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REVIEW ARTICLE

Emerging contaminants migration from pipes used in drinking water distribution systems: a review of the scientific literature

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Abstract

Migration of emerging contaminants (ECs) from pipes into water is a global concern due to potential human health effects. Nevertheless, a review of migration ECs from pipes into water distribution systems is presently lacking. This paper reviews, the reported occurrence migration of ECs from pipes into water distribution systems in the world. Furthermore, the results related to ECs migration from pipes into water distribution systems, their probable sources, and their hazards are discussed. The present manuscript considered the existing reports on migration of five main categories of ECs including microplastics (MPs), bisphenol A (BPA), phthalates, nonylphenol (NP), perfluoroalkyl, and polyfluoroalkyl substances (PFAS) from distribution network into tap water. A focus on tap water in published literature suggests that pipes type used had an important role on levels of ECs migration in water during transport and storage of water. For comparison, tap drinking water in contact with polymer pipes had the highest mean concentrations of reviewed contaminants. Polyvinyl chloride (PVC), polyamide (PA), polypropylene (PP), polyethylene (PE), and polyethylene terephthalate (PET) were the most frequently detected types of microplastics (MPs) in tap water. Based on the risk assessment analysis of ECs, levels of perfluorooctanoic acid (PFOA), perfluorononanoic acid (PFNA), perfluorodecanoic acid (PFDA), perfluorohexane sulfonate (PFHxS), and perfluorooctane sulfonate (PFOS) were above 1, indicating a potential non-carcinogenic health risk to consumers. Finally, there are still scientific gaps on occurrence and migration of ECs from pipes used in distribution systems, and this needs more in-depth studies to evaluate their exposure hazards on human health.

Keywords Emerging contaminants · Human health · Risk assessment · Tap water · Water supply network

Introduction

Emerging contaminants (ECs) are natural or synthetic chemicals that have the potential to enter the environment and cause adverse ecological or/and human health effects

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(Ahmed et al. 2017; Geissen et al. 2015; Ouda et al. 2021). ECs are not commonly monitored in the environment because of their emerging nature (Geissen et al. 2015; Ouda et al. 2021). If left unregulated, contaminants represent a main concern. The major threats from ECs are related to environmental and human toxicological effects that have not yet been properly studied (Sharma et al. 2019). ECs include a wide variety of compounds such as pharmaceuticals, endocrine-disrupting compounds (EDCs), personal care products (PCPs), flame retardants, pesticides, surfactants, and industrial additives among others (Ahmed et al. 2017; Jiang et al. 2013; Matamoros et al. 2016). ECs can cause different risks to humans and to the environment. A risk assessment of ECs is mostly based on the persistence, toxicity, and bioaccumulation (Haddaoui and Mateo-Sagasta 2021). Although the occurrence of ECs has been reported in different environmental media, as yet there are not enough reports on their potential environmental or human health risks (Naidu et al. 2016). Exposure to ECs may cause many different types of effects in humans, such as mutagenic and carcinogenic effects (Ouda et al. 2021; Yadav et al. 2019).

Tap water is potentially transferred over large distances through the distribution system to reach the consumer. A worsening in the quality of tap water cannot be ruled out after the water leaves a treatment plant (Douterelo et al. 2014; Machell et al. 2010; Ramos et al. 2010).). Well-managed distribution systems are an important factor in ensuring the integrity tap water and in protecting it from contamination. Nevertheless, the management of water supply networks often receives too little attention. There is widespread evidence that the insufficient management of tap water has led to outbreaks of disease. The reasons for these outbreaks and the level of chemical risks involved are various (Brunkard et al. 2011; WHO 2014). There are many chemical risks that could pollute drinking water, such as compounds coming from substances or reacting with substances in the water networks, chemicals which have accumulated and migrated from deposits and scales, and compounds entering the water networks through defects and fractures (WHO 2014). Chlorine as the most common chemical added to drinking water for water disinfection and the control of targeted pathogens. Chlorination of drinking water generates potentially carcinogenic disinfection by-products such as haloacetic acids and trihalomethanes (He et al. 2017; Richardson and Kimura 2016). Most developed countries have created regulations or guidelines to minimize human exposure to disinfection by-products (Richardson 2003). One serious problem facing drinking water distribution networks is the migration of contaminants from pipes, and this concern has important consequences for substance choice, the operation of a system, and regulatory compliance. Various organic and inorganic additives such as lubricants, antioxidants and other stabilizers, softeners, and coloring agents are used in pipes to increase the life of the material, and to aid the manufacturing, transport, and installation (Zhang and Liu 2014; Zhang et al. 2014). These additives, as well as their degradation products, may leach into water distribution systems and contaminate tap water (inorganic or/and organic) (Brocca et al. 2002). Thus, it is probable that pipes can be an extra source of contaminants (regulated and/or unregulated) in water. Consequently, the existence of ECs in drinking water distribution systems has been recognized and become a subject of public concern. The release of emerging and other contaminants from pipes that may adversely influence the chemical quality of drinking water and their effects on the health of people have been studied worldwide, in countries such as Turkey (Endirlik et al. 2019), Iran (Abdolahnejad et al. 2019), China (Gao et al. 2019), Germany (Mintenig et al. 2019), and Portugal (Santana et al. 2014). Among the ECs released from pipes into tap water, microplastics (MPs), bisphenol A (BPA), phthalates, nonylphenol (NP), perfluoroalkyl, and polyfluoroalkyl substances (PFAS) have received more attention. The presence of these contaminants in drinking water is concerning due to their effects on health. Some of the health effects of these contaminates are noted in Table 1.

To date (October, 2021), there are several studies on migration of MPs, BPA, phthalates, NP, and PFAS from pipes into tap water. But, the knowledge about the migration of ECs from the pipes used in drinking water distribution systems and the potential risks of these contaminants is still lacking. Also,

Table 1 Some health effects of MPs, BPA, phthalates, NP, and PFAS

Contaminant	Health effect	Ref
MPs[1]	Immunosuppression, immune activation, and abnormal inflammatory responses	(Prata 2018; Prata et al. 2020; Rahman et al. 2020)
BPA[2]	Estrogenic impacts, developing prostate cancer, and decreased fertility	(Batista and Rocha 2013; Nascimento and Rocha 2018)
Phthalate	Developmental and reproductive harm, neurological, metabolic, and immune effects	(Abtahi et al. 2019; Net et al. 2015)
NP[3]	Effects on nervous system, immune systems, and reproductive system	(Acir and Guenther 2018; Gan et al. 2015; Liu et al. 2020)
PFAS[4]	Cancer, immune system dysfunction, liver damage, developmental and reproductive harm, and tumor induction	(Liu et al. 2020; Ojo et al. 2020; Pelch et al. 2019)

[1]Microplastics; [2]bisphenol A; [3]nonylphenol; [4]perfluoroalkyl and polyfluoroalkyl substances

there is no review article especially related to the transmission of ECs from drinking water pipes into water. Hence, in this review, we study existing literature on the migration of ECs from pipes used in drinking water distribution networks into tap water, focusing on MPs, BPA, phthalates, NP, and PFAS. Even then, there are other compounds besides the compounds mentioned here, but in this study we only considered the studies that directly considered the mentioned ECs. Therefore, in the present review, the mentioned ECs are reviewed with a specific emphasis on their occurrence and source of their existence in tap water related to pipe types. Finally, the potential hazards of these contaminants in tap water are evaluated.

Materials and methods

Review methodology

In order to investigate the migration of ECs from the pipes used in drinking water distribution systems, published manuscripts were gathered by a search of the electronic literature Scopus, PubMed, Science Direct, Web of Science, ProQuest, Springer Link, and Publons from January 1, 2000 to October 30, 2021, using the keywords ("contaminants migration from pipes into water" OR "drinking water distribution systems" OR "tap water" OR "microplastics" OR "bisphenol A" OR "phthalates" OR "nonylphenol" OR "perfluoroalkyl and polyfluoroalkyl substances"). Our research was limited to peer-reviewed publications in the "English language." Also, we reviewed the references of the screened papers in order to find additional published manuscripts that were not found in the initial search.

After eliminating the duplicate papers, the adopted documents from different databases were selected and then screened with regard to the aim and scope of this review. The unsuitable studies, abstracts, reviews and editorial articles, book chapters, and conference proceedings were not considered. Finally, 92 papers articles were selected for inclusion in the present study. A flow diagram of the study selection process for this review is presented in Fig. 1. Also, the number of studies investigating the migration of contaminants from pipes into drinking water distribution systems included in the present study is shown in Fig. 2. It should be noted that some of the reports studied more than one of contaminants considered in the present study.

Inclusion and exclusion criteria

The initial search of the databases found 12,580 articles. Duplicate articles numbered 3572 articles were removed by using EndNote X8.2.0 software. Also, the titles and abstracts of the remaining articles were controlled for inclusion. After this screening, 8856 articles including editorials, book chapters, review articles, and irrelevant studies were excluded. Most of the screened papers were excluded from this review because they were not related to our topics. Furthermore, 11 related articles were included that were detected in the reference lists of remaining and review articles. The number scientific papers chosen for full-text review was 132, and these were examined closely in order to ensure they met the inclusion criteria. Among these remaining articles, the papers were filtered by the following final criteria: (1) the articles measured MPs, BPA, phthalate, NP, and PFAS in drinking water distribution systems, and (2) the articles were published in English. Finally, after applying the mentioned criteria, 92 articles were included in the present review (Fig. 1).

Human health–risk assessment

Ingestion is the major route of exposure to chemicals in drinking water (Abtahi et al. 2019). In this work, the chronic daily intake (CDI) of contaminants via ingestion was calculated according to the following equation (Bortey-Sam et al. 2015; Wongsasuluk et al. 2014):

$$\mathrm{CDI}_{\mathrm{Ingestion}} = \frac{C \times \mathrm{IR} \times \mathrm{ED} \times \mathrm{EF}}{\mathrm{BW} \times \mathrm{AT}} \quad (1)$$

where C is the maximum level of target compound (items/L, ng/L, μg/L, and mg/L), IR is the consumption rate of the water being studied (3.45 and 2 L/day for adults and children, respectively), ED is the exposure duration (70 years for adults and 10 years for children), EF is the frequency of exposure (365 days/year), BW is the average body weight (60 kg for adults and 25 kg for children), and AT is the average time, which is equal to 25,550 days for adults (i.e., 70 years × 365 days/year) and 3650 days for children (i.e., 10 years × 365 days/year).

The non-carcinogenic hazard index (HI) is estimated with dividing the value of CDI by the reference dose (RfD). The computation of HI for one contaminant can be conducted by the following equation (Kamunda et al. 2016; Wongsasuluk et al. 2014):

$$\mathrm{HI} = \frac{\mathrm{CDI}}{\mathrm{RfD}} \quad (2)$$

The RfD is the reference doses of exposure to contaminant via ingestion. The HI values are divided in two categories: less than 1 indicating no significant risk of relevant health effects and more than 1 with a significant risk of relevant health effects (Bortey-Sam et al. 2015; Wongsasuluk et al. 2014).

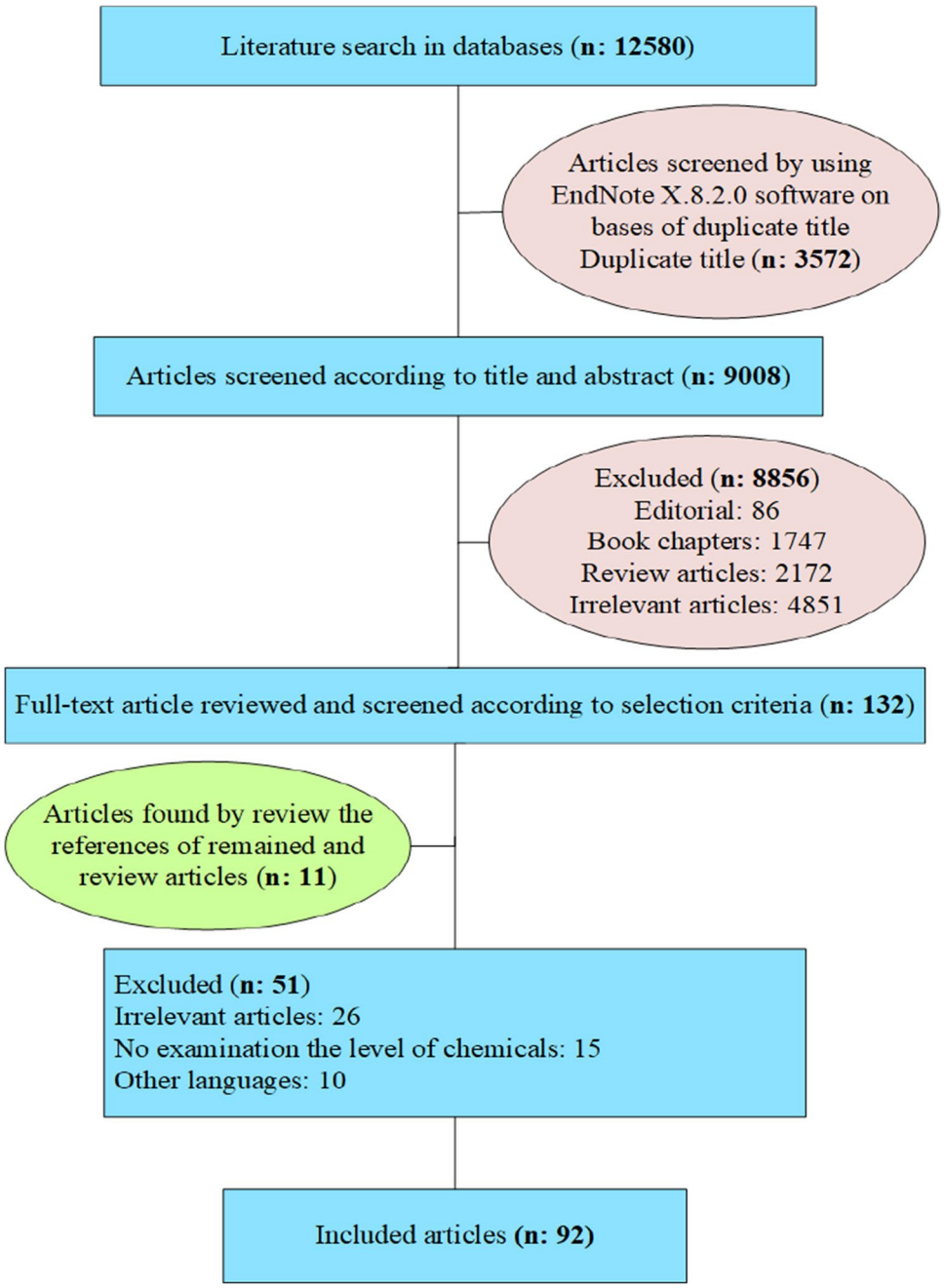

Fig. 1 Flowchart of study selection for MPs, BPA, phthalate, NP, and PFAS in drinking water distribution systems

Cancer risks (CR) were assessed as the incremental probability of an individual developing cancer over a lifetime as a result of exposure to a potential carcinogen. The following equation (Eq. 3) was used for the calculation of the carcinogenic risk (Titilawo et al. 2018):

$$CR = CDI \times CSF \tag{3}$$

where CSF is the cancer slope factor $(mg/kg/day)^{-1}$. Finally, the CR for each carcinogen compound was compared with the acceptable risk (Man et al. 2013; Titilawo et al. 2018).

Major emerging contaminants' release from pipes used in water distribution systems

Microplastics (MPs)

In spite of the irrefutable advantages of plastics in daily life (e.g., packaging, medical devices, electronic and electrical parts), there is a growing concern due to probable harmful influences of plastics and MPs on human health (Koelmans

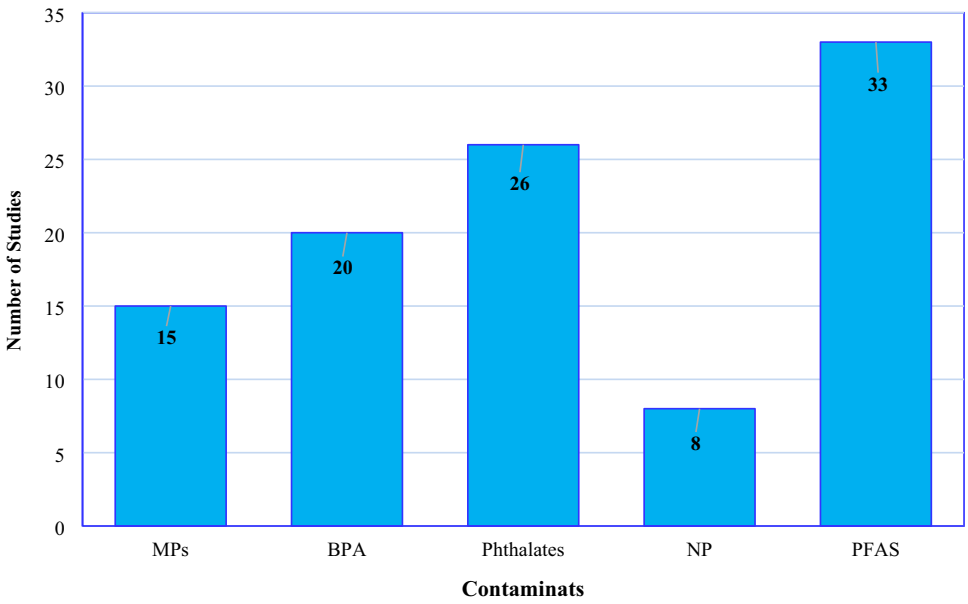

Fig. 2 The number of studies investigating migration of contaminants [microplastics (MPs), bisphenol A (BPA), phthalate, nonylphenol (NP), and perfluoroalkyl and polyfluoroalkyl substances (PFAS)] from pipes into drinking water distribution systems in the present study (More than one contaminant has been investigated in some studies)

et al. 2019a; Kumar et al. 2021b; Zuccarello et al. 2019). MPs are plastic particles with a size of smaller than 5 mm and have received substantial consideration as a new emerging contaminant class due to their global distribution, both from research societies and the community (Akhbarizadeh et al. 2021a, 2020b; Dobaradaran et al. 2018). Chemical toxicity, physical damages, and microbial risks are related to the effects of MPs, and these effects are probably dose-dependent (Koelmans et al. 2019a; Prata et al. 2020; Rahman et al. 2020; Rist et al. 2018). Oxidative stress, immunological responses, sugar biosynthesis, and hemocyte mortality are some of the toxicity mechanisms of MPs (Avio et al. 2015; Lagarde et al. 2016; Paul-Pont et al. 2016). In recent years, MP particles have been identified in different matrixes such as air (Abbasi et al. 2019; Akhbarizadeh et al. 2021b), food (Akhbarizadeh et al. 2020a; Liebezeit and Liebezeit 2014, 2015), water sources (Akhbarizadeh et al. 2020b; Li et al. 2020), wastewater effluents (Picó et al. 2021; Takdastan et al. 2021), marine environments (Akhbarizadeh et al. 2021a; Dobaradaran et al. 2018), wetlands (Kumar et al. 2021a; Su et al. 2019), and rivers (Eo et al. 2019; Kataoka et al. 2019). Also, the impacts of MPs on biota and ecological systems have been recognized (Fu et al. 2020; Jung et al. 2021; Prata et al. 2020). According to the present studies, plastic pipes in water distribution networks are a significant source of MPs. However, data concerning the existence of MPs in tap water are very scarce and to the present time only 13 works focused on these contaminants. The information reported in the included studies is shown in Table 2.

In the study of Mintenig et al. (2019) in Germany, the average number of MPs, with a size distribution of 50–150 µm, in tap water samples was reported to be less than 1 particle/L (Mintenig et al. 2019), and this number is very low compared to the average number of MPs reported in other countries worldwide. Also, in another study in Germany, Weber et al. (2021) investigated MPs, with a size distribution of 10–1000 µm in tap water samples collected from three house junctions, one transmission station, and five drinking taps. Based on the their findings, no MPs were identified in the tap water samples of consumption taps (Weber et al. 2021). Besides the differences present in the treatment methods employed in water treatment plants, the variations in the reported numbers of detected MPs, as shown in Table 2, can be due to differences in types of plastic pipes, fittings, and tanks used in the water distribution systems (Mintenig et al. 2019). Based on Table 2, the highest abundance of MPs was detected in a study by Pivokonsky et al. (2018) in Czech Republic. Also, in this study, the MPs in the tap water samples were investigated with micro Raman spectrometry, and it was discovered that up to 95% of the detected MPs had a size of 1–10 µm (Pivokonsky et al. 2018). However, these tiny fine particles have not detected in other studies performed on tap water (Kankanige and Babe 2020; Kosuth et al. 2018; Mintenig et al. 2019; Pivokonsky et al. 2018; Tong et al. 2020; Zhang et al. 2020a). The differences between the published data can be ascribed to various parameters such as limitations in the techniques and analytical methods used, and differences in the sample volumes, pipe materials used in the water distribution networks, and the study areas (Tong et al. 2020). In a study from China, Zhang et al. (2019) found the average number of MPs in 7 tap water samples, with size categories of < 100 µm [1.2%], 100–500 µm [26%], and > 500 µm [72.8%], was 0.7 items/L. The lowest size of MPs reported in this study, of < 100 µm, was due to the analytical methods used for the determination of MPs size. Also, rayon, polyethylene terephthalate (PET),

Table 2 The size distribution and average number per liter of MPs in drinking water distribution systems

Total number of samples	Size distribution				Country (city or area; year of sampling)	Instrumental analysis	Ref
	Number of MPs/L						
18	10–5000 µm				Germany (Russelsheim; 2019–2020)	Micro Raman spectroscopy	(Weber et al. 2021)
	0						
21	> 50 µm				Spain (Barcelona; 2019)	Stereomicroscope	(Dalmau-Soler et al. 2021)
	0.02						
32	6–50 µm				Brazil (Brasilia; 2018)	Fluorescence microscope	(Pratesi et al. 2021)
	316						
4	1–100 µm				China (Changsha; 2019)	FTIR[2]	(Shen et al. 2021)
	343.5						
24	50–150 µm				Germany (Nethen, Holdorf, Grossenkneten, Sandelermoens, and Thuelsfelde; 2014)	FTIR	(Mintenig et al. 2019)
	0. 7*						
159	100–500 µm				Cuba, Ecuador, England, France, Germany, India, Indonesia, Ireland, Italy, Lebanon, Slovakia, Switzerland, Uganda, and USA (156 cities; 2017)	Stereomicroscope	(Kosuth et al. 2018)
	4.34						
38	< 100 µm		> 100 µm		China (38 cities; NR[1])	Micro Raman spectroscopy	(Tong et al. 2020)
	137		303				
17	10–100 µm		> 100 µm		Denmark (17 areas; 2018)	Stereomicroscope and µFTIR[3]	(Feld et al. 2021)
	0.2		5.62				
110	2.7–149 µm		≥ 150 µm		China (Hong Kong; 2018)	Stereomicroscope	(Lam et al. 2020)
	1.51		0.67				
45	6.5–53 µm	53–500 µm		> 500 µm	Thailand (Rangsit campus; 2019)	FTIR	(Kankanige and Babe 2020)
	56	34		6			
42	19–50 µm	50–100 µm		> 100 µm	United States, France, Finland, and Germany (Kashiwa, Noda, Toshima, Shinjuku, Tachikawa, Machida, Yokohama, Yamagata, California and Nevada, Helsinki, Paris, and Munich; 2019–2020)	FTIR	(Mukotaka et al. 2021)
	15	17		7.3			
15	< 10 µm	10–140 µm		> 140 µm	Sweden (south Sweden; 2019)	µFTIR	(Kirstein et al. 2021)
	4	237		10			
42	< 500 µm	500–1000 µm		1000–5000 µm	Mexico (Mexico City; 2019)	Micro Raman spectroscopy	(Shruti et al. 2020)
	9	4.5		4.5			
7	< 100 µm	100–500 µm		≥ 500 µm	China (Qingdao; 2018)	FTIR	(Zhang et al. 2020a)
	0.008	0.182		0.51			
3	1–5 µm	5–50 µm		50–100 µm	Czech Republic (NR; 2017–2018)	FTIR and Micro Raman spectroscopy	(Pivokonsky et al. 2018)
	667	740		2			

[1]Not reported; [2]fourier transform infrared spectroscopy; [3]micro Fourier transform interferometer; *MPs/m^3 (based on reported values in the study)

and polyethylene (PE) were the types of polymer MPs most frequently detected in the tap water samples in further research from China (Zhang et al. 2020a). Based on the findings of a research in China, the detected average number of MPs was 2.2 items/L. The number of MPs in tap water, in the size classes of 2.7–149 and ≥ 150 μm, were 69.2% and 38.2%, respectively. According to the results of the research reviewed, the presence of MPs in water can be ascribed to the release these particles following the mechanical abrasion of plastic-coated or plastic-lined water pipes and tanks (Lam et al. 2020). In the study of Tong et al. (2020) in China, the average number of MPs in water samples collected from distribution systems was reported to be 440 items/L with 137 items/L in a size of < 100 μm and 303 items/L in a size of > 100 μm. The most abundant identified polymer types were PE, PP, PS, and PET. As the pipes used in the water supply networks in China are mostly plastic pipes, this may cause MPs pollution and enhance the MPs numbers in tap water samples (Tong et al. 2020). Based on the results of the a study in China, average number of MPs, in the size range of 1 to 100 μm, in tap water samples was reported to be 343.5 items/L, and this was dependent on the materials used in the transport pipelines in the drinking water distribution network (Shen et al. 2021). The differences present in the results of the studies done in China by Shen et al. (2021) [in Changsha] (Shen et al. 2021), Lam et al. (2020) [in Hong Kong] (Lam et al. 2020), Tong et al. (2020) [in 38 cities of China] (Tong et al. 2020), and Zhang et al. (2020a) [in Qingdao] (Zhang et al. 2020a) may be due to differences in the regions studied, the geographical conditions, the pipelines used in the drinking water distribution systems, and the analytical methods. In a recent study in Sweden, Kirstein et al. (2021) investigated distribution pipes (mainly of stainless steel, cement, PE, and cast iron) with different ages to determine the potential differences in the abundance of MPs. The presence of PE pipes with an age of more than 10 years had considerable effect on the abundance of MPs in the distribution system. Eight polymers of various types, comprising PA, polyester, acrylic, PVC, PS, PE, polyurethane (PU), and PP, were identified in tap water in varying amounts. Also, a very low number of MPs was identified in the outlets of the water treatment plants compared to the water samples taken from the distribution network. Kirstein et al. concluded that the occurrences of MPs in drinking water distribution systems may be due to abrasion and/or damage during pipeline construction (Kirstein et al. 2021). The reason for the dominance of the various polymer types (PVC, PE, PA and epoxy resin) in tap water samples can be explained by the abrasion of pipes and fittings in the distribution network, which are mostly built of PVC, PE, and PA coated with epoxy resin. Though plastic is a resistant and durable substance, abrasion may happen and this is a probable explanation for the occurrence of the specified particles of plastic in tap water (Kankanige and Babe 2020; Mintenig et al. 2019). High contact time of water with polymer pipes can cause the breakdown of polymers to a smaller size and damage to external structures. This damage helps promote the migration of more MPs into the drinking water. These damages help promote the migration of more MPs into drinking water (Ye et al. 2020). Additives to plastics and the components of plastics may also leach from the MP particles into drinking water distribution systems during transport and storage. All the additives present in MPs may leach and be absorbed in the human body after the drinking of tap water (Brocca et al. 2002; Whelton and Nguyen 2013). These components may have various toxicological impacts on the health of people (Brown et al. 2001; Schirinzi et al. 2017). It should be noted that the application of Raman microscopy or FTIR for the identification of smaller MPs compared to the manual sorting and subsequent identification of MPs is relatively easy to assess due to the measuring area of a filter (Koelmans et al. 2019b). Thus, the identification of smaller MPs with Raman microscopy or FTIR increases the number of detected MPs.

Different identification methods with different capabilities in the counting of MPs, different size categories in studies, and a general lack of a uniform detection and identification method for MPs are the main problems present when comparing the results of studies on the occurrence of MPs in tap water. It should be noted that there is still a big scientific limitation to the ability to count and identify the MPs in water (as well as in any sample matrix), especially the MPs with a size of less than 50 μm. Because of the importance to health of the daily intake of drinking water, the scientific community they should improve the identification method for MPs and also extend the studies to include nanoplastics (NPs), with the size range of 1–1000 nm (Schwaferts et al. 2019). Therefore, more sophisticated research on the amount, type, size, and source of MPs in water distribution systems, particularly by considering various types of plastic pipes, are needed to cover this scientific gap.

Bisphenol A (BPA) 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 additives mainly used in plastics to enhance their transparency, durability, flexibility, and longevity (Arnold et al. 2013; Sakhi et al. 2014; Shi et al. 2012). These plasticizers have entered widely and simply into the environment as they are not chemically bound to the products (Yang et al. 2018). It was predicted that the worldwide usage of phthalate plasticizers increased by 1.3% each year from 2017 to 2022 (Luo et al. 2018). BPA and the most common phthalates including butyl benzyl phthalate (BBP), di-n-butyl phthalate (DBP), diethyl phthalate (DEP), di-(2-ethylhexyl) phthalate (DEHP), di-isononyl phthalate (DiNP), di-isobutyl phthalate

(DiBP), di-isodecyl phthalate (DiDP), di-n-butyl phthalate (DnBP), di-methyl phthalate (DMP), and di-n-octyl phthalate (DnOP) are recognized as EDCs in tap water that can be connected to chronic health effects (Abtahi et al. 2019; Moazzen et al. 2018; Santhi et al. 2012). BPA can cause endocrine disruption, reproductive and developmental toxicity, neurotoxicity, and immunotoxicity (Ma et al. 2019; Qiu et al. 2019). Phthalates may cause endocrine disruption, oxidative stress, and reproductive toxicity (Sedha et al. 2021; Zhang et al. 2021). The guidelines of the US Environmental Protection Agency (EPA) for the TDI values of BPA, BBP, DBP, DEHP, DEP, DiDP, DMP, and DnOP are 50, 200, 100, 20, 800, 150, 100, and 10 µg/kg-bw/day, respectively (USEPA 2011). Furthermore, a standard level of 8 µg/L for DEHP in water is recommended by the World Health Organization (WHO 2008).

Until now, there have been only a limited number of studies on the concentrations of BPA and phthalates in water supply networks and the results are given in Tables 3 and 4, respectively. In a recent report, Cantoni et al. (2021) in Italy evaluated BPA release from pipes into water with high-performance liquid chromatography–tandem mass spectrometry (HPLC–MS/MS). The average level of BPA was 1129 ng/L, and it was clearly highlighted that the leakage of BPA from plastic constituents used in drinking water distribution systems pipelines is a major source of BPA in tap drinking water (Cantoni et al. 2021). The findings of previous studies on BPA corroborate that this plasticizer may be leached from polymer pipes into water supply networks (Colin et al. 2014; Goeury et al. 2019; Rajasärkkä et al. 2016; Santhi et al. 2012; Sodré et al. 2010; Tang et al. 2012; Zhang et al. 2019). This contamination can result from polymer decomposition during transport and storage in drinking water distribution systems (Abtahi et al. 2019). The concentration of BPA was 87.33 ng/L in tap drinking water samples from pipelines with epoxy resin lining in Finland. The existence of BPA in tap water might be due to epoxy pipelines upstream of the sampled water (Rajasärkkä

Table 3 The average concentration of BPA in drinking water distribution systems

Total number of samples	Concentration (ng/L)	Country (city or area; year of sampling)	Instrumental analysis	Ref
3	1129	Italy (NR[1]; NR)	HPLC–MS/MS[2]	(Cantoni et al. 2021)
155	1.55	Malaysia (Putrajaya; 2018)	LC–MS/MS[3]	(Wee et al. 2020)
20	12.80	China (20 areas; 2017)	UPLC-MS/MS[4]	(Zhang et al. 2019)
11	1.50	Canada (Quebec and Ontario provinces; 2017–2018)	UHPLC-MS/MS[5]	(Goeury et al. 2019)
20	28.83**	South Africa (Pretoria and Cape Town; 2013–2014)	UPLC-MS/MS	(Van Zijl et al. 2017)
6	87.33	Finland (Helsinki; 2015)	GC-MS[6]	(Rajasärkkä et al. 2016)
18	12**	Taiwan (Taipei and Kaohsiung; NR)	UPLC-MS[7]	(Cheng et al. 2016)
27	174	USA (New York; 2012)	HPLC–MS/MS	(Subedi et al. 2015)
291	<9	France (Paris, Rennes, Lille, Strasbourg, Lyon, Bordeaux, Toulouse, Nice, Marseille, Corsica, Guadeloupe, Reunion, Martinique, and Guyana; NR)	Online SPE-LC–MS/MS[8]	(Colin et al. 2014)
2	1.1	Japan (NR; 2012)	LC–MS/MS	(Kosaka et al. 2012)
3	37.33	Poland (South Poland; 2017)	GC–MS/MS[9]	(Kmiecik et al. 2020)
6	14.8*	Spain (Madrid; 2012)	LC-LC–MS/MS[10]	(Esteban et al. 2014)
11	6	Spain (Valencian Community; 2012)	LC–MS/MS	(Carmona et al. 2014)
35	1.56*	Italy (35 cities; 2012)	LC-LC–MS/MS	(Maggioni et al. 2013)
30	14.10	Malaysia (Kuala Lumpur; 2008–2009)	GC–MS	(Santhi et al. 2012)
30	160**	China (Henan province; NR)	NR	(Tang et al. 2012)
123	0.99**	Canada (Ontario; NR)	LC–MS/MS	(Kleywegt et al. 2011)
12	160	Brazil (Campinas; 2006)	GC–MS	(Sodré et al. 2010)
6	99	China (Guangzhou; NR)	GC–MS	(Li et al. 2010)
7	0.25**	Spain (Valles area; NR)	GC–MS	(Casajuana and Lacorte 2003)

[1]Not reported; [2]high-performance liquid chromatography–tandem mass spectrometry; [3]liquid chromatography–tandem mass spectrometry; [4]ultra-performance liquid chromatography–tandem mass spectrometer; [5]ultra-high-performance liquid chromatography–tandem mass spectrometry; [6]gas chromatography–mass spectrometry; [7]ultra-performance liquid chromatography–mass spectrometer; [8]online solid-phase extraction-liquid chromatography–tandem mass spectrometry; [9]gas chromatography–tandem mass spectrometry; [10]liquid chromatography–liquid chromatography–tandem mass spectrometry

*Median (based on reported values in the study)

**Maximum (based on reported values in the study)

Table 4 The average concentration of phthalates in drinking water distribution systems

Total number of samples	Compounds	Concentration (μg/L)	Country (city or area; year of sampling)	Instrumental analysis	Ref	Total number of samples	Compounds	Concentration (μg/L)	Country (city or area; year of sampling)	Instrumental analysis	Ref
5	BBP[1]	0.024***	China (Changzhou, Suzhou, Wuxi, Xuzhou, and Yancheng; 2010)	LC–ESI–MS/MS[15]	(Shi et al. 2012)	7	BBP	1.40	Vietnam (Hanoi; 2020)	GC-MS[16]	(Le et al. 2021)
	DBP[2]	0.11***					DMP[10]	0.35			
	DEHP[3]	0.28***					DBP	0.79			
	DiBP[4]	0.11***					DEP	1.09			
	DiDP[5]	0.09***					DCHP[11]	0.54			
	DiNP[6]	0.29***					DiBP	0.45			
	DEP[7]	0.003***					DEHP	0.45			
	DnBP[8]	0.93***					DnHP[12]	0.41			
	DnOP[9]	0.07***					DnOP	5.34			
							DPP[13]	0.37			
65	DEHP	0.08**	China (Wuhan; NR)	UHPLC-MS/MS[17]	(Li et al. 2019)	23	BBP	0.03	Taiwan (Kaohsiung; 2011–2013)	LC–ESI–MS/MS	(Yang et al. 2014b)
	DiBP	0.11**					DEHP	0.17			
	DEP	0.06**					DEP	0.03			
	DnBP	0.43**					DiDP	0.01			
7	BBP	0.017***	Spain (Valles area; NR)	GC–MS	(Casajuana and Lacorte 2003)		DiNP	0.30			
	DBP	0.032***					DMP	0.002			
	DEHP	0.33***					DnBP	0.09			
	DEP	0.09***					DnOP	0.03			
12	BBP	0.53	China (Hong Kong; 2017–2018)	HPLC–MS/MS[18]	(Li et al. 2021b)	89	BBP	< 0.05	France (NR; 2015–2016)	Online SPE-LC–MS/MS[19]	(Bach et al. 2020)
	DBP	0.34					DBP	0.95			
	DEHP	0.08					DEHP	< 0.5			
	DEP	0.10					DEP	0.25			
	DMP[10]	0.86					DiBP	1.29			
	DnOP	0.21									
40	BBP	Average level of total phthalates: 0.76 *	Iran (Tehran; 2018)	GC-FID[20]	(Abtahi et al. 2019)	225	BBP	0.02	China (35 cities; 2009–2012)	GC–MS	(Liu et al. 2015)
	DBP						DBP	0.01			
	DEHP						DEHP	0.77			
	DEP						DEP	0.03			
	DMP						DMP	0.07			
	DnOP						DnOP	0.02			
1	BBP	0.03	Portugal (NR[14]; NR)	GC–MS	(Serôdio and Nogueira 2006)	5	BBP	0.002	Czech Republic (Prague; NR)	GC[21]	(Prokůpková et al. 2002)
	DEHP	0.06					DEHP	0.06			
	DBP	0.52					DEP	0.07			
	DEP	0.19					DMP	0.08			
	DMP	0.04					DnBP	0.05			

Table 4 (continued)

Total number of samples	Compounds	Concentration (µg/L)	Country (city or area; year of sampling)	Instrumental analysis	Ref	Total number of samples	Compounds	Concentration (µg/L)	Country (city or area; year of sampling)	Instrumental analysis	Ref
33	BBP	0.05	Iran (Isfahan; 2017)	GC–MS	(Abdolahnejad et al. 2019)	146	DEHP	0.17	China (24 cities; 2015–2018)	GC–MS	(Ding et al. 2019)
	DBP	0.01					DiBP[7]	0.10			
	DEHP	0.17					DEP	0.005			
	DEP	0.04					DMP	0.01			
7	DBP	0.79	Vietnam (Hanoi; 2020)	GC–MS	(Le et al. 2021)	4	DEHP	0.03	Taiwan (NR; 2011–2012)	LC–ESI–MS/MS	(Yang et al. 2014a)
	DEP	1.09					DiBP	0.034			
	DMP	0.35					DnBP	0.02			
	DiBP	0.45									
20	DBP	1.06***	South Africa (Pretoria and Cape Town; 2013–2014)	UPLC-MS/MS[22]	(Van Zijl et al. 2017)	3	DBP	0.6***	France (Paris; 2008)	HRGC[23]	(Martine et al. 2013)
	DEHP	5.15***					DEHP	0.66***			
	DiNP	1.25***					DEP	0.9***			
6	BBP	0.53	China (Tianjin; 2019)	GC–MS	(Wang et al. 2021)	5	DEHP	0.93	Greece (NR; NR)	GC–MS	(Psillakis and Kalogerakis 2003)
	DBP	0.54					DEP	0.30			
	DEHP	1.33					DnBP	1.04			
30	DBP	0.93	China (Henan province; NR)	NR	(Tang et al. 2012)	NM	DBP	0.19	Spain (North-West of Spain; NR)	GC–MS	(Regueiro et al. 2008)
	DEHP	12.48				NM	DBP	0.17	Poland (Zabrze; NR)	GC-FID	(Bodzek et al. 2004)
	DEP	44									
7	DBP	0.63	Spain (Madrid; 2012)	GC	(Domínguez-Morueco et al. 2014)	4	DEHP	0.32	Portugal (Lisbon and three small neighbor cities; NR)	GC–MS	(Santana et al. 2014)
	DEP	0.38					DiBP	0.17			
3	DBP	0.001	China (Wuhan; NR)	GC–MS	(Luo et al. 2012)	6	DEP	0.09	China (Wuhan; NR)	GC-FID	(Xu et al. 2007)

[1]Butyl benzyl phthalate; [2]di-n-butyl phthalate; [3]di-(2-ethylhexyl) phthalate; [4]di-isobutyl phthalate; [5]di-isodecyl phthalate; [6]di-isononyl phthalate; [7]diethyl phthalate; [8]di-n-butyl phthalate; [9]di-n-octyl phthalate; [10]di-methyl phthalate; [11]dicyclohexyl phthalate; [12]di-n-hexyl phthalate; [13]diphenyl phthalate; [14]not reported; [15]liquid chromatography–electrospray ionization–tandem mass spectrometry; [13]liquid chromatography–tandem mass spectrometry; [16]gas chromatography–mass spectrometry; [17]ultra-high-performance liquid chromatography–tandem mass spectrometry; [18]high-performance liquid chromatography–tandem mass spectrometry; [19]online solid-phase extraction–liquid chromatography–tandem mass spectrometry; [20]gas chromatography–flame ionization detector; [21]gas chromatography; [22]ultra-performance liquid chromatography–tandem mass spectrometer; [23]high-resolution gas chromatography

*Based on reported values in the study

**Median (based on reported values in the study)

***Maximum (based on reported values in the study)

et al. 2016). In a study in South wAfrica, the maximum concentration of BPA was 28.83 ng/L. Based on the findings of the studies reviewed, although various processes in water treatment plants can eliminate BPA from the water leaving the plants, this contaminant may migrate from the pipes used in the water distribution system and contaminate the water available to the consumer (Van Zijl et al. 2017). Also, the the difference in the results of the studies done in Spain by Esteban et al. (2014) and Cantoni et al. (2021) may be due to differences in the pipelines used in drinking water supply networks and the geographical conditions as well as in the analytical methods. It should be noted that the use of different analytical methods, such as such as gas chromatography–mass spectrometry (GC–MS), liquid chromatography–mass spectrometry (LC–MS), and high-performance liquid chromatography (HPLC), is one of the reasons for the difference between BPA concentrations found in the published studies (Xue et al. 2013). The findings of Santhi et al. (2012) indicated that the average level of BPA in tap water was higher for polymer pipes than in pipes made of other substances (Santhi et al. 2012). Most of the preliminary substances such as epoxy resins that are used to produce polymer pipes would not be envisaged to include BPA, but cross pollution of BPA through the production of polymer material may be associated with tap water contamination (Colin et al. 2014). Epoxy resins are commonly applied as lacquers to protect water pipes and water supply reservoirs against corrosion, especially when the water is left standing in the pipes. The use of epoxies in small-diameter pipes (such as water service lines), which have high proportions of surface area to volume and flow intermittently, maximizes the potential for BPA to leach into water distribution systems (Lane et al. 2015). Due to the few studies available at present, more study is required to specify the exact impact of pipelines in enhancing BPA release into drinking water distribution systems as well as the health effects of BPA from drinking water for humans.

Regarding phthalates, several scientific studies investigated their presence in drinking water pipes as shown in Table 4. The impact of pipe type on the potential release of phthalates into the water distribution systems have been examined in various studies. For instance, Abtahi et al. (2019) examined the effects of plumbing pipe type on the phthalate concentrations of tap water, and reported all that polymer pipes increased the levels of phthalate including DBP, BBP, DEP, DMP, DEHP, and DnOP in drinking water distribution systems. These findings showed that water phthalate levels can increase after even a short time contact of tap water with plastic materials (Abtahi et al. 2019). Likewise, Abdolahnejad et al. (2019) evaluated the concentrations of BBP [0.05 µg/L], DBP [0.01 µg/L], DEHP [0.17 µg/L], and DEP [0.04 µg/L] with GC–MS in water samples taken from iron and plastic pipes used in water distribution systems. They reported that, except for BBP, the average levels of phthalates in plastic pipes were more than in metal pipes (Abdolahnejad et al. 2019). Ding et al. (2019) reported a detection frequency of investigated phthalates including DEHP, DiBP, DEP, and DMP of more than 90% in 24 cities throughout China, with the exception of DnOP which was found in only 9% of the water samples (Ding et al. 2019). Liu et al. (2015) also detected the six target phthalates including DBP, BBP, DEP, DEHP, DMP, and DnOP from plastic pipes with GC–MS in drinking water distribution systems with average concentrations of 0.02, 0.01, 0.77, 0.03, 0.07, and 0.02 µg/L, respectively (Liu et al. 2015). It is worth to mention that the different results in studies from China may be due to differences in the regions studied, the geographical conditions, the pipelines used in the water distribution systems, and the analytical methods. Similar findings demonstrated that the average levels of phthalates in water samples gathered from polymer pipes were more than from other pipes (Abdolahnejad et al. 2019; Abtahi et al. 2019; Serôdio and Nogueira 2006). According to the findings of research in Greece, the mean levels of phthalates in tap water, analyzed with GC–MS, including DEP, DEHP, and DnBP, were 0.93, 0.3, and 1.04 µg/L, respectively (Psillakis and Kalogerakis 2003). The results were higher than in all the other studies in China [analyzed with a gas chromatography–flame ionization detector (GC-FID)] (Xu et al. 2007) and in the Czech Republic [analyzed with by gas chromatography (GC)] (Prokůpková et al. 2002). This may be due to the use of polymer equipment in water supply network and to different used techniques and analytical methods being used (Psillakis and Kalogerakis 2003). Despite the removal of phthalates in water treatment processes, these contaminants may migrate from pipes into water (Casajuana and Lacorte 2003; Van Zijl et al. 2017). The use of different techniques, such as liquid–liquid extraction (LLE), semi-automated solid-phase extraction (SPE), and solid-phase micro-extraction (SPME)], and analytical methods [such as GC–MS and LC–MS] for the analysis of phthalates in the drinking water supply networks may be among the reasons for the differences in the concentrations of this contaminant found in the various studies (Bach et al. 2020). The use of polymer pipes in the urban distribution system or polymer pipes and reservoirs in the domestic distribution system can influence the concentration level of phthalates in tap water (Abdolahnejad et al. 2019). Scission of polymer chains and degradation of additives can cause the entry of phthalates from pipes into drinking water during transport and storage (Whelton and Nguyen 2013). High surface areas of polymer pipes and, in consequence, the high contact of water to pipes will accelerate the release process of phthalates such as DBP. More phthalate compounds can migrate to water freely, since additives were physically dispersed in the polymer structure rather than being linked

through bonds (Ye et al. 2020). Therefore, the selection of suitable additives in the production of pipes can reduce the release of additives, such as phthalates and BPA, into drinking water distribution systems.

According to the present studies, the use of plastic pipes and reservoirs in municipal distribution systems are the major sources of phthalate pollution in tap water (Serôdio and Nogueira 2006). Thus, further research on the levels of phthalates in tap drinking water during the transfer and storage of drinking water are needed.

Nonylphenol (NP)

Alkylphenols (APs) are a group of EDCs that has raised much environmental concern due to their estrogenic activity (Jie et al. 2017). NP is one of the most common APs that are widely utilized in the production of paints and latex paints, inks, adhesives, pesticides, petroleum recovery chemicals, paper industry, washing agents, textile and leather industry, metal working liquids, cleaners and detergents, personal care products, plastics, additives, and resins (Priac et al. 2017). In recent years, the consumption of NP has increased, especially in developing countries (Barber et al. 2015; Jie et al. 2017) and polluted water and food are the major sources of human exposure to NP (Raecker et al. 2011). NP is widely used in the production of polymer pipes, as an additive in epoxy resins, to enhance some properties, such as polymerization, drying, and plasticity (Liu et al. 2020; Ruczyńska et al. 2020; Saravanan et al. 2019). Growth and developmental toxic effects, the triggering of respiratory toxicity in cells, an estrogenic effect and reproductive toxic effects, are some toxicity mechanisms of NP (Soares et al. 2008; Zha et al. 2008). The EPA guideline for TDI of NP is 5 µg/kg-bw/day (USEPA 2011).

There is little data about NP concentrations in tap drinking water (Table 5). In a study in China, the NP level in 10 tap water samples was investigated and it was in a range of 0.32–5.43 µg/L. The findings of this study indicate that NP can migrate from polymer pipes into the water distribution network, and the NP concentration in the tap water increased as the contact time in the polymer pipes increased (Jie et al. 2017). Cheng et al. (2016) evaluated the presence of NP in tap drinking water with different pipes. The levels of NP in tap water samples taken from PVC pipes were more than the NP concentrations in tap water samples taken from other pipes, such as stainless steel and galvanized (Cheng et al. 2016). In two research studies, in France (Colin et al. 2014) and China (Sodré et al. 2010), it was stated that the occurrence of NP in tap water may be due to the presence of pipes coated with epoxy resins. The differences in results of research done in Italy by Maggioni et al. (2013) [analyzed by liquid chromatography-electrospray ionization-tandem mass spectrometry (LC–ESI–MS/MS) in 35 cities] (Maggioni et al. 2013) and Loos et al. (2007) [analyzed by liquid chromatography-tandem mass spectrometry (LC–MS/MS) in 7 cities] (Loos et al. 2007) may be because of differences in the regions studied and the geographical conditions, the pipes used in water supply networks and the analytical methods.

Although NP can be eliminated from water by water treatment processes, NP may still migrate into drinking water from the pipes used in the distribution systems (Casajuana and Lacorte 2003; Van Zijl et al. 2017). Epoxy coatings that are applied in water distribution network and

Table 5 The range of NP concentration in drinking water distribution systems

Total number of samples	Concentration (µg/L)	Country (city or area; year of sampling)	Instrumental analysis	Ref
10	0.32–5.43	China (Zunyi; 2015)	HPLC[1]	(Jie et al. 2017)
20	<LOD*	South Africa (Pretoria and Cape Town; 2013–2014)	UPLC-MS/MS[2]	(Van Zijl et al. 2017)
18	0.06–0.19	Taiwan (Taipei and Kaohsiung; 2011–2012)	UPLC-MS[3]	(Cheng et al. 2016)
291	<LOD to 0.14	France (Paris, Rennes, Lille, Strasbourg, Lyon, Bordeaux, Toulouse, Nice, Marseille, Corsica, Guadeloupe, Reunion, Martinique, and Guyana; 2011–2012)	LC-MS[4]	(Colin et al. 2014)
6	0.002–0.02	Spain (Madrid; 2012)	LC-LC–MS/MS[5]	(Esteban et al. 2014)
35	<0.007–0.08	Italy (35 cities; 2009)	LC–ESI–MS/MS[6]	(Maggioni et al. 2013)
12	0.19–1.07	Brazil (Campinas; 2006)	GC-MS[7]	(Sodré et al. 2010)
6	<0.01	Italy (Arolo, Ispra, Angera, Sesto Calende, Arona, Stresa, and Verbania; 2006)	LC–MS/MS[8]	(Loos et al. 2007)

[1]High-performance liquid chromatography; [2]ultra-performance liquid chromatography–tandem mass spectrometry; [3]ultra-performance liquid chromatography–mass spectrometry; [4]liquid chromatography–mass spectrometry; [5]liquid chromatography–liquid chromatography-tandem mass spectrometry; [6]liquid chromatography–electrospray ionization–tandem mass spectrometry; [7]gas chromatography–mass spectrometry; [8]liquid chromatography–tandem mass spectrometry

*Limit of detection

household water supply pipelines can release NP into tap drinking water (Liu et al. 2020; Ruczyńska et al. 2020). NP concentrations in water increase with the increase of contact time with the pipe materials (Cheng et al. 2016). Pipe type is an important factor in water quality that can affect the levels of NP released from pipelines into tap drinking water. Further studies are required for considering the impact of this factor on release of NP in drinking water distribution systems.

Perfluoroalkyl and polyfluoroalkyl substances (PFAS)

PFAS are known as a category of man-made contaminants that include a completely or partly fluorinated hydrophobic alkyl chain linked to a hydrophilic end group. From the 1940s, PFAS have been widely applied in different household and industrial usages because of their specific chemical and physical characteristics such as oxidative resistance and thermal stability (Arvaniti and Stasinakis 2015; Thomaidi et al. 2020). They are widely utilized in cookware, paper products, surfactants, fire-fighting foams, and textiles. Furthermore, PFAS are applied in the aviation and automotive industries, electronics, and semiconductor production (Ahrens 2011; De Voogt and Sáez 2006). PFAS can cause neurotoxicity, developmental toxicity, and immunotoxicity (Gaballah et al. 2020; Neagu et al. 2021). According to the recent studies, perfluoropentanoic acid (PFPeA), perfluorobutanoic acid (PFBA), perfluoroheptanoic acid (PFHpA), perfluorohexanoic acid (PFHxA), PFNA, perfluorooctanoic acid (PFOA), perfluorodecanoic acid (PFDA), perfluorohexane sulfonate (PFHxS), perfluorobutane sulfonate (PFBS), and perfluorooctane sulfonate (PFOS) are the main compounds of PFAS that have been identified in tap drinking water (Endirlik et al. 2019; Lu et al. 2017; Park et al. 2018; Schwanz et al. 2016). PFAS are relatively new chemicals and, although under scrutiny from water providers, there are at present few standards on the acceptable values for them in water. In the present standards, the values of 70 ng/L for combined PFOS and PFOA and 70 ng/L for PFOA and PFHxS are recommended for the lifetime drinking water health (Park et al. 2018). Also, the European Commission (EC) has determined a standard level of 0.5 μg/L for total PFAS in water (European-Commission 2020). Furthermore, the EPA guidelines for TDI levels of PFOA and PFOS are identical, with a level of 20 ng/kg-bw/day (USEPA 2016a). These levels are estimated for acute exposure while long-term exposures may be more appropriate for water (Schwanz et al. 2016).

The concentrations of PFAS as reported in former studies are given in Table 6. In a recent study in China, 16 PFAS compounds in 72 tap water samples, examined by HPLC–MS/MS, were investigated. More compounds of PFAS were detected in this study compared to the other studies. Also, the concentrations of more detected compounds of in this study were higher than in the other studies (Chen et al. 2021). In another study in China, Chen et al. (2019) investigated the levels of PFAS in tap water by high-performance liquid chromatography–liquid chromatography-mass spectrometry (HPLC-LC/MS) and compared the results with global levels (Chen et al. 2019). The PFAS levels in tap water in this study were higher than in those reported in some tap water samples analyzed by HPLC–MS/MS in the USA (Dasu et al. 2017) and Ghana (Essumang et al. 2017) but at the same levels as those reported in tap water samples analyzed by HPLC–MS in China (Lu et al. 2017). According to the findings of a study done by Chen et al. (2019), the pipeline distribution process may have a considerable effect on the levels of PFAS in tap water. Short-chain PFAS have great stability in the water phase and may travel to the far end of the pipeline (Chen et al. 2019). Park et al. (2018) investigated 44 samples of tap water samples from South Korea with high-performance liquid chromatography–electrospray ionization–tandem mass spectrometry (HPLC–ESI–MS/MS). The most commonly found PFAS were PFHxS, PFOA, PFHeA, and PFPxA, with the average levels of 15.1, 5.83, 5.51, and 5.52 ng/L, respectively (Park et al. 2018). These results were comparable to the results of former studies in different countries (Schwanz et al. 2016; Zafeiraki et al. 2015). In 2016, Schwanz et al. investigated 16 PFAS in 58 tap water samples with LC–MS/MS from three different countries (France, Spain, and Brazil with 8, 29, and 21 samples, respectively). The findings revealed that PFBS was the most prevalent compound in Brazilian tap water with a 61.3% occurrence (Schwanz et al. 2016). The findings of a study by Heo et al. (2014) revealed that tap water was the main human exposure route for PFAS, accounting for up to 50% of the PFOA exposure for an adult, but it has to be mentioned that this research was restricted by the tap water samples being gathered from one particular area (Heo et al. 2014). The difference between the PFAS concentrations in drinking water distribution systems in various studies may be due to the many physical, chemical, and even biological processes that happen in the pipelines of a water network (Chen et al. 2019). On the other hand, the differences between the published data may be explained by the different analytical methods (online and off-line methodologies) used to determine of the PFAS concentrations in the drinking water distribution systems (Haug et al. 2010; Llorca et al. 2012). A longer distance from a water treatment plant to a consumer in a water distribution system would increase the contact of the water with the pipe materials, and therefore increase the possibility raising the PFAS level (Park et al. 2018).

Short-chain PFAS (mainly PFBA) levels were nearly stable from water treatment plant to tap waters, but long-chain

Table 6 The range of PFAS concentration in drinking water distribution systems

Total number of samples	Compounds	Concentration (ng/L)	Country (city or area; year of sampling)	Instrumental analysis	Ref	Total number of samples	Compounds	Concentration (ng/L)	Country (city or area; year of sampling)	Instrumental analysis	Ref
77	PFBA[1]	0.25–64.94	China (Eastern and northern China; NR[19])	HPLC–MS/MS[20]	(Chen et al. 2021)	22	PFBA	<LOD* to 25	China (Huantai County, Shandong Province; 2018)	UPLC-MS/MS[21]	(Xie et al. 2021)
	PFPeA[2]	1.04–19.89					PFPeA	ND** to 13.9			
	PFHxA[3]	ND[18] to 34.61					PFHxA	ND to 14.1			
	PFHpA[4]	ND to 10.38					PFHpA	ND to 2.27			
	PFOA[5]	0.07–115.04					PFOA	5.95–19.3			
	PFNA[6]	ND to 6.98					PFDA	ND-4.33			
	PFDA[7]	ND to 2.63					PFHpS	ND-0.31			
	PFUdA[8]	ND to 1.87					PFHxS	ND to 5.1			
	PFDoA[9]	ND to 1.57					PFOS[1]	ND to 1.79			
	PFTrDA[10]	ND to 1.20				12	PFHxA	0.40–2.2	China (Hong Kong; 2017–2018)	HPLC–MS/MS	(Li et al. 2021a)
	PFTeDA[11]	ND to 1.04					PFHpA	1.54–7.59			
	PFHxDA[12]	ND to 0.49					PFOA	5.67–39.7			
	PFBS[13]	ND to 13.90					PFOS	0.11–8.63			
	PFHxS[14]	ND to 139.67					PFNA	0.37–1.02			
	PFOS[15]	0.45–14.62					PFDA	0.17–0.37			
	PFDS[16]	ND to 2.17					PFDDA[17]	0.06–0.85			
14	PFBA	0.20–5.60	China (Hangzhou, Hai'ning, Beijing, Shanghai, Changsh, and Suzhou; 2015)	HPLC–MS	(Lu et al. 2017)	94	PFBA	0.27–1.93	Turkey (33 provinces; 2017–2018)	UHPLC-MS/MS[22]	(Endirlik et al. 2019)
	PFPeA	0.80–7.20					PFPeA	0.08–1.23			
	PFHxA	1–41.20					PFHxA	0.08–2.90			
	PFHpA	1.10–5.20					PFHpA	0.08–1.65			
	PFOA	5.60–115.40					PFOA	0.10–2.37			
	PFNA	0.40–1.80					PFNA	0.07–0.44			
	PFDA	0.30–0.60					PFDA	0.08–0.41			
	PFBS	1.10–29					PFBS	0.11–0.85			
	PFHxS	2.90–6.90					PFHxS	0.10–2.18			
	PFOS	1.60–6.67					PFOS	0.09–2.04			
59	PFBA	0.16–3.58	Canada, USA, Japan, China, France, Norway, Chile, Burkina Faso, Ivory Coast, and Guadeloupe (41 cities; 2015)	UHPLC-MS[23]	(Kaboré et al. 2018)	6	PFBA	8.60–10	The Netherlands (Dordrecht, Alblasserdam, Gouda, Rotterdam, Spijkenisse, and Goedereede; 2016)	HPLC-MS[24]	(Brandsma et al. 2019)
	PFPeA	0.20–2.24					PFPeA	4.90–5.10			
	PFHxA	0.24–3.49					PFHxA	1.60–5.90			
	PFHpA	0.13–0.96					PFHpA	2.10–3.20			
	PFOA	0.22–4.86					PFOA	1.90–7.10			
	PFNA	0.12–0.47					PFNA	<0.50			
	PFDA	0.12–0.98					PFDA	<0.50			
	PFBS	0.12–0.96					PFBS	2.50–11			
	PFHxS	0.12–0.97					PFHxS	0.60–1.30			
	PFOS	0.12–4.99					PFOS	1.10–2.60			

Table 6 (continued)

Total number of samples	Compounds	Concentration (ng/L)	Country (city or area; year of sampling)	Instrumental analysis	Ref	Total number of samples	Compounds	Concentration (ng/L)	Country (city or area; year of sampling)	Instrumental analysis	Ref
44	PFPeA	0.87–17	South Korea (Seoul, Incheon, Suwon, Daejeon, Gwangju, Daegu, Andong, and Busan; 2017)	HPLC–ESI–MS/MS[25]	(Park et al. 2018)	87	PFBA	ND to 56.84	China (eastern China; 2018)	HPLC-LC/MS	(Chen et al. 2019)
	PFHxA	0.56–24.1					PFPeA	ND to 30.80			
	PFHpA	0.38–13.5					PFHxA	0.46–14.93			
	PFOA	0.75–27.70					PFOA	ND to 56.84			
	PFNA	0.22–2.95				2	PFOA	1.88–2.32	China (Henxin Chemical Plant; 2012)	HPLC-ESI-/MS/MS	(Gao et al. 2015)
	PFDA	0.16–0.94					PFOS	2.96–3.42			
	PFBS	0.33–5.73				5	PFOA	4.70–12	Japan (Tokyo; 2007–2009)	LC–MS/MS	(Kuroda et al. 2014)
	PFHxS	0.38–189.60					PFOS	1.70–11			
	PFOS	0.31–1.78									
3	PFBA	0.36–2.92	Sweden (Stockholm, Lovo, Norrvatten, Bollebygd and Umea; 2013)	UPLC-MS/MS	(Filipovic and Berger 2015)	NM	PFBA	0.57–0.82	Island (Kornvatn and Havnadal; 2012)	UPLC-MS[26]	(Eriksson et al. 2013)
	PFPeA	0.85–1.50					PFPeA	< 0.029			
	PFHxA	0.06–1.35					PFHxA	< 0.06–0.08			
	PFHpA	0.04–0.67					PFHpA	0.20–0.22			
	PFOA	0.07–1.50					PFOA	0.23–0.25			
	PFNA	0.02–0.30					PFNA	0.16–0.17			
	PFDA	0.05–0.14					PFDA	0.02–0.03			
	PFBS	0.007–2.31					PFBS	< 0.016			
	PFHxS	0.026–2.15					PFHxS	0.02–0.04			
	PFOS	0.026–4.64					PFOS	0.17–0.61			
2	PFBA	0.54–0.57	Germany (Leipzig; 2015)	UHPLC-MS/MS	(Shafique et al. 2017)	120	PFPeA	0.80–19.80	Netherlands and Greece (34 areas in the Netherlands and 35 cities in Greece; 2013–2014)	LC–MS/MS[27]	(Zafeiraki et al. 2015)
	PFPeA	0.51–0.59					PFHxA	2.40–4.90			
	PFHxA	0.91–0.93					PFHpA	1–3			
	PFHpA	0.16–0.21					PFOA	1.40–11.10			
	PFOA	5.80–6.50					PFNA	< 0.6			
	PFNA	0.30–0.48					PFDA	< 0.6			
	PFDA	0.24–0.26					PFBS	0.70–13.70			
	PFBS	1.23–1.30					PFHxS	0.70–2.30			
	PFHxS	0.04–0.06					PFOS	3–5			

Table 6 (continued)

Total number of samples	Compounds	Concentration (ng/L)	Country (city or area; year of sampling)	Instrumental analysis	Ref	Total number of samples	Compounds	Concentration (ng/L)	Country (city or area; year of sampling)	Instrumental analysis	Ref
58	PFHxA	ND to 58.21	Brazil, France, and Spain (Porto Alegre, Toulouse, Montpelier, Nimes, Avignon, Valence, Grenoble, Lyon, Perpignan, and Barcelona; NR)	LC–MS/MS	(Schwanz et al. 2016)	27	PFHxA	ND to 2.07	USA (New York; 2012)	HPLC–MS/MS	(Subedi et al. 2015)
	PFHpA	1.99–41.58					PFHpA	ND to 1.31			
	PFOA	2.98–45.62					PFNA	ND to 1.22			
	PFNA	< 2.1–46.75					PFDA	ND to 0.49			
	PFDA	< 3.1–24.99					PFBS	ND to 0.48			
	PFBS	0.48–23.82					PFHxS	ND to 0.54			
	PFHxS	< 2.3					PFOS	ND to 1.42			
	PFOS	1.99–140.48									
10	PFPeA	0.03–0.09	Ghana (Central and Western Regions of Ghana; 2015)	HPLC–MS/MS	(Essumang et al. 2017)	89	PFBA	2.40–27	Germany and Spain (32 cities in Germany and Spain; 2010–2012)	LC–MS/MS	(Llorca et al. 2012)
	PFHxA	0.01–0.08					PFPeA	0.76–17			
	PFHpA	< LOD to 0.06					PFHxA	0.23–11			
	PFOA	66–190					PFHpA	0.23–16			
	PFDA	0.05–0.08					PFOA	0.16–35			
	PFOS	16.20–168					PFOS	0.04–1			
NR	PFHxA	1.81–6.57	Japan (Osaka; 2015–2016)	LC–MS[28]	(Shiwaku et al. 2016)	25	PFHpA	0.64–10	USA (Ohio and Northern Kentucky; 2003–2006)	HPLC–MS/MS	(Dasu et al. 2017)
	PFHpA	1.78–9.53					PFOA	2.50–108			
	PFOA	6.17–25.55					PFBS	2.70–11.70			
	PFNA	0.81–7.03					PFHxS	0.77–6			
	PFDA	0.30–2.50					PFOS	1.90–98.60			
34	PFOA	12.90***	South Korea (Busan; 2011–2012)	LC/MS–MS	(Heo et al. 2014)	15	PFOA	0.17–33	South Korea (Seoul, Busan, Daegu, Kwangju, Ansan, Shiheung, Cheongju, and Jeonju; 2009)	HPLC–MS/MS	(Kim et al. 2011)
	PFOS	2.62***					PFOS	0.17–3.60			
3	PFOA	0.65–2.50	Norway (Oslo; 2008–2009)	UPLC-MS/MS	(Haug et al. 2010)	4	PFOA	6.80–40	China (Taihu; 2009)	HPLC-ESI-/MS/MS	(Qiu et al. 2010)
	PFOS	0.07–0.31					PFOS	3.20–14			
21	PFOA	0.1–45.9	China (21 cities; NR)	HPLC-ESI[29]	(Jin et al. 2009)	26	PFOA	0.35–2.82	Brazil (Rio de Janeiro and Sao Paulo; 2008)	HPLC–ESI–MS/MS	(Quinete et al. 2009)
	PFOS	0.1–14.8					PFOS	0.58–6.70			
6	PFOA	1–2.90	Italy (Arolo, Ispra, Angera, Sesto Calende, Arona, Stresa, and Verbania; 2006)	LC–MS-MS	(Loos et al. 2007)	11	PFOA	22–519	Germany (Ruhr area; 2006)	HPLC–MS/MS	(Skutlarek et al. 2006)
	PFOS	6.20–9.70					PFOS	3 – 22			
30	PFOA	0.12–40	Japan (Osaka area and Tohoku district; NR)	LC/MS	(Saito et al. 2004)	62	PFOA	< 0.50–9.66	Australia (34 sampling locations; 2010)	LC–MS-MS	(Thompson et al. 2011)
	PFOS	< LOD to 12					PFOS	< 0.66–15.6			

Table 6 (continued)

Total number of samples	Compounds	Concentration (ng/L)	Country (city or area; year of sampling)	Instrumental analysis	Ref	Total number of samples	Compounds	Concentration (ng/L)	Country (city or area; year of sampling)	Instrumental analysis	Ref
7	PFOA PFOS	0.30–8.56 0.39–8.81	Sweden, Italy, Belgium, the Netherlands, Norway, and Germany (Stockholm Ispra, Antwerp, Belgium, Amsterdam, Tromsø, Schmallenberg; 2010)	HPLC-HRMS[30]	(Ullah et al. 2011)	84	PFOA	2–47	Italy (Milan; 2010–2013)	HPLC–MS/MS	(Castiglioni et al. 2015)

[1]Perfluorobutanoic acid; [2]perfluoropentanoic acid; [3]perfluorohexanoic acid; [4]perfluoroheptanoic acid; [5]perfluorooctanoic acid; [6]perfluorononanoic acid; [7]perfluorodecanoic acid; [8]perfluoroundecanoic acid; [9]perfluorododecanoic acid; [10]perfluorotridecanoic acid; [11]perfluorotetradecanoic acid; [12]perfluorohexadecanoic acid; [13]perfluorobutane sulfonate; [14]perfluorohexane sulfonate; [15]perfluorooctane sulfonate; [16]perfluorodecane sulfonate; [17]perfluorododecanoic acid; [18]not mentioned; [19]not reported; [20]high-performance liquid chromatography–tandem mass spectrometry; [21]ultra-performance liquid chromatography–tandem mass spectrometry; [22]ultra-high-performance liquid chromatography–tandem mass spectrometry; [23]ultra-high-performance liquid chromatography–mass spectrometry; [24]high-performance liquid chromatography–mass spectrometry; [25]high-performance liquid chromatography–electrospray ionization–tandem mass spectrometry; [26]ultra-performance liquid chromatography–mass spectrometry; [27]liquid chromatography–tandem mass spectrometry; [28]liquid chromatography–mass spectrometry; [22]high-performance liquid chromatography–tandem mass spectrometry; [29]High-performance liquid chromatography–electrospray ionization; [30]high-performance liquid chromatography–high-resolution mass spectrometry

*Limit of detection

**Measured but not detected

***Average (based on reported values in the study)

PFAS (mainly PFOA) showed a considerable decrease in level, which can be due to their accumulation by the loose deposits in water networks (Chen et al. 2019). Higher levels of short-chain PFAS may be due to the breakdown of longer chain PFAS precursors and contamination coming from the polymer water pipes (or other PFAS-containing materials) during the distribution process (Li et al. 2022). PFAS have wide-ranging applications in polymer pipes that may be released into tap water via supply networks during transport and storage of water in storage tanks, pipe leaks, and corroded parts (Lee and Schwab 2005; Park et al. 2018). Based on the current knowledge, sources of PFAS in water distribution systems due to pipe types are not known yet. Thus, further studies are required on the PFAS levels in tap water with a focus on pipe types, especially polymer pipes. Also, further works are needed to reveal the hazard of PFAS due to tap drinking water.

Main factors affecting on emerging contaminants' release from pipes used in water distribution systems

Several factors, such as the features of the distribution network, the water quality, and the environmental conditions, can potentially influence the leaching of contaminants from pipes used in water distribution system into drinking water (Makris et al. 2014). Different studies have reported various parameters affecting the release of contaminants from pipes into drinking water. For example, the findings of a study in Sweden showed a direct relationship between the age of polymer pipes and abundance of MPs (Kirstein et al. 2021). The chemical composition (such as metal ions) and hardness of drinking water may influence the MP release from plastic materials (Shi et al. 2022). Paint peeling and aging in cast iron pipes can cause the release of epoxy resin. The aging of plastic pipes and fittings may lead to the appearance of PE, PA, and PP (Mintenig et al. 2019). The pH and surface structure of the pipe scales play an important role on the distribution of MPs in tap water (Chu et al. 2022). In the case of BPA, a study in Malaysia reported the impact of temperature on the release of this contaminant in tap drinking water. The concentration of this contaminant was considerably greater in dry and warm months compared to rainy months (Santhi et al. 2012). The results of another study in Taiwan indicated higher release of NP and BPA into drinking water distribution systems with increased contact time with the pipes, especially in the case of polymer pipes, and with ambient temperature (Cheng et al. 2016). According to the findings of two studies, phthalate concentrations were higher in summer than in winter, as the higher temperature led to an increase of phthalate migration into the drinking water (Abdolahnejad et al. 2019; Rudel and Perovich 2009). In a study in China, pH and ionic strength were found to have only a minor effect on phthalate release from PVC material (Yan et al. 2021). Also, in another study, increasing the contact time of water with polymer pipes increased the phthalate concentrations in tap drinking water (Abtahi et al. 2019). Based on the result of a study in China, plastic features such as plasticizer content, particle size, and aging of plastics had a big impact on the leaching of DBP (Yan et al. 2021). The dynamic behavior of water is a factor that may cause the release of phthalates from pipes and reservoirs (Casajuana and Lacorte 2003).

In the case of PFAs, it was reported that water quality factors (such as dissolved organic carbon) and pipeline distribution process factors (such as the transfer distance of the water, the presence of loose deposits in the pipes in the distribution system, and hydraulic disturbances due to the presence of a pressure booster) may have an influence on the fate and the migration of PFAS in water distribution systems (Chen et al. 2019). Therefore, there is a high need for further studies on the ECs levels in tap drinking water which consider the features of the distribution network, the water quality, and the environmental conditions during the transfer and storage of the drinking water, such as contact time with pipe materials, transfer distance of water in the distribution network, temperature and season, the practical lifetime of pipes, and the water quality parameters.

Risk assessment

Drinking water is an essential commodity for human beings that has to be protected from contamination to avoid it becoming a relevant source of contaminant uptake (Zhang et al. 2019). Human exposure to ECs such as MPs, BPA, phthalate, NP, and PFAS may cause adverse health effects. In recent years, some health impacts have become known of MPs (such as oxidative stress, cytotoxicity, neurotoxicity, reproductive toxicity, and disruption of immune function) (Prata et al. 2020; Rahman et al. 2020), BPA (such as breast cancer, infertility, cognitive dysfunction, and cardiovascular diseases (Catenza et al. 2020; Nascimento and Rocha 2018), phthalate (such as diabetes, obesity, insulin resistance, renal effects) (Net et al. 2015; Radke et al. 2019), NP (such as fecundity reduction, mutations, gonadal development inhibition, and fertility reduction) (Liu et al. 2020; Vargas-Berrones et al. 2020), and of PFAS (such as cancer, immune system dysfunction, liver damage, developmental and reproductive harm, and hormone disruption) (Ojo et al. 2020; Pelch et al. 2019).

The determined CDI, HI, and CR values for adults and children according to the maximum concentration of MPs, BPA, phthalate, NP, and PFAS in tap drinking water are presented in Table 7. Based on the data in Table 7, some

contaminants in the tap drinking water of some countries raise a potential health risk for humans as the HI values determined for them were above 1. PFAS, including PFOA in Germany, PFNA in France, and PFHxS in South Korea for adult and children as well as PFOS in Ghana for children, had values > 1 for non-carcinogenic health effects. Also, the CR values determined for DEHP and BBP showed a carcinogenic health risk for adults and children. So, as presented here, the output of drinking water from pipelines can be an important pathway for exposure to emerging contaminants. Besides tap drinking water, there are other exposure routes for emerging contaminants, such as beverages, food, and inhalation (Colin et al. 2014; Kosuth et al. 2018; Schwanz et al. 2016; Sodré et al. 2010), and these also need to be considered in order to have a better understanding of the risk of these contaminants for human health.

Challenges and recommendations

A drinking water distribution system includes diverse components such as pipes, valves, and water reservoir tanks (Abdolahnejad et al. 2019). During the distribution of drinking water through a pipeline, the quality of water may be affected by different processes, such as the leaching of chemicals from the pipes (Liu et al. 2017). Based on the results of some studies, contaminants can leach from pipes into drinking water supply network and cause adverse human health effects (Abdolahnejad et al. 2019; Chen et al. 2019; Weber et al. 2021; Whelton and Nguyen 2013). Generally, the release of contaminants, especially ECs, from pipelines into drinking tap water, is an important global concern. Some of the recommendations for the effective mitigation of the release of ECs from pipes into drinking water are listed as follow:

- ECs are rarely monitored in a worldwide scale (Yadav et al. 2021), especially in tap water, so more studies on these contaminants and their potential ecological and human health effects are needed.
- The features of pipes such as material pipes, aging pipes, and loose deposits are important parameters that can influence the quality of water due to release of contaminants (Liu et al. 2017). For example, in some studies the release of MPs (Chu et al. 2022), BPA (Cheng et al. 2016), phthalates (Jin et al. 2009), NP (Cheng et al. 2016), and PFAS (Chen et al. 2019) from pipes, especially from polymer pipes, are reported. Therefore, the choice of a suitable type of water pipe that will not pose undesirable environmental or human health consequences under all circumstances is of the utmost importance in public health and safety. Also, regular pipe cleaning of drinking water distribution systems can be helpful to minimize ECs level in tap water.
- The released contaminants into water distribution network, especially ECs, may be in low concentrations or under the detection limits due to dilution with a large volume of water (Liu et al. 2017). Measurements at multiple locations in the distribution system, including at the beginning of a network (after the treatment plant) and throughout the network, can provide more accurate data for comparison.
- Reduction of the usage of ECs and the introduction of statutory/regulatory limitations to the use of ECs are required (Kumar et al. 2021b). Also, rigorous control is needed over the various substances and processes that are linked to the diverse components of drinking water distribution system (during manufacturing and with evaluation before use).
- Therefore, a greater focus on developing strategies is required to reduce or/and prevent the migration of ECs migration from pipes into drinking water distribution systems and consequently of their potential adverse health effects.

It should be noted that the detection of ECs in the environment can be a challenge due to their trace concentrations. This problem may be solved by the development of analytical methods that are highly sensitive and selective (Gogoi et al. 2018). Thus, the application of an instrument that is highly efficient for the analysis of ECs would be very useful for the detection of these contaminants.

Research directions

The type of pipe used in water distribution may have a marked influence on the release of ECs into tap water. These contaminants can effect on human health. Research on ECs in tap drinking water has been done in some parts of the world, with the amount being done being greater in some countries than in others. The status of research on ECs including MPs, BPA, phthalates, NP, and PFAS migration from pipes into drinking water in the worldwide has been shown in Fig. 3. These contaminants are categorized in three groups including a high number of studies (> 10), medium research (3–10) and low research (< 3) (Ouda et al. 2021) that have been highlighted in blue, green, and red, respectively. Also, high-risk countries for PAFS (calculated in Table 6) are shown in purple.

While the studies in China on ECs in tap drinking water had the highest number in world, low number of researches were done in some countries (Fig. 3). Based on Fig. 3, most studies on ECs in tap water in worldwide have focused on phthalate and PFAS, and only a low

Table 7 Estimated CDI, HI, and CR of ECs (maximum concentration of each contaminant) through the consumption of tap drinking water

Contaminant	TDI[1] (µg/kg-bw/day)/ CSF (µg/kg-bw/day) [Ref]	Concentration (unit)Country [Ref]	CDI[2] (unit)		HI[3]/CR[4]	
			Adult	Children	Adult	Children
Non-carcinogenic effects						
MPs	-	1409 (items/L) Czech Republic (Pivokonsky et al. 2018)	≈811.76 (items /kg-bw/day)	≈112.72 (items /kg-bw/day)	-	-
BPA	50 (USEPA 2011)	160 (ng/L) Italy (Cantoni et al. 2021)	≈92.11 (ng/kg-bw/day)	≈92.11 (ng/kg-bw/day)	0.001	0.0002
Phthalates						
BBP	200 (USEPA 2011)	0.54 (µg/L) China (Li et al. 2021b)	≈0.30 (µg/kg-bw/day)	≈0.04 (µg/kg-bw/day)	0.001	0.0002
DBP	100 (USEPA 2011)	1.06 (µg/L) South Africa (Van Zijl et al. 2017)	≈0.61 (µg/kg-bw/day)	≈0.08 (µg/kg-bw/day)	0.006	0.0008
DEHP	20 (USEPA 2011)	12.48 (µg/L) China (Tang et al. 2012)	≈7.19 (µg/kg-bw/day)	≈0.99 (µg/kg-bw/day)	0.35	0.04
DEP	800 (USEPA 2011)	44 (µg/L) China (Tang et al. 2012)	≈25.34 (µg/kg-bw/day)	≈3.52 (µg/kg-bw/day)	0.03	0.004
DiDP	150 (SCHER 2008)	0.09 (µg/L) China (Shi et al. 2012)	≈0.05 (µg/kg-bw/day)	≈0.007 (µg/kg-bw/day)	0.0003	0.00004
DMP	100 (USEPA 2011)	0.86 (µg/L) China (Li et al. 2021b)	≈0.49 (µg/kg-bw/day)	≈0.06 (µg/kg-bw/day)	0.004	0.006
DnOP	10 (USEPA 2011)	0.21 (µg/L) China (Li et al. 2021b)	≈0.12 (µg/kg-bw/day)	≈0.01 (µg/kg-bw/day)	0.12	0.016
NP	5 (USEPA 2011)	5.43 (µg/L) China (Jie et al. 2017)	≈3.12 (µg/kg-bw/day)	≈0.43 (µg/kg-bw/day)	0.62	0.08
PFAS						
PFOA	0.02 (USEPA 2016b)	519 (ng/L) Germany (Skutlarek et al. 2006)	≈299.01 (ng/kg-bw/day)	≈41.52 (ng/kg-bw/day)	14.95	2.07
PFNA	0.0003 (Reade et al. 2019)	46.75 (ng/L) France (Schwanz et al. 2016)	≈26.93 (ng/kg-bw/day)	≈3.74 (ng/kg-bw/day)	89.77	12.46
PFBS	10 (USEPA 2018)	29 (ng/L) China (Lu et al. 2017)	≈16.70 (ng/kg-bw/day)	≈2.32 (ng/kg-bw/day)	0.001	0.0002
PFHxS	0.002 (Reade et al. 2019)	189.6 (ng/L) South Korea (Park et al. 2018)	≈109.40 (ng/kg-bw/day)	≈14.92 (ng/kg-bw/day)	54.70	7.46
PFOS	0.02 (USEPA 2016a)	168 (ng/L) Ghana (Essumang et al. 2017)	≈96.78 (ng/kg-bw/day)	≈13.44 (ng/kg-bw/day)	4.83	0.67
Carcinogenic effects						
DEHP	0.014 (USEPA 2011)	0.54 (µg/L) China (Li et al. 2021b)	≈7.19 (µg/kg-bw/day)	≈0.99 (µg/kg-bw/day)	0.10	0.01
BBP	0.0019 (USEPA 2011)	12.48 (µg/L) China (Tong et al. 2020)	≈0.30 (µg/kg-bw/day)	≈0.04 (µg/kg-bw/day)	0.0006	0.0008

[1]Tolerable daily intake; [2]chronic daily intake; [3]hazard index; [4]cancer risk

level of research attention has been given to other ECs including MPs, BPA, and NP. Also, research on ECs in Asia, Africa, and America is less widespread compared to Europe. It should be noted some countries having no or only a small number of studies on ECs may be due to a lack of advanced analytical facilities and qualified researchers (Ouda et al. 2021). According to Fig. 3 and Table 6, PFAS in South Korea (PFNA), France, the USA, and Ghana (PFHxS) were the contaminants with health risk for humans. Thus, more research on these contaminants in tap drinking water with a particular focus on pipe type are needed in these countries, as well as in other

countries. While ECs including MPs, BPA, phthalates, NP, and PFAS has been considered in various aquatic environments such as sea water, surface water, and bottled water (Akhbarizadeh et al. 2020b; Bhandari et al. 2021; Courtene-Jones et al. 2017; Egessa et al. 2020; Gao et al. 2019; Groffen et al. 2021; Lan et al. 2019; Ouda et al. 2021; Ozhan and Kocaman 2019; Zhang et al. 2020b), the migration of these contaminants from pipes into drinking water distribution systems and their health risk has not been fully investigated. Therefore, more studies are needed to measure these contaminants, their occurrence and their quantities in actual water supply systems, to understand better the factors that promote leaching and their interaction, and to improve knowledge on the processes that control the release of contaminants from pipelines into water. In future studies on the levels of ECs in tap water in tap water, the effects should be considered of the analytical techniques used, the method of sample collection (with or without previous flushing), and the sample pretreatment. A critical approach is needed to expanding to the fact that these contaminants that are included are just a small fraction of what is actually there in tap water. Also, more studies need to obtain data from other surface water bodies and expand them in the context of drinking water distribution systems.

Conclusions

Although contaminants can be removed by various water treatment processes, they can also migrate from pipes into drinking water. Several contaminants including MPs, BPA, phthalates, NP, and PFAS in drinking water distribution networks may stem from migration from pipes or reservoirs. This review showed that the pipes type, especially polymer pipes had an important role on ECs release from pipes into tap water during transport and storage. The risk assessment of studied ECs also showed that PFAS (including PFOA, PFNA, and PFHxS) and phthalates (including DEHP and BBP) in tap water had non-carcinogenic and carcinogenic effects for consumers in some countries, respectively. Therefore, more research is required to indicate trace levels of the various types of ECs that migrate from pipes into drinking water distribution networks. According to the findings obtained in this review, the pipes have an irrefutable role in the release of contaminants into the drinking water. Furthermore, the knowledge about the migration of ECs from pipes into drinking water distribution systems is not yet complete. Overall, this review highlights the significant need for further work on the migration of ECs from pipes into drinking water distribution networks the in the world.

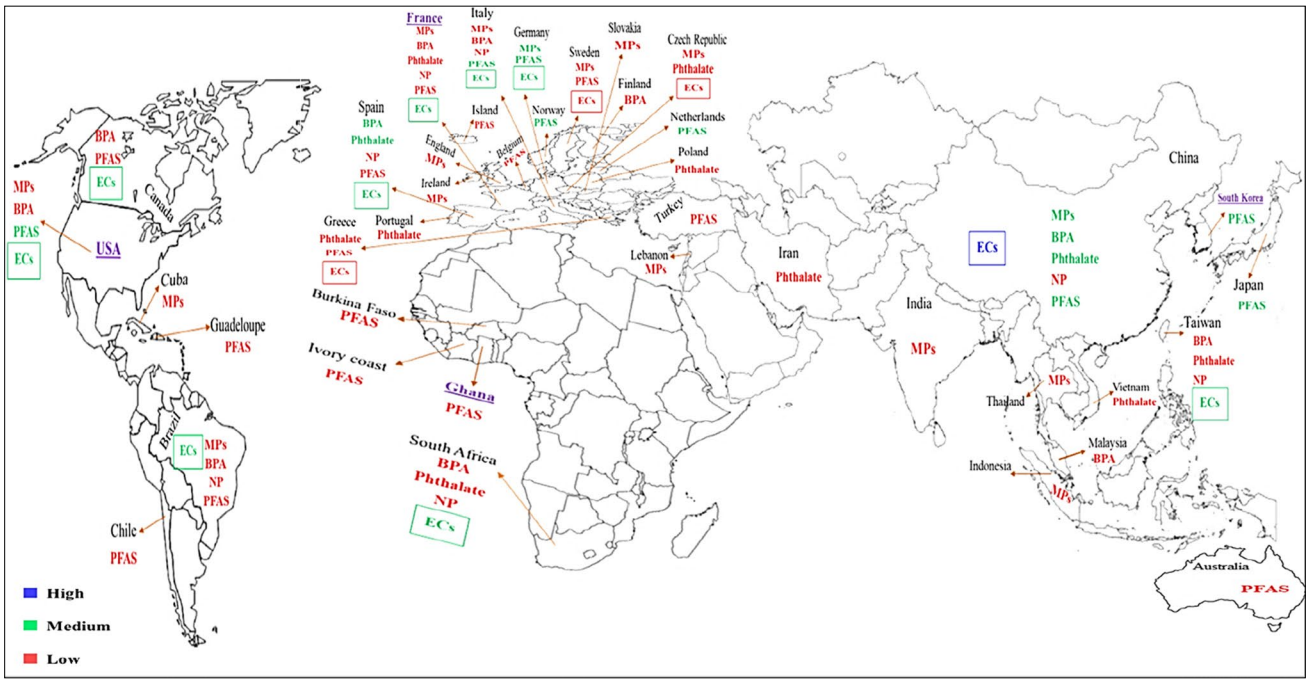

Fig. 3 Numbers of studies (published in English) on ECs including microplastics (MPs), bisphenol A (BPA), phthalates, nonylphenol (NP), perfluoroalkyl, and polyfluoroalkyl substances (PFAS) and sum numbers of studies on these contaminants (displayed with ECs) migration from pipes into water distribution systems in the world [high: > 10 (blue); medium: 3–10 (green); low: < 3 (red)] and as well as high-risk countries for PFAS (purple)

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Declarations

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