Journal of Hazardous Materials 190 (2011) 588–596

Contents lists available at ScienceDirect

Journal of Hazardous Materials

journal homepage: www.elsevier.com/locate/jhazmat

Occurrence and phase distribution of selected pharmaceuticals in the Yangtze Estuary and its coastal zone

Y. Yang[a,b], J. Fu[a], H. Peng[a], L. Hou[b], M. Liu[a], J.L. Zhou[c,*]

[a] School of Resources and Environmental Sciences, East China Normal University, Shanghai 200062, China
[b] State Key Laboratory of Estuarine and Coastal Research, East China Normal University, Shanghai 200062, China
[c] Department of Applied Sciences, London South Bank University, 103 Borough Road, London SE1 0AA, UK

ARTICLE INFO

Article history:
Received 12 January 2011
Received in revised form 23 March 2011
Accepted 23 March 2011
Available online 30 March 2011

Keywords:
Pharmaceuticals
Yangtze Estuary
Aquatic colloids
Sewage effluent
Mass balance

ABSTRACT

The occurrence and geochemical behavior of nine pharmaceutical compounds were investigated along the Yangtze River Estuary and its coastal area, by sampling and analysis of pharmaceuticals in sediment, suspended particulate matter (SPM), colloidal and soluble phases. In addition, the impact of sewage input was examined by sampling from sewage treatment plants (STP) effluent and its upstream and downstream in the Yangtze River. Although at relatively low concentrations in SPM and sediments, several pharmaceuticals were found at elevated concentration in filtered water samples from STP-affected sites. STP is therefore an important input of pharmaceuticals in the study area. Colloidal phase was further separated from bulk water samples using cross-flow ultrafiltration (CFUF), confirming it being an effective sorbent for pharmaceuticals with high sorption capacity which are 2–4 orders of magnitude higher than SPM. Moreover, mass balance calculations showed that significant percentages of selected pharmaceutical compounds were associated with aquatic colloids, indicating colloids as a reservoir for these contaminants in the Yangtze estuarine system.

1. Introduction

Pharmaceuticals of different classes are produced in large quantities in China (approximately 1.9 million tons in 2009), for treating many kinds of diseases such as gastrointestinal, inflammation and hypertension; in Shanghai alone more than 25,000 tons were produced. Once used, most pharmaceutical compounds are excreted with feces and urine and released into natural aquatic systems via different routes, the most important of which are effluents from sewage treatment plants (STP), and hospital wastewater, because of their incomplete elimination in STP. Some pharmaceuticals are considered as "pseudopersistent organic compounds" due to large quantities being used and continuous discharge into the environment. With the development of advanced analytical techniques such as liquid chromatography–tandem mass spectrometry (LC–MS/MS), pharmaceuticals have been detected in STP effluents [1], river water [2], estuarine water [3], groundwater [4] and even drinking water [5]. However, to our knowledge no report has been made of pharmaceuticals in aquatic systems in Shanghai, a major population centre in China. In addition, data for pharmaceutical presence in aquatic sediments and SPM is scarce [6]. Furthermore, the exact distribution of pharmaceutical pollutants between different particle size fractions such as colloids in aquatic systems has hardly been addressed.

Aquatic colloids are broadly defined as particles in the range of 1 nm to 1 μm with different origin, composition, formation, and physicochemical properties [7]. Although ubiquitous in natural aquatic systems, colloids are often neglected in investigations for contaminant fate and behavior. The traditional method of water sample preparation involves separating solid phases from water using membrane filters (e.g. pore size of 0.45 μm), to obtain SPM and so-called dissolved phase. In fact, the "dissolved" phase includes complex fractions such as colloids of different sizes and the truly soluble phase. Due to their large surface area and large surface site density, colloids may exhibit an enhanced sorption affinity for organic contaminants such as pharmaceuticals. Bowman et al. [8] found that the sorption capacity for 17α-ethynylestradiol (EE2) was 2 orders of magnitude higher in estuarine colloids than in sediments. More recently, with the application of CFUF Maskaoui et al. [9] experimentally determined the partition of selected pharmaceuticals between colloidal and dissolved phases and suggested that aquatic colloids may play a significant role in regulating the environmental behavior of pharmaceuticals. These studies have demonstrated that natural aquatic colloids show relatively high affinity for pharmaceuticals and could act as strong sorbents for pharmaceuticals.

* Corresponding author. Tel.: +44 20 78157933; fax: +44 20 78157699.
E-mail address: zhouj9@lsbu.ac.uk (J.L. Zhou).

0304-3894/$ – see front matter © 2011 Elsevier B.V. All rights reserved.
doi:10.1016/j.jhazmat.2011.03.092

In this study we aim to determine the concentrations of selected pharmaceutical compounds along the Yangtze River Estuary and its coastal area, a very important region both economically and ecologically in China. To gain further insight, water samples from STP effluent and the Yangtze Estuary were separated into SPM, filtrate, colloid and soluble phases, to determine the intrinsic association of pharmaceuticals between different phases. Such knowledge is essential to assess the long-term fate and potential risks of pharmaceuticals in estuaries and coastal zones.

2. Experimental

2.1. Chemicals

Pharmaceuticals including propranolol, sulfamethoxazole, mebeverine, thioridazine, carbamazepine, tamoxifen, indomethacine, diclofenac and meclofenamic acid were purchased from Sigma, UK. Internal standard (diuron-d_6 and ^{13}C-phenacetin) were purchased from Cambridge Isotope Laboratories, USA. All solvents used were of HPLC grade.

2.2. Sampling

As this is the first study of its kind in the Yangtze ecosystem, the overall aim was to determine the occurrence of selected pharmaceutical compounds in the water (including soluble, colloidal and SPM phases) and sediment. Hence the sampling strategy was first to cover key sources, including the largest STP (Bailonggang or BLG, 5#) in Asia, and the turbidity maximum (2#) where sediment resuspension could act as a secondary source (Fig. 1). Second consideration was to take sediment samples along the coastline, specifically Yanyang (YY), Daxinggang (DXG), Xupu (XP), Liuhekou (LHK), Wusongkou (WSK), Bailonggang (BLG), Luchao (LC) and Chongming (CM), in order to identify contamination levels in sediments. Thirdly sampling consideration was given to cover the salinity gradient in the estuary, to assess potential hotspots of contamination.

Prior to sampling, the brown glass bottles and jars were thoroughly cleaned by acetone, deionized water and Milli-Q water, before being ashed at 400 °C, in order to minimize potential contamination and photo transformation. All samples were collected in triplicate. Surface tidal sediment samples were collected along the Yangtze River Estuary and its nearby coastal areas in November 2009; sampling sites included YY, DXG, XP, LHK, WSK, BLG, LC and CM (Fig. 1). To ensure homogeneity, sediments in triplicate were combined, rendering a total wet sediment mass of at least 2 kg for each sample. All sediment samples were immediately stored at −20 °C until further processing.

In addition, sediments (2 kg) and water samples (50 L) were collected at sites 1#–6# (Fig. 1), together with the measurement of their properties (Table 1). Sampling site 1# is at the river mouth, 2# at the turbidity maximum zone and 3# in coastal zone. Sampling sites 4# and 6# are around the BLG STP discharge while site 5# is approximately 2 km away from the STP. As the largest STP in Asia, BLG STP has a capacity to treat more than one third of industrial and domestic sewage in Shanghai, using primary sedimentation, activated sludge process and abyss discharge. Sodium azide (0.01 mol/L sample) was added to water samples in order to minimize biological activity. Once transported to the laboratory, water samples were immediately filtered through 0.7 μm GF/F filters (Whatman, pre-combusted at 400 °C) to obtain soluble and colloidal phases. In marine organic chemistry, GF/F filters (0.7 μm) are the most widely used for the filtration of water samples, as a standard procedure. SPM samples on filters and sediments were stored at −20 °C until extraction. As a result, the soluble and colloidal phases obtained were not subject to potential changes resulting from procedures such as freezing.

2.3. Colloidal isolation

Owing to the dynamic nature of aquatic colloids, the definition of their characteristics is operational in nature, although they are often defined by size as particles between 1 nm and 1 μm. Practically, aquatic colloids are operationally defined by the pore size of separating devices used, e.g. filter papers. Filtered water samples were further separated into soluble and colloidal phases by CFUF (Pellicon System, Millipore), following the method developed by Wilding et al. [10,11]. Briefly, a 1-kDa regenerated cellulose Pellicon 2 PLAC ultrafiltration membrane was used to concentrate aquatic colloids. The CFUF operation was carried out in a sampling mode in which the colloidal phase was directed back to the feed container as retentate flow, and soluble phase as permeate flow was directed to a separate container. Hence in this work, colloids are defined as particles between 1-kDa and 0.7 μm.

2.4. Sample treatment and analyses

Samples in triplicate were processed and analyzed following a method by Zhang and Zhou [12]. Briefly, prior to the solid phase extraction (SPE), 100-ng of diuron-d_6 and ^{13}C-phenacetin were spiked into aqueous samples (filtrates, retentates and permeates) as the internal standards. All the SPE cartridges (Oasis HLB, Waters) were first conditioned with 10 mL of methanol, followed by ultrapure water (3× 5 mL) at a rate of 1 mL/min. Water samples were extracted at a flow rate of 5–10 mL/min. The analytes were eluted from SPE cartridges using methanol, and then reduced to 1 mL by a rotary evaporator. Sediment and SPM samples were extracted by methanol following an established method [13,14], in an accelerated solvent extractor (ASE 300, Dionex) under high pressure (100 bars) and high temperature (100 °C). Upon exaction, each extract was reduced to 1 mL and dissolved back into pure water, then extracted by SPE similar to water samples.

The LC–MS/MS analyses were conducted using a method developed by the authors [12]. Briefly, compound separation was accomplished on a Waters 2695 HPLC module (Milford, MA, USA) equipped with a Waters Symmetry C_{18} column (2.1 mm × 100 mm, particle size 3.5 μm). The mobile phase was 0.1% formic acid in ultrapure water (eluent A), acetonitrile (eluent B) and methanol (eluent C). At a flow rate of 0.2 mL/min, the elution started with 10% of eluent B, followed by a 25 min gradient to 80% of eluent B and a 3 min gradient to 100% of eluent C within 8 min and held for 10 min. The MS/MS analysis was completed with a Micromass Quattro triple-quadrupole mass spectrometer equipped with a Z-spray electrospray interface, in positive ion mode. The recovery of all compounds was between 51% and 103% for aqueous samples and 43–88% for SPM and sediment samples, with the exception of thioridazin being 25% and 21%, respectively. None of the target compounds was detected in blank samples. Full quality assurance such as the limit of detection (LOD) and limit of quantification (LOQ) has been reported in our earlier publications [12,14]. In summary, LOD and LOQ values were 1–144 pg/L, and 4–503 pg/L in water samples (soluble, colloidal), and 0.2–64 ng/g and 1–214 ng/g dry weight in SPM and sediment samples, respectively.

2.5. Organic carbon (OC) and particle size analysis

Sediment and SPM samples were treated with HCl (1 M) to remove carbonates, with particulate organic carbon (POC) contents being determined by elemental analysis (Vario EL, Elementar, Germany). Dissolved organic carbon (DOC) in filtrate, retentate, and permeate samples was analyzed using liquiTOC II (Elementar,

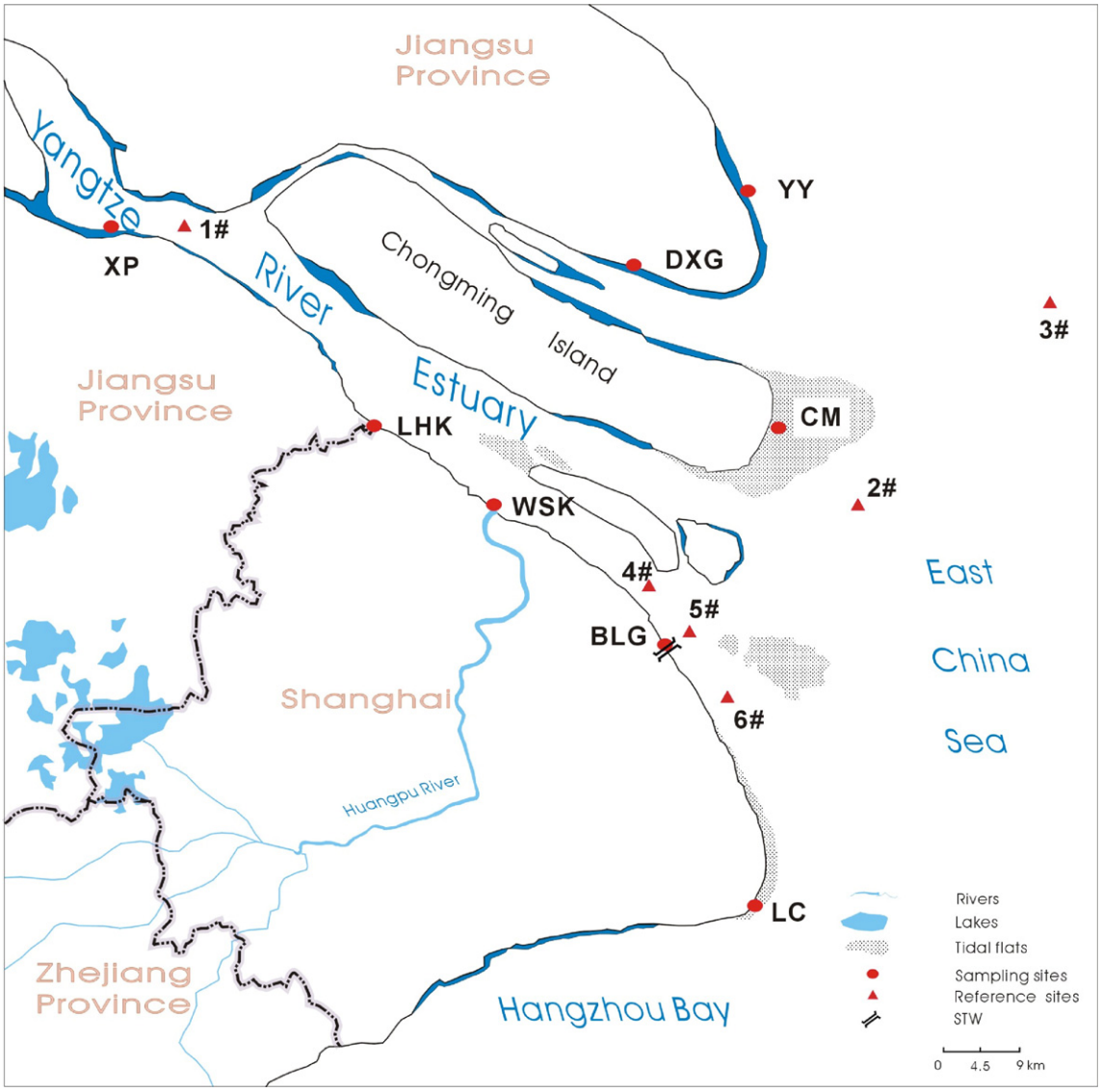

Fig. 1. Sampling sites along the Yangtze River Estuary and its coastal water. The BLG STP employs primary sedimentation and activated sludge process before discharge at sea.

Germany). A Hitachi S4700 scanning electron microscope (SEM) was used to obtain colloid sizes and morphologies. The samples were air dried before mounting on carbon stubs and coating with gold.

3. Results and discussion

3.1. Occurrence of pharmaceuticals in sediments

Selected pharmaceuticals were detected in coastal tidal sediment samples, suggesting the ubiquitous occurrence and potential persistence of some compounds. The total concentrations of selected nine pharmaceutical compounds were in the range of 414–872 ng/g on a dry weight basis, with the highest level at XP and lowest at CM site. The distribution pattern of selected pharmaceuticals in each sampling site was similar, indicating a uniform source of these contaminants. In detail, tamoxifen (212–431 ng/g), mebeverine (18–415 ng/g), and indomethacine (12–164 ng/g) were the most abundant pharmaceutical compounds (Fig. 2). The pharmaceuticals in XP sediments were dominated by the highest mebeverine concentration.

The distribution pattern of pharmaceuticals in estuarine and coastal sediments is also shown (Fig. 2), where sites 4#–6# are influenced by STP discharge. Concentrations of the target phar-

Table 1
Details of water sampling sites in the Yangtze Estuary and its coastal area.

Code	Location	Temperature (°C)	Salinity (‰)	SPM (mg/L)	DOC (mg/L)
1#	Yangtze River Estuary	14.0	0.2	39.1	3.5
2#	Turbidity maximum zone	10.9	18.5	298.9	2.8
3#	Coastal water	14.8	30.5	108.1	5.8
4#	Upstream of STP outfall	13.2	1.5	218.1	5.0
5#	STP outfall	12.7	2.5	219.8	4.0
6#	Downstream of STP outfall	13.6	6.5	352.2	4.6

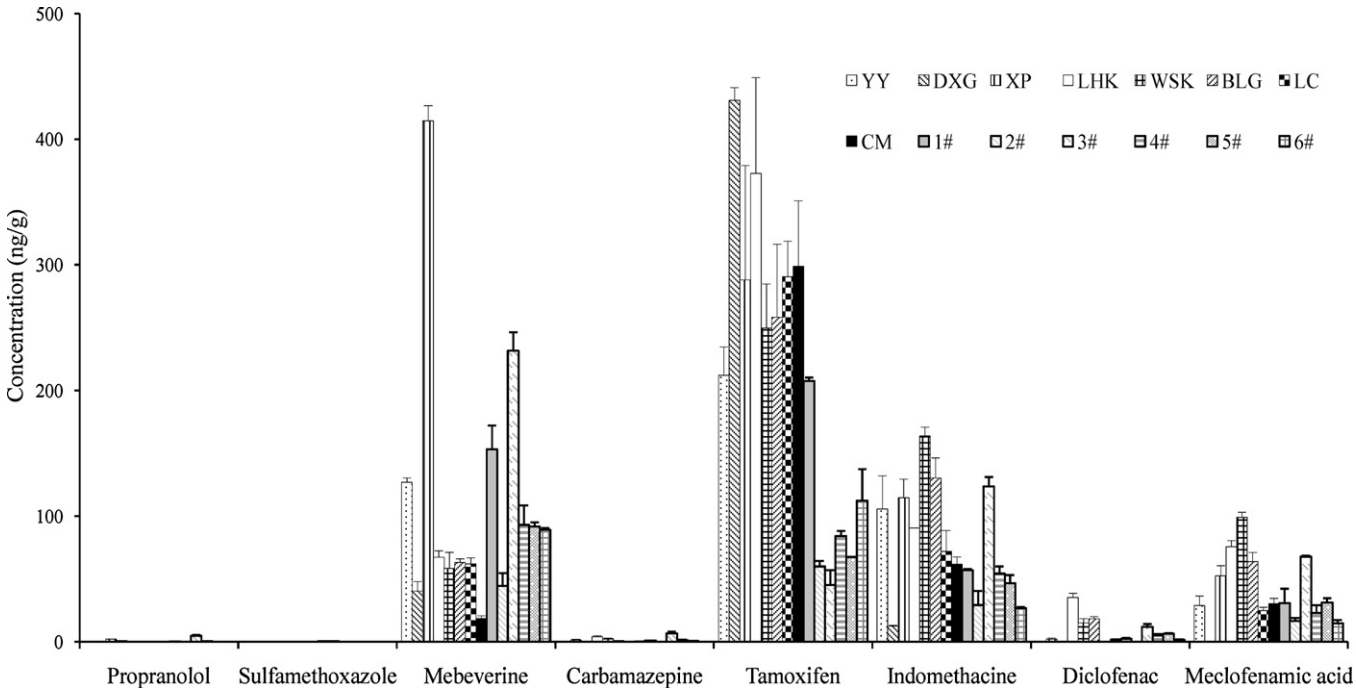

Fig. 2. Pharmaceutical distribution in sediments along the Yangtze Estuary and its coastal zone.

maceuticals were 452, 150 and 492 ng/g in sediments at sites 1#, 2# and 3#, respectively. Interestingly, sediments from STP-affected sites showed relatively low concentrations of 244–261 ng/g. Similar to the distribution pattern of coastal surface sediment, mebeverine, tamoxifen and indomethacine were the dominant pharmaceuticals with mean concentrations of 117 ng/g, 96 ng/g and 56 ng/g, respectively.

This study is, to our knowledge, the first time that the occurrence of pharmaceuticals in coastal sediments was reported. It is noticeable that tamoxifen was detected in almost all sediment samples with high abundance. With a high $\log K_{ow}$ value of 6.3 [15], this compound is highly hydrophobic and tends to associate with particles and consequently being detected in sediments. POC content showed a good correlation to total pharmaceutical concentrations ($r^2 = 0.49$, $p < 0.01$) in all sediment samples. If marine sediments were excluded, a better correlation ($r^2 = 0.85$, $p < 0.005$) was obtained between pharmaceutical concentrations and POC, indicating POC being one of the dominant sorbents for pharmaceuticals occurring in sediments.

3.2. Occurrence of pharmaceuticals in the water column

To investigate the impact of Yangtze River discharge and BLG STP effluent on the occurrence of pharmaceutical compounds in coastal water, water samples taken from sites 1# to 6# were filtered to obtain SPM and filtrate samples.

3.2.1. SPM

SPM samples were even less well studied than sediment samples for pharmaceutical pollution. Compared to the corresponding concentrations in sediments, SPM samples showed approximately 2–5 times higher concentrations of pharmaceuticals in the range of 451–2453 ng/g. Mebeverine, tamoxifen, indomethacine and meclofenamic acid were the most abundant pharmaceutical compounds, with average concentrations of 163 ng/g, 289 ng/g, 261 ng/g and 204 ng/g, respectively (Fig. 3). The spatial distribution pattern was similar to the corresponding sediments with the highest concentration at the offshore site, and SPM samples from STP sites showed relatively low concentration, suggesting the significant impact of complex hydrodynamics on water and sediment movement. POC content in SPM was in the range of 1.5–3.7%, which is higher than that in sediments. A positive correlation was found between pharmaceutical concentrations and POC ($r^2 = 0.77$, $p < 0.05$), again suggesting that POC is a significant sorbing phase in SPM for pharmaceuticals.

3.2.2. Filtrate

Selected pharmaceuticals were investigated in water samples filtered through 0.7-μm filters, which are traditionally classified as the dissolved phase (Table 2). Other than the sediment and SPM samples, water samples from the STP-affected sites showed obviously higher pharmaceuticals concentrations than the other sites in the estuary and offshore. Site 5# at the BLG STP showed the highest total concentration of 3808 ng/L. Site 1# in the Yangtze River had a combined pharmaceutical concentration of 1122 ng/L, higher than that at sites 2# and 3#. Mebeverine was not detected in the STP-affected water, while meclofenamic acid was not found in the Yangtze estuarine and offshore water samples. The occurrence of mebeverine in water samples from site 2# and especially site 3# suggested local input of this compound rather than from the STP.

Table 2
Selected pharmaceutical concentrations in filtered water samples (ng/L).

	Yangtze Estuary			STP-affected sites		
	1#	2#	3#	4#	5#	6#
Propranolol	25	0.3	N.D.	142	51	31
Sulfamethoxazole	485	40	4.2	225	701	765
Mebeverine	N.D.	71	154	N.D.	N.D.	N.D.
Carbamazepine	46	19	17	382	675	291
Tamoxifen	127	129	120.	224	172	141
Indomethacine	352	167	159	545	979	672
Diclofenac	N.D.	N.D.	N.D.	762	843	283
Meclofenamic acid	87	N.D.	N.D.	679	387	348
Total concentration	1122	426	454	2959	3808	2531

N.D., not detected.

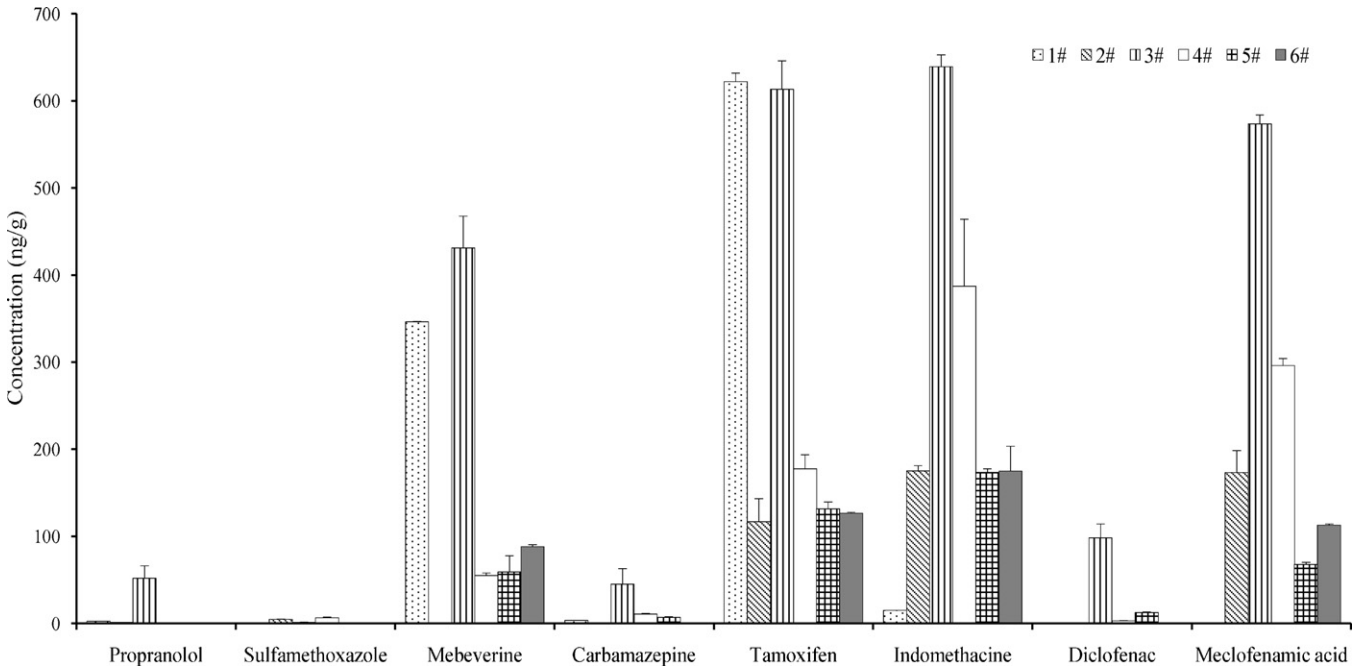

Fig. 3. Distribution pattern of pharmaceuticals in SPM samples at sites 1#–6#.

Most pharmaceuticals are not highly persistent, but as a consequence of their large quantity of usage, they are likely to have a continuous presence in the aquatic environment, that is, to be pseudopersistent [16]. Moreover, a mixture of pharmaceuticals may show a significantly increased toxicity, e.g. towards the growth inhibition of algae [17].

In the present study, sulfamethoxazole, carbamazepine, tamoxifen, indomethacine, diclofenac and meclofenamic acid were all detected in BLG STP effluent with a high abundance (>100 ng/L), and most of them were found in the Yangtze estuarine water. As a widely prescribed, nonselective β-adrenergic receptor-blocking agent, propranolol has been detected in municipal effluents from the ng/L to the low μg/L range [2,18–21]. It is also shown to be the most toxic to Daphnia and alga among the three major active ingredients of β blockers [22]. With a photodegradation half-life of 16.8 days [23], propranolol concentrations were in the range of 31–142 ng/L in the STP effluent which is similar to that found in England [2], suggesting a potential risk to the surrounding aquatic organisms. Sulfamethoxazole, as a common antibiotic, was found in lower levels than those in STP effluent samples from USA [24] and Canada [25], but higher than those found in England [2] and Korea [26]. The average STP removal efficiency of sulfamethoxazole was found to be ca. 42% [27]. Acute toxicity level for algae was found in the order of mg/L, while chronic toxicity appeared at con-

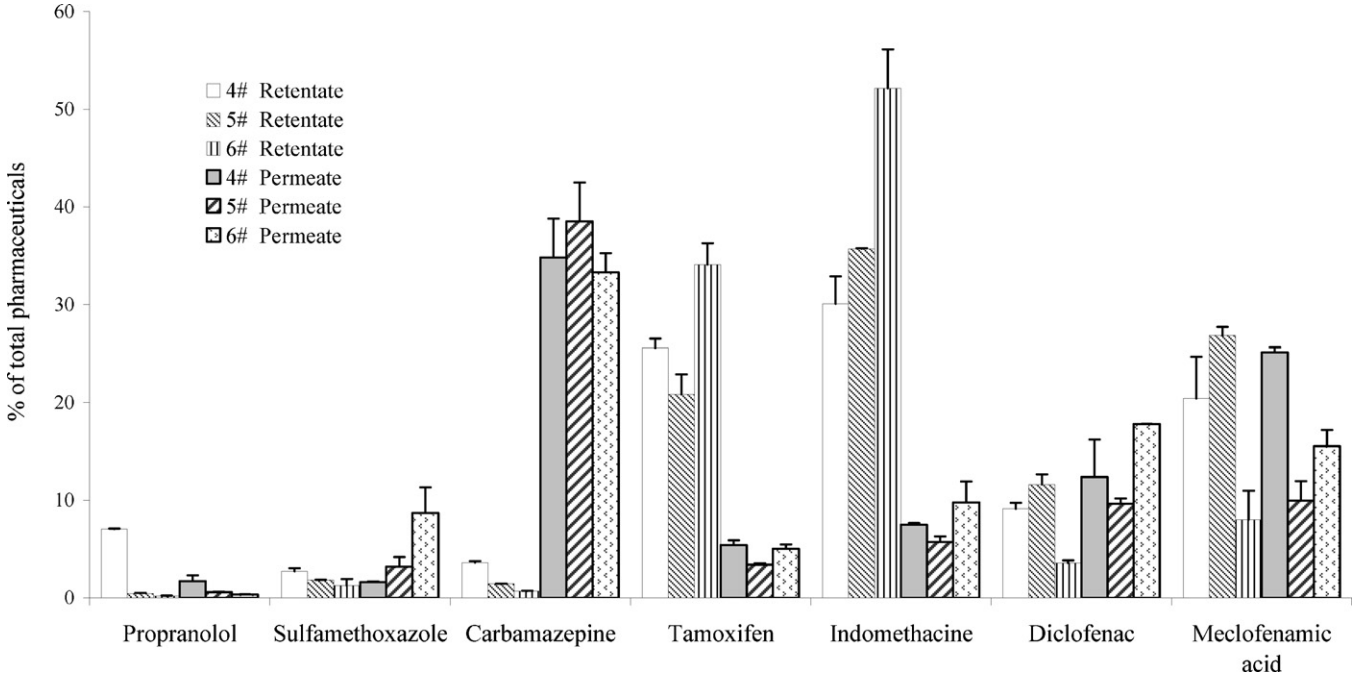

Fig. 4. Pharmaceuticals distribution pattern in retentate and permeate samples, from CFUF separation of filtered water samples.

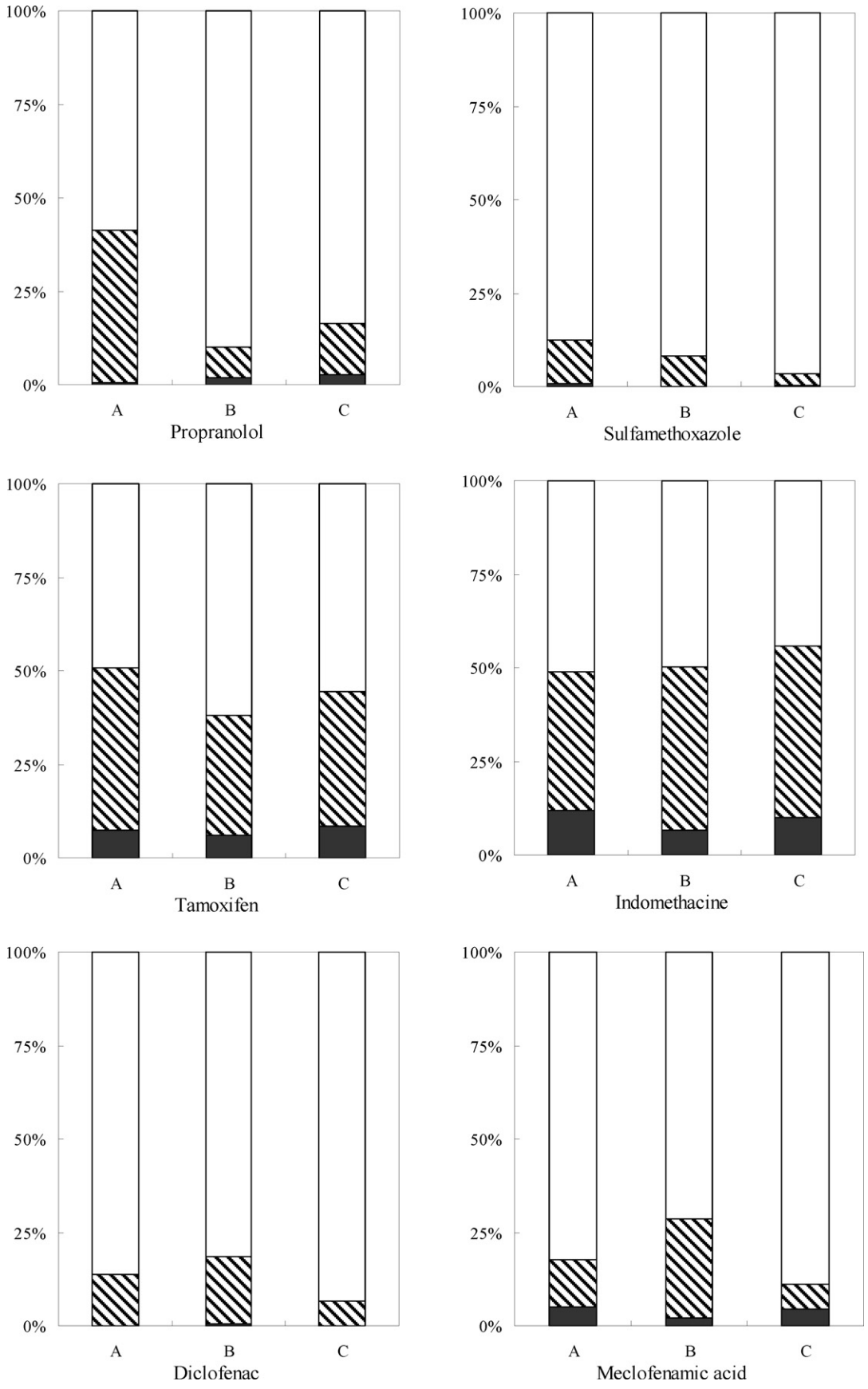

Fig. 5. Mass balance of selected pharmaceuticals among soluble (□), colloidal (▨) and SPM (■) phases in the Yangtze aquatic system.

Table 3
Organic carbon normalized partition coefficients between colloids and permeates (K_{coc}), between SPM and filtrate ($K_{oc}^{obs.}$), and between SPM and permeates ($K_{oc}^{int.}$), and relevant values from the literature ($K_{coc}^{lit.}$, $K_{oc}^{lit.}$).

	$\log K_{ow}$	$\log K_{coc}$	$\log K_{coc}^{lit.}$	$\log K_{oc}^{obs.}$	$\log K_{oc}^{int.}$	$\log K_{oc}^{lit.}$
Propranolol	1.2	6.97 ± 0.37	5.25[a]	2.65 ± 0.44	2.9 ± 0.48	3.64[b]
Sulfamethoxazole	0.48	6.5 ± 0.12	4.95[a]	2.53 ± 0.51	2.8 ± 0.3	1.3–2.72[b,c]
Tamoxifen	6.3	7.65 ± 0.27	–	4.35 ± 0.12	4.55 ± 0.23	–
Indomethacine	4.27	7.65 ± 0.3	5.5[a]	4.14 ± 0.22	4.49 ± 0.14	–
Diclofenac	4.51	6.7 ± 0.16	5.29[a]	2.8 ± 0.55	3.03 ± 0.6	2.08, 3.36[b]
Meclofenamic acid	5.12	6.8 ± 0.37	–	4.2 ± 0.12	4.35 ± 0.11	–

[a] Maskaoui et al. [9].
[b] Drillia et al. [37].
[c] Löffler et al. [38].

centration in the order of μg/L [28]. Carbamazepine is one of the most persistent pharmaceuticals in the environment with a mean 50% dissipation time (DT_{50}) of 82 days [29]. In this study, although its abundance in the sediment was very low, carbamazepine was found in water samples with relatively high concentrations. The level of carbamazepine in STP effluents (291–675 ng/L) was comparable to those from Spain [30] and Finland [31], while lower than those in England [2] and Germany [32]. Tamoxifen is an antiestrogen used in the therapy of breast cancer, to reduce the incidence in high risk women and treat male oligoasthenozoospermia. However, once released to the aquatic system tamoxifen, as the hormone antagonist, is found to cause a significant increase of plasma vitellogenin levels in male Japanese medaka at low-dose exposure (approximately 1 μg/L) [33]. The levels of tamoxifen in STP effluent and Yangtze estuarine water samples were similar, which are comparable to those found in some UK STP and River water samples [20]. Indomethacine, diclofenac and meclofenamic acid are widely used as non-steroidal anti-inflammatory drugs (NSAIDs). Indomethacine was detected in all water samples from Yangtze River Estuary and coastal water with relatively high concentrations compared to other samples worldwide [2,21,34,35]. As the most toxic NSAIDs [17], diclofenac was not found in water samples from sites 1# to 3#, due probably to its relatively low persistence in the environment; for example, Kim and Tanaka [36] reported a very rapid and efficient photodegradation of diclofanac under UV light. The concentrations of diclofenac in STP effluent samples (283–843 ng/L) were however higher than those in England [2] and Greece [34], which could pose an environmental risk in the study area.

3.3. Colloidal control of pharmaceutical distribution in water

With high concentrations of the target pharmaceuticals in water samples, STP effluent poses a high risk to the estuarine and coastal ecosystem. Filtered water samples were further separated into permeate (truly soluble phase) and retentate (colloidal phase) samples by CFUF to elucidate the potential importance of colloids for pharmaceuticals in study area.

Firstly, recovery (R) of OC was calculated from the following equation:

$$R\ (\%) = 100 \times \frac{C_p + C_r}{C_f} \quad (1)$$

where C_f, C_p and C_r are the concentrations of OC in filtrate, permeate and retentate samples, respectively. With a satisfying recovery of 85–122% for OC, therefore the application of CFUF was validated. Colloidal organic carbon (COC) concentrations (C_c) were estimated according to the equation based on mass balance:

$$C_c = \frac{C_r - C_p}{cf} \quad (2)$$

where cf is the concentration factor.

The colloid size and morphology at site 4# was characterized by SEM (Fig. 1, supplemental information), indicating that the colloids were in the 100–200 nm range in size with a visible globular structure.

Pharmaceuticals determined in permeate and retentate samples represent the true soluble and colloidal contaminants, respectively. Fig. 4 shows the distribution pattern of the target pharmaceuticals in truly soluble phase and colloidal phase in water samples from sites 4# to 6#. Contrast to those in sediment and SPM samples, instead of tamoxifen, indomethacine and meclofenamic acid, carbamazepine with a K_{ow} value of 0.5 was the most abundant pharmaceutical truly soluble in water. However in retentate samples, tamoxifen, indomethacine and meclofenamic acid were the dominant pharmaceuticals (Fig. 4), which is also the case for sediment and SPM samples. The findings suggested that pharmaceuticals can be transported by aquatic colloids, aggregated in SPM, and consequently stored in bottom sediments.

According to the equation:

$$\frac{C_r}{C_p} = 1 + K_{coc}[\mathrm{COC}], \quad (3)$$

the distribution coefficient K_{coc} value (mL/g) for each compound was calculated as shown in Table 3, except for carbamazepine due to poor mass balance. The average $\log K_{coc}$ values were calculated to be 6.97 for propranolol, 6.5 for sulfamethoxazole, 7.65 for tamoxifen and indomethacine, 6.7 for diclofenac and 6.8 for meclofenamic acid. These are higher than those determined in laboratory controlled sorption experiments [9]. Therefore, colloids could act as strong sorbents of pharmaceuticals with high sorption capacity in the aquatic systems.

3.4. Distribution and mass balance of pharmaceuticals in water

One of the key processes controlling the transport and fate of pharmaceuticals in water is sorption to SPM. The distribution of pharmaceuticals between SPM and water is represented by the partition coefficient (K_p), which is often normalized by the organic carbon content of SPM samples to obtain K_{oc} (organic carbon normalized partition coefficient).

Traditionally, K_{oc} value was calculated according to the ratio of contaminant concentrations between SPM and filtrates, and the presence and function of colloids were simply omitted. In this study, we calculated two sets of K_{oc} values, including both $K_{oc}^{obs.}$ and $K_{oc}^{int.}$ which represent observed K_{oc} and intrinsic K_{oc}. The intrinsic values considering colloids in water were calculated according to the ratio of pharmaceutical concentrations between SPM and permeates from CFUF. Intrinsic values were in general 1.1–2.5 times higher than the corresponding observed values. The apparent observed K_{oc} values investigated in this study were comparable to data from other studies, except for indomethacine which showed a higher value in this study. Moreover, the $\log K_{oc}$ values showed a positive correlation to $\log K_{ow}$ values ($r^2 = 0.69$, $p < 0.05$). In addi-

tion, the calculations showed that K_{coc} values were 2–4 orders of magnitude higher than the K_{oc} values, indicating that aquatic colloids act as more powerful sorbents for pharmaceuticals than SPM in the aquatic system.

To further evaluate the contribution of colloidal pharmaceuticals in the aquatic system, the mass balance of pharmaceuticals was calculated by including the SPM, colloidal and soluble phases. As shown in Fig. 5, the colloidal phase contributed 10–40% of propranolol, 4–12% of sulfamethoxazole, 31–43% of the total tamoxifen, 36–45% of total indomethacine, up to 18% of diclofenac, and 6.5–26% of meclofenamic acid in the aquatic system. However, the contribution of SPM was much lower compared to the colloids, due to small quantity of SPM per liter of water sample. Therefore, in this study aquatic colloids have been demonstrated to be a very important sink, and further as a carrier of pharmaceuticals in the aquatic systems.

4. Conclusions

A multi-phase study of selected pharmaceuticals was conducted along the Yangtze River Estuary and its coastal area. Sites located around a STP showed higher pharmaceuticals concentrations in water samples but lower pharmaceutical concentrations in sediments and SPM samples, than other estuarine and coastal sites. The results suggest potential multiple sources of pharmaceutical inputs to the Yangtze, and complex interactions between pollution sources, transport and degradation in this highly dynamic system. Aquatic colloids, often neglected, showed a significant sorption capacity for the pharmaceuticals with K_{coc} values 2–4 orders of magnitude higher than SPM. In addition, mass balance calculations confirmed that a high percentage of pharmaceuticals in the Yangtze ecosystem were bound with colloids. As colloids are a strong sorbent for pharmaceuticals, highly abundant and highly mobile in the aquatic systems, they act as an effective carrier for such contaminants, thereby affecting their long-term fate and toxicity. Further studies are needed to assess the temporal variation of pharmaceutical concentrations in the Yangtze ecosystem, by using appropriate sampling frequency and mode.

Acknowledgements

This study was funded by the National Natural Science Foundation of China (40901256), the Fundamental Research Funds for the Central Universities (78210030), and the State Key Laboratory of Estuarine and Coastal Research (2010RCDW01). We thank Yong Li and Qiandong Zhang at ECNU for their assistance with sampling.

Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at doi:10.1016/j.jhazmat.2011.03.092.

References

[1] A. Nikolaus, S. Meric, D. Fatta, Occurrence patterns of pharmaceuticals in water and wastewater environments, Anal. Bioanal. Chem. 387 (2007) 1225–1234.
[2] J.L. Zhou, Z.L. Zhang, E. Banks, D. Grover, J.Q. Jiang, Pharmaceutical residues in wastewater treatment works effluents and their impact on receiving river water, J. Hazard. Mater. 166 (2009) 655–661.
[3] K.V. Thomas, M.J. Hilton, The occurrence of selected human pharmaceutical compounds in UK estuaries, Mar. Pollut. Bull. 49 (2004) 436–444.
[4] K.K. Barnes, D.W. Kolpin, E.T. Furlong, S.D. Zaug, M.T. Meyer, L.B. Barber, A national reconnaissance of pharmaceuticals and other organic wastewater contaminants in the United States—(1) Groundwater, Sci. Total Environ. 402 (2008) 192–200.
[5] M.J. Benotti, R.A. Trenholm, B.J. Vanderford, J.C. Holady, B.D. Stanford, S.A. Snyder, Pharmaceuticals and endocrine disrupting compounds in U.S. drinking water, Environ. Sci. Technol. 43 (2009) 597–603.
[6] V. Matamoros, J.M. Bayona, Elimination of pharmaceuticals and personal care products in subsurface flow constructed wetlands, Environ. Sci. Technol. 41 (2006) 8171–8177.
[7] J.R. Lead, K.J. Wilkinson, Aquatic colloids and nanoparticles: current knowledge and future trends, Environ. Chem. 3 (2006) 159–171.
[8] J.C. Bowman, J.L. Zhou, J.W. Readman, Sediment–water interactions of natural oestrogens under estuarine conditions, Mar. Chem. 77 (2002) 263–276.
[9] K. Maskaoui, A. Hibberd, J.L. Zhou, Assessment of the interaction between aquatic colloids and pharmaceuticals facilitated by cross-flow ultrafiltration, Environ. Sci. Technol. 41 (2007) 8038–8043.
[10] A. Wilding, R. Liu, J.L. Zhou, Validation of cross-flow ultrafiltration for sampling of colloidal particles from aquatic systems, J. Colloid Interf. Sci. 280 (2004) 102–112.
[11] A. Wilding, R. Liu, J.L. Zhou, Dynamic behavior of colloidal and dissolved organic matter through cross-flow ultrafiltration system, J. Colloid Interf. Sci. 287 (2005) 152–158.
[12] Z.L. Zhang, J.L. Zhou, Simultaneous determination of various pharmaceutical compounds in water by solid-phase extraction liquid chromatography–tandem mass spectrometry, J. Chromatogr. A 1154 (2007) 205–213.
[13] A. Hibberd, K. Maskaoui, Z. Zhang, J.L. Zhou, An improved method for the simultaneous analysis of phenolic and steroidal estrogens in water and sediment, Talanta 77 (2009) 1315–1321.
[14] K. Maskaoui, J.L. Zhou, Colloids as a sink for certain pharmaceuticals in the aquatic environment, Environ. Sci. Pollut. Res. 17 (2010) 898–907.
[15] X. Liu, J. Zhang, J. Yin, H. Duan, Y. Wu, B. Shao, Analysis of hormone antagonists in clinical and municipal wastewater by isotopic dilution liquid chromatography tandem mass spectrometry, Anal. Bioanal. Chem. 396 (2010) 2977–2985.
[16] S.K. Khetan, T.J. Collins, Human pharmaceuticals in the aquatic environment: a challenge to green chemistry, Chem. Rev. 107 (2007) 2319–2364.
[17] L.H.M.L.M. Santos, A.N. Araujo, A. Fachini, A. Pena, C. Delerue-Matos, M.C.B.S.M. Montenegro, Ecotoxicological aspects related to the presence of pharmaceuticals in the aquatic environment, J. Hazard. Mater. 175 (2010) 45–95.
[18] M.J. Gomez, M. Petrovic, A.R. Fernandez-Alba, D. Barcelo, Determination of pharmaceuticals of various therapeutic classes by solid-phase extraction and liquid chromatography–tandem mass spectrometry analysis in hospital effluent wastewaters, J. Chromatogr. A 1114 (2006) 224–233.
[19] M.J. Hilton, K.V. Thomas, Determination of selected human pharmaceutical compounds in effluent and surface water samples by high-performance liquid chromatography–electrospray tandem mass spectrometry, J. Chromatogr. A 1015 (2003) 129–141.
[20] P.H. Roberts, K.V. Thomas, The occurrence of selected pharmaceuticals in wastewater effluent and surface waters of the lower Tyne catchment, Sci. Total Environ. 356 (2006) 143–153.
[21] J.W. Kim, H.S. Jang, J.G. Kim, H. Ishibashi, M. Hirano, K. Nasu, N. Ichikawa, Y. Takao, R. Shinohara, K. Arizono, Occurrence of pharmaceutical and personal care products (PPCPs) in surface water from Mankyung River, South Korea, J. Health Sci. 55 (2009) 249–258.
[22] M. Cleuvers, Aquatic ecotoxicology of pharmaceuticals including the assessment of combination effects, Toxicol. Lett. 142 (2003) 185–194.
[23] R. Andreozzi, M. Raffaele, N. Paxéus, Pharmaceuticals in STP effluents and their solar photodegradation in aquatic environment, Chemosphere 50 (2003) 1319–1330.
[24] A.L. Batt, M.S. Kostich, J.M. Lazorchak, Analysis of ecologically relevant pharmaceuticals in wastewater and surface water using selective solid-phase extraction and UPLC–MS/MS, Anal. Chem. 80 (2008) 5021–5030.
[25] X.S. Miao, F. Bishay, M. Chen, C.D. Metcalfe, Occurrence of antimicrobials in the final effluents of wastewater treatment plants in Canada, Environ. Sci. Technol. 38 (2004) 3533–3541.
[26] K. Choi, Y. Kim, J. Park, C.K. Park, M. Kim, H.S. Kim, P. Kim, Seasonal variations of several pharmaceutical residues in surface water and sewage treatment plants of Han River, Korea, Sci. Total Environ. 405 (2008) 120–128.
[27] R.H. Lindberg, P. Wennberg, M.I. Johansson, M. Tysklind, B.A.V. Andersson, Screening of human antibiotic substances and determination of weekly mass flows in five sewage treatment plants in Sweden, Environ. Sci. Technol. 39 (2005) 3421–3429.
[28] M. Isidori, M. Lavorgna, A. Nardelli, A. Parrella, L. Previtera, M. Rubino, Ecotoxicity of naproxen and its phototransformation products, Sci. Total Environ. 348 (2005) 93–101.
[29] M.W. Lam, C.J. Young, R.A. Brain, D.J. Johnson, M.A. Hanson, C.J. Wilson, S.M. Richards, K.R. Solomon, S.A. Mabury, Aquatic persistence of eight pharmaceuticals in a microcosm study, Environ. Toxicol. Chem. 23 (2004) 1431–1440.
[30] M.J. Gomez, M.J. Martinez Bueno, S. Lacorte, A.R. Fernandez-Alba, A. Aguera, Pilot survey monitoring pharmaceuticals and related compounds in a sewage treatment plant located on the Mediterranean coast, Chemosphere 66 (2007) 993–1002.
[31] N.M. Vieno, T. Tuhkanen, L. Kronberg, Analysis of neutral and basic pharmaceuticals in sewage treatment plants and in recipient rivers using solid phase extraction and liquid chromatography–tandem mass spectrometry detection, J. Chromatogr. A 1134 (2006) 101–111.
[32] T.A. Ternes, Occurrence of drugs in German sewage treatment plants and rivers, Water Res. 32 (1998) 3245–3260.
[33] L. Sun, J. Zha, P.A. Spear, Z. Wang, Tamoxifen effects on the early life stages and reproduction of Japanese medaka (*Oryzias latipes*), Environ. Toxicol. Pharmacol. 24 (2007) 23–29.
[34] H.B. Lee, T.E. Peart, M.L. Svoboda, Determination of endocrine-disrupting phenols, acidic pharmaceuticals, and personal-care products in sewage by

solid-phase extraction and gas chromatography–mass spectrometry, J. Chromatogr. A 1094 (2005) 122–129.
[35] M.J.M. Bueno, A. Aguera, M.D. Hernando, M.J. Gomez, A.R. Fernandez-Alba, Evaluation of various liquid chromatography–quadrupole-linear ion trapmass spectrometry operation modes applied to the analysis of organic pollutants in wastewaters, J. Chromatogr. A 1216 (2009) 5995–6002.
[36] I. Kim, H. Tanaka, Photodegradation characteristics of PPCPs in water with UV treatment, Environ. Int. 35 (2009) 793–802.
[37] P. Drillia, K. Stamatelatou, G. Lyberatos, Fate and mobility of pharmaceuticals in solid matrices, Chemosphere 60 (2005) 1034–1044.
[38] D. Löffler, R. Römbke, M. Meller, T.A. Ternes, Environmental fate of pharmaceuticals in water/sediment systems, Environ. Sci. Technol. 39 (2005) 5209–5218.