



Meta-analysis of environmental contamination by phtalates

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Alexandre Bergé, Mathieu Cladière, Johnny Gasperi, Bruno Tassin, Régis Moilleron, et al..
Meta-analysis of environmental contamination by phtalates. 2013. <hal-00845814>

HAL Id: hal-00845814

<https://hal-enpc.archives-ouvertes.fr/hal-00845814>

Submitted on 17 Jul 2013

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Environmental Science and Pollution Research

Meta-analysis of environmental contamination by phthalates.

--Manuscript Draft--

| | |
|--|--|
| Manuscript Number: | |
| Full Title: | Meta-analysis of environmental contamination by phthalates. |
| Article Type: | Review Article |
| Keywords: | phthalates; DEHP; review; state-of-the-art |
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| Order of Authors Secondary Information: | |
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Meta-analysis of environmental contamination by phthalates.

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Abstract

Introduction: Phthalate Acid Esters (PAE), commonly named Phthalates, are toxics classified as endocrine-disrupting compounds; they are primarily used as additives to improve the flexibility in polyvinyl chloride.

Occurrence: Many studies have reported the occurrence of phthalates in different environmental matrices, however none of these studies has yet establish a complete overview for those compounds in the water cycle within an urban environment. This review summarizes PAE concentrations for all environmental media throughout the water cycle, from atmosphere to receiving waters. Once the occurrences of compounds have been evaluated for each environmental compartment (urban wastewater, wastewater treatment plants, atmosphere and the natural environment), data are reviewed in order to identify the fate of PAE in the environment and establish whether geographical and historical trends exist. Indeed, geographical and historical trends appear between Europe and other countries such as USA/Canada and China, however they remain location-dependent.

Discussion: This study aimed at identifying both the correlations existing between environmental compartments and the processes influencing the fate and transport of these contaminants into the environment. In Europe, the concentrations measured in waterways today represent the background level of contamination, which provides evidence of a past diffuse pollution. In contrast, an increasing trend has actually been observed for developing countries, especially for China.

Keyword: phthalates, DEHP, review, state-of-the-art.

Introduction

Most of studies, published during the last three decades, have reported several categories of man-made chemicals, classified as “endocrine-disrupting

1 compounds” (EDCs). Among the most frequently cited EDCs, phthalates are of
2 particular concern due to their ubiquity and generally higher levels found in
3 environment comparatively to other EDCs (Sanchez-Avila et al. 2009; Fauser et
4 al. 2003; Staples et al. 1997; Fatoki and Mathabatha 2001). The predominant use
5 of Phthalic Acid Esters (PAE) is for improving flexibility in polyvinyl chloride
6 (PVC) (Giam et al. 1984). For instance, the amount of DEHP in PVC depends on
7 plastic composition. Some products can contain up to 50%, but typically there will
8 be approximately 30% DEHP in most PVC products (Ranke 2005). Moreover,
9 their applications extend to industrial and/or domestic sectors, depending on their
10 molecular weight (Table 1). Low molecular weight, especially Dimethyl phthalate
11 (DMP) and Diethyl phthalate (DEP) have therefore been incorporated into
12 cosmetics, fragrance and other personal care-products. Besides, Di-n-butyl
13 phthalate (DnBP) is also used in epoxy resins, cellulose esters and special
14 adhesive formulations. Additionally, high molecular weight and branching alkyl
15 chain PAEs, primarily Butylbenzyl phthalate (BBP), Di-n-octyl phthalate (DnOP)
16 and Di-(2-ethylhexyl) phthalate (DEHP) are being incorporated into food-
17 packaging, building materials, home furnishing, clothing and medical products
18 (Liu et al. 2010; Staples et al. 1997; Jackson and Sutton 2008; Cespedes et al.
19 2004). Despite their various uses, PAE have in recent studies leveled off in
20 American, Canadian and European production, mainly as a result of highly
21 restrictive regulations (EU 2005, 2004, 2007, 2008; CPSI 2008), while worldwide
22 output has been stagnating to 6,000,000 tons in 2006 (Peijnenburg and Struijs
23 2006). This difference may stem from the increase in consumption from emerging
24 nations, such as Brazil, Russia, India and China. The European and North
25 American productions have however evolved because of the replacement of
26 DEHP by heavier and more stable phthalates (Di-iso-nonyl phthalate and Di-iso-
27 decyl phthalate) therefore less subject to release into the environment.

34 Despite their societal and economical benefits, PAE consumption has lead to the
35 release of PAE into the environment, where they are now ubiquitous and can be
36 found in air, water, sediments and soils. This study focuses more specifically on 6
37 most commonly studied congeners of this family (Table 1): Dimethyl phthalate
38 (DMP), Diethyl phthalate (DEP), Di-n-butyl phthalate (DnBP), Butylbenzyl
39 phthalate (BBP), Di-n-octyl phthalate (DnOP) and Di-(2-ethylhexyl) phthalate
40 (DEHP). Most studies in the literature examine the occurrence and behavior of
41 these six compounds in various environmental matrices (surface water,
42 wastewater, atmosphere and treated water) given that these compounds are
43 regularly found in urban and environmental compartments at significant levels
44 (tens or more $\mu\text{g.l}^{-1}$) (Peijnenburg and Struijs 2006; Vethaak et al. 2005; Cespedes
45 et al. 2004; Fauser et al. 2003; Abb et al. 2009).

49 Strategy of this review

54 Most of the literature on PAEs in the environment, as published since 1990
55 (Staples et al. 1997; Vethaak et al. 2005; Deblonde et al. 2011) have been
56 concerned with one or more compartments, primarily downstream compartments
57 such as WWTP effluent and receiving waters, but none of these works has dealt
58 with all environmental compartments, in particular urban wastewater, nor with
59 PAE behavior throughout the whole water cycle. As similarly done for the meta-
60 analysis of environmental contamination by alkylphenols (Bergé et al. 2012a), the
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1 purpose of this review is to collect a substantial dataset for all compartments (Fig.
2 1) from the atmosphere (gaseous phase and rain water) and extending to urban
3 areas (industrial, residential and man-made wastewater, WWTP influent and
4 effluent, WWTP sludge and stormwater) and the natural environment (surface
5 water, sediments and soils). Unfortunately, the set of congeners reported in the
6 various papers has not been consistent and equivalent between compartments.
7 Some papers have reported on as few as one or two congeners (DEHP, DnBP or
8 DEP), while others have reported on four or six (DMP, DEP, DnBP, BBP, DEHP
9 and DnOP). In this review, PAE refers systematically to the sum of these six
10 congeners. From the database we built, statistical calculations have been derived
11 for certain sample sets. The year of sample extraction was not specified in all
12 references, so it was assumed that the samples were extracted on the article
13 publication date. It was also decided that this review focused exclusively on
14 environmental contamination and not on biota or bioaccumulation processes.
15 Moreover, the concentration ranges reported in all tables correspond to minimal
16 and maximal concentrations of each compound for each compartment. The other
17 values represent median concentrations calculated by authors. This review is also
18 aimed at determining whether temporal and/or geographical trends can be drawn
19 in PAE contamination as observed for alkylphenols (Bergé et al. 2012a). To
20 achieve this objective, the data were examined from geographical perspective.
21 Recommendations for further research will also be provided.
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26 **ATMOSPHERE**

27 **Air contamination**

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34 PAE in the atmosphere or in urban areas are generated by various emission
35 sources, including volatilization from materials, industrial processes, waste
36 combustion and wastewater treatment processes (Salapasidou et al. 2011). It has
37 been calculated that approximately 4% of the quantity of DEHP can evaporate
38 from materials such as floorings, rain clothes, toys, soles of shoes (Ranke 2005).
39 DEHP was indeed detected in aerosols emitted from the aeration tank of a WWTP
40 ranging from 71.1 to 228 ng/m³ (Lepri et al. 2000). As shown in Table 2, a small
41 number of air samples (10 references in all) has been reported for PAE. From data
42 collected, PAE contamination globally lies in the 1-50 ng.m⁻³ range for all
43 congeners, except for DEHP presenting higher levels (up to 3,640 ng.m⁻³). The
44 analysis of this database reveals that European samples were consistently above
45 the values reported in the world (USA, China, etc.), and DEHP was the most
46 abundant phthalate in air, with concentrations ranging from 0.08 to 3,640 ng/m³
47 (Müller et al. 2003). Recent studies conducted by Tlili et al. (2010) and Teil et al.
48 (2006), in the Parisian area (France), underscored that phthalates are preferentially
49 associated with the gaseous phase rather than aerosols. For instance, Tlili et al.
50 (2010) reported that between 60% to 70% against 40% to 30%, respectively, of
51 phthalates are associated to gaseous phase. In addition, the same authors reported
52 that the phthalates exhibiting short alkyl chains and high vapor pressures
53 (especially, DMP and DEP) are predominant in the gaseous phase and inversely
54 for the others compounds. Study conducted by Xie et al. (2005) showed that the
55 air-sea exchange was preferentially dominated by the deposition, especially for
56 DnBP (-60 to -686 ng/m²/d) and BBP (-4 to -28 ng/m²/d). Additionally to this
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1 latter point, the fluxes calculated for DEHP (-95 to 686 ng/m²/d) highlighted a
2 more complicated mechanism. Moreover, DnBP and DEHP total (dry and wet)
3 depositions, to the Great Lakes (Canada), provided by Eisenreich et al. (1981)
4 were estimated at 3.7 and 16 tons per year, respectively. This indicates that the
5 atmosphere could be one of the major contamination sources of PAEs. However,
6 according to Staples et al. (1997) and, more recently to Xie et al. (2007),
7 phthalates are subject to photo-degradation, and therefore generally do not persist
8 in the atmosphere. Authors reported photo-degradation half-lives of common
9 phthalates ranging from 0.3 to 15 days. A study led by Salapasidou et al. (2011)
10 underlined that concentrations of DEHP were significantly higher on urban-traffic
11 areas (4.63 to 45.0 ng/m³; median 19.4 ng/m³) than on urban-industrial site (up to
12 6.50 ng/m³; median 2.80 ng/m³) implying generally an input from vehicular
13 emissions. Similarly, Wang et al. (2008) reported that concentrations were about
14 3.5 higher above urban sites than above suburban sites. These authors attributed
15 this difference to both many point-sources and environmental recycling.
16 Additionally, Dargnat (2008) underlined the presence of seasonal variability
17 above the Parisian area, with smaller concentrations for spring and during
18 summer. This was previously observed by Guidotti et al. (2000) in Italy and Teil
19 et al. (2006) in France above the same area. Due to the limited number of studies
20 available, no distinct temporal or geographical trend could be drawn.
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25 **Rain water**

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29 A few measurements of PAE in rain water have been undertaken (see Table 3, 10
30 references in all). Outdoor levels of DEHP lie on the order of 0.02 to 39.0 µg.l⁻¹
31 and 0.03 to 11.0 µg.l⁻¹ for DnBP. At the scale of Europe, data have shown that
32 PAE concentrations in rain water seem to decline, which implies that the
33 environment in European countries is exposed to decreasing contamination. In
34 addition, data show that PAE concentrations in rain water are more important in
35 Northern European countries than Southern European countries, which implies
36 that the fate of PAE in the atmosphere could be governed by atmospheric currents.
37 A similar phenomenon has been reported for alkylphenols (Bergé et al. 2012a).
38 Dargnat (2008) emphasized that passive volatilization from buildings, in urban
39 areas, was the prevailing source of phthalate in rain water. Author therefore
40 concluded that phthalate uses in building materials constitutes a significant source
41 of contamination. Finally, Vethaak et al. (2005), in the Netherlands, reported that
42 most of phthalates present in rain water were at concentrations comparable to
43 those in surface water.
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48 **URBAN AREAS**

49 **Sewer contamination**

50 *Industrial, man-made and residential wastewater*

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60 As reported in Table 4 and Figure 2, only a few measurements of PAE have been
61 performed in wastewater (4-5 references for each type of wastewater). Industrial
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1 DEHP levels were in the 0.01 to 4,400 $\mu\text{g}\cdot\text{l}^{-1}$ range (median: 34.6 $\mu\text{g}\cdot\text{l}^{-1}$), i.e. twice
2 as low as levels in residential wastewater (3.30-160 $\mu\text{g}\cdot\text{l}^{-1}$; median: 61.3 $\mu\text{g}\cdot\text{l}^{-1}$)
3 and in man-made wastewater (0.60-470 $\mu\text{g}\cdot\text{l}^{-1}$; median: 66.0 $\mu\text{g}\cdot\text{l}^{-1}$). In Europe and
4 North America, it has been reported that high DEHP and other phthalate
5 concentrations were measured in industrial wastewater. In the USA, for instance,
6 DEHP levels of 4,400 $\mu\text{g}\cdot\text{l}^{-1}$ were measured in untreated industrial wastewater
7 (Clark et al. 2003). Similarly, Jackson and Sutton (2008) quoted DEHP, DEP and
8 DnBP levels reaching 2,700, 100 and 120 $\mu\text{g}\cdot\text{l}^{-1}$ in industrial laundry and adhesive
9 manufacturer wastewaters. Both studies led by Vethaak et al. (2005) and Sanchez-
10 Avila et al. (2009) showed that levels between 45 and 100 $\mu\text{g}\cdot\text{l}^{-1}$ have also been
11 measured in residential wastewater in the Netherlands and Spain. Some authors
12 mentioned that the high concentrations measured in Maresme wastewater
13 (Catalonia, Spain) and Parisian sewer network (France) would point out that PAE
14 are still being produced and used in industrial, household and agricultural
15 formulations despite the implementation of European regulations (EU 2005, 2004,
16 2007) restricting their use within the EU to levels $< 0.1\%$ (Sanchez-Avila et al.
17 2009; Bergé et al. 2012c).

21 *Wastewater Treatment Plant Influent*

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25 As opposed to industrial, man-made and residential wastewater, a large number of
26 WWTP samples have been analyzed for PAEs (see Table 5, i.e. 13 references).
27 Moreover, according to these concentrations, the PAE median levels reported for
28 WWTP influent were in the same order of magnitude than residential and man-
29 made wastewater but in the lower range of industrial wastewater (PAE: 50.7 $\mu\text{g}\cdot\text{l}^{-1}$
30 in WWTP influent vs 72.6, 42.8 and 139.9 $\mu\text{g}\cdot\text{l}^{-1}$ in these types of wastewater,
31 respectively, Fig. 2) (Jackson and Sutton 2008; Sanchez-Avila et al. 2009; Bergé
32 et al. 2012b). The variations in PAE concentrations among WWTP influent have
33 been further investigated using published data, which have provided sufficient
34 data for geographical areas including Europe and North America (Fig. 3).
35 Surprisingly, no data was available for Chinese plants. This figure also points out
36 the 10th and 90th percentiles (i.e. “the whiskers”) as well as the 25th and 75th
37 percentiles (“boxes”); the medians are also highlighted. PAE median
38 concentrations in samples from North America and Europe were quite similar:
39 49.5 and 51.8 $\mu\text{g}\cdot\text{l}^{-1}$, respectively. It is also interesting to note that European
40 countries display only a few outliers to the high side (see Table 5). In Spain and
41 France, these outlier concentrations typically exceed 160 and 100 $\mu\text{g}\cdot\text{l}^{-1}$, i.e. 2-3
42 times the average level, respectively. These outliers are not yet explained,
43 although the presence of untreated loaded industrial effluent could be
44 hypothesized. Additionally, the database analysis indicates that the PAE
45 concentrations measured in French influent were quite similar to those measured
46 in Europe except in Spain, where DEHP concentrations in influent were
47 significantly lower: i.e. 5 times the average level (Reyes-Contreras et al. 2011;
48 Martin-Ruel et al. 2010; Bergé et al. 2012b). This difference could be due to the
49 lower DEHP consumptions in Spain where this congener was found in lower
50 levels than in other European countries.
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Wastewater Treatment Plant Effluent

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4 Understanding the biodegradation processes of phthalic compounds proves to be a
5 critical factor in predicting the fate of these compounds in the environment. From
6 an overall standpoint, the removal of PAE by conventional activated sludge
7 WWTPs has been well documented (Dargnat et al. 2009; Tan et al. 2007; Barnabé
8 et al. 2008; Marttinen et al. 2003). Most of studies reported removal up to 90%.
9 Additionally, recent studies conducted by Bergé et al. (2012b) and Gasperi et al.
10 (2010) explained that biofiltration coupled to physicochemical lamellar
11 clarification could be a promising alternative to activated sludge tanks for today
12 plant built in large urbanized areas where the building pressure makes available
13 land scarce. A large number of samples from WWTP effluent have been analyzed
14 with respect to PAE (see Table 6). Generally speaking, effluent concentrations lie
15 in the range of 0.02-49.9 $\mu\text{g}\cdot\text{l}^{-1}$ (mean: 0.80 $\mu\text{g}\cdot\text{l}^{-1}$) for DEP and 0.02-69.0 $\mu\text{g}\cdot\text{l}^{-1}$
16 (mean: 2.44 $\mu\text{g}\cdot\text{l}^{-1}$) for DEHP, with PAE ranges extending from 0.07 to 108 $\mu\text{g}\cdot\text{l}^{-1}$.
17 As with WWTP influents, no clear difference has been observed for PAE
18 concentrations in final effluent whatever the location (Fig. 3). Median PAE
19 concentrations in the samples from North America and Europe were quite similar,
20 lying in the 4.85-5.35 $\mu\text{g}\cdot\text{l}^{-1}$ range. As previously mentioned for WWTP influent,
21 European effluent has exhibited a few high outliers. In Spain, these outlier
22 concentrations typically exceeded 50 $\mu\text{g}\cdot\text{l}^{-1}$; i.e. 10 times the average level. The
23 little availability of phthalate data before 2000 does not allow defining a trend of
24 these compounds in the discharges of WWTP. Surprisingly, the PAE median
25 concentration measured in Spanish WWTP effluent exceeded that in WWTP
26 influent (de los Rios et al. 2012; Bizkarguenaga et al. 2012; Sanchez-Avila et al.
27 2009; Sanchez-Avila et al. 2012). However, no explanation could be brought by
28 authors to elucidate this difference, but the WWTP efficiency could however be
29 subject to questioning.
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36 According to our review, PAE can display moderate (50%) to high (94%) removal
37 rates. In addition, a study conducted by Vogelsang et al. (2006) explained that
38 chemical and biological treatment eliminates from 50-60% of phthalates in a
39 Norwegian WWTP. In Spain, Sanchez-Avila et al. (2009) reported PAE removal
40 efficiency of 68%. Finally, a study led by Marttinen et al. (2003) explained that
41 DEHP has been removed from wastewater with an efficiency of 94%. In addition,
42 both studies conducted by Fauser et al. (2003) and in EU (2008) reported similar
43 removal efficiencies for DEHP. Ordinarily, phthalates are removed by different
44 processes occurring within WWTP, including solid settlement, sorption to sludge,
45 volatilization, biodegradation, hydrolysis and/or photolysis (Rogers 1996).
46 Sorption on sludge is considered to be one of the major pathways for PAE
47 removal in WWTP. Moreover, even if DEHP may be considered inherently
48 biodegradable under aerobic conditions (Staples et al. 1997), it was not removed
49 by the biological pathway. A study conducted by Gavala et al. (2004) showed that
50 enzymatic pretreatment increased DEHP biodegradation rate in secondary sludge.
51 Furthermore, it was showed that recirculating sludge and thus retaining specific
52 micro-organisms could stabilize removal capacity up to 86% whereas continuous
53 flow had a varying removal of 77-88% (Oliver et al. 2007, 2005). Alternatively,
54 Vogelsang et al. (2006) and Staples et al. (1997) reported that DEP, a soluble and
55 biodegradable compound under aerobic conditions, has been removed during
56 biological treatment process, although it has not been significantly eliminated by
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1 any chemical or mechanical sedimentation processes. As a general rule, in WWTP
2 effluent, DEHP is expected to be the most abundant congener. Phthalates'
3 distribution is however lightly different, compared to WWTP effluents, with a
4 higher proportion of DnBP and DEP (Fig. 4).
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6 **Wastewater Treatment Plant Sludge**

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9 As mentioned above, sorption on sludge is considered to be one of the major
10 pathways for PAE removal in WWTP (Kinney et al. 2006; Cai et al. 2007;
11 Barnabé et al. 2008; Clara et al. 2010) (Fig. 5). A large number of samples have
12 been analyzed for PAEs (see Table 7, i.e. 30 references). In accordance with its
13 predominance in wastewater, Harrison et al. (2006) and Tan et al. (2007) showed
14 that DEHP was the most abundant phthalate in sludge (Fig. 4). In addition, most
15 studies have reported significant DEHP levels in sludge, i.e. in the range of 150-
16 600 mg/kg.dw. For example, levels up to 3,514 mg/kg.dw in digested sludge from
17 Spain (Abad et al. 2005), 661 mg/kg.dw in digested sludge from Sweden
18 (Sweetman 1994) and 578 mg/kg.dw in digested sludge from USA (Staples et al.
19 1997) have been recorded. Additionally, a recent study conducted by Clara et al.
20 (2010) explained that PAE sorption importance and phthalate removals via sludge
21 increased with the molecular weight, and is therefore depending on the molecular
22 weight (Table 1). As a complement, authors reported that proportional mass
23 fraction removed with the sludge amounts to 3% for DMP, 1% for DEP, 76% for
24 DnBP, 21% for BBP and 78% for DEHP. Generally speaking, sludge contents lie
25 in the range of 0.02-2.00 mg/kg.dw (mean: 0.19 mg/kg.dw) for DMP, 0.01-11.0
26 mg/kg.dw (mean: 0.45 mg/kg.dw) for DEP and 0.32-3,514 mg/kg.dw (mean: 60.3
27 mg/kg.dw) for DEHP. It is interesting to note that certain phthalates display only a
28 few outliers to the high side. For DnBP and BBP, the outlier contents typically
29 exceed 260 and 35 mg/kg.dw, respectively, i.e., 490 and 150 times the average
30 level (0.53 and 0.23 mg/kg.dw, respectively). The little availability of phthalate
31 removal efficiencies in biosolids suggested that heat drying and anaerobic
32 digestion were less effective at reducing phthalates, particularly DEHP, than
33 composting. As a complement, both studies conducted by Gibson et al. (2005) and
34 Williams (2007) reported that composting removes between 64 and 70% of
35 DEHP, respectively. Based on our database, PAE contents are statistically close
36 regardless of the location considered, especially between European and Chinese
37 samples, and moreover DEHP contents have been decreasing in both
38 anaerobically-digested sludge and final sludge.
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47 **Stormwater**

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50 Although it has been established that stormwater is responsible for the spread of
51 pollutants, especially in urban areas, data concerning the emission, occurrence and
52 fate of PAE in stormwater remains poorly reported (Björklund et al. 2009; Rule et
53 al. 2006; Makepeace et al. 1995; Clara et al. 2010; Pitt et al. 1999; Zgheib et al.
54 2012). According to these authors, phthalates in stormwater originated from
55 plasticizers in PVC, paints, building materials, etc. For instance, wastewater from
56 car washes can also make a significant contribution to the emission of DEHP
57 (Vikelsee et al. 1998). Interestingly, PAEs are not chemically bound to the
58 material and may migrate from products during use and disposal (Cadogan et al.
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1993). DEHP has been measured at concentrations between 0.45 and 24.0 $\mu\text{g}\cdot\text{l}^{-1}$ in urban stormwater in Austria (Clara et al. 2010). DEHP was also measured in all urban and suburban samples between 3 and 58 $\mu\text{g}\cdot\text{l}^{-1}$ (Zgheib et al. 2012). Such levels were higher to those previously reported for stormwater in Sweden (5 $\mu\text{g}\cdot\text{l}^{-1}$, (Björklund et al. 2009)) and in London (0.75 to 1.25 $\mu\text{g}\cdot\text{l}^{-1}$, (Rule et al. 2006)). Finally, screening performed in the 1990s revealed higher concentrations of phthalates in stormwater compared to results from studies performed in 2000s, from less than ten times higher for DnOP and DEHP (Makepeace et al. 1995) up to 140 times higher for BBP (Pitt et al. 1999; Björklund et al. 2009). The higher concentrations measured in previous studies may be explained by the difference between phthalate uses a decade ago and the current situation.

NATURAL ENVIRONMENT

Surface water

Surface water is commonly considered as the natural compartment most affected by human pressures, since this water is subjected to the discharges of treated and/or untreated wastewater and/or stormwater (Lin et al. 2009). Therefore, throughout the world, the occurrence and fate of PAE in surface water have been well documented, as demonstrated by the 28 references pertaining to surface water contamination (see Table 8). First, close attention must be paid when comparing concentrations reported in literature, since industrialized and urbanized watershed have been monitored (Vitali et al. 1997; Tan 1995; Yuan et al. 2002; Long et al. 1998; Sha et al. 2007; Zhu and Qiu 2011; Dargnat 2008). Globally and at the world scale, PAE contamination in surface water varies from few $\mu\text{g}\cdot\text{l}^{-1}$ to several tens of $\mu\text{g}\cdot\text{l}^{-1}$. The variations in PAE concentrations in surface water can be examined using published data, which provide sufficient details for determining the statistical distribution of concentrations. Figure 3 summarizes data for three geographical areas, namely Europe, North America and China. The median DEHP concentrations for Europe and China are very similar, i.e. 1.05 and 1.11 $\mu\text{g}\cdot\text{l}^{-1}$, respectively; the median concentration in the North American samples is relatively lower (0.27 $\mu\text{g}\cdot\text{l}^{-1}$). Moreover, median PAE concentrations in the North American samples (0.29 $\mu\text{g}\cdot\text{l}^{-1}$) are notably lower than the Chinese (1.24 $\mu\text{g}\cdot\text{l}^{-1}$) and European samples (1.18 $\mu\text{g}\cdot\text{l}^{-1}$). It is also interesting to note that all geographical areas displayed a few high outliers. In United Kingdom (Long et al. 1998), these outlier concentrations typically exceeded 15 $\mu\text{g}\cdot\text{l}^{-1}$ (i.e., 12 times the average level). In addition, in China (Sha et al. 2007), they were above 28 $\mu\text{g}\cdot\text{l}^{-1}$ (i.e., 20 times the average level).

It is obvious from the database that PAE concentrations in European surface water first increased before exhibiting a decrease due to tighter regulations (EU 2005, 2004, 2007). It would be appropriate to treat these datasets as two separate time series (Fig. 6). The first series contains the samples from European countries before adoption of the Water Framework Directive (Thüren 1986; Ernst et al. 1988; Fatoki and Vernon 1990; Law et al. 1991; Vitali et al. 1997; Long et al. 1998; Belfroid et al. 1999), all of them exhibit PAE concentrations from 0.04 to 15.8 $\mu\text{g}\cdot\text{l}^{-1}$. The second series is composed of samples from European countries after the WFD implementation (Penalver et al. 2001; Fromme et al. 2002; Vethaak

1 et al. 2005; Bendz et al. 2005; Prieto et al. 2007; Sanchez-Avila et al. 2012),
2 revealing lower concentrations, i.e. of about $1.16 \mu\text{g.l}^{-1}$ for the sum of PAE. The
3 PAE concentrations plotted as a function of time for these two series are shown in
4 Figure 6. In contrast, concentrations measured in the Chinese surface water have
5 been increasing for the last decade (see Table 8). This is the result of the constant
6 increase of industrialization, consumptions of phthalates and absence of regulation
7 in China and more generally in developing countries (Zhu and Qiu 2011; Zeng et
8 al. 2008; Sha et al. 2007; Li et al. 2006; Chen et al. 2012).
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10 Most studies focusing on surface water analyze only the dissolved phase, hence
11 only a few results are available for suspended solids (Table 9; four references
12 available). According to these four studies, levels of phthalates in suspended
13 solids range from a few to 630 mg/kg.dw (Table 9). Such contamination levels
14 may be explained by the high hydrophobicity of DEHP and DnBP (with respect to
15 their Log K_{ow} ; see Table 1), which leads to sorption and accumulation of these
16 compounds on suspended solids (Gounaris et al. 1993; Sha et al. 2007). For less
17 hydrophobic compounds, contents are lower, i.e. about between 0.05 and 0.20
18 mg/kg for all compounds, except DEHP (0.70-630 mg/kg). Finally, suspended
19 solids may, however, play a key role in aquatic systems since, under low-flow
20 conditions, particles can settle and contribute to sediment formation, thus yielding
21 contaminant stocks (Fig. 1 and Fig. 5). This phenomenon has been demonstrated
22 for some heavy metals, pesticides and PCBs by Zgheib et al. (2012).
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27 Sediments

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31 Several sediment samples have been collected from rivers and lakes across
32 Europe, North America and developing countries (particularly China and India)
33 (Vitali et al. 1997; Fromme et al. 2002; Yuan et al. 2002; Vethaak et al. 2005; Sha
34 et al. 2007; Zeng et al. 2008; Liu et al. 2010). The amount of DnBP and DEHP
35 measured in these samples (see Table 10; 23 references) varied from 0.01 to 115
36 mg/kg.dw (median, 0.44 and 1.90 mg/kg.dw, respectively). Generally speaking,
37 DMP and DEP were not detected in sediments (Fig. 4), mainly due to their low
38 hydrophobicity. In addition, a recent study led by Liu et al. (2010) reported that
39 DEP is easily degraded in top material and cannot be eluted in deep sediments.
40 Moreover, under anaerobic conditions, DnBP and BBP are easily degraded
41 whereas DEP and DEHP were poorly removed (Yuan et al. 2002). From the
42 exhaustive data in literature, it is clear that PAE are often present in sediment
43 regardless of the location (Fig. 2). The variation in phthalate contents among
44 sediments was examined using published data, which provided sufficient details
45 for determining the statistical distribution of these concentrations. Huang et al.
46 (2005) examined the evolution of DEHP sediment content along a river transects
47 densely populated. They revealed the existence of dynamic processes occurring in
48 surface water, such as sorption to settleable particles and atmospheric
49 volatilization governing the fate of phthalates in the environment and could lead
50 to the formation of sinks in sediments and atmospheric compartment.
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56 A comparison between Tables 9 and 10 indicates that the PAE contents found in
57 sediments are close to those found in suspended solids (Fig. 5). As previously
58 mentioned and highlighted in Figure 1, sediments and suspended solids are linked
59 through sedimentation during low-flow periods and through re-suspension during
60 high-flow periods (Mitsunobu and Takahashi 2006). Presumably, the high K_{ow} of
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1 certain phthalates explains their sorption to particles in water, which in turn settle
2 to form sediment (Staples et al. 1997). Additionally, this phthalate accumulation
3 in sediments is fed and promoted by anthropogenic suspended solids, such as
4 particles released by municipal WWTP and untreated water (Srivastava et al.
5 2009; Huang et al. 2005).

6 7 **Soils**

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10 Several measurements of phthalates in soils have been undertaken (see Table 11,
11 12 references); these have mainly concerned agricultural and urban soils. DEHP
12 and DnBP levels ranged, respectively, from 0.02 to 264 mg/kg.dw (median: 3.33
13 mg/kg.dw) and 0.01 to 30.1 mg/kg.dw (median: 0.96 mg/kg.dw) for both types of
14 soils. A study performed by Michael et al. (1984) highlighted that these two
15 compounds could enter soils via irrigation and pesticide application. In addition,
16 both studies conducted by Wang et al. (2003) and Dolgen et al. (2007) reported
17 that sewage sludge application could also lead to soil contamination,
18 phytotoxicity, and could cause the accumulation of phthalates in the food supply.
19 Some studies have observed high biodegradation rates of DnBP and DEHP in
20 soils (Juneson et al. 2001; DiGennaro et al. 2005). Biodegradation however is not
21 the only pathway for eliminating DEHP and DnBP in soils. Other processes, such
22 as hydrolysis and photolysis may affect DEHP and DnBP concentrations and
23 occurrence rates. These processes however are recognized to be less important
24 than biodegradation (Yan et al. 1995). A study led by Chang et al. (2004)
25 explained that optimal PAE degradation is enhanced when DnBP and DEHP are
26 present simultaneously. It may be due to the large carbon source and energy
27 provided by both compounds. In addition, in specific physicochemical conditions,
28 DEP biodegradation can lead to the formation of DMP and its monoester
29 (Cartwright et al. 2000). Interestingly and according to the available literature on
30 soils (Table 11), similar contents for agricultural and urban soils have been
31 observed. This homogeneity is quite surprising but could be explained by a global
32 contamination through atmospheric deposits (Fig. 1). Nonetheless, a direct
33 comparison of soil contamination is rather difficult and subtle, since various
34 processes may be occurring. The differences or homogeneity observed can in fact
35 reflect different inputs, environmental factors, including oxygen availability and
36 nutrient amounts in soil, and sources or either different dynamic pollutants in the
37 soils (Semple et al. 2001; Namkoong et al. 2002).

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40 Compost particle sizes have been investigated by several studies (Amellal et al.
41 2001; Delhomenie et al. 2002; Chang et al. 2009). Authors explained that the
42 smaller the particle was, the higher the specific surface was and the lower the
43 porosity was. This favored the microorganism settlement and therefore promoted
44 microbial degradation activity. Finally, the use of straw compost in remediating
45 contaminated soil by phthalates is therefore effective to solid wastes recycling.
46 The variation in PAE contents among soils was examined using published data,
47 which provided sufficient details for determining the statistical distribution of
48 these contents. Figure 3 shows the data from Europe and China. Soils in China
49 (3.09 mg/kg.dw) seem to be slightly more contaminated than European soils (0.17
50 mg/kg.dw). This difference may be linked to several factors: i) greater use or
51 consumption of phthalate-containing goods in China; ii) a higher contamination of
52 air in China, and iii) an absence of pertinent regulations on sludge amendment.
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DISCUSSION AND RECOMMENDATIONS FOR FURTHER STUDY

It is now obvious from the concentrations and contents measured in the different environmental compartments that phthalates are ubiquitous environmental contaminants. Their concentrations and trends in most environmental compartments however remain location-dependent. This review has highlighted some geographical disparities, especially between Europe, North America and China. In Europe, historical trends were also highlighted. In pursuit of the WFD implementation, more specific data for most congeners are needed in upstream compartments to enable source elucidation and to elaborate potential source control action. It is obviously important to measure all industrially-significant phthalates across all samples. It is also important to monitor chemicals used to replace phthalates after the implementations of restrictive regulations worldwide, such as Di-iso-nonyl phthalate (DiNP) and Di-iso-decyl phthalate (DiDP). Finally, phthalate metabolites have to be investigated, which are supposed to be the active molecule of the estrogenic effect (Rais-Bahrami et al. 2004).

As mentioned above, DEHP is always present in all environmental compartments. It is moreover the abundant phthalate in all matrices, particularly in solid matrices, such as sludge and suspended solids (Fig. 4). The result is that what is left over from many industrial and commercial products of this compound leads to the contamination of the wastewater of urban areas.

In Europe, regulations banning products containing certain phthalates (DEHP, DnBP and BBP) have been implemented since 2000. It is therefore relevant to evaluate the effectiveness of these regulations by means of reliable and accurate measurements of the PAE contamination levels for sediments, soils and surface water as a function of time. Indeed, each day without consistent restrictions means that phthalates are produced in huge quantities and it can subsequently result on a widespread use in a lot of products leading to uncontrolled environmental discharges and dissemination. Recent observations in Europe have however shown that PAE concentrations have not been increasing. Moreover, a decreasing trend has actually been observed just over the last few years. Nevertheless, it is important to track these changes closely. As previously noted however, sediments and atmosphere behave like major sinks for PAE in the environment. The role of atmosphere has to be better understood since atmosphere seems to play a key role in PAE widespread. Recent measurements in China, whatever the media, have indicated increasing phthalate contents and concentrations. In addition, the environmental data exhibit a large spatial variability, reflecting differences in phthalate contaminations in various areas of China, which might be caused by uneven development of heavy industry, the imbalance of enforcement or the specific geographical location. Finally, even if trends show decreases across industrialized countries, especially in Europe and North America, the levels being recovered are still significant (up to $100 \mu\text{g}\cdot\text{l}^{-1}$) and still contribute to build the background level. A similar trend has been observed for alkylphenols (Bergé et al. 2012a). In addition, and based on the data collected, there are a few samples with very high levels as compared to the average (see the high outliers in all environmental media). A closer attention needs to be paid to these data – why are

1 these few samples so highly contaminated? Is it related to industrial activities or
2 to unknown diffuse sources? Understanding these outliers may contribute to
3 elucidate the mechanisms by which these samples have become contaminated.

4 The mechanisms involving these contaminants between the environmental
5 compartments cannot be easily distinguished; possibilities include:
6 biodegradation, volatilization, sorption to biomass, and particle sedimentation.
7 The three latter mechanisms may be important in explaining the distribution
8 changes throughout the whole water cycle from wastewater to the natural
9 environment. This change is featured by an increase of DnBP and a decrease of
10 DEHP and DEP (Fig. 4). Research on these mechanisms, particularly the
11 processes acting in WWTP, is required in order to improve better understanding
12 and to quantify what proportion is being volatilized into the atmospheric
13 compartment and how much is adsorbed to biomass. The latest research on
14 WWTP has shown high efficiency of the processes acting on sewage treatment
15 plants as well as an insignificant contribution from discharges (Dagnat et al.
16 2009; Bergé et al. 2012b; Martin-Ruel et al. 2010). Additionally, latest research
17 showed that DEHP removal was dependent on various parameters, but that the
18 type of biomass could significantly enhance removal (Oliver et al. 2007, 2005).
19 The removal of phthalates from wastewater can, however, be improved by adding
20 tertiary treatment such as nanofiltration, reverse osmosis, ozone oxidation, UV
21 irradiation or activated carbon filters to existing processes, in spite of the high cost
22 of these processes (Bodzek et al. 2004; Agenson et al. 2003; Verliefde et al.
23 2007; Oh et al. 2006).

24 A serious element dictating the fate of micro-pollutants in urban areas is their
25 release into the environment during storms. During wet-weather periods, the
26 operations of wastewater treatment plants are modified and may influence the
27 quality of effluent discharged into the environment (Bergé et al. 2012a; Gilbert et
28 al. 2011; Bergé et al. 2012b). Additionally, combined sewer overflow discharges
29 can happen (Zgheib et al. 2012; Gasperi et al. 2008). During wet-weather events,
30 the re-suspension of in-sewer deposits provides for additional pollution at the
31 WWTP. This additional contamination can then increase the load of phthalates
32 and disturb usual plant operations and, in return, efficiency. Finally, nowadays, no
33 information is available to evaluate the runoff contribution to wastewater.

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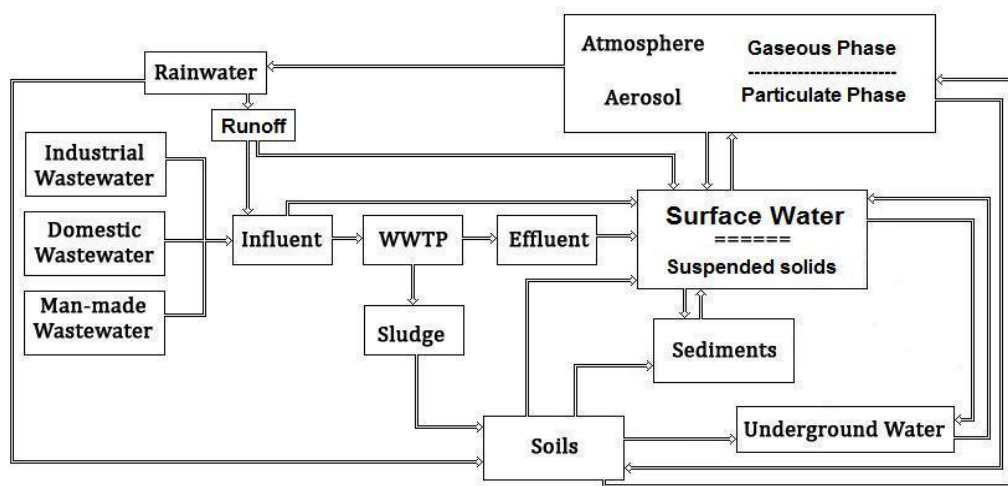


Figure 1: Interactions between the various environmental compartments – Cited from Bergé et al. (2012a)

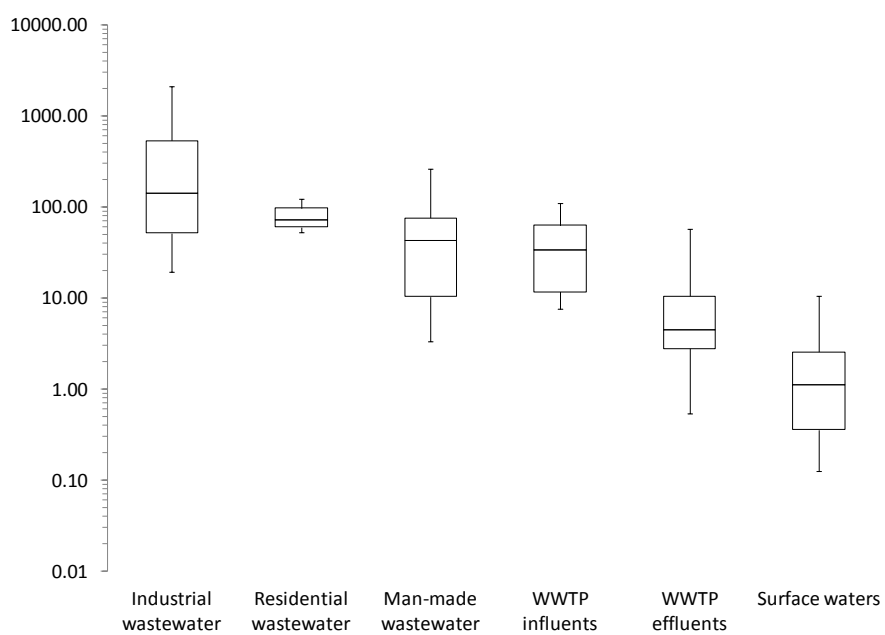


Figure 2: Total PAE (DMP + DEP + DnBP + BBP + DEHP + DnOP) concentrations (in $\mu\text{g.l}^{-1}$) in liquid matrices (WWTP: Wastewater Treatment Plant - global scale)

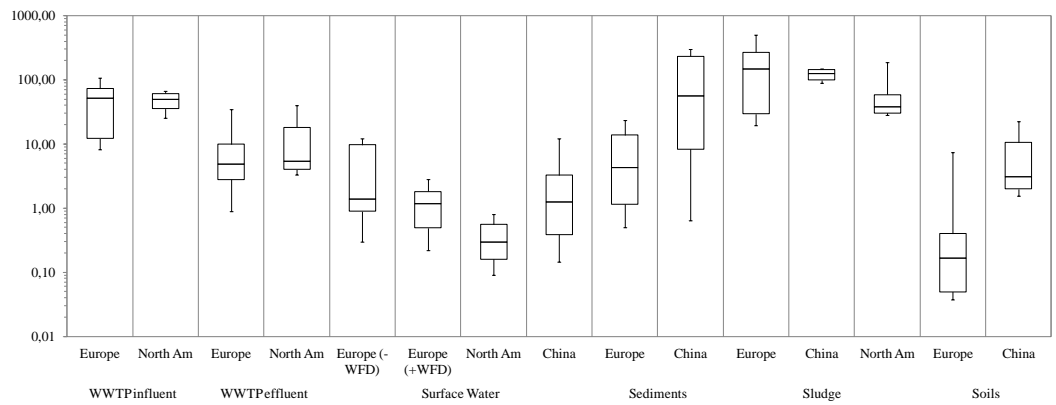


Figure 3: Distributions of total PAE (DMP + DEP + DnBP + BBP + DEHP + DnOP) concentrations ($\mu\text{g.l}^{-1}$) and contents (mg/kg.dw) in WWTP influent and effluent, surface water, sediments, sludge and soils from various locations (WFD: Water Framework Directive; North Am: North America)

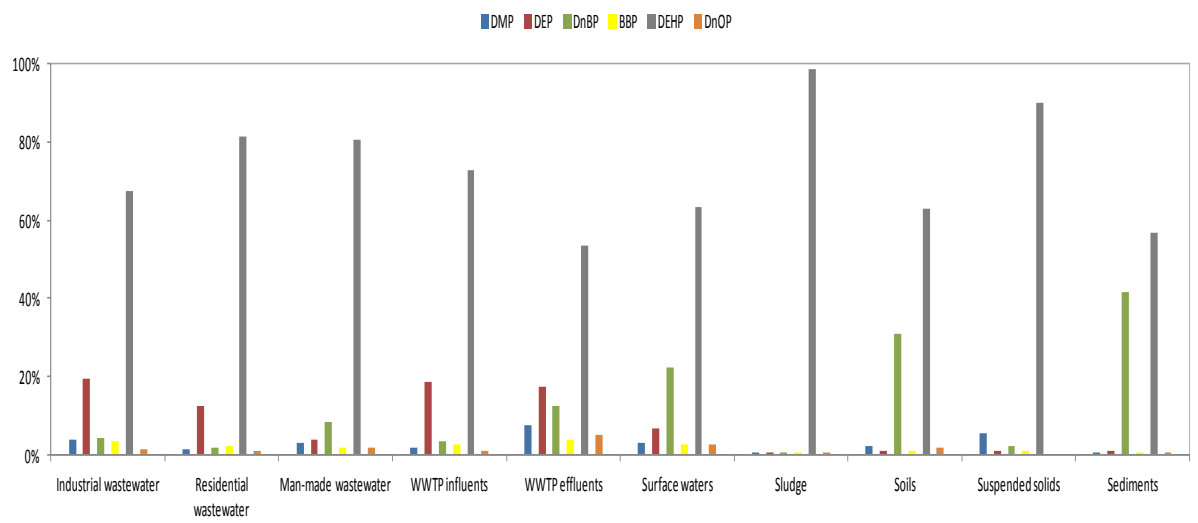


Figure 4: Evolution in the distribution of DMP, DEP, DnBP, BBP, DEHP and DnOP through the environmental cycle (WWTP: Wastewater Treatment Plant)

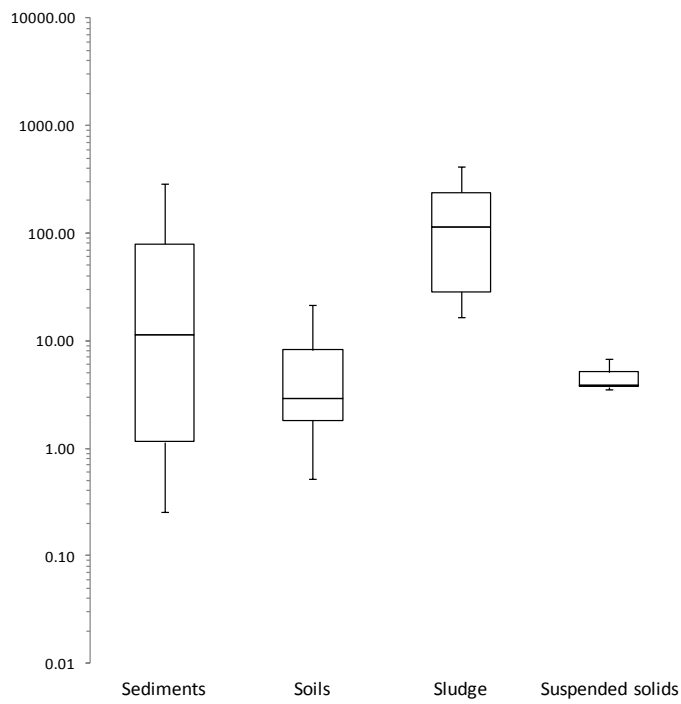


Figure 5: Total PAE (DMP + DEP + DnBP + BBP + DEHP + DnOP) contents (in mg/kg.dw) in solids matrices - global scale

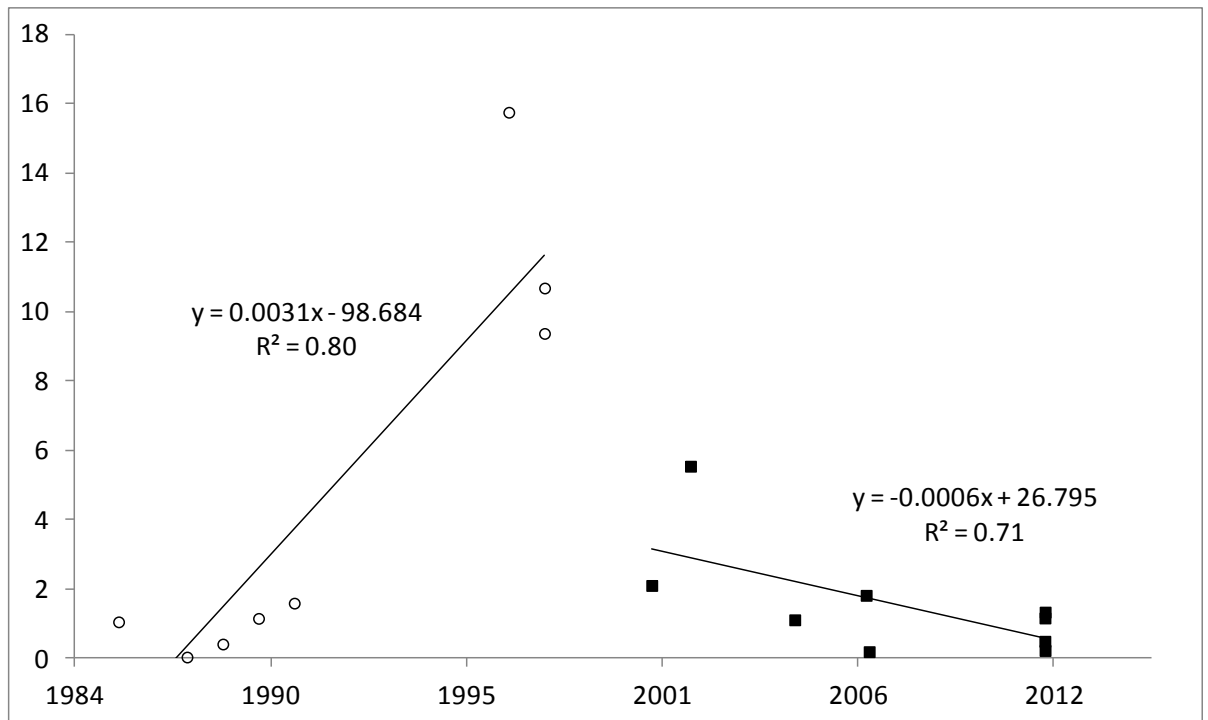


Figure 6: Historical trend lines for the median total PAE (DMP + DEP + DnBP + BBP + DEHP + DnOP) concentrations (in $\mu\text{g.l}^{-1}$) in European surface water. The dots represent samples collected before implementation of the Water Framework Directive, whereas the squares depict samples collected after its implementation

Table 1: Physicochemical properties of PAE

| Compound | Formula | MW (g/mol) | Water solubility at 20 °C (mg/l) | Log K _{ow} | Log K _{oc} | Log K _d | H (Atm.m ³ /mol) |
|--|--|--------------------|---|---|---|--|--|
| DMP | C ₁₀ H ₁₀ O ₄ | 194.2 ^c | 4,20 ^c | 1.61 ^c | 1.74 ^c 1.90- 2.56 ^b | | 1.22 E ^{-07c} |
| DEP | C ₁₂ H ₁₄ O ₄ | 222.2 ^c | 1,10 ^c | 2.42 ^c 2.47- 2.51 ^b | 1.84 ^c 2.85- 3.24 ^b | | 2.66 E ^{-07c} 7.80 E ^{-07b} |
| DnBP | C ₁₆ H ₂₂ O ₄ | 278.4 ^c | 11.2 ^c | 4.57 ^c | 3.14 ^c 4.17 ^b | | 8.83 E ^{-07c} |
| BBP | C ₁₉ H ₂₀ O ₄ | 312.4 ^c | 2.70 ^c | 4.84 ^c | 4.23 ^c 3.95 ^b | 3.55 ^c | 7.61 E ^{-07c} |
| DEHP | C ₂₄ H ₃₈ O ₄ | 390.6 ^c | 0.003 ^c | 7.50 ^c | | 4.12 ^c 4.18 ^a | 1.71 E ^{-05c} |
| DnOP | C ₂₄ H ₃₈ O ₄ | 390.6 ^c | 0.0005 ^c | 8.06 ^c | 4.94 ^c 5.68 ^b 5.71 ^a | 4.46 ^c | 1.03 E ^{-04c} |
| a: (Lützhof et al. 2008); b: (IPCS 2003); c: (Staples et al. 1997) | | | | | | | |

Table 2: Atmospheric contamination by PAE (in ng/m³)

| Location | DMP | DEP | DnBP | BBP | DEHP | DnOP | Reference |
|---------------|-----------|-----------|------------|-----------|------------|-----------|--------------------------------|
| USA | | | 0.60-5.00 | | | | (Atlas and Giam 1988) |
| Sweden | | | 0.23-49.9 | | 0.28-77.7 | | (Thüren and Larsson 1990) |
| China | | 56.0 | 32.0 | 1,910 | 550 | | (Wang et al. 2002) |
| China | | | 224 | | 56.0 | | (Wang et al. 2002) |
| USA | | | | | 5.00-132 | | (Clark et al. 2003) |
| Denmark | | | 1.50-2,480 | | 5.30-3,640 | | (Müller et al. 2003) |
| France | 0.10-21.2 | 1.70-24.6 | 2.90-59.3 | 0.50-12.2 | 3.40-25.7 | <log-1.10 | (Teil et al. 2006) |
| Netherlands | | | 2.00-70.0 | | 0.70-333 | | (Peijnenburg and Struijs 2006) |
| Norwegian Sea | 0.01-0.22 | 0.18-0.90 | 0.16-0.43 | 0.02-0.07 | 0.08-0.46 | | (Xie et al. 2007) |
| France | 1.11 | 3.53 | 1.09 | 0.21 | 1.66 | 0.08 | (Tlili et al. 2010) |
| Greece | | | 1.20-3.36 | 0.11-0.80 | <log-6.50 | <log-0.11 | (Salapavidou et al. 2011) |
| Greece | | | 0.43-2.40 | 0.04-0.98 | 4.63-45.0 | | (Salapavidou et al. 2011) |
| Min | 0.01 | 0.18 | 0.16 | 0.02 | 0.08 | 0.08 | n = 10 |
| Max | 21.2 | 56.0 | 2,480 | 1,910 | 3,640 | 1.10 | |
| Med | 2.93 | 8.70 | 2.50 | 0.48 | 19.7 | 0.24 | |

Table 3: Rain water concentrations of PAE (in $\mu\text{g.l}^{-1}$)

| Location | DMP | DEP | DnBP | BBP | DEHP | DnOP | Reference |
|-------------|-----------|-----------|-----------|-----------|-----------|-----------|----------------------------|
| USA | | 0.02-0.10 | 0.03-0.06 | 0.02-0.07 | 0.02-0.10 | | (Ligocki et al. 1985) |
| Sweden | | | <log-0.50 | | 0.01-0.43 | | (Thüren and Larsson 1990) |
| Canada | | | 0.50-11.0 | | 7.00-39.0 | | (Makepeace et al. 1995) |
| Denmark | | | 1.30 | | 32.0 | | (Kjohlholt et al. 1997) |
| Denmark | | | <log-1.60 | | 1.30-30.0 | | (Boutrup and Plesner 2001) |
| Netherlands | 0.01-0.02 | 0.24-0.43 | 0.28-0.88 | 0.14-0.26 | 0.69-1.70 | 0.04-0.25 | (Vethaak et al. 2005) |
| France | 0.12 | 0.33 | 0.59 | 0.08 | 0.42 | 0.01 | (Teil et al. 2006) |
| France | 0.07-0.11 | 0.14-0.25 | 0.10-0.16 | 0.02-0.06 | 0.36-0.85 | <log-0.03 | (Dargnat 2008) |
| Norway | | | | | 5.00 | | (Björklund et al. 2009) |
| Min | 0.01 | 0.02 | 0.03 | 0.02 | 0.02 | 0.01 | n = 9 |
| Max | 0.12 | 0.43 | 11.0 | 0.26 | 39.0 | 0.25 | |
| Med | 0.06 | 0.33 | 0.61 | 0.16 | 0.77 | 0.05 | |

Table 4: Industrial, man-made and residential wastewater concentrations (in $\mu\text{g.l}^{-1}$)

| Location | Type | DMP | DEP | DnBP | BBP | DEHP | DnOP | Reference |
|------------------------|------------------------|-----------|-----------|-----------|-----------|------------|-----------|-----------------------------|
| USA | Textile | | 3.20 | | | | | (IPCS 2003) |
| France | Textile | | | | | 282 | | (INERIS 2007) |
| France | Laundry | | | | | 470 | | (INERIS 2007) |
| USA | Diaper service | | <loq-1.00 | 12.0 | 0.20 | 0.63 | <loq-0.15 | (Jackson and Sutton 2008) |
| USA | Pet wash | | 1.30 | 0.76 | 2.30 | 6.50 | 1.60 | (Jackson and Sutton 2008) |
| USA | Veterinary clinic | | <loq-51.0 | <loq-18.0 | <loq-7.10 | <loq-11.0 | <loq-7.70 | (Jackson and Sutton 2008) |
| USA | Hospital | | <loq-1.00 | <loq-0.36 | 0.82 | 2.70 | <loq-0.15 | (Jackson and Sutton 2008) |
| USA | Medical clinic | | <loq-0.98 | 0.66 | 0.74 | 1.00 | <loq-0.15 | (Jackson and Sutton 2008) |
| Spain | Industrial-Residential | 0.28-4.94 | 27.5-192 | <loq-33.6 | <loq-5.70 | 9.88-287 | | (Sanchez-Avila et al. 2009) |
| Netherlands | Residential | 0.39-6.20 | 4.10-44.0 | 0.38-51.0 | 0.56-4.90 | 13.0-100 | 0.26-2.40 | (Vethaak et al. 2005) |
| USA | Residential | | 4.00 | <loq-0.36 | 0.76 | 9.10 | 0.60 | (Jackson and Sutton 2008) |
| USA | Residential | | 9.10 | <loq-0.34 | 1.00 | 3.30 | <loq-0.14 | (Jackson and Sutton 2008) |
| Spain | Residential | 0.45-1.12 | 44.0-45.9 | | 0.23-0.29 | 19.5-20.2 | | (Sanchez-Avila et al. 2009) |
| France | Sewer network | | 5.23-17.7 | 0.60-3.31 | 0.46-3.91 | 39.48-161 | | (Bergé et al. 2012c) |
| USA | Industrial activities | | | | | 0.01-4,400 | | (Clark et al. 2003) |
| USA | Manufactured factory | | 60.0 | | | | | (IPCS 2003) |
| Netherlands | Industrial effluents | <loq-1.30 | 0.35-5.20 | 0.69-21.0 | 0.17-1.30 | 1.00-150 | 0.01-2.80 | (Vethaak et al. 2005) |
| France | Paper pulp | | | | | 61.0 | | (INERIS 2007) |
| France | Dyeing | | | | | 41.0 | | (INERIS 2007) |
| France | Leather | | | | | 68.0 | | (INERIS 2007) |
| France | Painting | | | | | 1,300 | | (INERIS 2007) |
| France | Chemistry | | | | | 280 | | (INERIS 2007) |
| USA | Plastic manufacturer | | <loq-1.00 | 0.36 | 2.30 | 49.0 | 10.0 | (Jackson and Sutton 2008) |
| USA | Paper manufacturer | | <loq-1.00 | <loq-0.36 | 0.14 | 6.80 | <loq-0.15 | (Jackson and Sutton 2008) |
| USA | Beverage manufacturer | | <loq-20.0 | <loq-7.1 | <loq-2.70 | <loq-4.10 | <loq-2.90 | (Jackson and Sutton 2008) |
| USA | Adhesive manufacturer | | <loq-100 | 120 | 39.0 | 47.0 | <loq-15.0 | (Jackson and Sutton 2008) |
| USA | Industrial laundry | | <loq-24.0 | 86.0 | 95.0 | 2,700 | | (Jackson and Sutton 2008) |
| USA | Residential laundry | | 16.0 | <loq-3.6 | <loq-1.40 | 66.0 | 13.0 | (Jackson and Sutton 2008) |
| Spain | Industrial effluents | 0.15-7.05 | 22.2-53.6 | | <loq-10.0 | 8.00-36.4 | | (Sanchez-Avila et al. 2009) |
| Man-made wastewater | Min | 0.28 | 1.30 | 0.76 | 0.20 | 0.63 | 0.15 | n = 4 |
| | Max | 4.94 | 192 | 33.6 | 5.70 | 470 | 7.70 | |
| | Med | 2.61 | 3.20 | 6.90 | 1.50 | 66.0 | 1.60 | |
| Residential wastewater | Min | 1.00 | 0.35 | 0.34 | 0.23 | 3.30 | 0.07 | n = 4 |
| | Max | 1.12 | 45.0 | 51.0 | 4.90 | 161 | 2.40 | |
| | Med | 1.06 | 9.48 | 1.29 | 1.57 | 61.3 | 0.60 | |
| Industrial | Min | 0.15 | 0.35 | 0.36 | 0.14 | 0.01 | 0.01 | n = 5 |
| | Max | 7.05 | 100 | 120 | 95.0 | 4,400 | 15.0 | |

| | | | | | | | | |
|------------|-----|------|------|------|------|------|------|--|
| wastewtaer | Med | 1.96 | 10.0 | 2.20 | 1.83 | 34.6 | 0.80 | |
|------------|-----|------|------|------|------|------|------|--|

Table 5: WWTP influent concentrations of PAE (in $\mu\text{g.l}^{-1}$)

| Location | DMP | DEP | DnBP | BBP | DEHP | DnOP | Reference |
|-----------|-----------|-----------|-----------|-----------|-----------|-----------|-------------------------------|
| Finland | <log-1.00 | 6.00-11.0 | 6.00-8.00 | 3.00-5.00 | 28.0-66.0 | <log-1.00 | (Marttinen et al. 2003) |
| Finland | <log-1.00 | 5.00-74.0 | 3.00-9.00 | <log-3.00 | 28.0-122 | <log-2.00 | (Marttinen et al. 2003) |
| Finland | | <log-10.0 | 4.00-6.00 | <log-2.00 | 83.0 | <log-1.00 | (Marttinen et al. 2003) |
| Denmark | | | <log-1.03 | 0.05-0.97 | 13.1-44.3 | 0.22-0.79 | (Fauser et al. 2003) |
| Sweden | | 0.19 | 0.15 | | 0.27 | | (Bendz et al. 2005) |
| Australia | | 0.39 | 0.14 | 0.06 | 20.3 | | (Tan et al. 2007) |
| France | | | | | 55.0 | | (INERIS 2007) |
| Spain | | 0.68 | 0.62 | | 1.90 | | (Regueiro et al. 2008) |
| USA | | <log-10.0 | <log-3.60 | 14.0 | 33.0 | 4.20 | (Jackson and Sutton 2008) |
| USA | | <log-10.0 | <log-3.60 | 1.90 | 9.20 | <log-1.5 | (Jackson and Sutton 2008) |
| Denmark | | | | | 8.10-31.0 | | (Seriki et al. 2008) |
| Canada | | | | | 70.0 | | (Barnabé et al. 2008) |
| Canada | | | | | 41.0 | | (Barnabé et al. 2008) |
| France | 0.82 | 7.71 | 1.10 | 1.12 | 22.5 | 0.10 | (Dargnat et al. 2009) |
| Spain | 0.60 | 50.7 | 46.8 | 0.67 | 47.9 | | (Sanchez-Avila et al. 2009) |
| Austria | <log-2.40 | 0.77-9.20 | <log-8.70 | 0.31-3.20 | 3.40-34.0 | <log-1.10 | (Clara et al. 2010) |
| Austria | 0.26-0.41 | 1.20-2.00 | <log-0.47 | <log-0.11 | 4.40-8.80 | | (Clara et al. 2010) |
| Austria | 0.43-0.81 | 2.20-2.70 | 0.15-0.41 | 0.11-0.26 | 4.10-13.0 | <log-0.10 | (Clara et al. 2010) |
| France | | | | | 52.8 | | (Martin-Ruel et al. 2010) |
| Spain | | 1.05-2.59 | 0.20-0.55 | | 4.23-4.65 | | (Reyes-Contreras et al. 2011) |
| Spain | | 1.90-3.98 | 0.12-0.20 | | 7.50-9.91 | | (Reyes-Contreras et al. 2011) |
| Spain | <log-7.47 | 2.11-5.76 | 0.23-1.99 | 0.25-0.31 | | 0.24-5.91 | (Bizkarguenaga et al. 2012) |
| Spain | 0.31-9.13 | 0.14-5.91 | <log-0.76 | | | 0.27 | (Bizkarguenaga et al. 2012) |
| Spain | 0.36-93.3 | 0.12-4.34 | 0.59-2.24 | | | 0.22-0.25 | (Bizkarguenaga et al. 2012) |
| Spain | 3.94-10.6 | <log-0.96 | 0.23-1.54 | | | <log-3.54 | (Bizkarguenaga et al. 2012) |
| France | | 7.00-36.0 | 1.86-6.01 | 0.97-2.29 | 32.4-71.9 | | (Bergé et al. 2012b) |
| Min | 0.26 | 0.19 | 0.14 | 0.06 | 0.27 | 0.10 | n = 16 |
| Max | 93.3 | 74.0 | 46.8 | 14.0 | 122 | 5.91 | |
| Med | 0.89 | 9.81 | 1.86 | 1.29 | 38.10 | 0.50 | |

Table 6: WWTP effluent concentrations of PAE (in $\mu\text{g.l}^{-1}$)

| Location | DMP | DEP | DnBP | BBP | DEHP | DnOP | Reference |
|-------------|-----------|-----------|-----------|-----------|-----------|-----------|-----------------------------|
| Finland | | | 2.00 | | 2.00 | <loq-1.00 | (Martinen et al. 2003) |
| Finland | <loq-1.00 | <loq-4.00 | <loq-6.00 | <loq-1.00 | 2.00-8.00 | | (Martinen et al. 2003) |
| Finland | | | | | 4.00 | | (Martinen et al. 2003) |
| Denmark | | | 0.18-2.50 | 0.06-0.27 | 0.11-2.65 | 0.01-0.03 | (Fauser et al. 2003) |
| Netherlands | <loq-0.32 | 0.30-0.93 | 0.42-0.84 | 0.07-0.29 | 0.47-2.40 | <loq-0.02 | (Vethaak et al. 2005) |
| Sweden | | 0.02 | 0.03 | | 0.02 | | (Bendz et al. 2005) |
| Norway | | 7.90-9.90 | | | 0.50 | | (Vogelsang et al. 2006) |
| Australia | | 0.02 | 0.05 | 0.02 | 0.36 | | (Tan et al. 2007) |
| France | | 0.43 | | | 0.55 | | (INERIS 2007) |
| Spain | | <loq-1.0 | <loq-0.36 | 0.84 | 2.90 | | (Regueiro et al. 2008) |
| USA | | <loq-1.0 | 0.57 | 0.74 | 1.00 | | (Jackson and Sutton 2008) |
| USA | | <loq-1.0 | 5.50 | <loq-0.14 | 0.21 | | (Jackson and Sutton 2008) |
| Denmark | | | | | 0.30-6.10 | | (Seriki et al. 2008) |
| Canada | | | | | 54.0 | | (Barnabé et al. 2008) |
| France | 0.08 | 0.78 | 0.15 | 0.30 | 5.42 | 0.02 | (Dargnat et al. 2009) |
| Spain | 0.13 | 49.8 | | 0.01 | 9.43 | | (Sanchez-Avila et al. 2009) |
| Austria | <loq-0.19 | <loq-1.10 | <loq-2.40 | 0.09-1.40 | 0.08-6.60 | <loq-0.26 | (Clara et al. 2010) |
| Austria | | <loq-0.10 | | | <loq-0.28 | | (Clara et al. 2010) |
| Austria | | | | | <loq-1.30 | | (Clara et al. 2010) |
| France | | | | | 4.20 | | (Martin-Ruel et al. 2010) |
| Spain | <loq-7.18 | 0.23-0.40 | 0.22-7.92 | | 0.39-0.41 | 0.21-30.8 | (Bizkarguenaga et al. 2012) |
| Spain | 6.25-11.7 | 0.20-1.02 | <loq-10.9 | | | 0.22-0.27 | (Bizkarguenaga et al. 2012) |
| Spain | 2.07-11.2 | 0.36-1.06 | 0.21-1.34 | | 0.31 | 0.23-0.29 | (Bizkarguenaga et al. 2012) |
| Spain | 7.97-13.4 | 0.15-0.51 | 0.20-1.90 | 1.14 | | 3.26-9.06 | (Bizkarguenaga et al. 2012) |
| Austria | | | | | <loq-6.60 | | (Clara et al. 2012) |
| Spain | | 15.7-40.9 | 3.33-58.9 | | 4.17-69.0 | | (de los Rios et al. 2012) |
| Spain | | 0.03-3.21 | | | 1.72-9.22 | | (Sanchez-Avila et al. 2012) |
| France | | 0.46-6.77 | 0.01-0.93 | 0.01-0.21 | 0.95-6.43 | | (Bergé et al. 2012b) |
| Min | 0.08 | 0.02 | 0.01 | 0.01 | 0.02 | 0.01 | n = 20 |
| Max | 13.4 | 49.8 | 58.9 | 1.40 | 69.0 | 30.8 | |
| Med | 0.34 | 0.80 | 0.57 | 0.18 | 2.44 | 0.24 | |

Table 7: PAE in WWTP sludge contents (in mg/kg.dw)

| Location | Type | DMP | DEP | DnBP | BBP | DEHP | DnOP | Reference |
|-----------|----------------|-----------|-----------|-----------|-----------|------------|-----------|-----------------------------|
| Canada | Final | | | | | 3.00-176 | | (Webber and Lesage 1989) |
| Canada | Final | | | | | 1.60-245 | | (Webber and Nichols 1995) |
| Germany | Final | | | 9.70 | | 8.00 | | (Petrovic and Barcelo 2000) |
| Austria | Final | | | <loq-0.69 | | <loq-47.0 | | (Gangl et al. 2001) |
| China | Final | 0.02-2.00 | 0.01-11.0 | 0.04-3.70 | 0.02-35.0 | <loq-108 | <loq-6.60 | (Cai et al. 2007) |
| Norway | Final | | | | | 27.0-115 | | (Jaganyi 2007) |
| Sweden | Final | | | | | 25.0-661 | | (Jaganyi 2007) |
| Denmark | Final | | | | | 3.90-170 | | (Jaganyi 2007) |
| Denmark | Final | | | | | 3.46-40.6 | | (Seriki et al. 2008) |
| Australia | Primary | | | | | 134 | | (Tan et al. 2007) |
| Austria | Primary | 0.07-0.09 | 0.04-0.09 | 0.27-0.85 | 0.14-0.38 | 20.0-27.0 | 0.13-0.26 | (Clara et al. 2010) |
| USA | Secondary | | | | | 3.46-31.7 | | (Kinney et al. 2006) |
| Sweden | Digested | | | | | 25.0-661 | | (Sweetman 1994) |
| Finland | Digested | | | | | 163 | | (Martinen et al. 2003) |
| USA | Digested | | | | | 3.33 | | (Kinney et al. 2008) |
| USA | Amended | | | | | 163-578 | | (Staples et al. 1997) |
| Denmark | Amended | | | 0.02-260 | | 3.00-170 | | (Torslov et al. 1997) |
| Germany | Amended | | | | | 170 | | (Schnaak et al. 1997) |
| Germany | Amended | | | | | 1.50-5.10 | | (Schaecke and Kape 2003) |
| UK | Amended | 0.30 | 0.02 | 0.39 | 0.20 | 0.32-0.55 | 0.57 | (Gibson et al. 2005) |
| Spain | Amended | | | | | 2.00-3,514 | | (Abad et al. 2005) |
| Denmark | Amended | | 1.52 | 1.30 | 1.17 | | 1.24 | (Laturus and Gron 2007) |
| Denmark | Amended | | 0.91 | 0.56 | | | | (Laturus and Gron 2007) |
| Spain | Amended | | | | | 13.0-345 | | (Aparicio et al. 2009) |
| Norway | Compost | | | | | 1.00-140 | | (Paulsrud et al. 2000) |
| Taiwan | Aerobically | | | | | 143 | | (Cheng et al. 2001) |
| Taiwan | Anaerobically | | | | | 105-153 | | (Cheng et al. 2001) |
| Germany | Spin-dried | | | 0.20-1.70 | | 28.0-154 | | (Fromme et al. 2002) |
| France | Spin-dried | | | | | 72.1 | | (Dargnat et al. 2009) |
| Austria | Excess | <loq-0.06 | 0.06-0.13 | 0.64-1.20 | 0.12-0.25 | 22.0-29.0 | 0.06-0.12 | (Clara et al. 2010) |
| Canada | Press-filtered | | | | | 80.0-90.0 | | (Barnabé et al. 2008) |
| USA | | | | | | <loq-58.3 | | (Clark et al. 2003) |
| Germany | | | | | | 0.90-110 | | (Fragermann 2003) |
| USA | | | | | | <loq-310 | | (EPA 2009) |
| | Min | 0.02 | 0.01 | 0.04 | 0.02 | 0.32 | 0.08 | n = 29 |
| | Max | 2.00 | 11.0 | 260 | 35.0 | 3,514 | 6.60 | |
| | Med | 0.19 | 0.45 | 0.53 | 0.23 | 60.3 | 0.20 | |

Table 8: Surface water concentrations of PAE (in $\mu\text{g.l}^{-1}$)

| Location | Type | DMP | DEP | DnBP | BBP | DEHP | DnOP | Reference |
|-------------|-----------------|-----------|-----------|-----------|-----------|-----------|-----------|-----------------------------|
| USA | River | | | | | 0.50-1.00 | | (Sheldon and Hites 1979) |
| USA | Estuary | | | | | 0.21-0.77 | | (Ray et al. 1983) |
| USA | River | | | | | 0.01-0.70 | | (DeLeon et al. 1986) |
| Sweden | River | | | | | 0.30-1.80 | | (Thüren 1986) |
| Germany | River | | | | | 0.03-0.04 | | (Ernst et al. 1988) |
| UK | River | | | | | 0.40-1.90 | | (Fatoki and Vernon 1990) |
| UK | Bay | | | | | 0.98-2.20 | | (Law et al. 1991) |
| Malaysia | Estuary | | | | | 3.10-64.3 | | (Tan 1995) |
| Italy | River | | | | | 0.30-31.2 | | (Vitali et al. 1997) |
| UK | River | | | | | 0.74-18.0 | | (Long et al. 1998) |
| Netherlands | Estuary | | | | <loq-0.50 | 0.04-1.90 | | (Belfroid et al. 1999) |
| Spain | River | | 0.60 | 0.40 | | 1.10 | | (Penalver et al. 2001) |
| Spain | Fishing port | 1.60 | 1.40 | 1.30 | 0.50 | 2.10 | | (Penalver et al. 2001) |
| Spain | Industrial port | 2.10 | 1.80 | 1.90 | 1.10 | 3.20 | | (Penalver et al. 2001) |
| Japan | Bay | 0.01-0.09 | <loq-0.31 | 0.01-0.54 | <loq-0.06 | | | (Suzuki et al. 2001) |
| Taiwan | Bay | | <loq-2.50 | 1.00-13.5 | | <loq-18.5 | | (Yuan et al. 2002) |
| Germany | River | | | 0.50 | | 0.08-10.0 | | (Fromme et al. 2002) |
| Netherlands | River | | | | 0.01-1.00 | | | (Vethaak et al. 2002) |
| Netherlands | Sea | | | | 0.01-1.80 | | | (Vethaak et al. 2002) |
| China | River | | | 3.39-12.6 | | <loq-1.90 | | (Zhang and Chen 2003) |
| Netherlands | River | 0.01-0.19 | 0.07-2.30 | 0.07-3.10 | 0.01-1.80 | 0.90-5.00 | <loq-0.08 | (Vethaak et al. 2005) |
| Sweden | River | | 0.01-0.03 | 0.02-0.06 | | 0.01-0.04 | | (Bendz et al. 2005) |
| China | River | <loq-1.40 | 0.26-1.28 | 0.05-3.91 | | | 0.10-0.80 | (Li et al. 2006) |
| Spain | Estuaries | 0.01-0.18 | 0.31-1.31 | 0.25-0.58 | 0.01-0.03 | 0.32-0.46 | 0.08-0.09 | (Prieto et al. 2007) |
| Spain | Sea | 0.01 | 0.03 | 0.08 | 0.01 | 0.06 | | (Prieto et al. 2007) |
| China | River | 0.10-0.25 | 0.16-0.44 | 4.28-21.0 | | 0.34-24.0 | <loq-0.79 | (Sha et al. 2007) |
| China | Tributaries | <loq-0.58 | 0.01-1.09 | 9.24-26.0 | | 3.91-31.8 | <loq-7.10 | (Sha et al. 2007) |
| USA | River | | | 0.14-4.14 | 0.04-0.35 | | | (Solis et al. 2007) |
| USA | River | | | 0.16-1.36 | 0.07-0.14 | | | (Solis et al. 2007) |
| Spain | Estuary | 0.21-0.28 | 0.07 | 1.25-1.26 | 0.05 | 0.22 | 0.03 | (Prieto et al. 2008) |
| Spain | Sea | 0.22-0.25 | 0.04 | 0.25-0.40 | 0.06 | 0.17 | 0.03 | (Prieto et al. 2008) |
| China | River | <loq-0.09 | 0.02-0.32 | 0.94-3.60 | | | | (Zeng et al. 2008) |
| France | River | | 0.07-0.18 | 0.07-0.32 | | 0.16-0.31 | | (Dargnat et al. 2009) |
| China | River | | | | | 0.62-15.2 | 0.04-0.21 | (Zhu and Qiu 2011) |
| Iran | River | 0.87 | 0.67 | | | | | (Hadjmohammadi et al. 2011) |
| Iran | Sea | 0.49 | 0.52 | | | | | (Hadjmohammadi et al. 2011) |
| Canada | Sea | | | 0.18-3.00 | | 0.01-0.95 | | (Keil et al. 2011) |
| USA | Sea | | | | | 0.06-0.64 | | (Keil et al. 2011) |
| China | River | | | 0.11-0.29 | | <loq-0.84 | | (He et al. 2011) |
| Spain | Sea | <loq-0.14 | 0.02-0.48 | | <loq-0.10 | 0.03-0.62 | | (Sanchez-Avila et al. 2012) |

| | | | | | | | | |
|-------|-----------|-----------|-----------|-----------|-----------|-----------|-----------|-----------------------------|
| Spain | Port | <loq-0.01 | 0.02-0.87 | | <loq-0.80 | 0.06-5.97 | | (Sanchez-Avila et al. 2012) |
| Spain | River-sea | | 0.07-0.16 | | <loq-0.08 | 0.02-0.21 | | (Sanchez-Avila et al. 2012) |
| Spain | River | | 0.05-0.28 | | <loq-0.02 | 0.12-4.98 | | (Sanchez-Avila et al. 2012) |
| China | River | 0.03-1.45 | 0.03-0.71 | 0.02-1.35 | 0.01-0.86 | 0.02-5.58 | 0.04-0.12 | (Shi et al. 2012) |
| China | River | 0.02-0.13 | 0.01-0.09 | 0.06-7.19 | 0.02-0.07 | 0.23-28.4 | 0.01-0.34 | (Zhang et al. 2012) |
| Min | | 0.01 | 0.01 | 0.01 | 0.01 | 0.01 | 0.01 | n = 33 |
| Max | | 2.10 | 2.50 | 26.0 | 1.80 | 64.3 | 7.10 | |
| Med | | 0.05 | 0.11 | 0.37 | 0.04 | 1.00 | 0.04 | |

Table 9: PAE contents in suspended solids (in mg/kg.dw)

| Location | DMP | DEP | DnBP | BBP | DEHP | DnOP | Reference |
|-------------|-----------|-----------|-----------|-----------|-----------|------|--------------------------------|
| Netherlands | <loq-16.0 | <loq-2.69 | <loq-4.10 | <loq-3.00 | <loq-19.0 | | (Vethaak et al. 2005) |
| Netherlands | | | | | 0.79-11.4 | | (Peijnenburg and Struijs 2006) |
| Netherlands | | | | | 0.70-14.6 | | (Peijnenburg and Struijs 2006) |
| Netherlands | | | | | 0.97-19.3 | | (Peijnenburg and Struijs 2006) |
| China | 0.49-3.01 | <loq-0.13 | 17.6-57.8 | | 5.40-630 | | (Sha et al. 2007) |
| France | | | | | 0.91-25.1 | | (Gasperi et al. 2008) |
| Min | | | | | 0.70 | | n = 4 |
| Max | 16.0 | 2.69 | 57.8 | 3.00 | 630 | | |
| Med | 0.22 | 0.04 | 0.09 | 0.02 | 4.23 | | |

Table 10: PAE contents in sediments (in mg/kg.dw)

| Location | Type | DMP | DEP | DnBP | BBP | DEHP | DnOP | Reference |
|-------------|-------------|-----------|-----------|-----------|-----------|-----------|-----------|--------------------------------|
| USA | River | | | | | 0.04-16.0 | | (Ray et al. 1983) |
| UK | River | | | | | 1.20 | | (Preston and Al-Omran 1989) |
| Malaysia | River | | | 0.25 | | 0.49-15.0 | | (Tan 1995) |
| Singapore | Bay | | | | | 0.89-2.79 | | (Chee et al. 1996) |
| Italy | River | | | <loq-0.03 | | 0.06-0.49 | | (Vitali et al. 1997) |
| UK | River | | | | | 0.84-115 | | (Long et al. 1998) |
| Germany | River | | | 0.45 | | 0.21-8.44 | | (Fromme et al. 2002) |
| Taiwan | River | | 0.10-1.10 | 0.30-30.3 | <loq-1.80 | 0.50-23.9 | | (Yuan et al. 2002) |
| China | River | | | 0.02-0.05 | | 0.03-0.05 | | (Zhang and Chen 2003) |
| Netherlands | River | <loq-2.50 | 0.07-1.20 | 0.03-1.00 | <loq-0.06 | 0.10-7.60 | <loq-0.55 | (Vethaak et al. 2005) |
| Spain | River | | <loq-0.24 | 0.02-0.79 | 0.50-0.95 | 10.1-16.8 | | (Cortazar et al. 2005) |
| France | River | | | | | 0.55 | | (Quenea et al. 2005) |
| Taiwan | River | | | | | <loq-8.25 | | (Huang et al. 2005) |
| Japan | River | | | | | 1.00-2.00 | | (Yuwatini et al. 2006) |
| Netherlands | River | | | 0.09 | | 4.30 | | (Peijnenburg and Struijs 2006) |
| China | River | 0.14-0.42 | | 18.1-34.1 | | 9.29-50.7 | | (Sha et al. 2007) |
| China | Tributaries | <loq-1.04 | <loq-0.01 | 3.63-72.2 | | 5.35-259 | | (Sha et al. 2007) |
| China | Urban lake | 0.04 | 0.13 | 0.28 | 0.03 | 1.30 | 0.02 | (Zeng et al. 2008) |
| Taiwan | River | | 0.60 | 0.40 | | <loq-46.5 | | (Huang et al. 2008) |
| Taiwan | River | | | 0.04-1.88 | | | | (Huang et al. 2008) |
| China | River | 0.01-0.41 | 0.55-6.81 | 0.50-155 | | 0.40-324 | 0.01-1.19 | (Wang et al. 2008) |
| India | Urban River | <loq-0.02 | | <loq-0.02 | | <loq-0.02 | | (Srivastava et al. 2009) |
| India | Rural River | <loq-0.01 | <loq-0.01 | <loq-0.02 | | <loq-0.01 | | (Srivastava et al. 2009) |
| India | Urban River | <loq-0.05 | <loq-0.04 | <loq-0.04 | | <loq-0.32 | <loq-0.05 | (Srivastava et al. 2009) |
| Taiwan | River | | | | | 0.10-20.2 | | (Lin et al. 2009) |
| France | River | | | | | 0.91-26.6 | | (Gasperi et al. 2009) |
| China | River | 0.02 | 0.18 | 0.10 | 0.02 | 0.39 | | (Liu et al. 2010) |
| China | River | 0.03 | 0.26 | 0.04 | 0.01 | 0.22 | | (Liu et al. 2010) |
| | Min | 0.01 | 0.01 | 0.02 | 0.01 | 0.01 | 0.02 | n = 23 |
| | Max | 2.50 | 1.20 | 72.2 | 1.80 | 115 | 0.55 | |
| | Med | 0.02 | 0.20 | 7.45 | 0.02 | 10.2 | 0.04 | |

Table 11: PAE contents in soils (in mg/kg.dw)

| Location | Type | DMP | DEP | DnBP | BBP | DEHP | DnOP | Reference |
|-------------|--------------|-----------|-----------|-----------|-----------|-----------|-----------|--------------------------------|
| USA | Urban | | | | | nd-1.20 | | (Russel and Mc Duffie 1986) |
| Denmark | Amended | | | | | 0.03-0.04 | | (Vikelsøe et al. 2002) |
| China | Urban | <loq-0.20 | <loq-2.61 | <loq-1.66 | | 0.20-7.11 | | (Hu et al. 2003) |
| China | Urban | <loq-0.02 | <loq-0.05 | 0.34-1.66 | | 0.22-0.74 | 0.09 | (Ma et al. 2003) |
| UK | Rural | | | 0.01 | | 0.02-0.08 | 0.01 | (Gibson et al. 2005) |
| UK | Amended | | | 0.01 | | 0.32-0.55 | | (Gibson et al. 2005) |
| China | Vegetable | <loq-0.07 | <loq-1.77 | <loq-20.6 | <loq-1.48 | 2.82-25.1 | <loq-0.92 | (Cai et al. 2005) |
| Netherlands | | | | 0.01 | | 0.03 | | (Peijnenburg and Struijs 2006) |
| China | Urban | <loq-0.07 | <loq-0.25 | 0.28-3.82 | <loq-0.06 | 0.17-6.49 | <loq-0.17 | (Li et al. 2006) |
| Denmark | Agricultural | | 0.01 | 0.07 | | 0.18 | | (Laternus and Gron 2007) |
| Denmark | Compost | | 0.46 | 0.29 | | 12.2 | 1.24 | (Laternus and Gron 2007) |
| China | Agricultural | | | 2.75-29.4 | | 0.49-7.99 | | (Xu et al. 2008) |
| China | Agricultural | <loq-0.16 | <loq-0.18 | <loq-2.74 | <loq-1.58 | 0.11-29.4 | <loq-0.08 | (Zeng et al. 2008) |
| China | Roadsides | 0.02-0.35 | 0.01-0.20 | 0.29-30.1 | <loq-1.58 | 1.41-264 | <loq-2.31 | (Zeng et al. 2009) |
| China | Resident | 0.01-0.13 | 0.01-0.10 | 0.21-7.49 | <loq-0.16 | 1.40-97.2 | <loq-0.08 | (Zeng et al. 2009) |
| China | Parks | 0.02-0.10 | 0.01-0.07 | 0.21-7.49 | <loq-0.16 | 0.89-154 | <loq-0.03 | (Zeng et al. 2009) |
| China | Urban | | | | | 0.04 | | (Zhou and Liu 2010) |
| | Min | 0.01 | 0.01 | 0.01 | | 0.02 | 0.01 | n = 13 |
| | Max | 0.35 | 2.61 | 30.1 | 1.58 | 264 | 2.31 | |
| | Med | 0.08 | 0.09 | 0.96 | 0.05 | 3.33 | 0.09 | |