li et al. 2012). The MLR equation was expressed as Eq. (4):

$$\widehat{Z}_{\mathrm{sum}} = uB_iFS_i \tag{4}$$

where $\widehat{Z}_{\mathrm{sum}}$ $\widehat{Z}_{\mathrm{sum}}$ $\widehat{Z}_{\mathrm{sum}}$, $B_i$, and $\mathrm{FS}_i$ were the normalized standard deviation of the sum of the antibiotic concentrations ($Z_{\mathrm{sum}}$), regression coefficient value, and factor scores derived from the PCA analysis.

Therefore, the mean percentage contribution was obtained from $B_i/\sum B_i$, and the individual contribution of each source $i$ was calculated using Eq. (5):

$$\text{Contribution of source } i\ (\mathrm{ng/L}) = \mathrm{mean}\ [Z_{\mathrm{sum}}] \times (B_i/\sum B_i) + B_i\sigma\mathrm{FS}_i \tag{5}$$

where $\sigma$ was the standard deviation of the $Z_{\mathrm{sum}}$.

Statistical analysis

The experimental data were calculated using Microsoft Excel 2010. In addition, Pearson's correlation analysis was applied to evaluate the determined correlations with IBM SPSS Statistics (version 22.0). Source apportionment analysis was conducted using PCA–MLR via SPSS. The figures were drawn by OriginPro 2017.

## Results and discussions

### Detection frequencies and concentrations of antibiotics in Guangzhou urban rivers

In total, 18 antibiotics were analyzed (Table 1) and 15 antibiotics were detected in most of the sampling sites. Six SAs (SQX, TMP, SMD, SMR, SDZ, and SPD) and three QUs (OFX, CFX, and NFX) were found in all the sampling sites, while the detection frequency of $LC_S$ was the lowest.

The concentrations of target antibiotics were all in the ng/L level, except SMD that presented in the μg/L level. SAs were the most predominant type of antibiotics, the concentration of which contributed to 60.4–65.0% of the total antibiotics. SMD and ETM-$H_2O$ were with the highest concentrations in the two seasons, suggesting these antibiotics may be used at the largest amount in both seasons. In August, the concentrations of SMD and ETM-$H_2O$ were in the ranges of 25.3–3145 ng/L (mean: 333 ng/L) and ND-550.41 ng/L (mean: 40.9 ng/L), respectively. In December, the respective ranges were 32.4–2385 ng/L (mean: 604 ng/L) and ND-577 ng/L (mean: 137 ng/L).

Nine SAs (SQX, TMP, SCP, SDM, SMD, SMR, SMX, SDZ, and SPD) were detected at most sampling sites in both dry and wet seasons, implying that these antibiotics were used throughout the year. The concentration of sulfa antibiotics' ∑(SAs) in August and December were ND-3530 ng/L and ND-3097 ng/L, respectively, with the average levels were 412 ng/L and 851 ng/L. Notably, the concentrations of SQX and SDM were on the top levels among all SAs, reflecting the widespread use of these antibiotics in the Pearl River basin. SAs were the most common existence in this study, probably due to their extremely high stability in the surface water system (Hu et al. 2018; Yan et al. 2013). The high stability of SAs might be resulted from their low solid sorption coefficients, low organic

**Table 1** Concentration of antibiotics in surface water (ng/L) from Pearl River in wet season and dry season ($n = 24$)

| Group | | SAs | | | | | | | | | QUs | | | MLs | | | LCs | TCs | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Compounds | | SQX | TMP | SCP | SDM | SMD | SMR | SMX | SDZ | SPD | OFX | CFX | NFX | ROM | CLM | ETM-$H_2O$ | LIM | OTC | TC | ∑SAs | ∑QUs | ∑TCs | ∑LCs | ∑MCs |
| Wet season | Min. | 8.07 | 13.8 | NDa | 7.31 | 25.3 | 21.1 | 3.57 | 13.1 | 0.07 | 0.60 | 4.24 | 5.49 | ND | ND | ND | ND | 0.28 | 30.5 | ND | ND | 10.6 | ND | ND |
| | Max. | 155 | 37.0 | 252 | 16.5 | 3150 | 176 | 214 | 19.8 | 31.0 | 19.4 | 8.33 | 9.11 | ND | 19.4 | 550 | ND | 165 | 78.3 | 3530 | 35.4 | 210 | ND | 550 |
| | Avg. | 48.9 | 18.5 | 10.4 | 10.6 | 333 | 36.3 | 16.3 | 14.4 | 5.26 | 3.73 | 5.48 | 6.95 | ND | 1.74 | 40.9 | ND | 25.8 | 47.1 | 412 | 14.2 | 64.3 | ND | 37.3 |
| | Med. | 33.6 | 17.4 | ND | 10.2 | 106 | 26.2 | 5.68 | 13.7 | 1.88 | 1.84 | 5.27 | 6.66 | ND | ND | ND | ND | 9.03 | 45.2 | 198 | 13.0 | 55.1 | ND | ND |
| | Fred.(%) | 100 | 100 | 4 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 100 | 0 | 21 | 17 | 0 | 56 | 88 | 88 | 88 | 100 | 0 | 33 |
| Dry season | Min. | 22.2 | 14.8 | ND | ND | 32.4 | 6.71 | ND | 10.9 | 1.52 | 2.41 | 6.31 | 9.23 | 4.03 | ND | ND | 0.35 | 10.7 | 43.3 | ND | ND | ND | 0.35 | 10.18 |
| | Max. | 520 | 26.3 | ND | 4.03 | 2390 | 171 | 10.6 | 16.7 | 8.20 | 60.3 | 49.9 | 27.2 | 26.7 | ND | 577 | 21.3 | 310 | 90.8 | 3097 | 137 | 355 | 21.3 | 593 |
| | Avg. | 209 | 19.1 | ND | 0.93 | 604 | 37.3 | 1.38 | 12.5 | 3.66 | 13.3 | 14.4 | 13.7 | 13.1 | ND | 137 | 6.52 | 63.6 | 51.4 | 851 | 39.0 | 69.7 | 6.52 | 150 |
| | Med. | 191 | 19.1 | ND | ND | 540 | 27.1 | ND | 11.9 | 3.17 | 9.89 | 12.2 | 12.1 | 12.5 | ND | 93.1 | 4.49 | 29.6 | 46.6 | 785 | 33.4 | 52.2 | 4.49 | 112 |
| | Fred.(%) | 100 | 100 | 0 | 40 | 100 | 100 | 40 | 100 | 100 | 100 | 100 | 100 | 100 | 0 | 84 | 100 | 40 | 96 | 96 | 96 | 96 | 100 | 100 |

[a] Not detected ( Concentrations < LOQ)

carbon-normalized sorption coefficients (Koc), and low sorption affinity to soils and sediments (Table S2).

OFX, OFL, and NFX in the group of QUs showed 100% detection frequencies. QUs were a class of antibacterial compounds employed excessively in both veterinary and human medicine, and exhibited large chemical stability (Thiele-Bruhn 2003; Wammer et al. 2013). Therefore, the QUs were widespread in the aquatic environment.

MLs were large complex molecules with large numbers of stereocenters and were widely used for humans (Murata et al. 2011; Stepanić et al. 2012; Yan et al. 2013). The average detection frequencies of three MLs (ROM, CLM, and ETM-$H_2O$) ranged from 11 to 50%. The lower detection rates of the three macrolides in the surface water system might be due to their high hydrophobicity and strong sorption affinity to sediments (Huang et al. 2011; Yan et al. 2013).

Normally TCs were seldom found in water samples due to their interaction with cations and strong binding capacity to particulate matter (Díaz-Cruz et al. 2003; MacKay and Canterbury 2005). However, in the group of TCs, TC, and OTC were detected in the most sites during the two seasons. The concentrations of TC and OTC were ND-90.8 ng/L and ND-310 ng/L, respectively. OTC exhibited a high residue level with a detection frequency of more than 56%, which may be due to the excessive usage of OTC in both residential and industrial areas (Van Rennings et al. 2015).

The comparison of the antibiotic concentrations in the urban river located in Guangzhou and the other rivers in domestic and abroad were shown in Table 2. The concentrations of SAs in the Guangzhou urban river (ND-3150 ng/L) were higher than those in other domestic and abroad rivers, except the Tagus River (ND-5960 ng/L). The concentrations of ∑QUs showed the following sequence: Wenyu River (25.1–1440 ng/L) > Tagus River (2.5–786 ng/L) > Yellow River (675 ng/L) > Liao River (10.2–393 ng/L) > Ter River (232–377 ng/L) > Pearl River (ND-137 ng/L) > Yangtze River (10.5–105 ng/L) > Hong Kong River (ND-101 ng/L) > Hanjiang (ND-20.4 ng/L) > Songhua River (ND-8.50 ng/L) > Yitong River (ND-1.36 ng/L). ∑MLs concentrations (ND-604 ng/L) were similar to those in the Yangtze River (7.63–542 ng/L), Liao River (17.4–496 ng/L), and Yellow River (458 ng/L, China), but significantly higher than those in the Hong Kong River (ND-2.8 ng/L, China) and Chester River (3.4 ng/L) and Songhua River (ND-7.3 ng/L). ∑TCs concentrations showed the following sequence: Liao River (ND-850 ng/L) > Pearl River (30.8–401 ng/L) > Wenyu River (ND-201 ng/L) > Yangtze River (26.4–41.9 ng/L) > Hongkong River (ND-31.5 ng/L) > Hanjiang (ND-21.8 ng/L) > Soeste River (21 ng/L) > Yellow River (7.55 ng/L) > Yitong River (ND-0.29 ng/L). Overall, the concentrations of the residues in Guangzhou river were comparable or higher than other urban rivers. However, it should be noted that the sampling sites form the reference may include these locate the upstream far away from city, which may result in the lower concentrations. Comparing the usage values for the antibiotics with the other countries such as Germany, the USA, and Spain, China showed higher per capital consumptions of antibiotics. Meanwhile, the Pearl River Basin located in South China showed the highest emission densities in China (Zhang et al. 2015). Therefore, the concentration range and detection frequency of the antibiotics in this work were relatively high.

## Seasonal and spatial distribution of antibiotics in Guangzhou

Figures 2 and 3 showed the seasonal distribution pattern of five groups of antibiotics at 24 sampling sites. The concentrations of antibiotics were higher in dry season for all sampling sites, except site *Z*3 and *Z*21. The number and frequency of antibiotics in the dry season were comparable to those in the wet season. Up to 11 antibiotics (61%) exhibited 100% frequency in both seasons. Our results showed that the predominant antibiotics were the same in both seasons. SMD and OFX were the dominant compounds for SAs and QUs in the two seasons at sampling sites, whereas ETM-$H_2O$ and OTC were the dominant compounds for MLs and TCs group, which were consistent with the usage of various antibiotics in China (Zhang et al. 2015).

The ranges of surface water temperature in wet and dry seasons were 30.1–32.4 °2 and 16.7–19.8 °9, respectively. The most concentrations were higher in dry season than those in wet season, which demonstrated that the concentrations of antibiotics were higher at cold weather than those at warm weather and was consistent with previous studies (Jiang et al. 2011; Yan et al. 2013; Yang et al. 2011; Zhang et al. 2020). The stream flow of the Pearl River in the two sampling seasons was different, and the higher flow in wet season diluted the antibiotics and thus reduced their concentrations. Moreover, the capability to decompose the antibiotics microbes in sewage treatment system in cool water in dry season was lower than warm

**Table 2** Concentrations of main antibiotic classes in the Pearl River and other regions

| Sampling locations | Concentrations (ng/L) | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| | SQX | SMD | SDZ | SMR | SMX | TMP | OFX | CFX |
| Pearl River, Guangzhou, China | 8.07–520 | 25.3–3150 | 10.9–19.83 | 6.71–176 | ND-213 | 13.8–37.0 | 0.60–60.3 | 4.24–49.9 |
| Yitong River, Changchun, China | NAb | NDa-0.39 | ND-0.44 | ND-1.08 | ND-0.55 | ND-0.39 | ND-1.36 | ND-0.62 |
| Songhua River, Jilin, China | ND-3.9 | NA | NA | NA | ND-17.1 | NA | ND-8.0 | ND-0.5 |
| Yangtze River, Chongqing, China | NA | 11.4–68.4 | 27.9–359 | NA | 3.51–173 | 12.3–210 | 10.8–105 | NA |
| Yangtze Estuary, Shanghai, China | 0.4–74.1 | 1.90–206 | 0.40–242 | ND-1.60 | 8.4–62.5 | NA | NA | NA |
| Liao River, Jilin, China | NA | NA | ND-1.5 | ND-10.8 | ND-44.1 | NA | 10.2–285 | ND-23.7 |
| Hanjiang, Hubei, China | NA | NA | ND-24 | ND-14 | ND-22 | ND-9.6 | ND-9.50 | ND-4.5 |
| Hongkong River, Hong Kong, China | NA | ND-79.9 | NA | NA | ND-29.9 | NA | ND-75.5 | ND-25.7 |
| Wenyu River, Beijing, China | ND-2.1 | ND-267 | 46.9–739 | NA | 33.2–528 | 12.8–120 | 25.1–1214 | ND-24.1 |
| Yellow River, China | 1.26 | 29.5 | 75.4 | NA | 14.1 | 19.0 | 9.72 | 16.2 |
| Soeste River, Cloppenburg, Germany | NA | NA | 147 | NA | 114 | 62 | NA | NA |
| Tagus River, Madrid, Spain | NA | NA | NA | NA | 0.1–5960 | ND-1288 | NA | 2.5–786 |
| Ter River, Catalonia, Spain | NA | NA | NA | NA | 14–67 | 7–36 | 86–186 | 146–191 |
| Chester River, Maryland, USA | NA | NA | NA | NA | 14.8 | NA | NA | NA |

| Sampling locations | Concentrations (ng/L) | | | | | References |
|---|---|---|---|---|---|---|
| | NFX | OTC | TC | ERM-$H_2O$ | ROM | |
| Pearl River, Guangzhou, China | 5.49–27.2 | 0.28–310 | 30.5–90.8 | ND-577 | ND-26.7 | The present study |
| Yitong River, Changchun, China | NA | ND-0.02 | ND-0.27 | NA | NA | (Yu et al. 2019) |
| Songhua River, Jilin, China | ND | NA | NA | ND-7.3 | NA | (He et al. 2018) |
| Yangtze River, Chongqing, China | ND | 6.14–15.6 | 20.3–26.3 | NA | NA | (Yan et al. 2018) |
| Yangtze Estuary, Shanghai, China | NA | NA | NA | 7.43–535 | 0.20–6.5 | (Zhao et al. 2017) |
| Liao River, Jilin, China | ND-83.8 | ND-835 | ND-14.6 | 12.0–366 | 5.40–130 | (Dong et al. 2016) |
| Hanjiang, Hubei, China | ND-6.40 | ND-9.80 | ND-12.0 | NA | NA | (Hu et al. 2018) |
| Hongkong River, Hong Kong, China | NA | NA | ND-31.5 | NA | ND – 2.80 | (Deng et al. 2018) |
| Wenyu River, Beijing, China | ND-199 | ND-110 | ND-90.7 | NA | NA | (Zhang et al. 2014) |
| Yellow River, China | 649 | 4.46 | 3.09 | 453 | 4.87 | (Zhang et al. 2015) |
| Soeste River, Cloppenburg, Germany | NA | NA | 21.0 | 86 | NA | (Burke et al. 2016) |
| Tagus River, Madrid, Spain | NA | NA | NA | NA | NA | (Arenas-Sánchez et al. 2019) |
| Ter River, Catalonia, Spain | NA | NA | NA | NA | NA | (Lekunberri et al. 2017) |
| Chester River, Maryland, USA | 94.1 | NA | NA | ND | 3.40 | (He et al. 2019) |

[a] Under detection limit

[b] Not analyzed

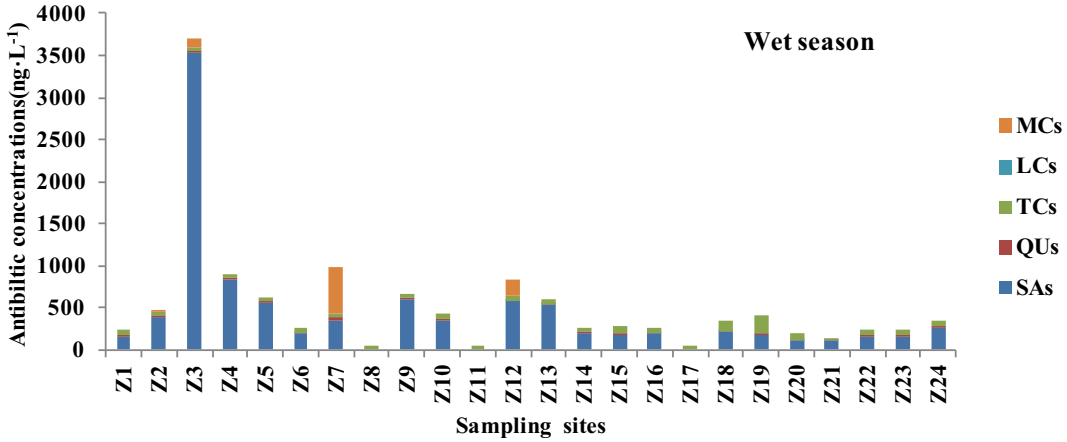


**Fig. 2** Distribution of five classes of 18 antibiotics concentrations in the urban surface water of Pearl River in the wet season

water in wet season, leading to the levels of antibiotics higher in cold conditions (Kim and Carlson 2007). Seasonal differences in antibiotic concentrations were associated with seasonal consumption of antibiotics in this region, but the correlation between antibiotic consumption and season was weak (Zhang et al. 2020).

Generally, the total concentrations of antibiotics at different locations could be arranged in the following order: West channel (Z1-Z4, mean: 479-3520 ng/L) > Front channel (Z5-Z11, mean: 445-1262 ng/L) > Back channel (Z12-Z24, mean: 97-1040 ng/L). The individual spatial distribution of five types of antibiotics showed similar characteristics with that of total antibiotics. The highest total concentration was found at Z3 and Z4 in dry season (3338 ng/L and 2736 ng/L) and wet season (3703 ng/Land 904 ng/L). The water in the west channel and the front channel flowed through the downtown with a large population and high emission of domestic sewage, which could be the main reason for the higher concentrations of antibiotics in west channel and front channel. Additionally, the concentration of antibiotics was higher in West channel compared with Front channel might be due to the former was affected by both industrial and domestic sewage (Huang et al. 2019). While for the back channel, antibiotic concentration was reduced by the inputs of fresh water from Dongping Waterway and increased sewage treatment systems nearby.

## Ecological risk assessment of antibiotics

In this work, the MEC and PNEC values of each antibiotic for aquatic organisms (e.g. algae, daphnia, and fish) were used to calculate the RQs and evaluate the ecological risks. The PNEC values were obtained from previous research on thechronic or acute toxicity of

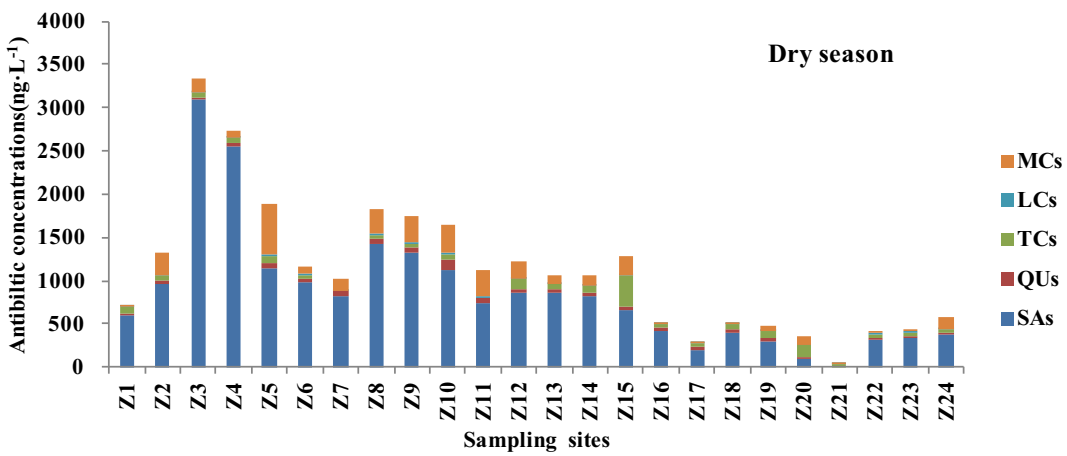


**Fig. 3** Distribution of five classes of 18 antibiotics concentrations in the urban surface water of Pearl River in the dry season

aquatic organisms, and the detailed toxicity data and PNEC values were presented in the Table S2.

As shown in Fig. 4, the results of risk assessment revealed that most RQs were below 1.0, which indicated medium-low risks to organisms at different trophic levels during the wet and dry season. However, among the 18 antibiotics, ETM-$H_2O$ showed relatively higher RQ, which suggested its high risk to aquatic organisms, especially to Anabaena sp. ETM-$H_2O$ was the main form of ETM in surface water which was easily enriched in sediments because of their high Koc and octanol-water partition coefficient (Log Kow) (Table S3). The release of ETM-$H_2O$ from sediments resulted in long-term exposure of aquatic organisms, which may cause the chronic and combined risks. It was reported that ETM posed medium to high risk in the Haihe River and the Songhua River (Chen et al. 2018). Compared to these two major rivers in China, the ecological risk of ETM-$H_2O$ in the Guangzhou urban river, Pearl River was higher. Therefore, more attention should be paid for the contamination from ETM-$H_2O$ in Guangzhou urban rivers.

The analytical results revealed that RQs of all antibiotics showed clear seasonal signatures, except SDM, SMX, and CLM, with higher values in the dry season (Fig. 5). ETM-$H_2O$ was the dominant ecological risk contributor in the sampling sites followed by TC. The remaining 16 antibiotics caused medium-low risk. Overall, about 79.2% sites faced a high risk, which were mostly located in west channel, front channel, and upper reaches of the back channel, due to the presence of ETM-$H_2O$ and TC. 20.8% sites faced a medium risk (sample sites Z16–18 and Z22–23), which were located near the Guangzhou university town. Most of these antibiotics posed high risks at Z5 and Z7, because these two sites were located near the heavily polluted city drains and the hospitals, respectively. The control measures for antibiotic pollution risks in this urban water should be made based on the seasonal and spatial distribution characteristics of antibiotic types with high risk.

## Possible sources of antibiotics

The source contribution was identified by PCA, followed by MLR. The results of PCA were presented in Fig. 6 and Table 3. Six principal components (PC1-6, eigenvalue of > 1) were obtained after varimax rotation, and each component accounted for 23.4%, 20.4%, 11.9%, 11.5%, 10.9%, and 6.24% of the total variance, respectively, which indicated the presence of multiple sources of antibiotic residues. In the sampling sites, there are 26 large and medium-sized hospitals, 23 small and medium-sized pharmaceutical enterprises, 6 municipal wastewater treatment plants with a daily handling capacity of 50,000 tons, 4 large-scale aquaculture farms, and 1 livestock farm, mainly distributed in the front channel and the upper reaches of the back channel. PC1contained the high factor loadings of five antibiotics, such as SQX, SDM, SMR, SDZ, and SMD, which were often detected with high concentration in the municipal sewage treatment plants (Wei et al. 2018). Moreover, the correlation analysis showed that these five

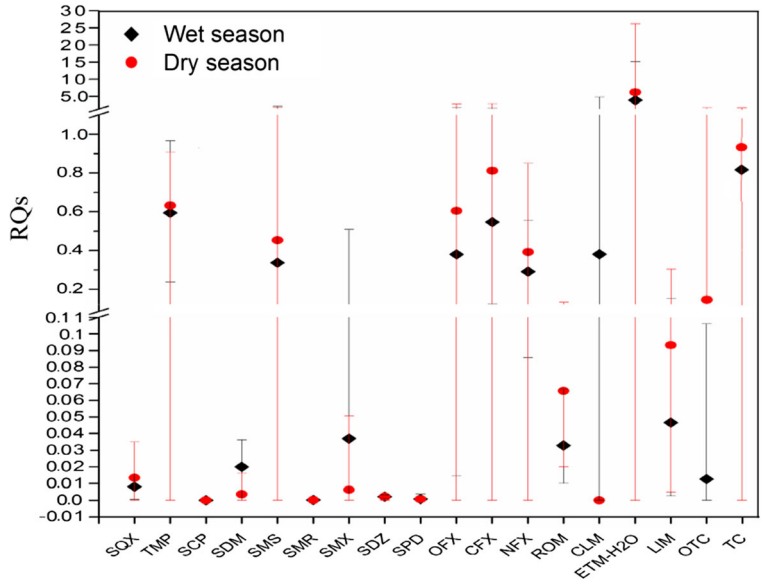


**Fig. 4** Risk quotients (RQs) of target antibiotics in the surface water of the Pearl River in the wet and dry seasons. Different symbols represent the mean RQs for two seasons, and the bars represent their maximum and minimum RQs

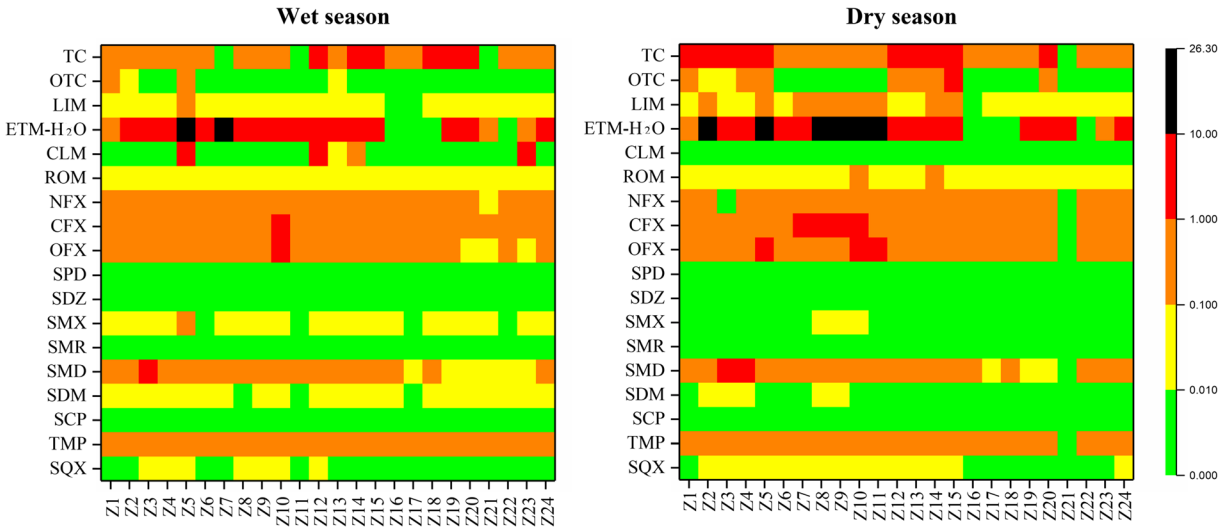


**Fig. 5** Ecological risk assessment of the antibiotics in the river water at each sampling site in the wet and dry season.

antibiotics were significantly correlated and thus they were probably from the same source (Table S4). Thus, PC1 indicated the municipal wastewater treatment plants. PC2 was mainly associated with OFX, CFX, and NFX. The concentration of OFX was positively correlated with CFX and NFX (Table S4). OFX was commonly used in the treatment of human diseases and often detected in hospital sewage (Chen et al. 2012; Dinh et al. 2017; Liu et al. 2017). Furthermore, OFX was also the main type of antibiotic discharged by pharmaceutical factory of Guangzhou (Wei et al. 2018). Therefore, PC2 represented hospital wastewater and pharmaceutical wastewater. PC3 was characterized by high loadings of CLM and SPD. CLM was only allowed to be used to treat human diseases in China (Xue et al. 2013) and SPD was mainly discharged from hospitals (Wei et al. 2018). Consequently, PC3 indicated the source of hospital wastewater. PC4 was only highly loaded by SMX, that was a commonly used veterinary drug (Chen et al. 2017; Selvam et al. 2017; Spielmeyer et al. 2017; Wei et al. 2018). SMX was often found in aquaculture wastewater and thus PC4 represented the source of aquaculture wastewater. The profile of PC5 showed high loadings of TC and OTC. These compounds were frequently employed to treat and prevent animal diseases as well as enhance animal growth due to their broad spectrum and low price (Bound and Voulvoulis

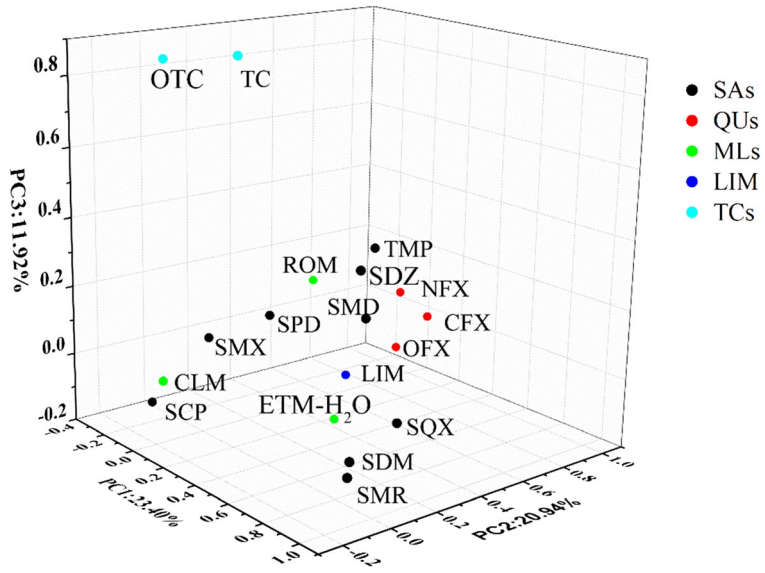


**Fig. 6** The principal component analysis diagram for antibiotics in the Pearl River

**Table 3** Varimax-rotated component matrix following principal component analysis of all water samples

| Antibiotics | Component | | | | | |
|---|---|---|---|---|---|---|
| | 1 | 2 | 3 | 4 | 5 | 6 |
| SQX | 0.863 | 0.188 | 0.129 | 0.155 | − 0.045 | − 0.173 |
| TMP | 0.479 | 0.460 | 0.595 | 0.017 | 0.333 | 0.156 |
| SCP | − 0.105 | − 0.079 | − 0.084 | − 0.031 | − 0.058 | 0.866 |
| SDM | 0.744 | 0.212 | 0.152 | 0.138 | 0.212 | 0.273 |
| SMD | 0.940 | − 0.093 | − 0.057 | 0.065 | − 0.117 | − 0.159 |
| SMR | 0.958 | − 0.091 | − 0.055 | 0.029 | − 0.072 | − 0.118 |
| SMX | 0.060 | − 0.006 | 0.077 | 0.910 | 0.091 | 0.087 |
| SDZ | 0.781 | 0.155 | 0.349 | − 0.019 | 0.354 | 0.230 |
| SPD | 0.176 | 0.174 | 0.832 | 0.109 | 0.131 | − 0.110 |
| OFX | 0.043 | 0.887 | − 0.079 | 0.146 | − 0.099 | − 0.074 |
| CFX | 0.231 | 0.904 | 0.056 | 0.057 | 0.023 | 0.027 |
| NFX | 0.049 | 0.931 | 0.098 | 0.084 | 0.076 | 0.101 |
| ROM | − 0.224 | 0.652 | 0.087 | 0.248 | 0.091 | − 0.258 |
| CLM | − 0.055 | − 0.134 | 0.913 | 0.084 | − 0.047 | − 0.035 |
| ETM-$H_2O$ | 0.212 | 0.423 | 0.204 | 0.689 | − 0.200 | − 0.167 |
| LIM | 0.129 | 0.549 | 0.014 | 0.779 | − 0.102 | − 0.114 |
| OTC | − 0.135 | − 0.050 | − 0.003 | 0.024 | 0.867 | − 0.089 |
| TC | 0.192 | 0.063 | 0.128 | − 0.078 | 0.896 | 0.028 |
| Eigen value | 5.789 | 3.320 | 2.246 | 1.661 | 1.228 | 1.047 |
| Percentage variance explained (%) | 23.40 | 20.94 | 11.92 | 11.54 | 10.90 | 6.24 |

Extraction method: Principal component analysis

Rotation method: Varimax with Kaiser normalization

2004), and were frequently detected in aquaculture farms (Wei et al. 2018). Therefore, PC5 could represent the contribution from aquaculture farms and livestock farms. PC6 correlated only with SCP. Because this compound was commonly used in veterinary drugs, PC6 would represent aquaculture wastewater. In summary, there were five potential sources of antibiotics including municipal wastewater treatment plants, hospitals, pharmaceutical factories, aquaculture, and livestock farms.

The resulting equation by PCA-MLR model was as follows:

$$\widehat{Z}_{\mathrm{sum}} = 0.976\mathrm{FS}_1 + 0.008\mathrm{FS}_2 + 0.137\mathrm{FS}_3 + 0.027\mathrm{FS}_4 + 0.067\mathrm{FS}_5 + 0.088\mathrm{FS}_6 (R2 = 0.985)$$

By expanding $\widehat{Z}_{\mathrm{sum}}$ , the MLR equation was written as:

$$Z_{\mathrm{sum}} = 0.976\,\sigma\,\mathrm{FS}_1 + 0.008\,\sigma\,\mathrm{FS}_2 + 0.137\sigma\mathrm{FS}_3 + 0.027\,\sigma\,\mathrm{FS}_4 + 0.067\,\sigma\,\mathrm{FS}_5 + 0.088\,\sigma\,\mathrm{FS}_6 + \mathrm{mean}[Z_{\mathrm{sum}}]$$

where $\sigma$ and mean $[Z_{\mathrm{sum}}]$ were 698.80 ng/L and 826.98 ng/L, respectively. Notably, the six components (PC1–6) were retained, and the mean percentage contribution Bk/∑Bk was 74.9% for municipal wastewater treatment plants (Factor 1), 11.1% for hospitals and pharmaceutical factories (Factor 2 and Factor 3), and 14.0% for livestock farms and aquaculture source (Factor 4, Factor 5, and Factor 6). Therefore, municipal wastewaters could serve as the primary sources for antibiotic pollution in Guangzhou urban rivers.

## Conclusions

This study investigated the contamination situation, seasonal variation, spatial distribution, ecological risks, and potential sources of 18 antibiotics in the urban rivers, Pearl River located in Guangzhou. The concentration of antibiotic residues was at the nanogram per liter level, except SMD (ug/L). SAs were the predominant antibiotics, contributing to 60.4–65.0% of the total antibiotics. SAs, QUs, MLs, LCs, and TCs concentrations in dry season were markedly higher than those in wet season at most sampling sites. The concentration of the residues in Guangzhou was present at high or moderate-average levels compared to other urban rivers. Among these antibiotics, the ETM-$H_2O$ and TC posed high risks to aquatic organisms. The primary source of these antibiotics was municipal wastewater treatment plants. These results would be helpful for the formulation of pollution control and mitigation policies on antibiotics in this area.

**Supplementary Information** The online version contains supplementary material available at https://doi.org/10.1007/s10661-021-08887-5.

**Acknowledgements** The authors sincerely thank Professor Jingwei Xu (Changchun Institute of Applied Chemistry, Chinese Academy of Sciences) for the contribution to this article.

**Funding** This research was supported by National Natural Science Foundation of China (No. 21777150), Guangdong Provincial Key R&D Programme (2020B1111350002) and Guangzhou Science and Technology Project (No. 201803030042).

## References

Arenas-Sánchez, A., Rico, A., Rivas-Tabares, D., Blanco, A., Garcia-Doncel, P., Romero-Salas, A., et al. (2019). Identification of contaminants of concern in the upper Tagus river basin (central Spain). Part 2: spatio-temporal analysis and ecological risk assessment. *Science of the Total Environment, 667*, 222–233.

Białk-Bielińska, A., Stolte, S., Arning, J., Uebers, U., Böschen, A., Stepnowski, P., & Matzke, M. (2011). Ecotoxicity evaluation of selected sulfonamides. *Chemosphere, 85*(6), 928–933.

Bound, J. P., & Voulvoulis, N. (2004). Pharmaceuticals in the aquatic environment—a comparison of risk assessment strategies. *Chemosphere, 56*(11), 1143–1155.

Burke, V., Richter, D., Greskowiak, J., Mehrtens, A., Schulz, L., & Massmann, G. (2016). Occurrence of antibiotics in surface and groundwater of a drinking water catchment area in Germany. *Water Environment Research, 88*(7), 652–659.

Chen, H., Li, X., & Zhu, S. (2012). Occurrence and distribution of selected pharmaceuticals and personal care products in aquatic environments: a comparative study of regions in China with different urbanization levels. *Environmental Science and Pollution Research, 19*(6), 2381–2389.

Chen, C.-E., Zhang, H., Ying, G.-G., & Jones, K. C. (2013a). Evidence and recommendations to support the use of a novel passive water sampler to quantify antibiotics in wastewaters. *Environmental Science & Technology, 47*(23), 13587–13593.

Chen, Y., Leung, K. S.-Y., Wong, J. W.-C., & Selvam, A. (2013b). Preliminary occurrence studies of antibiotic residues in Hong Kong and Pearl River Delta. *Environmental Monitoring and Assessment, 185*(1), 745–754.

Chen, C.-E., Chen, W., Ying, G.-G., Jones, K. C., & Zhang, H. (2015). In situ measurement of solution concentrations and fluxes of sulfonamides and trimethoprim antibiotics in soils using o-DGT. *Talanta, 132*, 902–908.

Chen, K.-L., Liu, L.-C., & Chen, W.-R. (2017). Adsorption of sulfamethoxazole and sulfapyridine antibiotics in high organic content soils. *Environmental Pollution, 231*, 1163–1171.

Chen, L., Lang, H., Liu, F., Jin, S., & Yan, T. (2018). Presence of antibiotics in shallow groundwater in the Northern and Southwestern Regions of China. *Groundwater, 56*(3), 451–457.

Chen, H., Bai, X., Li, Y., Jing, L., Chen, R., & Teng, Y. (2019). Characterization and source-tracking of antibiotic resistomes in the sediments of a peri-urban river. *Science of The Total Environment, 679*, 88–96.

Deng, W., Li, N., Zheng, H., & Lin, H. (2016). Occurrence and risk assessment of antibiotics in river water in Hong Kong. *Ecotoxicology and Environmental Safety, 125*, 121–127.

Deng, W.-J., Li, N., & Ying, G.-G. (2018). Antibiotic distribution, risk assessment, and microbial diversity in river water and sediment in Hong Kong. *Environmental Geochemistry and Health, 40*(5), 2191–2203.

Díaz-Cruz, M. S., López de Alda, M. J., & Barceló, D. (2003). Environmental behavior and analysis of veterinary and human drugs in soils, sediments and sludge. *TrAC Trends in Analytical Chemistry, 22*(6), 340–351.

Dinh, Q., Moreau-Guigon, E., Labadie, P., Alliot, F., Teil, M.-J., Blanchard, M., Eurin, J., & Chevreuil, M. (2017). Fate of antibiotics from hospital and domestic sources in a sewage network. *Science of The Total Environment, 575*, 758–766.

Dong, D., Zhang, L., Liu, S., Guo, Z., & Hua, X. (2016). Antibiotics in water and sediments from Liao River in Jilin Province, China: occurrence, distribution, and risk assessment. *Environmental Earth Sciences, 75*(16), 1202.

Feng, Y., Qing, W., Kong, L., Li, H., Wu, D., Fan, Y., Lee, P. H., & Shih, K. (2019). Factors and mechanisms that influence the reactivity of trivalent copper: a novel oxidant for selective degradation of antibiotics. *Water Research, 149*, 1–8.

He, S., Dong, D., Zhang, X., Sun, C., Wang, C., Hua, X., Zhang, L., & Guo, Z. (2018). Occurrence and ecological risk assessment of 22 emerging contaminants in the Jilin Songhua River (Northeast China). *Environmental Science and Pollution Research, 25*(24), 24003–24012.

He, K., Hain, E., Timm, A., Tarnowski, M., & Blaney, L. (2019). Occurrence of antibiotics, estrogenic hormones, and UV-filters in water, sediment, and oyster tissue from the Chesapeake Bay. *Science of The Total Environment, 650*, 3101–3109.

Hernando, M., Mezcua, M., Fernandezalba, A., & Barcelo, D. (2006). Environmental risk assessment of pharmaceutical residues in wastewater effluents, surface waters and sediments. *Talanta, 69*(2), 334–342.

Hu, Y., Yan, X., Shen, Y., Di, M., & Wang, J. (2018). Antibiotics in surface water and sediments from Hanjiang River, Central China: Occurrence, behavior and risk assessment. *Ecotoxicology and Environmental Safety, 157*, 150–158.

Huang, C. H., Renew, J. E., Smeby, K. L., Pinkston, K., & Sedlak, D. L. (2011). Assessment of potential antibiotic contaminants in water and preliminary occurrence analysis. *Journal of Contemporary Water Research & Education, 120*(1), 30–40.

Huang, Y. H., Liu, Y., Du, P. P., Zeng, L. J., Mo, C. H., Li, Y. W., et al. (2019). Occurrence and distribution of antibiotics and antibiotic resistant genes in water and sediments of urban rivers with black-odor water in Guangzhou, South China. *Science of The Total Environment, 670*, 170–180.

Jiang, L., Hu, X., Yin, D., Zhang, H., & Yu, Z. (2011). Occurrence, distribution and seasonal variation of antibiotics in the Huangpu River, Shanghai, China. *Chemosphere, 82*(6), 822–828.

Jiang, J. J., Lee, C. L., Brimblecombe, P., Vydrova, L., & Fang, M. D. (2016). Source contributions and mass loadings for chemicals of emerging concern: chemometric application of pharmaco-signature in different aquatic systems. *Environmental Pollution, 208*, 79–86.

Khetan, S. K., & Collins, T. J. (2007). Human pharmaceuticals in the aquatic environment: a challenge to green chemistry. *ChemInform, 38*(36).

Kim, S. C., & Carlson, K. (2007). Temporal and spatial trends in the occurrence of human and veterinary antibiotics in aqueous and river sediment matrices. *Environmental Science and Technology, 41*(1), 50–57.

Kümmerer, K. (2009). Antibiotics in the aquatic environment – a review – Part I. *Chemosphere, 75*(4), 417–434.

Lee, Y.-J., Lee, S.-E., Lee, D. S., & Kim, Y.-H. (2008). Risk assessment of human antibiotics in Korean aquatic environment. *Environmental Toxicology and Pharmacology, 26*(2), 216–221.

Lekunberri, I., Villagrasa, M., Balcázar, J. L., & Borrego, C. M. (2017). Contribution of bacteriophage and plasmid DNA to the mobilization of antibiotic resistance genes in a river receiving treated wastewater discharges. *Science of The Total Environment, 601-602*, 206–209.

Li, W. H., Tian, Y. Z., Shi, G. L., Guo, C. S., Li, X., & Feng, Y. C. (2012). Concentrations and sources of PAHs in surface sediments of the Fenhe reservoir and watershed, China. *Ecotoxicology and Environmental Safety, 75*(1), 198–206.

Li, A., Chen, L., Zhang, Y., Tao, Y., Xie, H., Li, S., Sun, W., Pan, J., He, Z., Mai, C., Fan, Y., Xian, H., Zhang, Z., & Wen, D. (2018). Occurrence and distribution of antibiotic resistance genes in the sediments of drinking water sources, urban rivers, and coastal areas in Zhuhai, China. *Environmental Science and Pollution Research, 25*(26), 26209–26217.

Liang, X. M., Nie, X. P., & Shi, Z. (2013). Preliminary studies on the occurrence of antibiotic resistance genes in typical aquaculture area of the Pearl River Estuary. *Journal of Environmental Sciences (China)., 34*(10), 4073–4080.

Liu, S., Zhao, H., Lehmler, H.-J., Cai, X., & Chen, J. (2017). Antibiotic pollution in marine food webs in Laizhou Bay, North China: trophodynamics and human exposure implication. *Environmental Science & Technology, 51*(4), 2392–2400.

Liu, J.-J., Diao, Z.-H., Liu, C.-M., Jiang, D., Kong, L.-J., & Xu, X. R. (2018a). Synergistic reduction of copper (II) and oxidation of norfloxacin over a novel sewage sludge-derived char-based catalyst: performance, fate and mechanism. *Journal of Cleaner Production, 182*, 794–804.

Liu, X., Lu, S., Guo, W., Xi, B., & Wang, W. (2018b). Antibiotics in the aquatic environments: a review of lakes, China. *Science of The Total Environment, 627*, 1195–1208.

MacKay, A. A., & Canterbury, B. (2005). Oxytetracycline sorption to organic matter by metal-bridging. *Journal of Environmental Quality, 34*(6), 1964–1971.

Marathe, N. P., Pal, C., Gaikwad, S. S., Jonsson, V., Kristiansson, E., & Larsson, D. G. J. (2017). Untreated urban waste contaminates Indian river sediments with resistance genes to last resort antibiotics. *Water Research, 124*, 388–397.

Murata, A., Takada, H., Mutoh, K., Hosoda, H., Harada, A., & Nakada, N. (2011). Nationwide monitoring of selected antibiotics: distribution and sources of sulfonamides, trimethoprim, and macrolides in Japanese rivers. *Science of The Total Environment, 409*(24), 5305–5312.

Niu, Z.-G., Zhang, K., & Zhang, Y. (2016). Occurrence and distribution of antibiotic resistance genes in the coastal area of the Bohai Bay, China. *Marine Pollution Bulletin, 107*(1), 245–250.

Park, S., & Choi, K. (2008). Hazard assessment of commonly used agricultural antibiotics on aquatic ecosystems. *Ecotoxicology, 17*(6), 526–538.

Qiao, M., Ying, G.-G., Singer, A. C., & Zhu, Y.-G. (2018). Review of antibiotic resistance in China and its environment. *Environment International, 110*, 160–172.

Reardon, S. (2014). Antibiotic resistance sweeping developing world. *Nature, 509*(7499), 141–142.

Rodriguez-Mozaz, S., Chamorro, S., Marti, E., Huerta, B., Gros, M., Sànchez-Melsió, A., Borrego, C. M., Barceló, D., & Balcázar, J. L. (2015). Occurrence of antibiotics and antibiotic resistance genes in hospital and urban wastewaters and their impact on the receiving river. *Water Research, 69*, 234–242.

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.

Selvam, A., Kwok, K., Chen, Y., Cheung, A., Leung, K. S. Y., & Wong, J. W. C. (2017). Influence of livestock activities on residue antibiotic levels of rivers in Hong Kong. *Environmental Science and Pollution Research, 24*(10), 9058–9066.

Spielmeyer, A., Höper, H., & Hamscher, G. (2017). Long-term monitoring of sulfonamide leaching from manure amended soil into groundwater. *Chemosphere, 177*, 232–238.

Stepanić, V., Žiher, D., Gabelica-Marković, V., Jelić, D., Nunhuck, S., Valko, K., & Koštrun, S. (2012). Physicochemical profile of macrolides and their comparison

with small molecules. *European Journal of Medicinal Chemistry, 47*, 462–472.

Tang, J., Shi, T., Wu, X., Cao, H., Li, X., Hua, R., Tang, F., & Yue, Y. (2015). The occurrence and distribution of antibiotics in Lake Chaohu, China: seasonal variation, potential source and risk assessment. *Chemosphere, 122*, 154–161.

Thiele-Bruhn, S. (2003). Pharmaceutical antibiotic compounds in soils – a review. *Journal of Plant Nutrition and Soil Science, 166*(2), 145–167.

Van Boeckel, T. P., Gandra, S., Ashok, A., Caudron, Q., Grenfell, B. T., Levin, S. A., & Laxminarayan, R. (2014). Global antibiotic consumption 2000 to 2010: an analysis of national pharmaceutical sales data. *The Lancet Infectious Diseases, 14*(8), 742–750.

Van Rennings, L., Von Münchhausen, C., Ottilie, H., Hartmann, M., Merle, R., Honscha, W., et al. (2015). Cross-sectional study on antibiotic usage in pigs in Germany. *PLoS ONE, 10*(3), e0119114.

Wammer, K. H., Korte, A. R., Lundeen, R. A., Sundberg, J. E., McNeill, K., & Arnold, W. A. (2013). Direct photochemistry of three fluoroquinolone antibacterials: Norfloxacin, ofloxacin, and enrofloxacin. *Water Research, 47*(1), 439–448.

Wang, B., Cao, M., Zhu, H., Chen, J., Wang, L., Liu, G., Gu, X., & Lu, X. (2013). Distribution of perfluorinated compounds in surface water from Hanjiang River in Wuhan, China. *Chemosphere, 93*(3), 468–473.

Wei, X., Liu, Y., Zhou, Z., Ou, H., Zhang, J., Zhao, J., et al. (2018). Occurrence and removal of antibiotics from wastewater of typical emission sources in Guangzhou China. *Journal of South China Normal University (Natural Science Edition)., 50*(1), 11–20.

Weng, X., Owens, G., & Chen, Z. (2020). Synergetic adsorption and Fenton-like oxidation for simultaneous removal of ofloxacin and enrofloxacin using green synthesized Fe NPs. *Chemical Engineering Journal, 382*, 122871.

Xu, J., Xu, Y., Wang, H., Guo, C., Qiu, H., He, Y., Zhang, Y., Li, X., & Meng, W. (2015). Occurrence of antibiotics and antibiotic resistance genes in a sewage treatment plant and its effluent-receiving river. *Chemosphere, 119*, 1379–1385.

Xue, B., Zhang, R., Wang, Y., Liu, X., Li, J., & Zhang, G. (2013). Antibiotic contamination in a typical developing city in south China: occurrence and ecological risks in the Yongjiang River impacted by tributary discharge and anthropogenic activities. *Ecotoxicology and Environmental Safety, 92*, 229–236.

Yan, C., Yang, Y., Zhou, J., Liu, M., Nie, M., Shi, H., & Gu, L. (2013). Antibiotics in the surface water of the Yangtze Estuary: occurrence, distribution and risk assessment. *Environmental Pollution, 175*, 22–29.

Yan, M., Xu, C., Huang, Y., Nie, H., & Wang, J. (2018). Tetracyclines, sulfonamides and quinolones and their corresponding resistance genes in the Three Gorges Reservoir, China. *Science of The Total Environment, 631-632*, 840–848.

Yang, J.-F., Ying, G.-G., Zhao, J.-L., Tao, R., Su, H.-C., & Liu, Y.-S. (2011). Spatial and seasonal distribution of selected antibiotics in surface waters of the Pearl Rivers, China. *Journal of Environmental Science and Health, Part B, 46*(3), 272–280.

Yang, W., Tang, Z., Zhou, F., Zhang, W., & Song, L. (2013). Toxicity studies of tetracycline on microcystis aeruginosa and Selenastrum capricornutum. *Environmental Toxicology and Pharmacology, 35*(2), 320–324.

Yu, Y., Zhu, X., Wu, G., Wang, C., & Yuan, X. (2019). Analysis of antibiotic resistance of Escherichia coli isolated from the Yitong River in North-east China. *Frontiers of Environmental Science & Engineering, 13*(3), 39.

Zhang, Q., Jia, A., Wan, Y., Liu, H., Wang, K., Peng, H., Dong, Z., & Hu, J. (2014). Occurrences of three classes of antibiotics in a natural river basin: association with antibiotic-resistant Escherichia coli. *Environmental Science & Technology, 48*(24), 14317–14325.

Zhang, Q.-Q., Ying, G.-G., Pan, C.-G., Liu, Y.-S., & Zhao, J.-L. (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.

Zhang, X. P., Zhang, Y. Y., Mai, L., Liu, L. Y., Bao, L. J., & Zeng, E. Y. (2020). Selected antibiotics and current-use pesticides in riverine runoff of an urbanized river system in association with anthropogenic stresses. *Science of The Total Environment, 739*, 140004.

Zhao, H., Cao, Z., Liu, X., Zhan, Y., Zhang, J., Xiao, X., Yang, Y., Zhou, J., & Xu, J. (2017). Seasonal variation, flux estimation, and source analysis of dissolved emerging organic contaminants in the Yangtze Estuary, China. *Marine Pollution Bulletin, 125*(1-2), 208–215.