CrossMark

RESEARCH ARTICLE

Sorption of pharmaceuticals and personal care products to polyethylene debris

Chenxi Wu[1] · Kai Zhang[1,2] · Xiaolong Huang[1,2] · Jiantong Liu[1]

Abstract Presence of plastic debris in marine and freshwater ecosystems is increasingly reported. Previous research suggested plastic debris had a strong affiliation for many pollutants, such as polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), and heavy metals. In this study, the sorption behavior of pharmaceuticals and personal care products (PPCPs), including carbamazepine (CBZ), 4-methylbenzylidene camphor (4MBC), triclosan (TCS), and 17α-ethinyl estradiol (EE2), to polyethylene (PE) debris (250 to 280 μm) was investigated. The estimated linear sorption coefficients (K_d) are 191.4, 311.5, 5140, and 53,225 L/kg for CBZ, EE2, TCS, and 4MBC, and are related to their hydrophobicities. Increase of salinity from 0.05 to 3.5 % did not affect the sorption of 4MBC, CBZ, and EE2 but enhanced the sorption of TCS, likely due to the salting-out effect. Increase of dissolved organic matter (DOM) content using Aldrich humic acid (HA) as a proxy reduced the sorption of 4MBC, EE2, and TCS, all of which show a relatively strong affiliation to HA. Results from this work suggest that microplastics may play an important role in the fate and transport of PPCPs, especially for those hydrophobic ones.

[1] State Key Laboratory of Freshwater Ecology and Biotechnology, Institute of Hydrobiology, Chinese Academy of Sciences, Donghu South Road #7, Wuhan 430072, People's Republic of China

[2] Graduate University of the Chinese Academy of Sciences, Beijing, People's Republic of China

Introduction

Plastics are considered to be the most versatile human-invented materials. Nowadays, nearly all of the products used in our daily lives contain plastics. World plastic production increased drastically in the last 50 years. In 2012, 280 million tons of plastics were produced globally with less than a half end up into landfills or recycled and the rest may still be in use or otherwise be discarded into the environment (Rochman et al. 2013a). In the environment, larger plastic items can slowly breakdown into small pieces via physical, chemical, and biological processes (O'Brine and Thompson 2010; Singh and Sharma 2008). Plastic debris can be transported into oceans with current and accumulate in marine habitats (Browne et al. 2011; Sadri and Thompson 2014; Thompson et al. 2004). Plastic debris observed in the environment are mainly millimeters or smaller in size, and those less than 5 mm are generally referred to microplastics (Hidalgo-Ruz et al. 2012; Thompson et al. 2004).

Microplastics can be ingested accidently by aquatic organisms during their feeding. Both laboratory experiment and field observation showed that microplastics can be taken up by aquatic organisms including crustaceans (Murray and Cowie 2011), bivalves (Van Cauwenberghe and Janssen 2014), zooplanktons (Cole et al. 2013), and fishes (Foekema et al. 2013). Once ingested, micro polystyrene (PS) particles were found able to affect the survival, development, and fecundity of the copepod *Tigriopus japonicas* (Lee et al. 2013). Notable histological changes and a strong inflammatory response were observed upon the uptake of micro high-density polyethylene (HDPE) by blue mussel *Mytilus edulis* L. (von

Moos et al. 2012). Setälä et al. (2014) experimentally proved that microplastics can be transferred into higher trophic level organisms via food chain.

Due to their physic-chemical properties, microplastics were found able to accumulate waterborne contaminants including polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), perfluoroalkyl acids (PFAA), and metals (Holmes et al. 2012; Lee et al. 2014; Rochman et al. 2012; Wang et al. 2015). Microplastics can therefore act as carriers for the transport of those contaminants. When ingested by aquatic organisms, contaminants carried by microplastics can be released and induce associated harmful effects (Bakir et al. 2014a; Besseling et al. 2012; Rochman et al. 2013b).

Other than those persistent organic compounds (POPs) and metals, pharmaceuticals and personal care products (PPCPs) are another important group of pollutants of emerging concern frequently detected in the environment (Caliman and Gavrilescu 2009). However, there is a lack of information on the sorption behavior of PPCPs to microplastics. PPCPs include various compounds with diverse properties; their sorption behavior can be different from that of POPs. Therefore, the purpose of this work is to characterize the sorption of PPCPs by microplastics. The effects of salinity and dissolved organic matter (DOM) content on the sorption of PPCPs were studied. Four PPCPs commonly detected in the environment were selected for the determination (Table 1). Their concentrations were typically observed in the nanograms per liter range in surface water and were found up to few micrograms per liter in wastewater (Fair et al. 2009; Ratola et al. 2012; Sánchez Rodríguez et al. 2015).

Materials and methods

Materials and chemicals

Standards of carbamazepine (CBZ), 4-methylbenzylidene camphor (4MBC), triclosan (TCS), and internal standard pentachloronitrobenzene (PCNB) were purchased from Sigma-Aldrich (St. Louis, MO, USA). Standard of 17α-ethinyl estradiol (EE2) was purchased from Dr. Ehrenstorfer GmbH (Augsburg, Germany). Aldrich humic acid (HA) was also obtained from Sigma-Aldrich and was used to study the effects of dissolved organic matter on the sorption of PPCPs to microplastics. Solvents were high-performance liquid chromatography (HPLC) grade and were obtained from Tedia (Fairfield, OH, USA). Other chemicals were from Sinopharm Group (Shanghai, China). Deionized water (DI water, 18.2 MΩ-cm) was provided by an Elga Purelab Ultra water purification system (High Wycombe, UK). All standards were prepared in methanol at 100 mg/L, and diluted to the desired concentrations before use.

Polyethylene (PE), which was one of the most abundant plastic types observed in the environment (Klein et al. 2015; Zhang et al. 2015), was selected for the examination. Plastic resin pellets of low density PE (0.93 g/mL) were obtained from Sinopec (Beijing, China). The pellets were grinded using an IKA A11 basic analytical mill (Staufen, Germany), and were sieved to obtain plastic debris between 250 to 280 μm in size. The pH_{pzc} (the point of zero charge) was determined for the PE particles and was 7.12. Obtained plastic debris was washed twice by soaking in methanol overnight and then dried in an oven at 60 °C. Prepared plastic debris was used as a model to study the sorption behavior of microplastics.
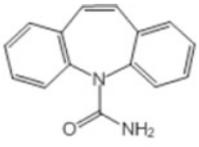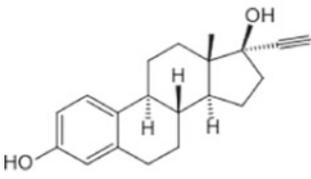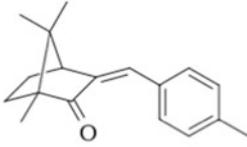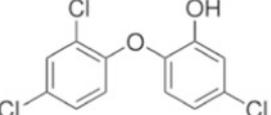
Experiment

The sorption experiment was performed using a batch equilibrium approach. Aliquots of 10 or 50 mg prepared plastic debris were added to 60-mL glass vials. Each vial was filled with 50 mL of 0.01 M CaCl₂, and NaN₃ was added to a concentration of 0.01 % (w/v) to inhibit microbial activity. In a kinetic experiment, each PPCP standard was added to the vial to a concentration of 1 mg/L. The vials were shaken on a reciprocal shaker for 168 h. Three vials for each PPCP were withdrawn at specified time intervals, plastic debris were removed by passing through Whatman GF/C glass fiber filters, and the water samples were analyzed for PPCP residuals. PPCPs reached sorption equilibrium within a short time with no significant variation in liquid phase residual within 48 h (Fig. 1). Then, standard solution of each PPCPs was added to the vial to achieve a concentration of 10, 20, 50, 100, and 200 μg/L, respectively. The vials were equilibrated on a reciprocal shaker for 5 days. Plastic debris was removed by passing through Whatman GF/C glass fiber filters. The water samples were used for the analysis of PPCP residuals. All experiments were performed in triplicates.

To study the effect of salinity on the sorption of PPCPs to microplastics, solutions of different salinity were prepared by dissolving a certain amount of sea salt in DI water. A final salinity of 0.05, 0.1, 0.5, 1.5, and 3.5 % was achieved. Aliquots of 50 mL prepared saline solution were added to 60-mL glass vials. Then, standard solution of each PPCPs was added to achieve a concentration of 100 μg/L. The vials were equilibrated on a reciprocal shaker for 5 days, and the concentrations of the PPCPs in the liquid phase were measured.

To study the effect of DOM on the sorption of PPCPs to microplastics, HA solution was prepared by dissolving HA in 0.01 M CaCl₂ solution with 0.01 % (w/v) NaN₃ and passed through a Whatman GF/C filter. The dissolved organic carbon (DOC) content of the filtered HA solution was 187 mg/L as determined by a vario cube TOC analyzer (Elementar GmbH, Germany), and the solution was diluted to a DOC content of 0, 1, 5, 10, and 20 mg/L, respectively. Aliquots of 50 mL

Table 1 Structures and selected physicochemical properties of the studied compounds

Compound	CAS number	Structure	Log K_{ow}[a]	pK_a	Application
Carbamazepine (CBZ)	298-46-4		2.45	2.3	Anticonvulsant
17α-Ethinyl estradiol (EE2)	57-63-6		3.67	10.33	Contraceptive
4-Methybenzylidene camphor (4MBC)	36861-47-9		5.1	NA[b]	UV filter
Triclosan (TCS)	3380-34-5		4.76	7.9	Antimicrobial

[a] Database value from the U.S. Environmental Protection Agency EPI Suite Ver.4.11

[b] None ionizable

prepared HA solution were added to 60-mL glass vials. Then, standard solution of each PPCPs was added to achieve an initial concentration of 100 μg/L. The vials were equilibrated on a reciprocal shaker for 5 days, and the concentrations of the PPCPs in the liquid phase were measured.

The sorption of PPCPs to HA was also examined using a solid phase extraction method with a slight modification (Ding et al. 2013). Briefly, each PPCP was added to 60-mL glass vials containing 50 mL of HA solution (DOC content 75 mg/L) to a concentration of 100 μg/L. The vials were equilibrated on a reciprocal shaker for 5 days, and then the solution was passed through a 6-mL, 200-mg Phenomenex Strata-X cartridge (Torrance, CA, USA) conditioned with

2×3 mL DI water and 2×3 mL methanol. HA-associated PPCPs were not retained in the cartridge during loading. Freely dissolved PPCPs were then eluted from the cartridge with 2×3 mL methanol. The sorption coefficients between HA and water (K_{DOC}) of the PPCPs were calculated based on the measured PPCP concentrations in the eluates with and without the addition of HA in the solution.

Chemical analysis

Water samples were adjusted to pH 5 and were concentrated by solid phase extraction (SPE) using 6-mL, 200-mg Phenomenex Strata-X cartridges. Cartridges were conditioned

Fig. 1 The sorption kinetics of the PPCPs to PE debris (*error bars* represent the standard deviation of three replicates)
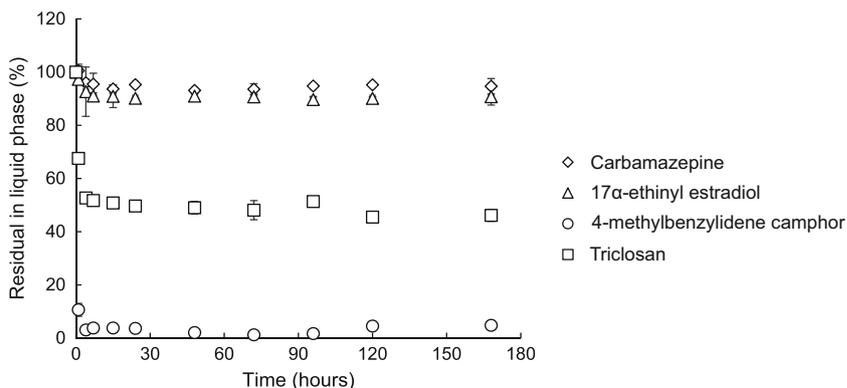
with 2×3 mL DI water and 2×3 mL methanol. Samples were loaded at a flow rate of 5 mL/min approximately. After loading, cartridges were vacuum-dried for 10 min and were eluted with 2×3 mL methanol. The elute was reduced to 1 mL under a gentle nitrogen stream, transferred to a 2-mL amber glass vial, and stored at 4 °C until analysis. Recoveries for the compounds were 93–98 %.

The concentrations of PPCPs were determined using a Waters 2695 high-performance liquid chromatography (HPLC) with a 2996 photodiode array detector (Milford, MA, USA). A Phenomenex Kinetex C18 column (75 mm × 4.6 mm × 2.6 μm) was used for the separation (Torrance, CA, USA). The column temperature was kept at 30 °C; the flow rate was 0.7 mL/min. The mobile phase A was 0.05 % trifluoroacetic acid and the mobile phase B was acetonitrile. The elution gradient started with 30 % B and kept for 3 min, ramped to 70 % in 9 min, held for 2 min, returned to 30 % in 1 min, and re-equilibrated for 3 min. The retention time was 3.91, 8.03, 9.97, 12.01, 13.47, and 14.09 min, and the wavelength used for quantification was 285, 230, 230, 230, 298, and 211 nm for CBZ, EE2, TCS, 4MBC, and PCNB respectively.

Data analysis

Linear model $C_s = K_d \times C_w$ was used to describe the sorption of PPCPs to PE, where C_s is the equilibrium solid phase concentration, C_w is equilibrium liquid phase concentration, and K_d is the linear sorption coefficient, respectively. The value of C_w was instrumentally determined, while C_s was calculated based on mass balance. Sorption parameters were estimated using SPSS v15.0 software (Chicago, IL, USA).

Results and discussion

Sorption isotherms

The sorption isotherms of the selected PPCPs to PE particles were presented in Fig. 2. Sorption parameters estimated using linear model were provided in Table 2. The estimated K_d values varied from 191 to 53,225 L/kg. 4MBC had the highest sorption capacity to PE particles followed by TCS, EE2, and CBZ. The Log K_d values of the compounds were found positively correlated with their octanol-water partition coefficients (Log K_{ow}) ($r^2 = 0.83$), suggesting more hydrophobic

Fig. 2 The sorption isotherms of the PPCPs to PE debris (*error bars* represent the standard deviation of three replicates)
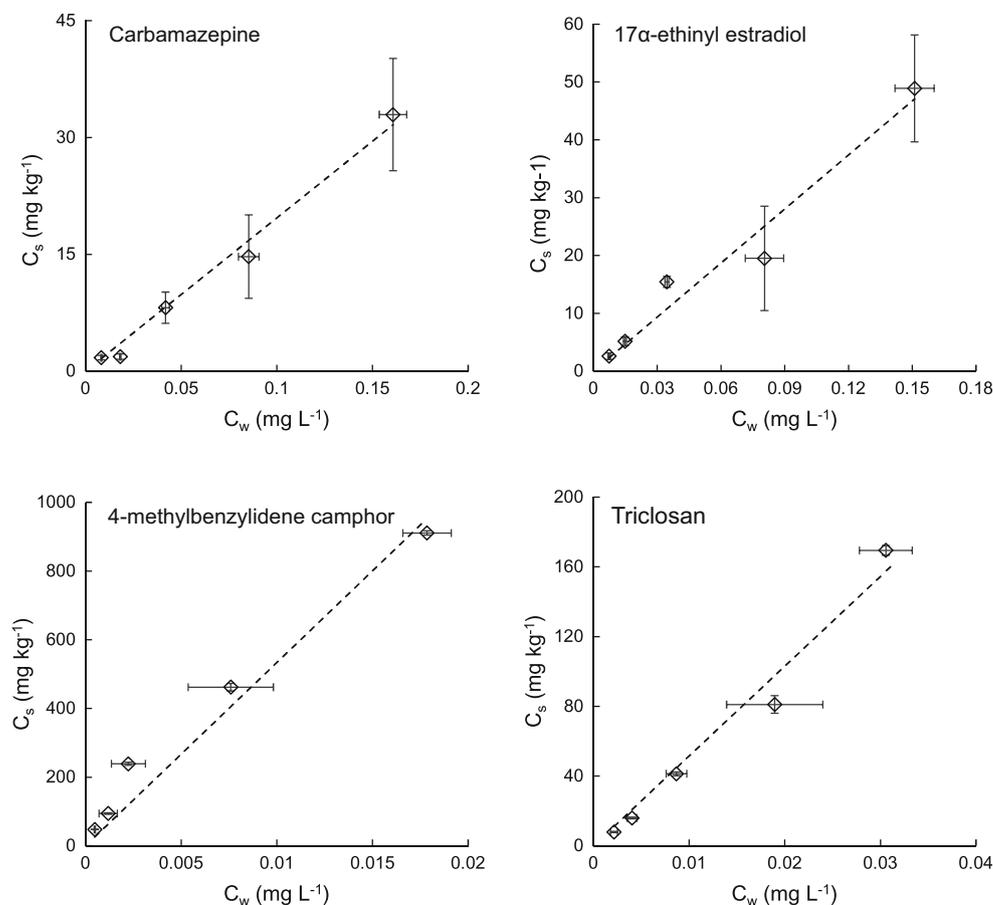
Table 2 Parameters (mean ± standard error) for the sorption of the selected PPCPs to PE particles estimated using linear sorption model

Compound	K_d (L/kg)	r^2
CBZ	191 ± 6.4	0.991
EE2	312 ± 21.5	0.959
4MBC	53,225 ± 3715	0.957
TCS	5140 ± 290	0.973

PPCPs can have higher affiliations to PE particles. Our results demonstrate that microplastics could also be a potential carrier for PPCPs, especially for those highly hydrophobic ones.

Previously, Lee et al. (2014) studied the sorption of hydrophobic organic compounds including PAHs, hexachlorocyclohexanes, and chlorinated benzenes using a third-phase partitioning method. The Log K_d values of the studied compound between PE particles and seawater were between 2.04 and 7.87. Their sorption capacity was also found related to the Log K_{ow} values of the compounds. Fries and Zarfl (2012) investigated the sorption of six PAHs to low- and high-density PE. Results indicated that PAHs had higher diffusion coefficients in low-density PE than in high-density PE, and the diffusion efficiency decreased with the increase of the molecular weight of the PAHs. Sorption of hydrophobic organic compounds to PEs with different crystallinity suggested the importance of rubbery domain within polymer in regulating the sorption of hydrophobic organic compounds (Guo et al. 2012).

PPCPs are a group of bioactive chemicals of diverse physicochemical properties. Some PPCPs can also be considered as hydrophobic such as 4MBC and TCS examined here, which showed a high sorption capacity comparable to those of PAHs and organochlorine pesticides. Whereas many PPCPs are polar and hydrophilic, their sorption to PE can be less pronounced due to a weak interaction with the hydrophobic surface of PE particles. However, weathering can greatly alter the surface property of plastics and therefore change the sorption behavior (Holmes et al. 2014; Zbyszewski et al. 2014). Carbonyl groups were identified in plastic samples collected from Lake Huron beaches, indicating the oxidation of the weathered plastics (Zbyszewski and Corcoran 2011). The introduction of polar functional groups may enhance the electrostatic interaction between hydrophilic PPCPs and PE particles.

Influence of salinity

As microplastics transport from rivers to oceans, salinity of the surrounding water will change gradually. The influence of salinity on the sorption behavior of PPCPs may play an important role on their fate and transport in aquatic environment. As presented in Fig. 3, increase of salinity from 0.05 to 3.5 % (from freshwater to saline water) did not significantly affect the sorption of 4MBC, CBZ, and EE2, whereas the sorption capacity of

TCS increased gradually with the increase of salinity. Increase of salinity can reduce solubility (known as salting-out effect) and thus enhance the sorption of organic compounds, which was observed in marine sediments (Soubaneh et al. 2014; Xu and Li 2009; Yang and Zheng 2010). Increased sorption of TCS at higher salinity levels might be related to the salting-out effect. However, no influence of salinity on the sorption was found for the rest of the compounds, suggesting the salting-out effect on the sorption is compound-specific and hydrophobic compounds are more likely to be affected.

In a previous study, salinity showed no significant effect on the sorption of phenanthrene to PE and polyvinyl chloride (PVC) but increase of salinity decreased the sorption of dichlorodiphenyltrichloroethane (DDT) (Bakir et al. 2014b). Study on the sorption of perfluorooctanesulfonate (PFOS) and perfluorooctanesulfonamide (FOSA) on microplastics found that increase of ionic strength (CaCl$_2$ and NaCl) had no influence on the sorption of FOSA but increased the sorption of PFOS on PE and PS, suggesting electrostatic interaction plays an important role in the sorption of PFOS (Wang et al. 2015). Based on results from this work and previous research, the effect of salinity on the sorption of organic pollutants to microplastics can vary among compounds and plastic types.

Influence of dissolved organic matter

DOM is another important component in water that may affect the sorption behavior of organic pollutants. Here, we use HA as a proxy of DOM, and the influence of HA with DOC content varied from 0 to 20 mg/L on the sorption of PPCPs to PE particles was presented in Fig. 4. Increase of DOC content decreased the sorption capacity of the 4MBC, EE2, and TCS while no significant influence on CBZ was observed. The influence of DOM on the sorption of organic pollutants may involve several processes. Firstly, organic pollutants can interact with DOM via hydrophobic interaction or complexation (Ilani et al. 2005; Nelson et al. 2000). Secondly, DOM can compete with organic pollutants for the sorption sites on the sorbent (Cox et al. 2007). Additionally, DOM can be adsorbed to the sorbent first and then interact with the sorbate (Ling et al. 2006; Sun et al. 2008). Sorption experiment showed that HA have negligible affiliation to the PE particles (data not show), likely due to the hydrophilic properties of the HA. Therefore, reduced sorption of 4MBC, EE2, and TCS could be attributed to their interaction with HA, whereas the sorption of CBZ was not affected, indicating that the interaction between CBZ and HA was of minor importance likely because CBZ is more polar than the rest of the compounds. As the interaction between HA and PE particles is weak, the influence of HA on the sorption of the PPCPs will be primarily determined by the interaction between HA and the PPCPs.

4MBC, EE2, and TCS showed strong affiliations to the HA with estimated K_{DOC} of 1268 ± 32, 1506 ± 23, and 776 ± 9 L/
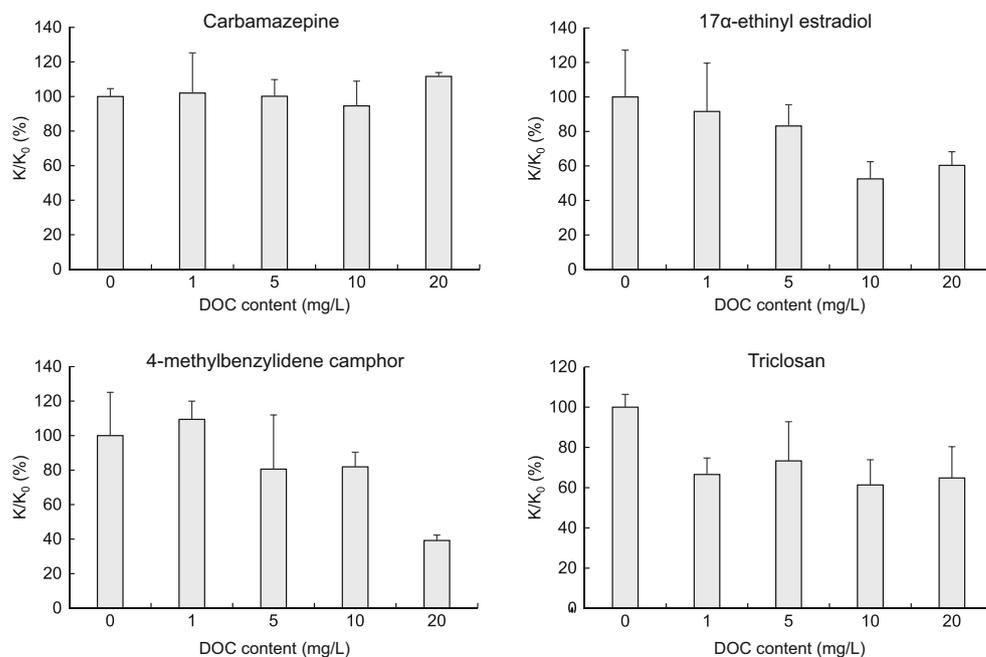
Fig. 3 Influence of salinity on the sorption of the PPCPs to PE debris (*error bars* represent the standard deviation of three replicates)
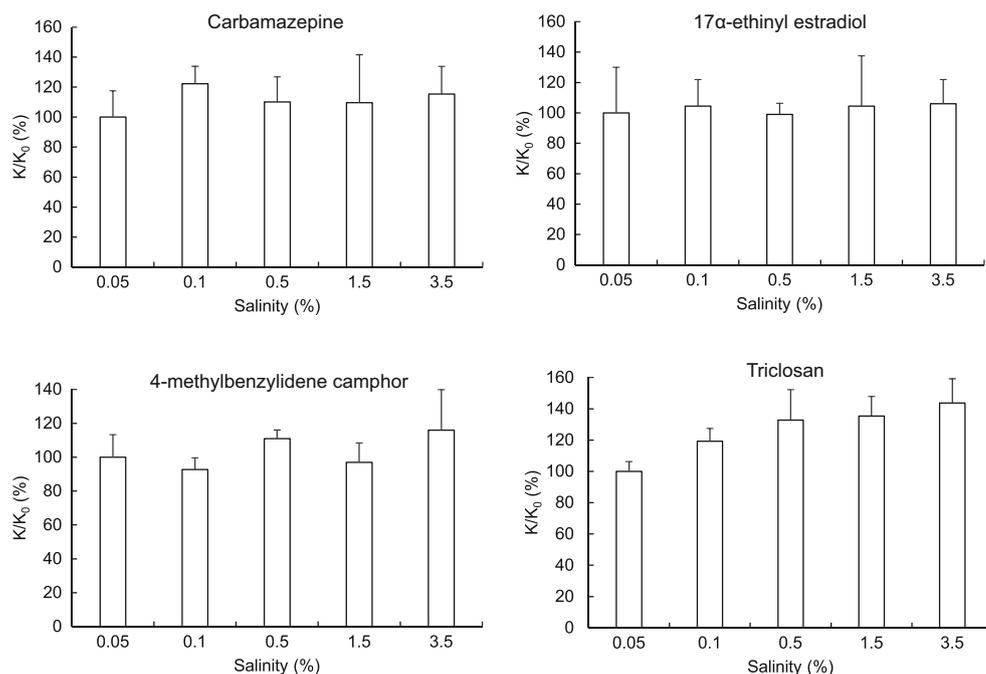

kg when the DOC content of HA is 75 mg/L in the solution, whereas CBZ showed a weak interaction with estimated K_{DOC} of 139 ± 1 L/kg (Fig. 5). Relatively higher K_{DOC} values of 4MBC, EE2, and TCS explain the decreased sorption strength as DOC content increase, and a low K_{DOC} of CBZ supports the invariance sorption observed in Fig. 4, whereas a higher K_{DOC} was estimated for EE2, which is less hydrophobic than 4MBC and TCS, suggesting that other processes such as electrostatic interaction and hydrogen binding might be involved in the sorption of EE2 to HA.

Previously, Maoz and Chefetz (2010) investigated the sorption of CBZ and naproxen to different fractions of sludge-derived DOM. Results show that their interactions are pH dependent, and hydrophobic fractions are less important than the hydrophilic ones for the sorption at the environmental relevant pH conditions. Ding et al. (2013) investigated the interactions between pharmaceuticals (carbadox, lincomycin, and tetracycline) and dissolved humic substances (Leonardite humic acid and Aldrich humic acid). Leonardite humic acid shows a higher affiliation to the compounds than the Aldrich

Fig. 4 Influence of HA with different DOC contents on the sorption of PPCPs to PE debris (*error bars* represent the standard deviation of three replicates)

Fig. 5 Sorption of the PPCPs to HA with a DOC content of 75 mg/L (*error bars* represent the standard deviation of three replicates)
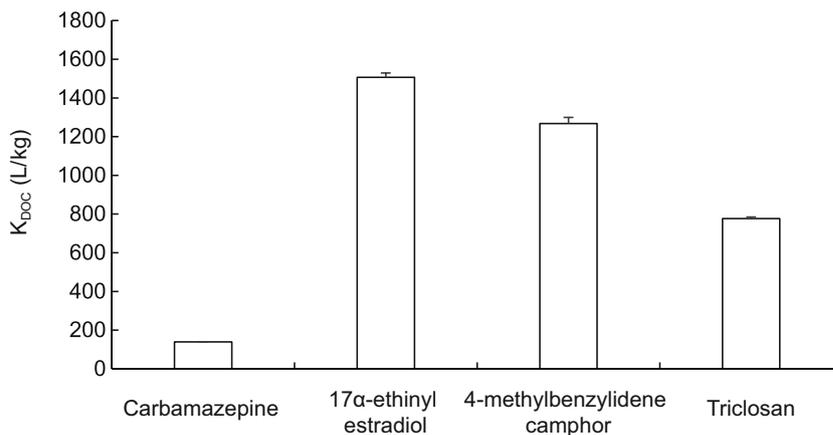
humic acid, likely due to a larger number of carboxylic and phenolic functional moieties within its structure. Therefore, the sorption of PPCPs to DOM can be affected by both hydrophobic and electrostatic interactions, which are related to the structure the compounds as well as the pH condition of the solution.

Conclusions

In this work, the sorption of four PPCPs to PE particles was studied, and the influence of salinity and DOM on the sorption capacity was evaluated. The sorption capacity of the PPCPs was related to their hydrophobicity. Increase of salinity enhanced the sorption of TCS but showed minor influence for the rest of the compounds. Increase of DOM using HA as a proxy reduced the sorption of 4MBC, EE2, and TCS but not CBZ, due to the interaction between the compounds and HA. Our results showed that microplastics might also act as an important carrier for the transport of PPCPs, especially for those hydrophobic ones. Salinity and DOM can have a significant effect on the sorption of PPCPs to plastics but no general trend was observed.

Compliance with ethical standards

References

Bakir A, Rowland SJ, Thompson RC (2014a) Enhanced desorption of persistent organic pollutants from microplastics under simulated physiological conditions. Environ Pollut 185:16–23

Bakir A, Rowland SJ, Thompson RC (2014b) Transport of persistent organic pollutants by microplastics in estuarine conditions. Estuar Coast Shelf Sci 140:14–21

Besseling E, Wegner A, Foekema EM, van den Heuvel-Greve MJ, Koelmans AA (2012) Effects of Microplastic on fitness and PCB bioaccumulation by the lugworm Arenicola marina (L.). Environ Sci Technol 47:593–600

Browne MA, Crump P, Niven SJ, Teuten E, Tonkin A, Galloway T, Thompson R (2011) Accumulation of microplastic on shorelines worldwide: sources and sinks. Environ Sci Technol 45:9175–9179

Caliman FA, Gavrilescu M (2009) Pharmaceuticals personal care products and endocrine disrupting agents in the environment—a review. CLEAN – Soil, Air, Water 37:277–303

Cole M, Lindeque P, Fileman E, Halsband C, Goodhead R, Moger J, Galloway TS (2013) Microplastic ingestion by zooplankton. Environ Sci Technol 47:6646–6655

Cox L, Velarde P, Cabrera A, Hermosín MC, Cornejo J (2007) Dissolved organic carbon interactions with sorption and leaching of diuron in organic-amended soils. Eur J Soil Sci 58:714–721

Ding Y, Teppen BJ, Boyd SA, Li H (2013) Measurement of associated pharmaceuticals with dissolved humic substances using solid phase extraction. Chemosphere 91:314–319

Fair PA, Lee H-B, Adams J, Darling C, Pacepavicius G, Alaee M, Bossart GD, Henry N, Muir D (2009) Occurrence of triclosan in plasma of wild Atlantic bottlenose dolphins (Tursiops truncatus) and in their environment. Environ Pollut 157:2248–2254

Foekema EM, De Gruijter C, Mergia MT, van Franeker JA, Murk AJ, Koelmans AA (2013) Plastic in north sea fish. Environ Sci Technol 47:8818–8824

Fries E, Zarfl C (2012) Sorption of polycyclic aromatic hydrocarbons (PAHs) to low and high density polyethylene (PE). Environ Sci Pollut Res 19:1296–1304

Guo X, Wang X, Zhou X, Kong X, Tao S, Xing B (2012) Sorption of four hydrophobic organic compounds by three chemically distinct polymers: role of chemical and physical composition. Environ Sci Technol 46:7252–7259

Hidalgo-Ruz V, Gutow L, Thompson RC, Thiel M (2012) Microplastics in the marine environment: a review of the methods used for identification and quantification. Environ Sci Technol 46:3060–3075

Holmes LA, Turner A, Thompson RC (2012) Sorption of trace metals to plastic resin pellets in the marine environment. Environ Pollut 160: 42–48

Holmes LA, Turner A, Thompson RC (2014) Interactions between trace metals and plastic production pellets under estuarine conditions. Mar Chem 167:25–32

Ilani T, Schulz E, Chefetz B (2005) Interactions of organic compounds with wastewater dissolved organic matter. J Environ Qual 34:552–562

Klein S, Worch E, Knepper TP (2015) Occurrence and spatial distribution of microplastics in river shore sediments of the Rhine-Main area in Germany. Environ Sci Technol 49:6070–6076

Lee H, Shim WJ, Kwon JH (2014) Sorption capacity of plastic debris for hydrophobic organic chemicals. Sci Total Environ 470–471:1545–1552

Lee KW, Shim WJ, Kwon OY, Kang JH (2013) Size-dependent effects of micro polystyrene particles in the marine copepod Tigriopus japonicus. Environ Sci Technol 47:11278–11283

Ling W, Xu J, Gao Y (2006) Dissolved organic matter enhances the sorption of atrazine by soil. Biol Fertil Soils 42:418–425

Maoz A, Chefetz B (2010) Sorption of the pharmaceuticals carbamazepine and naproxen to dissolved organic matter: role of structural fractions. Water Res 44:981–989

Murray F, Cowie PR (2011) Plastic contamination in the decapod crustacean Nephrops norvegicus (Linnaeus, 1758). Mar Pollut Bull 62:1207–1217

Nelson SD, Letey J, Farmer WJ, Williams CF, Ben-Hur M (2000) Herbicide application method effects on napropamide complexation with dissolved organic matter. J Environ Qual 29:987–994

O'Brine T, Thompson RC (2010) Degradation of plastic carrier bags in the marine environment. Mar Pollut Bull 60:2279–2283

Ratola N, Cincinelli A, Alves A, Katsoyiannis A (2012) Occurrence of organic microcontaminants in the wastewater treatment process. A mini review. J Harzardous Mater 239–240:1–18

Rochman CM et al (2013a) Policy: classify plastic waste as hazardous. Nature 494:169–171

Rochman CM, Hoh E, Hentschel BT, Kaye S (2012) Long-term field measurement of sorption of organic contaminants to five types of plastic pellets: implications for plastic marine debris. Environ Sci Technol 47:1646–1654

Rochman CM, Hoh E, Kurobe T, Teh SJ (2013b) Ingested plastic transfers hazardous chemicals to fish and induces hepatic stress. Scientific Reports 3:3263

Sadri SS, Thompson RC (2014) On the quantity and composition of floating plastic debris entering and leaving the Tamar Estuary, Southwest England. Mar Pollut Bull 81:55–60

Sánchez Rodríguez A, Rodrigo Sanz M, Betancort Rodríguez JR (2015) Occurrence of eight UV filters in beaches of Gran Canaria (Canary Islands). An approach to environmental risk assessment. Chemosphere 131:85–90

Setälä O, Fleming-Lehtinen V, Lehtiniemi M (2014) Ingestion and transfer of microplastics in the planktonic food web. Environ Pollut 185:77–83

Singh B, Sharma N (2008) Mechanistic implications of plastic degradation. Polym Degrad Stab 93:561–584

Soubaneh YD, Gagné J-P, Lebeuf M, Gouteux B, Nikiforov V, Awaleh MO (2014) Sorption behaviors of a persistent toxaphene congener on marine sediments under different physicochemical conditions. Chemosphere 114:310–316

Sun Z, Mao L, Xian Q, Yu Y, Li H, Yu H (2008) Effects of dissolved organic matter from sewage sludge on sorption of tetrabromobisphenol A by soils. J Environ Sci 20:1075–1081

Thompson RC et al (2004) Lost at sea: where is all the plastic? Science 304:838

Van Cauwenberghe L, Janssen CR (2014) Microplastics in bivalves cultured for human consumption. Environ Pollut 193:65–70

von Moos N, Burkhardt-Holm P, Köhler A (2012) Uptake and effects of microplastics on cells and tissue of the blue mussel Mytilus edulis L. after an experimental exposure. Environ Sci Technol 46:11327–11335

Wang F, Shih KM, Li XY (2015) The partition behavior of perfluorooctanesulfonate (PFOS) and perfluorooctanesulfonamide (FOSA) on microplastics. Chemosphere 119:841–847

Xu XR, Li XY (2009) Sorption behaviour of benzyl butyl phthalate on marine sediments: equilibrium assessments, effects of organic carbon content, temperature and salinity. Mar Chem 115:66–71

Yang G-P, Zheng X (2010) Studies on the sorption behaviors of phenanthrene on marine sediments. Environ Toxicol Chem 29:2169–2176

Zbyszewski M, Corcoran P (2011) Distribution and degradation of fresh water plastic particles along the beaches of Lake Huron, Canada. Water Air Soil Pollut 220:365–372

Zbyszewski M, Corcoran PL, Hockin A (2014) Comparison of the distribution and degradation of plastic debris along shorelines of the Great Lakes, North America. J Great Lakes Res 40:288–299

Zhang K, Gong W, Lv J, Xiong X, Wu C (2015) Accumulation of floating microplastics behind the Three Gorges Dam. Environ Pollut 204:117–123