Potential risk of BPA and phthalates in commercial water bottles: a minireview

Rouse da Silva Costa, Tatiana Sainara Maia Fernandes, Edmilson de Sousa Almeida, Juliene Tomé Oliveira, Jhonyson Arruda Carvalho Guedes, Guilherme Julião Zocolo, Francisco Wagner de Sousa and Ronaldo Ferreira do Nascimento

ABSTRACT

The global water bottling market grows annually. Today, to ensure consumer safety, it is important to verify the possible migration of compounds from bottles into the water contained in them. Potential health risks due to the prevalence of bisphenol A (BPA) and phthalates (PAEs) exposure through water bottle consumption have become an important issue. BPA, benzyl butyl phthalate (BBP), di-*n*-butyl phthalate (DBP) and di (2-ethylhexyl) phthalate (DEHP) can cause adverse effects on human health. Papers of literature published in English, with BPA, BBP, DBP and DEHP detections during 2017, by 2019 by liquid chromatography and gas chromatography analysis methods were searched. The highest concentrations of BPA, BBP, DBP and DEHP in all the bottled waters studied were found to be 5.7, 12.11, 82.8 and 64.0 μg/L, respectively. DBP was the most compound detected and the main contributor by bottled water consumption with 23.7% of the Tolerable Daily Intake (TDI). Based on the risk assessment, BPA, BBP, DBP and DEHP in commercial water bottles do not pose a serious concern for humans. The average estrogen equivalent level revealed that BPA, BBP, DBP and DEHP in bottled waters may induce adverse estrogenic effects on human health.

Key words | bisphenol A, estrogenic effects, phthalates, risk assessment, water bottles

HIGHLIGHTS

- DBP was the most compound detected.
- An estimated intake of BPA, BBP, DBP and DEHP was far below their TDIs.
- The risk assessment of BPA, BBP, DBP and DEHP does not raise serious concern for humans.
- The average estrogen equivalent level for BPA, BBP, DBP and DEHP may induce adverse estrogenic effects on human health.
- BPA, BBP, DBP and DEHP in bottled water need more accurate data to avoid their effects on human health.

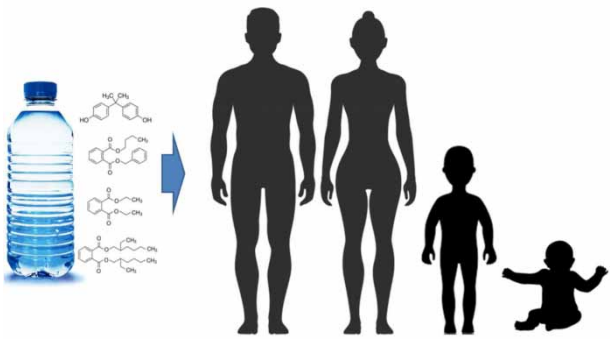
Rouse da Silva Costa
Tatiana Sainara Maia Fernandes (corresponding author)
Edmilson de Sousa Almeida
Juliene Tomé Oliveira
Jhonyson Arruda Carvalho Guedes
Ronaldo Ferreira do Nascimento
Department of Analytical Chemistry and Physical Chemistry,
Federal University of Ceará,
R. Humberto Monte S/N, 60455700 Fortaleza, CE,
Brazil
E-mail: *tatisainara@yahoo.com.br*

Jhonyson Arruda Carvalho Guedes
Guilherme Julião Zocolo
Embrapa Tropical Agroindustry,
R. Dra Sara Mesquita 2270, 60511-110 Fortaleza, CE,
Brazil

Francisco Wagner de Sousa
Department of Education – Chemistry Licenciate,
Federal Institute of Education, Science and Technology,
R. Francisco da Rocha Martins S/N, 61609-090 Caucaia, CE,
Brazil

doi: 10.2166/wh.2021.202

GRAPHICAL ABSTRACT



INTRODUCTION

Reports show that, in 2018, 64% of produced bottles were made of polyethylene terephthalate (PET), 34% of high-density polyethylene (HDPE), 1.8% of polypropylene and 1% other (polycarbonate (PC) included here) (ACC 2019). According to the American Chemistry Council (ACC), in 2018, 0.31 million pounds of postconsumer PC bottles were collected for recycling. PET and HDPE continued to dominate as selected resins to produce plastic bottles (97.1% by weight of produced bottles has made of PET or HDPE) (ACC 2019).

The bottled water industry is a phenomenon in practically every region of the world. First, bottled water became a mainstream commercial beverage category in Western Europe and later grew into a truly global beverage (IBWA 2018). The bottled industry produces mainly two types of packaged water: packaged natural mineral drinking water and packaged drinking water. The last is water derived from any source of a potable water (ground, well, bore well water, etc.), which must be subjected to different treatment processes such as filtration, aeration, decantation, and reverse osmosis (Jain *et al.* 2019). In 2018, for the first time, global bottled water consumption has surpassed that 100 billion gallons is estimated to, and the per capita consumption exceeded 42 gallons (158,987 liters). It should be stressing that per capita consumption by individual regions or countries can differ from the global average (IBWA 2019). In 2018, the rank of the 10 leading countries' consumption was China, United States, Mexico, Indonesia, Brazil, India, Thailand, Germany, Italy and France, respectively (IBWA 2019).

In 2018, approximately 7.7% (27.64 million tons out of the total plastic production of 359 million tons) of the plastic demand was constituted by PET worldwide was used in bottles for water, soft drinks, juices, and cleaners (Plastics Europe 2020). PET is the packaging most used in water bottles (Coniglio *et al.* 2020). PET and PC as the packing materials have been widely used for Chinese bottled water (Wang *et al.* 2020).

Bisphenol A (BPA), benzyl butyl phthalate (BBP), di-*n*-butyl phthalate (DBP) and di (2-ethylhexyl) phthalate (DEHP) have recently been detected in commercial water bottles raising concerns and discussions on possible risks for human health (Dada *et al.* 2018; Pinsrithong & Bunkoed 2018; Karayaka *et al.* 2019; Wu *et al.* 2019). Many countries included BPA, BBP, DBP and DEHP in the priority list of pollutants (Pignotti *et al.* 2017; Goeury *et al.* 2019; Li *et al.* 2019; Fard *et al.* 2020). Acceptable exposure levels for these compounds have been created to protect human health (Čelić *et al.* 2020; Fard *et al.* 2020). The maximum contaminant level (MCL) is the highest level of a contaminant that is allowed in drinking water (US EPA 2021). The MCL for drinking water for BPA, BBP, DBP and DEHP is in the section 'Extraction techniques for detection'.

According to Hassan *et al.* (2020), BPA and PAEs exhibit similar toxicogenomics and health effects. How BPA and PAEs are not bound to the matrix, they can leach out into

the surroundings by delicate changes in the environment, like temperature, pH and pressure alterations (Hassan *et al.* 2020). The Regulation (EU) No. 10/2011 (EC 2011) defines the Specific Migration Limit (SML) as the maximum permitted amount of a given substance released from a material or article into food or food simulants. The SML values by the EU for BBP, DBP and DEHP are 30, 0.3 and 1.5 mg/kg, respectively (EFSA 2019). The detection of very low BPA, BBP, DBP and DEHP in water can be carry out by high-performance liquid (HPLC) and gas (GC) chromatography (Gorji *et al.* 2019; Karayaka *et al.* 2019; Li *et al.* 2019; Yin *et al.* 2019). The detection power can be improved by preconcentrating analytes before instrumental measurement and the type of detector (Kumar *et al.* 2014; Chang *et al.* 2017; Farajzadeh *et al.* 2019; Karayaka *et al.* 2019; Li *et al.* 2019).

In this context, due to the increasing popularity of bottled water consumption, the potential health effects of possible migration of chemical compounds from the bottles into the water can pose a health risk to consumers. The purpose of this minireview is to verify if recent BPA, BBP, DBP and DEHP detections in commercial water bottles around the world using HPLC and gas GC may pose a risk to human health.

Papers of literature published in English, that detected BPA and PAEs (BBP, DBP and DEHP) in commercial bottles during 2017, by 2019 were searched. Papers with storage studies were also taken into account. For data sources for further analysis were identified a total of 41 publications from 17 countries. PC bottles were not considered. Thus, this work hopes to aid decision-making in future research focusing on BPA, BBP, DBP and DEHP in commercial water bottles using HPLC and GC. Moreover, this review hopes to avoid consumer exposure to these chemicals and to guarantee consumer safety.

BPA AND PAES IN PET BOTTLED WATER

The production process of water bottles uses PC plastics containing BPA (antioxidant or monomer) (Alfarhani *et al.* 2019; Fikarová *et al.* 2019; Liu *et al.* 2019). Although BPA is not used in the manufacture of PET, it should consider the use of recycled PET (R-PET) as a possible source of BPA coming from cross-contamination, not only during the recycling process but also during the manufacture of virgin PET (Dreolin *et al.* 2019). BPA leachable from polymer packaging due to its moderate water solubility (120–300 mg/L: pH 7.0 at 25 °C) and low log K_{ow} (3.32) in water (Borrirukwisitsak *et al.* 2012; Fikarová *et al.* 2019). Guart *et al.* (2011) not detected BPA in PET bottles cut in pieces, but on the other hand, detected BPA in HDPE caps at concentrations of 0.145 μg/dm^2. Bach *et al.* (2012) also indicated that the containers' caps, in PET bottled water, could be a source of BPA.

The manufacturing of beverage bottles widely uses PAEs (Li *et al.* 2019) and like they are not chemically bound to polymers, they may also enter drinking samples. This process can occur through the production, packaging and storage (bottling lines and water refinement centers) (Manzo *et al.* 2019; Pacyga *et al.* 2019). According to Bach *et al.* (2014), background pollution, as a source of PAEs, cannot be excluded. PAEs' presence in PET bottled water can be associated with PAEs in the source of water (groundwater or tap water) used to fill in the bottles (Jeddi *et al.* 2015). The type of closure ('cap') on the bottles could be a more important source of PAEs than the bottle material (glass or PET) (EFSA 2019). The caps of plastic bottles are made of high- and low-density polyethylene (HDPE and LDPE) and polystyrene (PS) (Guart *et al.* 2011). Guart *et al.* (2011) identified BPA in HDPE, LDPE and PS plastics. The adhesive used for sticking the bottle labels could thus be considered one of the sources of PAEs in water samples (Cincotta *et al.* 2018). Aznar *et al.* (2011) identified DBP and DEHP in adhesive based on vinyl acetate-ethylene.

PAEs are hydrophobic organic compounds under normal conditions (25 °C), very insoluble in water (BBP: 2.69 mg/L, DBB: 11.2 mg/L and DEHP: 0.27 mg/L) and have a particular affinity for fats and alcohols (Grinbaum *et al.* 2019; PubChem 2020). However, exposure to these low levels in water may also cause significant risks to humans under long-term chronic exposure by resulting in a considerable total health risk (Abtahi *et al.* 2019; Chen *et al.* 2019a; Abdelghani *et al.* 2020). Exposure to that low level can cause problems such as spasms in arms and legs, bronchial obstruction in children, irritation of the eyes and endocrine disruption (Abdelghani *et al.* 2020).

CHROMATOGRAPHIC AND EXTRACTION TECHNIQUES FOR DETECTION

Chromatographic techniques for detection

A wide range of methods analyzes BPA and PAEs. The liquid chromatography (LC) and gas chromatography (GC) analysis methods for detection and respective extraction techniques used for the determination in commercial water bottles are presented in Table 1. The choice of the detector and extraction influences the detection limit (LOD) and the quantification limit (LOQ) values obtained.

HPLC coupled with diode-array detection (HPLC–DAD) was the most used in BPA detections. HPLC is adequate for the analysis of BPA since it is a relatively polar compound. The DAD detector allows simultaneous collection of chromatograms over a range of wavelengths during a single run, providing more information on sample composition than is provided by the use of a single wavelength detector (Waksmundzka-Hajnos & Sherma 2010). DAD is preferable since it is sufficiently selective for compound identification (McGowin 2006).

GC coupled with mass spectrometry (GC–MS) was the most used technique in PAEs detections. GC can separate volatile and semi-volatile compounds with high resolution, and its combination with MS can identify them, providing detailed structural information on most compounds such that they can be identified correctly (Hussain & Maqbool 2014). Only Karayaka *et al.* (2019) analyzed BPA by GC–MS and derivatization is not used. BPA has volatility and thermal stability suitable for detection and quantification by GC–MS. However, derivatization can improve the sensitivity, selectivity and performance of the chromatographic properties (Nollet 2005).

BPA analysis underivatized by GC–MS can be found in the literature because sensitivity can be improved using preconcentration and liquid–liquid extraction (Oca *et al.* 2013). Karayaka *et al.* (2019) used the switchable liquid–liquid microextraction (SLLME) to preconcentrate BPA and improving the detection power of GC–MS. Microextraction methods are eco-friendly because they use too small quantities of chemicals, no compromising extraction efficiency and agree with green chemistry (Armenta *et al.* 2015).

Extraction techniques for detection

The sample preparation has been considered as the Achilles' heel (Fumes *et al.* 2015). Matrix-related compounds can be co-extracted and can interfere in the analysis; so, the sample preparation has a multifarious role related to target analyte extraction, preconcentration and clean-up from co-existing species (Gao *et al.* 2015). A preconcentration step is usually necessary before the final analysis of compounds (Gao *et al.* 2015; Feizi *et al.* 2017). However, some methods often require high amounts of organic solvents that are harmful to the environment (Gao *et al.* 2015; Feizi *et al.* 2017; Płotka-Wasylka *et al.* 2017). A concept that has been approached is the green analytical chemistry, which decreases or eliminates organic solvents during the extraction procedure (Fumes *et al.* 2015; Płotka-Wasylka *et al.* 2017). Karayaka *et al.* (2019) developed a method to extract BPA from drinking water bottles using a switchable polarity solvent (*N*,*N*-dimethylbenzylamine), which is a green solvent. Also, it is very important to use a proper sample preparation to reach the required lower LODs (Gao *et al.* 2015). Discoveries in materials science may supply new tools for the preparation of samples (Jalili *et al.* 2020). Mohammadnezhad *et al.* (2017) developed ionic liquid-bonded fused silica as a new solid-phase microextraction (SPME) fiber for the liquid chromatographic determination of BPA in mineral water bottled in PET. Wei *et al.* (2018) synthesized a novel magnetic solid-phase extraction (MSPE) for the determination of six phthalic acid esters in mineral water (including BBP, DBP and DEHP). The development of natural sorbents has also been investigated, which are cheap and readily available and sometimes their performance was comparable with synthetic sorbents (Sajid *et al.* 2016).

Some works in Table 1 developed extraction methods. González-Sálamo *et al.* (2017) used the first application of core–shell poly (dopamine) magnetic nanoparticles as a sorbent for the extraction of a group of 11 phthalic acid esters of interest. Pinsrithong & Bunkoed (2018) synthesized a hierarchically porous composite nanostructure of polypyrrole, reduced graphene oxide, magnetite nanoparticles and alginate hydrogel microspheres (PPy-rGO*x*-Fe_3O_4). They applied as a magnetic solid-phase extraction

Table 1 | Extraction methods for the determination of BPA, BBP, DBP and DEHP in commercial water bottles

Without migration study

Detected analyte (s)	Extraction method	Chromatographic technique	LODs (μg/L)	LOQs (μg/L)	Reference
BPA	SPE	UFLC–MS/MS	0.004–0.055[a]	1.4×10^{-2}–1.2×10^{-2}[a]	Zhou *et al.* (2019)
BPA	SBSE	HPLC–UV/Vis	0.02	0.06	Gorji *et al.* (2019)
BPA	USAE-MIP-μ-SPE	HPLC–DAD	0.07	0.15	Rozaini *et al.* (2017)
BPA	MDMIP- SPE	HPLC–DAD	0.083	0.114	Chang *et al.* (2017)
BPA	SPME	HPLC–DAD	0.20	Not stated	Mohammadnezhad *et al.* (2017)
BPA	SLLME	GC–MS	0.54	1.8	Karayaka *et al.* (2019)
BPA BBP; DBP; DEHP	LLE	LC–MS/MS GC–MS	Not stated	Not stated	Wu *et al.* (2019)
BBP; DBP	MIP-SPE	HPLC–MS	0.16; 0.84	0.55; 2.81	Barciela-Alonso *et al.* (2017)
BBP; DBP	IT-UAA-LLME	GC–MS	1.67; 0.75	5.50; 2.46	Farahani *et al.* (2017)
BBP; DBP	m-μdSPE	UHPLC–MS/MS	Not stated	6×10^{-3}; 11×10^{-3}	Santana-Mayor *et al.* (2018)
BBP; DBP; DEHP	MSPE	HPLC–UV/Vis	0.0103; 0.003[b]; 0.0167	0.0342; 0.022[b]; 0.0556	Yin *et al.* (2019)
BBP; DBP; DEHP	SPE	GC–MS/MS	0.18; 0.021; 0.036	0.60; 0.070; 0.12	Li *et al.* (2019)
BBP; DBP; DEHP	LLE	GC–MS/MS	1.0; 1.0; 0.5	3.0; 3.0; 0.15	Tran-Lam *et al.* (2018)
BBP; DBP; DEHP	MSPE	GC–MS	5.0; 1.0; 5.0	Not stated	Wei *et al.* (2018)
DBP	TSP-LLME	GC–MS	0.007	0.021	Chen *et al.* (2019b)
DBP	SVA-LLME	GC–MS	0.15	0.50	Mohebbi *et al.* (2017)
DBP	DSPE–DLLME	GC–FID	1.24	4.11	Farajzadeh *et al.* (2019)
DBP	SPE	HPLC–UV/Vis	2.4	7.9	Salazar-Beltrán *et al.* (2017)
DBP	MISPME	HPLC–UV/Vis	3	10	Soheilifar *et al.* (2018)
DBP	m-μdSPE	GC–MS/MS	Not stated	0.009	González-Sálamo *et al.* (2017)
DBP	LLE	HPLC–UV/Vis	Not stated	Not stated	Dada *et al.* (2018)
DBP	HF-LPME	GC–MS/MS	Not stated	Not stated	González-Sálamo *et al.* (2018)
DBP; DEHP	MEPS–DLLME	GC–FID	0.001; 0.005	0.003; 0.015	Amiri & Ghaemi (2017)
DBP; DEHP	MSPE	GC–MS/MS	0.005; 0.008	0.02; 0.03	Pinsrithong & Bunkoed (2018)
DBP; DEHP	RDSE	GC–MS	0.01; 0.03	0.04; 0.10	Manzo *et al.* (2019)
DBP; DEHP	LLE	GC–MS	0.01–0.05[c]	0.03–0.15[c]	Tri *et al.* (2018)
DBP; DEHP	MEPS	GC–FID	0.05; 0.10	0.10; 0.25	Amiri *et al.* (2017)
DBP; DEHP	LLE	GC–FID	Not stated	Not stated	Szendi *et al.* (2018)
DEHP	DMIMS–SPE	GC–MS	0.00039	0.0013	Özer *et al.* (2017)
DEHP	MSPE	GC–FID	0.02	Not stated	Chahkandi & Amiri (2019)
DEHP	DLLME	GC–FID	2	4	Notardonato *et al.* (2018)
Detected analyte (s)	Extraction method	Chromatographic technique	LODs (mg/kg)	LOQs (mg/kg)	Reference

(*continued*)

Table 1 | continued

Without migration study

Detected analyte (s)	Extraction method	Chromatographic technique	LODs (μg/L)	LOQs (μg/L)	Reference
BBP; DBP; DEHP	LLE	GC–MS	0.00031; 0.00025; 0.00042	0.00096; 0.0008; 0.00122	Yang *et al.* (2017)
With migration study					
Detected analyte (s)	Extraction method	Chromatographic technique	LODs (μg/L)	LOQs (μg/L)	Reference
BPA	PT-μ-SPE	HPLC–FLD	0.001	0.0032	Kaykhaii *et al.* (2020)
BBP; DBP; DEHP	μSPE	GC–FID	0.025; 0.017; 0.031	Not stated	Abtahi *et al.* (2019)
BBP; DBP; DEHP	Not stated	LC–MS/MS	0.20; 0.20; 0.30	0.64; 0.60; 0.94	Surhio *et al.* (2017)
DBP	MIP-SPME	GC–FID	0.12	Not stated	Hashemi-Moghaddam & Maddah (2018)
DBP	HS-SPME	GC–MS	0.17	0.57	Cincotta *et al.* (2018)
DBP; DEHP	SPE	GC–MS	0.015[d]	Not stated	Sulentic *et al.* (2018)
DBP; DEHP	LLE	GC–MS/MS	0.043; 0.062	Not stated	Zaki & Shoeib (2018)
DBP; DEHP	AALLME	GC–MS	0.3; 0.2	Not stated	Yousefi *et al.* (2019)
DEHP	UA-DLLME	GC–MS	10–100[e]	50–500[e]	Annamalai & Namasivayam (2017)

BPA, bisphenol A; BBP, benzylbutyl phthalate; DBP, di-*n*-butyl phthalate; DEHP, di(2-ethylhexyl) phthalate; SPE, solid-phase extraction; SBSE, stir bar sorptive extraction; USAE, ultrasound-assisted emulsification; MIP, molecularly imprinted polymer; μ-SPE, micro-solid-phase extraction; MDMIP, magnetic dummy molecularly imprinted polymer; SPME, solid-phase microextraction; SLLME, switchable liquid–liquid microextraction; LLE, liquid–liquid extraction; IT-UAA, in tube ultrasonic and air-assisted; LLME, liquid–liquid microextraction; m-μdSPE, magnetic micro-dispersive solid-phase extraction; MSPE, magnetic solid-phase extraction; TSP, temperature-sensitive polymer; SVA, solvent vapor-assisted; DSPE, dispersive solid-phase extraction; DLLME, dispersive liquid–liquid microextraction; MISPME, molecularly imprinted solid-phase microextraction; HF-LPME, hollow fiber liquid-phase microextraction; MEPS, microextraction in packed syringe; RDSE, rotating disk sorptive extraction; DMIMS, dual-template molecularly imprinted mesoporous silica; PT-μ-SPE, pipette-tip micro-solid-phase extraction; μSPE, micro-solid-phase extraction; HS, headspace; AALLME, air-assisted liquid–liquid microextraction; UA, ultrasound-assisted; GC, gas chromatography; MS, mass spectrometry; HPLC, high-performance liquid chromatography; DAD, diode-array detection; FID, flame ionization detector; UV/Vis, dual-wavelength ultraviolet/visible; MS/MS, tandem mass spectrometry; UHPLC, ultra-high-performance liquid chromatography; UFLC, ultra-fast liquid chromatography; UPLC–MS, ultra-performance liquid chromatography–mass spectrometry; FLD, fluorescence detector.

[a]Corresponds to four bisphenols – BPA; BPB: bisphenol B; BPF: bisphenol F; BPS: bisphenol S.

[b]Corresponds to two phthalates – DBP: bis(2-butoxyethyl) phthalate/DBP.

[c]Corresponds to nine phthalates – DEP: diethyl phthalate; DPP: dipropyl phthalate; DiBP: di-*iso*butyl phthalate; DBP; DCHP: dicyclohexyl phthalate; DnHP: dihexyl phthalate; BzBP: benzyl butyl phthalate; DnOP: di(*n*-octyl)phthalate; DEHP.

[d]Corresponds to three phthalates – DiBP; DBP; DEHP.

[e]Corresponds to 14 phthalates – BBP; DBP; DEHP; DPP; DEP; DiBP; DCHP; DNOP: di-*n*-octyl phthalate; DMP: dimethyl phthalate; DHP: dihexyl phthalate; DiNP: di-isononyl phthalate; DiDP: di-isodecyl phthalate; BBEP: bis (2-*n*-butoxyethyl) phthalate; BMEP: bis (2-methoxy ethyl) phthalate.

adsorbent for PAEs, including BBP, DBP and DEHP. Farajzadeh *et al.* (2019) developed a natural and costless adsorbent for the accomplishment of a dispersive solid-phase extraction (DSPE) procedure followed by dispersive liquid–liquid microextraction (DLLME) for the extraction and preconcentration of PAEs and alkylphenols. None of the methods of Table 1 present LOD and LOQ values lower than the MCL to BPA in drinking water by EC (0.1 μg/L), but are lower than in China (10 μg/L) (EC 2020; GB-5749-2006).

Currently, MCL has not been established for BBP (US EPA 2019a), although, in 1990, US EPA proposed an MCL of 100 μg/L (Parks *et al.* 1993). In 2004, New Jersey State Primary and Secondary Drinking Water Standards derived the same value, multiplying the drinking water equivalent level of 7 mg/L by the relative source contribution factor of 20% and dividing the result by the additional uncertainty factor of 10 for possible human carcinogens (NJDEP 2004). All methods show LOD and LOQ below this proposed MCL value for BBP.

Almost all methods exhibit LOD and LOQ lower than DBP by China for drinking water (3 μg/L) (GB-5749-2006). All methods show LOD and LOQ lower than DEHP by US FDA for bottled water (6 μg/L) and by WHO, Codex Alimentarius, China for drinking water (8 μg/L) (Codex Alimentarius 2001; GB-5749-2006; WHO 2017; ECFR 2020). Yang *et al.* (2017) analyzed BBP, DBP and DEHP by GC–MS/LLE. The values of LOD and LOQ are given in mg/kg. The LOD and LOQ are lower than the SML values by the EU for BBP (30 mg/kg), DBP (0.3 mg/kg) and DEHP (1.5 mg/kg) (EFSA 2019).

DETECTIONS OF BPA AND PAES

The detected levels of BPA, BBP, DBP and DEHP in commercial water bottles without the storage study are present in Table 2 and Figure 1. Wei *et al.* (2018) and Yang *et al.* (2017) are not included in Figure 1 because the units are in mg/kg. The detected levels of BPA, BBP, DBP and DEHP in commercial water bottles with the storage study are present in Table 3 and Figure 2. For articles with concentration ranges, averages were used to generate Figures 1 and 2. A better understanding of the methods used in storage studies can be verified in their respective articles.

Some papers presented values above the MCL to BPA (0.1 μg/L) by EC (EC 2020). All the papers exhibited levels lower than MCL to BPA by China (GB-5749-2006). Even though there is no specific legislation for BBP so far, all the papers showed levels lower than MCL of 100 μg/L proposed by US EPA. It should be noted that this is a proposed value and has not been defined as a standard, but the proposed value serves to analyze the results for the moment. Almost all papers exhibited DBP levels low than 3 μg/L (GB-5749-2006) and displayed DEHP levels low than 6 or 8 μg/L (Codex Alimentarius 2001; GB-5749-2006; WHO 2017; ECFR 2020). The values of BBP, DBP and DEHP obtained by Wei *et al.* (2018) and Yang *et al.* (2017) are lower than the SML values by the EU for BBP (30 mg/kg), DBP (0.3 mg/kg) and DEHP (1.5 mg/kg) (EFSA 2019).

The countries with the reported highest levels of BPA, BBP, DBP and DEHP were Turkey (5.7 μg/L – Figure 1), Pakistan (12.11 μg/L – Figure 2), Mexico (82.8 μg/L – Figure 1) and Thailand (64.0 μg/L – Figure 1), respectively. The PAE values detected were highest than those established by legislation. Thailand also was the country with the first rank with DEHP (94.1 μg/L) in bottled waters in the review by Luo *et al.* (2018). The value was obtained by Uansiri *et al.* (2016) in bottled water contained in plastic containers. DEHP is known as a dominant PAE in bottled water (Keresztes *et al.* 2013; Guart *et al.* 2014; Zaki & Shoeib 2018; Abtahi *et al.* 2019).

DBP was the most compound detected. Luo *et al.* (2018) also verified that DBP was the PAE with more detection frequency in bottled water. All the samples (10 brands) analyzed by Soheilifar *et al.* (2018) present DBP. Among 16 PAEs studied by Zhang *et al.* (2018), DBP was the most ubiquitous and dominant contaminant in the study population. Soheilifar *et al.* (2018) optimized a molecularly imprinted polymer as a highly selective sorbent toward DBP. Dada *et al.* (2018) also analyzed packaged sachet water, and DBP concentrations were almost four-time higher (160 μg/L) relative to bottled water. Sachet water is packaging in plastic bags (Semey *et al.* 2020) made of LDPE (Jnr *et al.* 2018), and it is relatively cheaper than a water bottle (Dada *et al.* 2018).

Kaykhaii *et al.* (2020) verified that the water sample presented more BPA migration (Figure 2) when brought to boiling in a steel jar, quickly poured into the bottle and after cooling at ambient temperature (Figure 2). Surhio *et al.* (2017) detected the highest value of BBP migration studied with the influence of sunlight in Pakistan (Figure 2). The intensity of sunlight may affect the degradation degree of PAEs (Lertsirisopon *et al.* 2009), and the occurrence of PAEs in water stored in PET bottles depended mainly on the country of origin of the bottle (Schmid *et al.* 2008; Keresztes *et al.* 2013). All the papers that specified the type of bottle demonstrated DBP levels above the MCL (3 μg/L). Yousefi *et al.* (2019) also studied PET bottled water exposed to sunlight and as well as Surhio *et al.* (2017) verified an increase in DBP concentration. DBP values at room temperature were lower than at freezing for Hashemi-Moghaddam & Maddah (2018), while the reverse occurred for Sulentic *et al.* (2018). The presence of DBP may be due to different production facilities used by the different brands tested (Al-Saleh *et al.* 2011; Guart *et al.* 2014). Annamalai & Namasivayam (2017) obtained bigger values to DEHP at 4 °C and smaller values at 37 °C. To Zaki &

Table 2 | Levels of BPA, BBP, DBP and DEHP in commercial water bottles without the storage study

Detected analyte	Sample	Country	Type of bottle	Number of brands or samples	Concentration (μg/L)	Reference
BPA	Drinking water bottle	Turkey	Not stated	3	5.7	Karayaka *et al.* (2019)
BPA	Mineral water bottle	Iran	PET[a]	1	5.5	Mohammadnezhad *et al.* (2017)
BPA	Mineral water	Malaysia	Not stated	6	1.25	Rozaini *et al.* (2017)
BPA	Plastic bottled mineral water	China	Not stated	1	0.127	Chang *et al.* (2017)
BPA	Bottled mineral water	Iran	Not stated	3	0.07	Gorji *et al.* (2019)
BPA	Bottled water	China	Not stated	Not stated	0.05–0.08	Zhou *et al.* (2019)
BPA	Bottled water	China	Not stated	17	0.01	Wu *et al.* (2019)
BBP	Mineral water	Iran	PET	3	2.9–5.5	Farahani *et al.* (2017)
BBP	Bottled water	China	Not stated	17	1.86	Wu *et al.* (2019)
BBP	Bottled water in plastic	Spain	Not stated	4	0.75–1.9	Barciela-Alonso *et al.* (2017)
BBP	Mineral water	China	Not stated	5	0.515–0.690	Yin *et al.* (2019)
BBP	Mineral water	Vietnam	Not stated	14	0.30–0.95	Tran-Lam *et al.* (2018)
BBP	Bottled drinking water	China	Not stated	60	0.019–0.032	Li *et al.* (2019)
BBP	Mineral water bottled in plastic	Spain	Not stated	1	<LOQ	Santana-Mayor et al. (2018)
DBP	Plastic bottled Water	Nigeria	Not stated	15	42	Dada *et al.* (2018)
DBP	Drinking water	Mexico	PET	10	20.5–82.8	Salazar-Beltrán *et al.* (2017)
DBP	Mineral water	China	Not stated	5	8.98–11.5	Yin *et al.* (2019)
DBP	Bottled water in plastic	Spain	Not stated	4	4.6–8.2	Barciela-Alonso *et al.* (2017)
DBP	Plastic bottled water	Thailand	Not stated	1	17.0	Pinsrithong & Bunkoed (2018)
DBP	Plastic bottled water	Iran	Not stated	1	5.2	Mohebbi *et al.* (2017)
DBP	Mineral water	Cyprus	Not stated	Not stated	4.35	Farajzadeh *et al.* (2019)
DBP	Bottled mineral water	Iran	Not stated	1	4.5	Amiri & Ghaemi (2017)
DBP	Mineral water	China	Not stated	1	2.68	Chen *et al.* (2019b)
DBP	Bottled water	China	Not stated	17	1.34	Wu *et al.* (2019)
DBP	Mineral water	Iran	Not stated	3	1.1–2.5	Amiri *et al.* (2017)
DBP	Mineral water	Iran	PET	3	1.1–1.7	Farahani *et al.* (2017)
DBP	Mineral water	Spain	Not stated	1	<1	González-Sálamo *et al.* (2018)
DBP	Mineral bottled water	Spain	PET	1	0.36	González-Sálamo *et al.* (2017)
DBP	Water packed in plastic bottle (still, sparkling and light sparkling)	Chile	Not stated	5 (2 – still, 2 – sparkling, 1 – light sparkling)	0.353–2.756	Manzo *et al.* (2019)
DBP	Plastic bottled water	Iran	Not stated	10	0.26–1.13	Soheilifar *et al.* (2018)
DBP	Plastic bottled beverages (water)	Vietnam	Not stated	8	0.24–1.86	Tri *et al.* (2018)

(*continued*)

Table 2 | continued

Detected analyte	Sample	Country	Type of bottle	Number of brands or samples	Concentration (μg/L)	Reference
DBP	Mineral water bottled in plastic	Spain	Not stated	1	0.184	Santana-Mayor et al. (2018)
DBP	Mineral water	Vietnam	Not stated	14	0.09–0.95	Tran-Lam *et al.* (2018)
DBP	Bottled drinking water	China	Not stated	60	0.021–0.51	Li *et al.* (2019)
DBP	Bottled mineral water	Hungary	PET	4	<0.005–0.2	Szendi *et al.* (2018)
DEHP	Plastic bottled water	Thailand	Not stated	1	64.0	Pinsrithong & Bunkoed (2018)
DEHP	Bottled water	Italy	Not stated	2	22.9–24.4	Notardonato *et al.* (2018)
DEHP	Plastic bottled beverages (water)	Vietnam	Not stated	8	10.3–42.3	Tri *et al.* (2018)
DEHP	Plastic bottled water	Turkey	Not stated	Not stated	10.06–11.90	Özer *et al.* (2017)
DEHP	Bottled mineral water	Iran	Not stated	1	3.0	Amiri & Ghaemi (2017)
DEHP	Bottled mineral water	Iran	Not stated	2	2.6	Chahkandi & Amiri (2019)
DEHP	Bottled water	China	Not stated	17	2.50	Wu *et al.* (2019)
DEHP	Water packed in plastic bottle (still, sparkling and light sparkling)	Chile	Not stated	5 (2 – still, 2 – sparkling, 1 – light sparkling)	1.258–4.321	Manzo *et al.* (2019)
DEHP	Mineral water	Iran	Not stated	3	0.5–3.5	Amiri *et al.* (2017)
DEHP	Mineral water	Vietnam	Not stated	14	0.46–1.8	Tran-Lam *et al.* (2018)
DEHP	Mineral water	China	Not stated	5	<LOQ–0.733	Yin *et al.* (2019)
DEHP	Bottled mineral water	Hungary	PET	4	<0.29–11.289	Szendi *et al.* (2018)
DEHP	Bottled drinking water	China	Not stated	60	0.013–0.021	Li *et al.* (2019)
Detected analyte	**Sample**	**Country**	**Type of bottle**	**Number of brands or samples**	**Concentration (mg/kg)**	**Reference**
BBP	Mineral water	China	Not stated	1	0.001	Wei *et al.* (2018)
DBP					0.014	
DEHP					0.018	
BBP	Mineral water Soda water	China	Not stated	Not stated	0.32×10^{-4}–1.1×10^{-4} <LOD–1.3×10^{-4}	Yang *et al.* (2017)
DBP	Mineral water Soda water				1.3×10^{-4}–10.2×10^{-4} 1.6×10^{-4}–63.4×10^{-4}	
DEHP	Mineral water Soda water				2.2×10^{-4}–43.9×10^{-4} 5.7×10^{-4}–72.9×10^{-4}	

BPA, bisphenol A; BBP, benzylbutyl phthalate; DBP, di-*n*-butyl phthalate; DEHP, di(2-ethylhexyl) phthalate.
[a]Polyethylene terephthalate.

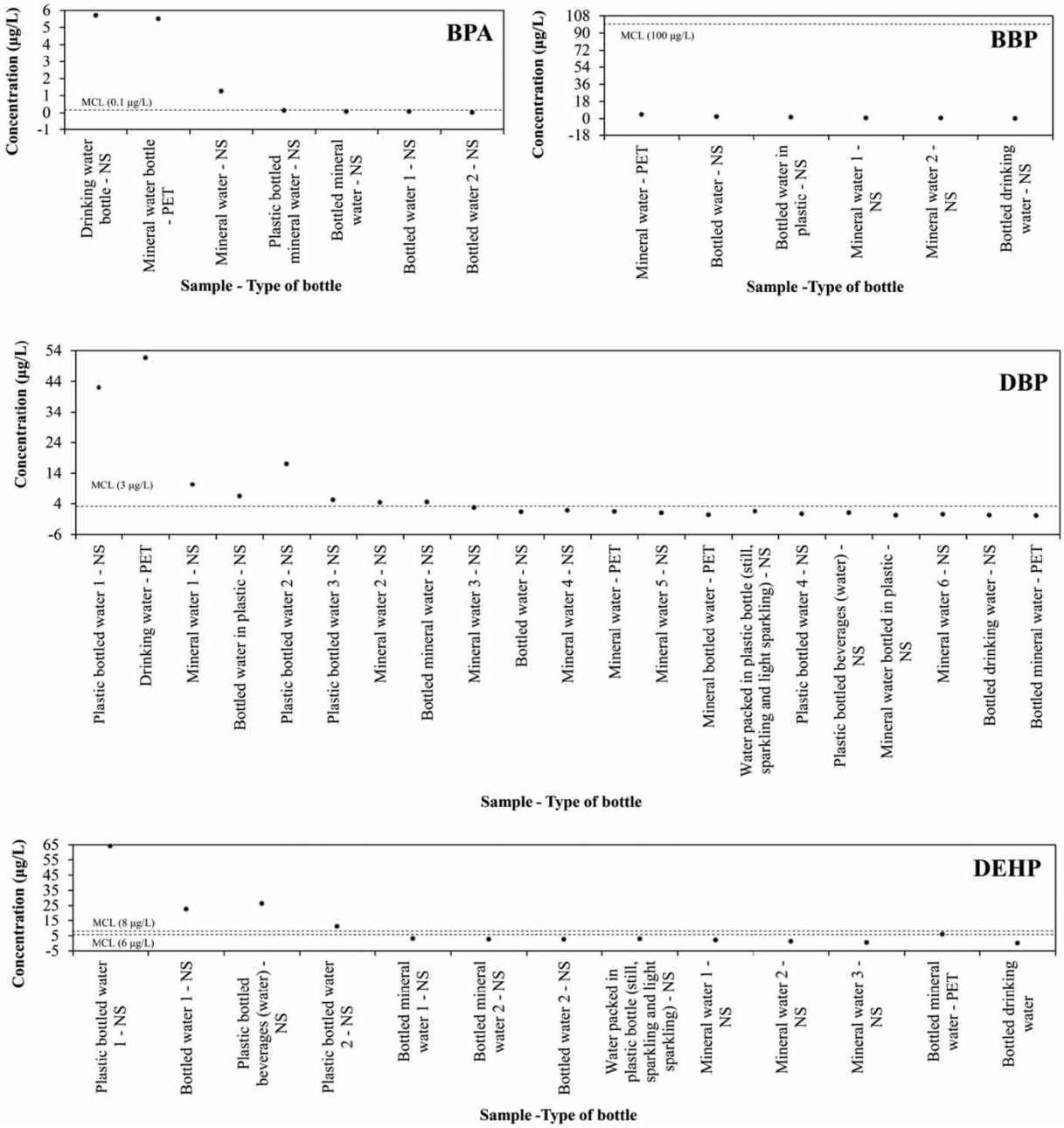


Figure 1 | BPA, BBP, DBP and DEHP concentrations detected in commercial water bottles without the storage study. The number in 'Sample-Type of bottle' represents different samples. NS is 'Not stated'.

Shoeib (2018) occurred the reverse. These authors analyzed DEHP in PET bottled water.

The migration of PAEs in bottled water results from the combined effects of multiple factors, as reported by Luo *et al.* (2018). The possible reason for the migration of PAEs is the usage of low-quality plastic as well as solubility in water (Saeed *et al.* 2010). The plastic type is that influences the presence of specific contaminants, where the migration of plasticizers from the cap material plays an important role (Guart *et al.* 2014). Jeddi *et al.* (2016) noted that the effect of temperatures and sunlight exposure on the release of the BBP, DBP and DEHP into the water is more than the effect

Table 3 | Levels of BPA, BBP, DBP and DEHP in commercial water bottles with the storage study

Detected analyte	Sample	Country	Type of bottle	Number of brands or samples	Storage study	Concentration (μg/L)	Reference
BPA	Bottled drinking water	Iran	Not stated	4	Freezing temperature (24 h)[a]	0.0023	Kaykhaii *et al.* (2020)
					Sunlight (for a week)	0.007	
					Boiled in a steel jar and quickly poured into the bottle (cooled to ambient temperature)[a]	0.016	
BBP	Mineral water bottle	Pakistan	Not stated	5	Sunlight (7 days with 10 h/day – 46–48 °C)	ND[b] – 12.11 (median: 7.43)	Surhio *et al.* (2017)
BBP	Bottled water	Iran	PET[c]	10	Sunlight (roof on sunny days for 1 week)	0.03–0.13	Abtahi *et al.* (2019)
DBP	Mineral water bottle	Pakistan	Not stated	5	Sunlight (7 days with 10 h/day – 46–48 °C)	ND[b] – 26.16 (median: 21.7)	Surhio *et al.* (2017)
DBP	Bottled water	Iran	PET	10	Sunlight (roof on sunny days for 1 week)	ND[b] – 0.12 (median: 0.10)	Abtahi *et al.* (2019)
DBP	Bottled water	Romania	Not stated	Not stated	Room temperature 1–4 °C[a]	6.11 5.12	Sulentic *et al.* (2018)
DBP	Water in plastic bottle	Iran	Not stated	3	Room temperature[a] Freezing temperature[a]	5.32 10.12	Hashemi-Moghaddam & Maddah (2018)
DBP	Bottled mineral water	Italy	PET	15	6 months at 25 °C 12 months at 60 °C 18 months at 60 °C	1.23 3.14 6.01	Cincotta *et al.* (2018)
DBP	Bottled water	Egypt	PET	5	1 months (4 ± 1 °C) 2 months(4 ± 1 °C) 4 months(4 ± 1 °C) 1 months (40 ± 5 °C) 2 months (40 ± 5 °C) 4 months (40 ± 5 °C) 2 months (25 ± 5 °C) 6 months (25 ± 5 °C)	0.107 0.128 0.173 0.124 0.167 0.229 0.136 0.227	Zaki & Shoeib (2018)
DBP	Drinking water bottled	Iran	PET	5	First week of the production Sunlight (23 ± 2 °C at 5 days) Incubator (25 °C for 75 days) Incubator (42 °C for 15 days)	0.80 5.86 Not stated Not stated	Yousefi *et al.* (2019)
DEHP	Mineral water bottle	Pakistan	Not stated	5	Sunlight (7 days with 10 h/day – 46–48 °C)	20.23	Surhio *et al.* (2017)

(*continued*)

Table 3 | continued

Detected analyte	Sample	Country	Type of bottle	Number of brands or samples	Storage study	Concentration (μg/L)	Reference
DEHP	Bottled water	Iran	PET	10	Sunlight (roof on sunny days for 1 week)	0.7–0.12	Abtahi *et al.* (2019)
DEHP	Drinking water bottled	Iran	PET	5	First week of the production Sunlight (23 ± 2 °C at 5 days) Incubator (25 °C for 75 days) Incubator (42 °C for 15 days)	0.77 Not stated 9.62[d] and 12.67[e] 10.33[f]	Yousefi *et al.* (2019)
DEHP	Bottled water	Romania	Not stated	Not stated	Room temperature 1 to 4 °C[a]	0.52 2.00	Sulentic *et al.* (2018)
DEHP	Bottled water	India	PET	Not stated	2 months at 4 °C 2 months at 37 °C 4 months at 4 °C 4 months at 37 °C 6 months at 4 °C 6 months at 37 °C	0.303 0.081 0.40 0.2010 1.09 0.59	Annamalai & Namasivayam (2017)
DEHP	Bottled water	Egypt	PET	5	1 months (4 ± 1 °C) 2 months(4 ± 1 °C) 4 months(4 ± 1 °C) 1 months (40 ± 5 °C) 2 months (40 ± 5 °C) 4 months (40 ± 5 °C) 2 months (25 ± 5 °C) 6 months (25 ± 5 °C)	0.135 0.235 0.307 0.190 0.306 0.432 0.274 0.396	Zaki & Shoeib (2018)

BPA, bisphenol A; BBP, benzylbutyl phthalate; DBP, di-*n*-butyl phthalate; DEHP, di(2-ethylhexyl) phthalate.
[a]Unspecified temperature.
[b]Not detected.
[c]Polyethylene terephthalate.
[d]Mean concentration.
[e]Highest mean concentration.
[f]Maximum amount.

due to storage duration. Keresztes *et al.* (2013) analyzed identical brands of water samples in PET containers having different volumes. The authors verified that how much higher is the contact surface between water and PET material, higher concentrations of BBP, DBP and DEHP were observed.

RISK ASSESSMENT

Daily intake-associated risk assessment

To compare the health risk *via* commercial water bottle consumption was used the risk assessment. The highest levels of BPA, BBP, DBP and DEHP in commercial water bottles are presented in Table 4 and Figure 3. The BPA in PET bottled water suggests other sources of contamination beside the packaging itself. The presence of BPA in PC packaging is known. In the case of PET bottled water, BPA can result from leaching by bottle caps or contamination of the water before bottling (Guart *et al.* 2011; Bach *et al.* 2012; Rowell *et al.* 2016). The water quality intended for bottling can be affected by the leaching of pollutants from unprotected agricultural and industrial areas (Bono-Blay *et al.* 2012). Bono-Blay *et al.* (2012) studied Spanish water sources intended for bottling, where BPA was one of the most frequently

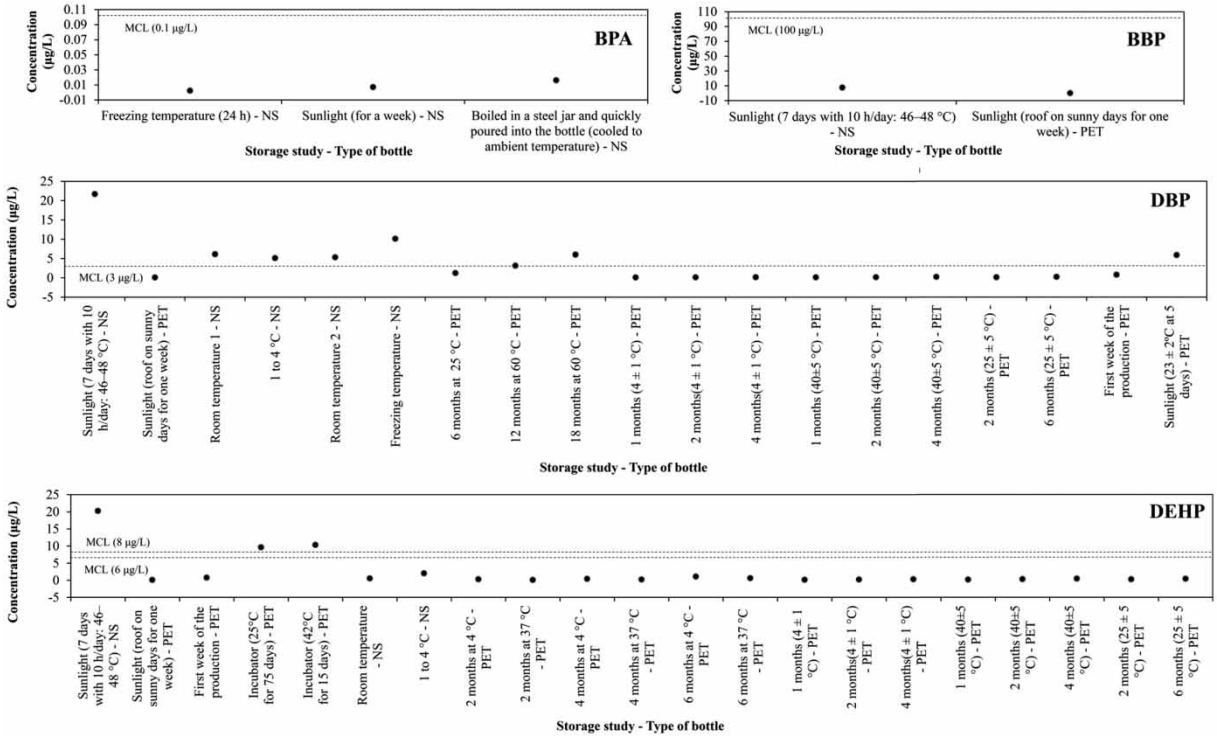


Figure 2 | Levels of BPA, BBP, DBP and DEHP variation in commercial water bottles with the storage study. The number in 'Sample-Type of bottle' represents different samples. NS is 'Not stated'.

Table 4 | Estimation of exposure to BPA, BBP, DBP and DEHP in commercial water bottles

Without the storage study

Detected analyte	Country	Concentration (µg/L)[a]	EDI (µg/kg-bw/day)[b]	Contribution *via* bottled water (%)[c]	ELCR[d]	Reference
BPA	Turkey	5.7	≈0.163	≈4.1	–	Karayaka *et al.* (2019)
BPA	Iran	5.5	≈0.157	≈3.9	–	Mohammadnezhad *et al.* (2017)
BPA	Malaysia	1.25	$\approx 3.6 \times 10^{-2}$	≈0.9	–	Rozaini *et al.* (2017)
BPA	China	0.127	$\approx 3.6 \times 10^{-3}$	≈0.09	–	Chang *et al.* (2017)
BPA	China	0.08	$\approx 2.3 \times 10^{-3}$	≈0.058	–	Zhou *et al.* (2019)
BPA	Iran	0.07	2.0×10^{-3}	≈0.05	–	Gorji *et al.* (2019)
BPA	China	0.01	$\approx 2.9 \times 10^{-4}$	$\approx 7.3 \times 10^{-3}$	–	Wu *et al.* (2019)
BBP	Iran	5.5	≈0.157	≈0.03	–	Farahani *et al.* (2017)
BBP	Spain	1.9	$\approx 5.4 \times 10^{-2}$	$\approx 1.1 \times 10^{-2}$	–	Barciela-Alonso *et al.* (2017)
BBP	China	1.86	5.3×10^{-2}	$\approx 1.1 \times 10^{-2}$	–	Wu *et al.* (2019)
BBP	Vietnam	0.95	$\approx 2.7 \times 10^{-2}$	$\approx 5.4 \times 10^{-3}$	–	Tran-Lam *et al.* (2018)
BBP	China	0.690	$\approx 2.0 \times 10^{-2}$	$\approx 4.0 \times 10^{-3}$	–	Yin *et al.* (2019)
BBP	China	0.032	$\approx 9.1 \times 10^{-4}$	$\approx 1.8 \times 10^{-4}$	–	Li *et al.* (2019)
BBP	Spain	<LOQ (0.006)	$< 1.7 \times 10^{-4}$	$< 3.4 \times 10^{-5}$	–	Santana-Mayor *et al.* (2018)
DBP	Mexico	82.8	2.37	23.7	–	Salazar-Beltrán *et al.* (2017)

(*continued*)

Table 4 | continued

Without the storage study

Detected analyte	Country	Concentration (μg/L)[a]	EDI (μg/kg-bw/day)[b]	Contribution *via* bottled water (%)[c]	ELCR[d]	Reference
DBP	Nigeria	42	1.2	12	–	Dada *et al.* (2018)
DBP	Thailand	17.0	≈0.486	≈4.86	–	Pinsrithong & Bunkoed (2018)
DBP	China	11.5	≈0.329	≈3.29	–	Yin *et al.* (2019)
DBP	Spain	8.2	≈0.234	≈2.34	–	Barciela-Alonso *et al.* (2017)
DBP	Iran	5.2	≈0.149	≈1.49	–	Mohebbi *et al.* (2017)
DBP	Iran	4.5	0.129	1.29	–	Amiri & Ghaemi (2017)
DBP	Iran	4.35	≈0.124	1.24	–	Farajzadeh *et al.* (2019)
DBP	China	2.68	$\approx 7.7 \times 10^{-2}$	≈0.77	–	Chen *et al.* (2019b)
DBP	Iran	2.5	$\approx 7.1 \times 10^{-2}$	≈0.71	–	Amiri *et al.* (2017)
DBP	Vietnam	1.86	5.3×10^{-2}	0.53	–	Tri *et al.* (2018)
DBP	Iran	1.7	$\approx 4.9 \times 10^{-2}$	≈0.49	–	Farahani *et al.* (2017)
DBP	China	1.34	$\approx 3.8 \times 10^{-2}$	≈0.38	–	Wu *et al.* (2019)
DBP	Iran	1.13	$\approx 3.2 \times 10^{-2}$	≈0.32	–	Soheilifar *et al.* (2018)
DBP	Spain	<1	$< 2.9 \times 10^{-2}$	<0.29	–	González-Sálamo *et al.* (2018)
DBP	Vietnam	0.95	$\approx 2.7 \times 10^{-2}$	≈0.27	–	Tran-Lam *et al.* (2018)
DBP	China	0.51	$\approx 1.5 \times 10^{-2}$	0.15	–	Li *et al.* (2019)
DBP	Spain	0.36	1.0×10^{-2}	0.1	–	González-Sálamo *et al.* (2017)
DBP	Hungary	<0.2	$< 5.7 \times 10^{-3}$	<0.057	–	Szendi *et al.* (2018)
DBP	Spain	0.184	$\approx 5.3 \times 10^{-3}$	≈0.053	–	Santana-Mayor et al. (2018)
DEHP	Thailand	64.0	≈1.83	≈3.66	$\approx 3.0 \times 10^{-5}$	Pinsrithong & Bunkoed (2018)
DEHP	Vietnam	42.3	≈1.21	2.42	$\approx 2.0 \times 10^{-5}$	Tri *et al.* (2018)
DEHP	China	2.50	$\approx 7.1 \times 10^{-2}$	≈1.42	$\approx 1.2 \times 10^{-6}$	Wu *et al.* (2019)
DEHP	Italy	24.4	≈0.697	≈1.39	$\approx 1.1 \times 10^{-5}$	Notardonato *et al.* (2018)
DEHP	Turkey	11.90	0.34	0.68	$\approx 5.6 \times 10^{-6}$	Özer *et al.* (2017)
DEHP	Hungary	<11.289	<0.323	<0.646	$\approx < 5.3 \times 10^{-6}$	Szendi *et al.* (2018)
DEHP	Chile	4.321	≈0.123	≈0.246	$\approx 2.0 \times 10^{-6}$	Manzo *et al.* (2019)
DEHP	Iran	3.5	0.1	≈0.2	$\approx 1.6 \times 10^{-6}$	Amiri *et al.* (2017)
DEHP	Iran	3.0	0.086	0.172	$\approx 1.4 \times 10^{-6}$	Amiri & Ghaemi (2017)
DEHP	Vietnam	1.8	5.1×10^{-2}	≈0.1	$\approx 8.53 \times 10^{-7}$	Tran-Lam *et al.* (2018)
DEHP	China	0.733	$\approx 2.1 \times 10^{-2}$	≈0.042	$\approx 3.4 \times 10^{-7}$	Yin *et al.* (2019)
DEHP	China	0.021	6.0×10^{-4}	1.2×10^{-3}	$\approx 9.9 \times 10^{-9}$	Li *et al.* (2019)

Detected analyte	Country	Concentration (μg/kg)[a]	EDI (μg/kg-bw/day)[b]	Contribution *via* bottled water (%)[c]	CR[d]	Reference
BBP	China	Mineral water (MW): 0.11 Soda water (SW): 0.13	$\approx 3.1 \times 10^{-3}$ $\approx 3.7 \times 10^{-3}$	$\approx 6.2 \times 10^{-4}$ $\approx 7.4 \times 10^{-4}$	–	Yang *et al.* (2017)

(*continued*)

Table 4 | continued

Without the storage study

Detected analyte	Country	Concentration (μg/L)[a]	EDI (μg/kg-bw/day)[b]	Contribution *via* bottled water (%)[c]	CR[d]	Reference
DBP		Mineral water: 1.02 Soda water: 6.34	$\approx 2.9 \times 10^{-2}$ ≈0.181	≈0.29 ≈1.8	–	
DEHP		Mineral water: 4.3 Soda water: 7.29	≈0.123 ≈0.208	≈0.246 ≈0.416	$\approx 5.2 \times 10^{-8}$ $\approx 3.4 \times 10^{-6}$	

With storage study

Detected analyte	Country	Concentration (μg/L)[a]	EDI (μg/kg-bw/day)[b]	Contribution *via* bottled water (%)[c]	CR[d]	Reference
BPA	Iran	0.016	$\approx 4.6 \times 10^{-4}$	$\approx 1.2 \times 10^{-2}$	–	Kaykhaii *et al.* (2020)
BBP	Pakistan	12.11	0.346	≈0.069	–	Surhio *et al.* (2017)
BBP	Iran	0.13	$\approx 3.7 \times 10^{-3}$	$\approx 7.4 \times 10^{-4}$	–	Abtahi *et al.* (2019)
DBP	Pakistan	26.16	≈0.747	7.47	–	Surhio *et al.* (2017)
DBP	Iran	10.12	≈0.289	≈2.89	–	Hashemi-Moghaddam & Maddah (2018)
DBP	Iran	8.45	≈0.241	≈2.41	–	Yousefi *et al.* (2019)
DBP	Romania	6.11	≈0.175	≈1.75	–	Sulentic *et al.* (2018)
DBP	Italy	6.01	≈0.172	≈1.72	–	Cincotta *et al.* (2018)
DBP	Egypt	0.229	$\approx 6.5 \times 10^{-3}$	≈0.065	–	Zaki & Shoeib (2018)
DBP	Iran	0.12	$\approx 3.4 \times 10^{-3}$	≈0.034	–	Abtahi *et al.* (2019)
DEHP	Pakistan	20.23	0.578	≈1.16	$\approx 9.5 \times 10^{-6}$	Surhio *et al.* (2017)
DEHP	Iran	12.67	0.362	≈0.724	$\approx 6.0 \times 10^{-6}$	Yousefi *et al.* (2019)
DEHP	Romania	2.00	$\approx 5.7 \times 10^{-2}$	≈0.114	9.4×10^{-7}	Sulentic *et al.* (2018)
DEHP	Egypt	0.432	$\approx 1.2 \times 10^{-2}$	≈0.024	$\approx 2.0 \times 10^{-7}$	Zaki & Shoeib (2018)
DEHP	Iran	0.12	$\approx 3.4 \times 10^{-3}$	$\approx 6.8 \times 10^{-3}$	$\approx 5.6 \times 10^{-8}$	Abtahi *et al.* (2019)
DEHP	India	1.09	$\approx 3.1 \times 10^{-2}$	≈0.062	$\approx 5.1 \times 10^{-7}$	Annamalai & Namasivayam (2017)

BPA, bisphenol A; BBP, benzylbutyl phthalate; DBP, di-*n*-butyl phthalate; DEHP, di(2-ethylhexyl) phthalate.

[a]The worst-case scenario (the maximum level of each compound) was employed.

[b]EDI = (C × IR)/BW, where *C* is the concentration of target compounds (μg/L or mg/kg), ingestion rate (IR) is the daily consumption rate of bottled water (L/day or g/day), and BW is body weight (Luo *et al.* 2018), and the IR was assumed to be 2.0 L/day or and 2.0 kg/day for a 70 kg for adult (BW) (WHO 2005). The value of 2.0 L/day refers to all water sources that includes water from all supply sources such as community water supply (i.e., tap water), bottled water, etc.

μg/kg-bw/day: microgram per kilogram of the body weight of the person taking per day.

[c]Contribution *via* drinking water = (EDI/TDI) × 100 (Zaki & Shoeib 2018), where the TDI for BPA, BBP, DBP, and DEHP are available for reference as established by EFSA (4, 500, 10 and 50 μg/kg-bw/day).

[d]ELCR is the Excess Lifetime Cancer Risks due to exposure to chemicals through the use of bottled water. ELCR = DWUR × MC, where Drinking Water Unit Risk is equal to 4.7×10^{-7} μg/L of DEHP in water, and MC is the maximum concentration (μg/L or μg/kg) of DEHP in bottled water (Jeddi *et al.* 2015). Here was considered the value of 4.7×10^{-7} μg/kg for papers with concentrations give in mg/kg.

detected compounds at concentrations between 0.031 and 0.203 μg/L.

Although the estimated daily intake (EDI) of BPA, BBP, DBP and DEHP detected in PET bottled waters analyzed was below the legislative values (Table 4), considering all types of food, it may contribute to the total daily intake of these compounds. The highest contributions *via* commercial water bottles of BPA, BBP, DBP and DEHP in all the bottled waters studied were 4.1, 0.069, 23.7 and 3.66% of TDI, respectively (Figure 3). The results demonstrate that

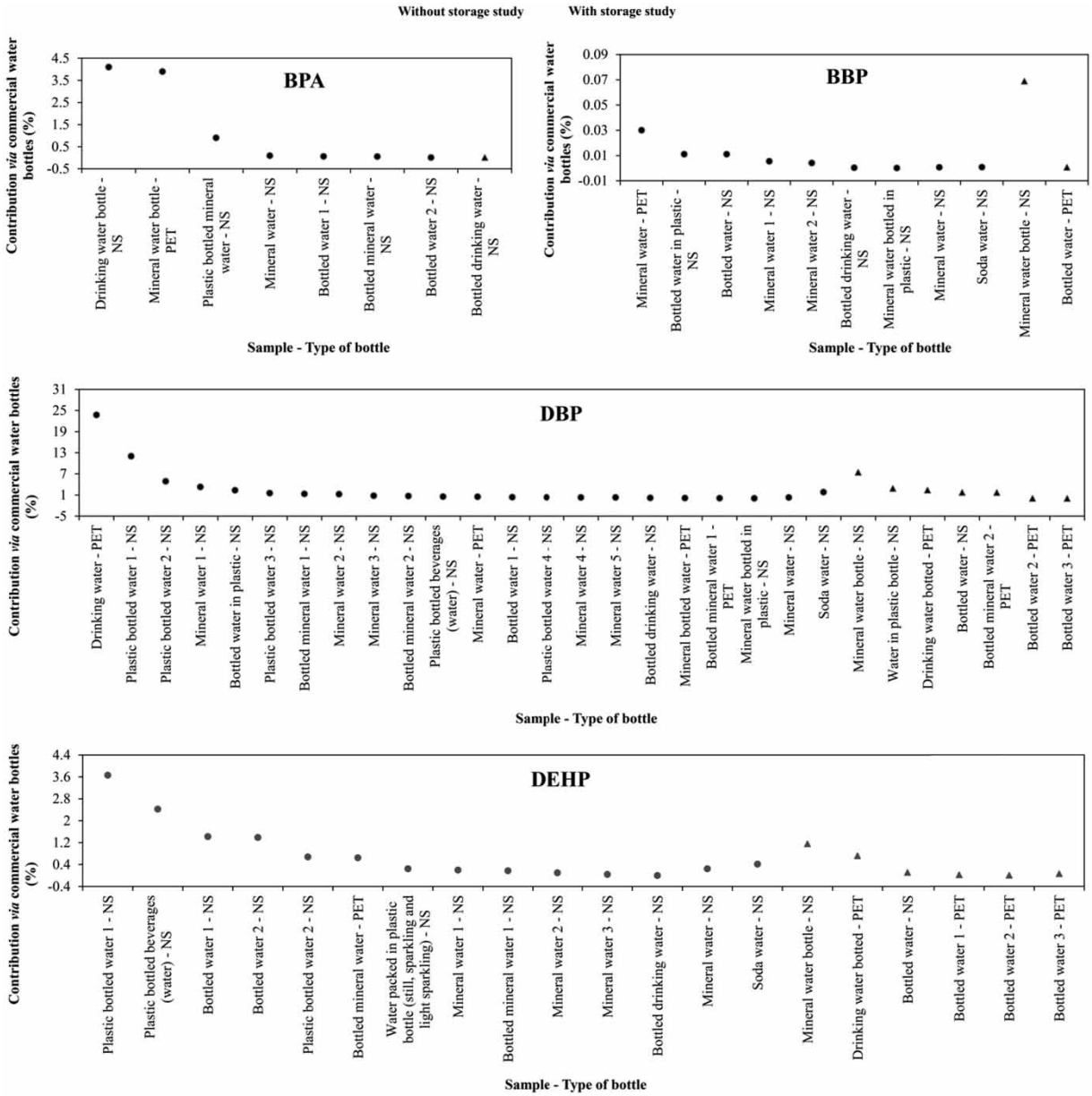


Figure 3 | BPA, BBP, DBP and DEHP contribution *via* commercial water bottles. The number in parentheses represents articles from the same country with different concentrations. MW is mineral water and SW is soda water.

contribution *via* commercial water bottles (Table 4) could represent a substantial source of exposure to these compounds (considering the highest contributions), when the daily consumption rate, of 2.0 L/day of bottled water and body weight of 70 kg, is used according to the standard WHO (2005). If Reference Dose (RfD) were considered (Table 5), which is more restrictive for DEHP, the contribution would be much higher (9.15%).

The carcinogenic risk (Excess Lifetime Cancer Risks – ELCR) posed by the highest concentration of DEHP in bottled water was negligible for all papers, with extremely below or between the accepted risk level of 10^{-6}–10^{-4} cancer risk (WHO 2017). As mentioned by WHO (2017), daily water intake can vary significantly in different parts of the world and location-specific data on drinking water consumption are preferred. As reported by Leung *et al.*

Table 5 | Estimated human exposure and estrogenic effects of BPA, BBP, DBP and DEHP via commercial bottled water ingested for other population groups

	BPA	BBP	DBP	DEHP
Reference	Karayaka *et al.* (2019)	Surhio *et al.* (2017)	Salazar-Beltrán *et al.* (2017)	Pinsrithong & Bunkoed (2018)
Maximum concentration (μg/L)	5.7	12.11	82.8	64.0
EDI (μg/kg-bw/day)				
Infants (birth to <12 months)*[a]	≈0.042	≈0.089	≈0.61	≈0.471
Children (1 to <3 years)*[b]	≈0.106	≈0.225	≈1.539	≈1.19
Children (3 to <11 years)*[c]	≈0.072	≈0.154	≈1.051	≈0.813
Teenage (11 to <16 years)*[d]	≈0.053	≈0.112	≈0.767	≈0.593
Young adult (16 <21 years)*[e]	≈0.061	≈0.129	≈0.882	≈0.682
Adult (≥21 years)*[f]	≈0.060	≈0.128	≈0.876	≈0.677
Pregnant (15–44)*[g]	≈0.046	≈0.097	≈0.662	0.512
Elderly (≥65 years)*[h]	≈0.057	≈0.120	≈0.821	≈0.635
Tolerable daily intake (μg/kg-bw/day)	4	500	10	50
Contribution *via* bottled water (%)[#]				
Infants (birth to <12 months)*[a]	≈1.05	$\approx 1.78 \times 10^{-2}$	≈6.10	≈0.942
Children (1 to <3 years)*[b]	≈2.65	$\approx 4.50 \times 10^{-2}$	≈15.40	≈2.38
Children (3 to <11 years)*[c]	≈1.80	$\approx 3.08 \times 10^{-2}$	≈1.05	≈1.63
Teenage (11 to <16 years)*[d]	≈1.33	$\approx 2.24 \times 10^{-2}$	≈7.67	≈1.19
Young adult (16 to <21 years)*[e]	≈1.53	$\approx 2.58 \times 10^{-2}$	≈8.82	≈1.36
Adult (≥21 years)*[f]	≈1.50	$\approx 2.56 \times 10^{-2}$	≈8.76	≈1.35
Pregnant (15–44)*[g]	≈1.15	$\approx 1.94 \times 10^{-2}$	≈6.62	1.02
Elderly (≥65 years)*[h]	≈1.43	$\approx 1.2 \times 10^{-2}$	≈8.21	≈1.27
RfD (μg/kg-bw/day)[i]	50	500	100	20
HQ[j]				
Infants (birth to <12 months)*[a]	$\approx 8.40 \times 10^{-4}$	$\approx 1.78 \times 10^{-4}$	$\approx 6.10 \times 10^{-3}$	$\approx 2.36 \times 10^{-2}$
Children (1 to <3 years)*[b]	$\approx 2.12 \times 10^{-3}$	$\approx 4.50 \times 10^{-4}$	$\approx 1.54 \times 10^{-2}$	$\approx 5.95 \times 10^{-2}$
Children (3 to <11 years)*[c]	$\approx 1.44 \times 10^{-3}$	$\approx 3.08 \times 10^{-4}$	$\approx 1.05 \times 10^{-2}$	$\approx 4.07 \times 10^{-2}$
Teenage (11 to <16 years)*[d]	$\approx 1.06 \times 10^{-3}$	$\approx 2.24 \times 10^{-4}$	$\approx 7.67 \times 10^{-3}$	$\approx 2.97 \times 10^{-2}$
Young adult (16 to <21 years)*[e]	$\approx 1.22 \times 10^{-3}$	$\approx 2.58 \times 10^{-4}$	$\approx 8.82 \times 10^{-3}$	$\approx 3.41 \times 10^{-2}$
Adult (≥21 years)*[f]	$\approx 1.20 \times 10^{-3}$	$\approx 2.56 \times 10^{-4}$	$\approx 8.76 \times 10^{-3}$	$\approx 3.39 \times 10^{-2}$
Pregnant (15–44)*[g]	$\approx 9.20 \times 10^{-4}$	$\approx 1.94 \times 10^{-4}$	$\approx 6.62 \times 10^{-3}$	2.56×10^{-2}
Elderly (≥65 years)*[h]	$\approx 1.14 \times 10^{-3}$	$\approx 2.4 \times 10^{-4}$	$\approx 8.21 \times 10^{-3}$	$\approx 3.18 \times 10^{-3}$
SF (based on maximum concentration)				
Infants (birth to <12 months)*[a]	1,190	5,620	164	42.5
Children (1 to <3 years)*[b]	472	2,220	65	16.8
Children (3 to <11 years)*[c]	694	3,250	95.1	24.6
Teenage (11 to <16 years)*[d]	943	4,460	130	33.7
Young adult (16 <21 years)*[e]	820	3,880	113	29.3
Adult (≥21 years)*[f]	833	3,910	114	29.5

(*continued*)

Table 5 | continued

	BPA	BBP	DBP	DEHP
Pregnant (15–44)*[g]	1,090	5,150	151	39.1
Elderly (≥65 years)*[h]	877	4,170	122	31.5
Estrogenic potency (EP)	$5.9E^{-05}$[l]	$2E^{-4}$[m]	$4.1E^{-5}$[m]	$3E^{-7}$[n]
EEQ (ng E2/L)[&]	0.336	2.42	3.39	0.0192
Total compounds				6.1652

*The values of weights and bottled water ingested are based on US EPA (2011) and US EPA (2019b): [a]0.0685 L/day for 9.3 kg, [b]0.2305 L/day for 12.45 kg, [c]0.3365 L/day for 26.5 kg, [d]0.517 L/day for 55.8 kg, [e]0.753 L/day for 70.7 kg; [f]0.84 L/day for 79.4 kg; [g]0.6 L/day for 75 kg; [h]0.749 L/day for 75.5 kg (Table 8–24, 8–25, 8–29, 3–34, 3–71, and Table A-2).
[#]Contribution *via* drinking water = (EDI/TDI) × 100 (Zaki & Shoeib 2018), where the TDI for BPA, BBP, DBP, and DEHP are available for reference as established by EFSA.
[i]US EPA (1987a, 1987b, 1988, 2019a).
[j]Hazard Quotient (HQ) = EDI/RfD, where HQ is associated with the exposure via the specified exposure route (unitless) (Jeddi *et al.* 2015).
[k]Safety factor (SF) = RfD/EDI (Luo *et al.* 2018).
[&]EEQ = $EP_i \times c_i$, where EP and *c* denote the estrogenic potency of an individual estrogenic compound (*in vitro* bioassays) and its corresponding concentration (Liu *et al.* 2009).
[l]Legler *et al.* (1999).
[m]Kim & Ryu (2006).
[n]Cavanagh *et al.* (2018).

(2013), infants and children have been subject to increased risks that are approximately six times greater than those in adolescents and adults due to their high drinking water consumption based on body weight. As specified by US EPA (2011), older adults (≥65 years of age) and pregnant are other susceptible groups due to their physiological properties change. As reported by Gerba *et al.* (1996), the elderly may be less able to create an effective defense against contaminants because of a pre-existing disease or weakened immune system. The risk is inherent to the pregnant and also to the fetus (Wee & Aris 2019).

Table 5 shows more detailed the risk assessment to other population groups based on only the ingestion of bottled water. The values of bottled water ingestion are based on US EPA (2011) and US EPA (2019b). US EPA (2019b) includes only bottled water consumed directly as a beverage, not including bottled water used in the preparation of foods. As stated by Hossain *et al.* (2013), the regional variability in water intake can be due to differences in weather conditions and food intake habits of the population. As can be seen in Table 5, the results demonstrate that bottled water can represent a substantial source of exposure to these compounds. Children (1 to <3 years) had a higher EDI, and as a result, a high contribution via bottled water consumption (15.4%) and high Hazard Quotient (HQ). As previously reported, if RfD for DEHP were considered, the contribution would be much higher (5.95%). The highest HQ for the compounds via bottled water consumption was much lower than 1 for all compounds, indicating an absence of risk (US EPA 2011). The safety factor (SF) calculated for the selected compounds with the maximum concentrations was all far above 1 for all groups, denoting that the BPA, BBP, DBP and DEHP concentrations disclosed in bottled water should not represent grave safety concerns, corroborating with the review by Luo *et al.* (2018).

Potential estrogenic effect of BPA and PAEs

Despite the safety factor indicates that the levels of the compounds in bottled waters are acceptable in terms of water safety, the potential estrogenic effects of the compounds by an average Estrogen Equivalent (EEQ) level in bottled waters are based on the highest concentrations that were evaluated (Table 5). The EEQ provides valuable information on human exposure to estrogen-like compounds, aiding in the estimation of the total dietary intake of estrogenicity (Schilirò *et al.* 2013). The potential estrogenic effects of BBP and DBP in bottled water should not be ignored due to their relatively high concentrations. As can be seen in Table 5, the average EEQ level in the bottled waters is significantly at 6.1652 ng E2/L, which was 22.8 times higher than those that cause adverse estrogenic effects on zebrafish (0.27 ng E2/L) as reported by Soares *et al.* (2009). Thus, the

average EEQ level indicated that BPA, BBP, DBP and DEHP in bottled waters may induce adverse estrogenic effects on human health.

CONCLUSIONS

Although the governments have published the guideline tolerance values of bisphenol A and PAEs in drinking water, they are still detected in water bottles. HPLC–DAD was the most used in BPA detections, while GC–MS was the most used in PAE detections. New methods to improve the extraction of BPA, BBP, DBP and DEHP from commercial water bottles have been developed. DBP and DEHP have still been detected in concentrations greater than those established by legislation. Contradictory observations, with decreasing and increasing concentrations on PAE concentration in bottled water, are reported. No consistent or clear trends regarding the effects of storage conditions, on PAE concentration in bottled water, are demonstrated. Based on the risk assessment, BPA, BBP, DBP and DEHP in commercial water bottles do not raise serious concern for humans. The average EEQ level revealed that BPA, BBP, DBP and DEHP in bottled waters may induce adverse estrogenic effects on human health. Besides that, the use of bottled water kept in unsuitable conditions is not appropriate and especially for sensitive groups. Thus, the occurrence of individual BPA, BBP, DBP and DEHP and their association in bottled water need to be verified to avoid their synergistic effects on human health.

DATA AVAILABILITY STATEMENT

All relevant data are included in the paper or its Supplementary Information.

REFERENCES

Abdelghani, J. I., Freihat, R. S. & El-Sheikh, A. H. 2020 Magnetic solid phase extraction of phthalate products from bottled, injectable and tap waters using graphene oxide: effect of oxidation method of graphene. *Journal of Environmental Chemical Engineering* **8** (2), 103527. https://doi.org/10.1016/j.jece.2019.103527.

Abtahi, M., Dobaradaran, S., Torabbeigi, M., Jorfi, S., Gholamnia, R., Koolivand, A., Darabi, H., Kavousi, A. & Saeedi, R. 2019 Health risk of phthalates in water environment: occurrence in water resources, bottled water, and tap water, and burden of disease from exposure through drinking water in Tehran, Iran. *Environmental Research* **173**, 469–479. https://doi.org/10.1016/j.envres.2019.03.071.

ACC 2019 *United States National Postconsumer Plastic Bottle Recycling Report (2019)*. American Chemistry Council. Available from: https://plastics.americanchemistry.com/Reports-and-Publications/2018-National-Post-Consumer-Plastics-Bottle-Recycling-Report.pdf (accessed 7 February 2020).

Alfarhani, B. F., Al-Tameem, M., Fadhel, A. A., Hammza, R. A. & Kadhem, M. I. 2019 Endocrine disrupting bisphenol A detection in different water samples in Iraq. *Journal of Physics: Conference Series* **1294** (5), 052045. https://doi.org/10.1088/1742-6596/1294/5/052045.

Al-Saleh, I., Shinwari, N. & Alsabbaheen, A. 2011 Phthalates residues in plastic bottled waters. *The Journal of Toxicological Sciences* **36** (4), 469–478. https://doi.org/10.2131/jts.36.469.

Amiri, A. & Ghaemi, F. 2017 Microextraction in packed syringe by using a three-dimensional carbon nanotube/carbon nanofiber–graphene nanostructure coupled to dispersive liquid-liquid microextraction for the determination of phthalate esters in water samples. *Microchimica Acta* **184** (10), 3851–3858. https://doi.org/10.1007/s00604-017-2416-8.

Amiri, A., Chahkandi, M. & Targhoo, A. 2017 Synthesis of nano-hydroxyapatite sorbent for microextraction in packed syringe of phthalate esters in water samples. *Analytica Chimica Acta* **950**, 64–70. https://doi.org/10.1016/j.aca.2016.11.027.

Annamalai, J. & Namasivayam, V. 2017 Determination of effect of pH and storage temperature on leaching of phthalate esters from plastic containers by ultrasound-assisted dispersive liquid–liquid micro-extraction. *Journal of Food Measurement and Characterization* **11** (4), 2222–2232. https://doi.org/10.1007/s11694-017-9607-1.

Armenta, S., Garrigues, S. & de la Guardia, M. 2015 The role of green extraction techniques in Green Analytical Chemistry. *TrAC Trends in Analytical Chemistry* **71**, 2–8. https://doi.org/10.1016/j.trac.2014.12.011.

Aznar, M., Vera, P., Canellas, E., Nerín, C., Mercea, P. & Störmer, A. 2011 Composition of the adhesives used in food packaging multilayer materials and migration studies from packaging to food. *Journal of Materials Chemistry* **21** (12), 4358–4370. https://doi.org/10.1039/C0JM04136 J.

Bach, C., Dauchy, X., Chagnon, M. C. & Etienne, S. 2012 Chemical compounds and toxicological assessments of drinking water stored in polyethylene terephthalate (PET) bottles: a source of controversy reviewed. *Water Research* **46** (3), 571–583. https://doi.org/10.1016/j.watres.2011.11.062.

Bach, C., Dauchy, X., Severin, I., Munoz, J. F., Etienne, S. & Chagnon, M. C. 2014 Effect of sunlight exposure on the release of intentionally and/or non-intentionally added substances from polyethylene terephthalate (PET) bottles into water: chemical analysis and in vitro toxicity. *Food Chemistry* **162**, 63–71. https://doi.org/10.1016/j.foodchem.2014.04.020.

Barciela-Alonso, M. C., Otero-Lavandeira, N. & Bermejo-Barrera, P. 2017 Solid phase extraction using molecular imprinted polymers for phthalate determination in water and wine samples by HPLC-ESI-MS. *Microchemical Journal* **132**, 233–237. https://doi.org/10.1016/j.microc.2017.02.007.

Bono-Blay, F., Guart, A., de la Fuente, B., Pedemonte, M., Pastor, M. C., Borrell, A. & Lacorte, S. 2012 Survey of phthalates, alkylphenols, bisphenol A and herbicides in Spanish source waters intended for bottling. *Environmental Science and Pollution Research* **19** (8), 3339–3349. https://doi.org/10.1007/s11356-012-0851-y.

Borrirukwisitsak, S., Keenan, H. E. & Gauchotte-Lindsay, C. 2012 Effects of salinity, pH and temperature on the octanol-water partition coefficient of bisphenol A. *International Journal of Environmental Science and Development* **3** (5), 460. http://www.ijesd.org/papers/267-CD0081.pdf

Cavanagh, J. A. E., Trought, K., Mitchell, C., Northcott, G. & Tremblay, L. A. 2018 Assessment of endocrine disruption and oxidative potential of bisphenol-A, triclosan, nonylphenol, diethylhexyl phthalate, galaxolide, and carbamazepine, common contaminants of municipal biosolids. *Toxicology In Vitro* **48**, 342–349. https://doi.org/10.1016/j.tiv.2018.02.003.

Čelić, M., Škrbić, B. D., Insa, S., Živančev, J., Gros, M. & Petrović, M. 2020 Occurrence and assessment of environmental risks of endocrine disrupting compounds in drinking, surface and wastewaters in Serbia. *Environmental Pollution* **262**, 114344. https://doi.org/10.1016/j.envpol.2020.114344.

Chahkandi, M. & Amiri, A. 2019 Hydroxyapatite/Fe_3O_4 nanocomposite as efficient sorbent for the extraction of phthalate esters from water samples. *Inorganic Chemistry Research* **2** (1), 50–64. https://doi.org/10.22036/icr.2019.184834.1045.

Chang, T., Yan, X., Liu, S. & Liu, Y. 2017 Magnetic dummy template silica sol–gel molecularly imprinted polymer nanospheres as magnetic solid-phase extraction material for the selective and sensitive determination of bisphenol A in plastic bottled beverages. *Food Analytical Methods* **10** (12), 3980–3990. https://doi.org/10.1007/s12161-017-0969-0.

Chen, H., Mao, W., Shen, Y., Feng, W., Mao, G., Zhao, T., Yang, L., Yang, L., Meng, C., Li, Y. & Wu, X. 2019a Distribution, source, and environmental risk assessment of phthalate esters (PAEs) in water, suspended particulate matter, and sediment of a typical Yangtze River Delta City, China. *Environmental Science and Pollution Research* **26** (24), 24609–24619. https://doi.org/10.1007/s11356-019-05259-y.

Chen, X., Xin, L., Xu, Y., Liu, J., Li, Z., Wang, Y. & Zhao, J. 2019b Polymer phase transition characteristics coupled with GC-MS for the determination of phthalate esters. *Journal of Separation Science* **42** (19), 3095–3101. https://doi-org.ez11.periodicos.capes.gov.br/10.1002/jssc.201900410.

Cincotta, F., Verzera, A., Tripodi, G. & Condurso, C. 2018 Non-intentionally added substances in PET bottled mineral water during the shelf-life. *European Food Research and Technology* **244** (3), 433–439. https://doi.org/10.1007/s00217-017-2971-6.

Codex Alimentarius 2001 General standard for bottled/packaged drinking waters (other than natural mineral waters). Available from: http://www.fao.org/input/download/standards/369/CXS_227e.pdf (accessed 18 June 2020).

Coniglio, M. A., Fioriglio, C. & Laganà, P. 2020 Polyethylene terephthalate. In: *Non-Intentionally Added Substances in PET-Bottled Mineral Water* (M. A. Coniglio, C. Fioriglio & P. Laganà, eds). Springer International Publishing, Cham.

Dada, E. O., Osidipe, V. A., Iyaomolere, K. E., Itoje, S. O. & Akinola, M. O. 2018 Concentrations of phthalates and metals in commercially packaged sachet and plastic bottled water sold in Lagos, Nigeria. *Journal of Food Quality and Hazards Control* **5** (4), 134–139. https://doi.org/10.29252/jfqhc.5.4.4.

Dreolin, N., Aznar, M., Moret, S. & Nerin, C. 2019 Development and validation of a LC–MS/MS method for the analysis of bisphenol A in polyethylene terephthalate. *Food Chemistry* **274**, 246–253. https://doi.org/10.1016/j.foodchem.2018.08.109.

EC 2011 *Commisssion Regulation (EU) No. 10/2011 on Plastic Materials and Articles Intended to Come Into Contact with Food*. The European Commission, Brussels. Available from: https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?

EC 2020 *Directive (EU) 2020/2184 of the European Parliament and of the Council of 16 December 2020 on the Quality of Water Intended for Human Consumption (Recast) (Text with EEA Relevance)*. The European Commission, Brussels. Available from: https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX%3A32020L2184&qid=1612894729778 (accessed 9 February 2021).

ECFR 2020 *§165.110 Bottled Water. Electronic Code of Federal Regulations*. Available from: https://www.ecfr.gov/cgi-bin/text-idx?SID=a345042f6bd438d7949b0270596b60fd&mc=true&node=se21.2.165_1110&rgn=div8 (accessed 19 June 2020).

EFSA 2005a *Opinion of the Scientific Panel on Food Additives, Flavourings, Processing Aids and Materials in Contact with Food (AFC) on A Request From the Commission Related to Di-Butylphthalate (DBP) for use in Food Contact Materials.* Available from: https://efsa.onlinelibrarywiley-com.ez11.periodicos.capes.gov.br/doi/epdf/10.2903/j.efsa.2005.242 (accessed 20 June 2020).

EFSA 2005b *Opinion of the Scientific Panel on Food Additives, Flavourings, Processing Aids and Materials in Contact with Food (AFC) on a Request From the Commission Related to Di-(2-Ethylhexyl) Phthalate (DEHP) for use in Food Contact Materials.* European Food Safety Authority. Available from: https://www.efsa.europa.eu/en/efsajournal/pub/243 (accessed 20 June 2020).

EFSA 2005c *Opinion of the Scientific Panel on Food Additives, Flavourings, Processing Aids and Materials in Contact with Food (AFC) on A Request From the Commission Related to Butyl Benzyl Phthalate (BBP) for use in Food Contact Materials*. European Food Safety Authority. Available from: https://www.efsa.europa.eu/en/efsajournal/pub/241 (accessed 21 June 2020).

EFSA 2019 Update of the risk assessment of di-butylphthalate (DBP), butyl-benzyl-phthalate (BBP), bis (2-ethylhexyl) phthalate (DEHP), di-isononylphthalate (DINP) and di-isodecylphthalate (DIDP) for use in food contact materials. *EFSA Journal* **17**, e05838. Available from: https://doi-org.ez11.periodicos.capes.gov.br/10.2903/j.efsa.2019.5838 (accessed 20 June 2020).

Farahani, A., Ramezani, M., Hassan, J. & Niazi, A. 2017 In tube ultrasonic and air assisted liquid-liquid microextraction-gas chromatography-mass spectrometry determination: a novel method for the determination of phthalate esters in aqueous samples. *Journal of the Brazilian Chemical Society* **28** (6), 967–974. https://doi.org/10.21577/0103-5053.20160247.

Farajzadeh, M. A., Pezhhanfar, S. & Mohebbi, A. 2019 Development of a dispersive solid phase extraction procedure using a natural adsorbent as an efficient and costless sorbent followed by dispersive liquid–liquid microextraction. *International Journal of Environmental Analytical Chemistry* 1–14. https://doi.org/10.1080/03067319.2019.1685667.

Fard, S. M. B., Ahmadi, S. H., Hajimahmodi, M., Fazaeli, R. & Amini, M. 2020 Preparation of magnetic iron oxide nanoparticles modified with imidazolium-based ionic liquids as a sorbent for the extraction of eight phthalate acid esters in water samples followed by UPLC-MS/MS analysis: an experimental design methodology. *Analytical Methods* **12** (1), 73–84. https://doi.org/10.1039/c9ay02073j.

Feizi, N., Yamini, Y., Moradi, M. & Salamat, Q. 2017 Nanostructured Gemini-based supramolecular solvent coupled with ultrasound-assisted back extraction as a preconcentration step before GC–MS. *Journal of Separation Science* **40** (24), 4788–4795. https://doi.org/10.1002/jssc.201700548.

Fikarová, K., Cocovi-Solberg, D. J., Rosende, M., Horstkotte, B., Sklenářová, H. & Miró, M. 2019 A flow-based platform hyphenated to on-line liquid chromatography for automatic leaching tests of chemical additives from microplastics into seawater. *Journal of Chromatography A* **1602**, 160–167. https://doi.org/10.1016/j.chroma.2019.06.041.

Fumes, B. H., Silva, M. R., Andrade, F. N., Nazario, C. E. D. & Lanças, F. M. 2015 *Recent advances and future trends in new materials for sample preparation*. *TrAC Trends in Analytical Chemistry* **71**, 9–25. https://doi.org/10.1016/j.trac.2015.04.011.

Gao, Y., Xia, B., Liu, J., Ji, B., Ma, F., Ding, L., Li, B. & Zhou, Y. 2015 Development and characterization of a nanodendritic silver-based solid-phase extraction sorbent for selective enrichment of endocrine-disrupting chemicals in water and milk samples. *Analytica Chimica Acta* **900**, 76–82. https://doi.org/10.1016/j.aca.2015.10.019.

GB 9685-2016 2016 *Standards for the use of Additives for Food Contact Materials and Products*. Available from: https://www.chinesestandard.net/PDF.aspx/GB9685-2016 (accessed 18 June 2020).

Gerba, C. P., Rose, J. B. & Haas, C. N. 1996 Sensitive populations: who is at the greatest risk? *International Journal of Food Microbiology* **30** (1–2), 113–123. https://doi.org/10.1016/0168-1605(96)00996-8.

Goeury, K., Duy, S. V., Munoz, G., Prévost, M. & Sauvé, S. 2019 Analysis of Environmental Protection Agency priority endocrine disruptor hormones and bisphenol A in tap, surface and wastewater by online concentration liquid chromatography tandem mass spectrometry. *Journal of Chromatography A* **1591**, 87–98. https://doi.org/10.1016/j.chroma.2019.01.016.

González-Sálamo, J., Socas-Rodríguez, B., Hernández-Borges, J. & Rodríguez-Delgado, M. Á. 2017 Determination of phthalic acid esters in water samples using core-shell poly (dopamine) magnetic nanoparticles and gas chromatography tandem mass spectrometry. *Journal of Chromatography A* **1530**, 35–44. https://doi.org/10.1016/j.chroma.2017.11.013.

González-Sálamo, J., González-Curbelo, M. Á., Socas-Rodríguez, B., Hernández-Borges, J. & Rodríguez-Delgado, M. Á. 2018 Determination of phthalic acid esters in water samples by hollow fiber liquid-phase microextraction prior to gas chromatography tandem mass spectrometry. *Chemosphere* **201**, 254–261. https://doi.org/10.1016/j.chemosphere.2018.02.180.

Gorji, S., Bahram, M. & Biparva, P. 2019 Optimized stir bar sorptive extraction based on self-magnetic nanocomposite monolithic kit for determining bisphenol A in bottled mineral water and bottled milk samples. *Analytical and Bioanalytical Chemistry Research* **6** (1), 137–156. https://pdfs.semanticscholar.org/e27c/30138336e1c6bba176b9d20dfa4ceadde943.pdf.

Grinbaum, M., Camponovo, A., Desseigne, J. M., Poupault, P., Meisterman, E., Chatelet, B., Davaux, F. & Lempereur, V. 2019 Phthalates: potential sources and control measures. *BIO Web of Conferences* **12**, 04008. https://doi.org/10.1051/bioconf/20191204008.

Guart, A., Bono-Blay, F., Borrell, A. & Lacorte, S. 2011 Migration of plasticizersphthalates, bisphenol A and alkylphenols from plastic containers and evaluation of risk. *Food Additives and Contaminants* **28** (5), 676–685. https://doi.org/10.1080/19440049.2011.555845.

Guart, A., Bono-Blay, F., Borrell, A. & Lacorte, S. 2014 Effect of bottling and storage on the migration of plastic constituents in Spanish bottled waters. *Food Chemistry* **156**, 73–80. https://doi.org/10.1016/j.foodchem.2014.01.075.

Hashemi-Moghaddam, H. & Maddah, S. 2018 Coating of optical fiber with a smart thermosensitive polymer for the separation of phthalate esters by solid-phase microextraction. *Journal of Separation Science* **41** (4), 886–892. https://doi-org.ez11.periodicos.capes.gov.br/10.1002/jssc.201700994.

Hassan, S., Ali, R., Shah, D., Sajjad, N. & Qadir, J. 2020 Bisphenol A and phthalates exhibit similar toxicogenomics and health effects. In: *Handbook of Research on Environmental and Human Health Impacts of Plastic Pollution* (A. W. Khursheed, A. Lutfah, & S. M. Zuber, eds). Hershey, IGI Global, Engineering Science Reference, pp. 263–287.

Hossain, M. A., Rahman, M. M., Murrill, M., Das, B., Roy, B., Dey, S., Maity, D. & Chakraborti, D. 2013 Water consumption patterns and factors contributing to water consumption in arsenic affected population of rural West Bengal, India. *Science of the Total Environment* **463**, 1217–1224. https://doi.org/10.1016/j.scitotenv.2012.06.057.

Hussain, S. Z. & Maqbool, K. 2014 GC-MS: principle, technique and its application in food science. *International Journal of Current Science* **13**, 116–126. http://www.currentsciencejournal.info/issuespdf/Syed%2014.pdf.

IBWA 2018 *Statistics: 2017 Market Report findings*. International Bottled Water Association. Virginia, United States. Available from: https://www.bottledwater.org/eco-nomics/industry-statistics (accessed 12 June 2020).

IBWA 2019 *IBWA Buyers' Guide Edition July/August 2019*. International Bottled Water Association. Virginia, United States. Available from: https://www.bottledwater.org/newsroom/bottled-water-reporter (accessed 13 June 2020).

Jain, B., Singh, A. K. & Susan, M. A. B. H. 2019 The world around bottled water. In: *Bottled and Packaged Water* (A. Grumezescu & A. M. Holban, eds). Woodhead Publishing, Kidlington, UK, pp. 39–62.

Jalili, V., Barkhordari, A. & Ghiasvand, A. 2020 New extraction media in microextraction techniques. A review of reviews. *Microchemical Journal* **153**, 104386. https://doi.org/10.1016/j.microc.2019.104386.

Jeddi, M. Z., Rastkari, N., Ahmadkhaniha, R. & Yunesian, M. 2015 Concentrations of phthalates in bottled water under common storage conditions: do they pose a health risk to children? *Food Research International* **69**, 256–265. https://doi.org/10.1016/j.foodres.2014.11.057.

Jeddi, M. Z., Rastkari, N., Ahmadkhaniha, R. & Yunesian, M. 2016 Endocrine disruptor phthalates in bottled water: daily exposure and health risk assessment in pregnant and lactating women. *Environmental Monitoring and Assessment* **188** (9), 534. https://doi.org/10.1007/s10661-016-5502-1.

Jnr, A. K. L., Yunana, D., Kamsouloum, P., Webster, M., Wilson, D. C. & Cheeseman, C. 2018 Recycling waste plastics in developing countries: use of low-density polyethylene water sachets to form plastic bonded sand blocks. *Waste Management* **80**, 112–118. https://doi.org/10.1016/j.wasman.2018.09.003.

Karayaka, S., Chormey, D. S., Fırat, M. & Bakırdere, S. 2019 Determination of endocrine disruptive phenolic compounds by gas chromatography mass spectrometry after multivariate optimization of switchable liquid-liquid microextraction and assessment of green profile. *Chemosphere* **235**, 205–210. https://doi.org/10.1016/j.chemosphere.2019.06.079.

Kaykhaii, M., Yavari, E., Sargazi, G. & Ebrahimi, A. K. 2020 Highly sensitive determination of bisphenol A in bottled water samples by HPLC after its extraction by a novel Th-MOF pipette-tip micro-SPE. *Journal of Chromatographic Science* **58** (4), 373–382. https://doi.org/10.1093/chromsci/bmz111.

Keresztes, S., Tatár, E., Czégény, Z., Záray, G. & Mihucz, V. G. 2013 Study on the leaching of phthalates from polyethylene terephthalate bottles into mineral water. *Science of the Total Environment* **458**, 451–458. https://doi.org/10.1016/j.scitotenv.2013.04.056.

Kim, Y. J. & Ryu, J. C. 2006 Evaluation of estrogenic effects of phthalate analogues using in vitro and in vivo screening assays. *Molecular & Cellular Toxicology* **2** (2), 106–113. https://www.koreascience.or.kr/article/JAKO200617317912011.pdf.

Kumar, R., Gaurav, H., Malik, A. K., Kabir, A. & Furton, K. G. 2014 Efficient analysis of selected estrogens using fabric phase sorptive extraction and high performance liquid chromatography-fluorescence detection. *Journal of Chromatography A* **1359**, 16–25. https://doi.org/10.1016/j.chroma.2014.07.013.

Legler, J., van den Brink, C. E., Brouwer, A., Murk, A. J., van der Saag, P. T., Vethaak, A. D. & van der Burg, B. 1999 Development of a stably transfected estrogen receptor-mediated luciferase reporter gene assay in the human t47d breast cancer cell line. *Toxicological Sciences* **48** (1), 55–66. https://doi.org/10.1093/toxsci/48.1.55.

Lertsirisopon, R., Soda, S., Sei, K. & Ike, M. 2009 Abiotic degradation of four phthalic acid esters in aqueous phase under natural sunlight irradiation. *Journal of Environmental Sciences* **21** (3), 285–290. https://doi.org/10.1016/S1001-0742(08)62265-2.

Leung, H. W., Jin, L., Wei, S., Tsui, M. M. P., Zhou, B., Jiao, L., Cheung, P. C., Chun, Y. K., Murphy, M. B. & Lam, P. K. S. 2013 Pharmaceuticals in tap water: human health risk assessment and proposed monitoring framework in China. *Environmental Health Perspectives* **121** (7), 839–846. https://doi-org.ez11.periodicos.capes.gov.br/10.1289/ehp.1206244.

Li, H., Li, C., An, L., Deng, C., Su, H., Wang, L., Zhang, C. & Jin, F. 2019 Phthalate esters in bottled drinking water and their human exposure in Beijing, China. *Food Additives & Contaminants: Part B* **12** (1), 1–9. https://doi.org/10.1080/19393210.2018.1495272.

Liu, Z. H., Ito, M., Kanjo, Y. & Yamamoto, A. 2009 Profile and removal of endocrine disrupting chemicals by using an ER/AR competitive ligand binding assay and chemical analyses. *Journal of Environmental Sciences* **21** (7), 900–906. https://doi.org/10.1016/S1001-0742(08)62356-6.

Liu, Z., Li, Y., Sun, L., Yang, H., Zheng, X. & Wang, L. 2019 Investigation of diazo-derivatization of bisphenol A and its applicability for quantitation in food safety inspections using high-performance liquid chromatography. *Biomedical Chromatography* **33** (3), e4419. https://doi-org.ez11.periodicos.capes.gov.br/10.1002/bmc.4419.

Luo, Q., Liu, Z. H., Yin, H., Dang, Z., Wu, P. X., Zhu, N. W., Lin, Z. & Liu, Y. 2018 Migration and potential risk of trace

phthalates in bottled water: a global situation. *Water Research* **147**, 362–372. https://doi.org/10.1016/j.watres.2018.10.002.

Manzo, V., Becerra-Herrera, M., Arismendi, D., Molina-Balmaceda, A., Caraballo, M. A. & Richter, P. 2019 Rotating-disk sorptive extraction coupled to gas chromatography mass spectrometry for the determination of phthalates in bottled water. *Analytical Methods* **11** (48), 6111–6118. https://doi.org/10.1039/C9AY02076D.

Mohammadnezhad, N., Matin, A. A., Samadi, N., Shomali, A. & Valizadeh, H. 2017 Ionic liquid-bonded fused silica as a new solid-phase microextraction fiber for the liquid chromatographic determination of bisphenol A as an endocrine disruptor. *Journal of AOAC International* **100** (1), 218–223. https://doi.org/10.5740/jaoacint.16-0189.

Mohebbi, M., Heydari, R. & Ramezani, M. 2017 Solvent-vapor-assisted liquid–liquid microextraction: a novel method for the determination of phthalate esters in aqueous samples using GC–MS. *Journal of Separation Science* **40** (22), 4394–4402. https://doi-org.ez11.periodicos.capes.gov.br/10.1002/jssc.201700755.

McGowin, A. E. 2006 Polycyclic aromatic hydrocarbons. In: *Chromatographic Analysis of the Environment* (L. M. L. Nollet & D. A. Lambropoulou, eds). CRC Press, Taylor and Francis Group, LLC, Boca Raton, pp. 555–616.

NJDEP 2004 *Ground Water Quality Standards. N.J.A.C. 7:9-6 as N.J.A.C. 7:9C*. New Jersey Department of Environmental Protection, New Jersey, United States. Available from: https://www.nj.gov/dep/rules/adoptions/7.9-6_7.9C.pdf (accessed 3 July 2020).

Nollet, L. M. 2005 *Chromatographic Analysis of the Environment*. Taylor & Francis – CRC Press, London.

Notardonato, I., Russo, M. V. & Avino, P. 2018 Phthalates and bisphenol-A residues in water samples: an innovative analytical approach. Rendiconti Lincei. *Scienze Fisiche e Naturali* **29** (4), 831–840. https://doi.org/10.1007/s12210-018-0745-0.

Oca, M. L., Ortiz, M. C., Herrero, A. & Sarabia, L. A. 2013 Optimization of a GC/MS procedure that uses parallel factor analysis for the determination of bisphenols and their diglycidyl ethers after migration from polycarbonate tableware. *Talanta* **106**, 266–280. https://doi.org/10.1016/j.talanta.2012.10.086.

Özer, E. T., Osman, B. & Yazıcı, T. 2017 Dummy molecularly imprinted microbeads as solid-phase extraction material for selective determination of phthalate esters in water. *Journal of Chromatography A* **1500**, 53–60. https://doi.org/10.1016/j.chroma.2017.04.013.

Pacyga, D. C., Sathyanarayana, S. & Strakovsky, R. S. 2019 Dietary predictors of phthalate and bisphenol exposures in pregnant women. *Advances in Nutrition* **10** (5), 803–815. https://doi.org/10.1093/advances/nmz029.

Parks, W. S., Mirecki, J. E. & Kingsbury, J. A. 1993 *Hydrogeology, Ground-Water Quality, and Potential for Water-Supply Contamination Near an Abandoned Wood-Preserving Plant Site at Jackson, Tennessee*. Memphis, Tennessee.

Pignotti, E., Farré, M., Barceló, D. & Dinelli, E. 2017 Occurrence and distribution of six selected endocrine disrupting compounds in surface- and groundwaters of the Romagna area (North Italy). *Environmental Science and Pollution Research* **24** (26), 21153–21167. https://doi.org/10.1007/s11356-017-9756-0.

Pinsrithong, S. & Bunkoed, O. 2018 Hierarchical porous nanostructured polypyrrole-coated hydrogel beads containing reduced graphene oxide and magnetite nanoparticles for extraction of phthalates in bottled drinks. *Journal of Chromatography A* **1570**, 19–27. https://doi.org/10.1016/j.chroma.2018.07.074.

Plastics Europe 2020 *Plastics – the Facts 2019. An Analysis of European Plastics Production, Demand and Waste Data*. Brussels, Belgium. Available from: https://www.plasticseurope.org/download_file/force/3183/181 (accessed 10 April 2020).

Płotka-Wasylka, J., Rutkowska, M., Owczarek, K., Tobiszewski, M. & Namieśnik, J. 2017 Extraction with environmentally friendly solvents. *TrAC Trends in Analytical Chemistry* **91**, 12–25. https://doi.org/10.1016/j.trac.2017.03.006.

PubChem 2020 *Explore Chemistry*. Available from: https://pubchem-ncbi-nlm-nih.ez11.periodicos.capes.gov.br/ (accessed 15 January 2020).

Rowell, C., Kuiper, N. & Preud'Homme, H. 2016 Is container type the biggest predictor of trace element and BPA leaching from drinking water bottles? *Food Chemistry* **202**, 88–93. https://doi.org/10.1016/j.foodchem.2016.01.109.

Rozaini, M. N. H., Yahaya, N., Saad, B., Kamaruzaman, S. & Hanapi, N. S. M. 2017 Rapid ultrasound assisted emulsification micro-solid phase extraction based on molecularly imprinted polymer for HPLC-DAD determination of bisphenol A in aqueous matrices. *Talanta* **171**, 242–249. https://doi.org/10.1016/j.talanta.2017.05.006.

Saeed, M., Niaz, A., Shah, A., Afridi, H. I. & Rauf, A. 2010 Fast voltammetric assay of water soluble phthalates in bottled and coolers water. *Analytical Methods* **2** (7), 844–850. https://doi.org/10.1039/C0AY00156B.

Sajid, M., Basheer, C., Alsharaa, A., Narasimhan, K., Buhmeida, A., Al Qahtani, M. & Al-Ahwal, M. S. 2016 Development of natural sorbent based micro-solid-phase extraction for determination of phthalate esters in milk samples. *Analytica Chimica Acta* **924**, 35–44. https://doi.org/10.1016/j.aca.2016.04.016.

Salazar-Beltrán, D., Hinojosa-Reyes, L., Ruiz-Ruiz, E., Hernández-Ramírez, A. & Guzmán-Mar, J. L. 2017 Determination of phthalates in bottled water by automated on-line solid phase extraction coupled to liquid chromatography with UV detection. *Talanta* **168**, 291–297. https://doi.org/10.1016/j.talanta.2017.03.060.

Santana-Mayor, Á., Socas-Rodríguez, B., del Mar Afonso, M., Palenzuela-López, J. A. & Rodríguez-Delgado, M. Á. 2018 Reduced graphene oxide-coated magnetic-nanoparticles as sorbent for the determination of phthalates in environmental samples by micro-dispersive solid-phase extraction followed by ultra-high-performance liquid chromatography tandem

mass spectrometry. *Journal of Chromatography A* **1565**, 36–47. https://doi.org/10.1016/j.chroma.2018.06.031.

Schilirò, T., Porfido, A., Longo, A., Coluccia, S. & Gilli, G. 2013 The E-screen test and the MELN gene-reporter assay used for determination of estrogenic activity in fruits and vegetables in relation to pesticide residues. *Food and Chemical Toxicology* **62**, 82–90. https://doi.org/10.1016/j.fct.2013.07.067.

Schmid, P., Kohler, M., Meierhofer, R., Luzi, S. & Wegelin, M. 2008 Does the reuse of PET bottles during solar water disinfection pose a health risk due to the migration of plasticisers and other chemicals into the water? *Water Research* **42** (20), 5054–5060. https://doi.org/10.1016/j.watres.2008.09.025.

Semey, M. D., Dotse-Gborgbortsi, W., Dzodzomenyo, M. & Wright, J. 2020 Characteristics of packaged water production facilities in Greater Accra, Ghana: implications for water safety and associated environmental impacts. *Journal of Water, Sanitation and Hygiene for Development* **10** (1), 146–156. https://doi.org/10.2166/washdev.2020.110.

Soares, J., Coimbra, A. M., Reis-Henriques, M. A., Monteiro, N. M., Vieira, M. N., Oliveira, J. M. A., Guedes-Dias, P., Fontaínhas-Fernandes, A., Parra, S. S., Carvalho, A. P., Castro, L. F. C. & Santos, M. M. 2009 Disruption of zebrafish (*Danio rerio*) embryonic development after full life-cycle parental exposure to low levels of ethinylestradiol. *Aquatic Toxicology* **95** (4), 330–338. https://doi.org/10.1016/j.aquatox.2009.07.021.

Soheilifar, S., Rajabi-Moghaddam, M. J., Karimi, G., Mohammadinejad, A., Motamedshariaty, V. S. & Mohajeri, S. A. 2018 Application of molecularly imprinted polymer in solid-phase microextraction coupled with HPLC-UV for analysis of dibutyl phthalate in bottled water and soft drink samples. *Journal of Liquid Chromatography & Related Technologies* **41** (9), 552–560. https://doi.org/10.1080/10826076.2018.1488138.

Sulentic, R. O., Dumitrascu, I., Deziel, N. C. & Gurzau, A. E. 2018 Phthalate exposure from drinking water in Romanian adolescents. *International Journal of Environmental Research and Public Health* **15** (10), 2109. https://doi.org/10.3390/ijerph15102109.

Surhio, M. A., Talpur, F. N., Nizamani, S. M., Talpur, M. K., Afridi, H. I., Khaskheli, A. A., Bhurgri, S. & Surhio, J. A. 2017 Leaching of phthalate esters from different drinking stuffs and their subsequent biodegradation. *Environmental Science and Pollution Research* **24** (22), 18663–18671. https://doi.org/10.1007/s11356-017-9470-y.

Szendi, K., Gyöngyi, Z., Kontár, Z., Gerencsér, G., Berényi, K., Hanzel, A., Fekete, J., Kovács, A. & Varga, C. 2018 Mutagenicity and phthalate level of bottled water under different storage conditions. *Exposure and Health* **10** (1), 51–60. https://doi.org/10.1007/s12403-017-0246-x.

Tran-Lam, T. T., Dao, Y. H., Nguyen, D. T., Ma, H. K., Pham, T. Q. & Le, G. T. 2018 Optimization of sample preparation for detection of 10 phthalates in non-alcoholic beverages in Northern Vietnam. *Toxics* **6** (4), 69. https://doi.org/10.3390/toxics6040069.

Tri, T. M., Anh, N. T. N., Thao, P. T. P. & Trung, N. Q. 2018 Determination and distribution of phthalate diesters in plastic bottled beverages collected in Hanoi, Vietnam. *VNU Journal of Science: Natural Sciences and Technology* **34** (4), 97–104. https://doi.org/10.25073/2588-1140/vnunst.4822.

Uansiri, S., Vichapong, J. & Kanchanamayoon, W. 2016 Ultrasound-assisted low density solvent based dispersive liquid-liquid microextraction for determination of phthalate esters in bottled water samples. *Chemical Research in Chinese Universities* **32** (2), 178–183. https://doi.org/10.1007/s40242-016-5343-z.

US EPA 1987a *Integrated Risk Information System (IRIS) – Dibutyl phthalate (CASRN 84-74-2)*. United States Environmental Protection Agency, Washington, United States. Available at: https://cfpub.epa.gov/ncea/iris/iris_documents/documents/subst/0038_summary.pdf (accessed 3 July 2020).

US EPA 1987b *Integrated Risk Information System (IRIS), Di (2-ethylhexyl) phthalate (CASRN 117-81-7)*. United States Environmental Protection Agency, Washington, United States. Available at: https://cfpub.epa.gov/ncea/iris/iris_documents/documents/subst/0014_summary.pdf (accessed 3 July 2020).

US EPA 1988 *Integrated Risk Information System (IRIS), Butyl benzyl phthalate (CASRN 85-68-7)*. United States Environmental Protection Agency, Washington, United States. Available at: https://cfpub.epa.gov/ncea/iris/iris_documents/documents/subst/0293_summary.pdf (accessed 4 July 2020).

US EPA 2011 *Exposure Factors Handbook 2011 Edition (Final Report)*. United States Environmental Protection Agency. Washington, United States. Available at: https://cfpub.epa.gov/ncea/risk/recordisplay.cfm?deid=236252 (accessed 5 July 2020).

US EPA 2019a *Proposed Designation of Butyl Benzyl Phthalate (CASRN 85-68-7) as a High-Priority Substance for Risk Evaluation*. United States Environmental Protection Agency. Washington, United States. Available from: https://www.epa.gov/sites/production/files/2019-08/documents/butylbenzylphthalate_85-68-7_highpriority_proposeddesignation_082319.pdf (accessed 3 July 2020).

US EPA 2019b *Update for Chapter 3 of the Exposure Factors Handbook Ingestion of Water and Other Select Liquids*. U.S. EPA Office of Research and Development, Washington, DC. EPA/600/R-18/259F, 2019. United States Environmental Protection Agency. Available from: https://www.epa.gov/sites/production/files/2019-02/documents/efh__chapter_3_update.pdf (accessed 3 July 2020).

US EPA 2021 *Ground Water and Drinking Water*. National Primary Drinking Water Regulations. Washington, United States. Available from: https://www.epa.gov/ground-water-and-drinking-water/nationalprimary-drinking-water-regulations (accessed 11 March 2021).

Waksmundzka-Hajnos, M. & Sherma, J. 2010 Overview of the field of high performance liquid chromatography in phytochemical analysis and the structure of the book. In: *High Performance Liquid Chromatography in Phytochemical Analysis* (M. Waksmundzka-Hajnos & J. Sherma, eds). CRC Press, Boca Raton.

Wang, H., Liu, Z. H., Tang, Z., Zhang, J., Yin, H., Dang, Z., Wu, P. & Liu, Y. 2020 Bisphenol analogues in Chinese bottled water: quantification and potential risk analysis. *Science of The Total Environment* **713**, 136583. https://doi.org/10.1016/j.scitotenv.2020.136583.

Wee, S. Y. & Aris, A. Z. 2019 Occurrence and public-perceived risk of endocrine disrupting compounds in drinking water. *NPJ Clean Water* **2** (1), 1–14. https://doi-org.ez11.periodicos.capes.gov.br/10.1038/s41545-018-0029-3.

Wei, S. L., Liu, W. T., Huang, X. C. & Ma, J. K. 2018 Preparation and application of a magnetic plasticizer as a molecularly imprinted polymer adsorbing material for the determination of phthalic acid esters in aqueous samples. *Journal of Separation Science* **41** (19), 3806–3814. https://doi-org.ez11.periodicos.capes.gov.br/10.1002/jssc.201800535.

WHO 2005 *Sustainable Development and Healthy Environments Cluster. Nutrients in Drinking Water.* World Health Organization, Geneva. Available from: https://apps.who.int/iris/handle/10665/43403 (accessed 4 July 2020).

WHO 2017 *Guidelines for Drinking-Water Quality. Fourth Edition, Incorporating the First Addendum.* World Health Organization, Geneva. Available from: https://apps.who.int/iris/bitstream/handle/10665/254637/9789241549950-eng.Pdf;jsessionid=4DF865392EEC754552D7D46490287573?sequence=1 (accessed 8 January 2020).

Wu, H., Wu, L. H., Wang, F., Gao, C. J., Chen, D. & Guo, Y. 2019 Several environmental endocrine disruptors in beverages from South China: occurrence and human exposure. *Environmental Science and Pollution Research* **26** (6), 5873–5884. https://doi.org/10.1007/s11356-018-3933-7.

Yang, J. F., Yang, L. M., Zheng, L. Y., Ying, G. G., Liu, C. B. & Luo, S. L. 2017 Phthalates in plastic bottled non-alcoholic beverages from China and estimated dietary exposure in adults. *Food Additives and Contaminants: Part B* **10** (1), 44–50. https://doi.org/10.1080/19393210.2016.1245679.

Yin, S., Yang, Y., Yang, D., Li, Y., Jiang, Y., Wu, L. & Sun, C. 2019 Determination of 11 phthalate esters in beverages by magnetic solid-phase extraction combined with high-performance liquid chromatography. *Journal of AOAC International* **102** (5), 1624–1631. https://doi.org/10.1093/jaoac/102.5.1624.

Yousefi, Z., Ala, A., Babanezhad, E. & Ali Mohammadpour, R. 2019 Evaluation of exposure to phthalate esters through the use of various brands of drinking water bottled in polyethylene terephthalate (PET) containers under different storage conditions. *Environmental Health Engineering and Management Journal* **6** (4), 247–255. https://doi.org/10.15171/EHEM.2019.28.

Zaki, G. & Shoeib, T. 2018 Concentrations of several phthalates contaminants in Egyptian bottled water: effects of storage conditions and estimate of human exposure. *Science of the Total Environment* **618**, 142–150. https://doi.org/10.1016/j.scitotenv.2017.10.337.

Zhang, S. H., Shen, Y. X., Li, L., Fan, T. T., Wang, Y. & Wei, N. 2018 Phthalate exposure and high blood pressure in adults: a cross-sectional study in China. *Environmental Science and Pollution Research* **25** (16), 15934–15942. https://doi.org/10.1007/s11356-018-1845-1.

Zhou, J., Chen, X. H., Pan, S. D., Wang, J. L., Zheng, Y. B., Xu, J. J., Zhao, Y. G., Cai, Z. X. & Jin, M. C. 2019 Contamination status of bisphenol A and its analogues (bisphenol S, F and B) in foodstuffs and the implications for dietary exposure on adult residents in Zhejiang Province. *Food Chemistry* **294**, 160–170. https://doi.org/10.1016/j.foodchem.2019.05.022.

First received 2 September 2020; accepted in revised form 29 March 2021. Available online 8 April 2021