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Occurrence of phthalate acid esters (PAEs) in the northwestern Mediterranean Sea and the Rhone River

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ABSTRACT

Phthalate acid esters (PAEs) which are mainly anthropogenic molecules with endocrine disrupting effects in animals and humans, have been detected in terrestrial and aquatic environments. However, little is known about their distribution in the Mediterranean Sea, mainly because of analytical difficulties and the high possibility of ambient sample contamination. Here, we report the optimization of an existing protocol for the determination of PAEs in seawater and freshwater samples, as well as the first estimation of the source and distribution of phthalates acid esters (PAEs) in coastal waters from the NW Mediterranean Sea. By passing 1 L of sample through glass cartridges packed with 200 mg of Oasis HLB and eluted with 6 mL of ethyl acetate followed by gas chromatography and mass spectrometry (GC/MS) analyses, the recoveries for DMP, DEP, DPP, DiBP, DnBP, BzBP, DEHP and DnOP were 101, 98, 115, 110, 99, 98, 103 and 95%, respectively, with acceptable blank values (below 0.4–4.0% of the masses measured in different seawater samples). By using this method, we detected PAEs in the Marseilles coastal area, offshore (2000 m depth) and in the Rhone River with total concentrations ranging from 75.3 ng/L offshore in surface water to 1207.1 ng/L a few meters above the bottom of the Marseilles Bay. High concentrations were also observed in deep waters offshore (310.2 ng/L) as well as in the Rhone River (615.1 ng/L). These results suggest that Marseilles urban area, Rhone River and sediment are potential sources of PAEs in the areas studied.

In the Rhone River, DEHP was the most abundant PAE (66.1%) followed by DiBP (20.5%) and DnBP (6.6%), whereas a gradual change was observed in the plume of the river with increasing salinity. In the Marseilles Bay, DiBP was the most abundant PAE at the surface (47.3% of total PAEs) followed by DEHP (22.7%) and DnBP (19.1%), whereas DnBP was predominant (38.6%) a few meters above the bottom. By contrast, DEHP was the dominant species in the first 50 m (66.9–76.7%) offshore followed by DnBP (10.9–15.2%), whereas DnBP was the most abundant (57.0–72.6%) followed by DEHP (20.1–33.1%) in the deepest waters. This study suggests that in addition to direct PAEs injection in marine waters, different processes regulate PAE distribution in Mediterranean Sea including photochemical oxidation reactions, bacterial degradation and possible diffusion following release from marine litter near the bottom.

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1. Introduction

Phthalates or phthalic acid ester (PAEs) with low volatility and low water solubility are colorless and odorless. They are mainly produced by anthropogenic activity such as the manufacture and processing of plastic products (Simoneit et al., 2005; Serôdio and Nogueira, 2006; Lyche et al., 2009). Their annual production reached 8 million tons in 2011 (Net et al., 2015a, 2015b and references therein) and they represent approximately 70% of the world

consumption of plasticizer in 2014 (CEH, 2015). The structure of PAEs consists of a di-alkyl ester of *o*-benzenedicarboxylic acid. The most commercially significant PAEs, sorted by increasing number of carbons in the ester side chains, are dimethyl phthalate (DMP), diethyl phthalate (DEP), di-*n*-butyl phthalate (DnBP), benzylbutyl phthalate (BzBP), di-ethylhexyl phthalate (DEHP) and di-*n*-octyl phthalate (DnOP) the later being the most abundant (Holahan and Smith, 2015). PAEs of lower molecular weight are used as essential components of solvents, adhesives, waxes, pharmaceutical products, insecticide materials and cosmetics, whereas PAEs of higher molecular weight are instead used to improve the flexibility and handling of industrial materials (IARC, 2000;

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Halden, 2010). PAEs are mixed with the polymers and are not covalently bound to them. During aging, they are likely to migrate out of the plastic and to be released directly into the environment or inside an animal's stomach or tissue (Andrady, 2011).

As widely used compounds, PAEs are ubiquitous in terrestrial, marine and atmospheric environments (Xie et al., 2007; Fu et al., 2013; Dargnat et al., 2009; Gao et al., 2014; Staples et al., 1997; Net et al., 2014; Net et al., 2015a, 2015b; Wang et al., 2014). PAEs have been detected in surface seawater (Turner and Rawling, 2000; Horn et al., 2004; Brossa et al., 2005), in indoor air (Becker et al., 2004), in urban environments in soils (Ferreira and Morita, 2012), in urban lakes (Zeng et al., 2008), in drinking water (Yang et al., 2014), in terrestrial and marine remote aerosols (Xie et al., 2006, 2007; Fu et al., 2008, 2009, 2013), and in marine sediment, as well as in biota, including several aquatic species (Wofford et al., 1981), algae (Gao and Chi, 2015), and the tissues of mammals (Staples et al., 1997) and humans (Liou et al., 2014). PAEs can be degraded by some microalgal species (Gao and Chi, 2015), fish and shellfish (Munshi et al., 2013), bacteria (Chang et al., 2004) and intracellular and extracellular algal enzymes (Gao and Chi, 2015) as well as by the combined action of UV and H₂O₂ (Xu et al., 2007). PAEs are of particular concern to humans since they are considered as emerging environmental contaminants due to the endocrine disruption and carcinogenic effects detected in laboratory animal experiments and in humans (Crisp et al., 1998; Latini, 2005; Kamrin, 2009; Meeker et al., 2009).

The Mediterranean Sea is a semi-enclosed basin covering an area of 2.5 million km² with a slow turnover time of ~80 years implying a sensitive response to anthropogenic impact (The Mermex group, 2011). Urbanization has been growing particularly along the coastline, with the result of a substantial modification of the coast and adverse effects on the quality of the environment (Pergent et al., 2012). There is a large range of industrial activities spread all along the Mediterranean basin, and a number of highly industrialized spots that are concentrated mainly in the northwestern Mediterranean Sea (Zorita et al., 2007). All these activities constitute a source of pollution through direct disposal, continental runoff and atmospheric transport (UNEP Chemicals, 2002; The Mermex group, 2011 and references therein). These industrial, agricultural and domestic activities generate a large volume of wastewater, which provides nutrients, trace metals (Cossa et al., 2018) organic matter, persistent organic pollutants (POPs) and related anthropogenic contaminants such as polycyclic aromatic hydrocarbons (PAHs) and polychlorinated dibenzo-p-dioxins and dibenzofurans (PCDD/Fs) (The Mermex group, 2011; Berrojalbiz et al., 2011; Guigue et al., 2011, 2014; Castro-Jimenez et al., 2008, 2010, 2012) that impact the coastal marine ecosystems. The Mediterranean Sea is also impacted by marine litter and microplastics (Deudero and Alomar, 2015; Cozar et al., 2015; Faure et al., 2015; Pedrotti et al., 2016; Suaria et al., 2016; Schmidt et al., 2018), which can supply PAEs to surrounding waters during their degradation (Andrady, 2011). Although PAEs have already been detected in aquatic environments (Xie et al., 2005; Net et al., 2015a, 2015b), only a few studies have reported the occurrence of individual phthalates in the Mediterranean Sea (Sanchez-Avila et al., 2012; Brossa et al., 2005), and there is no study dealing with the distribution of a complete series of PAEs in the Mediterranean basin, including the sea and rivers, mainly because of analytical difficulties.

Indeed, PAE analysis of environmental samples is a challenging task because of the low concentrations, which is associated with the risk of contamination during sampling and processing in relation with their ubiquity in the laboratory atmosphere, tools, glassware and various containers (Net et al., 2015b and references

therein). For environmental waters, several extraction procedures have already been used, including liquid-liquid (EPA, 1996; Zeng et al., 2008; Li et al., 2013; Gao et al., 2014), solid-liquid (SLE), solid phase (SPE), magnetic (MSPE) dispersive (DSPE), solid phase micro extraction (SPME) (Zhang et al., 2013; Liou et al., 2014; Yang et al., 2014) and high volume sample techniques with PAD-2 resin columns and Soxhlet extractor (Xie et al., 2005). SPE appears to be an appropriate technique for seawater because of its ease of implementation, its ability to save time and solvent (compared to liquid-liquid extraction) and the decreased contamination (Zhang et al., 2013; Liou et al., 2014; Yang et al., 2014; Net et al., 2015b). However, the SPE protocol uses a tube made of polyethylene or polypropylene, which can release DiBP, DnBP and DEHP (Tienpont, 2004; Fasano et al., 2012; Bonini et al., 2008; Paluselli unpublished results). This technique has been used by a great number of reported protocols, concerning the type, amount and conditioning of the phase (Liou et al., 2014; Sanchez-Avila et al., 2011, 2012; Yang et al., 2014) and the sample volume as well as the volume and type of solvent (He et al., 2013; Sanchez-Avila et al., 2011, 2012; Yang et al., 2014; Liou et al., 2014). Previous protocols have shown a recovery variability ranging from 50–60% to 120–150% (Del Carlo et al., 2008; Wang et al., 2008; He et al., 2013; Sanchez-Avila et al., 2011), especially for DEHP, which ranges from 60–80% (He et al., 2013) to 151% (Sanchez-Avila et al., 2011).

The objectives of this study are to examine, for the first time, the concentration levels and distribution of a homologous series of PAEs in the coastal northwestern Mediterranean Sea. This environmental study was associated with an optimization of a detection protocol for PAEs at trace level in seawater. Based on previous studies (Liou et al., 2014; Gao and Chi, 2015; Yang et al., 2014), conditions for the SPE extraction of PAEs have been modified and optimized for seawater. The method we report here was tested in seawater spiked with authentic standards and then applied to coastal and offshore Mediterranean seawater as well as to Rhone River freshwater in the framework of the 'Particule-MERMEX' project.

2. Experimental section

2.1. Materials and reagents

The selected phthalates are dimethyl phthalate (DMP), diethyl phthalate (DEP), dipropyl phthalate (DPP), di-isobutyl phthalate (DiBP), di-n-butyl phthalate (DnBP), benzylbutyl phthalate (BzBP), di-ethylhexyl phthalate (DEHP) and di-n-octyl phthalate (DnOP). The deuterated internal standards were DEP-d₄, DnBP-d₄ and DEHP-d₄. All native and labeled standards were of high purity grade (>98%) and were purchased from Supelco (Bellefonte, USA). Stock solutions were prepared in hexane and stored in the dark at 4 °C. Working solutions (unlabeled and labeled standards) were prepared by dilution of these solutions to 20 mg/L. All solvents were glass-distilled grade and supplied by Rathburn Chemicals Ltd. (Walkerburn, UK). Ultra-pure water was produced on-site by a Milli-Q system, Millipore (Molsheim, France) with a specific resistivity of 18.2 MΩ·cm (25 °C) and a total organic carbon content of <2 ppb. The extractions were carried out in controlled air conditions in an ISO class 6 cleanroom (temperature: 22 °C; SAS pressure: +15 Pa; SAS brewing rate: 30 vol/h; lab pressure: +30 Pa; brewing rate: 50 vol/h). All material was produced in borosilicate glass and that was previously kept in an acid bath overnight (10% hydrochloric acid), rinsed with ultrapure water, combusted at 450 °C for 6 h and rinsed with methanol and dichloromethane just before use.

2.2. Solid phase extraction (SPE)

The protocol was adapted from that of Yang et al. (2014) and Liou et al. (2014) and optimized for seawater in this study. The mentioned methods were modified in order to perform easy and rapid analysis on a large-scale with low contamination and to improve the recovery. The amount of Oasis HLB phase, the elution solvent type and volume, the sample volume, deuterated internal standards and glass materials were considered to improve the recoveries. Extraction cartridges were prepared in our laboratory with 200 mg of Oasis HLB sorbent (Waters Corporation, 30 μm , 100 g) in a precombusted 6 mL glass reaction tube equipped with methanol-precleaned Teflon frits (Supelco, Sigma-Aldrich). SPE was conducted on a Visiprep vacuum manifold from Sigma-Aldrich (Saint Quentin Fallavier, France). Before sample extraction, the SPE cartridges were sequentially cleaned three times with 5 mL of acetone and 5 mL of dichloromethane. The cartridges were then conditioned with 5 mL of ethyl acetate, 5 mL of acetone and 5 mL of ultrapure water prior to sample processing. Seawater samples were spiked with DEP-d4, DnBP-d4 and DEHP-d4 at 50, 200 and 400 ng/L, respectively, and percolated onto the cartridges under vacuum at a flow rate of 1–2 drops per second. After percolation, the cartridge was washed with 5 mL of ultrapure water to remove the remaining salt and dried under vacuum for 1 h. The retained phthalates were then eluted with 2×3 mL of ethyl acetate into precombusted (450 $^{\circ}\text{C}$, 6 h) 10 mL vials. Ethyl acetate was then gently evaporated under a gentle stream of nitrogen (purity >99,995%) at room temperature to a final volume of 200 μL . The final samples were immediately closed with Teflon-lined screw caps, wrapped with Teflon ribbon and stored in the dark at -20 $^{\circ}\text{C}$ until analysis.

2.3. GC–MS analysis

Analysis was performed using an Agilent Technologies 6850 GC system coupled to an Agilent Technologies 5975C mass spectrometer (GC/MS) operated with electron impact ionization (70 eV). Chromatographic separation was achieved on an Agilent HP-5MS capillary column (30 m \times 0.25 mm, 0.25 μm film thickness) with the GC oven programmed at 70 $^{\circ}\text{C}$ for 1 min and then up to 230 $^{\circ}\text{C}$ at 30 $^{\circ}\text{C}/\text{min}$, to 250 $^{\circ}\text{C}$ at 8 $^{\circ}\text{C}/\text{min}$ and to 280 $^{\circ}\text{C}$ at 30 $^{\circ}\text{C}/\text{min}$ with a final isothermal hold for 7.50 min. Helium was used as the carrier gas at a flow rate of 1.1 mL/min. The sample was manually injected (2 μL) in splitless mode at an injector temperature of 250 $^{\circ}\text{C}$. A Merlin Microseal injector was used as a low carryover septum and a gas purifier (Charcoal, CP17972) was fitted on the carrier gas line to prevent contamination during injection. Data were acquired in selected ion monitoring (SIM) mode for increased sensitivity (SIM parameters given in Table 1) with a dwell time of 100 ms. Two fragment ions were monitored per

compound, with the most abundant ion used for quantification, except for DMP, DPP, DiBP and DnBP, which were ionized in a unique fragment. Data were collected and analyzed with the Agilent ChemStation software. The PAEs were quantified by isotopic dilution of 3 deuterated PAEs. The use of internal standard homologues to the target molecules has been shown to improve the recoveries by accurately compensating for loss of analyte during sample treatment (Aminot et al., 2015). The response factors were assessed for each analytical sequence from freshly prepared standards.

2.4. Application to seawater and freshwater samples

Seawater samples were collected from SUNMEX station in the northwestern Mediterranean Sea, Bay of Marseilles (43 $^{\circ}$ 18'N, 05 $^{\circ}$ 22'E) in October 2014 and offshore at 'Antares station' (42 $^{\circ}$ 47'856N; 6 $^{\circ}$ 04'395E) in March 2015 (Fig. 1), with a 12 L GO-Flo bottle previously rinsed with 1% hydrochloric acid and ultrapure water to avoid contamination. The Go-Flo bottle tested with Milli-Q water showed only trace levels of PAEs which were not subtracted from our dissolved PAEs concentrations in this study. Immediately after collection, seawater samples were transferred into 5 L precombusted glass bottles. Additional samples were directly collected in the Rhone River and Rhone River plume in March 2014 using 5 L precombusted glass bottles. General information on the Marseilles Bay, the Antares station and the Rhone River were given by Sempéré et al. (2015), Martini et al. (2016) and Sempéré et al. (2000), respectively. Less than 2 h after collection, the samples were filtered through precombusted (450 $^{\circ}\text{C}$ for 6 h) GFF filters (47 mm filter diameter, rinsed with 2 L of Milli-Q water and 150 mL of sample prior to filtration) in a precombusted glass apparatus directly in a filtered-air clean laboratory. Milli-Q water rinse limit additional contamination (Yoro et al., 1999). After filtration, duplicate subsamples (10 mL) were poisoned with sulfuric acid to approximately pH 2 to avoid any biological activity and stored in the dark in precombusted Teflon-capped glass vials for dissolved organic carbon (DOC) analysis. DOC concentrations were measured using a Shimadzu TOC-5000 carbon analyzer (Sempéré et al., 2008). Some seawater samples (Bay of Marseilles) were spiked with authentic PAEs (from 5 to 800 ng/L final concentration) to perform recovery experiments. All the seawater analyses were conducted in duplicate in the laboratory in Marseilles.

3. Results and discussion

3.1. Extraction of PAEs in seawater samples (optimization of the method)

Numerous factors are known to significantly influence the efficiency of SPE extraction of phthalates from water. The most

Table 1
Retention time and ions for selected ion monitoring (SIM) of PAEs by GC/MS.

Compound	Molar mass (g mol^{-1})	Retention time (min)	Quantifier ion (abundance)	Qualifier ion (abundance)
DMP	194.2	5.55	163 (100)	194, 135, 77
DEP	222.2	6.12	149 (100)	222, 121
DPP	250.0	6.84	149 (100)	209, 191
DiBP	278.4	7.22	149 (100)	223, 205
DnBP	278.4	7.65	149 (100)	223, 205
BzBP	312.4	9.72	149 (100)	206, 238
DEHP	390.6	10.62	149 (100)	279, 167
DnOP	390.6	11.76	149 (100)	279
DEP-d4	226.2	6.11	153 (100)	181
DnBP-d4	282.4	7.64	153 (100)	209
DnBP-d4	282.4	7.64	153 (100)	209
DEHP-d4	394.6	10.61	153 (100)	171, 283

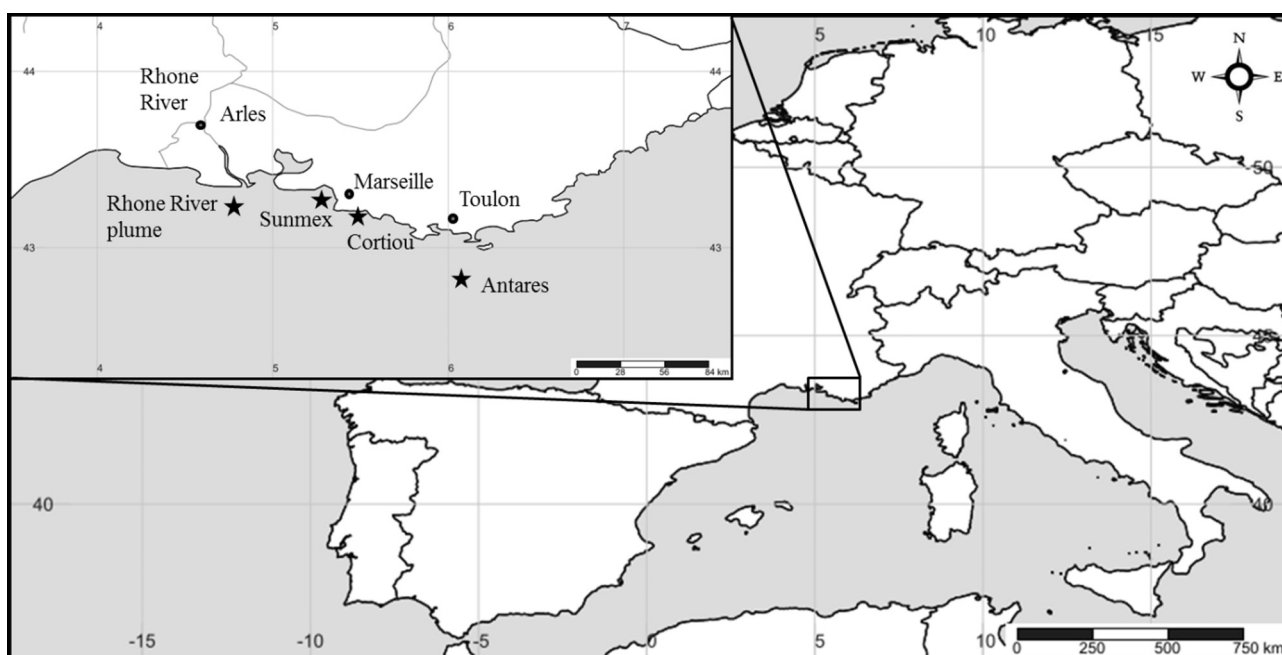


Fig. 1. Area of study. The map shows the sampling stations: SUNMEX in the Bay of Marseilles (October 2014), offshore 'Antares station' (March 2015), Arles in Rhone River and the river plume (March 2014).

influential factors are elution the solvent, sample volume and sorbent mass. To increase the recovery, the elution solvent, sample volume and sorbent mass were studied in order to optimize efficient and reliable conditions for SPE extraction of seawater samples.

3.1.1. Elution solvent effect

First, 10 μL of 20 mg/L PAE solution was passed through the cartridges, and two fractions of elution were collected. The first fraction was eluted with 5 mL of either acetone, dichloromethane, ethyl acetate, methanol or *n*-hexane, and the second fraction was systematically eluted with 5 mL of dichloromethane in order to elute the remaining PAEs off the cartridge. Internal standards were spiked directly in the eluted fraction. The repartition of PAEs between the two fractions is given in Fig. 2. Acetone, dichloromethane, ethyl acetate and methanol yielded acceptable recoveries ranging from 55 to 100% while *n*-hexane was satisfactory only for DEHP and DnOP (79–85%), with 12% for DMP, 36% for DEP and 1% for BzBP. *n*-Hexane is a better solvent for aliphatic compounds, unlike DMP, DEP (short carbon chain) and BzBP (higher aromaticity).

Acetone and methanol showed a similar elution strength with recoveries of 80–100% for DMP, DEP, DPP, DiBP, DnBP and BzBP in the first fraction and 70% and 50% for DEHP and DnOP, respectively. This result is consistent with the studies of Liou et al. (2014) and Yang et al. (2014), which performed the elution with 6 mL of methanol. Dichloromethane and ethyl acetate both indicated high recoveries for all PAEs in the first fraction and were the most appropriate elution solvents. Dichloromethane was the most appropriate elution solvent for DMP (100%), DEP (98%) and DPP (97%), whereas ethyl acetate was the most appropriate for the elution of DiBP (91%), DnBP (92%), BzBP (85%) and DEHP (90%). DnOP was efficiently eluted (100%) with both solvents. Previously, published protocols used 10 mL of dichloromethane/*n*-hexane (He et al., 2013) or 10 mL of acetone/dichloromethane (Sanchez-Avila et al., 2011, 2012) as the elution solvent. When comparing their recoveries and those from our developed protocol, the combination of dichloromethane with acetone seems to be more effective for PAE elution than its combination with *n*-

hexane. In these protocols, the ability to elute PAEs from the cartridges was improved by increasing the elution solvent volume to 10 and 20 mL, respectively. According to our results, ethyl acetate is the best solvent for PAE elution.

3.1.2. Seawater volume and sorbent mass effects

As high SPE enrichment factors are required to compensate for the likely low PAE levels in seawater, it is important to ensure that the analytes do not breakthrough the sorbent at high percolation volumes. Increasing volumes (100 mL, 200 mL, 500 mL and 1000 mL) of seawater collected at Sunmex station at a 1.5 m depth were spiked with 10 μL of 20 mg/L PAEs and percolated through either 200 mg (Liou et al., 2014) or 400 mg of sorbent. For each experimental condition, two additional samples were kept unspiked and were processed in the same conditions (blank samples). The recoveries were calculated as the difference between the concentration in the spiked and unspiked samples. In this experiment, the deuterated internal standards DEP-d4, DnBP-d4 and DEHP-d4 were spiked after the elution of the solvent. The recoveries of the extractions obtained with 4 different volumes of spiked seawater for 200 and 400 mg of HLB sorbent are given in Table 2.

The volume of the sample does not significantly affect the recovery of most of the PAEs, indicating the absence of breakthrough. Only DEP and DnBP were marginally affected by increasing sample volumes. The recovery of DEP increased from 60% with 100 mL to 93% with all other volumes on 200 mg of sorbent. However, the recovery was stable between 93 and 100% on 400 mg. DnBP showed the opposite trend, where the recovery decreased from 75–79% with 100–200 mL to 63–66% with 500–1000 mL on both 200 mg and 400 mg of sorbent. Acceptable recoveries (93–120%) were determined for DMP, DPP, DiBP and BzBP under all experimental conditions. The two most hydrophobic PAEs DEHP and DnOP exhibited lower recoveries regardless of the experimental conditions, attributed to their potential for adsorption onto glassware. Such losses were corrected for by the use of deuterated DEHP as an internal standard (Table 3).

Above all, the volume of the sample did not significantly influence the performance of the HLB sorbent toward PAEs. Similar

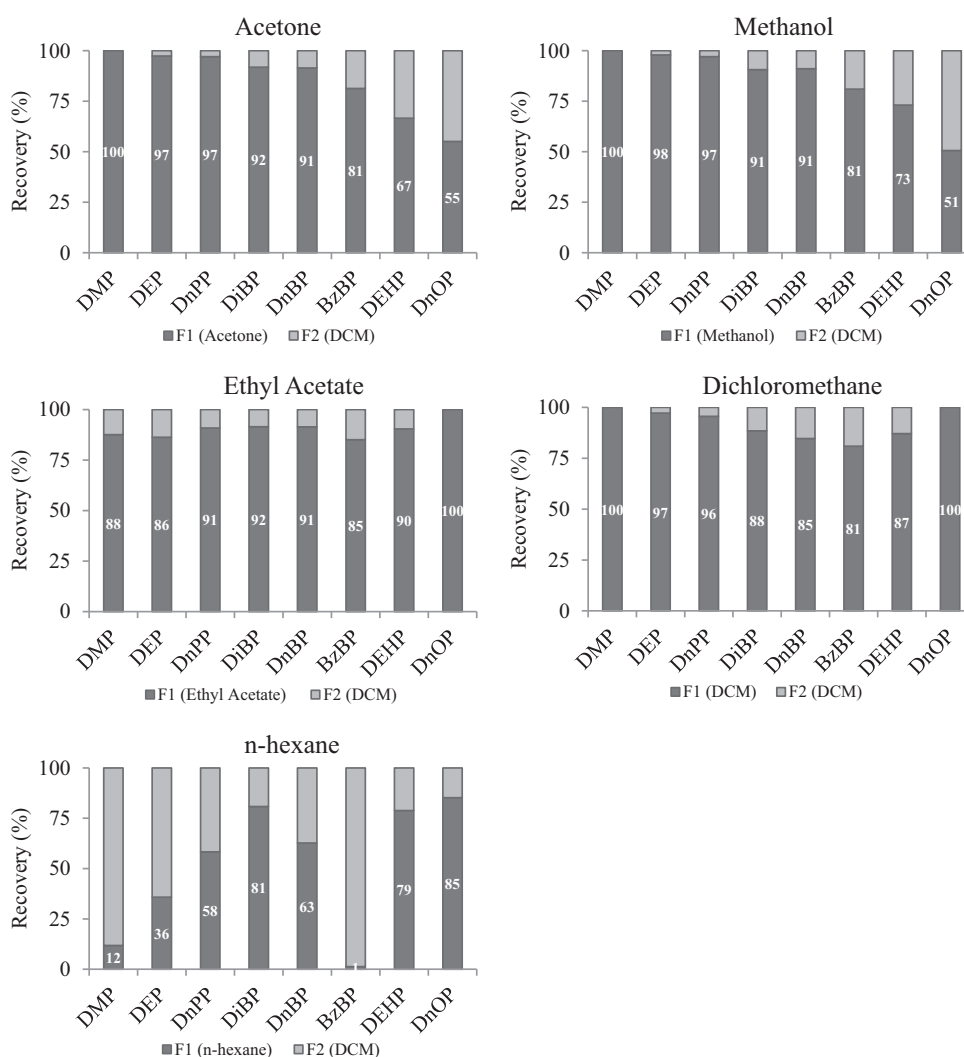


Fig. 2. Recovery of PAEs obtained in two elution fractions carried out with five different organic solvents (F1) and dichloromethane (F2).

Table 2

Recoveries of PAE extractions obtained with 4 different volumes of spiked seawater and 200 and 400 mg of HLB sorbent.

Compounds	Recoveries (%)							
	200 mg				400 mg			
	100 mL	200 mL	500 mL	1000 mL	100 mL	200 mL	500 mL	1000 mL
DMP	99	108	100	105	114	107	109	110
DEP	61	93	88	89	95	93	95	100
DPP	105	117	113	120	122	121	119	114
DiBP	96	122	111	123	121	100	107	115
DnBP	79	75	63	66	136	77	67	82
BzBP	102	125	94	106	126	104	87	97
DEHP	14	13	8	10	11	7	21	6
DnOP	8	10	15	10	10	6	24	6

results were observed for the sorbent mass. The DnOP concentration in seawater is expected to be low, indicating the need to increase the enrichment factor by extracting 1 L of sample. BzBP is also characterized by low concentrations in seawater and by relatively high instrumental detection limits, confirming the need for a large volume of sample. As the differences between 200 and 400 mg of sorbent did not justify the use of double the amount of sorbent, 200 mg of sorbent was selected to reduce the price and time for a single extraction. Previously, published protocols reported the use of 1 L of matrix, but different amounts of solid phase, from 200 mg (Liou et al., 2014) to 500 mg (Yang et al.,

2014). According to the results, our optimized conditions were 200 mg of HLB phase prepared in a combusted glass tube, 1 L of sample volume, and 6 mL of ethyl acetate as the elution solvent, split in two steps.

3.2. Method validation

3.2.1. Blank, response linearity and extraction validation

Procedural blank samples were examined by passing dry air and ethyl acetate directly through the pre-cleaned cartridge and measuring the PAEs directly in the elution solution. Linearity was

Table 3

Statistical data of the linear regression between the spiked concentration of PAEs and the measured peak areas. PAE recovery was reported.

Compound	Spiked concentration (ng/L)	n	R ²	Slope (μVs/ng/L)	y-intercept (μVs)	Recovery (%)	MDL (ng/L)
DMP	0, 5, 10, 20, 40	5	0.98	11,022	20,671	101.2	0.11
DEP	0, 5, 10, 20, 40, 100	6	0.99	13,978	40,049	98.1	0.21
DPP	0, 5, 10, 20, 40	5	0.98	12,072	2893	115.5	0.31
DiBP	0, 40, 100, 200, 400, 600, 800	7	0.99	11,006	5,477,511	110.4	0.23
DnBP	0, 40, 100, 200, 400, 600	6	0.99	19,424	1,965,660	99.7	0.28
BzBP	0, 5, 10, 20, 40	5	0.99	2718	1845	98.7	1.67
DEHP	0, 40, 100, 200, 400, 600, 800	7	0.99	1638	23,076	103.8	0.93
DnOP	0, 5, 10, 20, 40	5	0.96	268	323	95.0	2.73

investigated over an eight-point calibration ranging from 50 to 6000 pg injected. Instrumental repeatability was measured through the relative standard deviation of 10 injections of the same standard solution, corresponding to 100 pg injected, whereas instrumental detection limits (IDLs) were extrapolated for a signal-to-noise ratio of 3. The noise region was equivalent to the width of the peak before the peak, and the noise value was taken as the average height ("RMS" mode). The recoveries were calculated from the spiked seawater samples (10 μL of 20 mg/L of PAEs in 1 L) after subtraction of the concentrations measured for the unspiked seawater samples (n = 2). Such experiments were repeated with half the spiked concentration (5 μL of 20 mg/L of PAEs) in seawater samples (n = 2) to study the influence of the concentration level.

The method was applied to 1 L seawater samples to minimize contamination and to allow the detection of low concentrations of BzBP and DnOP in seawater. The samples were passed through 200 mg of HLB sorbent, and the cartridges were eluted twice with 3 mL of ethyl acetate. A typical gas chromatogram of the PAEs isolated from the seawater samples collected in the Marseilles Bay is given in Fig. 3. Identification of the PAEs was confirmed by GC/MS. Despite the precautions taken to prevent contamination, DEP, DiBP and occasionally DnBP were detected in the blanks at levels that remained below 0.4–2%, 2–3% and 0–4% of the masses measured in different seawater samples, respectively. The average recoveries of PAEs spiked in the seawater samples (Table 3) showed accept-

able recoveries for all the phthalates, ranging from 97% of DEP to 110% of DiBP. The high recoveries obtained at low spiking levels also validated the appropriate performance of the protocol. To assess the influence of the concentration level on the recoveries, 1 L aliquots of seawater collected in the Bay of Marseilles were spiked at increasing concentration levels (standard addition technique). Different ranges of spiked concentrations were used depending on the initial concentration of the compound in seawater (from 5 to 40 ng/L for DMP, DEP, DPP, BzBP and DnOP and from 40 to 800 ng/L for DiBP, DiBP and DEHP). Excellent correlation coefficients ($R^2 > 0.96$) and recoveries (between 95 and 116%) indicate that the method is linear and accurate for all compounds, including DEHP ($R^2 = 0.987$; n = 7) and DnOP ($R^2 = 0.962$; n = 5), in a range of environmentally realistic concentrations.

3.3. Concentration of PAEs in the northwestern Mediterranean Sea and Rhone River

All PAEs, excepted DPP and DnOP, were detected in the Mediterranean seawater and Rhone River samples (Table 4). We found that offshore (Antares site), the concentration of total PAEs ranged from 75.3 ng/L at the surface to 310.2 ng/L in the deepest water, from 244.5 ng/L to 1207.1 ng/L a few meters above the bottom in the Marseilles Bay and from 182.2 to 212.6 ng/L in the Rhone River plume and close to the Marseilles Cortiou water treatment plant outlet. Although particulate PAEs were not measured in this study,

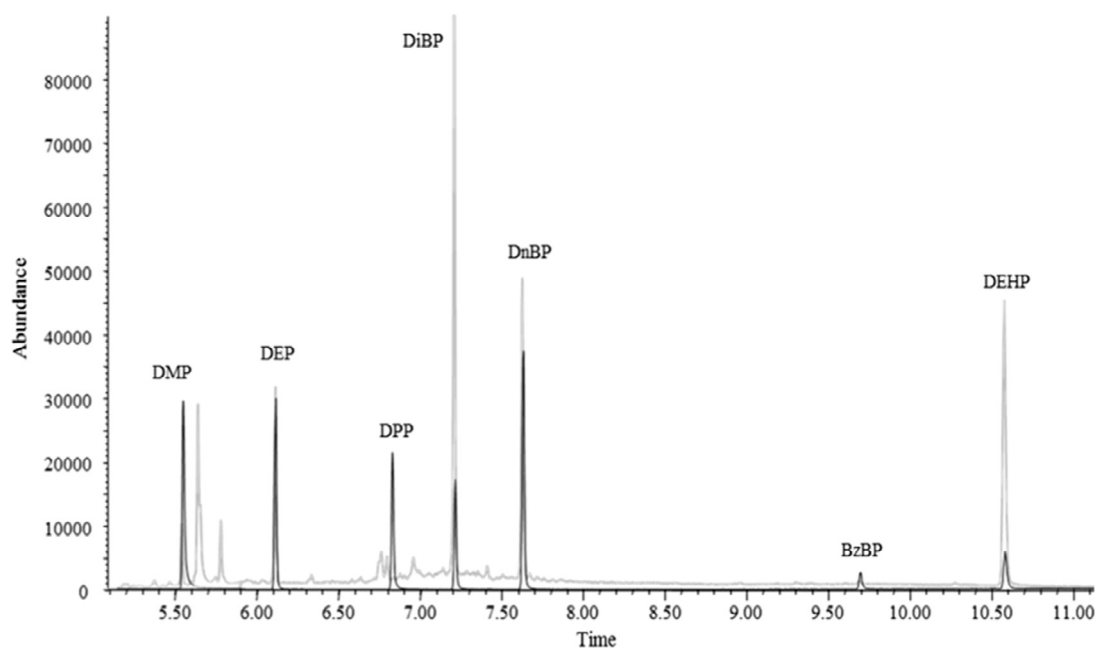


Fig. 3. Typical capillary gas chromatogram of PAEs (DMP, DEP, DiBP, DnBP, BzBP and DEHP) isolated from the seawater sample (grey line) collected from the Marseilles Bay (northwestern Mediterranean Sea) in October 2014 and typical PAEs standard solution chromatogram (black line, 0.1 mg/L). The identification of PAEs was confirmed by GC/MS. The mass spectra of the PAEs (not presented here) were in accordance with those published in the literature. DPP and DnOP were not detected in this sample.

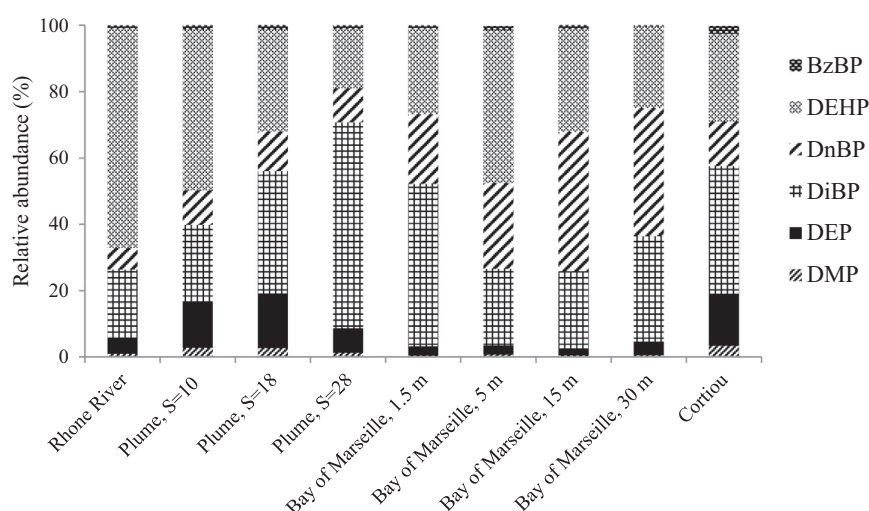


Fig. 4. PAE relative abundances (%) detected in different Mediterranean Sea samples: Surface Rhone River and associated Plume (March 2014), Bay of Marseilles (October 2014) and surface 'Cortiou' station (November 2014).

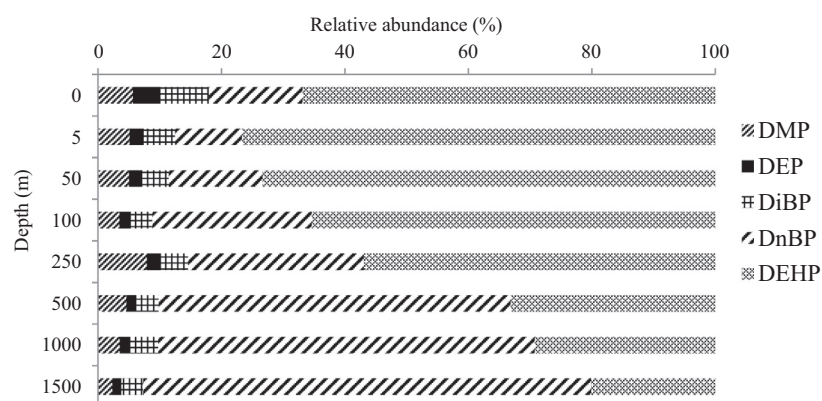


Fig. 5. PAE relative abundances (%) detected in offshore Antares station-samples collected at the surface and at depths of 5, 50, 100, 250, 500, 1000 and 1500 m in March 2015.

the relative abundance of DEHP was slightly lower (56.9–65.4%), whereas that of DnBP was higher (25.8–28.5%) up to the maximum salinity and minimum temperature. Under 500 m, DnBP was the most abundant PAE (57.0–72.6%) followed by DEHP (20.1–33.1%). DiBP, DMP and DEP were detected as minor fractions of the total PAEs detected in offshore waters, representing 7.9, 5.7 and 4.4% of total PAEs at the surface and 3.5, 2.3 and 1.5% in deepest waters, respectively (Fig. 5; Table 6). Note that this offshore station area is influenced by the shallow depth Northern Current (NC) with high seasonal variability (Millot, 1991), which may penetrate into the shelf of the Gulf of Lions (Barrier et al., 2016 and references therein).

This water originating eastward from the Italian coasts may carry plastic debris (Schmidt et al., 2018) and dissolved PAEs. The decrease in concentration in the first meter of the water column could be explained by biodegradation processes. The increase in concentration at 200–500 m suggests the production of PAEs from the aging of plastic debris, as was observed in Marseilles Bay. Indeed, the behavior of DnBP in the deepest water seems to reflect the same production observed close to the bottom of the Marseilles area. These data were the first attempt to investigate the vertical distribution of PAEs in the water column.

3.3.1. Rhone River and river mouth

The total PAE concentrations ranged from 615.1 ng/L to 182.2–220.0 ng/L in the Rhone River plume (Table 4) suggesting that the Rhone is clearly a source of PAEs in the coastal area. Previous studies dealing with individual PAEs reported DMP concentrations in the range of 2.5–5.5 ng/L (Sanchez-Avila et al., 2012) in six rivers in Spain (Muga, Fluvià, Ter, Besòs, Llobregat and Ebro River). DEP was detected in different studies in the Seine River (France) in a range slightly higher than our data, from 46–208 ng/L (Teil et al., 2007) to 71–181 ng/L (Dargnat et al., 2009), as well as 52–284 ng/L in Spain (Sanchez-Avila et al., 2012). BzBP was also reported at low concentration in the range of 5–23 ng/L (Dargnat et al., 2009; Sanchez-Avila et al., 2012). Dargnat et al. (2009) and Teil et al. (2007) found similar concentrations of DEHP, 161–314 ng/L and 323–779 ng/L, respectively, in the Seine River. We found that DEHP was the most abundant PAE (66.1%) in the Rhone River followed by DiBP (20.5%) and DnBP (6.6%), whereas DMP, DEP and BzBP represented the smallest fractions (Table 5; Fig. 4). The relative abundance of DEHP decreased from 66.1% of the total PAEs in the river ($S = 0$) to 48.4, 30.7 and 17.8% in the plume stations ($S = 10–28$). The opposite trend was observed for all other PAEs, especially for DnBP and DiBP, with DnBP values increasing from 6.6 to 11.9% and from 20.5 to 62.0% for DiBP. A similar trend was

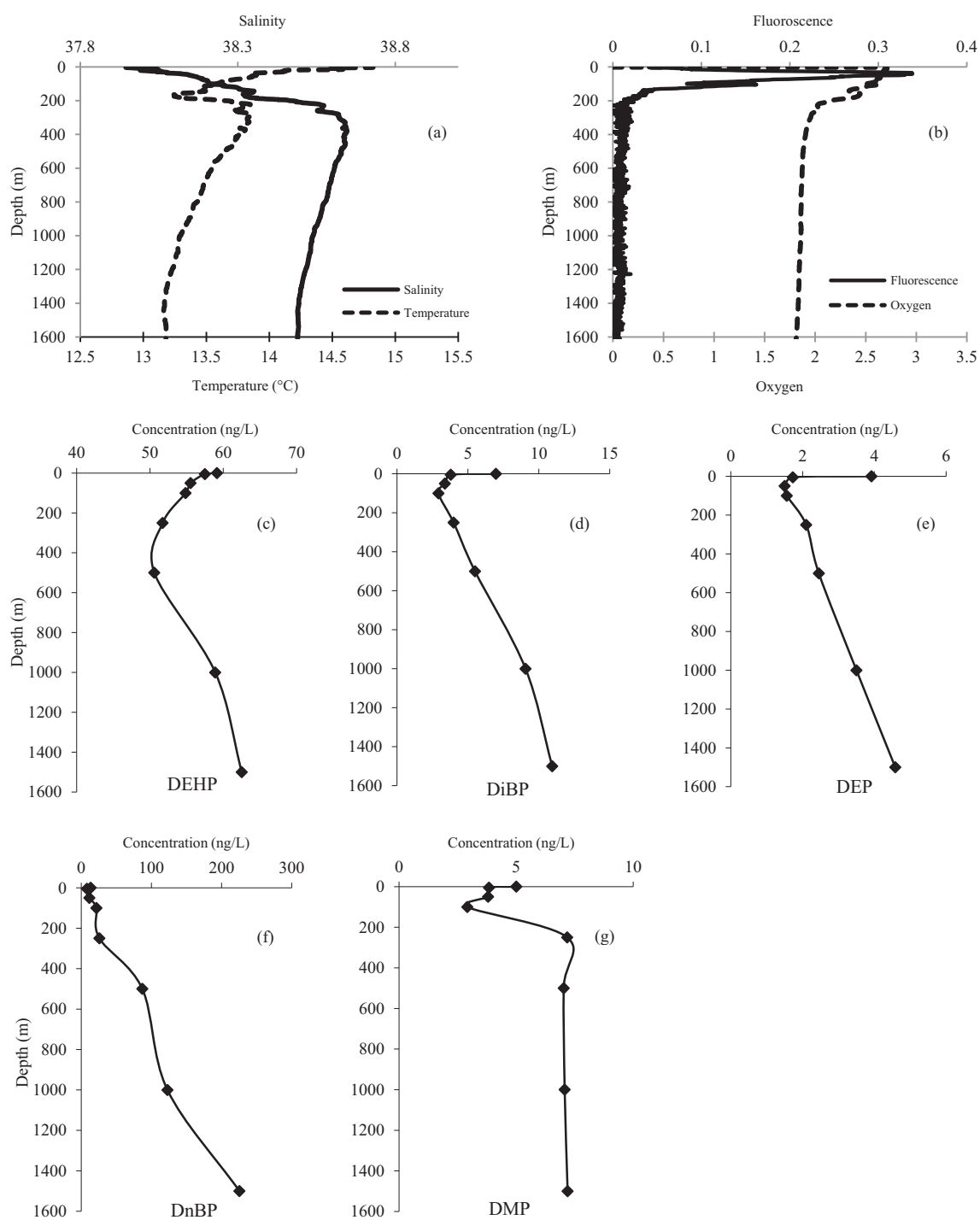


Fig. 6. Vertical profile (0–1500 m) of 5 PAEs, along with salinity, temperature (a), fluorescence and oxygen (b) detected at offshore Antares station in March 2015. The compounds detected were: (c) di-ethylhexyl phthalate (DEHP), (d) di-iso-butyl phthalate (DiBP), (e) diethyl phthalate (DEP), (f) di-n-butyl phthalate (DnBP) and (g) dimethyl phthalate (DMP).

Table 6
PAE relative abundances (%) detected in offshore Antares station-samples collected at the surface and at depths of 5, 50, 100, 250, 500, 1000 and 1500 m in March 2015.

Compounds	Depth (m)							
	0	5	50	100	250	500	1000	1500
DMP	5.7	5.1	5.0	3.5	7.9	4.6	3.5	2.3
DEP	4.4	2.3	2.0	1.9	2.3	1.6	1.7	1.5
DiBP	7.9	5.1	4.5	3.5	4.4	3.6	4.5	3.5
DnBP	15.2	10.9	15.2	25.8	28.5	57.0	61.0	72.6
DEHP	66.9	76.7	73.3	65.4	56.9	33.1	29.2	20.1

also observed for DMP and DEP. Changes in the relative abundance from the river to the plume could be due to different selective bacterial degradation and/or photodegradation processes occurring in the salty water. Only DiBP showed, after the concentration first decreased probably due to dilution effect, an increase in concentration and in relative abundance. This could be caused by some atmospheric input of this compound as it was already supposed in the Bay of Marseilles.

4. Conclusion

A reliable analytical technique for measuring PAEs in natural seawater samples at trace levels was slightly improved and allowed determination of PAEs in seawater. This technique consists of the adsorption on PAEs from acidified seawater samples onto 200 mg Oasis HLB glass cartridges followed by elution with ethyl acetate and determination by GC/MS. The recoveries of the PAEs from spiked seawater samples were found to be between 95% and 115%, and the contamination was strongly reduced by using a number of precautions. Our results showed that the Rhone River was a source of PAE in estuary and in coastal area. Although DEHP was predominant in the river, its relative abundance decreased with increasing salinities, DiBP being predominant in the plume for a salinity of 28. We reported, for the first time, the PAE vertical profile (0–1500 m) from the offshore water, which had lower concentrations compared to the Bay of Marseilles. We observed that DEHP was also predominant in surface marine waters including offshore (0–250 m) and below the surface in Marseille Bay (5 m). The opposite trend was found for DnBP for which concentrations significantly increased in mesopelagic waters. High concentrations of DnBP, DEHP and DiBP were detected close to the bottom (30 m) of the Bay of Marseilles. Such feature might be related to diffusion of PAEs released from the sediment surface very likely induced from plastic debris degradation and previously accumulated in the bottom. Precise origin of such material still needs to be determined. Other processes such as photochemical oxidation reactions in surface waters, bacterial degradation, are certainly important processes that regulate as well the distribution of PAEs in the water column. High PAE concentrations found in coastal area near Marseilles city strongly suggest that these organic compounds are very likely abundant in the whole Mediterranean that is surrounded by urbanized areas and that is submitted to relatively high inputs of rivers and atmospheric deposition compared to the global Ocean.

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