Journal of Hazardous Materials 392 (2020) 122271

Contents lists available at ScienceDirect

Journal of Hazardous Materials

journal homepage: www.elsevier.com/locate/jhazmat

Review

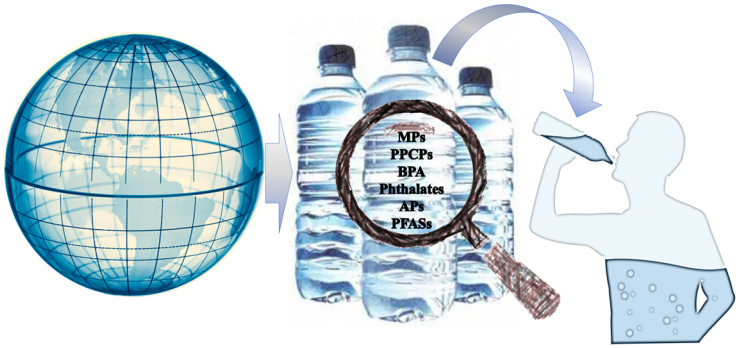
Worldwide bottled water occurrence of emerging contaminants: A review of the recent scientific literature

Razegheh Akhbarizadeh[a,b], Sina Dobaradaran[a,b,c,d,*], Torsten C. Schmidt[d], Iraj Nabipour[b], Jörg Spitz[e]

[a] Systems Environmental Health and Energy Research Center, The Persian Gulf Biomedical Sciences Research Institute, Bushehr University of Medical Sciences, Bushehr, Iran
[b] The Persian Gulf Marine Biotechnology Research Center, The Persian Gulf Biomedical Sciences Research Institute, Bushehr University of Medical Sciences, Bushehr, Iran
[c] Department of Environmental Health Engineering, Faculty of Health, Bushehr University of Medical Sciences, Bushehr, Iran
[d] Instrumental Analytical Chemistry and Centre for Water and Environmental Research (ZWU), Faculty of Chemistry, University of Duisburg-Essen, Essen, Germany
[e] Akademie für Menschliche Medizin GmbH, Krauskopfallee 27, 65388, Schlangenbad, Germany

Keywords:
Microplastics
Pharmaceuticals and personal care products
Bisphenol A
Phthalates
Alkylphenols
Perfluoroalkyl and polyfluoroalkyl substances

GRAPHICAL ABSTRACT



ARTICLE INFO

Editor: G. Lyberatos

ABSTRACT

Contaminants of emerging concern (CECs) have recently been detected in bottled water and have brought about discussions on possible risks for human health. However, a systematic review of CECs in bottled water is currently lacking due to the relatively new introduction and/or detection of these pollutants. Hence, this paper reviews the existing studies on the presence of six major groups of emerging contaminants including microplastics (MPs), pharmaceuticals and personal care products (PPCPs), bisphenol A (BPA), phthalates, alkylphenols (APs), and perfluoroalkyl and polyfluoroalkyl substances (PFASs) in bottled water from different countries. Also, the findings related to CECs' levels, their possible sources, and their risks are summarized. The gathered data indicate that MPs within the size range of 1 − 5 μm are the most predominant and potentially toxic classes of MPs

* Corresponding author at: Systems Environmental Health and Energy Research Center, Boostan 19 Alley, Imam Khomeini Street, Bushehr, 7514763448, Iran.
E-mail address: s.dobaradaran@bpums.ac.ir (S. Dobaradaran).

https://doi.org/10.1016/j.jhazmat.2020.122271
Received 26 November 2019; Received in revised form 21 January 2020; Accepted 10 February 2020
Available online 14 February 2020

in bottled water. In addition, PPCPs, PFASs, APs, and BPA occur in concentration levels of ng/L, while phthalates occur in the μg/L level in bottled water. The bottle type plays an important role in the contamination level. As expected, water in plastic bottles with plastic caps is more polluted than in glass bottles. However, other sources of contamination such as contact materials during cleaning, bottling, and storage are not negligible. Based on the gathered data in this review, the CEC levels except for MPs (no threshold values) in bottled water of most countries do not raise a safety concern for the human. However, the occurrence of individual CECs and their association in bottled water need more accurate data to understand their own/synergistic effects on human health.

1. Introduction

During the last decades, industries development and population growth led to the increase of anthropogenic contaminant emissions into the environment (Starling et al., 2019). Simultaneously, improvement of analytical techniques allowed the identification of these contaminants in various environmental media and foods (Starling et al., 2019; Wilkinson et al., 2017). Consequently, the presence of contaminants of emerging concern (CECs) (a variety of newly identified anthropogenic-source contaminants) such as microplastics (MPs), pharmaceuticals and personal care products (PPCPs), bisphenol A (BPA), phthalates, alkylphenols, and perfluoroalkyl and polyfluoroalkyl substances (PFASs) in different media and food could be detected and became a topic of public concern (Starling et al., 2019; Wilkinson et al., 2017; Gogoi et al., 2018). CECs may be industrial in origin or may originate from domestic, agricultural, hospital, and laboratory wastewater (Gogoi et al., 2018). In addition, these chemical compounds, their metabolites, and transformation products could be released into the environment from different pathways and both nonpoint (atmospheric deposition, septic tanks, stormwater discharge, agricultural runoff, and absorbing wells) and point (industrial, municipal, and hospital wastewater treatment plants) sources (Starling et al., 2019; Abtahi et al., 2019). Emerging contaminants are present and have been found in different environmental compartments such as soil, sediment, food, surface water, groundwater, and drinking water (Starling et al., 2019; Gogoi et al., 2018; Dobaradaran et al., 2018; Akhbarizadeh et al., 2019; Corsolini and Sara, 2017; Petersen, 2016; Eriksson et al., 2013; Hantoro et al., 2019; Ahrens et al., 2016). The fate of each CEC in the environment (sorption, dilution, photolysis, hydrolysis, oxidation, biodegradation, and complexation) depends on physicochemical properties as well as the interaction of the contaminant with different matrix of the environment (Starling et al., 2019).

Over the decades, the bottled water industry has been steadily developing with a global production of >6000 million gallons per year in 2015 (Luo et al., 2018a). Bottled water, a general name for natural mineral water, spring water, and treated water (i.e. bottled waters that are neither spring water nor natural mineral water) has emerged as popular drinking water in many areas (Guart et al., 2014; Rowell et al., 2016; Lardy-Fontan et al., 2017). These products are used both directly (drinking water) and indirectly (preparation of food products) as a part of human nourishment (Gellrich et al., 2013). In most parts of the world, bottled water is more appropriate than tap water due to its safety, taste, purity, quality, and portability (Luo et al., 2018a; Salazar-Beltrán et al., 2018; Santana et al., 2014). These waters are distinguished by their treatment, mineral composition, sources and quality requirements (Lardy-Fontan et al., 2017). In terms of packaging, two important types of bottles are glass and plastic (Salazar-Beltrán et al., 2018). Plastic bottles are mainly made of polyethylene terephthalate (PET), polycarbonate (PC), and high-density polyethylene (HDPE), while their caps are made of HDPE, low-density polyethylene (LDPE), and polystyrene (PS) (Guart et al., 2011). Different polymeric materials, bottle formats, shapes, and colors are used in the different bottling industries (Guart et al., 2011). Despite various efforts to keep the water clean and safe, the possibility of contamination during different stages of production is undeniable (Guart et al., 2014; Al-Saleh et al., 2011). Migration can occur between water and bottle materials and consumers

Table 1
The abundance and size distribution of MPs in bottled water.

Type of bottles	Total number of samples	Size distribution					Reference
		6.5- 100 μm				>100 μm	
Plastic bottles, brand Aqua	295	374				8	(Mason et al., 2018)
Plastic bottles, brand Aquafina		200				13	
Plastic bottles, brand Bisleri		338				9	
Plastic bottles, brand Dasani		109				10	
Plastic bottles, brand E-Pura		238				10	
Plastic bottles, brand Evian		114				14	
Plastic bottles, brand Gerolsteiner		1396				15	
Glass bottles, brand Gerolsteiner		159				9	
Plastic bottles, brand Minalba		63				4	
Plastic bottles, brand Nestle Pure Life		912				20	
Plastic bottles, brand San Pellegrino		27				2	
Plastic bottles, brand Wahaha		90				6	
		1−5 μm	5-10 μm	10−100 μm			
Single-use PET[1] bottles	32	2604	45	0			(Oßmann et al., 2018)
Reusable PET bottles		4664	142	83			
Glass bottles		4895	969	434			
		5-10 μm		10-20 μm	20-50 μm	50−100 μm	>100 μm
Single-use plastic bottles	38	6		4	3	2	0	(Schymanski et al., 2018)
Reusable plastic bottles		66		34	14	2	1	
Glass bottles		26		16	7	4	2	
Beverage cartons		4		3	2	1	1	
		≥ 3 μm					
Single-use PET bottles	18	148						(Winkler et al., 2019)

1: Polyethylene terephthalate.

may be exposed to the different types of chemicals including additives, processing aids, and un-reacted monomers (Bach et al., 2013).

The presence of different types of pollutants in water has been extensively studied in recent years. As to date, more than 80 studies exist that provide different CECs' concentration data in bottled water. However, a systematic review of MPs and other emerging contaminants in bottled water is currently lacking. This review critically examines the most influential research, encompassing six major groups of emerging contaminants, with a particular focus on their occurrence, association, and bio/photochemical transformation in bottled water.

2. Emerging contaminants of interest in bottled water

2.1. Microplastics

Despite the undeniable benefits of polymers in human daily activity (e.g. food packaging and management, medical devices, electrical and electronic components, etc.) there is an increasing concern regarding their possible adverse effects on human health (Zuccarello et al., 2019; Winkler et al., 2019). Recently, extensive replacement of glass and metal materials with plastics in food packaging such as bottling of water and beverages, as well as storage containers has occurred (Zuccarello et al., 2019). MPs are known as small plastic debris that are less than 5 mm in size (Koelmans et al., 2019; Mintenig et al., 2019; Arthur et al., 2009). Chemical toxicity, physical injuries, and microbial hazards are associated effects with MPs and such effects are likely to be dose-dependent (Koelmans et al., 2019). However, knowledge of their real exposure levels, translocation in a living organism, and toxic effects are currently lacking.

The presence of MPs of all sizes and shapes in various natural environments (Barboza and Gimenez, 2015; Kafilzadeh, 2015; He et al., 2018; Dehghani et al., 2017; Dris et al., 2016), terrestrial and aquatic organisms (Deng et al., 2017; Kolandhasamy et al., 2018; Ribeiro et al., 2018; Nelms et al., 2018; Ding et al., 2018; Huerta Lwanga et al., 2017), and some types of beverages and foods (Van Cauwenberghe and Janssen, 2014; Smith et al., 2018; Liebezeit et al., 2014; Kosuth et al., 2018; Wiesheu et al., 2016), has been investigated. The results of these studies and the high consumption of bottled water, raised the question if pollution of bottled drinking water also occurs?

To date, there are only a few studies that point out the presence of MPs in bottled water (Zuccarello et al., 2019; Winkler et al., 2019; Wiesheu et al., 2016; Mason et al., 2018; Oßmann et al., 2018; Schymanski et al., 2018). Wiesheu et al. (2016) in their study on the analysis of fibers in beverages, did not report significant differences between the controls and the sample, and they concluded that the detected MPs were from the background contamination of the laboratory environment (Wiesheu et al., 2016) The detailed data of these research is presented in Table 1. The highest numbers of MPs in bottled water were reported by Oßmann et al. (2018), ranging from 2649 items/L in single-use plastic bottles to 6292 items/L in glass bottles. It should be noted that most of the reported MPs by Oßmann et al. (2018) were within the size range of $1-5$ μm, while such small particle sizes were not reported by Mason et al. (2018) and Schymanski et al. (2018). Recently, Winkler et al. (2019) investigated the presence of MPs in bottled water before and after the mechanical stress (Winkler et al., 2019). They found 148 ± 253 MPs/L in single-use PET bottles before mechanical stress (the smallest analyzed size was 3 μm). These variations among published data could be explained by applied techniques and analytical limitations in different studies. The presence of PET (Winkler et al., 2019; Oßmann et al., 2018; Schymanski et al., 2018) and PE (Mason et al., 2018) in bottled water, suggest the possible degradation of packaging material: bottle and/or cap. Winkler et al. (2019) also analyzed the possibility of MPs released from PET bottlenecks and HDPE caps into water after 1, 10, and 100 times opening/closing bottles. Their results demonstrated a considerable increase of MPs (<5 μm) occurrence on the PET and HDPE materials especially after 100 times opening and closing the bottles (Winkler et al., 2019). Hence, frequent use of one single-use PET bottles, increase the chance of MPs ingestion by humans. As the plastic litter with small size (<1.5 μm) can easily pass through gut epithelial membranes (Oßmann et al., 2018; Eerkes-Medrano et al., 2019), their presence in bottled water is very important. Also, the effects of mechanical stress on MPs release from the inner surface of the PET bottles by squeezing treatment for 1 and 10 min were investigated. Surprisingly, the number of MPs in bottled water did not increase (Winkler et al., 2019). However, the presence of MPs was significantly higher in samples from the reusable (older) bottles (Oßmann et al., 2018; Schymanski et al., 2018). The high presence of fragments in bottled water and the fact that some of the particles were from cleaning, production, and refilling processes (Mason et al., 2018; Mason et al., 2018; Oßmann et al., 2018; Schymanski et al., 2018), suggest that some of the identified MPs in bottled water may be coming from the industrial processes. Moreover, the presence of plastic particles in glass bottles also suggests other sources of pollutants besides the packaging itself (Oßmann et al., 2018; Eerkes-Medrano et al., 2019).

Thermo oxidative and thermo mechanical reactions as results of the oxygen presence and high temperature in PET melt processes could enhance the migration of plastics additives and components (Guart et al., 2014). Plastic additives such as BPA, Tris(2, 4-ditert-butylphenyl) phosphite, and phthalates as well as sorbed contaminants such as heavy metals and organic pollutants could be also leached from them (Guart et al., 2014; Oßmann et al., 2018; Rocha-Santos and Duarte, 2015; Welle and Franz, 2018). There are two possible scenarios to explain the fate of MPs' additives in bottled water: 1) they could be leached from the MP particles into the water during storage (before opening the bottles), and/or 2) they could be leached after ingestion and absorbed in the consumer's body. However, regardless of the mode of action, all additives present in MPs could reach the human body after water consumption by using bottled water (Welle and Franz, 2018). Considering the fact that all plastics contain reactive oxygen species (ROS) because of their processing history and polymerization, the ingested MPs can produce oxidative stress in the human body (Winkler et al., 2019). Unfortunately, the effects of ingested MPs in human body is not investigated yet. Thus, more researches are needed to determine the risk of dietary exposure and toxicity of MPs for human.

In the case of nanoplastics (NPs-plastic debris within the size range from 1 to 100 nm), it should be noted that there is currently no information on the presence of NPs in foods and beverages (including bottled water) (Welle and Franz, 2018). However, when considering the potential sources of plastic particles in bottled water such as cleaning, production, and filling processes, it is improbable that significant amounts of NPs will be there (Welle and Franz, 2018). This hypothesis is supported by the results of Oßmann et al. (2018) study, where the number of 1–1.5 μm MPs did not show a significant increase when compared to $1.5-5$ μm MPs. However, it should be noted that most studies of MPs and NPs in bottled water ignored such small size because analytical methods to detect sub-μm particles based on Raman microscopy have only recently been developed.

2.2. Pharmaceuticals and personal care products

Pharmaceuticals are used to treat or prevent human and animal diseases, while PCPs are mainly used as substances to improve the quality of daily life (Ebele et al., 2017). To date, more than 3000 PPCPs have been used for the medication and enhancement of human as well as animal living standards (Yang et al., 2017). Possible sources of PPCPs in the environment are the discharge of hospital wastewater, domestic sewage, industrial wastewater, livestock breeding, and landfill leachates (Ebele et al., 2017; Yang et al., 2017; Xu et al., 2019; Sharma et al., 2019). As some of the PPCPs are bio-active even at low concentrations, they contain complex chemical structures, and they have potential to accumulate in organisms' body, their occurrence in

drinking water may pose health concerns (Wilkinson et al., 2017; Sharma et al., 2019; Perret et al., 2006). Despite the low concentration of PPCPs in drinking water (ng/L), some evidence showed that these contaminants (especially antibiotics) may pose serious health problems (Xu et al., 2019; Sharma et al., 2019). Glucocorticoids, sex hormones, growth hormones, and few non-steroidal pharmaceutical substances are categorized as endocrine disrupting pharmaceuticals (Ebele et al., 2017). The details of endocrine disrupting PPCPs and their synergic toxicity are presented in kinds of literature (Ebele et al., 2017; Cleuvers, 2003; Thorpe et al., 2001).

Data concerning the presence of PPCPs in bottled water are very rare and to date, only five studies focused on this important topic. This can be due to 1) very low concentration of PPCPs in bottled water and the analytical limitation to measure contaminants at ng/L or sub ng/L, and 2) the absence of defined regulatory limits for PPCPs (Devier et al., 2013). In the latest study on PPCPs in bottled water, the presence of 20 target PPCPs in 167 commercially bottled water from European countries was investigated by Lardy-Fontan et al. (2017). Only three PPCPs (metformin (12 ng/L), salicylic acid (16 ng/L), and gabapentin (10 and 12 ng/L)) were detected above the LOD, whereas all other investigated PPCPs were not detected. They concluded that low detected levels (below 20 ng/L) highlighting a negligible risk following the ingestion of PPCPs via bottled water (Lardy-Fontan et al., 2017). Devier et al. (2013) carried out a survey of pharmaceutically active compounds in 285 samples from two brands of natural mineral bottled water. Among 97 different pharmaceutical substances only salicylic acid, ketoprofen, and caffeine were detected (<1 ng/L). However, they concluded that the presence of these substances was due to the background contamination of the laboratory environment (Devier et al., 2013). González Alonso et al. (2012) performed a survey on the presence of 58 pharmaceutical substances in 10 bottled mineral water from Spain. The range of nicotine varied from 7 to 15 ng/L in 5 bottles and the concentration of other substances were not above the LOQs (González Alonso et al., 2012). In 2010, Li et al. investigated the presence of triclosan (TCS) in 21 bottled water including mineral water and pure drinking water from China. The concentration of TCS ranged from 0.6–9.7 ng/L, and its daily intake for adults through drinking bottled water was estimated to be 7 ng/day (Li et al., 2010). The potential presence of 11 sulfonamides in various Italian bottled water was also investigated by Perret et al. (2016). The range of detected sulfonamides in mineral waters was from 9 to 80 ng/L (Perret et al., 2006). Since sulfonamides (mostly used for veterinary practices) are both water soluble and polar, they have a high potential for running off to the surface waters and/or leaching to groundwaters. Consequently, they can easily lead to contamination of drinking water (i.e. natural mineral waters) (Yang et al., 2017). Although PPCPs have been entering the environment for many years and it is expected that some of them would have a tendency to be present in groundwater (drinking water) (Ebele et al., 2017; Sharma et al., 2019), but there is very little knowledge about their eco-toxicological impacts on human (Gogoi et al., 2018). The major concern about the toxicity of PPCPs is that they were mostly designed to maximize their biological activity at low doses and also they could be pharmacologically active in non-target organisms (Ebele et al., 2017). In addition, the synergistic interaction of complex mixture of PPCPs (even at low concentration) can exert considerable ecotoxicity (Ebele et al., 2017; Cleuvers, 2003). Hence, the chronic toxicity caused by PPCPs (single and mixture) could not be ignored and more accurate investigations are required to check the long-term toxicity of these compounds.

2.3. Bisphenol A and phthalates

Bisphenol A (BPA; 2,2-bis(4-hydroxyphenyl)propane) and phthalates (esters of phthalic acid-$C_6H_4(CO_2H_2)_2$) are currently applied in epoxy resins, polycarbonates, thermal paper, flame retardants, and polyacrylates (Starling et al., 2019; Abtahi et al., 2019; Salazar-Beltrán

Table 2
The level of bisphenol A (BPA) in bottled water.

Type of bottles	Total number of samples	Concentration (ng/L)	Reference
Not mentioned	Not mentioned	50 – 90	(Zhou et al., 2019a)
PET[1] bottles (natural water)	16	0.08- 6.74	(Chailurkit et al., 2017)
PE bottles (natural water)	1	20.7	
Polypropylene bottles (natural water)	4	<LOD[2]	
PET bottles (mineral water)	6	3.04–27.3	
Plastic bottles	2	<LOD	(Sakhi et al., 2014)
PET bottles	25	< LOD	(Colin et al., 2014)
New PC[3] bottles	13	250 – 4210	
Used PC bottles	14	70 – 1660	
PC bottles	22	50- 1370	(Dhaini and Nassif, 2014)
PET bottles at 4 °C	16	0.26 – 18.7	(Fan et al., 2014)
PET bottles at 25 °C	16	0.62 – 22.6	
PET bottles at 70 °C	16	2.89 – 38.9	
PET bottles	5	<0.73 – 1.13	(Maggioni et al., 2013)
PET bottles stored indoor	7	3.13 – 5.13	(Elobeid et al., 2012)
PET bottles stored outdoor	7	6.20 – 8.80	
PET bottles at 25 °C	9	3.3 ± 2.6	(Santhi et al., 2012)
PET bottles at 50 °C	9	11.3 ± 5.3	
PET bottles	131	31 – 203	(Bono-Blay et al., 2012)
PET bottles	10	<LOD	(Guart et al., 2011)
HDPE[4] bottles	7	<LOD	
PC bottles	10	1600 – 4440	
Plastic bottles	21	17.6 – 285	(Li et al., 2010)
Glass bottles	10	<LOD	(Cao and Corriveau, 2008)
Metal bottles	1	<LOD	
HDPE bottles	2	<LOD	
PET bottles	38	<LOD	
PC bottles	5	670 – 8800	
New PC bottles	3	0.08-1.33	(Le et al., 2008)
Used PC bottles	3	0.21-0.93	
New HDPE	3	0.01-0.19	

1: Polyethylene terephthalate; 2: limit of detection; 3: Polycarbonate; 4: High-density polyethylene.

et al., 2018; Zhou et al., 2019a; Colin et al., 2014; Sakhi et al., 2014). Due to the increasing popularity of plastic products and their lightweight and durability, the annual BPA production has reached more than 8 million metric tons in 2016 (Moreman et al., 2017; Almeida et al., 2018). In addition, it has been forecasted that the global consumption of phthalate plasticizers increases at the rate of 1.3 % per year during 2017–2022 (Luo et al., 2018a). It should be noted that BPA and phthalates are not chemically but only physically bound to the products and can easily and widely leach, migrate and/ or evaporate from packages to foods and air (Salazar-Beltrán et al., 2018; Sakhi et al., 2014; Prokůpková et al., 2002). Hence, food and drinking packaging could be considered as important sources of BPA and phthalates in food and beverages (Le et al., 2008; Serôdio and Nogueira, 2006; Biles et al., 1997). BPA and eight phthalates including diethyl phthalate (DEP), di-n-propyl phthalate (DPrP), dibutyl phthalate (DBP), benzyl butyl phthalate (BBP), bis-(2-ethylhexyl) phthalate (DEHP), dicyclohexyl phthalate (DCHP), dipentyl phthalate (DPP), dihexyl phthalate (DHXP) are known as an endocrine disrupting compound (EDC) that is known to interact and/or affect the thyroid receptors, male fertility, hemostasis of hormones, cancer progression, glucocorticoid receptors, and the cardiovascular system (Colin et al., 2014; Sakhi et al., 2014; Dreolin et al., 2019; Ehrlich et al., 2014; Welshons et al., 2006; Jeddi et al., 2015a). The tolerable daily intake (TDI) of BPA, DBP, BBP, and DEHP are 4, 10, 500, and 50 μg/kg-bw, respectively (Sakhi et al., 2014; EFSA, 2015; EFSA, 2005a; EFSA, 2005b; EFSA, 2005c). The BPA exposure via bottled water consumption has been estimated at 0.17 μg/day (Hernandez-Hernandez et al., 2019). Moreover, the guidelines of the World Health Organization (WHO) and the US Food and Drug Administration (US FDA) for DEHP in drinking water are 8 and 6 μg/L, respectively (Abtahi et al., 2019; Luo et al., 2018a; Zaki and Shoeib, 2018).

Till now, the presence of BPA and phthalates in bottled water has been investigated in some studies. Among them, the results of those studies that detected BPA and phthalates are presented in Tables 2 and 3.

In the case of BPA, the results of these studies confirmed that BPA can leach from PC containers under normal use conditions (Colin et al., 2014; Dhaini and Nassif, 2014; Le et al., 2008; Cao and Corriveau, 2008; Wagner and Oehlmann, 2009). Those findings might be explained by polymer degradation during transportation, storage, and recycling activities (Dhaini and Nassif, 2014). Interestingly, Colin et al. (2014) have reported that recently manufactured PC bottles had higher levels of BPA than older bottles. They suggested that the trend might have occurred as a result of faster migration of free BPA during the first year rather than next years. Furthermore, Le et al. (2008) have reported that BPA migration from PC bottles significantly increases (up to 55-fold) with temperature (100 ℃). This is caused by hydrolysis of carbonate linkage and can be avoided by sterilization below 80 ℃ (Vilarinho et al., 2019). Even though PET containers, especially those labeled as “BPA-free”, were not expected to release BPA, the results of some studies revealed that BPA could be present in all types of bottled water (Dreolin et al., 2019; Chailurkit et al., 2017; Maggioni et al., 2013; Fan et al., 2014). According to these studies, the source of detected BPA in PET bottled water could also be related to pollution of water prior to bottling, bottle closures, and use of recycled bottles (Chailurkit et al., 2017; Fan et al., 2014; Russo et al., 2019). Moreover, Fan et al. (2014) and Santhi et al. (2012) have reported that high temperature (70 ℃ and 50 ℃, respectively) induced BPA release from PET bottles. The increase in BPA concentration of bottled water after storage at high temperature suggests chemical reactions in the packaging materials (Santhi et al., 2012). In summary, most of the primary materials that are used to make plastic bottles would not be expected to comprise BPA, but cross contamination (background contamination) of BPA during the manufacture of virgin PET and also during the recycling process may be related to the water pollution (Colin et al., 2014; Dreolin et al., 2019). However, it should not be possible to assume it as a consistent source of contamination (Dreolin et al., 2019). In addition,

Table 3
The level of phthalates in bottled water.

Type of bottles	Total number of samples	Compounds	Concentration (μg/L)	Reference
PET[1] bottles	45	DEHP BBP DBP DEP DMP DnOP[3]	Mean level of total phthalates: 0.96 ± 0.1.06	(Abtahi et al., 2019)
PET bottles	60	DMP DEP DBP BBP DEHP DnOP DiBP DPrP DHXP DCHP DPP BMPP[4] DBEP[5] DPHP[6] DBzP DNP	0.010-0.41 0.011-0.071 0.021-0.51 0.019-0.032 0.013-0.021 0.015-0.039 0.049-0.40 0.011-0.016 0.017-0.022 0.013-0.019 0.015-0.021 0.012-0.020 0.021-0.027 0.026-0.031 0.031-0.048 0.019-0.027	(Li et al., 2019)
PET bottles	12	DBP DEHP	<LOD[9] – 0.171 <LOD – 0.298	(Zaki and Shoeib, 2018)
PET, PC[2] and glass bottles	12	DnBP DBP DEHP	<0.026 – 0.16 <0.005 – 0.2 <0.29 – 11.72	(Szendi et al., 2017)
PET bottles	10	DBP DMP[7]	14.5–25.6 4.1–50.6	(Salazar-Beltran et al., 2017)
PET bottles	30	DBP DEP DnOP DMP BBP DEHP	Mean: 0.198 Mean:0.098 Mean:0.039 Mean: 0.006 Mean: 0.005 Mean: 1.470	(Selvaraj et al., 2016)
PET bottles	12	DBP DEHP	Mean: 0.135 Mean: 0.217	(Jeddi et al., 2015b; Jeddi et al., 2016)
PET bottles	12	DEP	Mean: 0.231	(Jeddi et al., 2015a)
PET bottles	3	DBP DEHP	0.063 – 0.068 1.196 – 1.685	(Otero et al., 2015)
Plastic bottles	2	DEP DiBP[8] DnBP DEHP	0.037 0.079 0.34 0.17	(Sakhi et al., 2014)
Plastic and glass bottles	6	DiBP DEHP DnBP	0.06 – 6.5 0.02 – 0.18 0.1 – 1.89	(Santana et al., 2014)
PET and glass bottles	224	DEP DEHP DMP BBP	1.02–20.5 <LOD-1.52 <LOD- 0.022 0.619- 1.28	(Guart et al., 2014)
PET bottles	3	DiBP DBP BBP DEHP	<0.003- 0.2 <0.066 – 0.8 <0.06 – 0.1 <0.16 – 1.7	(Keresztes et al., 2013)
PET and PC bottles	18	DBP BBP DEHP	0.33- 4.64 0.08–52.35 3.86-15.2	(Mousa et al., 2013)
PET bottles	131	DEP	1.115	(Bono-Blay et al., 2012)
PET and PC bottles	6	DEHP DBP DEP	Median: 0.35 Median: 0.044 Median: 0.033	(Amiridou and Voutsa, 2011)
PET bottles	10	DMP DEP BBP DEHP	<LOD- 2.996 <LOD-1.778 1.194- 8.227 <LOD- 1.254	(Al-Saleh et al., 2011)
PET and glass bottles	26	DEHP	<LOD- 0.2	(Leivadara et al., 2008)

(*continued on next page*)

Table 3 (*continued*)

Type of bottles	Total number of samples	Compounds	Concentration (μg/L)	Reference
PET bottles	11	DEP	0.054-0.100	(Cao, 2008)
		DiBP	0.161-0.481	
		DBP	0.092-1.717	
		DEHP	0.052-0.338	
PET and glass bottles	142	DMP	<LOD-0.65	(Montuori et al., 2008)
		DEP	<LOD-0.8	
		DiBP	0.02-0.92	
		DBP	0.02- 0.98	
		DEHP	<LOD-0.17	
Not mentioned	3	DEP	Mean: 0.24 ± 0.02	(Regueiro et al., 2008)
Not mentioned	1	DEP	2.8	(Xu et al., 2007)
		DnBP	4.6	
Not mentioned	1	DEP	0.04	(Serôdio and Nogueira, 2006)
		DBP	0.35	
		BBP	0.02	
		DEHP	0.17	
Glass and PET bottles	2	DnBP	0.18-0.20	(Prokůpková et al., 2002)
		BBP	<LOD-0.002	
		DEHP	2.88-9.78	

1: Polyethylene terephthalate; 2: Polycarbonate; 3: Di-n-octyl phthalate; 4: Bis-methylphenyl phthalate; 5: Bis(2-butoxyethyl) phthalate; 6: Di-(2-propylheptyl) phthalate; 7: Dimethyl phthalate; 8: Di-iso-butyl phthalate; 9: limit of detection.

the sun's ultraviolet radiation and heat could cause chemical reactions in plastic bottles to be released chemicals with estrogenic activities (i.e. BPA) into the water of the bottles (Rowell et al., 2016; Chailurkit et al., 2017). Hence, care should be taken into account to avoid accidental exposure of the plastic bottles to direct sun radiation and heat during transportation and storage (Cao and Corriveau, 2008). However, more accurate research is needed to determine the parameter(s) that increase the BPA release in bottled water and also the exact health effects of the released BPA for humans. In the case of phthalates, in order to avoid misunderstanding, the data of two works from Thailand (Prapatpong and Kanchanamayoon, 2010; Kanchanamayoon et al., 2012) are not reported in Table 3 since the applied method had extremely high LODs of 25 − 50 mg/L, thus positive findings could not be expected. As presented in Table 3, the level of DEHP in some bottled water samples from Saudi Arabia (15.2 μg/L), Hungary (11.29 μg/L) and Czech Republic (9.8 μg/L) was higher than recommended values by WHO and FAD (8 and 6 μg/L). Szendi et al. (2017) reported that there were no significant differences between phthalate concentrations in different bottle types (PET and glass). However, other studies found that the concentration of phthalates in various types of bottles (i.e. PET, PC, and glass) are very different. For instance, Montuori et al. (2008) reported that the concentration of phthalates in water from PET bottles was 18 times higher than from glass bottles. In contrast, Santana et al. (2014) reported that the concentrations of DiBP and DnBP in glass bottles were higher than in PET bottles. For PET containers, the migration of phthalates from PET bottles and plastic caps could be the main source of phthalates in bottled water (Luo et al., 2018a; Li et al., 2019). While for glass bottles, migration of phthalates from metal caps that were sealed with polyvinyl chloride (PVC) inserts could be considered as the origin of contaminants (Santana et al., 2014). Besides the material of packaging, the detectable phthalates in bottled water may also originate from source water, production process, recycled bottles, and laboratory background contamination (Luo et al., 2018a; Devier et al., 2013; Szendi et al., 2017; Keresztes et al., 2013). Thus, the phthalates in bottled water do not necessarily mean that these compounds are present in packaging material (Bach et al., 2012). In conclusion, the exact sources of phthalates found in bottled water have not clearly established and should be investigated in the future.

In some previous studies, the effects of sunlight radiation, temperature, storage time, bottle size, and pH of water, on the possible migration of phthalates into the water of plastic bottles were also investigated. Jeddi et al. (2015) investigated the effects of sunlight exposure and storage duration, and reported that the concentration of DEHP level increased from 0.35 to 0.80 μg/L after 45 days sun exposure (Jeddi et al., 2015b). Similarly, Zaki and Shoeib (2018), reported that the levels of DiBP and DEHP were significantly increased after 1, 2, and 4 months of outdoor storage with direct sunlight exposure. Moreover, it seems that high temperature increases the migration of phthalates to water (Jeddi et al., 2015a; Zaki and Shoeib, 2018; Keresztes et al., 2013; Jeddi et al., 2016; Pourzamani et al., 2016; Zaater et al., 2013; Salazar-Beltran et al., 2017; Mousa et al., 2013). On the other hand, some researchers reported that the concentration of phthalate in the water of plastic bottles was significantly decreased under sunlight radiation (Luo et al., 2018a; Al-Saleh et al., 2011; Leivadara et al., 2008). The possible reason for this phenomenon (degradation of phthalates in water) is photolysis (Luo et al., 2018a; Peterson and Staples, 2003). In addition, some studies reported that the sunlight radiation, storage time and storage temperature did not affect the phthalates levels of bottled water (Guart et al., 2011; Bach et al., 2013; Devier et al., 2013; Amiridou and Voutsa, 2011). In the case of pH, most of the results indicated that higher pH leads to more pronounced phthalates migration from plastic bottles to water (Keresztes et al., 2013; Montuori et al., 2008; Rastkari et al., 2017). On the contrary, some studies reported that the acidic condition elevated the phthalates leaching from bottles (Martine et al., 2013; Bošnir et al., 2007). Based on the results, the migration of phthalates from plastic bottles to water is affected by the combined effects of various factors (Luo et al., 2018a). It should be noted that many studies had been attended in a short period of time (one or two months), but the quality warranty of most bottled water is at least one year. In this regard, Luo et al. (2018) provided some models to describe the possible migration of phthalates from bottles into water during long-time storage and different temperatures. Their results showed that the quality of water may change during the long-time storage and high temperature. In summary, there is still some disagreement regarding the effects of storage duration, pH, temperature, and sunlight exposure. Hence, further studies are needed to make these disagreements clear. It should be noted that recent toxicological studies have shown that DBP has estrogenic effects, DPrP has reproductive toxicity and decreased sperm concentration, DHXP caused testicular atrophy, and DEHP has estrogenic and carcinogenic effects (Luo et al., 2018a; Li et al., 2019). In addition, it is exhibited that the mixture of phthalates enhanced estrogenic activity (Luo et al., 2018a). Hence, considering the huge annual consumption of bottled water globally, human exposure to phthalates should be paid more attention in future studies.

2.4. Alkylphenols

Alkylphenols (APs) are a large family of organic compounds that are formed by a substituted phenolic ring and an alkyl chain (Valcarcel et al., 2018; Salgueiro-Gonzalez et al., 2017). Octylphenols (OPs) and nonylphenols (NPs) are the most common groups of APs and are widely used in a broad range of applications including pulp and paper production, pesticides, latex paints, detergents, industrial and household cleaners, plastics, oil industries, and flotation agents (Valcarcel et al., 2018; Salgueiro-Gonzalez et al., 2017; Acir and Guenther, 2018; Luo et al., 2018b). It is expected that APs' consumption is raised in the growing Asian markets (Chiu et al., 2010). OPs are already considered as priority substances and NPs are listed as priority hazardous substances in the Water Framework Directive of the European Union (Amiridou and Voutsa, 2011; Valcarcel et al., 2018; Acir and Guenther, 2018). In other words, both OP and NP, and their metabolites are well known endocrine disruptor (Luo et al., 2018b; Zhou et al., 2017). In a competitive manner, NPs can interact with the female hormonal system by binding to the estrogen receptor and displacing 17β-estradiol (Acir

and Guenther, 2018). The guidelines of the European Union for OP and NP in surface water are 100 and 300 ng/L, respectively (E.C. EC, 2015). In addition, the specified TDI value of 5 μg/kg-bw have been suggested for OP and NP (Danish, 2016).

Ingestion is the main exposure pathway for human exposure to APs and their metabolites (Acir and Guenther, 2018; Vidal et al., 2016). Consumption of water from contaminated surface water, mineral water, and groundwater and/or ingestion of polluted food from land that is irrigated with contaminated water can seriously endanger humans and wildlife (Acir and Guenther, 2018).

Several scientific studies investigated the presence of APs, and their metabolites in bottled water. Among them, the results of those studies that detected APs are presented in Table 4. In one of the latest studies, the presence of 8 types of APs were investigated in 40 bottled water samples from France, and only 2 parent compounds (OP and NP) were quantified in 2 different samples (Le Coadou et al., 2017). In another study, the mean levels of OP and NP in the bottled water from Spain were 18.5 and 177 ng/L, respectively (Fabregat-Cabello et al., 2016). While the levels of OP and NP in Argentinian bottled water were below 1 ng/L (Vidal et al., 2016). In another study, Colin et al. (2014) didn't detect APs in 25 PET bottled water, while they reported the 4-tert-butylphenol (4-terBP) in PC bottled water samples in the range of 44–247 ng/L. Li et al. (2010) also determined the level of NP in Chinese bottled water and found it in the range of 108–298 ng/L. The diversity of used methodologies and achieved LOQ is an important reason for the inconsistent results of different studies (Le Coadou et al., 2017). Moreover, the antioxidants in laboratory equipment and materials should also be considered as an undesirable source of detected APs in some analysis (Bach et al., 2012).

The effects of bottle type, cap's material, pH, and storage time of bottled water, on the possible migration of APs into the water were investigated in a study by Guart et al. (2014). Their results showed that the glass bottles with metallic crown caps and PC bottles with LDPE caps caused more migration rate of APs into water. The lower pH and higher storage time also increased the APs' level in the studied bottled water (Guart et al., 2014). In addition, the concentrations of OP and NP in various bottled water with different polymer materials were investigated by Loyo-Rosales et al. (2004). Their results showed that bottled water in PVC and HDPE containers had more OP(12 ng/L) and NP (300 ng/L), respectively (Loyo-Rosales et al., 2004). Since the APs' application in PET bottles and cleaner agents of bottling line was banned according to REACH regulation (Bergkamp, 2013), the PET bottles and bottling line could not be considered as possible sources of APs in European bottled water. However, other bottle types (i.e. PC, HDPE, and PVC containers), water contamination prior to bottling, and bottle caps could be considered as potential sources of APs in bottled water (Colin et al., 2014; Le Coadou et al., 2017; Loyo-Rosales et al., 2004; Pernica et al., 2015). Releasing APs from bottles during the washing process, bottling, transportation, and storage may vary depending on the technology as well as regulations in various countries (Pernica et al., 2015). In addition, according to Pernica et al. (2015) and Devier et al. (2013), the common samples' contamination during the analysis from laboratory air, solvent and chemicals, and septa vials could also be affected the level of APs in analyzed bottled water. Hence, more research is needed to determine the accurate level and possible sources of APs in bottled water in order to estimate the risks of bottled water for humans.

2.5. Perfluoroalkyl and polyfluoroalkyl substances

Per and polyfluoroalkyl substances (PFASs) are a large group of persistent synthetic contaminants (Dauchy, 2019; Zhou et al., 2019b). PFASs are a group of highly fluorinated aliphatic substances that are linked to a functional group with or without C—H bond (Papadopoulou et al., 2017; Zafeiraki et al., 2015). Some perfluoroalkyl acids widely used in water/grease-proof food packaging materials (e.g. polyfluoroalkyl phosphate esters and fluorotelomer alcohols) can easily migrate to food (Papadopoulou et al., 2017). The water solubility, resistance to degradation, resistance to heat, bioaccumulation potential, and biomagnification ability, make PFASs a global problem (Kabore et al., 2018; Ingelido et al., 2018; Llorca et al., 2012).

Human exposure to PFASs can occur via ingestion, inhalation, and dermal contact (Dauchy, 2019). However, recent studies demonstrated that dietary exposure is the main exposure pathway for PFASs (Zhou et al., 2019b; Ingelido et al., 2018; Thompson et al., 2011). Epidemiological and toxicological studies found that PFASs may adversely affect developmental, immune, and endocrine functions (Papadopoulou et al., 2017; Ballesteros et al., 2017; Heo et al., 2014). In 2016, the US EPA established a lifetime health advisory level at 70 ng/L in drinking water for sum of PFOS and PFOA (Dauchy, 2019). In addition, the

Table 4
The level of alkylphenols (octylphenol (OP), nonylphenol (NP), and 4-tert-butylphenol (4-terBP)) in bottled water.

Type of bottles	Total number of samples	Compounds	Concentration (ng/L)	Reference
PET[1] and glass bottles	40	NP	<LOD[6]-30.1	(Le Coadou et al., 2017)
		OP	<LOD-2.2	
PET bottles	5	NP	0.14-0.16	(Vidal et al., 2016)
		OP	0.44-0.51	
PET bottles	4	NP	Mean: 177 ± 3	(Fabregat-Cabello et al., 2016)
		OP	Mean: 18.5 ± 1.9	
PET bottles	6	OP	1.3	(Pernica et al., 2015)
New PC[2] bottles	13	4-terBP	44-247	(Colin et al., 2014)
Glass bottles with metallic crown	85	OP	4-3160	(Guart et al., 2014)
		NP	61-430	
Glass bottles with metallic screw-cap	20	OP	3-5	
		NP	64	
PC bottles with LDPE[3] cap	20	OP	4-37	
		NP	573	
PET bottles with HDPE[4] cap	224	OP	2-11	
		NP	62-538	
PET bottles	131	OP	2- 3	(Bono-Blay et al., 2012)
		NP	58	
PET and PC bottles	6	NP	Median: 7.9	(Amiridou and Voutsa, 2011)
Plastic bottles	21	NP	108-298	(Li et al., 2010)
HDPE bottles	6	NP	Mean: 180 ± 53	(Loyo-Rosales et al., 2004)
HDPE bottles	6	OP	Mean: 12 ± 2.8	
PVC[5] bottles	12	NP	Mean 300 ± 44	

1: Polyethylene terephthalate; 2: Polycarbonate; 3: l = Low-density polyethylene; 4: High-density polyethylene; 5: Polyvinyl chloride; 6: Limit of detection.

European food safety authority (EFSA) in 2008 established TDI values of PFOA and PFOS as 1500 and 150 ng/kg-day, respectively (E.F.S.A., 2008). It should be noted that all these values are calculated using the acute exposure scenario while for drinking water long-term exposure may be more accurate (Llorca et al., 2012; Schwanz et al., 2016). In addition, the focus on a few individual seems inadequate and the total of different PFASs should also be considered (Llorca et al., 2012).

According to reported data by some researchers, the level of PFASs in bottled water is much lower than in tap water or surface water (Gellrich et al., 2013; Zhou et al., 2019b; Heo et al., 2014; Schwanz et al., 2016; Ericson et al., 2008; Flores et al., 2013). However, Kunacheva et al. (2010) reported that the levels of PFASs in bottled water from Thailand were higher than those in tap water. The higher level of contaminants in bottled water supports the high influence of bottled materials in water (Schwanz et al., 2016). Results of studies that detected PFASs are presented in Table 5. In one of the latest studies, the presence of PFASs in 38 bottled water from different countries including Ivory Coast, China, Canada, Mexico, and Burkina Faso were

Table 5
The level of perfluoroalkyl and polyfluoroalkyl substances in bottled water.

Type of bottles	Total number of samples	Compounds	Concentration (ng/L)	Reference
Not mentioned	38	PFBA	Mean: 0.20	(Kabore et al., 2018)
		PFDoDA[1]	Mean: 0.10	
		ΣPFCAs	Mean: 0.62	
		PFBS	Mean: 0.10	
		PFHxS[2]	Mean: 0.10	
		ΣPFSAs	Mean: 0.50	
		6:2FTSA[3]	Mean: 0.20	
		Σ_{29}PFAS	Mean: 1.10	
PET[12] and glass bottles	40	PFBA	<LOD[13]-1.1	(Le Coadou et al., 2017)
		PFBS	<LOD-1.4	
		PFHpA	<LOD-1.4	
		PFHxS	<LOD-2.5	
		PFOA	<LOD-9.5	
		PFOS	<LOD-3.7	
		Σ_{10}PFAS	Mean: 6.7	
PET bottles	38	PFBS	<LOD- 23.82	(Schwanz et al., 2016)
		PFHxA[4]	<LOD- 58.21	
		PFHxS	<LOD- 6.73	
		PFHpA	<LOD- 41.58	
		PFOA	<LOD- 45.62	
		PFOS	<LOD- 140.48	
		PFNA[5]	<LOD- 46.75	
		PFUnDA	<LOD- 7.71	
		PFDA	<LOD- 24.99	
		PFDS[6]	<LOD- 22.83	
		PFDoDA[7]	<LOD – 41.83	
		FOSA[8]	<LOD- 8.68	
		PFTeDA[9]	<LOD- 29.03	
		PFHxDA[10]	<LOD- 23.17	
		PFODA[11]	<LOD-25.19	
		Σ_{15}PFAS	0.95-124.29	
Not mentioned	10	Σ_{11}PFAS	< 0.6	(Zafeiraki et al., 2015)
Not mentioned	8	PFBA	Mean: 0.039	(Heo et al., 2014)
		PFPeA	Mean: 0.084	
		PFOA	Mean: 0.158	
		PFNA	Mean: 0.040	
		PFDA	Mean: 0.014	
		PFUnDA	Mean: 0.082	
		PFOS	Mean: 0.064	
		Σ_7PFAS	Mean: 0.480	
Not mentioned	119	PFBS	Median: 2.6	(Gellrich et al., 2013)
		PFHxS	Median: 1.4	
		PFOS	Median: 1.5	
		PFPeA	Median: 1.7	
		PFOA	Median: 1.6	
		PFNA	Median: 3.0	
		Σ_{10}PFAS	<1-25	
Plastic bottles	6	PFHxA	0.17-0.23	(Llorca et al., 2012)
		PFHpA	0.23-17	
		PFOS	0.04-1	
Not Mentioned	2	Σ_6PFAS	0.75-1.45	(Thompson et al., 2011)
PET bottles	20	PFPA	Mean: 0.28 ± 0.34	(Kunacheva et al., 2010)
		PFHxA	Mean: 0.34 ± 0.30	
		PFHpA	Mean: 0.25 ± 0.18	
		PFOA	Mean: 10.55 ± 9.58	
		PFOS	Mean: 0.22 ± 0.37	
PET bottles	4	PFHpA	<LOD-0.40	(Ericson et al., 2008)
		PFOA	<LOD-0.67	
		PFNA	<LOD-0.20	

1: Perfluorododecanoic acid; 2: Perfluorohexane sulfonate; 3: 6:2 fluorotelomer sulfonate; 4: Perfluorohexanoic acid; 5: Perfluorononanoic acid; 6: Perfluorodecane sulfonic acid; 7: Perfluorododecanoic acid; 8: Perfluorooctyl sulfonamide; 9: Perfluorotetradecanoic acid; 10: perfluorohexadecanoic; 11: perfluorooctadecanoic; 12: Polyethylene terephthalate; 13: limit of detection.

investigated by Kaboré et al. (2018) and perflurohexanoic acid (PFBA) was the predominant PFAS compound which was found in 45 % of the studied samples. In 2016, Schwanz et al. (2016) analyzed 16 PFASs in 38 bottled water from 3 countries (19 samples from France, 10 samples from Spain, and 9 samples from Brazil) and they found that perfluoroheptanoic acid (PFHpA) was the most abundant compound in Brazilian bottled water with 67 % frequency. Hao et al. (2014) investigated 8 bottled water from Korea. Among 7 detected compounds in Korean bottled water, the dominant ones were perfluoroundecanoic acid-PFUnDA (50 %), PFOA (37.5 %), PFOS (25 %), and perfluorodecanoic acid-PFDA (25 %). PFOA was detected at the highest mean levels (0.158 ng/L) of all the detected PFASs. The presence of PFASs in 119 bottled water samples from different European countries (Germany, Switzerland, Czech Republic, France, and Italy) were measured by Gellrich et al. (2013). According to their results, the most prevalent compounds were perfluoropentanoic acid-PFPeA (28 %) and PFOA (26 %) followed by perfluorobutane sulfonate-PFBS (16 %) and PFOS (9%). The discrepancies in different studies could potentially be due to the different contamination levels in various areas, different analytical methodology, procedsural blanks, and probable contamination during analysis (Le Coadou et al., 2017; Zhou et al., 2019b). Schwanz et al. (2016) hypothesized that ink from labels and some dopants which are used in polymer production could be also the sources of PFASs in bottled water. While Llorca et al. (2012) believed that the origin of PFASs is the plastic of bottles. Thus, the plastic of the bottles could be considered as an important source of PFASs in bottled water (Llorca et al., 2012; Schwanz et al., 2016). However, other sources of contamination such as pollution of water prior to bottling, bottle closures, and contamination of contact materials during bottling, handling, and storage are not negligible (Llorca et al., 2012; Schwanz et al., 2016). Unfortunately, the toxicological data and standards for most of PFASs are still rare, their exact effects on human health remain unknown (Gellrich et al., 2013). Thus, more research is needed to determine the accurate sources of PFASs in bottled water and their toxicological effects in order to estimate the risks of bottled water for humans.

2.6. Risk assessment

In order to investigate the potential risks of CECs in drinking water via bottled water, the consumption rate was assumed to be 2 L/day for a 60 kg adult (WHO, 2008). In addition, the worst-case scenario (the maximum level of each compound) was employed. The estimated daily intake (EDI) of CECs via bottled water consumption can be calculated by using the following equation:

$$EDI = \frac{C \times IR}{BW}$$

Where C is the concentration of target compounds (ng/L, μg/L, items/L), IR is daily consumption rate of bottled water (L/day), and BW is body weight (Luo et al., 2018a; Jeddi et al., 2016).

Table 6 represents the calculated EDI of CECs from bottled water and comparison with TDI (if applicable). Under these conditions, the CEC levels except for MPs (no threshold values) in bottled water of most countries, do not raise a safety concern for the human because their calculated EDI values were well below the TDI. However, it should be noted that other beverages drinking, food ingestion, and inhalation are other potentially significant routes of CECs exposure (Eerkes-Medrano et al., 2019; Welle and Franz, 2018; Li et al., 2010; Colin et al., 2014; Schwanz et al., 2016). Due to the known differences in health-based levels for humans, more research is needed to determine the risk of dietary exposure and toxicity of CECs for humans.

3. Conclusions and outlook

The presence of emerging contaminants in bottled water represent leakages of man-made contaminants from technical cycles into biological cycles. However, analytical improvement is needed to detect the lower level of different types of contaminants in bottled water in order to investigate the possible risks for human health. Based on the obtained results from this review, the bottle types, caps' material, and pH

Table 6
Daily intake of contaminants of emerging concern (CEC) through the bottled water consumption.

Compound	TDI[1] (μg/kg-bw/day) (reference)	Concentration (unit) (reference)	EDI[2] of CECs
MPs in plastic bottles	–	2649 (items/L) (Oßmann et al., 2018)	≈ 85 (items/kg-bw/day)
MPs in glass bottles		6292 (items/L) (Oßmann et al., 2018)	≈ 220 (items/kg-bw/day)
Salicylic acid bottled water	8.3 (Khan and Nicell, 2015)	16 (ng/L) (Lardy-Fontan et al., 2017)	0.5 (ng/kg-bw//day)
Metformin in bottled water	79.4 (Schwab et al., 2005)	12 (ng/L) (Lardy-Fontan et al., 2017)	0.4 (ng/kg-bw/day)
Gabapentin in bottled water	1.7 (Khan and Nicell, 2015)	12 (ng/L) (Lardy-Fontan et al., 2017)	0.4 (ng/kg-bw/day)
Nicotine in bottled water	8 (EFSA, 2009)	15 (ng/L) (González Alonso et al., 2012)	0.5 (ng/kg-bw/day)
Triclosan in plastic bottles	0.05 (Li et al., 2010)	9.7 (ng/L) (Li et al., 2010)	≈ 0.3 (ng/kg-bw/day)
Sulfamethoxazole in bottled water	10 (Khan and Nicell, 2015)	80 (ng/L) (Perret et al., 2006)	≈ 3 (ng/kg-bw/day)
BPA in PC[3] bottles	4 (EFSA, 2015)	8800 (ng/L) (Cao and Corriveau, 2008)	≈ 0.3 (μg/kg-bw/day)
BPA in PET[4] bottles		203 (ng/L) (Bono-Blay et al., 2012)	≈ 0.007 (μg/kg-bw/day)
OPs in plastic bottles	5 (Danish, 2016)	37 (ng/L) (Guart et al., 2014)	≈ 1.5 (ng/kg-bw/day)
OPs in glass bottles		3160 (ng/L) (Guart et al., 2014)	≈ 100 (ng/kg-bw/day)
NPs in plastic bottles	5 (Danish, 2016)	538 (ng/L) (Guart et al., 2014)	≈ 20 (ng/kg-bw/day)
NPs in glass bottles		430 (ng/L) (Guart et al., 2014)	≈ 15 (ng/kg-bw/day)
DEHP in plastic bottles	50 (EFSA, 2005c)	15.2 (μg/L) (Mousa et al., 2013)	≈ 0.5 (μg/kg-bw/day)
DEHP in glass bottles		9.87 (μg/L) (Prokůpková et al., 2002)	≈ 0.3 (μg/kg-bw/day)
DNP in plastic bottles	150 (Sakhi et al., 2014)	27 (ng/L) (Li et al., 2019)	≈ 1 (ng/kg-bw/day)
DEP in plastic bottles	500 (Sakhi et al., 2014)	20.5 (μg/L) (Guart et al., 2014)	≈ 0.7 (μg/kg-bw/day)
DEP in glass bottles		9.34 (μg/L) (Guart et al., 2014)	≈ 0.3 (μg/kg-bw/day)
DBP in plastic bottles	10 (EFSA, 2005a)	25.6 (μg/L) (Salazar-Beltran et al., 2017)	≈ 0.9 (μg/kg-bw/day)
BBP in plastic bottles	500 (EFSA, 2005b)	52.35 (μg/L) (Mousa et al., 2013)	≈ 1.7 (μg/kg-bw/day)
BBP in glass bottles		0.8 (μg/L) (Guart et al., 2014)	≈ 0.03 (μg/kg-bw/day)
PFOA in plastic bottles	1.5 (E.F.S.A., 2008)	10.55 (ng/L) (Kunacheva et al., 2010)	≈ 0.35 (ng/kg-bw/day)
PFOS in plastic bottles	0.15 (E.F.S.A., 2008)	1 (ng/L) (Llorca et al., 2012)	≈ 0.03 (ng/kg-bw/day)

1: Tolerable daily intake; 2: Estimated daily intake; 3: Polycarbonate; 4: Polyethylene terephthalate.

of water play an undeniable role in the migration of CECs into the water. Hence, the container's type (both bottles and caps) should be improved by bottling industries in order to limit the migration of CECs to their products. In addition, the knowledge about the combined effects of emerging contaminants in the human body is still lacking. Hazard and exposure assessment of MPs and other emerging contaminants in bottled water need to be improved before the full risks to human health can be assessed and understood. Therefore, the development of analytical methods and new protocols for toxicological tests and risk assessment in humans seems necessary. Considering the fact that characterization of the significance and types of CECs' threat require an understanding of their toxicokinetic and toxicodynamic models, more accurate and exhaustive research is suggested to fill gaps in the association and toxicity of individual CECs in bottled water.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgments

The authors would like to extend their gratitude to the Systems Environmental Health and Energy Research Center for their logistic support.

References

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. Environ. Res. 173, 469–479.
Acir, I.H., Guenther, K., 2018. Endocrine-disrupting metabolites of alkylphenol ethoxylates - A critical review of analytical methods, environmental occurrences, toxicity, and regulation. Sci. Total Environ. 635, 1530–1546.
Ahrens, L., Gashaw, H., Sjoholm, M., Gebrehiwot, S.G., Getahun, A., Derbe, E., Bishop, K., Akerblom, S., 2016. Poly- and perfluoroalkylated substances (PFASs) in water, sediment and fish muscle tissue from Lake Tana, ethiopia and implications for human exposure. Chemosphere 165, 352–357.
Akhbarizadeh, R., Moore, F., Keshavarzi, B., 2019. Investigating microplastics bioaccumulation and biomagnification in seafood from the Persian Gulf: a threat to human health? J. Food Addit. Contam. Part A 1–13.
Almeida, S., Raposo, A., Almeida-González, M., Carrascosa, C., 2018. Bisphenol A: food exposure and impact on human health. Compr. Rev. Food Sci. Food 17, 1503–1517.
Al-Saleh, I., Shinwari, N., Alsabbaheen, A.J.T.J., 2011. Phthalates Residues in Plastic Bottled Waters 36. pp. 469–478.
Amiridou, D., Voutsa, D., 2011. Alkylphenols and phthalates in bottled waters. J. Hazard. Mater. 185, 281–286.
Arthur, C., Baker, J., Bamford, H., 2009. Proceedings of the International Research Workshop on the Occurrence, Effects, and Fate of Microplastic Marine Debris, September 9-11, 2008.
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 Res. 46, 571–583.
Bach, C., Dauchy, X., Severin, I., Munoz, J.-F., Etienne, S., Chagnon, M., 2013. Effect of temperature on the release of intentionally and non-intentionally added substances from polyethylene terephthalate (PET) bottles into water: chemical analysis and potential toxicity. Food Chem. 139, 672–680.
Ballesteros, V., Costa, O., Iniguez, C., Fletcher, T., Ballester, F., Lopez-Espinosa, M., 2017. Exposure to perfluoroalkyl substances and thyroid function in pregnant women and children: a systematic review of epidemiologic studies. Environ. Int. 99, 15–28.
Barboza, L.G.A., Gimenez, B.C.G., 2015. Microplastics in the marine environment: current trends and future perspectives. Mar. Pollut. Bull. 97, 5–12.
Bergkamp, L., 2013. The European Union REACH Regulation for Chemicals: Law and Practice. OUP Oxford.
Biles, J., McNeal, T., Begley, T., Hollifield, H.C., 1997. Determination of bisphenol-A in reusable polycarbonate food-contact plastics and migration to food-simulating liquids. J. Agric. Food Chem. 45, 3541–3544.
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. Environ. Sci. Pollut. Res. Int. 19, 3339–3349.
Bošnir, J., Puntarić, D., Galić, A., Škes, I., Dijanić, T., Klarić, M., Grgić, M., Čurković, M., Šmit, Z., 2007. Migration of phthalates from plastic containers into soft drinks and mineral water. Food Technol. Biotech. 45, 91–95.
Cao, X.L., 2008. Determination of phthalates and adipate in bottled water by headspace solid-phase microextraction and gas chromatography/mass spectrometry. J. Chromatogr. A 1178, 231–238.
Cao, X.L., Corriveau, J., 2008. Survey of bisphenol A in bottled water products in Canada. Food Addit. Contam. Part B Surveill. 1, 161–164.
Chailurkit, L., Srijaruskul, K., Ongphiphadhanakul, B., 2017. Bisphenol a in canned carbonated drinks and plastic-bottled water from supermarkets. Expo. Health 9, 243–248.
Chiu, T., Paterakis, N., Cartmell, E., Scrimshaw, M., Lester, J.N., 2010. A critical review of the formation of mono-and dicarboxylated metabolic intermediates of alkylphenol polyethoxylates during wastewater treatment and their environmental significance. Crit. Rev. Environ. Sci. Technol. 40, 199–238.
Cleuvers, M., 2003. Aquatic ecotoxicity of pharmaceuticals including the assessment of combination effects. Toxicol. Lett. 142, 185–194.
Colin, A., Bach, C., Rosin, C., Munoz, J.F., Dauchy, X., 2014. Is drinking water a major route of human exposure to alkylphenol and bisphenol contaminants in France? Arch. Environ. Contam. Toxicol. 66, 86–99.
Corsolini, S., Sara, G., 2017. The trophic transfer of persistent pollutants (HCB, DDTs, PCBs) within polar marine food webs. Chemosphere 177, 189–199.
Danish, E., 2016. Toxicological Evaluation and Limit Values for Nonylphenol, Nonylphenol Ethoxylates, Tricresyl, Phosphates and Benzoic Acid.
Dauchy, X., 2019. Per- and polyfluoroalkyl substances (PFASs) in drinking water: Current state of the science. Curr. Opin. Environ. Sci. Health 7, 8–12.
Dehghani, S., Moore, F., Akhbarizadeh, R., 2017. Microplastic pollution in deposited urban dust, Tehran metropolis, Iran. Environ. Sci. Pollut. Res. 24, 20360–20371.
Deng, Y., Zhang, Y., Lemos, B., Ren, H., 2017. Tissue accumulation of microplastics in mice and biomarker responses suggest widespread health risks of exposure. Sci. Rep. 7, 46687.
Devier, M.H., Le Menach, K., Viglino, L., Di Gioia, L., Lachassagne, P., Budzinski, H., 2013. Ultra-trace analysis of hormones, pharmaceutical substances, alkylphenols and phthalates in two French natural mineral waters. Sci. Total Environ. 443, 621–632.
Dhaini, H.R., Nassif, R.M., 2014. Exposure assessment of endocrine disruptors in bottled drinking water of Lebanon. Environ. Monit. Assess. 186, 5655–5662.
Ding, J., Zhang, S., Razanajatovo, R.M., Zou, H., Zhu, W., 2018. Accumulation, tissue distribution, and biochemical effects of polystyrene microplastics in the freshwater fish red tilapia (Oreochromis niloticus). Environ. Pollut. 238, 1–9.
Dobaradaran, S., Schmidt, T.C., Nabipour, I., Khajeahmadi, N., Tajbakhsh, S., Saeedi, R., Mohammadi, M.J., Keshtkar, M., Khorsand, M., Ghasemi, F., 2018. Characterization of plastic debris and association of metals with microplastics in coastline sediment along the Persian Gulf. Waste Manage. 78, 649–658.
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 Chem. 274, 246–253.
Dris, R., Gasperi, J., Saad, M., Mirande, C., Tassin, B., 2016. Synthetic fibers in atmospheric fallout: a source of microplastics in the environment? Mar. Pollut. Bull. 104, 290–293.
E.C. EC, 2015. Commission Implementing Decision (EU) 2015/495 of 20 March 2015. Establishing a Watch List of Substances for Union-wide Monitoring in the Field of Water Policy Pursuant to Directive 2008/105/EC of the European Parliament and of the Council.
E.F.S.A, 2008. EFSA, Perfluorooctane Sulfonate (PFOS), Perfluorooctanoic Acid (PFOA) and Their Salts Scientific Opinion of the Panel on Contaminants in the Food Chain 6. pp. 653.
Ebele, A.J., Abdallah, M.A.-E., Harrad, S., 2017. Pharmaceuticals and personal care products (PPCPs) in the freshwater aquatic environment. Emerging Contam. 3, 1–16.
Eerkes-Medrano, D., Leslie, H.A., Quinn, B., 2019. Microplastics in drinking water: a review and assessment. Curr. Opin. Environ. Sci. Health 7, 69–75.
EFSA, E., 2015. Panel on Food Contact Materials, Flavourings, P.A.J.E. Journal, Scientific Opinion on the Risks to Public Health Related to the Presence of Bisphenol a (BPA) in Foodstuffs 13. pp. 3978.
EFSA, E.F.S.A., 2009. Potential Risks for Public Health Due to the Presence of Nicotine in Wild Mushrooms 7. pp. 286r.
EFSA, E.F.S.A., 2005a. Opinion of the Scientific Panel on Food Additives, Flavourings, Processing Aids and Materials in Contact With Food (AFC) Related to Di-butylphthalate (DBP) for Use in Food Contact Materials 3. pp. 242.
EFSA, E.F.S.A., 2005b. Opinion of the Scientific Panel on Food Additives, Flavourings, Processing Aids and Materials in Contact With Food (AFC) Related to Butylbenzylphthalate (BBP) for Use in Food Contact Materials 3. pp. 241.
EFSA, 2005c. Opinion of the Scientific Panel on Food Additives, Flavourings, Processing Aids and Materials in Contact With Food (AFC) Related to Bis (2-ethylhexyl) Phthalate (DEHP) for Use in Food Contact Materials.
Ehrlich, S., Calafat, A.M., Humblet, O., Smith, T., Hauser, R., 2014. Handling of thermal receipts as a source of exposure to bisphenol A. JAMA 311, 859–860.
Elobeid, M.A., Almarhoon, Z.M., Virk, P., Hassan, Z.K., Omer, S.A., ElAmin, M., Daghestani, M.H., AlOlayan, E.M., 2012. Bisphenol a detection in various brands of drinking bottled water in Riyadh, Saudi Arabia using gas chromatography/mass spectrometer. Trop. J. Pharm. Res. 11, 455–459.
Ericson, I., Nadal, M., van Bavel, B., Lindström, G., Domingo, J.L., 2008. Levels of perfluorochemicals in water samples from Catalonia, Spain: is drinking water a significant contribution to human exposure? Environ. Sci. Pollut. Res. Int. 15, 614–619.
Eriksson, U., Karrman, A., Rotander, A., Mikkelsen, B., Dam, M., 2013. Perfluoroalkyl substances (PFASs) in food and water from Faroe Islands. Environ. Sci. Pollut. Res. Int. 20, 7940–7948.
Fabregat-Cabello, N., Pitarch-Motellón, J., Sancho, J.V., Ibáñez, M., Roig-Navarro, A.F., 2016. Method development and validation for the determination of selected

endocrine disrupting compounds by liquid chromatography mass spectrometry and isotope pattern deconvolution in water samples, Comparison of Two Extraction Techniques. Anal. Methods 8, 2895–2903.
Fan, Y.Y., Zheng, J.L., Ren, J.H., Luo, J., Cui, X.Y., Ma, L.Q., 2014. Effects of storage temperature and duration on release of antimony and bisphenol A from polyethylene terephthalate drinking water bottles of China. Environ. Pollut. 192, 113–120.
Flores, C., Ventura, F., Martin-Alonso, J., Caixach, J., 2013. Occurrence of perfluorooctane sulfonate (PFOS) and perfluorooctanoate (PFOA) in NE spanish surface waters and their removal in a drinking water treatment plant that combines conventional and advanced treatments in parallel lines. Sci. Total Environ. 461, 618–626.
Gellrich, V., Brunn, H., Stahl, T., 2013. Perfluoroalkyl and polyfluoroalkyl substances (PFASs) in mineral water and tap water. J. Environ. Sci. Health A. Tox. Subst. Environ. Eng. 48, 129–135.
Gogoi, A., Mazumder, P., Tyagi, V.K., Chaminda, G.T., An, A.K., Kumar, M., 2018. Occurrence and fate of emerging contaminants in water environment: a review. Groundwater Sustain. Develop. 6, 169–180.
González Alonso, S., Valcárcel, Y., Montero, J., Catalá, M., 2012. Nicotine occurrence in bottled mineral water: analysis of 10 brands of water in Spain. Sci. Total Environ. 416, 527–531.
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 Chem. 156, 73–80.
Guart, A., Bono-Blay, F., Borrell, A., Lacorte, S., 2011. Migration of plasticizers phthalates, bisphenol A and alkylphenols from plastic containers and evaluation of risk. Food Addit. Contam. Part A Chem. Anal. Control Expo. Risk Assess. 28, 676–685.
Hantoro, I., Löhr, A.J., Van Belleghem, F.G., Widianarko, B., Ragas, A.M.J., 2019. Microplastics in coastal areas and seafood: implications for food safety. Food Addit. Contam, Part A 36, 674–711.
He, D., Luo, Y., Lu, S., Liu, M., Song, Y., Lei, L., 2018. Microplastics in soils: analytical methods, pollution characteristics and ecological risks. Trac-Trend. Anal. Chem.
Heo, J.-J., Lee, J.-W., Kim, S.-K., Oh, J.-E., 2014. Foodstuff analyses show that seafood and water are major perfluoroalkyl acids (PFAAs) sources to humans in Korea. J. Hazard. Mater. 279, 402–409.
Hernandez-Hernandez, K.L., Tapia-Orozco, N., Gimeno, M., Espinosa-Garcia, A.M., Garcia-Garcia, J.A., Araiza-Olivera, D., Sanchez-Bartez, F., Gracia-Mora, I., Gutierrez-Aguilar, M., Garcia-Arrazola, R., 2019. Exposure to bisphenol A: current levels from food intake are toxic to human cells. Mol. Biol. Rep. 46, 2555–2559.
Huerta Lwanga, E., Mendoza Vega, J., Ku Quej, V., Chi, J.L.A., Sanchez Del Cid, L., Chi, C., Escalona Segura, G., Gertsen, H., Salanki, T., van der Ploeg, M., Koelmans, A.A., Geissen, V., 2017. Field evidence for transfer of plastic debris along a terrestrial food chain. Sci. Rep. 7, 14071.
Ingelido, A.M., Abballe, A., Gemma, S., Dellatte, E., Iacovella, N., De Angelis, G., Zampaglioni, F., Marra, V., Miniero, R., Valentini, S., Russo, F., Vazzoler, M., Testai, E., De Felip, E., 2018. Biomonitoring of perfluorinated compounds in adults exposed to contaminated drinking water in the Veneto Region, Italy. Environ. Int. 110, 149–159.
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. Environ. Monit. Assess. 188, 534.
Jeddi, M.Z., Rastkari, N., Ahmadkhaniha, R., Yunesian, M., Nabizadeh, R., Daryabeygi, R., 2015a. A margin of exposure approach to assessment of non-cancerous risk of diethyl phthalate based on human exposure from bottled water consumption. Environ. Sci. Pollut. Res. Int. 22, 19518–19528.
Jeddi, M.Z., Rastkari, N., Ahmadkhaniha, R., Yunesian, M., 2015b. Concentrations of phthalates in bottled water under common storage conditions: Do they pose a health risk to children? Food Res. Int. 69, 256–265.
Kabore, H.A., Vo Duy, S., Munoz, G., Meite, L., Desrosiers, M., Liu, J., Sory, T.K., Sauve, S., 2018. Worldwide drinking water occurrence and levels of newly-identified perfluoroalkyl and polyfluoroalkyl substances. Sci. Total Environ. 616–617, 1089–1100.
Kafilzadeh, F., 2015. Assessment of organochlorine pesticide residues in water, sediments and fish from Lake Tashk, Iran. Achiev. Life Sci. 9, 107–111.
Kanchanamayoon, W., Prapatpong, P., Chumwangwapee, S., Chaithongrat, S., 2012. Analysis of phthalate esters contamination in drinking water samples. Afr. J. Biotechnol. 11, 16263–16269.
Keresztes, S., Tatar, E., Czegeny, Z., Zaray, G., Mihucz, V.G., 2013. Study on the leaching of phthalates from polyethylene terephthalate bottles into mineral water. Sci. Total Environ. 458–460, 451–458.
Khan, U., Nicell, J., 2015. Human health relevance of pharmaceutically active compounds in drinking water. AAPS J. 17, 558–585.
Koelmans, A.A., Mohamed Nor, N.H., Hermsen, E., Kooi, M., Mintenig, S.M., De France, J., 2019. Microplastics in freshwaters and drinking water: critical review and assessment of data quality. Water Res. 155, 410–422.
Kolandhasamy, P., Su, L., Li, J., Qu, X., Jabeen, K., Shi, H., 2018. Adherence of microplastics to soft tissue of mussels: a novel way to uptake microplastics beyond ingestion. Sci. Total Environ. 610, 635–640.
Kosuth, M., Mason, S.A., Wattenberg, E.V., 2018. Anthropogenic contamination of tap water, beer, and sea salt. PLoS One 13, e0194970.
Kunacheva, C., Fujii, S., Tanaka, S., Boontanon, S.K., Poothong, S., Wongwatthana, T., Shivakoti, B.R., 2010. Perfluorinated compounds contamination in tap water and bottled water in Bangkok, Thailand. J WATER SUPPLY RES T —AQUA 59, 345–354.
Lardy-Fontan, S., Le Diouron, V., Drouin, C., Lalere, B., Vaslin-Reimann, S., Dauchy, X., Rosin, C., 2017. Validation of a method to monitor the occurrence of 20 relevant pharmaceuticals and personal care products in 167 bottled waters. Sci. Total Environ. 587, 118–127.
Le, H.H., Carlson, E.M., Chua, J.P., Belcher, S.M., 2008. Bisphenol a is released from polycarbonate drinking bottles and mimics the neurotoxic actions of estrogen in developing cerebellar neurons. Toxicol. Lett. 176, 149–156.
Le Coadou, L., Le Menach, K., Labadie, P., Devier, M.H., Pardon, P., Augagneur, S., Budzinski, H., 2017. Quality survey of natural mineral water and spring water sold in France: monitoring of hormones, pharmaceuticals, pesticides, perfluoroalkyl substances, phthalates, and alkylphenols at the ultra-trace level. Sci. Total Environ. 603–604, 651–662.
Leivadara, S.V., Nikolaou, A.D., Lekkas, T.D., 2008. Determination of organic compounds in bottled waters. Food Chem. 108, 277–286.
Li, X., Ying, G.G., Su, H.C., Yang, X.B., Wang, L., 2010. Simultaneous determination and assessment of 4-nonylphenol, bisphenol A and triclosan in tap water, bottled water and baby bottles. Environ. Int. 36, 557–562.
Li, H., Li, C., An, L., Deng, C., Su, H., Wang, L., Jiang, Z., Zhou, J., Wang, J., Zhang, C., Jin, F., 2019. Phthalate esters in bottled drinking water and their human exposure in Beijing, China. Food Addit. Contam. Part B Surveill. 12, 1–9.
Liebezeit, G., Liebezeit, E., 2014. Synthetic particles as contaminants in German beers. Food Addit. Contam. Part A Chem. Anal. Control Expo. Risk Assess. 31, 1574–1578.
Llorca, M., Farré, M., Picó, Y., Müller, J., Knepper, T.P., Barceló, D., 2012. Analysis of perfluoroalkyl substances in waters from Germany and Spain. Sci. Total Environ. 431, 139–150.
Loyo-Rosales, J.E., Rosales-Rivera, G.C., Lynch, A.M., Rice, C.P., Torrents, A., 2004. Chemistry, migration of nonylphenol from plastic containers to water and a milk surrogate. J. Agr. Food Chem. 52, 2016–2020.
Luo, Q., Liu, Z.H., Yin, H., Dang, Z., Wu, P.X., Zhu, N.W., Lin, Z., Liu, Y., 2018a. Migration and potential risk of trace phthalates in bottled water: a global situation. Water Res. 147, 362–372.
Luo, X., Zheng, H., Zhang, Z., Wang, M., Yang, B., Huang, L., Wang, M., 2018b. Cloud point extraction for simultaneous determination of 12 phenolic compounds by high performance liquid chromatography with fluorescence detection. Microchem. J. 137, 148–154.
Maggioni, S., Balaguer, P., Chiozzotto, C., Benfenati, E., 2013. Screening of endocrine-disrupting phenols, herbicides, steroid estrogens, and estrogenicity in drinking water from the waterworks of 35 Italian cities and from PET-bottled mineral water. Environ. Sci. Pollut. Res. Int. 20, 1649–1660.
Martine, B., Cendrine, D., Fabrice, A., Marc, C., 2013. Assessment of adult human exposure to phthalate esters in the urban centre of Paris (France). Bull. Environ. Contam. Toxicol 90, 91–96.
Mason, S.A., Welch, V.G., Neratko, J., 2018. Synthetic polymer contamination in bottled water. Front Chem. 6.
Mintenig, S.M., Loder, M.G.J., Primpke, S., Gerdts, G., 2019. Low numbers of microplastics detected in drinking water from ground water sources. Sci. Total Environ. 648, 631–635.
Montuori, P., Jover, E., Morgantini, M., Bayona, J.M., Triassi, M., 2008. Assessing human exposure to phthalic acid and phthalate esters from mineral water stored in polyethylene terephthalate and glass bottles. Food Addit. Contam. Part A Chem. Anal. Control Expo. Risk Assess. 25, 511–518.
Moreman, J., Lee, O., Trznadel, M., David, A., Kudoh, T., Tyler, C.R., 2017. Acute toxicity, teratogenic, and estrogenic effects of bisphenol A and its alternative replacements bisphenol S, bisphenol F, and bisphenol AF in zebrafish embryo-larvae. Environ. Sci. Technol. 51, 12796–12805.
Mousa, A., Basheer, C., Rahman Al-Arfaj, A., 2013. Determination of phthalate esters in bottled water using dispersive liquid–liquid microextraction coupled with GC–MS. J. Sep. Sci. 36, 2003–2009.
Nelms, S.E., Galloway, T.S., Godley, B.J., Jarvis, D.S., Lindeque, P.K., 2018. Investigating microplastic trophic transfer in marine top predators. Environ. Pollut. 238, 999–1007.
Oßmann, B.E., Sarau, G., Holtmannspotter, H., Pischetsrieder, M., Christiansen, S.H., Dicke, W., 2018. Small-sized microplastics and pigmented particles in bottled mineral water. Water Res. 141, 307–316.
Otero, P., Saha, S.K., Moane, S., Barron, J., Clancy, G., Murray, P., 2015. Improved method for rapid detection of phthalates in bottled water by gas chromatography-mass spectrometry. J. Chromatogr. B Analyt. Technol. Biomed. Life Sci. 997, 229–235.
Papadopoulou, E., Poothong, S., Koekkoek, J., Lucattini, L., Padilla-Sánchez, J.A., Haugen, M., Herzke, D., Valdersnes, S., Maage, A., Cousins, I.T., et al., 2017. Estimating human exposure to perfluoroalkyl acids via solid food and drinks: implementation and comparison of different dietary assessment methods. Environ. Res. 158, 269–276.
Pernica, M., Poloucka, P., Seifertova, M., Simek, Z., 2015. Determination of alkylphenols in water samples using liquid chromatography-tandem mass spectrometry after pre-column derivatization with dansyl chloride. J. Chromatogr. A 1417, 49–56.
Perret, D., Gentili, A., Marchese, S., Greco, A., Curini, R., 2006. Sulphonamide residues in italian surface and drinking waters: a small scale reconnaissance. Chromatographia 63, 225–232.
Petersen, A., 2016. Statement on the Presence of Microplastics and Nanoplastics in Food, With Particular Focus on Seafood.
Peterson, D.R., Staples, C.A., 2003. Degradation of phthalate esters in the environment. Series Anthropogenic Compounds. Springer, pp. 85–124.
Pourzamani, H., Falahati, M., Rastegari, F., Ebrahim, K., 2016. Freeze–melting process significantly decreases phthalate ester plasticizer levels in drinking water stored in polyethylene terephthalate (PET) bottles. Water Supply 17, 745–751.
Prapatpong, P., Kanchanamayoon, W., 2010. Determination of phthalate esters in drinking water using solid-phase extraction and gas chromatography. Appl. Sci. 10, 1987–1990.
Prokůpková, G., Holadová, K., Poustka, J., Hajšlová, J., 2002. Development of a solid-phase microextraction method for the determination of phthalic acid esters in water. Anal. Chim. Acta 457, 211–223.

Rastkari, N., Zare Jeddi, M., Yunesian, M., Ahmadkhaniha, R., 2017. The effect of storage time, temperature and type of packaging on the release of phthalate esters into packed acidic liquids. Food Technol. Biotechnol. 55, 562–569.
Regueiro, J., Llompart, M., Garcia-Jares, C., Garcia-Monteagudo, J.C., Cela, R., 2008. Ultrasound-assisted emulsification–microextraction of emergent contaminants and pesticides in environmental waters. J. Chromatogr. A 1190, 27–38.
Ribeiro, F., O'Brien, J.W., Galloway, T., Thomas, K.V., 2018. Accumulation and fate of nano-and micro-plastics and associated contaminants in organisms. Trac-trend Anal. Chem.
Rocha-Santos, T., Duarte, A.C., 2015. A critical overview of the analytical approaches to the occurrence, the fate and the behavior of microplastics in the environment. Trac-trend Anal. Chem. 65, 47–53.
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 Chem. 202, 88–93.
Russo, G., Barbato, F., Mita, D.G., Grumetto, L., 2019. Occurrence of Bisphenol A and its analogues in some foodstuff marketed in Europe. Food Chem. Toxicol. 131, 110575.
Sakhi, A.K., Lillegaard, I.T.L., Voorspoels, S., Carlsen, M.H., Løken, E.B., Brantsæter, A.L., Haugen, M., Meltzer, H.M., Thomsen, C., 2014. Concentrations of phthalates and bisphenol a in norwegian foods and beverages and estimated dietary exposure in adults. Environ. Int. 73, 259–269.
Salazar-Beltrán, D., Hinojosa-Reyes, L., Palomino-Cabello, C., Turnes-Palomino, G., Hernández-Ramírez, A., Guzmán-Mar, J.L., 2018. Determination of phthalate acid esters plasticizers in polyethylene terephthalate bottles and its correlation with some physicochemical properties. Polym. Test. 68, 87–94.
Salazar-Beltran, D., Hinojosa-Reyes, L., Ruiz-Ruiz, E., Hernandez-Ramirez, A., Luis Guzman-Mar, J., 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.
Salgueiro-Gonzalez, N., Muniategui-Lorenzo, S., Lopez-Mahia, P., Prada-Rodriguez, D., 2017. Trends in analytical methodologies for the determination of alkylphenols and bisphenol A in water samples. Anal. Chim. Acta 962, 1–14.
Santana, J., Giraudi, C., Marengo, E., Robotti, E., Pires, S., Nunes, I., Gaspar, E.M., 2014. Preliminary toxicological assessment of phthalate esters from drinking water consumed in Portugal. Environ. Sci. Pollut. Res. 21, 1380–1390.
Santhi, V., Sakai, N., Ahmad, E., Mustafa, A., 2012. Occurrence of bisphenol a in surface water, drinking water and plasma from malaysia with exposure assessment from consumption of drinking water. Sci. Total Environ. 427, 332–338.
Schwab, B.W., Hayes, E.P., Fiori, J.M., Mastrocco, F.J., Roden, N.M., Cragin, D., Meyerhoff, R.D., Vincent, J., Anderson, P.D., 2005. Human pharmaceuticals in us surface waters: a human health risk assessment. Regul. Toxicol. Pharm. 42, 296–312.
Schwanz, T.G., Llorca, M., Farré, M., Sotte, D., 2016. Perfluoroalkyl substances assessment in drinking waters from Brazil, France and Spain. Sci. Total Environ. 539, 143–152.
Schymanski, D., Goldbeck, C., Humpf, H.U., Furst, P., 2018. Analysis of microplastics in water by micro-Raman spectroscopy: release of plastic particles from different packaging into mineral water. Water Res. 129, 154–162.
Selvaraj, K.K., Mubarakali, H., Rathinam, M., Harikumar, L., Sampath, S., Shanmugam, G., Ramaswamy, B.R., 2016. Cumulative exposure and dietary risk assessment of phthalates in bottled water and bovine milk samples: a preliminary case study in Tamil Nadu, India. Hum. Ecol. Risk Assess. 22, 1166–1182.
Serôdio, P., Nogueira, J., 2006. Considerations on ultra-trace analysis of phthalates in drinking water. Water Res. 40, 2572–2582.
Sharma, B.M., Bečanová, J., Scheringer, M., Sharma, A., Bharat, G.K., Whitehead, P.G., Klánová, J., Nizzetto, L., 2019. Health and ecological risk assessment of emerging contaminants (pharmaceuticals, personal care products, and artificial sweeteners) in surface and groundwater (drinking water) in the Ganges River Basin, India. Sci. Total Environ. 646, 1459–1467.
Smith, M., Love, D.C., Rochman, C.M., Neff, R.A., 2018. Microplastics in seafood and the implications for human health. Curr. Environ. Health Rep. 5, 375–386.
Starling, M.C.V., Amorim, C.C., Leão, M.M.D., 2019. Occurrence, control and fate of contaminants of emerging concern in environmental compartments in Brazil. J. Hazard. Mater. 372, 17–36.
Szendi, K., Gyöngyi, Z., Kontár, Z., Gerencsér, G., Berényi, K., Hanzel, A., Fekete, J., Kovács, A., Varga, C., 2017. Mutagenicity and Phthalate Level of Bottled Water Under Different Storage Conditions, Exposure and Health 10. pp. 51–60.
Thompson, J., Eaglesham, G., Mueller, J., 2011. Concentrations of PFOS, PFOA and other perfluorinated alkyl acids in australian drinking water. Chemosphere 83, 1320–1325.
Thorpe, K.L., Hutchinson, T.H., Hetheridge, M.J., Scholze, M., Sumpter, J.P., Tyler, C.R., 2001. Assessing the biological potency of binary mixtures of environmental estrogens using vitellogenin induction in juvenile rainbow trout (*Oncorhynchus Mykiss*). Environ. Sci. Technol. 35, 2476–2481.
Valcarcel, Y., Valdehita, A., Becerra, E., Lopez de Alda, M., Gil, A., Gorga, M., Petrovic, M., Barcelo, D., Navas, J.M., 2018. Determining the presence of chemicals with suspected endocrine activity in drinking water from the Madrid region (Spain) and assessment of their estrogenic, androgenic and thyroidal activities. Chemosphere 201, 388–398.
Van Cauwenberghe, L., Janssen, C.R., 2014. Microplastics in bivalves cultured for human consumption. Environ. Pollut. 193, 65–70.
Vidal, R.B.P., Ibanez, G.A., Escandar, G.M., 2016. A green method for the quantification of plastics-derived endocrine disruptors in beverages by chemometrics-assisted liquid chromatography with simultaneous diode array and fluorescent detection. Talanta 159, 336–343.
Vilarinho, F., Sendón, R., van der Kellen, A., Vaz, M.F., Silva, A.S., 2019. Bisphenol A in food as a result of its migration from food packaging. Trends Food Sci. Technol.
Wagner, M., Oehlmann, J., 2009. Endocrine disruptors in bottled mineral water: total estrogenic burden and migration from plastic bottles. Environ. Sci. Pollut. Res. Int. 16, 278–286.
Welle, F., Franz, R., 2018. Microplastic in bottled natural mineral water - literature review and considerations on exposure and risk assessment. Food Addit. Contam. Part A Chem. Anal. Control Expo. Risk Assess. 35, 2482–2492.
Welshons, W.V., Nagel, S.C., vom Saal, F.S., 2006. Large effects from small exposures. iii. endocrine mechanisms mediating effects of bisphenol a at levels of human exposure. Endocrinology 147, s56–s69.
WHO, 2008. Guidelines for Drinking-water Quality, 3rd ed. Geneva.
Wiesheu, A.C., Anger, P.M., Baumann, T., Niessner, R., Ivleva, N.P., 2016. Raman Microspectroscopic Analysis of Fibers in Beverages. Anal. Methods 8, 5722–5725.
Wilkinson, J., Hooda, P.S., Barker, J., Barton, S., Swinden, J., 2017. Occurrence, fate and transformation of emerging contaminants in water: an overarching review of the field. Environ. Pollut. 231, 954–970.
Winkler, A., Santo, N., Ortenzi, M.A., Bolzoni, E., Bacchetta, R., Tremolada, P., 2019. Does mechanical stress cause microplastic release from plastic water bottles? Water Res. 166, 115082.
Xu, M., Huang, H., Li, N., Li, F., Wang, D., Luo, Q., 2019. Occurrence and ecological risk of pharmaceuticals and personal care products (PPCPs) and pesticides in typical surface watersheds, china. Ecotoxicol. Environ. Saf. 175, 289–298.
Xu, J., Liang, P., Zhang, T., 2007. Dynamic liquid-phase microextraction of three phthalate esters from water samples and determination by gas chromatography. Anal. Chim. Acta 597, 1–5.
Yang, Y., Ok, Y.S., Kim, K.-H., Kwon, E.E., Tsang, Y.F., 2017. Occurrences and removal of pharmaceuticals and personal care products (PPCPs) in drinking water and water/sewage treatment plants: a review. Sci. Total Environ. 596, 303–320.
Zaater, M.F., Tahboub, Y.R., Sayyed, A.N., 2013. Determination of phthalates in jordanian bottled water using GC–MS and HPLC–UV: environmental study. J. Chromatogr. Sci. 52, 447–452.
Zafeiraki, E., Costopoulou, D., Vassiliadou, I., Leondiadis, L., Dassenakis, E., Traag, W., Hoogenboom, R.L., van Leeuwen, S.P., 2015. Determination of perfluoroalkylated substances (PFASs) in drinking water from the Netherlands and Greece. Food Addit. Contam. Part A Chem. Anal. Control Expo. Risk Assess. 32, 2048–2057.
Zaki, G., Shoeib, T., 2018. Concentrations of several phthalates contaminants in Egyptian bottled water: effects of storage conditions and estimate of human exposure. Sci. Total Environ. 618, 142–150.
Zhou, Q., Lei, M., Li, J., Zhao, K., Liu, Y., 2017. Sensitive determination of bisphenol A, 4-nonylphenol and 4-octylphenol by magnetic solid phase extraction with Fe@MgAl-LDH magnetic nanoparticles from environmental water samples. Sep. Purif. Technol. 182, 78–86.
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., 2019a. 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 Chem. 294, 160–170.
Zhou, W., Zhao, S., Tong, C., Chen, L., Yu, X., Yuan, T., Aimuzi, R., Luo, F., Tian, Y., Zhang, J., 2019b. Shanghai Birth Cohort, Dietary intake, drinking water ingestion and plasma perfluoroalkyl substances concentration in reproductive aged Chinese women. Environ. Int. 127, 487–494.
Zuccarello, P., Ferrante, M., Cristaldi, A., Copat, C., Grasso, A., Sangregorio, D., Fiore, M., Oliveri Conti, G., 2019. Exposure to microplastics (<10mum) associated to plastic bottles mineral water consumption: the first quantitative study. Water Res. 157, 365–371.