Science of the Total Environment 872 (2023) 162121

Contents lists available at ScienceDirect

Science of the Total Environment

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

Tracking microplastics contamination in drinking water in Zahedan, Iran: From source to consumption taps

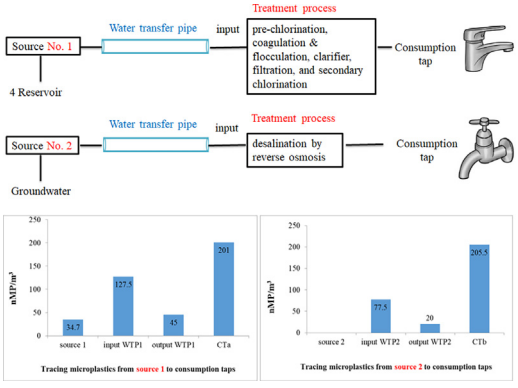
Hassan Taghipour [a], Mehdi Ghayebzadeh [b,c,*], Fatemeh Ganji [c], Saeid Mousavi [d], Nahid Azizi [e]

[a] Health and Environment Research Center, School of Public Health, Tabriz University of Medical Sciences, Tabriz, Iran
[b] Department of Environmental Health Engineering, Health Promotion Research Center, Zahedan University of Medical Sciences, Zahedan, Iran
[c] Department of Environmental Health Engineering, Zahedan University of Medical Sciences, Zahedan, Iran
[d] Department of Statistics and Epidemiology, Tabriz University of Medical Sciences, Tabriz, Iran
[e] Department of Environmental Health Engineering, School of Public Health, Tehran University of Medical Sciences, Tehran, Iran

HIGHLIGHTS

- There are MPs in all the samples (from the source to the consumption taps).
- The number of MPs was larger in spring than in winter.
- About 64 % and 75 % of particles were eliminated in WTP1 and WTP2, respectively.
- The average number of MPs in consumption taps (due to secondary contamination) was larger than in treatment water.
- According to the EDI results, children are more at risk of MPs.

GRAPHICAL ABSTRACT



ARTICLE INFO

Editor: Yolanda Picó

Keywords:
Microplastic
Drinking water
Spectroscopy
Contamination
Fate

ABSTRACT

Microplastics (MPs) that pollute drinking water are inherently toxic, act as an adsorbent of hazardous pollutants, and threaten human health. So, the fate of microplastics in drinking water from the source to consumption taps (CTs) was assessed in spring and winter in Zahedan city in Iran. Sampling was performed from 4 reservoirs (raw water), before and after two water treatment plants (WTPs), and 10 CTs. The reservoirs were sampled using a plankton net (pore size = 100 μm), and the remaining samples were taken using a sampling device (containing a stainless steel membrane as a filter with pore size = 5 μm). The combination of density separation techniques, digestion, observation, Micro-Raman and FTIR, and SEM analysis was performed to recognize MPs. The average number of MPs in raw water varied between 15.4 and 44.7 MP/m^3 (winter) and 22–51.8 MP/m^3 (spring). The results before and after the treatment plant showed that about 64 % and 75 % of particles were eliminated in WTP_1 and WTP_2, respectively. The average number of MPs in CTs was more than treatment water (CT_a = 85–390 MP/m^3 and CT_b = 75–400 MP/m^3), which is a probable confirmation of secondary contamination (abrasion from pipes, installations, and sealing materials). The dominant type of polymer detected in raw water, treated water, and consumption taps were PS. The estimated daily intake for children and adults was about 0.16–15 MP/kg/bw/year and 0.07–5.7 MP/kg/bw/year, respectively. The surface morphology of MPs showed that the particles were affected by continuous weathering, mechanical breakage, and oxidation. MPs threaten the environment and human health due to the adsorption and transport of hazardous pollution and their intrinsic toxicity, so a solution must be thought of to prevent the pollution of drinking water by MPs.

* Corresponding author.
E-mail address: m.ghayebzadeh@zaums.ac.ir (M. Ghayebzadeh).

http://dx.doi.org/10.1016/j.scitotenv.2023.162121
Received 4 October 2022; Received in revised form 4 February 2023; Accepted 5 February 2023
Available online 9 February 2023

1. Introduction

Currently, humans live with all kinds of plastic substances in their life (Kirstein et al., 2021). Since a few decades ago, global plastic generation has increased noticeably due to its potential for diverse applications (Thompson et al., 2009). High production levels, lightweight nature, versatility, durability, low price, and low recycling rate of plastics have led to the accumulation of plastic debris in all ecosystems and their pollution (Barnes et al., 2009; Ghayebzadeh et al., 2023; Lee et al., 2015). Currently, global plastic production is >400 million metric tons annually (Fahim et al., 2021). Plastics are difficult to degradable due to their durability, but over time become fragile and turn into ever-smaller fragments (macro and meso) and particles (micro and nano) (Al-Oufi et al., 2004). Microplastics (MPs) are particles from 1 μm to 5 mm. Plastic particles <1 μm are commonly referred to as nano-plastics (NPs) (Koelmans et al., 2019). Different studies have reported the existence of plastic particles and fragments in various environmental matrices (Koelmans et al., 2022), freshwater (Weber et al., 2021; Wong et al., 2020), seawater (Lorenz et al., 2019; Schönlau et al., 2020), sediments (Ghayebzadeh et al., 2020; Leslie et al., 2017; Xu et al., 2020), and air (Dris et al., 2017; Prata et al., 2020; Zhang et al., 2020). Fine plastic particles gradually enter the food chain. MPs also have been reported in bottled and drinking water (Koelmans et al., 2019; Oßmann et al., 2018; Zuccarello et al., 2019), shellfish and shrimp (Devriese et al., 2015; Lia et al., 2018), fish (Bessa et al., 2018; Jaafar et al., 2021) and so on. The entry of plastic particles into aquatic environments is through wastewater discharge (Mason et al., 2016; Mintenig et al., 2017), unsuitable waste management, various industrial activities, including plastic and textile manufacturing plants (da Costa et al., 2016), as well as the destruction and decomposition of plastic litter, and more (da Costa et al., 2016). Not all MPs are removed in wastewater treatment process plants (WWTPs) (Leslie et al., 2017; Wang et al., 2020), and considerable amounts of MPs are being discharged from WWTPs into raw water sources (Lee and Chae, 2021; Mohana et al., 2021). MPs particles may be re-fragmented (due to weathering) and become finer particles and, for some reason, pass through the drinking water treatment system and be consumed by humans (Lee and Chae, 2021; Mohana et al., 2021). The small size of these particles makes plastics easily pass through the inhalation/gastrointestinal system and transfer to other parts of the body (Pirsaheb et al., 2020; Senathirajah et al., 2021; Sharma and Chatterjee, 2017). A recent study has confirmed the presence of plastic particles in human blood (Leslie et al., 2022). Researchers have come to believe that the identification of fine particles (<500 μm) and very fine particles (<20 μm) MPs are very important and essential. Because it has been reported that MPs smaller than 500 μm can possibly pass through the intestinal wall (Lusher et al., 2020), and MPs smaller than 20 μm can accumulate in the liver, kidneys, and intestines of mice (Deng et al., 2017). In 2019, the World Health Organization (WHO) demanded to further research to assess the risks of the presence of MPs and to investigate possible damaging effects on human health. Hence, the investigation of MPs contamination in bottled water and drinking water was suggested by the WHO (Marsden et al., 2019). For this purpose, various studies have been carried out in different world areas. In South Korea (2022), MPs were traced in raw water (2.2 ± 1.3 MP/L) and also in treated drinking water (0.02 ± 0.02 MP/L) (Jung et al., 2022). In the Czech Republic (2018), the results of the occurrence of MPs in raw and treated drinking water showed that MPs were found in all water samples and their average abundance ranged from 1473 ± 34 to 3605 ± 497 MP/L in raw water and from 338 ± 76 to 628 ± 28 MP/L in treated water, depending on the WTP (Pivokonsky et al., 2018). In Germany (2021), the contamination of consumption taps water with MPs was not detected (Weber et al., 2021). Two studies were carried out in Mexico City on drinking water from outdoor refill kiosks (74.18 ± 48.76 MP/L) and urban decentralized pay-to-fetch drinking-water refill kiosks (11 to 860 MP/L), and the results showed that they were contaminated with MPs (Pérez-Guevara et al., 2022a; Shruti et al., 2022). Various studies have investigated MPs contamination in bottled water. According to the results, bottled water in Germany (14 MP/L) (Schymanski et al., 2018), the USA (3.57 MP/L) (Kosuth et al., 2018), and Iran (8.5 ± 10.2 MP/L) (Makhdoumi et al., 2021), were contaminated with MPs. It is necessary to explain that the sizes of the particles detected in these studies differed. Despite the studies conducted and the consequences of MPs for the environment and alive creatures' health, so far, limited research has been focused on MPs contamination in drinking water, especially in consumption taps. In Iran, no research has been carried out in this field, and globally, a handful of research has been carried out about the fate of MPs from source (raw water) to consumption tap (Dalmau-Soler et al., 2021; Dalmau-Soler et al., 2022). Given that the sources of drinking water supply are different, and the identification of MPs in consumption taps is also related to its source, in this study, the identification and fate of MPs from source (raw water) to consumption taps carried out in the Zahedan city located in southwest Iran.

2. Material and methods

2.1. study area

The study area was Zahedan (the capital city of Sistan and Baluchestan province), which has an area of about 80 square km^2 and a population of nearly 800 thousand people. About 60 %–70 % of this city's water is supplied from stored water in the 4 natural reservoirs located north of the city (as water source 1). The current capacity of the three main reservoirs is equal to 660 million cubic meters, and its useful capacity is approximately 340 million cubic meters. A storage reservoir (capacity of 800 million cubic meters) is for agriculture, but due to drought and water scarcity, this reservoir is currently also used for drinking water. These reservoirs are filled with water from the Helmand River (a river that originates in Afghanistan) in the spring. About 30 %–40 % of this city's water is supplied by desalination of brackish groundwater (as water source 2). There is a treatment plant for reservoir water and another for the desalination of brackish groundwater. The study area and location of source 1 (4 reservoirs) and field images of some plastic-polluted sites are presented in Fig. 1.

2.2. Sampling

Our goal in this study was to trace the fate of MPs from the source to consumption taps of drinking water in order to cover variability in MPs' abundance. For this purpose, sampling was performed in two seasons (spring and winter) from water supply sources (source 1 = 4 reservoirs), water transfer pipes before the treatment plants, water transfer pipes after the treatment plants, and consumption taps (including consumption taps in a university, an apartment, a villa house, a residential complex, and a health center). It is necessary to mention that sampling of source 2 (groundwater) was not accomplished due to some technical limitations. Additional information about more sampling details, the length of water transfer pipes, the length of the water distribution network, pipe material type, plastic storage tanks, and the water treatment system are given in the supplementary (Text S1, Table S1, and Fig. S1).

First, the sampling was carried out from sources 1 (4 reservoirs). Samples were collected in the winter and spring. These timings were chosen because of the low and high water levels in the reservoirs. In winter, water inflow to the reservoirs is low or zero. In spring, excess water from the Helmand River enters the reservoirs due to the melting snow in Afghanistan. For the sampling of water, a plankton net with a 100-μm pore size (a circle with a diameter of 0.25-m and a 1-m-long collecting bag) was used (Fig. S1, A). The plankton net was towed at the upper surface of water (at a depth of 0 to 20 cm from the water surface), for roughly 15 min in a boat with a speed of 1–1.5 $m.s^{-1}$. The volume of water filtered by the plankton net was between 44 and 66 cubic meters (an average of 55 m^3). The trapped material in the net was rinsed into a sterilized 1-L laboratory glass jar, carried in the cool box, and then stored at 4 °C before processing in the lab.

Sampling from other stations was performed using a sampling apparatus (made by the research team). The sampling apparatus (Fig. S1, B) includes a steel housing holding the filter, filter membrane (material = stainless steel, diameter = 100 mm and pore size = 5 μm), stainless steel and brass fittings and valves, a silicone hose, a flowmeter connected to the apparatus (Meister-Flow untitled model DGU-14, UK). The inlet of the silicone hose

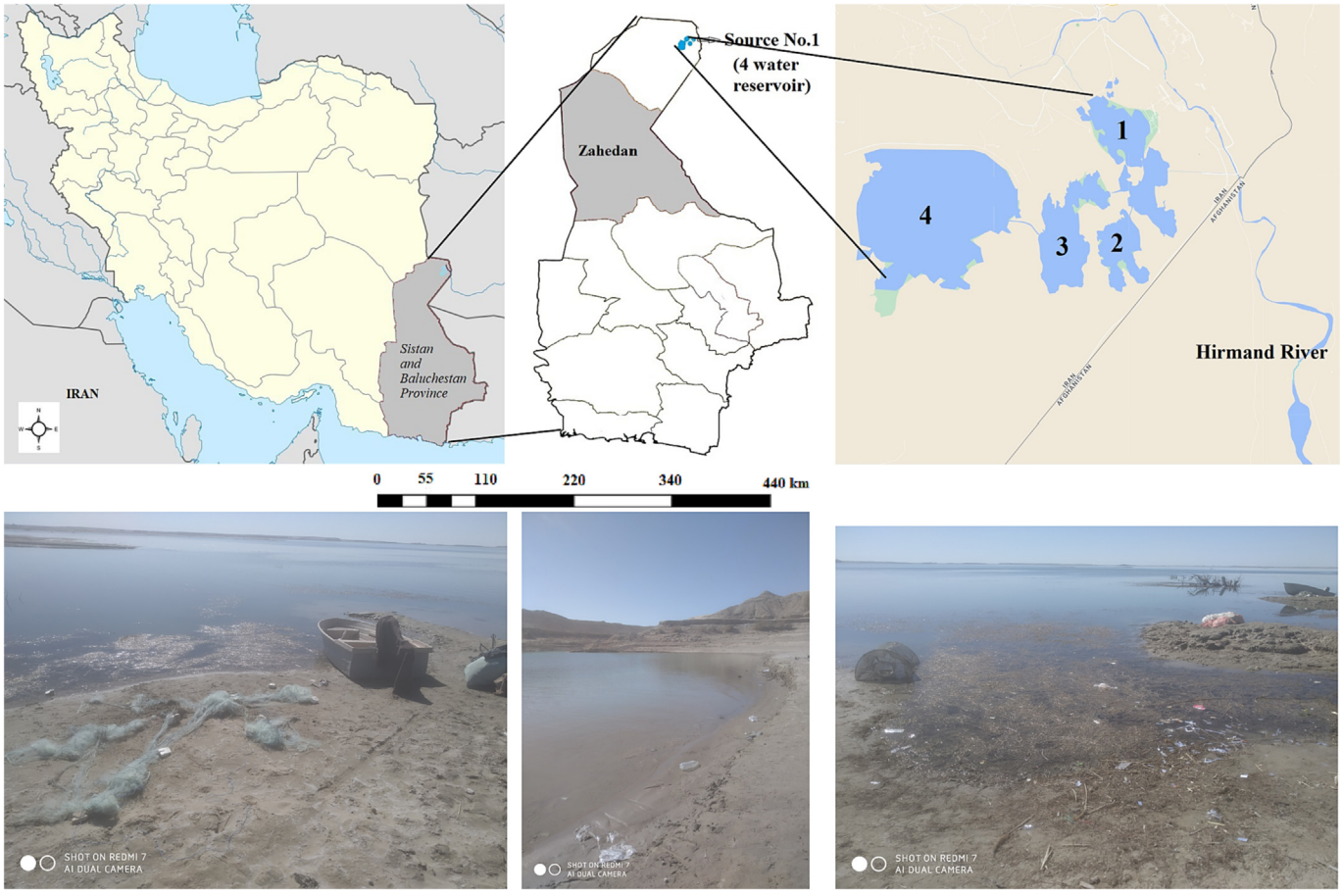


Fig. 1. The geographical position of source No.1 (4 reservoirs) along with field images of some plastic-polluted sites.

was connected to a water tap at the hydrant or to the pipe, and its outlet was connected to the flowmeter, and the outlet of the flowmeter was attached to the inlet of the sampling apparatus. These fittings were implemented using some pipe clamps and adapters. The fittings were completely caulked with special hemp. After the connection was established, first, the hose, flowmeter, and apparatus were passed for 10 min with maximum sampling water flow rate (bypass and outlet were open and there was no filter in the apparatus). After the initial wash, the filter (filter washed with ultra-pure water) was carefully placed in the filter holder housing. The outlet valve closed. After the connection was re-established, the tap was washed through a bypass for 15 min. Then the bypass valve was closed and using a flowmeter, the water flow was adjusted to roughly 10 L min^{-1}, and sampling was taken for 10 min. The volume of the water sample passed through the filter was equal to 100 L. Next, the filters were placed in covered glass Petri dishes containing 70 % ethanol and stored at 4 °C prior to processing. It is necessary to explain that the sampling apparatus was opened under the laboratory hood, and the stainless-steel filter was carefully taken out with tweezers and transferred to muffled glass petri dishes. All sampling was repeated twice.

2.3. Sample processing

Primary separation and extraction of plankton net MPs before processing: For the initial separation of the contents of the plankton net, first, the net was washed several times with high-pressure water (both outside and inside the net). Gradually, the contents were directed to the end of the net (net bucket), and the contents were collected in the net bucket. Then the bucket opened, and its contents were washed into a beaker.

The samples may contain inorganic precipitates and biogenic organic matter. These materials can make it difficult to identify MPs. Therefore, the method of density separation and simple digestion is required to decrease this problem. The extraction of MPs (separation based on density, in order to separate MPs from inorganic precipitates such as sand) was performed by $ZnCl_2$ (at a density of 1.8 g mL^{-1}). $ZnCl_2$ solution and the contents inside it (microplastics and other impurities) were stirred for 5 min by a mixer at 250 rpm. After about 0.5 h of the settlement, the supernatant containing plastic particles was carefully transferred to another breaker. Next, the digestion method was carried out to reduce biogenic organic matter using H_2O_2 30 %. Based on the available impurities, the amount of H_2O_2 30 % was variable from 20 to 30 mL, digestion time from 24 to 48 h, and digestion temperature from 50 to 60 °C. After digestion, the samples were filtrated through cellulose nitrate membrane filters (pore size = 0.45 μm, diameter = 47 mm, Company Sartorius, Germany), by vacuum filtration. The MPs were cleaned with deionized water before instrumental and optical analysis (Jung et al., 2022). Finally, the particles were classified according to size, shape, and color. Visual analysis of particles (determination of size, shape, and color) was performed using an Olympus microscope (model BX41 and BX42, Olympus Company, made in Japan) equipped with a controller and a DP12 camera.

Primary separation and extraction of filter contents (stainless steel filter, pore size 5 μm): First, the sampling device was opened under the hood, and the stainless steel filter was removed with tweezers. The filter and the inner surface of the filter housing were rinsed into a beaker. The filter was washed with a large volume of distilled water to rinse all the contents of the filter. Then contents of the beaker were digested by H_2O_2 (30 %) for 24 h at 50 °C. The analysis continuation was performed according to the previous step.

2.4. Fourier-transform infrared spectroscopy (FT-IR), Raman micro-spectroscopy, and field emission scanning electron microscope (FE-SEM)

Fourier-transform infrared spectroscopy (FT-IR model TENSOR 27 made by the BRUKER, Germany) and Raman micro-spectroscopy (XploRA™ PLUS-

HORIBA, Japan) were implemented to determine polymer structure. For analysis of FT-IR, samples were sent to the Department of Chemistry of the University of Sistan & Baluchestan of Iran. Among MPs extracted, 126 MPs (with size >100 μm) were randomly selected as composite samples and were analyzed. The FTIR spectra used for particle identification were in the wavelength range of 500 to 4000 $cm^{-1,}$ and the maximum resolution was 4 cm^{-1}.

Micro Raman was used to identify particles smaller than 100 μm. For analysis of Micro-Raman, samples were sent to the Institute for Color Science & Technology of Tehran, Iran. Among MPs extracted, 60 MPs (with size <100 μm) were randomly selected as composite samples and were analyzed. The micro Raman device in this study had a confocal microscope for microscopic point-by-point analysis and examination of the sample surface. This device had fast imaging with the SWIFT technique (typically 10× faster than conventional Raman imaging), multiple laser wavelengths, EMCCD detection, Raman polarization, and even Raman-AFM combination, with high sensitivity and power. The identified micro Raman spectra were accredited using SLoPP and SLoPP-E Raman spectral libraries for MPs research. Raman micro-spectroscopy had a multi-laser option (532, 638, 785 nm), confocal imaging 0.5 μm XY, and resolution standard >1.4 cm^{-1} to identify MPs.

Field emission scanning electron microscopy (FE-SEM) (model SU5000, made by Hitachi, Japan) was applied to characterize the MP's surface morphology. For FE-SEM analysis, four cellulose nitrate membrane filters (filtered samples of the reservoirs, before and after water treatment plants, and consumption taps) were randomly selected and sent to the Institute for Color Science & Technology of Tehran, Iran. Parts of the filters (in sizes of mm^2 and in the checkered pattern method) were cut and analyzed by the device. MPs on the surface of the cut parts of the filters (about 15 MPs) were randomly selected as composite samples and analyzed. FE-SEM is a type of SEM that has a field emission electron gun. The field emission electron gun does not require thermal energy to overcome surface potential, and by applying a very high electric field to the surface of the sample, it separates the electrons from the surface. The main specifications of the device are included a resolution-in-column top detector (2.0 nm at 1 kV), magnification [10–600,000× (based on 4″ × 5″picture), 18–1,000,000× (800 × 600 pixels on display) / 30–1,500,000× (1280 × 960 pixels on display)], and sensitivity-ultra-efficient photodiode BSE detector, unmatched low kV imaging to 100 V, and high probe current (>200 nA).

2.5. Estimated daily intake (EDI)

The estimated daily intake (EDI) of MP via drinking water was calculated by using Eq.1. The ingestion rate (IR) was determined according to the general consumption of water in Iran (2–2.5 L/day for adults and 1–1.5 L/day for children). C is the concentration of MPs (n/L), and the BW is the body weight assumed to be 70 kg for adults and 16 kg for children (Arena et al., 2015; Zuccarello et al., 2019). It is necessary to explain, according to the report of Zahedan Water and Wastewater Company (Sistan and Baluchestan Water and Wastewater Company, 2022), as well as the field survey done and the economic status of the studied area, most people (>95 %) are drinking from taps.

$$\mathrm{EDI\ (MP/kg/day)} = (\mathrm{C} \times \mathrm{IR})/\mathrm{BW} \quad (1)$$

2.6. Quality control and quality assurance

A blank test was carried out to avoid and assess the contamination of lab tools and equipment due to the commute, chemicals, air, and tester during the test process. For this purpose, ultrapure distilled water samples (7 samples) were poured into beakers and stored for 1 to 7 days in the testing space and also in the sample storage area. Cellulose nitrate filters (7 filters on glass plates) were used as a negative control (blank control), which were placed adjoining the microscope, the hood, and the surrounding environment of the experiment (Baldwin et al., 2016).

To avoid errors and pollution during the preparation, testing, and analysis of samples, doors and windows were closed, and air conditioning and fans were turned off during the testing process. Commute restrictions to the lab were applied at the time and space of the test. Polymer-free coats, clothing, and gloves were always used. Glass and steel tools and equipment were used. Glass tools and equipment (with nitric acid 65 % and ultra-pure water) and stainless steel (with ultra-pure water) were rinsed regularly. Finally, they were dried and sterilized in an oven (170 °C) for 1 h. Aluminum foil was used to cover and wrap samples and clean and sterilized equipment and tools. To prevent contamination with hair particles, dust, dirt, and others, forearms, hands, and work surfaces were rinsed and wiped regularly (Ghayebzadeh et al., 2021).

2.7. Recovery test

Recovery tests were performed to accredit and confirm the extraction process. For this aim, a standard method (with minor corrections) was used. A Number of 120 and 150 approximately spherical and fragment-shaped plastic particles were added to ultra-pure water in sizes between 20–35 μm, 50–100 μm, and 120–150 μm, respectively (Tiwari et al., 2019; Weber et al., 2021). These particles are artificial reference particles and were mainly PE and PS. For this purpose, PE and PS powder were prepared from a local plastic factory. Then they were classified by sieves (all of the used sieves were model ASTM, E: 11, made of stainless steel by Damavand, Iran) with Mesh of 600, 400, 270, 150, 125, and 100 in different sizes (categorization mentioned above). The Olympus microscope (model BX41 and BX42, Olympus Company, made in Japan) equipped with a controller and a DP12 camera was used to count MPs and confirm the size. Two samples were considered for each size class, and the recovery rate was determined twice, (the recovery rate was determined for the entire filter area and the subarea of the checkerboard pattern (a quarter filter) in order to assess representativity). In total, twelve samples (for each) were used to evaluate the recovery rate. A comparison of recovery rate based on two counting methods (entire filter area and checkerboard pattern) was performed using the Mann-Whitney *U* test.

2.8. Hot needle examination

A hot needle test was carried out to ensure the extracted particles were plastic. With this test, plastic particles melt and become sticky (De Witte et al., 2014). This method was performed for raw water samples and larger particles.

2.9. Statistical analysis

To compare the abundance of MPs in 4 reservoirs (raw water) according to season (spring and winter) and also the abundance distribution of MPs among reservoirs, the One-way Repeated Measures Analysis of Variance test was used. Comparison of the abundance of MPs based on the season (spring and winter) in consumption taps was investigated by *t*-test. The average abundance of particles after treatment plants and consumption taps was compared by using ANOVA test. The distribution and comparison of colors and shapes of particles from the source to consumption taps were analyzed by using the Chi-Square test. The distribution of color and shape of particles in spring and winter was compared by the Wilcoxon test.

3. Result

3.1. Quality control and quality assurance

3.1.1. Blanks

On average, 0.29 ± 0.21 MP particles per blank (blank = cellulose nitrate membrane filter) were detected. The background contamination was insignificant compared with the abundance of MPs detected in the samples. Common polymers like PS, PET, and PE were found (Table S2). In blank

samples (blank = ultra-pure distilled water samples), in order to control contamination in the sample storage area, no MPs were detected.

3.1.2. Recovery rates

For MPs particles in the 20–35 μm, 50–100 μm, and 120–150 μm size class, the recovery rate when counting 100 % of the filter area was about 72–78 %, 81–87 %, and 90–94 %, respectively, and when counting checkerboard pattern, we obtained 61–70 %, 83–101 %, and 87–109 %, respectively (Table S3). The comparison of particle counting based on two methods (entire filter area and the checkerboard pattern method) was done using the Mann-Whitney *U* test, and the results showed that the difference between the two methods was significant for small particles (20–35 μm) ($p < 0.05$), and not significant for medium (50–100 μm) ($p = 0.081$), and large particles (120–150 μm) ($p = 0.248$).

3.2. Tracing MPs in the water (from source to consumption taps)

3.2.1. MPs abundance

This study showed that MPs were present in all samples (from source to consumption). In general, MPs fluctuated during the measurement period (from the source to the consumption point) and also based on sampling seasons.

Investigation of plastic particles in the samples of source 1 (4 reservoirs) displayed that the average abundance of MPs was different in spring (15.4–44.7 MP/m^3) and winter (22–51.8 MP/m^3) ($p < 0.05$). The differences in the mean number of MPs in water reservoirs were not statistically significant ($p = 0.54$). Nonetheless, the abundance of MPs was higher in reservoir 2 (spring = 51.8 MP/m^3, winter = 44.7 MP/m^3) (Fig. 2).

The results of sampling before and after water treatment plants (WTPs) showed that the amounts of MPs were generally lower in the treated water. In fact, a significant part of the MPs was removed by the treatment processes, 65 % at WTP_1, and 74 % at WTP_2 on average (Fig. 2).

The average MPs before and after WTP_1 were equal to 127.5 MP/m^3 and 45 MP/m^3, respectively (the average is for two seasons, spring and winter). The average MPs before and after WTP_2 were equal to 77.5 MP/m^3 and 20 MP/m^3, respectively (the average is for two seasons, spring and winter). There was a considerable numerical difference between the mean number of MPs in treated water in the WTP_1 and WTP_2.

Comparing the average abundance of MPs based on the season (spring and winter) in consumption taps indicated that the abundance of particles

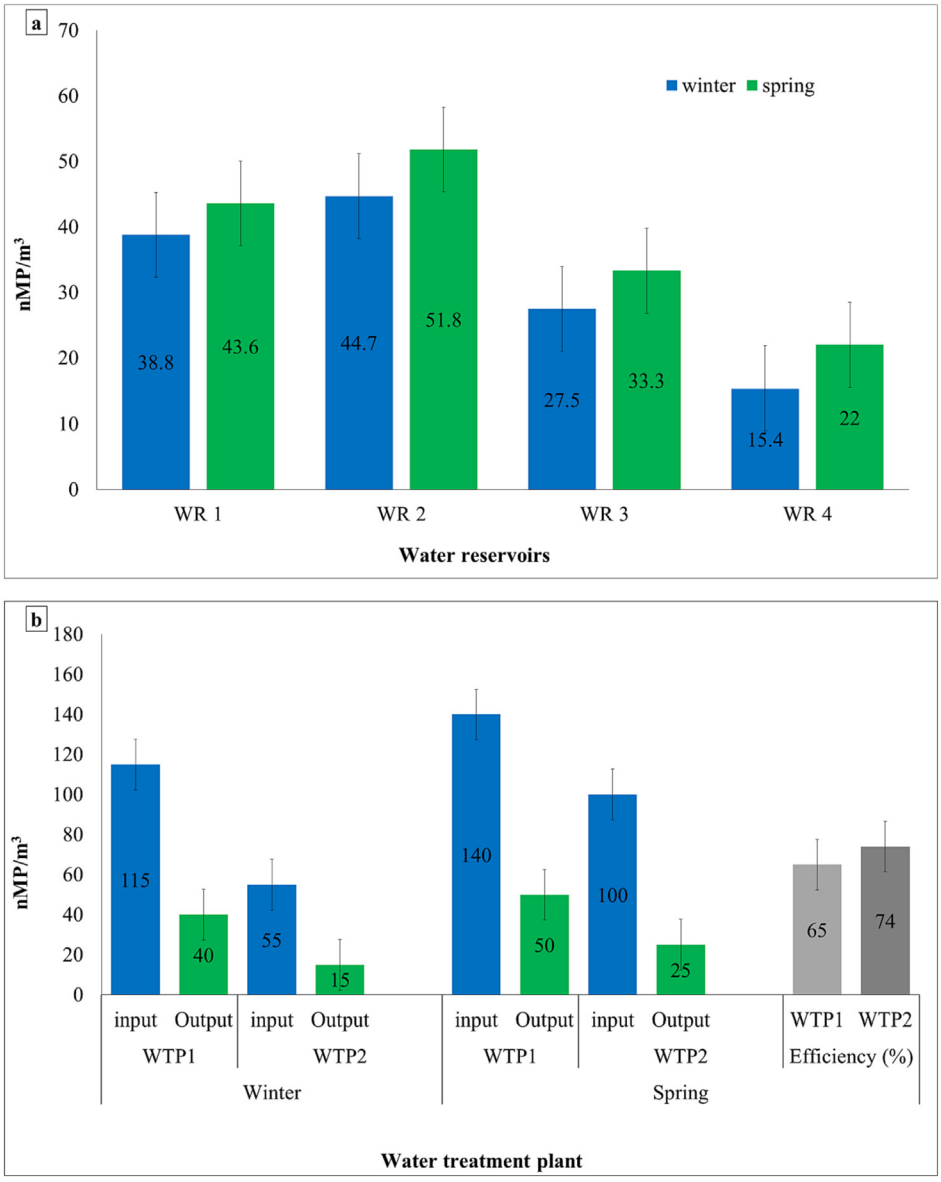


Fig. 2. Seasonal variation in MPs amounts in (a) water source samples (raw water), (b) water treatment plant (WTP) samples (before and after treatment), and (c) consumption taps (CT_a = consumption taps for WTP_1 and CT_b = consumption taps for WTP_2).

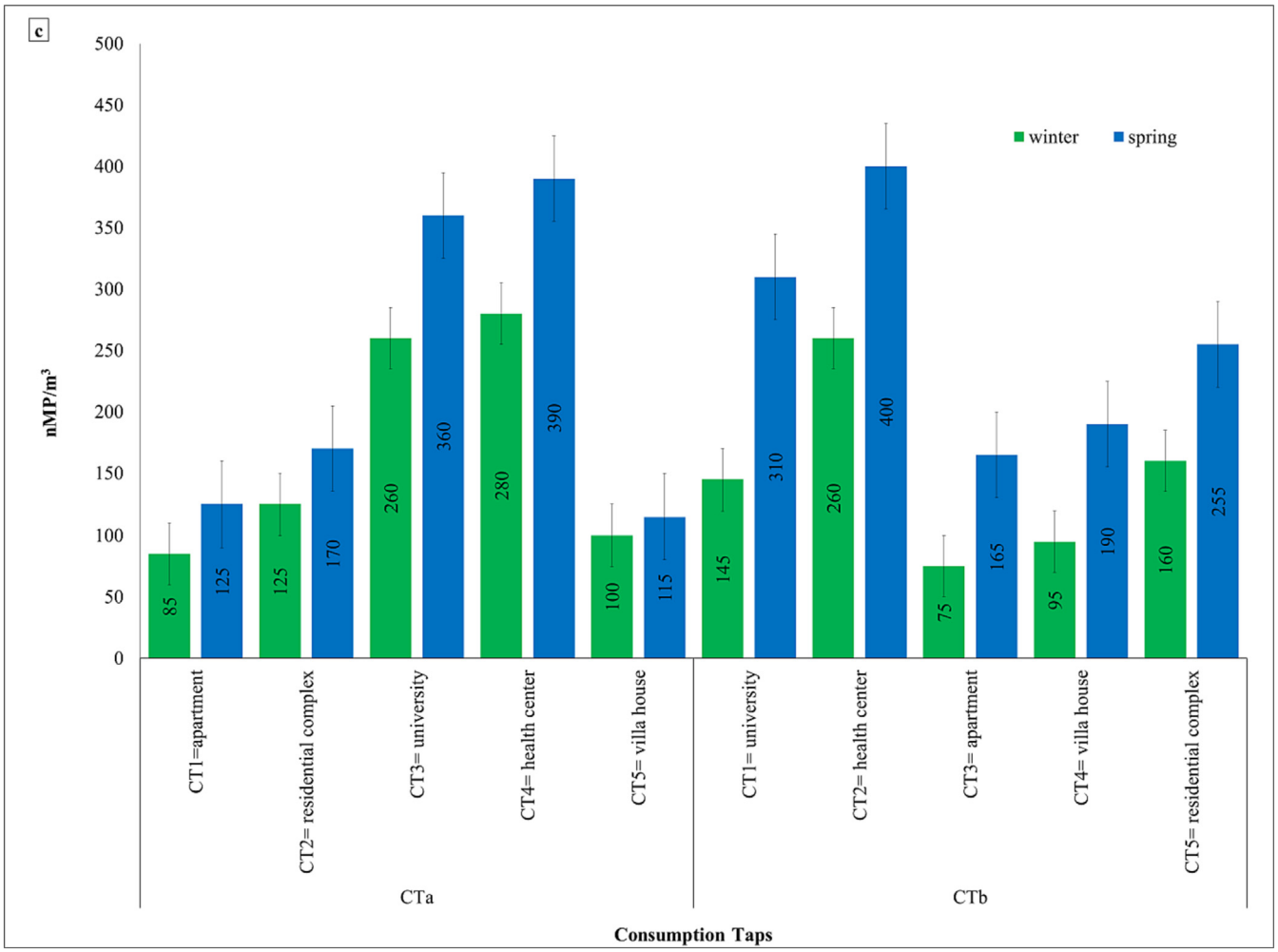


Fig. 2 (*continued*).

was different in spring and winter, and this difference was significant ($p < 0.05$). In the winter, the average MPs for CT_a and CT_b varied from 85 to 280 MP/m^3 and 75 to 260 MP/m^3, respectively. Also, in spring, average MPs for CT_a and CT_b varied from 115 to 390 MP/m^3 and 165 to 400 MP/m^3, respectively.

The investigation of the abundance of MPs in the consumption taps (CTs) demonstrated that the mean number of particles in CTs increased in comparison with the outlet of the water treatment plant (after WTPs), and this difference was also statistically significant ($p < 0.05$). Among CT_a, CT_3 belonging to the university (the average for two seasons of sampling was equal to 310 MP/m^3) and CT_4 belonging to the health center (the average for two seasons was equal to 335 MP/m^3) had the highest abundance of MPs and among CT_b, CT_1 belonging to the university (the average for two seasons was equal to 228 MP/m^3) and CT_2 belonging to the health center (the average for two seasons was equal to 330 MP/m^3) had the highest abundance of MPs (Fig. 2).

3.2.2. Distribution of size, shape, and type of MPs

MPs size distribution trends are shown in Fig. 3. Although there are distinctive differences in the numbers of MPs in the samples, a statistically significant difference was not observed in the size distribution of MPs particles. In sources (4 reservoirs), about 68–74 % of MPs were particles of 100–500 μm. 17–23 % of MPs were particles of 500–1000 μm. About 10 % of the particles were identified as 1000–5000 μm in size.

MPs size distribution in WTPs displayed that, before WTP_1, approximately 39–52 % of MPs were smaller than 50 μm, 26–32 % of MPs were 50–100 μm, 17–18 % of MPs were 100–500 μm, and about 4–11 % of the particles were 500–1000 μm in size. While after WTP_1, approximately 80–87.5 % of MPs were smaller than 50 μm, and 12.5–20 % of particles had a size of 50–100 μm (Fig. 3). Particles >100 μm were removed from WTP_1.

Before WTP_2, about 73–80 % of MPs were <50 μm in size, and 20–27 % of particles were the size of 50–100 μm. But after WTP_2, all particles were smaller than 50 μm in size (Fig. 3). Particles >50 μm were removed from WTP_2. As can be seen, the efficiency of WTP_2 in removing particles is better than WTP_1.

The size distribution of MPs in CTs indicated that >90 % of the particles were <50 μm in size and <10 % of the particles had sizes larger than 50 μm and even larger than 100 μm (Fig. 3).

MPs general shapes are displayed in Fig. S2. The types of MPs that have been found in source No.1 are fiber (46 %), fragments (43.5 %), foam (4.5 %), and film (6 %). These proportions are the average of two seasons (spring and winter). The shape and color distribution of MPs is shown in Figs. S3 and S4, respectively. The percentage of particle shapes and colors identified during the measurement period (from the source to the consumption point) and also based on the sampling seasons were distinguished, and the differences were statistically significant (color, $p < 0.05$ and shape, $p < 0.05$). Generally, in the water source (No.1), nearly 90 % of the particles were fragment-shaped and fiber-shaped, and <10 % were film-shaped and foam-shaped. The analysis of MPs shape distribution in WTPs (before and after WTPs) and CTs indicated that the fiber-shaped and fragment-shaped particles were in the water samples, but the fragment-shaped particles were dominant. While the foam and film-shaped MPs were removed in the water treatment system. <50 % of the particles before WTPs were fiber-

shaped, and >50 % were fragment-shaped. After WTPs, both particles were almost 50 %. In CTs, on average, >60 % of the particles were fragment-shaped, and <40 % were fiber-shaped.

The results presented about the color of the particles show that MPs identified in water source (No.1) had five different colors (red (14.7 %), blue (18.7 %), black (6.7 %), transparent (53.5 %), and white (6.4 %)). These proportions are the average of two seasons, spring and winter. >50 % of the identified particles were transparent in color. MPs color distribution in WTPs indicates that the identified particles are red, blue, and transparent. In general, blue MPs had a relatively high abundance before WTPs (nearly 50 % of particles before WTP_1 and 40 % of particles before WTP_2 were blue), but red particles were relatively abundant after WTPs (>30 % of particles after WTP_1 and >40 % of particles after WTP_2 were red). The results of color distribution in CTs show that, unlike WTPs, the color variation of the identified MPs was greater (five different colors including red (16.7 %), blue (23.3 %), black (11.5 %), transparent (23.9 %), and white (24.6 %)). Generally, the abundance of white MPs was relatively higher in CTs.

Fig. 4 illustrates the analysis of the surface morphology of MPs using a field-emission scanning electron microscope (FE-SEM). The surface morphology of the particles displays that some MPs have an uneven surface, and porosity, pits, roughness, cracks, and grooves are visible on the surface.

Identification of the type of polymers was performed for the samples by ATR-FT-IR and Micro-Raman. FT-IR and Micro-Raman spectra are presented in Fig. S5. Polymer-type distribution trends are demonstrated in Fig. 5. According to the results, except for the consumption taps, no distinguishable difference was observed in the polymer group of the investigated samples. In general, the polymer group identified for samples included polyethylene family (PE), polyethylene terephthalate (PET), polypropylene (PP), polystyrene (PS), expanded polystyrene (EPS), polyamide (PA), polyvinyl chloride (PVC), Poly-methyl methacrylate (PMMA), and rubber. (The raw data and additional descriptions are presented in Table S4). The

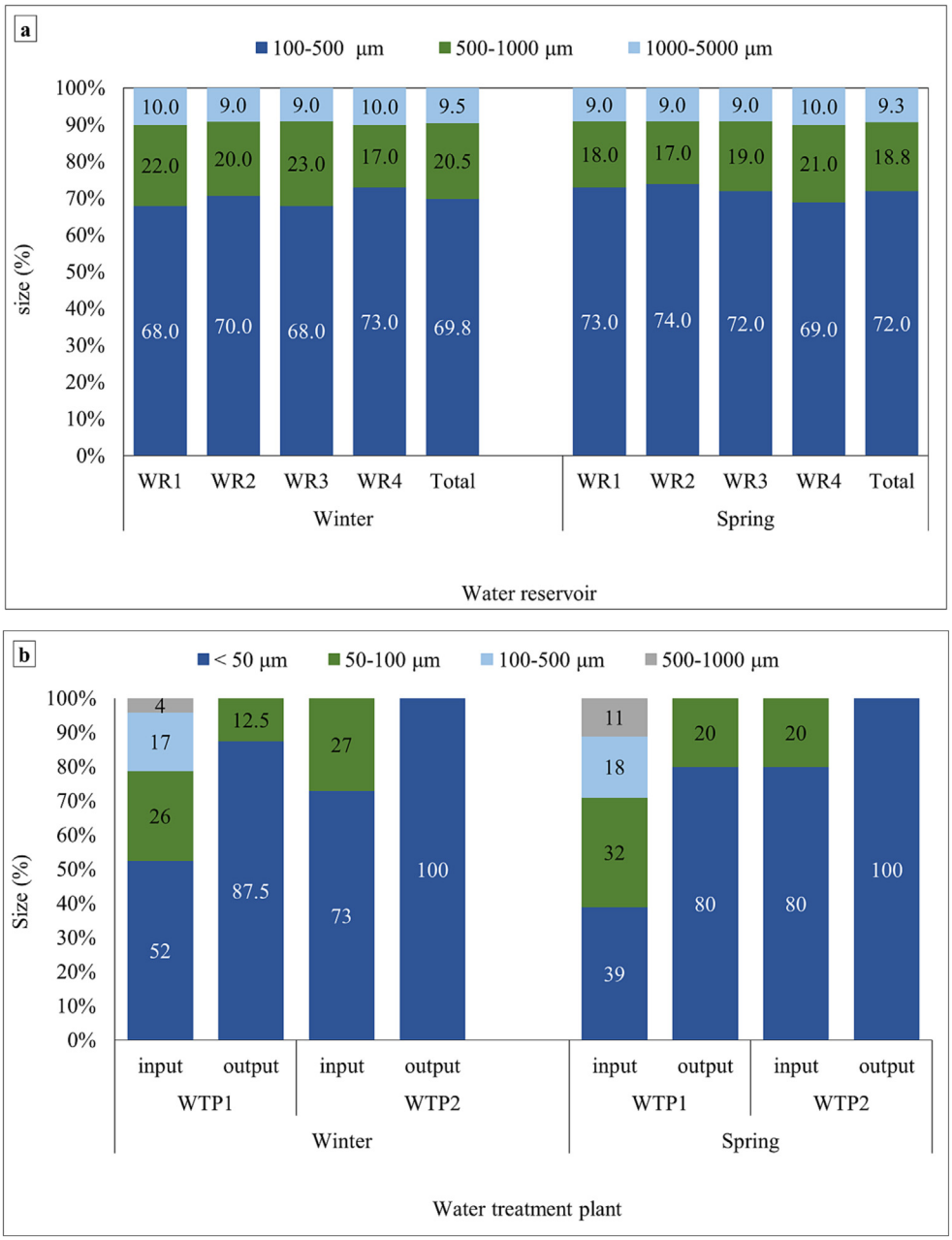


Fig. 3. Seasonal variation of microplastic based on size in (a) water source samples (raw water), (b) water treatment plant (WTP) samples (before and after treatment), and (c) consumption taps (CT_a = consumption taps for WTP_1 and CT_b = consumption taps for WTP_2).

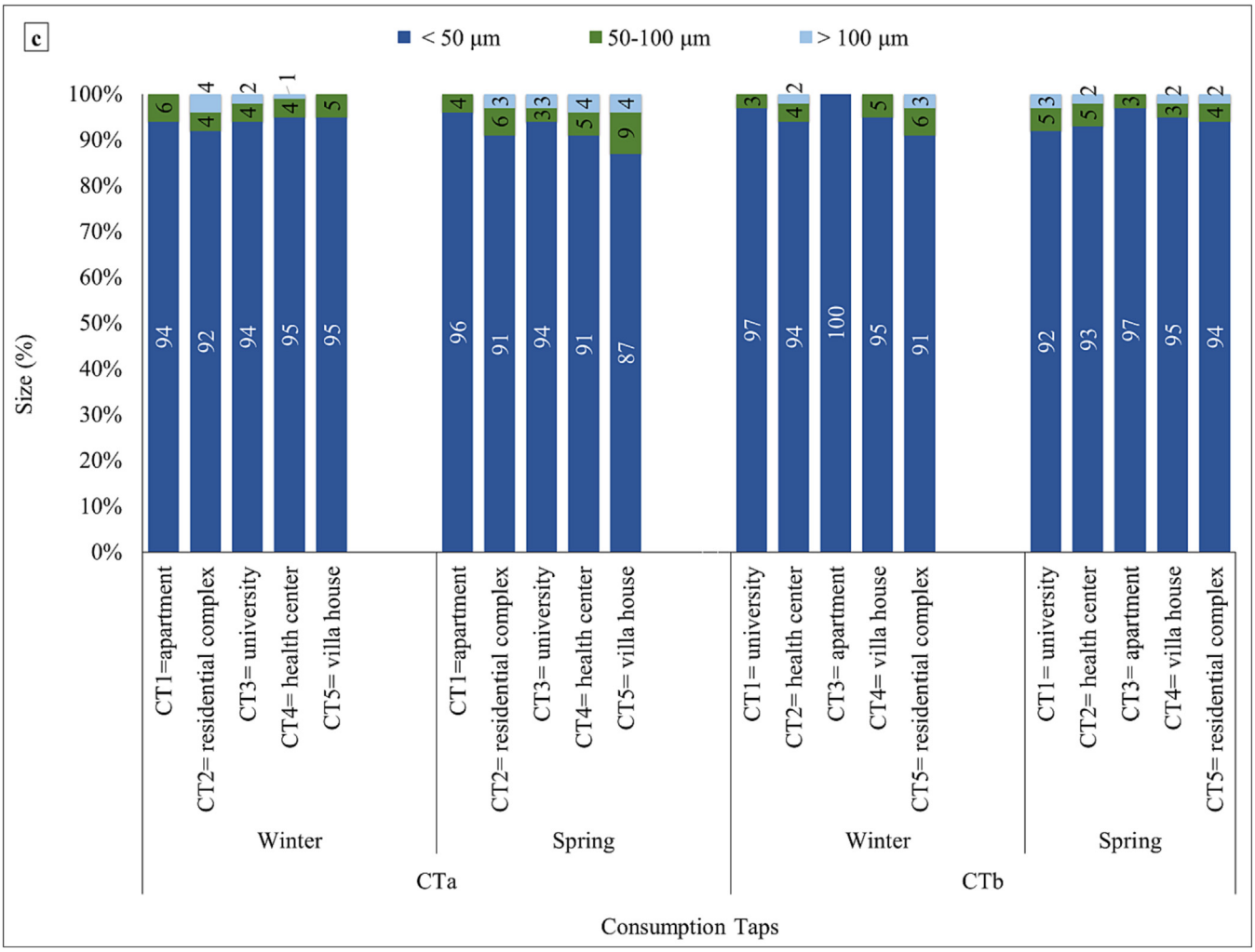


Fig. 3 (*continued*).

dominant MPs in sources (raw water), before and after WTPs, and CTs were PS, PE, PS, and PS, respectively.

4. Discussion

In this study, due to the stainless steel filter pore size (5 µm) and plankton net (100 µm), there were no limitations for the sampling volume due to obstruction.

4.1. Quality control and quality assurance

Low blank values indicate effective prevention of errors and contamination of samples in the study process and analysis.

The recovery rates showed that in both cases (recovery rate entire filter area and the checkerboard pattern), recovery was not 100 %. The extraction of particles in one of the steps (transfer, sample preparation, rinsing, processing, and filtration) had an error. Another reason may be due to the adhesion of particles to surfaces or to each other, causing errors. According to the results, it can be stated that the recovery rate decreases for smaller particles, and this has been clearly presented in the other studies (Pérez-Guevara et al., 2022b; Weber et al., 2021). The recovery rate of the checkerboard pattern had a higher range of counting oscillations and can lead to over- and under-estimation. The difference in the recovery rate in the entire filter area and the checkerboard pattern was approximately 7–8 %, and the difference was not recognizable and noticeable. Nevertheless, the recovery rate of MP particles can be estimated based on these data.

4.2. MPs abundance

MPs abundance was higher in spring than in winter (Fig. 2). In spring, due to the melting snow in Afghanistan, excess water from the Helmand River enters the reservoirs. Plastics may be affected by mechanical breakage, long-term oxidation, and external physical/chemical weathering (Mehdinia et al., 2020). Therefore, it is possible that due to high flow on the catchment path of the area, desolated and dispersed plastics (macro-, meso-, and microplastics) in the path were washed, and a massive volume of litter plastic fragments and particles entered the reservoirs. Also, the 120-day winds of the region during the hot period of the year (June to October) (Rajabi et al., 2006; Shahriar et al., 2018) most likely cause the atmospheric deposition of MPs and the transfer of plastic particles through the wind to the water surface of the reservoirs (Allen et al., 2019), moreover, probably the more decomposition and degradation of plastics have led to the increase of MPs in water during the spring season.

In source 1, all 4 water reservoirs are connected through waterways, and in the spring season (high water level in the reservoirs), water always flows between the reservoirs. In the winter (low water level in the reservoirs), the reservoirs that directly supply drinking water, the water level in the reservoir gradually decreases due to its use. To compensate for the

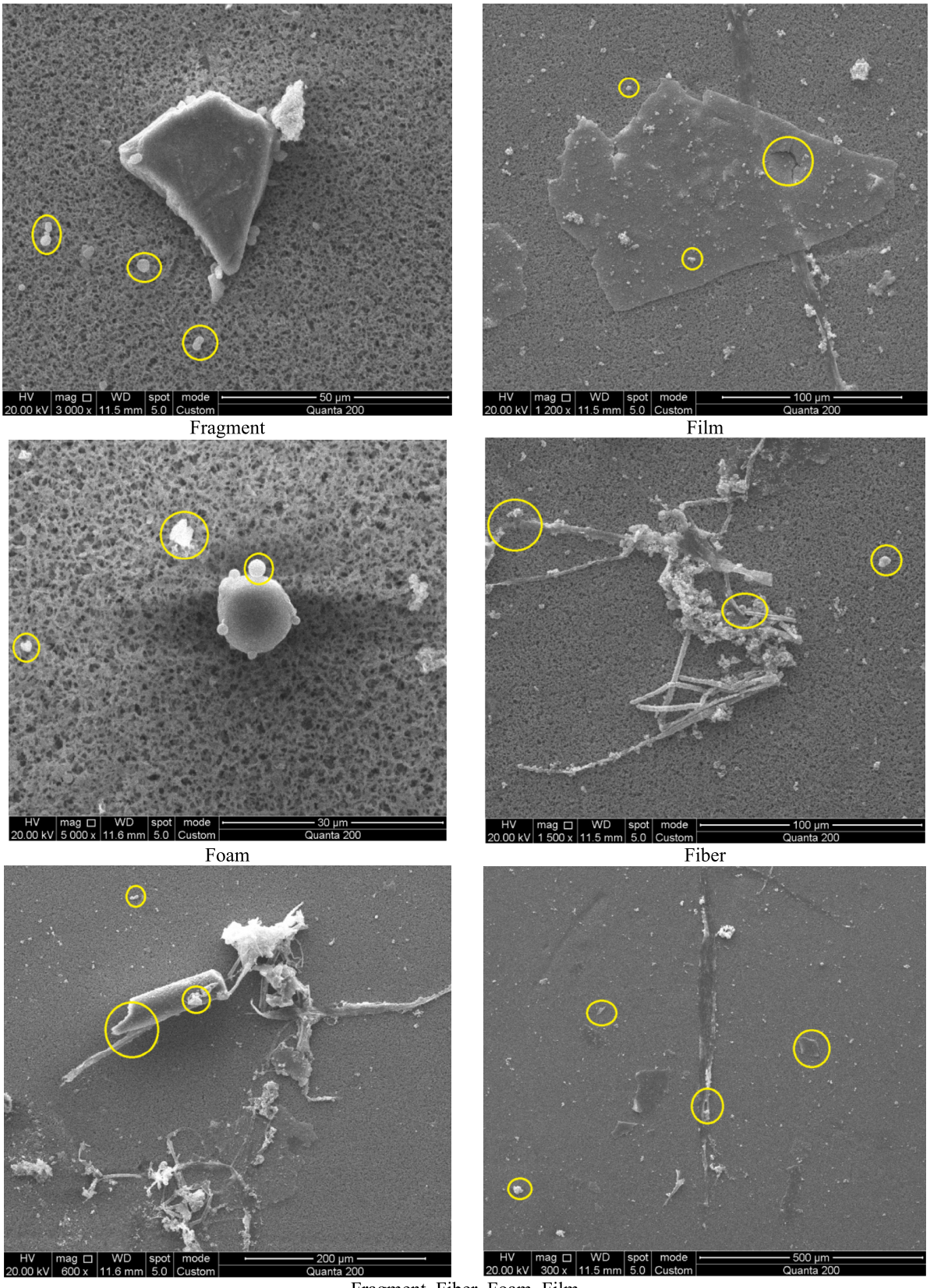


Fig. 4. High-resolution morphological characteristics of some MPs particles (yellow colors indicate the continued destruction and degradation of particles).

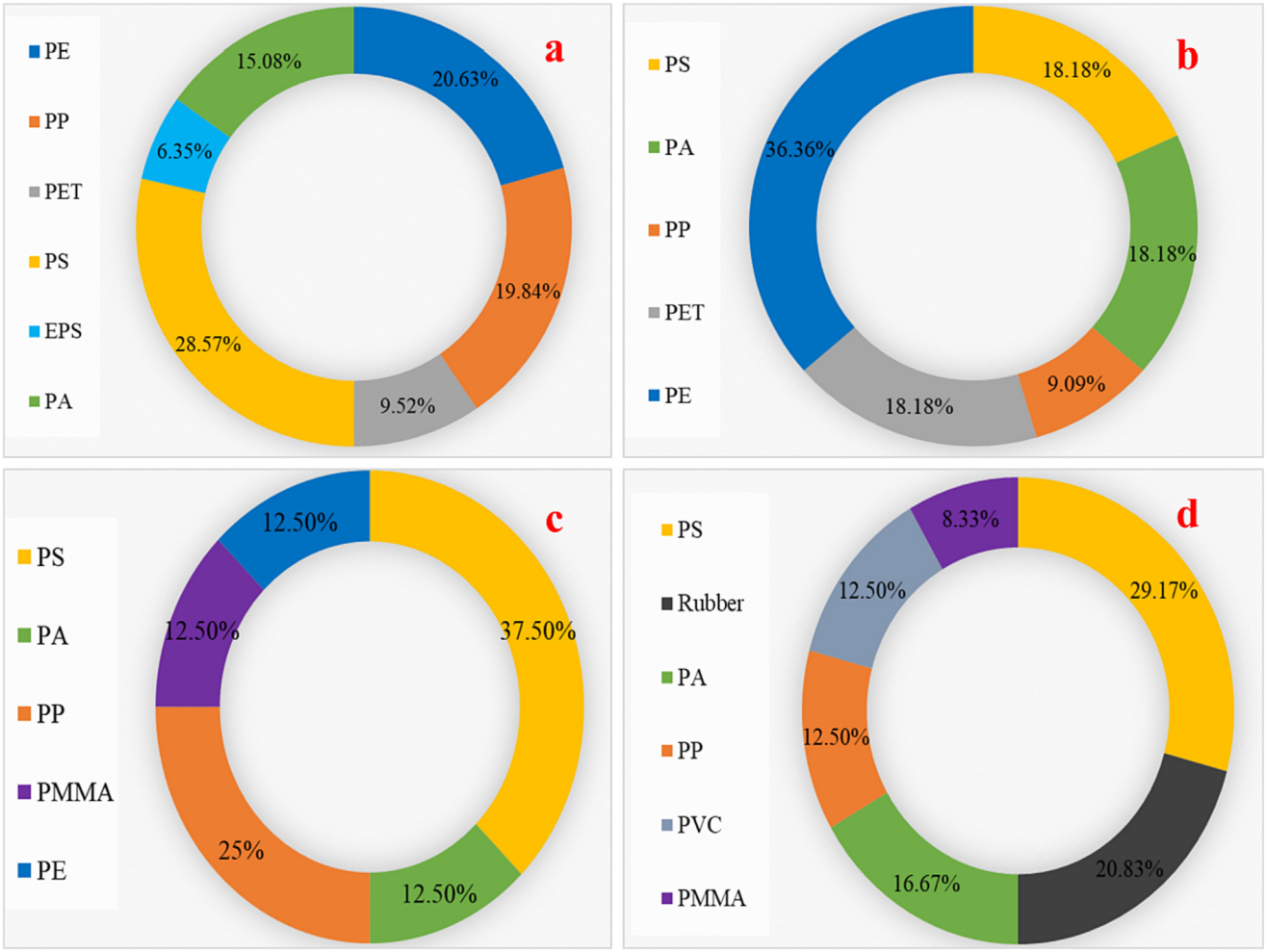


Fig. 5. The proportion of the types of MPs found in (a) raw water (source), (b) before water treatment plant, (c) after water treatment plant, and (d) consumption taps.

water shortage in these reservoirs, generally, water is directed into it by other reservoirs. Hence, observing non-difference (non-significance) in the average abundance of MPs in the reservoirs can be attributed to the connection of the reservoirs through waterways and water conduction (water flow) between those reservoirs. Nevertheless, the abundance of MPs in water reservoirs (WRs) was as follows: $WR_2 > WR_1 > WR_3 > WR_4$. The possible reason for the higher abundance of MPs in reservoir 2 is that there is a license for fishing activity only in this reservoir (the fishing tackle such as lines, net, rope, traps, and waders left on the reservoir margin) which may have led to a high abundance of MPs, too (Fig. 2). Reservoir 1 is the first recipient of Helmand River water. As mentioned above, plastics and microplastics in the catchment path can be washed and enter the reservoir. For this reason, it has higher MPs in comparison with reservoirs 3 and 4. On the other hand, reservoir 4 is the last water recipient/storage reservoir of the Helmand river water, and the water volume of this reservoir alone is higher than the sum of the three other reservoirs.

MPs difference in treatment plants 1 and 2 (before and after treatment plants) was not impressive, but there was a slight difference. The difference in abundance of MPs in raw water (before WTPs) can be caused by the kind of water body (water supply sources, including surface water and groundwater). Another reason for this abundance difference can be attributed to the increase of particles during transport in water pipes. As reported in previous studies, the use of polymer composition similar to plastic during water purification, transportation, and packaging, indicates the point of entry of MPs into drinking water (Eerkes-Medrano et al., 2019; Zhang et al., 2022). On the other hand, various other studies have stated that the abrasion of plastic equipment during water purification or transport can be the source of MPs in drinking water (Mintenig et al., 2019; Tong et al., 2020). In the present study, water is directed to the treatment plant through transfer pipes (made of polyethylene with different diameters and lengths). The length of the transfer pipes to WTP_1 is about 200 km, and the length of the transfer pipes to WTP_2 is about 30 km (Text S1). Therefore, because of the longer length of transfer pipe 1, more pipe abrasion may have occurred and affected the higher abundance of particles before WTP_1. However, more substantial evidence is needed to support the inference that plastic equipment, especially plastic pipes is a possible source of MPs in drinking water. The difference in abundance of MPs in treated water (after WTPs) is probably caused by the type of treatment process (WTP_1 = pre-chlorination, coagulation and flocculation, clarifier, filtration, and secondary chlorination and WTP_2 = desalination by reverse osmosis). Despite the good performance of WTP_2, it is still impossible to claim with certainty which purification process has better efficiency. Our knowledge on this matter needs to be improved and further study is needed. Because water quality (in terms of the presence of MPs, such as abundance, size, shape and type of polymer particles, as well as water quality parameters such as turbidity, total organic carbon "TOC" and similar items) affects the treatment performance (Jung et al., 2022).

The results indicated that the abundance of MPs increased in CTs. Disinfectants (such as chlorine) is widely employed in drinking water treatment plants, and water pipes (especially the pipes of the water distribution system) are usually exposed to an oxidative environment. Due to residual chlorine in water, plastic pipes are exposed to an oxidative environment for the long term, and the surface of the pipes undergoes chemical changes (Whelton et al., 2011). Other studies have reported that surface oxidation,

polymer chain scission, crack propagation, and mechanical failure can be observed in plastic pipes (especially aged plastic pipes) (Fischer et al., 2019; Fujii et al., 2019). Also, plastics exhibit a significant degree of aging when exposed to water disinfectants (Cosgriff et al., 2017; Fischer et al., 2016). Therefore, due to the aging mechanism and plastic material characteristics, it is likely that such processes lead to the release of more MPs from plastic pipes (Mintenig et al., 2019), and they can affect the high abundance of MPs in CTs.

On the other hand, due to conditions of global drought and water scarcity, and also because of the maximum water consumption at certain times of the day and reduced water pressure, almost the entire city uses plastic storage tanks (Fig. S1, C), to supply the daily water requirement. Using plastic water storage tanks and their frequent filling and emptying (due to possible water outages and lack of water pressure, especially in the hot season) can affect the abundance of MPs by causing abrasion in the tanker body. It is necessary to mention that the storage tanks are kept on rooftops of buildings or in the yard and are directly exposed to ultraviolet light or sunlight and in unsuitable conditions (suitability for photo degradation). So, photo-oxidation by incident ultraviolet (UV) light is one of the most important mechanisms causing chain scission reactions, and it leads to the rapid destruction of plastics (Gewert et al., 2018). Hence, photo-degradation may be one of the reasons for the release of MPs in tanks and the increase of particles in CTs (especially in the hot season). In this study, the water distribution system in health centers was older than the other places. In aged water pipes, the possibility of MPs release is higher. So, aged pipes can be the reason for high particles in health centers. The length of the water distribution system, the volume of water storage tanks, and the amount of water consumption in the university are more than the other places. Abrasion of plastic equipment during water transport is a source of the release of MPs. Therefore, long water transport can affect the abundance of MPs. Based on the results, it can be inferred that the age of the pipes has a higher effect on the abundance of MPs than the length of the water distribution system. Given that, the apartments and the water distribution system were new, and also the length of the water distribution system was less than the other places. Hence, the abundance of MPs in the apartment consumption tap was less.

4.3. Size and morphology

Fig. 3 displays MPs size distribution in raw water samples (source) to consumption tap. In raw water (source No. 1 or 4 reservoirs), particles larger than 100 μm were detected. At the time of conducting this study, plankton nets with smaller pore sizes (<100 μm) were not available. So, particles smaller than 100 remained unknown in the sources. Nevertheless, MPs' size distribution indicates a high abundance of fine particles. The dominant source of small MPs often is the fragmentation of larger plastics or product wear (Shim et al., 2018; Zeng, 2018). Biodegradation, optical oxidation, thermal oxidation, and weathering are the main reasons for plastic fragmentation (Jambeck et al., 2015), although the fragmentation rate under natural conditions is still unknown. Particles larger than 100 μm were eliminated in the water treatment plant system. An increased proportion of small-sized MPs shows that large-sized MPs may have been mostly eliminated by filtration (Jung et al., 2022).

Despite that, the water treatment plant system had removed particles >100 μm, yet a few particles >100 μm were detected at the consumption tap. Therefore, it can be concluded that secondary pollution (abrasion of pipes, installations, and sealing materials during water purification or transport, and also chemical and photo-oxidation and re-destruction of plastics) probably occurred at CTs. Various studies have also reported this issue (Gewert et al., 2018; Jung et al., 2022; Mintenig et al., 2019; Tong et al., 2020).

Considering MPs shape distribution in the samples and the complete elimination of foam and film-shaped particles by the water treatment plants, as well as the difference in removal between fiber and fragment-shaped particles (relative increase in the exclusion of fiber particles compared to fragment), it can probably be inferred that there is a relationship between the shape of MPs and their delete ability by different water treatment technologies (Pivokonsky et al., 2018), which was outside the objectives of our study though more study is needed for firm confirmation. Plastic fragments in water are expected to originate mostly from microbeads (MBs). Microbeads are usually >0.1 μm and ≤1 mm in size (are commercially available in sizes from 10 μm to 1 mm) and can be manufactured with different shapes and from various types of polymers (Bashir et al., 2021; Möhlenkamp et al., 2018). The most common types of polymers used are polyethylene (PE), polypropylene (PP), polystyrene (PS), polyethylene terephthalate (PET), poly methyl methacrylate (PMMA), Nylon (N6, PA6), ethylene vinyl acetate copolymers (EVA), polyurethane (PUR), polytetrafluoroethylene (PTFE), acrylic and polyester (PES) (Andrady, 2017; Yurtsever and Yurtsever, 2019). The most common shapes of microbeads are spherical or irregular (fragments) (Isobe, 2016). Microbeads are frequently used in the most common Personal Care and Cosmetic Products (PCCPs), including toothpaste, deodorant, face, and body cleansers, exfoliats, lotions, types of creams, hair coloring, nail polish, and other similar products (Cheung and Fok, 2016; Isobe, 2016). MBs are not solely used in Personal Care and Cosmetic Products (PCCPs). They are also used as abrasive beads in detergents and cleaning fluids, lubricants, pigment carriers, and separating agents; and are melted into plastic objects such as toys, packaging, and pharmaceuticals and finally are used in many industrial processes (EPA, 2016; Yurtsever and Yurtsever, 2019). Most of the particles found in cosmetic products are <100 μm (Kalčíková et al., 2017). It has been reported that microbeads are the main source of microplastics in wastewater, and >15 mg per person per day are released into the wastewater system (Kalčíková et al., 2017). Various studies have found that wastewater treatment plants are an important route of releasing microplastics, especially microbeads into rivers and freshwaters (Ding et al., 2020; Kalčíková et al., 2017). In this study, considering that most of the particles detected in CTs are <100 μm in size (the main size of microbeads is <100 μm as well), and the dominant shape of the particles identified in CTs is fragment (most common shapes of MBs is irregular or fragment as well), and nearly 70 % of the polymers detected in CTs are PP, PS, PMMA, and PA, which are the most common polymers used in MBs. Therefore, it can be said that the main origin of the identified particle in this study (from the source to the consumption taps) was probably microbeads.

Fig. 4 shows the analysis and investigation of the surface morphology of MPs using SEM. MP particles, such as film, showed unevenness of the surface, as well as other surface characteristics, including roughness and cracked surface. Fiber-shaped MPs particles show complexity, continued destruction, and breakage. The fragments depict a combination of relatively smooth surfaces with irregular edges. Foam particles indicate the possibility of continued destruction and turning into finer particles, as well as the sticking of smaller particles on the surface of particles. Analysis of particle surface morphology displays that MPs underwent persistent weathering in the water stream, mechanical breakage, and oxidation (Abbasi et al., 2018; Li et al., 2018; Wang et al., 2017).

The results of identifying the type of plastic polymer structure displayed that the family of polystyrene (PS) and polyethylene (PE) particles were the most common particle in the source and before WTP_1 (raw water). PS particles are mostly used in disposable containers, types of foam packaging, food packaging (dairy, fishery), Styrofoam, and more. PS plastics are used with a high proportion for daily plastic needs, and primary components of plastic debris are detected in the water environment (Barboza et al., 2019; Bayo et al., 2019; Facts, 2019; Kershaw and Rochman, 2015). Also, to keep fishing nets and ropes floating and suspended and to prevent some fishing tools from sinking in water reservoirs, the Styrofoam has been used according to field observations (Fig. S6). Meanwhile, plastic litter left (such as disposable containers, types of foam packaging, food packaging (dairy, fishery), and Styrofoam) in this aquatic ecosystem was abundantly visible in the study area (Fig. S6). Polystyrenes have poor UV light resistance behavior, and polymer weathering and photo-oxidation occur faster. So, these polymers can be converted into smaller particles (micro- and nano-plastics) in less time. Therefore, these reasons may have

led to the high abundance of polystyrenes in raw water (Feldman, 2002; Wypych, 2022).

PE is among the most abundant polymers in environmental samples. The entity of PE particles in raw water may be due to the presence of plastic bags, plastic bottles, food containers, and disposable packages (Barboza et al., 2019; Bayo et al., 2019; Facts, 2019; Kershaw and Rochman, 2015). It can be said that probably another major source of the presence of PE in the studied area is the discharge of wastewater into the Helmand catchment (as the origin of water supply for reservoirs). In addition, given that the type of transferring water pipes (from source No.1 to the treatment plant) in the Zahedan city water supply system is mostly PE. Consequently, there is a possibility that it has influenced the abundance of PE in the samples before the treatment plant.

According to the results of the current study, the predominant MPs after the treatment plant and consumption taps were Polystyrene (PS). As mentioned above, polystyrenes have poor resistance to weathering and chemical and photo-oxidation (Wypych, 2022). Therefore, there is a possibility that polystyrene MPs are poorly resistant to disinfectants (chlorine and ozone) and have caused their destruction and disintegration (Wypych, 2022), and so these particles have increased in number after the treatment plant and at consumption taps. Also, it is necessary to explain that the type of water storage tanks in the study area was mostly PP. Hence, it can be stated that particles separated from the tanks may have affected the abundance of PP in samples of consumption taps in Zahedan city.

Some of the particles (PMMA, PVC, and rubber) were identified in CTs samples, which were not detected even in raw water. Of course, it cannot be claimed that this particle does not exist in raw water. These particles can possibly be detected by repeated sampling and sampling by nets with finer pore sizes. PMMA is mostly used in making electronic components (televisions, laptops, smartphones, etc.), aquariums and canopies, medical materials, etc. (Avella et al., 2001), and is one of the most common polymers used in microbeads (Yurtsever and Yurtsever, 2019). So, with a low probability, the presence of PMMA in CTs can be attributed to microbeads (Bashir et al., 2021). PVC particles can be due to their use as pipes in municipal water supply distribution systems. The possibility of rubbers in water can be due to their use as fittings sealing.

Analyzing the type of particles polymer structure and tracking MPs in water (from the source to the point of consumption), especially the difference in the type of polymers before the treatment plant (raw water) and after the treatment plant (in CTs), indicates a possible relationship in the removal of particles based on polymer structure type. Some particles indicate more expulsion ability that can depend on various factors. These factors can even be the density, shape, size, and degradability of the particle. For example, PET was fully eliminated in the treatment plant (Fig. 5). Among the detected particles, PET has high resistance to weathering, chemical and photo-oxidation (Wypych, 2022). Therefore, they are difficult to destroy and disintegrate. In addition to the above (secondary pollution and weathering, chemical and photo-oxidation resistance), one of the treatment processes may have been defective (for example, the presence of a defective or damaged, broken, unusable nozzle and other similar cases). Therefore, the distribution of MPs' size, shape, and type varies when passing through each treatment process, and tracking their distribution is essential for monitoring and controlling MPs in water.

4.4. Comparison with other studies

Recently, the presence of MPs in drinking water has attracted significant attention. However, tracing MPs from the source (raw water) to consumption taps has been evaluated rarely. Sampling methods significantly vary in different studies. Nevertheless, the methodology was compared in a few slightly similar studies and presented in Table 1. It is necessary to explain that the present study was divided into three parts (samples of raw or source water (No.1), treated water, and consumption taps) and the results of each part were compared with different studies (Jung et al., 2022; Leslie et al., 2017; Pivokonsky et al., 2018; Wang et al., 2020; Weber et al., 2021). In order to better and easily compare the results of this study with those

Table 1
Comparison between the results of this study and other studies (MPs abundance, dominant MPs, and MPs type). The unit was standardized for comparison (Unit: MP/L).

Research	Raw water (RW)		Treated water (TW)		CTs[a]	Polymer type & MPs shape	Size	Treatment process
Jung et al., 2022	2.2 ± 1.3		0.02 ± 0.02	–	PE, PET & –	>10 μm	Pre-ozonation + Sedimentation Sand + filtration
Leslie et al., 2017	Dry season	1385	448.7	–	Fiber	6.5–500 μm	Screen-outlet Clarifier-outlet Filtration
	Rainy season	1796.6	769.4				
Wang et al., 2020	6614 ± 1132		930 ± 72	–	PET, PE, PP &fragment fiber	>1 μm	Coagulation/sedimentation + Deep bed filtration + Ozonation + GAC filtration[b]
Pivokonsky et al., 2018	RW 1	RW 2	TW 1	TW 2	TW 3	–		>1 μm	TW1: coagulation/flocculation and sand filtration.
	1473 ± 34	1812 ± 35	443 ± 10	338 ± 76	628 ± 28		PE, PP, PET & fragment fiber	>1 μm	TW2: coagulation/flocculation, sedimentation, sand and granular activated carbon filtration
		RW 3							TW3: coagulation-flocculation, flotation, sand filtration and granular activated carbon filtration
		3605 ± 497							
Weber et al., 2021	RW1	RW2	TW 1	TW 2	0	–	>10 μm	Aeration + flocculation + sand filtration
	–	–	0	0				
This study[e]	RW 1	RW 2	Before	After	Before	After	CT1	CT2	PS & fragment fiber	>100 μm[c]	TW1: pre-chlorination, coagulation and flocculation, clarifier, filtration, and secondary chlorination
			WTP1	WTP1	WTP2	WTP2					
	0.035 ±	–	0.128 ±	0.045 ±	0.078 ±	0.020 ±	0.201 ±	0.206 ±		>5 μm[d]	TW2: desalination by reverse osmosis
	0.012		0.022	0.01	0.03	0.008	0.113	0.106			

[a] Consumption taps.
[b] Granular activated carbons.
[c] Limit of size detection for RW.
[d] Limit of size detection for TWs and CTs.
[e] In this study, the results reported for raw water 1 are the average of 4 water reservoirs. By the way, the average and standard deviation are the samples of two seasons.

studies, the unit was standardized for comparison, and a unit based on "MP/L" was provided. However, some restrictions in the studies (for example, the lack of plankton nets with smaller sizes) have confined the possibility of comparison. As can be seen in Table 1, the results obtained in different studies, including our study, are completely different. There are many reasons that can lead to differences in the amount of MPs particles. The contamination of MPs varies geographically with locations and times (rainy season) (Shim et al., 2018). Other factors such as sampling method (volume, size, and method of extracting and preparing samples) (Kirstein et al., 2021; Pivokonsky et al., 2018), type of water supply source (river, dam or reservoir, underground water and fountain) (Kirstein et al., 2021; Pivokonsky et al., 2018), and anthropogenic factors (plastic waste management and environmental protection strategies) (Shahul Hamid et al., 2018) affect the abundance of MPs in water (especially raw water). In addition to those, the type of water treatment method and secondary pollution (abrasion of pipes, installations, and sealing materials during water purification or transport, and also chemical and photo-oxidation and re-destruction of plastics) may lead to a difference in the abundance of MPs in treated water and taps.

The MP found in our samples is much lower compared to other studies. Also, the size of MPs identified in other studies was smaller than that of the current study (about 90 % of particles reported smaller than 5 μm) (Oßmann et al., 2018; Pivokonsky et al., 2018). Therefore, it is not surprising that the abundance of MPs is much lower compared to other studies.

The appearance and type of polymer identified in different studies are usually similar. The most commonly used polymers in different areas and countries include PE, PS, PP, and PET. For this reason, they are abundant in the environment in most areas compared to other polymers. The predominant appearance of particles in different studies is fiber and fragment (Pivokonsky et al., 2018; Wang et al., 2020). Fiber and fragments are the most abundant form of MPs particles (Toumi et al., 2019). According to Table 1, common treatment processes can not completely remove these particles. Also, it has been confirmed that MPs from the supply network (especially fragment-shaped MPs) can enter drinking water occasionally (Kirstein et al., 2021), because plastics usually turn into irregular shapes when they are fragmented into pieces, and on the other hand, microbeads usually have irregular shapes. So, the high prevalence of these particles indicates MP contamination from a secondary origin and microbeads (Kalčíková et al., 2017; Toumi et al., 2019).

4.5. The calculation of estimated daily intake (EDI) and annual intake (EAI)

According to the ingestion rate of drinking water in adults (IR = 2–2.5 l) and children (IR = 1–1.5 l), the EDI and EAI were determined, and the results are presented in Table 2. EDI for adults and children was estimated 0.0002–0.0157 MP/kg/bw/day and 0.0004–0.0413 MP/kg/bw/day, respectively. EAI for adults and children was obtained 0.073–5.736 MP/kg/bw/year and 0.160–15.056 MP/kg/bw/year, respectively. In Iran, the bottled mineral water EDI for adults and children was reported to be about 0.005–0.011 MP/kg/bw/day and 0.022–0.065 MP/kg/bw/day, respectively. EAI for adults and children was reported about 1.831–5.35 MP/kg/bw/year and 8.011–23.43 MP/kg/bw/year, respectively (Makhdoumi et al., 2021). Zuccarello et al., have reported the results of EDI for adults and children about 1,531,524 MP/kg/bw/day (40.1 mg/kg/bw/day) and 3,350,208 MP/kg/bw/day (87.8 mg/kg/bw/day), respectively (Zuccarello et al., 2019). Many parameters may affect EDI results, including the type of drinking water supply sources, the quality of containers and water storage tanks (for example, the quality of water tanks and mineral water bottles), drinking water quality (type of treatment process), consumption behavior, secondary pollution (storage condition of bottled water and drinking water), frequency of usage of water bottled, and sampling, extraction, and detection methods of MPs (Koelmans et al., 2019). So far, there are no exact and complete data and results regarding toxicology and MPs. Therefore, the health and human risk assessments still need to be clarified.

Some identified plastic polymers have a lower risk scale (like PE, PET, and PP) or are known as less toxic polymers (Lithner et al., 2011; Rochman, 2015). Nevertheless, some plastic, such as PVC and PMMA have a higher risk scale and cause mutagenic and carcinogenic effects. Some studies have reported MPs as a vector to other pollutants (heavy metals and organic substances) which can lead to a higher risk of exposure to these known toxic substances, and it can also have a synergistic effect (Lithner et al., 2011; Rochman, 2015).

Recent studies on various animals have reported several problems focusing on exposure to micro- and nano-plastics; endocrine disruptions, effect on mobility and development, interference with immune response, impaired reproduction, carcinogenesis, transgenerational effects, and cytotoxicity in cerebral and epithelial human cells (Akdogan and Guven, 2019; Barnes et al., 2009; Lithner et al., 2009; Miranda et al., 2019). A recent study confirmed the plastic particle pollution in human blood (Leslie et al., 2022). Therefore, there is a possibility that all those mentioned effects can also be revealed in humans.

5. Conclusions

The current study is the first one that has tracked MPs in drinking water from the source to consumption taps. The results indicated that the abundance of particles (especially smaller particles) increased after water treatment and in the distribution network and also the consumption taps. Probably, the smaller particles have poor chemical resistance and have been re-disintegrated and re-destroyed, and their number has increased after the treatment plant. Some of the particles (PMMA, PVC, and rubber) were identified in CTs samples, which were not detected even in raw water. Therefore, it can be concluded that probably secondary pollution has occurred, including the release of MPs from plastic water storage tanks and aged pipes (due to abrasion). The shape and type of polymers were different before the treatment plant (raw water) and after the treatment plant (in CTs). This difference may indicate a relationship in the removal of particles based on shape and polymer structure type. The surface morphology of MPs showed that the particles were affected by continuous weathering, mechanical breakage, and oxidation. According to the EDI results of drinking water in the study, it is likely that children are more at risk of these toxic substances. Finally, it can be stated that only purifying water is not enough to reduce MPs, and these particles should be controlled and managed in the municipal water distribution network and consumption taps (to prevent secondary pollution). Identifying gaps and shortcomings to understand the occurrence of MPs pollution in drinking water in different seasons, based on the quality of drinking water (turbidity, pH, TOC, etc.) and minimum and maximum peak water consumption, and also study in smaller sizes (nano), still requires additional research. The knowledge concerning exposure to MPs and assessment of health effects remains largely unknown and requires further research.

Table 2
Comparative EDI and EAI for adults and children for different levels of consumption (MP/kg/bw/time).

Group	IR_{day}	IR_{year}	BW kg	Min	Average	Max	n/kg/bw/day			n/kg/bw/year		
							EDI_{min}	$EDI_{average}$	EDI_{max}	EAI_{min}	$EAI_{average}$	EAI_{max}
Adult	2		70	0.007	0.203	0.44	0.0002	0.0058	0.0126	0.073	2.117	4.589
	2.5		70	0.007	0.203	0.44	0.0003	0.0073	0.0157	0.091	2.646	5.736
Children	1		16	0.007	0.203	0.44	0.0004	0.0127	0.0275	0.160	4.631	10.038
	1.5		16	0.007	0.203	0.44	0.0007	0.0190	0.0413	0.240	6.946	15.056

Funding

Funded by Zahedan University of Medical Sciences.

CRediT authorship contribution statement

Conception and design of study: *M.Ghayebzadeh, H.Taghipour*.
Acquisition of data: *F. Ganji, N. Azizi*.
Analysis and/or interpretation of data: *S. Mousavi, M.Ghayebzadeh*.
Drafting the manuscript: *M.Ghayebzadeh, H.Taghipour*.
Revising the manuscript critically for important intellectual content: *M. Ghayebzadeh*.
Approval of the version of the manuscript to be published (the names of all authors must be listed): *H. Taghipour, M. Ghayebzadeh, F. Ganji, S. Mousavi, N. Azizi*.

Data availability

Data will be made available on request.

Declaration of competing interest

No conflict of interest exists.

Acknowledgments

Funding for this study was supported by Zahedan University of Medical Sciences, Iran (As a Research project, grant number 10342, Approval ID = IR.ZAUMS.REC.1400.245). The authors desire to thank financial suppliers and others who had logistical support. Especially we would like to thank Dr. Mohammad Reza Lakzaei for all their help and organizational support.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2023.162121.

References

Abbasi, S., Soltani, N., Keshavarzi, B., Moore, F., Turner, A., Hassanaghaei, M., 2018. Microplastics in different tissues of fish and prawn from the Musa Estuary, Persian Gulf. Chemosphere 205, 80–87.
Akdogan, Z., Guven, B., 2019. Microplastics in the environment: a critical review of current understanding and identification of future research needs. Environ. Pollut. 254, 113011.
Allen, S., Allen, D., Phoenix, V.R., Le Roux, G., Durántez Jiménez, P., Simonneau, A., Binet, S., Galop, D., 2019. Atmospheric transport and deposition of microplastics in a remote mountain catchment. Nat. Geosci. 12, 339–344.
Al-Oufi, H., McLean, E., Kumar, A.S., Claereboudt, M., Al-Habsi, M., 2004. The effects of solar radiation upon breaking strength and elongation of fishing nets. Fish. Res. 66, 115–119.
Andrady, A.L., 2017. The plastic in microplastics: a review. Mar. Pollut. Bull. 119, 12–22.
Arena, G., Copat, C., Dimartino, A., Grasso, A., Fallico, R., Sciacca, S., Fiore, M., Ferrante, M., 2015. Determination of total vanadium and vanadium (V) in groundwater from mt. Etna and estimate of daily intake of vanadium (V) through drinking water. J. Water Health 13, 522–530.
Avella, M., Errico, M.E., Martuscelli, E., 2001. Novel PMMA/CaCO3 nanocomposites abrasion resistant prepared by an in situ polymerization process. Nano Lett. 1, 213–217.
Baldwin, A.K., Corsi, S.R., Mason, S.A., 2016. Plastic debris in 29 Great Lakes tributaries: relations to watershed attributes and hydrology. Environ. Sci. Technol. 50, 10377–10385.
Barboza, L.G.A., Cózar, A., Gimenez, B.C.G., Barros, T.L., Kershaw, P.J., Guilhermino, L., 2019. Macroplastics pollution in the marine environment. World Seas: An Environmental Evaluation. Elsevier, pp. 305–328.
Barnes, D.K.A., Galgani, F., Thompson, R.C., Barlaz, M., 2009. Accumulation and fragmentation of plastic debris in global environments. Philos. Trans. R. Soc. B Biol. Sci. 364, 1985–1998. https://doi.org/10.1098/rstb.2008.0205.
Bashir, S.M., Kimiko, S., Mak, C.-W., Fang, J.K.-H., Gonçalves, D., 2021. Personal care and cosmetic products as a potential source of environmental contamination by microplastics in a densely populated Asian city. Front. Mar. Sci. 8, 604.
Bayo, J., Rojo, D., Olmos, S., 2019. Abundance, morphology and chemical composition of microplastics in sand and sediments from a protected coastal area: the mar menor lagoon (SE Spain). Environ. Pollut. 252, 1357–1366.
Bessa, F., Barría, P., Neto, J.M., Frias, J.P.G.L., Otero, V., Sobral, P., Marques, J.C., 2018. Occurrence of microplastics in commercial fish from a natural estuarine environment. Mar. Pollut. Bull. 128, 575–584.
Cheung, P.K., Fok, L., 2016. Evidence of microbeads from personal care product contaminating the sea. Mar. Pollut. Bull. 109, 582–585.
Cosgriff, E., Mantell, S., Bhattacharya, M., 2017. Method for degrading polyethylene sheet samples in an oxidative environment. in: 75th Annual Technical Conference and Exhibition of the Society of Plastics Engineers, SPE ANTEC Anaheim 2017. Society of Plastics Engineers, pp. 1228–1233.
da Costa, J.P., Santos, P.S.M., Duarte, A.C., Rocha-Santos, T., 2016. (Nano) plastics in the environment–sources, fates and effects. Sci. Total Environ. 566, 15–26.
Dalmau-Soler, J., Ballesteros-Cano, R., Boleda, M.R., Paraira, M., Ferrer, N., Lacorte, S., 2021. Microplastics from headwaters to tap water: occurrence and removal in a drinking water treatment plant in Barcelona metropolitan area (Catalonia, NE Spain). Environ. Sci. Pollut. Res. 1–11.
Dalmau-Soler, J., Ballesteros-Cano, R., Ferrer, N., Boleda, M.R., Lacorte, S., 2022. Microplastics throughout a tap water supply network. Water Environ. J. 36, 292–298.
De Witte, B., Devriese, L., Bekaert, K., Hoffman, S., Vandermeersch, G., Cooreman, K., Robbens, J., 2014. Quality assessment of the blue mussel (Mytilus edulis): comparison between commercial and wild types. Mar. Pollut. Bull. 85, 146–155.
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, 1–10.
Devriese, L.I., van der Meulen, M.D., Maes, T., Bekaert, K., Paul-Pont, I., Frère, L., Robbens, J., Vethaak, A.D., 2015. Microplastic contamination in brown shrimp (Crangon crangon, linnaeus 1758) from coastal waters of the southern North Sea and channel area. Mar. Pollut. Bull. 98, 179–187.
Ding, N., An, D., Yin, X., Sun, Y., 2020. Detection and evaluation of microbeads and other microplastics in wastewater treatment plant samples. Environ. Sci. Pollut. Res. 27, 15878–15887.
Dris, R., Gasperi, J., Mirande, C., Mandin, C., Guerrouache, M., Langlois, V., Tassin, B., 2017. A first overview of textile fibers, including microplastics, in indoor and outdoor environments. Environ. Pollut. 221, 453–458. https://doi.org/10.1016/j.envpol.2016.12.013.
Eerkes-Medrano, D., Leslie, H.A., Quinn, B., 2019. Microplastics in drinking water: a review and assessment. Curr. Opin. Environ. Sci. Heal. 7, 69–75.
EPA, N., 2016. Plastic microbeads in products and the environment. Environ. Prot. Auth. 1–12.
Facts, P.E.P., 2019. An Analysis of European Plastics Production, Demand and Waste Data.
Fahim, I., Mohsen, O., ElKayaly, D., 2021. Production of fuel from plastic waste: a feasible business. Polymers (Basel). 13, 915.
Feldman, D., 2002. Polymer weathering: photo-oxidation. J. Polym. Environ. 10, 163–173.
Fischer, J., Bradler, P.R., Lang, R.W., Wallner, G.M., 2016. Fatigue crack growth resistance of polypropylene in chlorinated water at different temperatures. Proceedings of the Plastic Pipes 18th Conference Proceedings, Berlin, Germany.
Fischer, J., Mantell, S.C., Bradler, P.R., Wallner, G.M., Lang, R.W., 2019. Effect of aging in hot chlorinated water on the mechanical behavior of polypropylene grades differing in their stabilizer systems. Mater. Today Proc. 10, 385–392.
Fujii, T., Matsui, Y., Hirabayashi, H., Igawa, K., Okada, S., Honma, H., Nishimura, H., Yamada, K., 2019. Influence of residual chlorine and pressure on degradation of polybutylene pipe. Polym. Degrad. Stab. 167, 1–9.
Gewert, B., Plassmann, M., Sandblom, O., MacLeod, M., 2018. Identification of chain scission products released to water by plastic exposed to ultraviolet light. Environ. Sci. Technol. Lett. 5, 272–276.
Ghayebzadeh, M., Aslani, H., Taghipour, H., Mousavi, S., 2020. Contamination of the Caspian Sea southern coast sediments with microplastics: a marine environmental problem. Mar. Pollut. Bull. 160. https://doi.org/10.1016/j.marpolbul.2020.111620.
Ghayebzadeh, M., Taghipour, H., Aslani, H., 2021. Abundance and distribution of microplastics in the sediments of the estuary of seventeen rivers: Caspian southern coasts. Mar. Pollut. Bull. 164. https://doi.org/10.1016/j.marpolbul.2021.112044.
Ghayebzadeh, M., Taghipour, H., Aslani, H., Hoseini, G., Farahani, V., 2023. Abundance and distribution of mesoplastics in the sediment of the southern coast of the Caspian Sea. Reg. Stud. Mar. Sci. 57, 102767.
Isobe, A., 2016. Percentage of microbeads in pelagic microplastics within japanese coastal waters. Mar. Pollut. Bull. 110, 432–437.
Jaafar, N., Azfaralariff, A., Musa, S.M., Mohamed, M., Yusoff, A.H., Lazim, A.M., 2021. Occurrence, distribution and characteristics of microplastics in gastrointestinal tract and gills of commercial marine fish from Malaysia. Sci. Total Environ. 799, 149457.
Jambeck, J.R., Geyer, R., Wilcox, C., Siegler, T.R., Perryman, M., Andrady, A., Narayan, R., Law, K.L., 2015. Plastic waste inputs from land into the ocean. Science (80-.) 347, 768–771. https://doi.org/10.1126/science.1260352.
Jung, J.-W., Kim, S., Kim, Y.-S., Jeong, S., Lee, J., 2022. Tracing microplastics from raw water to drinking water treatment plants in Busan, South Korea. Sci. Total Environ. 825, 154015.
Kalčíková, G., Alič, B., Skalar, T., Bundschuh, M., Gotvajn, A.Ž., 2017. Wastewater treatment plant effluents as source of cosmetic polyethylene microbeads to freshwater. Chemosphere 188, 25–31.
Kershaw, P.J., Rochman, C.M., 2015. Sources, fate and effects of microplastics in the marine environment: part 2 of a global assessment. Reports Stud. Jt. Gr. Expert. Sci. Asp. Mar. Environ. Prot. Eng No. 93.
Kirstein, I.V., Hensel, F., Gomiero, A., Iordachescu, L., Vianello, A., Wittgren, H.B., Vollertsen, J., 2021. Drinking plastics?–Quantification and qualification of microplastics in drinking water distribution systems by μFTIR and py-GCMS. Water Res. 188, 116519.
Koelmans, A.A., Nor, N.H.M., 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.
Koelmans, A.A., Redondo-Hasselerharm, P.E., Nor, N.H.M., de Ruijter, V.N., Mintenig, S.M., Kooi, M., 2022. Risk assessment of microplastic particles. Nat. Rev. Mater. 7, 138–152.
Kosuth, M., Mason, S.A., Wattenberg, E.V., 2018. Anthropogenic contamination of tap water, beer, and sea salt. PLoS One 13, e0194970.
Lee, J., Chae, K.-J., 2021. A systematic protocol of microplastics analysis from their identification to quantification in water environment: a comprehensive review. J. Hazard. Mater. 403, 124049.

Lee, J., Lee, J.S., Jang, Y.C., Hong, S.Y., Shim, W.J., Song, Y.K., Hong, S.H., Jang, M., Han, G.M., Kang, D., Hong, S., 2015. Distribution and size relationships of plastic marine debris on beaches in South Korea. Arch. Environ. Contam. Toxicol. 69, 288–298. https://doi.org/10.1007/s00244-015-0208-x.

Leslie, H.A., Brandsma, S.H., Van Velzen, M.J.M., Vethaak, A.D., 2017. Microplastics en route: field measurements in the dutch river delta and Amsterdam canals, wastewater treatment plants, North Sea sediments and biota. Environ. Int. 101, 133–142.

Leslie, H.A., Van Velzen, M.J., Brandsma, S.H., Vethaak, A.D., Garcia-Vallejo, J.J., Lamoree, M.H., 2022. Discovery and quantification of plastic particle pollution in human blood. Environ. Int. 163, 107199.

Li, J., Liu, H., Chen, J.P., 2018. Microplastics in freshwater systems: a review on occurrence, environmental effects, and methods for microplastics detection. Water Res. 137, 362–374.

Lia, J., Greenc, C., Reynoldsd, A., Shia, H., Rotchellb, J.M., 2018. Microplastics in Mussels Sampled from Coastal Waters and Supermarkets in the United Kingdom. 2.

Lithner, D., Damberg, J., Dave, G., Larsson, Å., 2009. Leachates from plastic consumer products–screening for toxicity with Daphnia magna. Chemosphere 74, 1195–1200.

Lithner, D., Larsson, Å., Dave, G., 2011. Environmental and health hazard ranking and assessment of plastic polymers based on chemical composition. Sci. Total Environ. 409, 3309–3324.

Lorenz, C., Roscher, L., Meyer, M.S., Hildebrandt, L., Prume, J., Löder, M.G.J., Primpke, S., Gerdts, G., 2019. Spatial distribution of microplastics in sediments and surface waters of the southern North Sea. Environ. Pollut. 252, 1719–1729.

Lusher, A.L., Welden, N.A., Sobral, P., Cole, Mja, 2020. Sampling, isolating and identifying microplastics ingested by fish and invertebrates. Analysis of Nanoplastics and Microplastics in Food. CRC Press, pp. 119–148.

Makhdoumi, P., Amin, A.A., Karimi, H., Pirsaheb, M., Kim, H., Hossini, H., 2021. Occurrence of microplastic particles in the most popular Iranian bottled mineral water brands and an assessment of human exposure. J. Water Process Eng. 39, 101708.

Marsden, P., Koelmans, A.A., Bourdon-Lacombe, J., Gouin, T., D'Anglada, L., Cunliffe, D., Jarvis, P., Fawell, J., De France, J., 2019. Microplastics in drinking water. World Health Organization 1, 1–124.

Mason, S.A., Garneau, D., Sutton, R., Chu, Y., Ehmann, K., Barnes, J., Fink, P., Papazissimos, D., Rogers, D.L., 2016. Microplastic pollution is widely detected in US municipal wastewater treatment plant effluent. Environ. Pollut. 218, 1045–1054.

Mehdinia, A., Dehbandi, R., Hamzehpour, A., Rahnama, R., 2020. Identification of microplastics in the sediments of southern coasts of the Caspian Sea, north of Iran. Environ. Pollut. 258, 113738.

Mintenig, S.M., Int-Veen, I., Löder, M.G.J., Primpke, S., Gerdts, G., 2017. Identification of microplastic in effluents of waste water treatment plants using focal plane array-based micro-Fourier-transform infrared imaging. Water Res. 108, 365–372.

Mintenig, S.M., Löder, 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.

Miranda, T., Vieira, L.R., Guilhermino, L., 2019. Neurotoxicity, behavior, and lethal effects of cadmium, microplastics, and their mixtures on Pomatoschistus microps juveniles from two wild populations exposed under laboratory conditions—implications to environmental and human risk assessment. Int. J. Environ. Res. Public Health 16, 2857.

Mohana, A.A., Farhad, S.M., Haque, N., Pramanik, B.K., 2021. Understanding the fate of nanoplastics in wastewater treatment plants and their removal using membrane processes. Chemosphere 284, 131430.

Möhlenkamp, P., Purser, A., Thomsen, L., 2018. Plastic microbeads from cosmetic products: an experimental study of their hydrodynamic behaviour, vertical transport and resuspension in phytoplankton and sediment aggregates. Elem. Sci. Anth. 6, 61.

Oßmann, B.E., Sarau, G., Holtmannspötter, H., Pischetsrieder, M., Christiansen, S.H., Dicke, W., 2018. Small-sized microplastics and pigmented particles in bottled mineral water. Water Res. 141, 307–316.

Pérez-Guevara, F., Roy, P.D., Elizalde-Martínez, I., Kutralam-Muniasamy, G., Shruti, V.C., 2022a. Human exposure to microplastics from urban decentralized pay-to-fetch drinking-water refill kiosks. Sci. Total Environ. 848, 157722.

Pérez-Guevara, F., Roy, P.D., Kutralam-Muniasamy, G., Shruti, V.C., 2022b. Coverage of microplastic data underreporting and progress toward standardization. Sci. Total Environ. 829, 154727.

Pirsaheb, M., Hossini, H., Makhdoumi, P., 2020. Review of microplastic occurrence and toxicological effects in marine environment: experimental evidence of inflammation. Process Saf. Environ. Prot. 142, 1–14.

Pivokonsky, M., Cermakova, L., Novotna, K., Peer, P., Cajthaml, T., Janda, V., 2018. Occurrence of microplastics in raw and treated drinking water. Sci. Total Environ. 643, 1644–1651.

Prata, J.C., Castro, J.L., da Costa, J.P., Duarte, A.C., Cerqueira, M., Rocha-Santos, T., 2020. An easy method for processing and identification of natural and synthetic microfibers and microplastics in indoor and outdoor air. MethodsX 7, 100762.

Rajabi, M.R., Rohani, F., Ekhtessassi, M.R., Ghazanfarpoor, N., 2006. Morphodynamic anlysis of wind faces using spatial sand rose model in Zabol and zahak region. Iran. Desert 11, 45–51.

Rochman, C.M., 2015. The complex mixture, fate and toxicity of chemicals associated with plastic debris in the marine environment. Marine Anthropogenic Litter. Springer, Cham, pp. 117–140.

Schönlau, C., Karlsson, T.M., Rotander, A., Nilsson, H., Engwall, M., van Bavel, B., Kärrman, A., 2020. Microplastics in sea-surface waters surrounding Sweden sampled by manta trawl and in-situ pump. Mar. Pollut. Bull. 153, 111019.

Schymanski, D., Goldbeck, C., Humpf, H.-U., Fürst, 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.

Senathirajah, K., Attwood, S., Bhagwat, G., Carbery, M., Wilson, S., Palanisami, T., 2021. Estimation of the mass of microplastics ingested–a pivotal first step towards human health risk assessment. J. Hazard. Mater. 404, 124004.

Shahriar, A., Mozaffari, G.A., Poudineh, S., 2018. The impact of drought periods and wind erosion on the physical development of desert cities (Case study: Zabol-Iran). Desert 23, 199–209.

Shahul Hamid, F., Bhatti, M.S., Anuar, Norkhairiyah, Anuar, Norkhairah, Mohan, P., Periathamby, A., 2018. Worldwide distribution and abundance of microplastic: how dire is the situation? Waste managRes. 36, 873–897.

Sharma, S., Chatterjee, S., 2017. Microplastic pollution, a threat to marine ecosystem and human health: a short review. Environ. Sci. Pollut. Res. 24, 21530–21547.

Shim, W.J., Hong, S.H., Eo, S., 2018. Marine microplastics: abundance, distribution, and composition. Microplastic Contamination in Aquatic Environments. Elsevier, pp. 1–26 https://doi.org/10.1016/B978-0-12-813747-5.00001-1.

Shruti, V.C., Kutralam-Muniasamy, G., Pérez-Guevara, F., Roy, P.D., Elizalde-Martínez, I., 2022. Free, but not microplastic-free, drinking water from outdoor refill kiosks: a challenge and a wake-up call for urban management. Environ. Pollut. 309, 119800.

Sistan and Baluchestan Water and Wastewater Company, 2022. Report of deputy of operational affairs. Statistics and information of Sistan and Baluchistan Water and Wastewater Company. Zahedan, Iran.

Thompson, R.C., Swan, S.H., Moore, C.J., Vom Saal, F.S., 2009. Our plastic age. https://doi.org/10.1098/rstb.2009.0054.

Tiwari, M., Rathod, T.D., Ajmal, P.Y., Bhangare, R.C., Sahu, S.K., 2019. Distribution and characterization of microplastics in beach sand from three different Indian coastal environments. Mar. Pollut. Bull. 140, 262–273.

Tong, H., Jiang, Q., Hu, X., Zhong, X., 2020. Occurrence and identification of microplastics in tap water from China. Chemosphere 252, 126493.

Toumi, H., Abidli, S., Bejaoui, M., 2019. Microplastics in freshwater environment: the first evaluation in sediments from seven water streams surrounding the lagoon of Bizerte (Northern Tunisia). Environ. Sci. Pollut. Res. 26, 14673–14682.

Wang, J., Peng, J., Tan, Z., Gao, Y., Zhan, Z., Chen, Q., Cai, L., 2017. Microplastics in the surface sediments from the Beijiang River littoral zone: composition, abundance, surface textures and interaction with heavy metals. Chemosphere 171, 248–258.

Wang, Z., Lin, T., Chen, W., 2020. Occurrence and removal of microplastics in an advanced drinking water treatment plant (ADWTP). Sci. Total Environ. 700, 134520.

Weber, F., Kerpen, J., Wolff, S., Langer, R., Eschweiler, V., 2021. Investigation of microplastics contamination in drinking water of a German city. Sci. Total Environ. 755, 143421.

Whelton, A.J., Dietrich, A.M., Gallagher, D.L., 2011. Impact of chlorinated water exposure on contaminant transport and surface and bulk properties of high-density polyethylene and cross-linked polyethylene potable water pipes. J. Environ. Eng. 137, 559–568.

Wong, K.H.J., Lee, K.K., Tang, K.H.D., Yap, P.-S., 2020. Microplastics in the freshwater and terrestrial environments: Prevalence, fates, impacts and sustainable solutions. Sci. Total Environ. 137512.

Wypych, G., 2022. Handbook of Polymers. Elsevier.

Xu, Q., Xing, R., Sun, M., Gao, Y., An, L., 2020. Microplastics in sediments from an interconnected river-estuary region. Sci. Total Environ. 729, 139025.

Yurtsever, M., Yurtsever, U., 2019. Use of a convolutional neural network for the classification of microbeads in urban wastewater. Chemosphere 216, 271–280.

Zeng, E.Y., 2018. Microplastic Contamination in Aquatic Environments: An Emerging Matter of Environmental Urgency. Elsevier.

Zhang, Q., Zhao, Y., Du, F., Cai, H., Wang, G., Shi, H., 2020. Microplastic fallout in different indoor environments. Environ. Sci. Technol. 54, 6530–6539.

Zhang, X., Lin, T., Wang, X., 2022. Investigation of microplastics release behavior from ozone-exposed plastic pipe materials. Environ. Pollut. 296, 118758.

Zuccarello, P., Ferrante, M., Cristaldi, A., Copat, C., Grasso, A., Sangregorio, D., Fiore, M., Conti, G.O., 2019. Exposure to microplastics (<10 μm) associated to plastic bottles mineral water consumption: The first quantitative study. Water Res. 157, 365–371.