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REVIEW

The occurrence and ecological risk assessment of phthalate esters (PAEs) in urban aquatic environments of China

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Abstract Phthalate esters (PAEs) are widely used in the manufacturing of plastics, and the demand for PAEs has grown rapidly, especially in China. This trend will lead to much more environmental PAE contamination. PAEs are listed as priority substances in the European Union and are therefore subject to ecological risk assessments. This paper reviews the literature concerning the pollution status of PAEs and their ecological risk to aquatic environments. Risk quotients (RQs) based on the predicted no effect concentration and PAE concentrations in aquatic environments demonstrated significant ($10 \leq RQ < 100$) or expected ($RQ \geq 100$) potential adverse effects for algae, *Daphnia*, and fish in aquatic environments near PAE-based industrial and urban areas. Thus, the ecological risk of PAEs in Chinese aquatic environments should be considered, especially in areas where commercial plastics are produced.

Keywords Phthalate esters (PAEs) · Environmental media · Pollution level · Ecological risk · China

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Introduction

Phthalate esters (PAEs) are restricted to the *ortho* form of benzenedicarboxylic acid prepared by reaction of phthalic acid with a specific alcohol to form the desired ester. Most of the esters are colorless liquids, have low volatility, and are poorly soluble in water but soluble in organic solvents and oils (Autian 1973). PAEs are endocrine disruptors and have been used as plasticizing agents in cellulosics and elastomers (Graham 1973). In addition, PAEs are widely applied in many daily products including floor tiles, various types of furnishings for households and transportation vehicles, food packaging systems, industrial tubing and conduits, medical tubing, catheters, blood containers, certain types of dental materials, drug coatings, and numerous other products. As a result, PAEs are ubiquitous pollutants in the environment and have been widely detected in the air, water, soil, and sediments (Peijnenburg and Struijs 2006; Teil et al. 2006). For example, according to the Toxics Release Inventory (TRI) database, the total release of PAEs in the USA in 2012 was 1, 492, 674 kg, with releases of 1, 354, 968 kg to air, 237 kg to water (direct), 1, 457 kg released on-site to land, 2, 204 kg transferred off-site to land, and 136, 010 kg to off-site disposal or other releases (USEPA 2012).

Due to their possible teratogenic, mutagenic, and carcinogenic effects (Koch and Calafat 2009; Meeker et al. 2009; Oehlmann et al. 2009; Talsness et al. 2009; Shaxson 2009), PAEs are classified as priority pollutants by the United States Environmental Protection Agency (USEPA) and several other countries (CCME 1989; USEPA 1980). Although several regulatory measures have been initiated at local, regional, and global levels to control the production and use of some of these

Table 1 Physical and chemical characteristics of 16 PAEs

Phthalate esters	CAS	Abbreviation	Molecular weight	Solubility in water (mg/L)	Koc (L/Kg) (soil/sediment)
Dimethyl phthalate	131-11-3	DMP	194	2014	55.0–360
Diethyl phthalate	84-66-2	DEP	222	287	69.0–1726
Diallyl phthalate	131-17-9	DAP	246	43.3	–
Diisobutyl phthalate	84-69-5	DiBP	278	5.06	–
Di-*n*-butyl phthalate	84-74-2	DnBP	278	2.35	1375–14,900
Bis(2-methylglycol) phthalate	117-82-8	DMGP	282	1737	–
Bis(2-ethoxyethyl) phthalate	605-54-9	DEEP	310	173	–
Butyl benzyl phthalate	85-68-7	BBP	312	0.950	9000–17,000
Dicyclohexyl phthalate	84-61-7	DCHP	330	0.0400	–
Bis(4-methyl-2-pentyl) phthalate	146-50-9	BMPP	334	0.0200	–
Dihexyl phthalate	84-75-3	DHP	334	0.0100	52,600
Hexyl-2-ethylhexyl) phthalate	75,673-16-4	HEHP	362	0.0200	–
Bis (2-*n*-butoxyethyl) phthalate	117-83-9	DBEP	366	1.67	–
Di-*n*-octyl phthalate	117-84-0	DnOP	390	0.00	–
Bis(2-ethylhexyl) phthalate	117-81-7	DEHP	390	0.00	87,420–510,000
Dinonyl phthalate	84-76-4	DiNP	418	0.00	–

chemicals (Andrady and Neal 2009), PAE use is not regulated in Asia. Furthermore, the domestic demand for PAEs in China has increased at an annual rate of 7.70 % from 2010 to 2015 (Emanuel 2011), and the total demand of PAEs was approximately 1.36×10^6 tons in 2010 (Wang et al. 2008a, b). In addition, China is currently the world's largest PAEs importer, aggravating PAE contamination.

Effective environmental management practices for pollutants are dependent on the clear understanding of the ecological risk of chemicals (Lei et al. 2008). Therefore, more and more studies have investigated the potential ecological risk from PAEs in the environment (Staples et al. 2000; Xia et al. 2011). Few studies, however, have evaluated the ecological risk of PAEs, even though PAEs have been detected in different types of media (water, soil/sediment, and air) in recent years. The toxicity of PAEs to aquatic organisms generally increases with increasing alkyl chain length up to the point where a critical body burden cannot be attained due to the low aqueous solubility of the ester (Adams et al. 1995). For phthalate esters with alkyl chain lengths of six or more carbons, the results of numerous acute and chronic aquatic toxicity studies using many species indicate no toxicity at the solubility limit (Staples et al. 1997a). Therefore, the aim of this study was to reveal the ecological risk for the most detected PAEs with lower molecular weights (DMP, DEP, DBP, BBP, DHP, and DEHP) in aquatic environments in China.

Materials and methods

Data collection strategies

To reflect the overall status of PAE research in aquatic environments of China, a systematic literature review was performed using an electronic search of Elsevier, Springer, Google Scholar, ISI Web of Knowledge, and PubMed. Literature published in Chinese was retrieved from the China Knowledge Resource Integrated Database and the Wanfang of E-Resources for China studies, with master's theses and doctoral dissertations included. Given the large number of studies in the literature, our study focused on those that are most relevant to Chinese aquatic environments. Studies that failed to report details of occurrence data and/or geographical information were excluded. Data on the toxicology of PAEs to non-target organisms were retrieved from the USEPA ECOTOXicology database (USEPA 2012) and supplemented by journal articles screened by the criteria of accuracy, relevance, and reliability (Klimisch et al. 1997). The collected toxicity data should be obtained under the guideline of good laboratory practice.

Analytical methods for PAEs in environmental media

This section will briefly introduce several PAE analysis methods. Cao (2010) reviewed the analytical methods that

Table 2 The supply–demand situation of PAEs in the mainland China (10^4 tons)

Year	Output (10^4 tons)	Net import (10^4 tons)	Apparent volume of consumption (10^4 tons)
2000	39.5	36.4	75.9
2001	46.4	36.7	83.1
2002	53.7	43.6	97.2
2003	54.0	51.0	105
2004	55.0	50.0	105
2005	50.5	45.8	96.3
2006	67.3	48.7	116
2007	66.6	46.9	113
2008	90.0	35.0	128
2009	104	32.0	136
2010	112	21.6	136

have been reported for the determination of PAEs in food. The main strategies developed included sample preparation, extraction, cleanup, separation, and detection. Extraction and cleanup are the most challenging steps for phthalate analysis in foods and are often the critical steps that determine the detection limits of the overall methods. The extraction method can be subdivided into solvent-based or liquid–liquid extraction (Page and Lacroix 1995) and solid-phase microextraction (SPME) (Arthur and Pawliszyn 1990). For separation and detection, studies have mainly focused on techniques such as liquid chromatography–mass spectrometry (LC–MS) (Lin et al. 2003), gas chromatography–mass spectrometry (GC–MS) (Farahani et al. 2007), and other methods (Kozyrod and Ziaziaris 1989; Cai et al. 2003; Sorensen 2006; Hogberg et al. 2008).

The analysis of PAEs in environmental media is generally similar to the analysis of PAEs in food, with the exception of the sample treatment process. Cai et al. (2003) developed a new analytical method for the analysis of PAEs in surface waters using solid-phase extraction, quantitative desorption with acetonitrile, and determination by high performance liquid chromatography (HPLC). In another study (Cortazar et al. 2005), a method to determine PAEs in sediment using solid phase extraction, desorption with MeOH, and GC–MS was developed. For atmospheric samples, Wang et al. (2008a, b) used liquid–liquid extraction and concentration, desorption with cyclohexane, and determination by GC–MS.

The fate of PAEs in ecosystems

Water solubility

Water solubility is an extremely important property that influences the biodegradation and bioaccumulation potential of a chemical as well as aquatic toxicity. Water solubility is also a determining factor that controls the environmental distribution of chemicals (Staples et al. 1997b). PAEs have water solubilities ranging from approximately 2014–0.00 mg/L. Most of the higher molecular weight phthalate esters (alkyl chain length of C6 or greater) are actually mixtures of closely related isomers. The physical and chemical characteristics of 16 PAEs are shown in Table 1.

Soil/sediment sorption

The sorption of phthalate esters to soil, sediment, or suspended solids is partially governed by the relative hydrophobicity of the chemical. The sorption is not always linear with the chemical concentration in the soil, and it may vary considerably with the particular solid used (Carlberg and Martinsen 1982). A number of authors have published soil or sediment and water partition coefficients (Table 1). In addition, several authors have examined the dissolved versus suspended particulate-bound fraction of phthalate esters in surface water samples. For example, Ritsema et al. (1989) used centrifugation to separate SPM from surface water samples collected from Lake Yssel and the Rhine River (Netherlands). The geometric mean of SPM values ranged from 4.00 to 100 mg/L; 98.0 % of the DBP present was dissolved, while only 2.00 % was SPM-bound.

Air–water partitioning

The equilibrium distribution of a chemical between water and air serves as a guide to estimate the tendency of a substance to escape from water into air. The ratio of the vapor pressure to the molar water solubility is an estimate of Henry's Law constant, which is a measure of the equilibrium distribution coefficient (Thomas 1982).

For lower molecular weight phthalate esters (DMP, DEP, DAP, DPP, DnBP, DiBP, and BBP), H values ranged from 1.2E−7 to 8.8E−7 atm-m^3/mole. Compounds with H values in the range of 1.0E−7 atm-m^3/mole are generally

Table 3 Concentrations of PAEs in aquatic environment in China, compared with other countries

Location	PAEs levels (ng/l)	Media	References
China			
Lake and reservoir water			
Beijing-Park Lakes	6400–138,100	Park lake water	Zhong et al. 2010
Suzhou-Taihu Lake	1888–126,100	Urban lake waters	Wang et al. 2003
Taiyuan-Fenhe Reservoir	37,490	Reservoir water	Guo et al. 2002
Guangzhou-Urban Lakes	1690–4720	Urban lakes water	Zheng et al. 2014
Beijing-Urban Lakes	386–3184	Urban lakes water	Zheng et al. 2014
River water			
Haihe River	3890–141,780	Urban river waters	Chi 2009
Yangtze River-Wuhan	34.0–91,220	Urban river water	Wang et al. 2008a, b
Yellow River-Taiyuan	87,230	Urban river water	Guo et al. 2002
Songhua River-Jilin	2500–68,960	Urban river water	Wei et al. 2011
Yellow River	3990–45,450	Middle and lower reaches	Sha et al. 2007
Taiwan Rivers	1000–36,300	River waters	Yuan et al. 2002
Yangtze River Delta	61.0–28,550	River water	Zhang et al. 2012
Xiangjiang River-Zhuzhou	22,390–27,400	River water	Tang et al. 2010
Qiantangjiang River-Zhejiang	4150–15,380	River water	Zhang et al. 2003
Jiangshu-Yangtze River	178–1474	River water	He et al. 2011
Surface water			
Shanghai	0.00–13,530	Surface water	Zhang et al. 2003
Yangzhou	0.00–10,430	Surface water	Zhang et al. 2003
Estuary/Port/Harbor water			
Yangtze Estuary area	3380	Marine water	Liu et al. 2006
Ground water			
Guangzhou-Dongguan	0.00–6700	Groundwater	Zhang et al. 2011a, b
Hubei-Jianghan Plain	80.1–1882	Groundwater	Zhang et al. 2009
Drinking water source			
Yangtze River Delta-Suzhou	5700–14,000	Drinking water source	Shi et al. 2012
Yangtze River Delta-Wuxi	6300–12,000	Drinking water source	Shi et al. 2012
Yangtze River Delta-Changzhou	3500–8300	Drinking water source	Shi et al. 2012
Yangtze River Delta-Yancheng	3000–3800	Drinking water source	Shi et al. 2012
Yangtze River Delta-Xuzhou	40.0	Drinking water source	Shi et al. 2012
Wastewater treatment plants			
Beijing-Wastewater treatment plants	41,440–69,880	Wastewater treatment plants influents	Lin et al. 2004
Harbin	21,010	Municipal wastewater	Huang et al. 2013
Beijing-Wastewater treatment plants	720–4240	Wastewater treatment plants effluents	Lin et al. 2004
Other countries			
Lake water			
USA-Lake Pontchartrain	0.00–20,000	Lake water	Liu et al. 2013
River water			
Nigeria-Ogun River	3,950,000–4,775,000	River water	Adeniyi et al. 2011
Malaysia-Klang River	5000–69,200	River water	Tan 1995
Italy-Rieti river	0.00–45,900	River water	Vitali et al. 1997
Sweden-Svartan River	320–3100	River water	Thurén 1986
UK-Trent River	740–1800	River water	Long et al. 1998
France-Seine River	464–771	River water	Dargnat et al. 2009
Spain-Embo river	0.00–700	River water	Penalver et al. 2000

Table 3 continued

Location	PAEs levels (ng/l)	Media	References
Spain-Galicia river	69.0	River water	Polo et al. 2005
Surface water			
Germany-Berlin	330–97,800	Surface water	Fromme et al. 2002
Netherlands	540–26,200	Freshwater	Peijnenburg and Struijs 2006
Italy-Rieti Rain Water	3700–11,400	Residental center rain water	Guidotti et al. 2000
Italy-Rieti Rain Water	3200–8300	City center rain water	Guidotti et al. 2000
Italy-Rieti Rain Water	7100–7800	Industrial center rain water	Guidotti et al. 2000
Estuary/Port/Harbor water			
Thailand	8640	Gulf water	Sirivithayapakorn and Thuyviang 2010
Canada-False Creek Harbor	3.30–1060	Marine water	Mackintosh et al. 2006
Spain-Industrial Port	0.00–2120	Industrial port water	Penalver et al. 2000
Drinking water source			
Japan-Tokyo	210–5700	Surface water	Fatoki and Noma 2002
Germany	1010	Tap water	Serôdio and Nogueira 2006
Germany	740	Bottled mineral water	Serôdio and Nogueira 2006
Wastewater treatment plants			
Germany-Berlin	1740–182,000	Sewage effluents	Fromme et al. 2002
Spain-Galicia wastewater treatment plants	6623	Wastewater treatment plants influent	Polo et al. 2005
France	2819–4682	Wastewater treatment plants effluents	Dargnat et al. 2009
Spain-Galicia wastewater treatment plants	1278	Wastewater treatment plants effluent	Polo et al. 2005

considered to have negligible volatility (Howard et al. 1985). For all higher molecular weight phthalate esters except BOP, the calculated H values ranged from approximately 1.7E−5 to 5.5E−4 atm-m^3/mole. The higher H values are due to a greater decrease in water solubility relative to vapor pressure with increasing alkyl chain length. The H values for some of the higher molecular weight phthalate esters (DiNP, DiDP, and DTDP) are difficult to calculate due to the extremely low vapor pressures and water solubilities that are not accurately known.

Degradation

In both aquatic and terrestrial systems (e.g., sewage, soils, sediments, and surface water), microbial action is thought to be the principal mechanism of PAE degradation (Staples et al. 1997b). Microorganisms that degrade PAEs can be aerobic (Wang et al. 1995), anaerobic (Shelton et al. 1984), or facultative. Precise measurements of biodegradation rates are considered important to accurately forecast the fates of potential pollutants and assess risk.

It is generally accepted that only the truly dissolved phase of a non-polar organic chemical is bioavailable (Steen et al. 1980). Therefore, the partitioning of phthalate esters into colloidal and particulate organic carbon should be considered in the analysis of field samples. Most historical measurements, however, are based on total concentrations, which fail to differentiate free and complex forms. Consequently, the available field data may significantly overestimate bioavailability, especially for the more hydrophobic phthalates that are expected to exist principally in the environment as complex forms (Staples et al. 1997a).

Ecological risk assessment

Ecological risk assessment of PAEs was conducted according to the European Commission's Technical Guidance Document (EC 2003) and previous studies (Staples et al. 2000). The risk quotient (RQ) approach based on the measured contaminant concentrations in surface waters was used to assess the potential ecological risk. In this study, the RQ was assessed on non-target organisms, as described in previous studies (Staples et al. 1997b, 2000; USEPA 1995). The RQs were calculated as the quotient of the measured environmental concentration (MEC) and the predicted no effect concentration (PNEC). PNEC was estimated as the quotient of toxicologically relevant concentration and a security factor (f). For this purpose, the LC_{50} or EC_{50} values for fish, *Daphnia,* and algae associated with DMP, DEP, DnBP, and BBP were used for the RQ calculations. The RQs of PAEs were calculated as follows:

Table 4 Concentrations of PAEs in sediment/soil in China, compared with other countries

Location	PAEs levels (ng/g dry weight)	Media	References
China			
River and lake sediments			
Yangtze River-Wuhan	76,300–450,000	Urban river sediments	Wang et al. 2008a, b
Yellow River	30,520–85,160	Urban section sediments	Sha et al. 2007
Guangzhou-Urban Lake	2270–74,940	Urban lake sediments	Zheng et al. 2014
Taihu Lake-Urban Lake	2590–56,170	Urban lake sediments	Wang et al. 2003
Taiwan-Urban Rivers	2900–27,100	River sediments	Yuan et al. 2002
Beijing-Urban Lakes	52.6–8216	Urban lakes sediments	Zheng et al. 2014
Shanghai-Suzhou River	980–7720	Urban river sediments	Hu and Han 2011
Zhejiang-Qiangtang River	590–6740	Urban river sediments	Sun et al. 2013
Haihe River	430–3320	River sediments	Chi 2009
Beijing-Guangting Reservoir	479–2119	Drinking water source sediments	Wang et al. 2006
Marine/Estuary/Harbor sediments			
Taiwan-Kaohsiung Harbor	400–34,800	Marine sediments	Chen et al. 2013
Urban soils			
Guangzhou	1670–322,000	Urban soils	Zeng et al. 2009
Beijing	1.90–31,417	Urban and rural soils	Xia et al. 2011
Beijing	508–7959	Urban soils	Li et al. 2006
Yangzhou	0.00–2592	Urban sediment/soils	Zhang et al. 2003
Hubei-Jianghan Plain	253–2516	Alluvial sediments	Liu et al. 2010
Shanghai-Jinshan	51.0–980	Rural sediment/soils	Zhang et al. 2003
Agricultural soil			
Guangzhou	9700–58,900	Green and organic vegetable soils	Li et al. 2006
Guangzhou and Shenzhen	3000–45,670	Vegetable soils	Cai et al. 2005
Guangzhou	195–33,600	Agricultural soils	Zeng et al. 2008a, b
Haihe River	4330–11,810	Root soils	Chi 2009
Tianjin-Vegetable	50.0–10,400	Vegetable soils	Kong et al. 2012
Zhejiang-Taizhou	1810–5770	Agricultural soils	Ma et al. 2013
Leizhoun peninsula	0.00–5450	Agricultural soils	Guan et al. 2007
Hangzhou	1900–4360	Greenhouse plastics vegetable soils	Chen et al. 2011
Beijing	1340–3150	Greenhouse soils	Ma et al. 2003
Tianjin-Farmland	91.0–2740	Farmland soils	Kong et al. 2012
Tianjin-Wasteland	106–1360	Wasteland soils	Kong et al. 2012
Tianjin-Orchard	53.0–1080	Orchard soils	Kong et al. 2012
Zhangjiagang	0.580–762	Agricultural soils	Zhang et al. 2011a, b
Nanchang	0.00–386	Agricultural soils	Xiong et al. 2008
Other countries			
River and lake sediments			
UK-Trent River	840–31,000	River sediment	Long et al. 1998
Malaysia-Klang River	568–16,244	River sediment	Tan 1995
Germany-Berlin	210–8440	Rivers, lakes, and channels sediments	Fromme et al. 2002
Sweden-Svartan River	1200–8030	River sediment	Thurén 1986
Nigeria-Ogun River	325–2880	River sediments	Adeniyi et al. 2011

Table 4 continued

Location	PAEs levels (ng/g dry weight)	Media	References
India-Comti River	0.00–364	Urban river sediments	Srivastava et al. 2010
Italy-Rieti river	10.0–525	River sediment	Vitali et al. 1997
Marine/Estuary/Harbor sediments			
Netherlands-North sea	92,700–727,500	Marine sediments	Peijnenburg and Struijs 2006
Thailand	14,510	Gulf Surface water	Sirivithayapakorn and Thuyviang 2010
Singapore-Tuas Bay	890–2790	Sediment samples	Chee et al. 1996
Canada-False Creek Harbor	4.00–2100	Marine sediments	Mackintosh et al. 2006
Urban sediments/soil			
Germany-Berlin	27,900–154,000	Sewage sludge	Fromme et al. 2002
Singapore-Jurong industrial park	1580–4390	Sediment samples	Chee et al. 1996
Turkey-Istanbul Wastewater treatment plants	1400–2700	Sewage sludge	Çifci et al. 2013
Singapore-Jurong industrial park	750–2040	Soil samples	Chee et al. 1996
Agricultural soils			
Israeli-Coastal Plain	6000	Agricultural soil	Muszkat et al. 1997
Czech Republic-Moravia	310–2320	Agricultural soils	Zorníkova et al. 2011
Denmark	198–303	Agricultural soil	Laturnus and Grøn 2007

$$RQ = \frac{MEC}{PNEC} = \frac{MEC}{\frac{L(E)C_{50}}{f}} \quad (1)$$

For data interpretation, the maximum probable risk for ecological effects from contaminated water was followed as recommended by Wentsel et al. (1996):

RQ < 1.00 (i.e., the exposure point concentration is less than the risk screening benchmark) indicates no significant risk;

1.00 ≤ RQ < 10.0 (i.e., the exposure point concentration is between one and ten times the risk screening benchmark) indicates a small potential for adverse effects;

10.0 ≤ RQ < 100 (i.e., the exposure point concentration is between ten to one hundred times the risk screening benchmark) indicates a significant potential for adverse effects;

RQ ≥ 100 (i.e., the exposure point concentration is equal to or greater than one hundred times the risk screening benchmark) indicates that potential adverse effects should be expected.

Results and discussions

Production and consumption of PAEs in China

Table 2 lists the total supply and demand volumes for PAEs in mainland China during 2000–2010; the variation trend for apparent consumption volume significantly increased during 2000–2010. Table S1 lists several major manufacturers of PAEs in China; the major manufacturers were located in Shangdong, Guangdong, Zhejiang, and Jiangshu province.

PAEs in aquatic environments

Table 3 lists the levels of PAEs detected in aquatic environments in different parts of China compared with other parts of the world. Most studies conducted in China reported PAE concentrations in river and lake waters to be higher than 8.00 μg/L, with the exception of three areas: Guangzhou (Urban Lakes), Beijing (Urban Lakes), and Yangtze River (Jiangsu section). In China, the Environmental Quality Standards for Surface Water (PRC-NS 2002) and Standards for Drinking Water Quality (PRCNS 2006) regulate the concentrations of two (DEHP, 8.00 μg/L and DBP, 3.00 μg/L) and three (DEHP, 8.00 μg/L, DBP, 3.00 μg/L and DEP, 300 μg/L) PAEs, respectively. Therefore, the potential ecological risk of PAEs cannot be ignored (Mo et al. 2001; Lan et al. 2012). Compared with other countries, the PAE concentrations in the waters of China are higher than global PAE levels (higher than 8.00 μg/L).

River water from Ogun River located in Southwestern Nigeria contained a rather high level of 395–4, 775 μg/L PAE, which was 597 times higher than the guideline value for PAEs in environmental waters. With the increasing consumption of PAEs in metropolitan areas, the concentrations of PAEs detected in urban water bodies were obviously higher than those in other areas of China. The data also showed that urban river waters may receive PAE

Table 5 Concentrations of PAEs in air in China, compared with other countries

Location	PAEs levels (ng/m^3)	Media	References
China			
Indoor air			
Zhejiang	330,000	Workplace area	He and Guo 2010
Hangzhou	12,096	Household area	Pei et al. 2013
Outdoor air			
Tianjin	136–203	Industrial area	Kong et al. 2013
Nanjing	41.2–135	Urban area	Wang et al. 2008a, b
Nanjing	10.5–45.1	Rural area	Wang et al. 2008a, b
Tibetan Plateau	1.10–4.10	Remote site	Li et al. 2013
Other countries			
Indoor air			
Japan-Tokyo	20.0–11,400	Metropolitan area	Otake et al. 2004
Sweden-Stockholm area	1200–7400	Home area	Bergh et al. 2011
USA-California	162–5715	Homes area	Rudel et al. 2010
Sweden-Stockholm area	1200–5600	Day care area	Bergh et al. 2011
Sweden-Stockholm area	740–3900	Work area	Bergh et al. 2011
Germany-Berlin	2829	Urban area	Weschler et al. 2008
Germany-Berlin	1188	Kindergarten area	Fromme et al. 2004
Germany-Berlin	1083	Apartment area	Fromme et al. 2004
USA-Cape code	80.0–1080	Home area	Rudel et al. 2003
France-Paris	20.6–109	Urban area	Teli et al. 2006
Germany-Düren	50.0	Indoor air	Fromme et al. 2013
Outdoor air			
USA-California	1.00–924	Homes area	Rudel et al. 2010
Netherlands	40.0–403	Industrialized area	Peijnenburg and Struijs 2006
Netherlands	61.0–89.0	Heavily populated area	Peijnenburg and Struijs 2006
Netherlands	20.0–82.0	Other sites	Peijnenburg and Struijs 2006
Sweden	1.70–75.3	Urban air	Thuren and Larsson 1990
Greece-Thessaloniki	5.10–48.4	Urban traffic area	Salapasidou et al. 2011
Greece-Thessaloniki	1.31–10.8	Urban industrial area	Salapasidou et al. 2011
Germany-Bight	0.00–3.40	North Sea	Xie et al. 2005
Arctic-North Sea	1.11–3.09	Remote area	Xie et al. 2007

discharges from industrial effluents and landfill leakages without effective treatment; for example, extremely high PAE concentrations were detected in the effluents (up to 182 μg/L) from wastewater treatment plants in Berlin, Germany. Furthermore, PAE pollution of water bodies was found not only in surface waters but also in underground waters; for instance, PAE concentrations in the range of 0.00–6.70 μg/L were detected in underground waters in Dongguan, Guangdong Province, China. However, the PAE concentrations (except DBP) in surface water (rivers, lakes, and reservoirs) were generally higher than those in groundwater (Liu et al. 2014).

PAEs in soil and sediment

Table 4 lists some of the reported data on the PAE levels in sediment from different parts of China in comparison to other countries. The concentrations (maximum of 450, 000 ng/g, dry mass) of PAEs in metropolitan sediment/soils were generally higher than those from other

Table 6 Acute toxicity (LC_{50} or EC_{50}) used for the risk assessment for Fish, *Daphnia,* and algae

Compound	Fish *(Lepomis macrochirus)*			Invertebrate *(Daphnia magna)*			Algae *(Selenastrum capricornutum)*		
	L(E)C_{50} (mg/l)	NOEC	References	L(E)C_{50} (mg/l)	NOEC	References	L(E)C_{50} (mg/l)	NOEC	References
DMP	50.0	15.3	Adams et al. 1995	33.0	<1.70	LeBlanc 1980	142	<64.7	Adams et al. 1995
DEP	16.7	1.65	Adams et al. 1995	86.0	37.5	Adams et al. 1995	16.0	3.65	Adams et al. 1995
DBP	0.480	0.420	Adams et al. 1995	3.00	1.70	Adams et al. 1995	0.400	0.21	Adams et al. 1995
BBP	1.70	0.360	Adams et al. 1995	3.70	1.00	Gledhill et al. 1980	0.210	<0.100	Adams et al. 1995
DHP	>0.110	0.110	Adams et al. 1995	>0.180	0.0300	Adams et al. 1995	>0.330	0.180	Adams et al. 1995
DEHP	>0.200	0.200	Adams et al. 1995	>1.00	1.00	Brown and Williams 1994	>0.100	0.100	Adams et al. 1995

areas of China. In addition, PAE concentrations in metropolitan sediment/soils were associated with the rather high levels detected in the water of the same area, confirming the influence of discharge from the local chemical plants associated with the manufacturing and processing of PAEs-based products. Compared with other countries, the PAE concentrations in the sediments/soils of China were higher than the global levels (Table 4). However, sediments from the North Sea in the Netherlands contained the highest level of 92.7–727.5 μg/g PAEs.

PAEs in air

To comprehensively understand the PAE pollution level in China, we analyzed many studies that reported PAE concentrations in air. However, the studies concerning PAE concentrations in air are limited. Studies conducted in China have mainly focused on the ambient air concentrations of PAEs or PAEs in PAE-based factories. Table 5 provides several useful data concerning PAE concentrations in the atmosphere in both China and other countries and regions. These data show that the PAE concentrations (0.00–330,000 ng/m^3) ranged over six orders of magnitude throughout the world, with a declining trend from continents to remote sites. In China, the level of PAEs was higher in Zhejiang than in the Tibetan Plateau, which is not an industrial district or a densely populated area. This result is reasonable because Zhejiang province has many PAE manufacturers.

At present, Rudel and Perovich (2009) reported that indoor air PAE concentrations are higher than outdoor concentrations and that concentrations in urban areas are higher than in rural and remote areas. Therefore, household goods and office furnishings have been demonstrated to be potential emission sources of PAEs (Wormuth et al. 2006). At the same time, outdoor sources of phthalates such as the wearing of tires are known to be secondary to indoor sources (Rakkestad et al. 2007). Kong et al. (2013) noted that the emission from cosmetics and personal care products, plasticizers and sewage and industrial wastewater may be important sources of PAEs in atmospheric particulate matter and that PAEs were preferentially concentrated in finer particles. These conclusions are consistent with the data in Table 4. Although no such data concerning PAEs in indoor or outdoor air and dust is available in China, the potential health hazards associated with the continued rise of the indoor use of PAEs, which may result in higher PAE concentrations in indoor air and dust, require more attention.

The ecological risk of PAEs in Chinese aquatic environments

The risk that PAEs pose to aquatic environments is still unknown. Continuous inputs and intrinsic toxicity are the main parameters that influence their effects on ecosystems. A risk assessment following the recommendation of the Technical Guidance Document on risk assessment (European Commission 2003) has been performed considering the L(E)C_{50} of fish, Daphnia, and algae; this assessment requires at least three trophic levels from the assessed environment. For those compounds, the toxicity data were selected from the review "Aquatic Toxicity of Eighteen Phthalate Esters" (Staples et al. 1997b). RQ values were calculated using the NOEC data along with the lowest L(E)C_{50} and a factor of 1000 (European commission 2003). Table 6 presents the toxicologically relevant concentrations (LC_{50}, EC_{50}, and NOEC) used for the RQ calculations. Tables 7, 8 and 9 present the RQ results for each compound and location. Because the locations that presented no significant, low or significant potential for adverse effects, and expected potential adverse effects were different for these three aquatic populations, three risk tables were used to illustrate all of these results. Among the PAEs, DBP, DEHP, and BBP presented the main contribution to the ecological risk values. The RQs for DMP varied from 0.00 to 2.78 for Bluegill *Lepomis macrochirus*, from 0.00 to 25.1 for *Daphnia magna*, and from 0.00 to 0.660 for *Selenastrum capricornutum*. In contrast, expected potential adverse effects (RQ > 100) for DEP, DBP, BBP, and DEHP were observed in some locations including

Table 7 Fish RQ results for DMP, DEP, DBP, BBP, DHP, and DEHP and the Sum of RQs for each location in China (μg/l)

Location	DMP	DEP	DBP	BBP	DHP	DEHP	Sum	References
China								
Lake and reservoir water								
Beijing-Yaowahu Park Lake	1.22	12.7	61.0	49.4	ND	160	284	Zhong et al. 2010
Beijing-Chaoyang Park Lake	0.144	1.64	12.6	136	ND	27.5	178	Zhong et al. 2010
Beijing-Yiheyuan Lake	0.0582	0.672	7.38	15.0	ND	46.7	69.8	Zhong et al. 2010
Beijing-Honglingjin Park Lake	0.229	2.97	16.2	15.3	ND	31.0	65.7	Zhong et al. 2010
Beijing-Rendinghu Park Lake	ND	ND	4.00	14.5	ND	45.3	63.8	Zhong et al. 2010
Beijing-Lianhuachi Park Lake	0.0183	0.285	10.4	13.1	ND	34.5	58.2	Zhong et al. 2010
Beijing-Longtanhu Park Lake	ND	ND	5.86	15.6	ND	32.2	53.6	Zhong et al. 2010
Beijing-Yuyuantang Park Lake	ND	ND	12.6	13.3	ND	25.7	51.6	Zhong et al. 2010
Suzhou-Taihu Lake	2.78	4.23	20.9	11.7	ND	ND	39.7	Wang et al. 2003
Beijing-Beihai Lake	0.0170	1.23	6.31	ND	ND	26.8	34.4	Zhong et al. 2010
Beijing-Shichahai Lake	ND	ND	8.50	13.3	ND	ND	21.8	Zhong et al. 2010
Taiyuan-Fenhe Reservoir	–	–	17.21	–	–	4.15	21.4	Guo et al. 2002
Beijing-Taorangting Park Lake	0.0458	0.606	ND	13.1	ND	ND	13.7	Zhong et al. 2010
Guangzhou-Urban Lakes	0.000600	0.00910	5.76	ND	ND	0.850	6.62	Zeng et al. 2008a, b
Guanting Reservoir	0.00370	ND	0.726	1.32	ND	0.435	2.49	Zheng et al. 2014
Beijing-Lakes in Summer palace	0.00410	0.0364	0.631	0.0167	ND	1.31	1.99	Zheng et al. 2014
Beijing-Shichahai	0.00530	0.00550	0.157	0.508	ND	1.20	1.87	Zheng et al. 2014
River water								
Jiangshu-Yangtze River	1.63	175	1161	58.3	ND	10.3	1407	He et al. 2011
Wuhan-Yangtze River Low water period	0.00650	ND	84.9	ND	ND	273	359	Wang et al. 2008a, b
Yellow River Tributary-Yiluo	0.0173	0.142	35.7	ND	ND	159	195	Sha et al. 2007
Yellow River Tributary-Mangqin	ND	0.208	61.9	ND	ND	115	177	Sha et al. 2007
Yellow River Main River-Xiaolangdi	ND	0.0976	50.0	ND	ND	120	170	Sha et al. 2007
Songhua River-Jilin	0.0399	1.02	163	ND	ND	ND	164	Wei et al. 2011
Yangtze River Delta	0.00840	0.0521	17.1	0.2	ND	142	159	Zhang et al. 2012
Yellow River Tributary-Luoyang	ND	0.0612	50.0	ND	ND	101	151	Sha et al. 2007
Haihe River-Urban Area	–	–	17.4	–	–	109	126	Chi 2009
Yellow River Tributary-Wenyan Channel	0.0380	0.116	ND	ND	ND	87.4	87.6	Sha et al. 2007
Yellow River Main River-Kaifeng	0.0164	0.268	ND	ND	ND	80.0	80.3	Sha et al. 2007
Yellow River Main River-Zhengzhou	0.00660	0.187	ND	ND	ND	15.0	75.0	Sha et al. 2007
Yellow River Main River-Dongming	0.0110	0.233	ND	ND	ND	70.0	70.2	Sha et al. 2007
Taiwan Rivers	ND	0.303	11.7	ND	ND	46.5	58.5	Yuan et al. 2002
Yellow River Tributary-Xinmang	ND	0.177	22.0	ND	ND	29.3	51.5	Sha et al. 2007
Yellow River Tributary-Mengzhou	0.0373	0.00730	31.0	ND	ND	19.6	50.6	Sha et al. 2007
Qiantangjiang River-Zhejiang	ND	5.19	17.0	ND	ND	10.0	32.2	Zhang et al. 2003
Yellow River-Taiyuan	ND	ND	18.1	ND	ND	3.95	22.0	Guo et al. 2002
Xiangjiang River-Zhuzhou	1.286	2.15	8.12	10.1	ND	ND	21.6	Tang et al. 2010
Yellow River Main River-Jiaodong	0.0164	0.258	ND	ND	ND	16.2	16.5	Sha et al. 2007
Yellow River Main River-Mengjin	0.0161	0.0958	10.2	ND	ND	1.74	12.0	Sha et al. 2007
Wuhan-Yangtze River High water period	0.0193	0.221	0.319	ND	ND	0.140	0.700	Wang et al. 2008a, b
Surface water								
Yangzhou	ND	1.07	3.79	ND	ND	14.9	19.8	Zhang et al. 2003
Shanghai	ND	1.47	3.60	ND	ND	12.7	17.7	Zhang et al. 2003
Estuary/Port/Harbor water								
Yangtze Estuary area	ND	0.594	2.62	ND	ND	6.50	9.71	Liu et al. 2006

Table 7 continued

Location	DMP	DEP	DBP	BBP	DHP	DEHP	Sum	References
Ground water								
Guangzhou-Dongguan	0.000700	0.0303	0.929	0.0278	ND	2.35	3.34	Zhang et al. 2011a, b
Drinking water source								
Yangtze River Delta-Suzhou	0.00390	0.0491	22.1	0.806	ND	4.90	27.9	Shi et al. 2012
Yangtze River Delta-Wuxi	0.00200	0.0376	18.9	0.0667	ND	2.80	21.8	Shi et al. 2012
Yangtze River Delta-Changzhou	0.00370	0.0200	12.9	0.972	ND	4.80	18.7	Shi et al. 2012
Yangtze River Delta-Yancheng	0.00420	0.0394	7.86	0.106	ND	0.700	8.71	Shi et al. 2012
Yangtze River Delta-Xuzhou	ND	ND	0.100	ND	ND	0.0550	0.155	Shi et al. 2012
Wastewater treatment plants								
Beijing-influents	ND	35.9	11.4	32.4	ND	38.0	118	Lin et al. 2004
Beijing-effluents	ND	ND	ND	9.75	ND	12.5	22.3	Lin et al. 2004

Table 8 *Daphnia magna* RQ results for DMP, DEP, DBP, BBP, DHP, and DEHP and the Sum of RQs for each sampled site in River

Location	DMP	DEP	DBP	BBP	DHP	DEHP	Sum	References
China								
Lake and reservoir water								
Beijing-Yaowahu Park Lake	10.9	0.560	15.1	17.8	ND	32.0	76.4	Zhong et al. 2010
Beijing-Chaoyang Park Lake	1.29	0.0720	3.12	49.0	ND	5.50	59.0	Zhong et al. 2010
Suzhou-Taihu Lake	25.1	0.186	5.17	4.22	ND	ND	34.6	Wang et al. 2003
Beijing-Shichahai Lake	0.0476	0.000200	25.8	0.183	ND	0.239	26.2	Zheng et al. 2014
Beijing-Honglingjin Park Lake	2.06	0.131	4.00	5.50	ND	6.20	17.9	Zhong et al. 2010
Beijing-Yiheyuan Lake	0.524	0.0296	1.82	5.40	ND	9.33	17.1	Zhong et al. 2010
Beijing-Rendinghu Park Lake	ND	ND	0.988	5.22	ND	9.06	15.3	Zhong et al. 2010
Beijing-Lianhuachi Park Lake	0.165	0.0125	2.57	4.71	ND	6.89	14.3	Zhong et al. 2010
Beijing-Longtanhu Park Lake	ND	ND	1.45	5.60	ND	6.43	13.5	Zhong et al. 2010
Beijing-Yuyuantang Park Lake	ND	ND	3.12	4.80	ND	5.13	13.0	Zhong et al. 2010
Beijing-Beihai Lake	0.153	0.0101	1.56	ND	ND	5.36	7.08	Zhong et al. 2010
Beijing-Shichahai Lake	ND	ND	2.10	4.79	ND	ND	6.89	Zhong et al. 2010
Beijing-Taorangting Park Lake	0.412	0.0267	ND	4.70	ND	ND	5.14	Zhong et al. 2010
Taiyuan-Fenhe Reservoir	–	–	4.25	–	–	0.830	5.08	Guo et al. 2002
Guangzhou-Urban Lakes	0.00530	0.000400	1.42	ND	ND	0.170	3.02	Zeng et al. 2008a, b
Beijing-Guanting Reservoir	0.0329	ND	0.179	0.476	ND	0.0870	0.775	Zheng et al. 2014
Beijing-Lakes in Summer palace	0.0365	0.000200	0.156	0.00600	ND	0.261	0.460	Zheng et al. 2014
River water								
Jiangshu-Yangtze River	14.7	7.68	287	21.0	ND	2.05	332	He et al. 2011
Wuhan—Yangtze River—Low water period	0.0588	ND	21.0	ND	ND	54.7	75.8	Wang et al. 2008a, b
Yellow River-Tributary-Yiluo	0.155	0.00620	8.82	ND	ND	31.8	40.8	Sha et al. 2007
Songhua River-Jilin	0.359	0.0448	40.4	ND	ND	ND	40.8	Wei et al. 2011
Yellow River-Tributary-Mangqin	ND	0.00910	15.3	ND	ND	23.0	38.3	Sha et al. 2007
Yellow River-Main River-Xiaolangdi	ND	0.00430	12.4	ND	ND	24.0	36.4	Sha et al. 2007
Yangtze River Delta	0.0759	0.00230	4.23	0.0720	ND	28.4	32.8	Zhang et al. 2012
Yellow River-Tributary-Luoyang	ND	0.00270	12.4	ND	ND	20.3	32.7	Sha et al. 2007
Haihe River-Urban Area	–	–	4.31	–	–	21.7	26.0	Chi 2009
Yellow River-Tributary-Wenyan Channel	0.342	0.00510	ND	ND	ND	17.5	17.8	Sha et al. 2007
Yellow River-Main River-Kaifeng	0.148	0.0118	ND	ND	ND	16.0	16.2	Sha et al. 2007
Yellow River-Main River-Zhengzhou	0.0594	0.00820	ND	ND	ND	15.0	15.1	Sha et al. 2007
Yellow River-Main River-Dongming	0.0988	0.0102	ND	ND	ND	14.0	14.1	Sha et al. 2007

Table 8 continued

Location	DMP	DEP	DBP	BBP	DHP	DEHP	Sum	References
Taiwan Rivers	ND	0.0133	2.88	ND	ND	9.30	12.2	Yuan et al. 2002
Yellow River-Tributary-Mengzhou	0.336	0.000300	7.65	ND	ND	3.91	11.9	Sha et al. 2007
Yellow River-Tributary-Xinmang	ND	0.00780	5.44	ND	ND	5.86	11.3	Sha et al. 2007
Xiangjiang River-Zhuzhou	11.6	0.0947	2.01	3.63	ND	ND	11.3	Tang et al. 2010
Qiantangjiang River-Zhejiang	ND	0.228	4.19	ND	ND	2.00	6.42	Zhang et al. 2003
Yellow River-Taiyuan	ND	ND	4.47	ND	ND	0.790	5.26	Guo et al. 2002
Yellow River-Main River-Jiaodong	0.148	0.0113	ND	ND	ND	3.24	3.40	Sha et al. 2007
Yellow River-Main River-Mengjin	0.145	0.00420	2.52	ND	ND	0.347	3.01	Sha et al. 2007
Wuhan-Yangtze River-High water period	0.174	0.00970	0.0788	ND	ND	0.0280	0.290	Wang et al. 2008a, b
Surface water								
Yangzhou	ND	0.0469	0.935	ND	ND	2.98	3.96	Zhang et al. 2003
Shanghai	ND	0.0651	0.888	ND	ND	2.53	3.48	Zhang et al. 2003
Estuary/Port/Harbor water								
Yangtze Estuary area	ND	0.0261	0.647	ND	ND	1.30	1.97	Liu et al. 2006
Ground water								
Guangzhou-Dongguan	0.00590	0.00130	0.229	0.0100	ND	0.470	0.717	Zhang et al. 2011a, b
Drinking water source								
Yangtze River Delta-Suzhou	0.0347	0.00220	5.47	0.290	ND	0.980	6.78	Shi et al. 2012
Yangtze River Delta-Wuxi	0.0176	0.00170	4.66	0.0240	ND	0.560	5.26	Shi et al. 2012
Yangtze River Delta-Changzhou	0.0329	0.000800	3.18	0.350	ND	0.960	4.52	Shi et al. 2012
Yangtze River Delta-Yancheng	0.0376	0.00170	1.94	0.0380	ND	0.140	2.16	Shi et al. 2012
Yangtze River Delta-Xuzhou	ND	ND	0.0247	ND	ND	0.0110	0.0357	Shi et al. 2012
Wastewater treatment plants								
Beijing-influents	ND	1.58	2.81	11.7	ND	7.60	26.5	Lin et al. 2004
Beijing-effluents	ND	ND	ND	3.51	ND	2.50	6.01	Lin et al. 2004

Jiangshu-Yangtze River for *Lepomis macrochirus* populations, Songhua River-Jilin for fish populations, and Beijing-Chaoyang Park Lake for *Selenastrum capricornutum* populations. Generally, algae are especially susceptible to PAEs, while the RQs for invertebrates (*Daphnia magna*) are at least two-fold less. With the exception of DMP, DEP, and DHP, most RQs were in the range of 10.0–100, indicating significant risk related to the current predicted concentrations in aquatic environments. However, DMP, DEP, and DHP were found to pose no or low risk towards fish, invertebrates, and algae by growth inhibition (Staples et al. 1997a, b).

To estimate the joint effects of these PAEs in China, a sum of RQs for each detected compound was calculated for each location. Based on these sums, no significant risk (RQ < 1.00) was observed in Yangtze River Delta-Xuzhou for any of the three populations. The RQ sums varied from 0.160 (Yangtze River Delta-Xuzhou) to 1407 (Jiangshu-Yangtze River) for fish populations, from 0.0400 (Yangtze River Delta-Xuzhou) to 333 (Jiangshu-Yangtze River) for *Daphnia magna* population, and from 0.310 (Yangtze River Delta-Xuzhou) to 2634 (Jiangshu-Yangtze River) for *Selenastrum capricornutum* population. The sums indicate that effects (RQ > 100) are expected in Jiangshu-Yangtze River for all of the three populations.

With the exception of Shichahai, the lakes in Summer Places, and Guanting reservoir, which showed no or low significant adverse effects, most of the urban lakes were observed to have expected or significant potential for adverse effects. Most of the rivers presented significant or expected potential for adverse effects, except for Wuhan-Yangtze River High Water Period, which presented no significant risk. For the other aquatic environments, the influents of wastewater treatment plants in Beijing presented expected potential for adverse effects, while most others presented low to significant potential for adverse effects. Thus, the ecological risk of PAEs in Chinese aquatic environments should be considered, and more short-term and long-term toxicological data on the synergistic effects of PAE mixtures in water at relevant urbanization environmental conditions are needed for a more reliable risk assessment.

Table 9 *Selenastrum capricornutum* RQ results for DMP, DEP, DBP, BBP, DHP, and DEHP and the sum of RQs for each sampled site in river

Location	DMP	DEP	DBP	BBP	DHP	DEHP	Sum of RQs	References
China								
Lake and reservoir water								
Beijing-Yaowahu Park Lake	0.288	5.75	122	178	ND	320	626	Zhong et al. 2010
Beijing-Chaoyang Park Lake	0.0340	0.739	25.2	490	ND	55.0	571	Zhong et al. 2010
Beijing-Yiheyuan Lake	0.0138	0.304	14.8	54.0	ND	93.3	162	Zhong et al. 2010
Beijing-Rendinghu Park Lake	ND	ND	8.00	52.2	ND	90.6	151	Zhong et al. 2010
Beijing-Honglingjin Park Lake	0.0541	1.34	32.4	55.0	ND	62.0	151	Zhong et al. 2010
Beijing-Lianhuachi Park Lake	0.00430	0.129	20.8	47.1	ND	68.9	137	Zhong et al. 2010
Beijing-Longtanhu Park Lake	ND	ND	11.7	56.0	ND	64.3	132	Zhong et al. 2010
Beijing-Yuyuantang Park Lake	ND	ND	25.2	48.0	ND	51.3	125	Zhong et al. 2010
Suzhou-Taihu Lake	0.658	1.912	41.9	42.2	ND	ND	86.6	Wang et al. 2003
Beijing-Beihai Lake	0.00400	0.104	12.6	ND	ND	53.6	66.3	Zhong et al. 2010
Beijing-Shichahai Lake	ND	ND	17.0	47.9	ND	ND	64.9	Zhong et al. 2010
Beijing-Taorangting Park Lake	0.0108	0.274	ND	47.0	ND	ND	47.3	Zhong et al. 2010
Taiyuan-Fenhe Reservoir	–	–	34.4	–	–	8.30	42.7	Guo et al. 2002
Guangzhou-Urban Lakes	0.000100	0.00410	11.5	ND	ND	1.70	13.2	Zeng et al. 2008a, b
Beijing-Guanting Reservoir	0.000900	ND	1.45	4.76	ND	0.870	7.08	Zheng et al. 2014
Beijing-Shichahai Lake	0.00130	0.00250	0.314	1.83	ND	2.39	4.54	Zheng et al. 2014
Beijing-Lakes in Summer palace	0.00100	0.00160	1.26	0.0600	ND	2.61	3.93	Zheng et al. 2014
River water								
Jiangshu-Yangtze River	0.386	78.9	2323	210	ND	20.5	2633	He et al. 2011
Wuhan-Yellow River-Low water period	0.00150	ND	170	ND	ND	547	717	Wang et al. 2008a, b
Yellow River-Tributary-Yiluo	0.00410	0.0641	71.4	ND	ND	318	389	Sha et al. 2007
Yellow River-Tributary-Mangqin	ND	0.0940	124	ND	ND	230	354	Sha et al. 2007
Yellow River-Main River-Xiaolangdi	ND	0.0441	100	ND	ND	240	340	Sha et al. 2007
Songhua River-Jilin	0.00940	0.460	327	ND	ND	ND	327	Wei et al. 2011
Yangtze River Delta	0.00200	0.0236	34.2	0.720	ND	284	319	Zhang et al. 2012
Yellow River-Tributary-Luoyang	ND	0.0277	100	ND	ND	203	303	Sha et al. 2007
Haihe River-Urban Area	–	–	34.9	–	–	217	252	Chi 2009
Yellow River-Tributary-Wenyan Channel	0.00900	0.0526	ND	ND	ND	174	175	Sha et al. 2007
Yellow River-Main River-Kaifeng	0.00390	0.121	ND	ND	ND	160	160	Sha et al. 2007
Yellow River-Main River-Zhengzhou	0.00160	0.0847	ND	ND	ND	150	150	Sha et al. 2007
Yellow River-Main River-Dongming	0.00260	0.105	ND	ND	ND	140	140	Sha et al. 2007
Taiwan Rivers	ND	0.137	23.3	ND	ND	93.0	116	Yuan et al. 2002
Yellow River-Tributary-Xinmang	ND	0.0800	44.0	ND	ND	58.6	103	Sha et al. 2007
Yellow River-Tributary-Mengzhou	0.00880	0.00330	61.9	ND	ND	39.1	101	Sha et al. 2007
Qiantangjiang River-Zhejiang	ND	2.35	34.0	ND	ND	20.0	56.3	Zhang et al. 2003
Xiangjiang River-Zhuzhou	0.304	0.973	16.2	36.3	ND	ND	53.8	Tang et al. 2010
Yellow River-Taiyuan	ND	ND	36.2	ND	ND	7.90	44.1	Guo et al. 2002
Yellow River-Main River-Jiaodong	0.00390	0.116	ND	ND	ND	32.4	32.5	Sha et al. 2007
Yellow River-Main River-Mengjin	0.00380	0.0433	20.4	ND	ND	3.47	23.9	Sha et al. 2007
Wuhan-Yangtze River-High water period	0.00460	0.100	0.638	ND	ND	0.280	1.02	Wang et al. 2008a, b
Surface water								
Yangzhou	ND	0.482	7.57	ND	ND	29.8	37.9	Zhang et al. 2003
Shanghai	ND	0.669	7.19	ND	ND	25.3	33.2	Zhang et al. 2003
Estuary/Port/Harbor water								
Yangtze Estuary area	ND	0.269	5.24	ND	ND	13.0	18.5	Liu et al. 2006

Table 9 continued

Location	DMP	DEP	DBP	BBP	DHP	DEHP	Sum of RQs	References
Ground water								
Guangzhou-Dongguan	0.000200	0.0137	1.86	0.100	ND	4.70	6.67	Zhang et al. 2011a, b
Drinking water source								
Yangtze River Delta-Suzhou	0.000900	0.0222	44.3	2.90	ND	9.80	57.0	Shi et al. 2012
Yangtze River Delta-Wuxi	0.000500	0.0170	37.6	0.240	ND	5.60	43.5	Shi et al. 2012
Yangtze River Delta-Changzhou	0.000900	0.00900	25.7	3.50	ND	9.60	38.8	Shi et al. 2012
Yangtze River Delta-Yancheng	0.00100	0.0178	15.7	0.380	ND	1.40	17.5	Shi et al. 2012
Yangtze River Delta-Xuzhou	ND	ND	0.200	ND	ND	0.110	0.310	Shi et al. 2012
Wastewater treatment plants								
Beijing-influents	ND	16.2	22.7	117	ND	76.0	232	Lin et al. 2004
Beijing-effluents	ND	ND	ND	35.1	ND	25.0	60.1	Lin et al. 2004

Some studies have indicated that PAEs can accumulate in biota (Gorsuch et al. 2008; Cheng et al. 2013; Wang and Zhang 2013). Cheng et al. (2013) measured PAEs in 20 fish species collected from Hong Kong market; the ΣPAEs concentration ranged from 1.66 to 3.14 μg/g wet weight (ww) in fresh water fish and from 1.57 to 7.10 μg/g ww in marine fish. DEHP and DBP were the predominant compounds in both freshwater and marine fish. Mo et al. (2009) measured six PAEs in 11 vegetable species from nine farms of the Pearl River Delta; the total concentrations of PAEs ranged from 0.0700 to 11.2 μg/g dw, with a mean value of 3.20 μg/g (dw). The highest levels of PAEs were found in *Brassica parachinensis*, and the bioconcentration factors of the PAEs ranged from <0.000100 to 0.610. These results indicate that these phthalate esters can accumulate during gastrointestinal digestion in biota.

Conclusions

As one of the world's most high produced and consumed chemicals, PAEs have become an ecological risk through their widespread and continuous exposure via food and drinking water. In general, the sources of PAEs in China were mainly derived from the manufacturing and processing of PAEs-based materials. Due to the demand for PAEs and PAEs-based materials, PAE pollution is predicted to become more serious in the future. The ecological risk assessment was performed based on measurements of PAE concentrations in aquatic environments and toxicity data. However, long term and bioaccumulation studies for these pollutants in aquatic environments are needed to define the environmental stress produced by the high concentrations of PAEs. There is an urgent need to monitor the sources, fates, toxicity, and ecological risk of PAEs in different environmental media in China, especially in highly urbanized areas or areas with PAE-based industries.

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Conflict of interest The authors declare that they have no conflict of interest.

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