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RESEARCH ARTICLE

National screening study investigating nine phthalates and one adipate in raw and treated tap water in France

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Abstract

The goal of this study was to determine the potential exposure of much of the French population to nine phthalates and bis (2-ethylhexyl) adipate (DEHA) due to water consumption. The occurrence of these compounds was investigated in raw and treated water from public water systems. Water samples were collected in one sampling campaign equally distributed across 101 French *départements* (a French administrative unit) from November 2015 to July 2016. In all, 271 raw water samples and 283 treated water samples were collected. A specific sampling protocol was conducted in order to assess phthalate pollution during sampling and analysis, and to produce reliable results. Field blanks were thus collected at the same time as real samples at each sampling point. The contamination detected in field blanks was due to diethyl phthalate (DEP), dibutyl phthalate (DBP), diisobutyl phthalate (DIBP), and di-2-ethylhexyl phthalate (DEHP), which are common phthalate interferences in blanks. Their concentrations were never ten times higher than the limits of quantification (LOQ). In tap water, the most frequently detected compound was DBP, at a maximum concentration of 1300 ng/L. In raw water, however, DEP was the most frequently detected analyte with concentrations ranging from 255 to 406 ng/L, while DIBP was observed at a maximum concentration of 1650 ng/L. It is worth mentioning that DEHP—the most widely used phthalate—was only detected in one sample of raw water. Phthalates are not concentrated in any particular area of France in either raw or treated water.

Keywords Phthalate ester · Raw water · Tap water · Field blank · Online SPE · LC-MS

Introduction

Phthalates and adipates are anthropogenic compounds used in a wide range of industrial applications. High-molecular-weight phthalates, such as di-2-ethylhexyl phthalate (DEHP) and di-iso-nonyl phthalate (DiNP), are mainly employed as plasticizers in the production of flexible vinyl, which is used in consumer products, food contact packaging, wall coverings, and medical devices. Low-molecular-weight phthalates, such as benzyl butyl phthalate (BBP) and di-iso-butyl phthalate (DIBP), are added as solvents to personal-care products and used as plasticizers for making lacquers, varnishes, and coatings (Directive 2000/60/CE; Hauser and Calafat 2005). In addition, traces of bis-2-ethylhexyl adipate (DEHA) are contained in polyvinyl chloride films for wrapping foodstuffs (Fasano et al. 2012).

Due to their use in various products of everyday life, these compounds have become ubiquitous contaminants present throughout the environment, including in water (Net et al. 2015a). The toxic potency of phthalates on human health is a matter of concern for the scientific community. Several studies have shown the acute toxicity, genotoxic effects, and potential endocrine-disrupting activity of phthalates and DEHA using in vitro and in vivo models (Boas et al. 2012; Caldwell 2012; Chen et al. 2014; Jarfelt et al. 2005).

The use of phthalates has been restricted worldwide due to the risk of exposure for the population. Dibutyl phthalate (DBP), BBP, and DEHP have been included in the EU's REACH regulation as very dangerous substances (Ventrice et al. 2013) and as priority pollutants by EPA in the USA. For drinking water, the maximum concentration of DEHP

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was set at 6 and 8 μg/L according to the WHO and EPA respectively. In Australia, Japan, and New Zealand, the maximum DEHP concentration was set at 9, 100, and 10 μg/L respectively. The EU guideline for environmental quality has set a concentration limit of 1.3 μg/L for DEHP in fresh and marine water (Net et al. 2015a).

The determination of plasticizers (and especially phthalates) in environmental samples has been comprehensively reviewed by Net et al. (2015a). Classical techniques such as liquid-liquid extraction (LLE) and semi-automated solid-phase extraction (SPE) have been widely used for water samples. However, free-solvent techniques requiring less sample preparation have also been used. These include solid-phase micro-extraction (SPME) or stir-bar sorptive extraction (SBSE). Not only are these treatments simple and fast, but they require less handling of the sample, thus reducing background pollution. In contrast, the polymeric support can be difficult to desorb as it can be quickly damaged by the high temperature needed. Gas chromatography coupled with mass spectrometry (GC-MS) is the most common technique for separating and identifying these compounds in electron ionization (EI) mode. Phthalates can also be analyzed by liquid chromatography coupled with mass spectrometry (LC-MS) in electrospray ionization mode (ESI). However, LC-MS has proved to be less sensitive than GC-MS. However, all phthalates give similar GC-MS mass spectra with the most abundant phthalic anhydride fragment at 149 m/z due to the high fragmentation pattern of the EI mode. The m/z 149 ion is ubiquitous in analytical instruments, glassware, solvents, and the lab environment (Fankhauser-Noti and Grob 2007). When the m/z 149 ion is used for quantification, there is a high risk of overestimating sample results due to the ion's lack of specificity. Although procedural blanks and precautions are implemented, the random background contamination in the lab, which depends on technicians, room temperature, and air quality among others, is clearly demonstrated (Capdeville and Budzinski 2011). Despite this difficulty, most of the reported methods use the m/z 149 ion for quantification probably due to the higher sensitivity obtained. LC-MS is a good alternative because the ESI mode generates the molecular ion and phthalates can be quantified using more specific transitions. Analytical environmental laboratories are constantly trying to lower the limits of detection and quantification. However, other factors should be taken into account when determining these limits for phthalate analysis. False-positive results can be obtained when samples are contaminated during handling. Few studies (especially those focused on DBP and DEHP) dealing with the sources of contamination for phthalates in routine analysis have been published (Furtmann 1994; Tienpont et al. 2005). Some tips for reducing background pollution and avoiding overestimated results have been published (Capdeville and Budzinski 2011). According to Capdeville and Budzinski (2011), the most common sources of sample contamination are solvents, reagents, glassware, analytical instruments, equipment, and the lab environment. Accordingly, procedural blanks must be processed at the same time as real samples throughout the analytical procedure in order to assess the contribution of contamination and produce reliable results. Several solutions to minimize background contamination by phthalates have also been reported (Fankhauser-Noti and Grob 2007; Furtmann 1994; Marega et al. 2013; Tienpont et al. 2005). Overall, these procedures include glassware calcination (450 °C, 2 h), the washing of lab materials with high-quality phthalate-free solvents (isooctane, hexane), and banishing the use of personal care products during sampling, handling, and pretreatment of the sample. Moreover, it is recommended to dedicate a room (equipped with air filters) and a hood exclusively for sample treatment. Despite these precautions, spot contaminations may occur in any step (INERIS 2009).

Another critical point when analyzing phthalates is the sampling step. The usual sampling protocol consists of collecting water samples in pre-cleaned and calcined (450 °C, 2 h) glass containers avoiding any intermediate plastic item for filling (Net et al. 2015a). Calcined aluminum foil can be used to wrap the neck of the sample containers in order to avoid contamination from the caps (Tienpont et al. 2005). Field blanks should be systematically carried out in parallel with real samples when phthalates are monitored in order to evaluate the potential environmental contamination. Field blanks undergo the same steps as real samples (collection, transport, treatment, analysis). According to EPA, a field blank is a sample of analyte-free water poured into the container on-site, preserved and shipped to the laboratory with the field samples (EPA 2009). Different types of water have been used for field blanks, including MilliQ water, UPLC water, and bottled water (Capdeville and Budzinski 2011). The literature rarely describes how field blanks were prepared and their contamination levels compared to real samples. To the best of our knowledge, only Dévier et al. (2013) investigated DEHP contamination in blanks of mineral water samples. Moreover, the presence of a phthalate in water samples was confirmed only if the analyte response was ten times higher than the response of its procedural blank. Several studies in environmental water have included field and procedural blanks to ensure the validity of their results (Bono-Blay et al. 2012; Hu et al. 2013; Liu et al. 2015; Liu et al. 2014; Net et al. 2015b), but none described in detail how field blanks were prepared and how they exploited their results when field blanks were found to be contaminated. Results on the occurrence of phthalates are thus always controversial because of the potential background pollution.

Table 1 Phthalate and adipate esters analyzed in this study with their abbreviation, CAS registry number, and molecular weight (MW)

Compound	Abbreviation	CAS number	MW (Da)
Dimethyl phthalate	DMP	131-11-3	194.2
Diethyl phthalate	DEP	84-66-2	222.2
Dibutyl phthalate	DBP	84-74-2	278.4
Benzyl butyl phthalate	BBP	85-68-7	312.4
Diisobutyl phthalate	DIBP	84-69-5	278.4
Dicyclohexyl phthalate	DCHP	84-61-7	330.4
Dihexyl phthalate	DHP	84-75-3	334.5
Di-2-ethylhexyl phthalate	DEHP	117-81-7	390.6
Bis-2-ethylhexyl adipate	DEHA	103-23-1	370.6
Dioctyl phthalate	DOP	117-84-0	390.6

The main purpose of this study was to evaluate potential exposure of the French population to nine phthalates and one adipate (see Table 1 for the list of target compounds). An analytical method using online SPE-LC-MS/MS was developed and validated. A detailed protocol for sampling field blanks and water was then drafted and applied to a sampling campaign using raw and treated tap water. The field blank results were monitored and compared with values found in the associated sample in order to determine the occurrence of target compounds in drinking water.

Materials and methods

Reagents and standards

A standard solution of nine phthalates and DEHA at 1000 mg/L in methanol (MeOH) was purchased from CPA Chen via ACSD (Trappes, France). The mass-labeled standards, benzylbutyl phthalate-3,4,5,6-d_4 and di-2-ethylhexyl adipate-d_4 at 100 μg/mL in cyclohexane and acetone respectively, were obtained from Dr. Ehrenstorfer via LGC Standards (Molsheim, France). The $^{13}C_2$-dihexyl phthalate at 100 μg/mL in nonane was purchased from Cambridge Isotope Laboratories (Andover, USA). The other deuterated-labeled phthalate standards at 100 μg/mL in MeOH were obtained from Techlab (St Julien-les-Metz, France). MeOH and water (both ULC-MS grade) were purchased from Biosolve BV (Dieuze, France).

When preparing the samples and standard solution, no plastics were allowed to be used to avoid contamination. Lab glassware (excluding volumetric flasks) was calcined for 4 h at 500 °C and then wrapped in aluminum foil and stored separately. Prior to its use, the glassware was washed several times with solvent in order to minimize contamination. Specific batches of solvents were dedicated to phthalate analysis.

Collection of field blanks and samples

Raw and treated water was collected during a sampling campaign equally distributed across 101 French *départements* from November 2015 to July 2016. Two sample sites were investigated in each *département*: the drinking water source with the greatest flow and a different, randomly selected drinking water source. Sixty-two additional samples were collected from sites suspected of being affected by the release of these target compounds due to industrial and commercial activities. Each *département* was allowed to select only one additional sample. Finally, a total of 271 raw water and 283 treated water samples were analyzed, representing approximately 20% of the national water supply flow.

A field blank was prepared for each sample collected using analyte-free bottled water. Unopened bottled water was used to prepare a field blank for each sample from each *département*. During sample collection, bottled water was poured into the sampling container through the sampling equipment prior to the collection of real samples. Each water sample collected corresponds to a field blank. This is how the contamination of total ambient conditions for each sample was assessed. In addition, laboratory sources of contamination were taken into account because field blanks were processed and analyzed like the real water samples. The bottled water used for field blanks was from the same batch.

Water samples were collected twice (aliquots 1 and 2) in 40-mL amber glass vials (vials 1 and 2) previously cleaned and calcined (4 h at 500 °C). After sampling, the necks of glass vials were immediately wrapped in aluminum foil and capped with their Teflon caps. Prior to sample collection, a field blank was prepared following the same protocol described for water samples. The detailed sampling procedure is represented in Fig. S1. Samples were shipped with cold packs and arrived at our laboratory within 24–48 h. Samples were immediately acidified with formic acid (FA) at 0.1% and stored at 4 °C before analysis.

Sample preparation

Treated water and field blanks (1.5 mL) were spiked directly in the injection vials with the mass-labeled standards at a concentration of 200 or 500 ng/L depending on the analytes. Treated water was analyzed without any pretreatment. On the other hand, raw water was systematically centrifuged to eliminate any suspended solids. As centrifugation was a potential source of contamination, field blanks of raw samples were also systematically centrifuged in order to assess the pollution of this additional step. For raw water samples and their associated field blanks, 3 mL of each acidified sample (0.1% FA)

was spiked with mass-labeled standards at a concentration of 200 or 500 ng/L depending on the analytes. Spiked samples were centrifuged (4000 rpm/min, 4 °C) for 2 min, then 1.5 mL of the supernatant was transferred to a vial for the injection.

Online SPE-LC-MS/MS

The samples were analyzed following the procedure described in an application note from Sciex (Schreider et al. 2011). A linear ion trap quadrupole (QTRAP) 5500 (Sciex, Framingham, MA, USA) coupled with an Ultra-Fast Liquid Chromatograph (UFLC) XR (Shimadzu, Columbia, MD, USA) with three LC-20AD pumps (A, B, and C) and a CTC PAL autosampler (Eksigent, Dublin, CA, USA) was used. Pumps A and B delivered solvents onto the chromatographic column. Pump C was used for SPE extraction. The SPE and LC steps were done online using a six-port switching valve. Water samples (1000 μL) were loaded on a Hypersil GOLD SPE column (20 × 2.1 mm, 12 μm) from Thermo Scientific using the mobile phase C (water - 0.1% formic acid) pre-filtered with a Hypersil GOLD column (50 × 2.1 mm, 3 μm) also from Thermo Scientific to avoid potential contaminants. The loading conditions were 0.7 mL/min for 4 min. The trapped analytes were then directly eluted from the SPE column onto the head of the analytical column. An Xterra C_{18} column (100 × 2.1 mm; 3.5 μm) from Waters was used for chromatographic separation at 40 °C. A Hypersil GOLD column (100 × 2.1 mm; 3 μm) was used as a filter for mobile phase A (water - 0.1% formic acid) and B (MeOH - 0.1% formic acid) to retain potential pollution. The optimal gradient conditions for LC separation were as follows: 0 min, 50% B; 12 min, 80% B; 12.1 min and for 2 min, 98% B; and 18.1 min and for 2 min, 50% B. The column temperature was kept constant at 40 °C, and the flow rate of the mobile phase was 300 μL/min. To prevent cross-contamination, the syringe and the sample loop were flushed twice with 5 mL of a solvent mixture (ULC-MS water 25%, ACN 25%, MeOH 25%, isopropanol 25%) and then twice with 5 mL of ULC-MS. The MS was operated in positive electrospray ionization multiple (ESI+) reaction monitoring (MRM) mode. The MS instrumental parameters were as follows: ion spray voltage, 5500 V; source temperature, 400 °C; curtain gas flow, 35 arbitrary units (au); ion source gas 1, 40 au; and ion source gas 2, 50 au. Two transitions for the analytes were monitored. The most intensive transition was used for the quantification. Specific MRM transitions for the target compounds as well as their optimized compound-dependent parameters are presented in Table 2. The criteria used to identify target compounds in MS were retention time, the presence of its specific transitions, and the relative intensities of the detected product ions (ratio qualifier/quantifier transitions). LC chromatograms for nine phthalates and DEHA can be found in Fig. S2.

Method performances

The analytical method was validated to prove its reliability and consistency for the identification and quantification of the analytes. The validation was conducted with matrices representative of the sampling campaign, namely raw water and tap water.

Procedural blanks and calibration solutions were prepared using acidified mineral water (0.1% FA). Ultra-pure water was not employed because of phthalate contamination possibly due to the water purification system filters. Mineral water was previously analyzed and found to be free of analytes. For raw water samples, the collected samples, calibration solutions, and procedural blanks were all centrifuged and then analyzed in order to assess the pollution of the whole analytical process.

Linearity was checked by spiking seven different equidistant concentrations of analytes in acidified mineral water (0.1% FA), with a quadratic fit over the entire range studied. Internal standard calibration was carried out using isotopically labeled compounds at a concentration of 200 or 500 ng/L depending on analytes. The method showed a quadratic response with R-squared values from regression analysis > 0.97 in all cases (Table S1).

Procedural blanks (acidified mineral water (0.1% FA)) were used to monitor the analytical procedure's background pollution. LOQ levels were determined on the basis of a signal-to-noise ratio of ten for analytes not detected in procedural blanks. The LOQ levels for DEP, DIBP, DBP, and DEHP were calculated on the basis of a signal-to-noise ratio of 20 for analytes in procedural blanks. This avoided having to subtract procedural blank values and/or calculate LOQ levels for each sample batch. The LOQ levels ranged from 50 to 500 ng/L depending on analytes (see Table 2).

Table 3 summarizes the method's performances. Recovery and accuracy experiments were performed by spiking each of the abovementioned matrices with three different concentrations of analytes: LOQ, medium level, and high level (Table 3). Both experiments were assessed by analyzing two spiked samples on the same day over 5 days. The accuracy was expressed as relative standard deviation (RSD). Mean recoveries ranged from 94 to 111% in raw water for the three levels investigated. For treated water, mean recoveries ranged from 90 to 114% for the three levels. The lower and upper limits (60 −140%) complied with the SANTE guidelines for both matrices (SANTE/11813/2017). With regard to the accuracy, the RSDs for raw and tap water spiked at three concentration levels were between 4 and 23% and 4–

Table 2 Limits of quantification, retention times, multiple reaction monitoring (MRM) transitions for quantification (in bold), and qualification and mass parameters for target compound analysis

Compound (IS)	LOQ (ng/L)	RT (min)	Transitions	DP (V)	CE (V)	CXP (V)
DMP (*DMP–d_4*)	50	2.94	**195 > 77**	16	43	4
			195 > 92	16	59	4
			199 > 167	*51*	11	12
DEP (*DEP–d_4*)	150	5.47	**223 > 149**	76	25	12
			223 > 177	76	15	12
			227 > 153	*36*	*21*	*8*
DBP (*DBP–d_4*)	500	12.21	**279 > 149**	121	25	16
			279 > 205	121	11	14
			283 > 153	*106*	*23*	*14*
BBP (*BBP–d_4*)	50	12.41	**313 > 91**	131	57	8
			313 > 149	131	17	10
			317 > 91	*126*	*51*	*4*
DIBP (*DBP–d_4*)	150	12.53	**279 > 149**	121	25	16
			279 > 205	121	11	14
			283 > 153	*106*	*23*	*14*
DCHP (*DCHP–d_4*)	50	14.73	**331 > 149**	71	41	16
			331 > 167	71	19	22
			335 > 153	*171*	*31*	*24*
DHP (*DHP–$^2C_{13}$*)	50	15.19	**335 > 149**	111	25	16
			335 > 233	111	13	8
			339 > 153	*116*	*15*	*8*
DEHP (*DEHP–d_4*)	500	15.85	**391 > 167**	136	21	20
			391 > 279	136	13	22
			395 > 283	*126*	*15*	*26*
DEHA (*DEHA–d_4*)	500	15.87	**371 > 129**	186	23	10
			371 > 101	186	43	10
			395 > 265	*176*	*25*	*10*
DOP (*DOP–d_4*)	150	16.06	**391 > 261**	161	13	22
			391 > 149	161	27	16
			395 > 283	*1*	*13*	*16*

LOQ, limit of quantification; *RT*, retention time; *DP*, declustering potential; *CE*, collision energy; *CXP*, cell exit potential

The italic values are for isotopically labeled compounds

16% respectively. These results were in good agreement with a maximum tolerance of 30% thus demonstrating the accuracy of the analytical method.

Quality assurance

For each sample batch, procedural blanks were prepared at several intervals to check the potential contamination and carry-over from sample to sample. The procedural blanks in this study never exceeded ten times the LOQ. The reliability of the results was checked using within-run and intra-sample controls that were performed for each sample batch. Two within-run controls corresponding to the fourth (QC 1) and the sixth (QC 2) point of the calibration curve were inserted in each sample batch. The concentrations of these within-run controls (QC 1 and QC 2) for each analyte are shown in Table S1. Batches were validated only when the bias between the experimental and theoretical concentration was $\leq 20\%$.

Intra-sample controls consisted of spiking one of the raw and treated water samples from each *département* with the target compounds. Samples were spiked at the fourth calibration point concentration (DOP). Intra-sample control concentrations are also shown in Table S1.

Results and discussion

Procedure for assessing the reliability of results

Water samples were systematically collected twice (in two vials of 40 mL) along with a field blank. The first aliquot

Table 3 Spiking levels, mean recoveries with relative standard deviation (RSD), inter-day accuracy expressed with the RSD for raw and treated water. Experiments were conducted by analyzing two spiked samples in the same day over a 5-day period ($n = 2 \times 5$)

		Raw water ($n = 2 \times 5$)		Tap water ($n = 2 \times 5$)	
Analytes	Spiking level (ng/L)	% mean recovery (RSD)	Accuracy, % inter-day RSD	% mean recovery (RSD)	Accuracy, % inter-day RSD
DMP	50	106 (4)	5	106 (4)	4
	150	108 (5)	5	106 (12)	12
	300	105 (6)	6	103 (8)	8
DEP	150	99 (8)	23	93 (6)	7
	200	98 (14)	14	93 (9)	9
	500	102 (9)	10	102 (6)	6
DBP	500	98 (7)	7	105 (3)	5
	700	102 (9)	9	105 (5)	5
	1000	101 (4)	4	104 (7)	7
DIBP	500	97 (3)	6	98 (4)	4
	700	105 (11)	11	104 (7)	7
	1000	99 (6)	6	100 (5)	5
BBP	50	94 (8)	10	102 (7)	7
	150	105 (7)	7	96 (7)	7
	300	100 (6)	6	96 (6)	6
DCHP	50	102 (5)	14	105 (5)	9
	150	106 (12)	12	102 (14)	14
	300	111 (9)	9	98 (11)	11
DHP	50	96 (8)	10	114 (10)	10
	150	102 (8)	8	92 (14)	14
	300	104 (8)	8	90 (10)	10
DEHA	500	94 (15)	15	103 (8)	8
	700	98 (10)	10	102 (6)	6
	1000	96 (16)	16	91 (14)	14
DEHP	500	100 (10)	10	98 (5)	9
	700	101 (11)	11	101 (8)	10
	1000	105 (10)	10	94 (13)	16
DOP	150	98 (24)	24	90 (17)	18
	200	100 (17)	17	95 (15)	15
	500	100 (12)	12	101 (6)	10

and its associated field blank were analyzed. When samples were positive for analytes, a second analysis was performed using the second aliquot in order to confirm the quantification and avoid false-positive results. Field blank concentrations and procedural blanks were also compared with the associated sample results to avoid false-positives. The following guarantees were established in order to clearly distinguish background contamination (analytes present in the blanks) from real analytes present in a sample:

i) The analytes in procedural and field blanks must not be ten times higher than the LOQ. If the concentration of one blank is higher than its LOQ, their associated sample results cannot be validated. This result must be considered as indicative information and sampling and analysis must be repeated.

ii) The concentration found in samples must be higher than its LOQ. Terms such as "traces," "detection," or "presence" (meaning between LOD and LOQ) are not employed in this study. The interpretation of "traces" should be used cautiously, especially for ubiquitous compounds, in order to avoid an ambiguous result.

iii) No statistically significant differences must be observed between concentrations found in both collected aliquots. A Wilcoxon matched pair test is used to compare two aliquot concentrations. Statistical analyses are conducted with Statistica 13 (TIBCO Software Inc., Palo Alto, CA, USA) and statistical significance is set at a p value lower than 0.05.

Field blanks

The contamination detected in field blanks was due to DEP, DBP, DIBP, and DEHP, which are common phthalate interferences in blanks (Fankhauser-Noti and Grob 2007; Tienpont et al. 2005). These compounds have a low molecular weight, high solubility in water, and are widely used in plastic production (Marega et al. 2013). Figure 1 shows the amount of contamination among field blanks collected during the sampling campaign. From November 2015 to the end of January 2016, DBP, DIBP, and DEHP concentrations were below the method's LOQ. In contrast, DEP concentrations in field blanks were slightly over or close to the method's LOQ, revealing pollution during the sampling, transport, or storage. Even so, DEP was not detected in their corresponding samples or in procedural blanks, thus clearly demonstrating the randomness of phthalate pollution. This problem is very difficult to overcome. Field blank contributions between February 3 and March 22, 2016, dramatically increased and reached a maximum concentration of 760 ng/L for DEP, 1592 ng/L for DBP, and 2076 ng/L for DIBP. Even so, DEHP concentrations remained below the LOQ with two random peaks of pollution in field blanks at 823 and 579 ng/L. At first, it was thought that this increase was due to a contribution by the person doing the sampling, but pollution occurred for different samplers and different French *départements*. The origin of this sudden increase was the combination of two problems, namely the way field blanks were sampled and storage time after baking the glass containers. After baking, sampling containers were capped and the neck wrapped with aluminum foil before storage. We noticed that after 3 months of storage, the bottles had absorbed these phthalates. Furthermore, sampling containers were not washed with mineral water before preparing the field blanks. Instead, they were rinsed with the water to be sampled before filling (Guart et al. 2011; Guart et al. 2014). That is why we found pollution in the field blanks but not in their corresponding water samples. As seen in Fig. 1, the pollution dropped dramatically after baking new containers for the same sample batch. However, random contamination occurred throughout the sampling campaign, especially for DBP and DEHP. We do not have any explanation for this contamination, confirming the randomness of phthalate pollution and the difficulties that arise. We can control the entire process, but random pollution is almost unavoidable (Capdeville and Budzinski 2011).

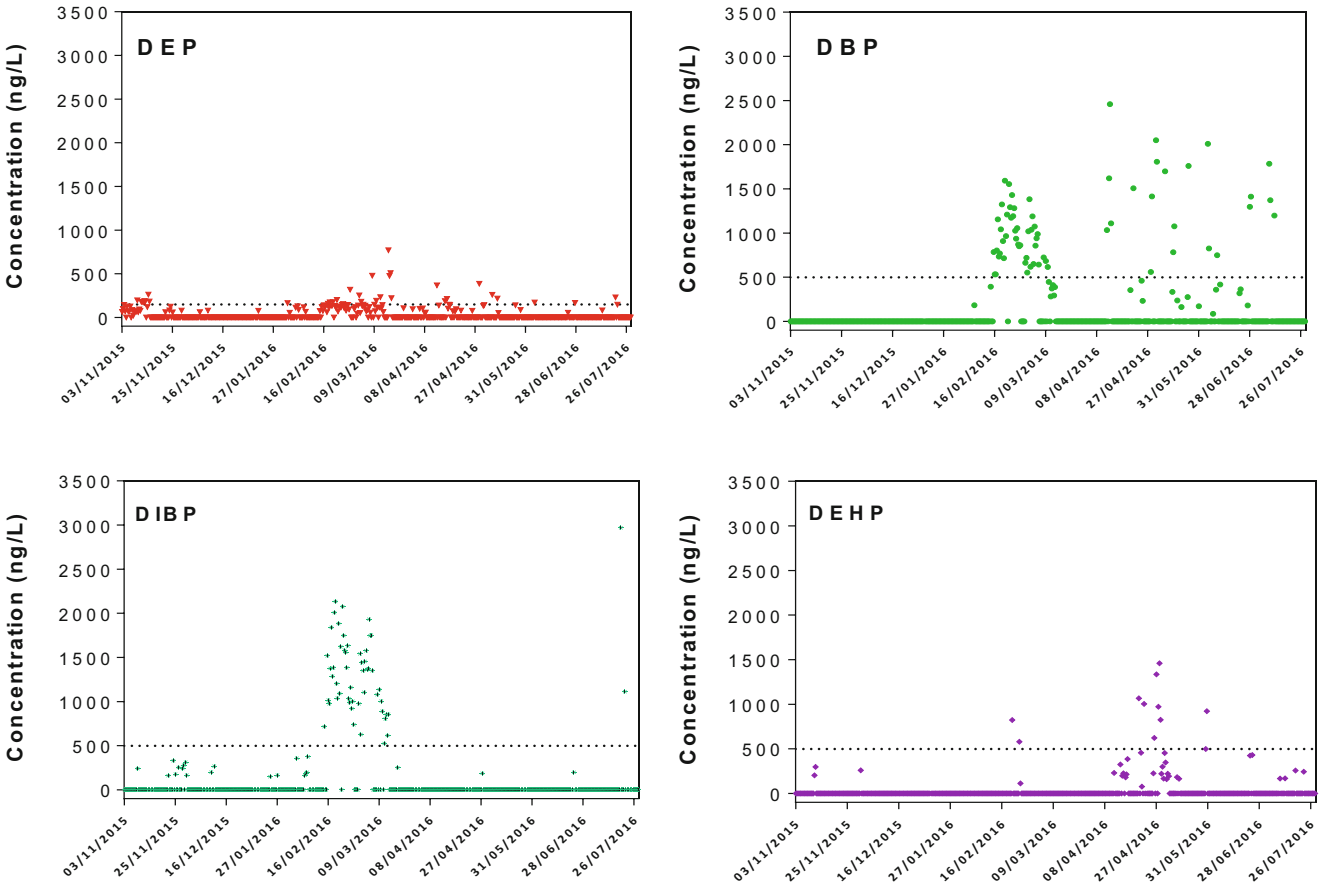

Fig. 1 Concentrations of DEP, DBP, DIBP, and DEHP in field blanks from November 2015 to January 2016

Occurrence of phthalates and DEHA in raw and treated water

The results of the sampling campaign are given in Table 4. This survey involved drinking water networks supplied by groundwater and surface water. Due to the sampling strategy, groundwater samples were predominant, representing 58% of the 271 raw water samples collected. In 261 collected samples (96%), all the analytes were below the LOQ. In only ten samples (4%), at least one analyte was detected at a concentration greater than the LOQ. DEP was the most frequently detected analyte, with concentrations ranging from 255 to 406 ng/L. DIBP was observed in two samples at a maximum concentration at 1650 ng/L. BBP was also found in two samples at a concentration of 52 and 516 ng/L. DBP and DEHP were individually detected in three different samples at a maximum concentration of 768 and 813 ng/L respectively. DMP, DCHP, DHP, DEHA, and DOP were not detected at all the sampling sites. Although the number of positive samples was very low, the frequency of detection was almost equivalent between surface water (2.3%) and groundwater (2.7%).

The treated water samples represented the quality of water at the customers' tap at least several hours after leaving the drinking water treatment plant. In this study, 283 different drinking water networks were investigated: 166 supplied by groundwater resources, 89 by surface water resources, and 28 by a mixture of surface and groundwater resources. In 279 samples (98%), no target compound was observed. In only four samples (1%), at least one target compound was quantified. DBP was the most frequently detected compound (found in three different samples) at a quite high maximum concentration of 1300 ng/L. DEP was only observed in one sample at a maximum concentration of 260 ng/L. DIBP and DBP were simultaneously detected in only one sample at 1300 and 950 ng/L respectively.

Two sample aliquots with a field blank were collected in separate vials at each sampling point. Figure 2 shows the phthalate concentrations found in water samples for the two vials, with the phthalate amounts in their field and procedural

Table 4 Phthalates found in raw and treated water collected during the sampling campaign

			N	*N* > LOQ	Maximum (ng/L)	Average (ng/L)	Frequency of detection
DEP	Raw water	SW	114	3	406	317	
		GW	157	1	324	324	2%
	Tap water	SW	89	1	255	255	
		GW	166	0	< 150	< 150	
		MW	28	0	< 150	< 150	1.1%
DBP	Raw water	SW	114	1	768	768	
		GW	157	0	< 500	< 500	0.4%
	Tap water	SW	89	1	951	951	
		GW	166	1	1340	1340	
		MW	28	1	1114	1114	1.2%
DIBP	Raw water	SW	114	1	1650	1650	
		GW	157	1	655	655	0.8%
	Tap water	SW	89	1	1296	1296	
		GW	166	0	< 500	< 500	
		MW	28	0	< 500	< 500	0.4%
BBP	Raw water	SW	114	0	< 50	< 50	
		GW	157	2	516	284	0.7%
	Tap water	SW	89	0	< 50	< 50	
		GW	166	0	< 50	< 50	
		MW	28	0	< 50	< 50	0%
DEHP	Raw water	SW	114	1	813	813	
		GW	157	0	< 500	< 500	0.4%
	Tap water	SW	89	0	< 500	< 500	
		GW	166	0	< 500	< 500	
		MW	28	0	< 500	< 500	0%

N, number of collected samples; *LOQ*, limit of quantification; *SW*, surface water; *GW*, groundwater; *MW*, mixture of surface and groundwater

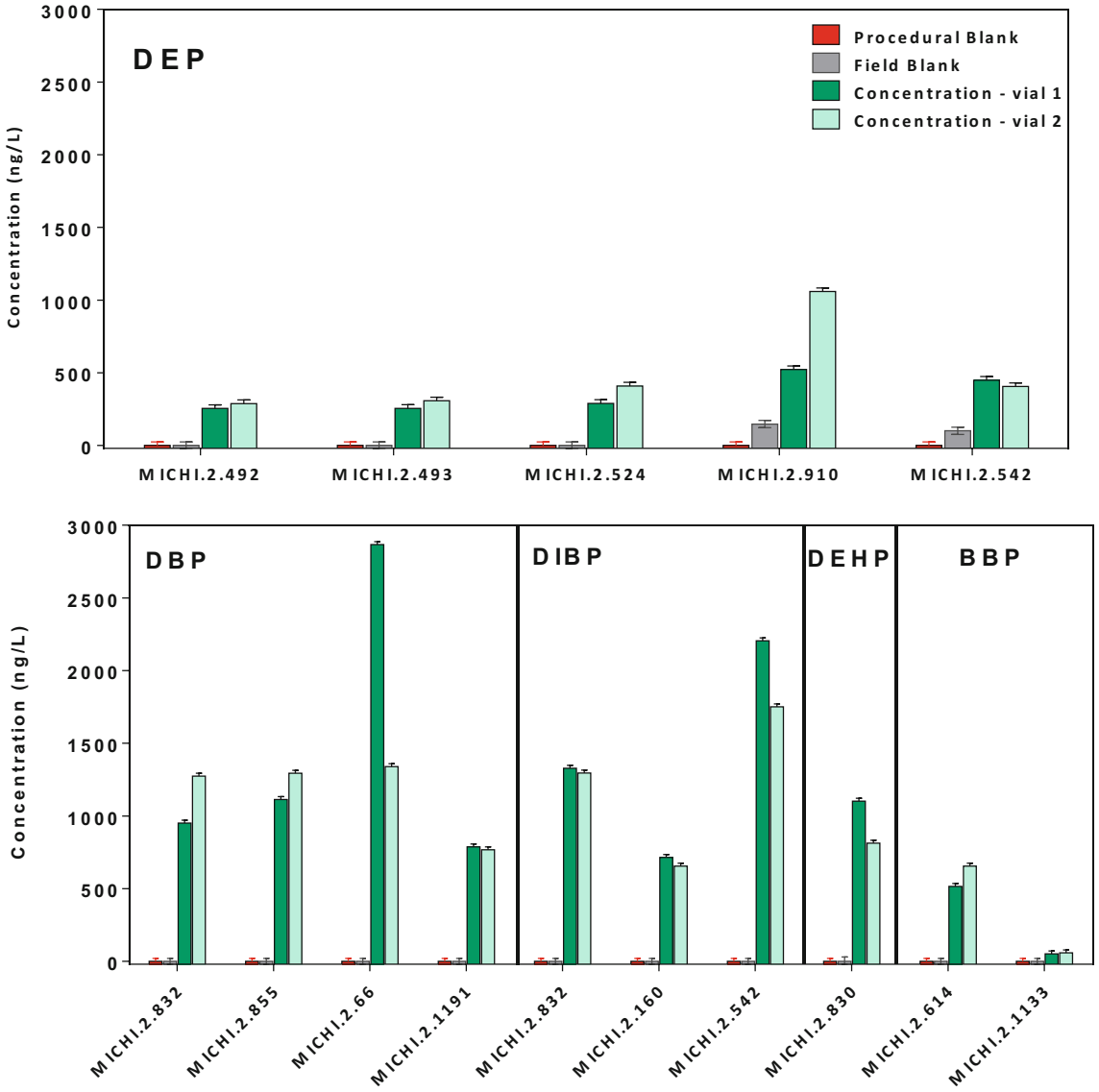

Fig. 2 Concentrations of phthalates found in the two aliquots of the water sample with their respective procedural and field blanks

blanks. As shown in Fig. 2, phthalates were not detected in most field blanks or their amounts were not ten times higher than the LOQ. Moreover, as previously described, the Wilcoxon test was applied to compare values between the two aliquots. No statistically significant differences were found between concentrations in both the aliquots for the same sample ($N = 15$; p value = 0.73).

The spatial distribution of the water sources where analytes were found does not show any hotspot in France for either type of water or their resources. The distribution (number of samples) between raw and tap water is plotted in Fig. S3 for each chemical compound with a concentration greater than the LOQ. With the exception of DBP, the target compounds were detected less frequently in tap water than in raw water. Moreover, the results were compared for the paired samples of raw and tap water. For nine raw water resources where at least one analyte was quantified, only two of the respective drinking water networks systematically contained a compound at a concentration greater than the LOQ.

Despite the few positive results, we compared our data with several previous studies (Abtahi et al. 2019; Cai et al. 2003; Domínguez-Morueco et al. 2014; Gou et al. 2016; Hu et al. 2013; Kong et al. 2017; Lee et al. 2019; Liou et al. 2014; Liu et al. 2015; Loraine and Pettigrove 2006). This comparison is provided in Tables S2 for raw water and S3 for treated water. DBP (92%) is the predominant phthalate in raw water, followed by DEHP (75%) and DEP (73%) according to the average detection frequencies estimated by previous studies (Table S2). In contrast, in our study, the most frequently detected phthalate was DEP (2%), followed by DIBP (0.8%) in raw water samples (Table 4). DEP and DIBP are widely used in perfumes, toys, inks, and nail varnish among other personal care products, which explains their presence in the

environment (Hauser and Calafat 2005). The limited presence of DBP and DEHP, only observed in one sample, could be explained by our higher LOQ compared to the other studies. Indeed, LOQ levels ten to 250 times lower have been reported for DBP and DEHP (Hu et al. 2013; Lee et al. 2019). However, a similar LOQ of 687 ng/L was used by Domínguez-Morueco et al. (2014). Moreover, the latter authors found an average concentration of DBP of 817 ng/L in river water, which is not far from our detected level (see Table S2). In contrast, DEHP was not found in any sample. For DEHP, Domínguez-Morueco et al. (2014) chose a LOQ almost two times higher than ours (970 ng/L compared to 500 ng/L in our study). This could explain the discrepancies.

As regards tap water, DBP was the most frequently detected phthalate, which is in accordance with previous studies (see Table S3). DBP is one of the most frequently produced and used phthalate ester after DEHP. However, DEHP was not observed in any tap water sample (Table 4). In contrast, DEHP is the second most frequently detected compound in the literature. These differences in the frequency of detection could be due to lower LOQ levels (1, 10, and 40 ng/L) compared to our study (500 ng/L). Indeed, DEHP was not found in any sample by Domínguez-Morueco et al. (2014), which reported a similar LOQ of 525 ng/L.

Conclusion

Plasticizers such as phthalates and adipates are among the most ubiquitous environmental and urban contaminants. Analysis is difficult because these chemicals are also present in the laboratory environment, e.g., reagents, lab glassware, and solvents. Their determination is particularly challenging because of the risk of contamination during the sampling to the analysis process. Field and procedural blanks must be continuously monitored in order to produce reliable results. Blank concentrations can only be subtracted from sample concentrations if measurements are repeatable. Moreover, limits of quantitation must be calculated in order to integrate the risk of background pollution. According to the literature and from our own experience, stringent precautions must be taken when analyzing phthalates and adipates:

- We recommend minimal sample preparation, or fully automated analysis;
- High-quality solvents and standards should be employed and continuously tested in order to ensure that they are free of contamination;
- Glassware should be calcinated at 500 °C for 4 h, sealed, and stored in a dedicated place;
- All lab material should be rinsed several times with solvents before use;
- Procedural and field blanks should be prepared in order to monitor the potential pollution and confirm that this pollution is not ten times higher than the limits of quantification;
- Two sample aliquots should be collected in order to confirm the presence and concentration of target compounds;
- The sample concentrations obtained should be reported in parallel with blank concentrations.

Any study investigating these compounds should be considered with caution unless the above recommendations are implemented. Despite these recommendations, spot contaminations may occur throughout the whole analytical and sampling procedure.

Our investigations show that phthalates are rarely found in raw and treated tap water in France. In raw water, DEP was the predominant compound at a maximum concentration of 406 ng/L. In contrast, DBP was the most frequently detected compound in treated water with a maximum concentration of 1300 ng/L. DEHP, which the most popular and widely used phthalate, was found in only one sample of raw water at 813 ng/L. This concentration value does not exceed the concentration limits of international regulations. Our frequency of detection in raw water is not in accordance with previous studies. This difference could be explained by our higher limits of quantification. More investigations are needed to elucidate the origin of phthalates in certain raw and treated water samples. The possible sources of phthalates and adipates could be materials used for water treatment and/or transport and storage through to the distribution point.

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