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Micro- and nanoplastics in drinking water: the review

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Abstract: Plastic pollution in water is becoming increasingly alarming due to the annual increase in global plastic production. Micro- and nanoplastics (MNPs) have been detected in drinking water sources and tap water, raising concerns about the effectiveness of drinking water treatment plants and the public health of people who consume tap or bottled water. This article presents a systematic review of the available knowledge on the global occurrence of MNPs in drinking water sources and treated water, tap water, and/or bottled water. Data on the occurrence and properties of MNPs (polymer type, shape, and size) are presented. It also discusses the effectiveness of MNPs removal in various unit processes used for drinking water treatment, including coagulation, flocculation, sedimentation, sand and membrane filtration, adsorption, advanced oxidation processes, and disinfection. Attention is also drawn to the health risks posed by MNPs present in drinking water. Currently, there still appear to be gaps in research on MNPs removal in drinking water treatment. This article therefore discusses the potential challenges, strategies, and research needs related to the occurrence and removal of MNPs in unit processes used in drinking water production.

Introduction

Global plastic production in 2024 amounted to 430,9 million tons, of which 15.4% was recycled plastic, and 1.5% was bio-based plastics (Plastics - The Fast Facts 2025, Plastics Europe). Due to their low cost, lightness, ease of processing, durability, flexibility, low weight, elasticity, corrosion resistance, high strength-to-weight ratio, and water resistance, plastics have replaced more traditional materials such as glass and metals (Kundu et al. 2021).

Plastic products in the environment undergo physical, chemical, and biological degradation into smaller fragments (Frias and Nash 2019). It is generally accepted that microplastics (MPs) are defined as plastic particles ranging in size from 5 mm to 1 µm, while particles smaller than 1 µm are usually referred to as nanoplastics (NPs) (Li et al. 2020a). In this study, the term "MNPs" is used as a collective term referring to both MPs and NPs, i.e., plastic particles smaller than 5 mm formed as a result of plastic degradation. Depending on their origin, MNPs are divided into primary and secondary types. Primary MNPs are usually manufactured in the form of small particles and are found in various products, primarily cosmetics (Mai et al. 2018). Secondary MNPs originate from larger plastic items, such as bags, bottles and tires, which have undergone the aforementioned degradation over time. MNPs can have various shapes, such as fibers, granules, or fragments (Akdogan and Guven 2019). Due to their high surface area and hydrophobic nature, both primary and secondary MNPs act as carriers of toxic pollutants present in the aquatic environment, such as heavy metals and endocrine-disrupting compounds (EDCs), including polycyclic aromatic hydrocarbons (PAHs), polybrominated diphenyl ethers (PBDEs), pharmaceutical pollutants, and others (Lim et al. 2022). The adsorption of these chemicals onto MNPs has adverse health effects.

MNPs have been found in aquatic environments, including rivers, lakes and oceans, as well as in freshwater, wastewater, tap water and bottled water, air, food products, and aquatic organisms (Paliulis et al. 2025, Tong et al. 2020, Mason et al. 2018). MNPs enter freshwater environments in many ways, mainly from surface runoff, industrial and municipal wastewater (both treated and untreated), as well as from degraded plastic waste and atmospheric deposition (Rajmohan et al. 2019). The presence of MNPs in freshwater and drinking water has become a subject of growing concern in recent years due to their potential impact on human health and the environment. Due to their small size, MNPs enter the human food chain through the consumption of various foods and beverages, such as table salt, beer, drinking water, seafood and others (Rist et al. 2018).

To date, only a limited number of studies have been conducted on the presence of MNPs in tap water, as well as in bottled water and beverages (Kosuth et al. 2018, Mintenig et al. 2019, 2020, Hart and Lenhart 2026, Rahman et al. 2026). Compared to the number of studies on the occurrence, characteristics, and removal efficiency of MNPs in wastewater treatment plants, the number of such studies conducted in drinking water treatment plants (DWTPs) is relatively limited. Since MNPs in drinking water pose a potential risk to human health, assessing their occurrence, characteristics (polymer type, size, and shape), and removal efficiency in DWTPs unit processes will help determine the current status and gaps in MNP removal in DWTPs, as well as support the development of new treatment strategies. For this reason, this analysis first identifies the sources of MNPs in surface and groundwater, followed by a review of the occurrence, properties, and removal efficiency of MNPs in DWTPs located in different parts of the world. Next, studies presenting the presence and characteristics of MNPs in tap water and bottled water are reviewed, and the impact of MNPs in drinking water on human health is analyzed. Finally, gaps in research on MNP in drinking water and recommendations for future research are presented.

MNPs in waters feeding DWTPs

Water from rivers, lakes, reservoirs, groundwater, and the sea is the main source of water for agriculture, industry, and human consumption (Ezugbe and Rathilal 2020). Natural waters are polluted by industry and agriculture, causing MNPs to enter the water sources that supply DWTPs. In addition, MNPs enter surface waters through the abrasion of plastic products, improperly managed plastic waste, and other sources (Alimi et al. 2018). Sources of MNPs in groundwater include leachate from solid waste landfills, release of MNPs accumulated in soil through irrigation and rainfall, interactions with surface water, and leaks from drainage systems (Selvam et al. 2021). In general, groundwater contains less MNPs than surface water due to less anthropogenic impact (Mintening et al. 2019). Wastewater treatment plants (WWTPs) effectively remove significant amounts of MNPs from the aqueous phase (72%–98%); however, due to the large loads of MNPs entering WWTPs, treated wastewater still contains considerable amounts, and these contaminants accumulate in sewage sludge (Murphy et al. 2016, Bodzek et al. 2024). In turn, the presence of large amounts of MNPs in sewage sludge, which is often applied in agriculture, contributes to the contamination of both surface and groundwater (Leslie et al. 2017).

The dominant polymer type in the feed water of DWTPs is polyethylene terephthalate (PET) (55-68%), followed by polyethylene (PE) and polypropylene (PP), as well as polyvinyl chloride (PVC), nylon, cellulose acetate (CA), polyester (PES), and polystyrene (PS) (Pivokonsky et.al. 2018, 2020, Wang et al. 2020a, Sarkar et al.2021, Hendrickson et al.2018, Shen et al. 2021). The most common shapes of MNPs in the water supplied to DWTPs are fibers, spheres, fragments, and films, with fragments being the dominant shape. MNP fragments in water originate mainly from the degradation of packaging, cosmetics, and cleaning products (Xue et al. 2022), while fibers come from plastic textiles and spherical MNPs primarily originate from personal hygiene products and cosmetics (Bodzek et al. 2024, Cai et al. 2020, Pivokonsky et al. 2020, Wang et al. 2020a). Studies of MNP shapes conducted by Pivokonsky et al. (2020) showed that fragments accounted for 71%–76% of all MNPs in reservoir waters, while raw river water contained lower proportions (42%–48%), although the proportion of fibers was also significant (37%–61%). Studies have shown that the concentration of MNPs in the water supplying DWTPs can vary significantly, from approximately 0 to several thousand particles/L (Koelmans et al. 2019, Lenaker et al. 2020). These differences depend on the type of water source, wastewater discharge from WWTPs, runoff from land or roads, the presence of plastic waste, as well as methodological conditions, population density and human activity in areas adjacent to water bodies.

The analytical methodology for determining MNPs is also of great importance, as it involves the use of different methods, equipment, detection limits, and sample volumes (Lv et al. 2019). Moreover, in many studies, blank samples are not analyzed in parallel, which would allow assessment of methodological limitations and the risk of cross-contamination during sample collection, preparation, and analysis. Consequently, the results of studies on MNPs in DWTPs are not directly comparable, and these limitations should be taken into account when interpreting and comparing the results. MNPs also vary in size. Previous studies of raw and treated water in DWTPs, as well as during treatment, have mainly focused on particles larger than 1 μm due to the limitations of the analytical methods. Some studies quantify the number of MPs larger than 1 μm in drinking water sources (Wang et al. 2020a, Pivokonsky et al. 2020, Shen et al. 2021, Adib et al. 2021, Mintening et al. 2019, Dronjak et al. 2022). According to Wang et al. (2020a), 54.6%-58.0% of MNPs detected in DWTP feedwater were in the size range of 1-5 μm, and 20.0%-27.6% were in the range of 5-10 μm, while Pivokonsky et al. (2018) reported that MNPs smaller than 10 μm predominated in raw water across all DWTPs studied. Other studies have shown that smaller particles, i.e., NPs, dominate in raw water supplying DWTPs (Pivokonsky et.al. 2018, 2020, Wang et al. 2020a, Sarkar et al. 2021). In terms of size, these particles are similar to colloids and are therefore difficult to detect using current analytical methods (Cai et al.2020). Table 1 summarizes the abundance and size of MNPs in selected freshwater environments worldwide.

Future studies on the concentration, shape and polymer types of MNPs in drinking water sources should include a comprehensive quantitative analysis of not only MPs larger than 1 μm but also smaller ones, i.e., NPs.

Removal of MNPs during water treatment in DWTPs

MNPs are found in both surface and groundwater; therefore understanding the effectiveness of water treatment in DWTPs is important due to the impact of these contaminants on human health (Kumar et al. 2023). The degree of MNP removal should be assessed at the inlet and outlet of DWTPs, as well as in individual unit processes. In addition, the effectiveness of WWTPs is important, as wastewater is the main source of MNPs in surface waters (Bodzek et al. 2024). This chapter reviews the effectiveness of individual processes used in the production of drinking water.

One of the key roles of drinking water treatment is to remove or reduce particles content; therefore, treatment processes should also be effective in removing MNPs from water, especially since they have properties similar to many natural particles, such as size, density, and surface charge (Mason et al. 2018, Murphy et al. 2016). To date, no dedicated techniques for removing MNPs have been developed. Conventional water treatment methods primarily remove dissolved contaminants and, incidentally, also particulate matter such as MNPs. However, the effectiveness of these processes with respect to MNPs has not yet been sufficiently investigated. Moreover, no standards have been established to define permissible levels of

Table 1: Concentration of MNPs in selected freshwater environments around the world

Location	Concentration (particles/L)	References
Lake Dongting, China	<1-2.8	Wang et al. 2018a
Lake Taihu, China	$3.4\text{-}2.6\cdot10^1$	Su et al. 2016
Lake Poyang, China	$5\text{-}3.4\cdot10^1$	Yuan et al. 2019
Yangtze River, China	$5\cdot10^{-1}\text{-}3.1$	Su et al. 2018
Suzhou and Huangpu Rivers, China	1.8-2.4	Luo et al. 2019
Hong Lake, China	1.3-4.7; mean: 2.9	Wang et al. 2018a
Urban lakes, Hanjiang and Wuhan Rivers, China	1.7-8.9	Wang et al. 2016
Three Gorges Reservoir, China	$1.6\text{-}1.3\cdot10^1$; mean: 4.7	Di and Wang 2018
River, France	$7.8 – 4.5\cdot10^1$	Barbier et al. 2022
Seine River, France	$3\cdot10^{-1}\text{-}5\cdot10^{-1}$; mean: $4\cdot10^{-1}$	Dris et al. 2015
Seine River, France	$3\text{-}10^6$; mean: $3.0\cdot10^1$	Dris et al. 2015
Groundwater, Germany	$0\text{-}7\cdot10^{-3}$; mean: $7\cdot10^{-4}$	Kosuth et al. 2018
Rivers, Germany	<1	Triebskorn et al. 2019
Western Lake Superior, USA	mean: $2.6\cdot10^{-4}$	Hendrickson et al. 2018
29 Great Lakes, USA	<1	Baldwin et al. 2016
Amsterdam Canal, Netherlands	$4.8\cdot10^1\text{-}1.5\cdot10^2$	Leslie et al. 2017
Lake, Switzerland	$3.8\cdot10^{-2}\pm1.5\cdot10^{-2}$	Velasco et al. 2022
River, Spain	4.2±1.2	Dronjak et al. 2022
Surface water, Hungary	<1	Bordos et al. 2019
Surface water, Czech Republic	$3.1\cdot10^3\text{-}4.5\cdot10^3$; mean: $3.1\cdot10^3$	Pivokonsky et al. 2018
Danube River, Austria	<1	Bergmann et al. 2015
River, lake, South Korea	2.1-2.4	Jung et al. 2022
River, India	$1.8\cdot10^1$	Sarkar et al. 2021
River, Thailand	$1.4\cdot10^3\text{-}1.8\cdot10^3$	Kankanige and Babel 2021

these contaminants in drinking water (Bodzek and Pohl 2022). As a result, these contaminants can be found in drinking water, potentially affecting human health.

The treatment of drinking water in DWTPs most often involves coagulation, flocculation, sedimentation, sand filtration, and disinfection, with MNP removal efficiencies ranging from 69.9% to 100% (Velasco et al. 2022, Jung et al. 2022, Acarer 2023a, Cherniak et al. 2022, Chu et al. 2022, Dalmau-Soler et al. 2021, Romphophak et al. 2024). The most commonly used system consists of coagulation/flocculation and sand filtration (Fig. 1a), with an efficiency of 40%-70% (Privokonsky et al. 2018, 2020), while systems additionally including ozonation and granular activated carbon (GAC) filtration (Fig. 1b) achieve MNP removal efficiencies of >80% (Privokonsky et al. 2018, 2020, Wang et al. 2020a). The process consisting of coagulation/flocculation, pulsed clarifier, and sand and membrane filtration (Fig. 1c), is also characterized by high MNP removal, reaching 84.6% (Sarkar et al. 2021). A

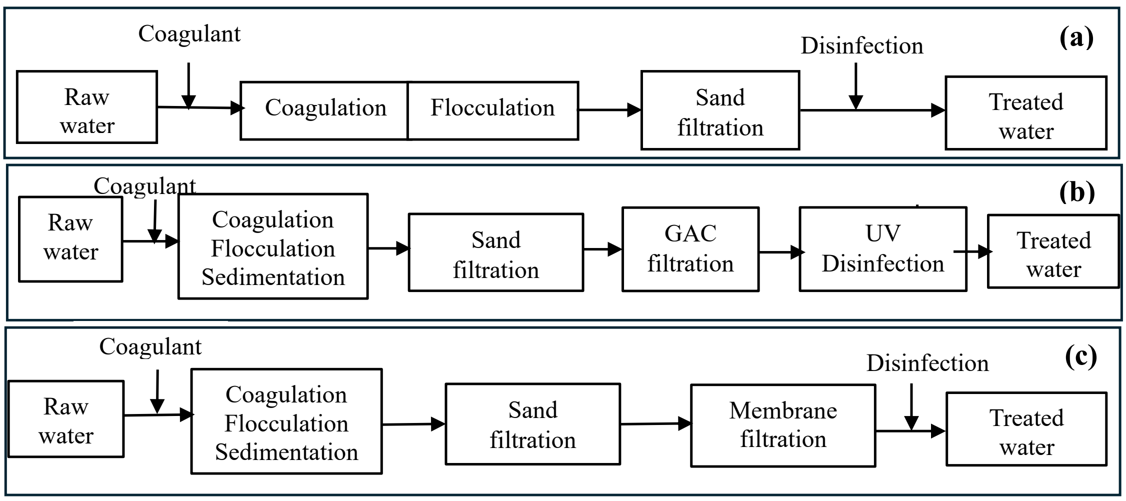

Fig. 1. Technological schemes used in DWTPs: (a) conventional scheme, (b) advanced scheme with GAC filtration and disinfection, and (c) advanced scheme with membrane filtration and disinfection, own work based on (Privokonsky et al. 2018, 2020)

Table 2: Comparison of relevant parameters and effectiveness of MPs removal from various DWTPs

Name of DWTP	Raw water	Purification processes	Efficiency,%	References
Milence, Czech	The Úhlava River	Flocculation and sand filtration	40.0	Privokonsky et al. 2020
Pilsen, Czech	The Úhlava River	Coagulation/flocculation, sedimentation, sand filtration, ozonation, and GAC filtration	88.0	Privokonsky et al. 2020
DWTP1, Czech	Large tank	Flocculation and sand filtration	70.0	Privokonsky et al. 2018
DWTP2, Czech	Small tank	Coagulation/flocculation, sedimentation, sand filtration, and GAC filtration.	81.0	Privokonsky et al. 2018
Sant Joan Despí, Spain	The Llobregat River	Coagulation/flocculation, sedimentation, sand filtration, ozonation, GAC filtration, and reverse osmosis	93.0	Dalmau-Soler et al. 2021
Barcelona, Spain	The Llobregat River	Coagulation/flocculation, sedimentation tanks, sand filters, carbon filters, reverse electrodialysis	98.3	Dronjak et al. 2022
Indira Gandhi, India	The Ganges River	Preliminary disinfection, coagulation/ flocculation, impulse clarification, sand filtration, and final disinfection	84.6	Sarkar et al. 2021
4 drinking DWTPs, Korea	The Nakdong River and two lakes	Preliminary ozonation, coagulation, sedimentation, sand filtration	>90.0	Jung et al. 2022
China	Yangtze River	Coagulation/flocculation, sedimentation, sand filtration, ozonation, GAC filtration	82.1–88.6	Wang et al. 2020a
DWTP in Zhengzhou, China	Yellow River	Coagulation/flocculation, sedimentation, sand filtration, GAC filtration, chlorination	69.4-74.5	Li et al. 2024a

disadvantage of some water treatment processes is the potential formation of MNPs in treated water, particularly during coagulation and flocculation using polymeric coagulants and flocculants, and in processes using polymeric membranes. For example, Wang et al. (2020a) and Dalmau-Soler et al. (2021) found that the coagulation/flocculation/sedimentation process resulted in a 114% increase in MNP concentration, with the MNPs having a different chemical composition than those in the raw water. Polymeric membranes used in water treatment can also contribute to MNP formation in treated water (Bodzek and Bodzek 2025).

Table 2 presents data on the effectiveness of MNPs removal in various DWTPs, with the efficiency of the processes depending on the type of unit processes used. In most of the DWTPs studied, MNP removal exceeds 80.0%, indicating relatively high efficiency.

Coagulation/flocculation/sedimentation

Coagulation/flocculation is a commonly used method of drinking water treatment in DWTPs that removes solid particles and colloids in the form of flocs and other contaminants such as bacteria, heavy metal ions, PAHs, and pesticides, which are then removed through sedimentation/flotation and filtration processes (Katrivesis et al. 2019, Crittenden et al. 2012). The coagulation process is usually carried out using aluminum and iron salts, with the choice of coagulant depending on water quality and the desired treatment objectives (Poleneni et al. 2019). Coagulation is followed by flocculation using cationic and anionic polymers and natural products (Crittenden et al. 2012), during which destabilized particles and coagulation products aggregate into larger flocs that are more easily separated from water by sedimentation or filtration. The appropriate selection and dosage of coagulants and flocculants are critical to achieving optimal treatment performance.

Dissolved air flotation (DAF) is an effective alternative for removing MNPs with lower density than water and other suspensions. A modified DAF process allows for the removal of 100% of MPs larger than 15 μm (Wang et al. 2021a). Pivokonsky et al. (2018) investigated the occurrence of MPs >1 mm in raw and treated water from full-scale DWTPs supplied by rivers and surface water reservoirs in the Czech Republic. The content of MPs in treated water was significantly reduced compared to raw water, with removal efficiencies of 70% for coagulation/flocculation and sand filtration, 81% for systems additionally equipped with activated carbon filtration, and 83% for a systems combining coagulation/flocculation, flotation, sand filtration, and activated carbon filtration.

Table 3 shows the effectiveness of MNP removal using various unit processes, primarily coagulation/flocculation/ sedimentation in full-scale DWTPs and laboratory-scale studies. Despite differences in MNP concentration and composition in raw water, the studies show comparable MNP removal efficiencies (40.4%-61.5%) in the coagulation/ flocculation process, despite differences in several factors such as raw water characteristics, MNP composition, coagulant type and dose, and operating conditions.

In DWTPs, aluminum chloride ($AlCl_3{\cdot}6H_2O$), ferric chloride ($FeCl_3{\cdot}6H_2O$) and aluminum polychloride (PACl) are used to remove MNPs during the coagulation process (Xue et al. 2022). Comparative studies of coagulants have shown

Table 3: Removal of MNPs by coagulation/flocculation (CF) and sedimentation (S) DWTPs tests

Water type, Country	Treatment stage	MPs type, concentration	Size range and concentration of MNPs	Removal (%)	Reference
Full-scale research					
India, River water	CF-pulsed tank	17.9 particle/L	<100 μm	60.9	Sarkar et al. 2021
Czech, River water	CFS	23 particle/L	1 –100 μm	61.5	Privokonsky et al. 2020
China, River water	CFS	6614 particle/L	1 – >100 μm	40.4-54.5[2)]	Wang et al. 2020a
Laboratory tests					
China, Simulated water	CFS	PS PE	- <5 mm	65-80[1)] 16-30[1)]	Zhou et al.2022
China, Simulated water	CFS		1-5 μm (balls) 5 μm (fibres)	98	Xia et al. 2020
China, Simulated water	CFS	PET	100-500 μm	69.8-79.35 [2)]	Zhang et al.2021
China, Simulated water	CFS	PET	0,5 μm	90-100	Lu et al. 2021
China, Simulated water	CFS	PET	<0,5-5 mm	2.4-13.5[1)] 17.7-61.3[1)]	Ma et al. 2019a
Singapore, Simulated water	CFS	PS	-	>80	Li et al. 2021
Canada, Lake water	CFS	PS	3, 6, 25, 45 and 90 μm	63.7–96.1[1)]	Xue et al. 2021
Canada, River water	CF	PE PS	15 and 140 μm 140 μm	90-92 83.7	Lapointe et al.2020
Korea, Simulated water	CFS	PE	10-100 μm	52[1)] 71[1)]	Shahi et al. 2020
Korea, Simulated water	CF–membrane filtration	PS	0.5 and 90 μm	97	Park et al. 2021
Switzerland, Simulated water	CFS	PS	500 nm	98.2	Peydayesh et al. 2021
UK, Simulated water	Carbon filter Sand filter	PS	10 μm	>95 60-80	Wang et al. 2020a
UK, Simulated water	Ultrafiltration	PE	13-690 nm	54	Enfrin et al. 2020
UK, River and groundwater	CFS	PA	63-90 μm	99.99	Jonson et al. 2020
Greece, Simulated water	CFS	PS, PE	53-500 μm	92.4	Arvaniti et al. 2021
Spain, Lake water	CFS	PP, PEST, PS, PAN, PE	20-500 μm	>90	Dalmau-Soler et al. 2021
South Korea, water	CF	PE PET	40–48 μm 300 μm	>80	Lee et al. 2023
China, tap water	CF	PS	10–1000 μm	48-97	Li et al. 2024b
Spain, tap water	CF	213 particle/L	<500 μm	43	Casella et al. 2025
Brazil, tap water	CF	PVC	<106 μm	78.2-99.6	Godoy et al. 2025

CFS – coagulation/flocculation/sedimentation, 1) depending on the size of the MNPs particles, 2) depending on the type and concentration of the coagulant

that PACl is the most effective in removing MNPs from PS and PE, followed by aluminum-based coagulants (Zhou et al. 2022, Ma et al. 2019b). These differences are related to hydrophobic and electrostatic interactions between MNPs and post-coagulation flocs. In addition, increased MNP removal efficiency is achieved through the addition of flocculants, e.g., polyacrylamide (PAM) (Ma et al. 2019b, Ma and You 2021, Tabatabaei et al. 2022), as well as through the additional use of ultrafiltration (Bodzek and Bodzek 2025). Despite the high efficiency of traditional coagulants in removing MNPs, new coagulants are being developed that are more cost-effective and based on natural and environmentally friendly materials, including polymer coagulants derived from starch, chitosan, or cellulose/lignin (Peydayesh et al. 2021, Hu et al. 2022, Pivokonsky et al. 2024).

The effectiveness of MNP removal in the CFS process depends on many factors, including coagulant dose, polymer type, and particle size and shape, as well as aqueous environment conditions such as pH, mixing speed, and sedimentation time.

The dose of coagulants used in the coagulation process in DWTPs is typically <20 mg/L in terms of Al and Fe (Ma et al. 2019b, 2021), with all types of coagulants initially showing an increasing and then stabilizing trend in MNP removal as the coagulant dose increases (Shahi et al. 2020, Tang et al. 2022, Xue et al. 2021, Lapointe et al.2020). High coagulant doses cause flocs to become loose and prone to breakage (Zhou et al. 2022), lower pH, increase chemical consumption, sludge production, and DWTP operating costs, and increase the risk of corrosion in water distribution systems (Sheng et al. 2022).

The effectiveness of MNP removal in the CFS process also depends on particle shape (Li et al. 2020a, Zhang et al. 2020a), with fibers generally being more effectively removed than other shapes. Such a phenomenon can be explained by the easier formation and deposition of coagulation flocs and the fact that fibrous MNPs are usually higher-density polymers (Katrivesis et al. 2019). The size of MNPs also affects their removal in the CFS process, with contradictory results. Shahi et al. (2020) found that smaller MNPs (10-30 μm) are more difficult to remove than larger ones (30-100 μm), while Xue et al. (2021) and Lapointe et al. (2020) observed lower removal of larger MNPs (140 μm) than smaller ones (15 μm). Very small MNPs (<10 μm) show low removal efficiency in the CFS process due to weak attachment to flocs and poor sedimentation (Adib et al. 2021, Na et al. 2021).

MNP removal in CFS is influenced not only by particle characteristics but also by the properties of the aquatic environment. Lapointe et al. (2020) found that the removal efficiencies of PES (99%) and PE (82%) fibers were consistent with turbidity removal. Studies examining various parameters, including zeta potential, pH, natural organic matter (NOM) (including humic acid), and turbidity show that pH values above 7-9 lead to larger floc sizes and, consequently, greater MNP capture within flocs (Skaf et al. 2020, Ma et al. 2019b, Ma and You 2021). Zeta potential analyses show that increasing pH reduces zeta potential, reflecting decreased particle stability at higher pH (Romphophak et al. 2024). The presence of NOM affects MNP removal during coagulation and flocculation, as it can either inhibit or enhance aggregation and sedimentation processes (Xue et al. 2022, Na et al. 2021).

Overall, further research is needed to improve the efficiency of removing different types of MPs using CFS, taking into account the properties of MNPs commonly found in DWTPs and the conditions under which the process is carried out. CFS is a promising approach for MNP removal and warrants further investigation; however, experimental setups should more closely reflect full-scale drinking water treatment conditions.

Electrocoagulation (EC) is an advanced water treatment process that involves generating coagulants (Fe^{3+} or Al^{3+} ions) in situ via an electric current, which then interact with contaminants, including MNPs, to form flocs that can be removed by sedimentation or filtration (Moussa et al. 2017, Shen et al. 2022). Studies conducted by Perren et al. (2018) have shown MNP removal efficiencies of over 90%, reaching 99.2% at pH 7.5. Pilco-Nuñez et al. (2024) investigated the removal of PE MPs (average particle size of 300-355 μm) from a synthetic solution using EC with aluminum electrodes and a current density of 20 A/m². The MPs removal efficiency was very high, reaching 98.04%. The advantages of MNP removal by electrocoagulation include the ability to remove MNPs across a wide range (0.1 - 100 μm) and the removal of other contaminants, such as heavy metals and various organic substances (Moussa et al. 2017). The limitations of electrocoagulation include the need for an external power supply and the potential production of sludge that is difficult to handle, which affects the overall process cost (Moussa et al. 2017).

Filtration

Sand filtration (SF), including rapid sand filtration (RSF), and granular activated carbon (GAC) filtration are commonly used in DWTPs to remove suspended solids and other contaminants, including MNPs. The principle of conventional filtration processes is to pass water through beds of sand, gravel, anthracite, or GAC (Romphophak et al. 2024, Chowdhury et al. 2023). The effectiveness of MNP removal is high in both in laboratory and full-scale studies. It depends on type, height, and pore size of the filter bed, the filtration rate, water quality (particle size and concentration), and operational parameters (pressure, backwashing, and bed regeneration) (Romphophak et al. 2024, Chowdhury et al. 2023). Studies have shown that SF can eliminate up to 99% of MNPs after coagulation and sedimentation (Babel and Dork 2021, Sarkar et al. 2021, Dalmau-Soler et al. 2021, Kankanige and Babel 2021, Dronjak et al. 2022). Pivokonsky et al. (2018, 2020) studied MNPs removal using coagulation-flocculation and sedimentation, as well as SF and RSF filtration at several DWTPs in the Czech Republic, achieving an overall MNP removal efficiency of 88%.

The effectiveness of MNP removal in SF and RSF is also influenced by particle shape and size. In CF and RSF processes, fiber-shaped MNPs are removed to a greater extent than other shapes (Wang et al. 2020a, Babel and Dork 2021). For example, Wang et al. (2020a) reported removal efficiencies of fiber-shaped MNPs in the range of 31–49%, spherical particles in the range of 24–51%, and fragments in the range of 19–28% after sedimentation and SF. Studies have shown that MNPs larger than the pore size of filter media are removed more effectively than smaller particles (Wang et al. 2020a, Kankanige and Babel 2021, Babel and Dork 2021). The elimination of small MNPs (e.g., 5–10 μm) in SF and RFS processes is challenging (Siegel et al. 2021) because they are removed only when attached to the grains of the filter layer. The mechanism of MNPs removal in filtration processes involves their retention on the filter surface or in the upper layers of the filter bed, while smaller particles may penetrate deeper into the lower layers. The low removal efficiency of small solids in SF suggests the need to optimize CF/CFS processes or implement an additional treatment. Nevertheless, SF and RSF are very effective in removing MNPs, especially those above 100 µm, although further research is needed on the influence of MNP morphology on their removal efficiency in SF processes.

The effectiveness of MNP removal is also influenced by the porosity of the filter bed (Talvitie et al. 2017, Sembiring et al. 2021, Fajar et al. 2022). A sand bed usually consists of several layers of sand with a specific particle size distribution, i.e., a layer of gravel (grain size 3 to 5 mm, height 1 m) and a layer of quartz (grain size 0.1 to 0.5 mm, remaining 0.5 m). In general, filter beds with lower porosity show higher MNP removal efficiencies (96.4% to 99.2%), while filter beds with higher porosity show lower efficiencies (90.5% to 96.5%) (Fajar et al. 2022). In DWTP studies conducted in Turku, Finland, the removal of MNPs from PE, PP, PS, and ethylene vinyl acetate (EVA) in an RSF installation exceeded 97% with a filter bed granulation of 0.39 mm. This efficiency dropped to 94% when the filter bed granulation increased to 0.68 mm (Talvitie et al. 2017, Sembiring et al. 2021).

In some advanced DWTPs, granular activated carbon (GAC) filtration is used as the final stage of water treatment, which is highly effective in removing MNPs of various sizes and shapes (Pivokonsky et al. 2018, Xu et al. 2020, Arenas et al. 2021). GAC can remove 74% of MNPs ranging in size from 1 to 5 µm and over 55% of larger MNPs. Thermally activated GAC increases the efficiency of removing larger particles (20-50 µm) to over 90% (Kim and Park 2021).

Membrane processes for removing MNPs

Pressure-driven membrane processes, i.e. microfiltration (MF), ultrafiltration (UF), nanofiltration (NF), and reverse osmosis (RO), are most commonly used in water and wastewater treatment (Ezughe et al. 2020, Bodzek 2019). The main disadvantage of membrane filtration is fouling, which occurs as a result of the adsorption of contaminants on the surface and within its pores (Bodzek 2019, Malankowska et al. 2021). UF, NF, and RO membranes have smaller pore sizes than MPs, but also smaller than NPs, which makes their removal efficiency superior to other technologies used in wastewater treatment and drinking water treatment (Malankowska et al. 2021, Ali et al. 2024).

The MF process uses symmetrical semi-permeable membranes with pore sizes of 0.05–10 µm, while the UF process mainly uses asymmetrical membranes with pore sizes of 1–100 nm (Bodzek 2019, Malankowska et al. 2021). Comparing the pore sizes of MF and UF membranes with the particle sizes of MPs (5 mm–1 µm) and NPs (< 1 µm), it is clear that the MF process can be used to remove MPs, and the UF process can eliminate both MPs and NPs (Xu and Wang 2020, Talvitie et al. 2017, Bodzek 2019, Malankowska et al. 2021). Many studies have analyzed the removal of MNPs in MF/UF processes in DWTPs at both laboratory scale (Ma et al. 2019a, 2019b, Li et al. 2021, 2020a) and industrial scale (Dalmau-Soler et al. 2021, Johnson et al. 2020). Most of these studies achieved complete removal because MNPs are larger than the membrane pore diameters (Table 4).

Pizzichetti et al. (2021) studied the efficiency and effectiveness of removing MPs from nylon 6 (PA) and PS using MF membranes made of polycarbonate, CA, and PTFE with a pore size of 5 µm. The removal efficiency of MNPs for all three membranes exceeded 94%, with CA proving to be the most effective membrane material. The main removal mechanism was sieve separation based on particle size, but

Table 4: Results of selected studies on the use of MF and UF for MNPs removal in full-scale DWTPs

Country,Water source	MNPs/sizes	Process type	Membrane type	Removal, %	Reference
China Synthetic water	MPs made of PE <0.5-5mm	CFS+UF	PVDF 100 KDa	100	Ma et al. 2019a, b
		UF		100	
		UF		100	
Korea Synthetic water	PS: 0.5 and 90 mm	Coagulation-UF	Filter paper: 11 mm	97	Park et al. 2021
Singapore Tap water, Raw water	PS: 0.1, 1, 10 and 18 mm	UF	Capillary fibres PVDF (0.03mm)	100	Li et al. 2021
		Coagulation -UF		100	
China Synthetic water	PVC (< 5 mm)	MBR[1)]	0.1 mm	99.9	Li et al. 2020b
UK Synthetic water	MPs from the EP	UF	Polysufone 30 kDa	100	Enfrin et al. 2020
	NPs - 13-690 nm			54	
Spain/ Ireland Synthetic water	PA 127,000 /L	MF (pore: 5 mm)	PC	99.6	Pizzichetti et al. 2021
	PS 33,000 /L			96.8	
Spain/ Ireland Synthetic water	PA 27 000/L	MF (pore: 5 mm)	Cellulose acetate	99.8	
	PS 8,000 /L			94.3	
Spain/ Ireland Synthetic water	PA 46,000 /L	MF (pore: 5 mm)	PTFE	99.6	
	PS 47,000 /L			96.0	

[1)] MBR - membrane bioreactor

membrane abrasion and fouling caused MPs to pass through the membrane to an extent dependent on membrane properties, MP-membrane interaction, particle size, and the applied transmembrane pressure (TMP).

Membrane processes, mainly UF or MF, are used in DWTPs after coagulation and flocculation, and often also after sedimentation (Xue et al. 2022). Ma et al. (2019a, 2019b) conducted studies on the removal of MNPs from PE in the drinking water treatment process using a UF membrane and coagulants $FeCl_3 \cdot 6H_2O$ and $AlCl_3 \cdot 6H_2O$. Because the density of PE is very close to that of water (0.92-0.97 g/cm^3), its removal by sedimentation or flotation is difficult. When introduced into the UF system, the overall PE removal efficiency was 86.14% due to the small membrane pore diameter. Studies have also shown that membrane fouling is low at conventional coagulant doses but increases with increasing coagulant dose due to the formation of a thicker sediment layer (Ma et al. 2019a, 2019b). The use of a flocculant (polyacrylamide, PAM) increased the removal efficiency of smaller MNP particles, improved process performance, and reduced fouling because the charges of the coagulant flocs are opposite to the charge of PAM (Ma et al. 2019a, 2019b).

Dalmau-Soler et al. (2021) used UF and RO, as well as ozonation and GAC filtration, in DWTPs in Spain to remove MNPs from contaminated groundwater and river water. The average concentration of MNPs in raw water was 0.96 particles/L (n=5), with PES and PP particles dominating, whereas the concentration in purified water was 0.06 particles/L, corresponding to an overall removal efficiency of 93%. The results showed that UF/RO (advanced treatment) is more effective in removing microplastic particles than ozonation and GAC filtration.

As already mentioned, membrane fouling has a significant impact on the effectiveness of MNP removal (Li et al. 2021, Enfrin et al. 2020, 2019), which directly results in a reduction in water flux and an increase in TMP. If MNPs are 0.1-10 times larger than the pore size of MF/UF membranes, they cause classic membrane fouling, while particles larger than 10 times the pore size form a filter cake (Enfrin et al. 2020, 2019). The intensity of fouling also depends on the concentration of MNPs, with higher concentrations leading to greater fouling (Wang et al. 2020b, Ma et al. 2019a, 2019b). Li et al. (2021) demonstrated that MNPs with a diameter of approximately 1 μm cause the greatest contamination among the tested MNPs with diameters of 0.1, 1, 10, and 18 μm, which may indicate that particle size is critical for membrane flux and removal efficiency. Luogo et al. (2022) compared the effectiveness of MF and UF ceramic membranes in removing MNPs and their impact on fouling. The studies showed significantly greater fouling of the MF membrane compared to the UF membrane, due to the adsorption of MNPs inside the MF membrane pores. Equally dangerous is biofouling, which occurs in natural surface waters as a result of the production of extracellular polymeric substances (Xu and Wang 2020). The effectiveness of MF and UF membranes therefore depends on controlling membrane fouling through effective membrane cleaning during operation or through pretreatment.

The main mechanism for removing MNPs through MF and UF membranes is size exclusion, i.e., particles larger than the pore size of the membrane are retained, while smaller ones pass through (Acarer 2023a). In addition, adsorption of MNPs on the membrane surface and within membrane pores, hydrophobic-hydrophilic interactions, and electrostatic repulsive forces are additional mechanisms involved in MNP removal through MF and UF membranes. Therefore, the characteristics of MNPs (size, shape, and polymer type) and membranes (structure, pore size, and membrane material) determine the mechanisms of MNP removal from the aquatic environment.

Nanofiltration (NF) is a pressure-driven process with properties intermediate between RO and UF. NF membranes are characterized by lower retention of monovalent ions compared to divalent and multivalent ions, as well as organic compounds with a molecular weight above 200-300 Da (Bodzek 2019, Malankowska 2021, Namla et al. 2025). The membranes used in NF do not have pores in the conventional sense, and the mechanism of separation of dissolved components is based on dissolution and diffusion within the membrane. A study was conducted to evaluate the effectiveness of NF in removing MNPs from raw river water and drinking water at DWTPs in the Paris region of France (Barbier et al. 2022). MNP concentrations in raw water ranged from 7.4 to 45.0 particles/L, while in drinking water from zero to 0.260 particles/L (99% removal). PE, PP, and PET were the main polymers detected in both raw and treated water, with significant differences in the proportions of each polymer. The results of this study suggest that the NF process may be more effective at removing MNPs than conventional processes.

Reverse osmosis (RO) is a pressure-driven membrane process in which TMP drives the selective movement of solvent molecules in the direction opposite to that resulting from osmotic pressure, i.e., from a solution with higher osmotic pressure to a solution with lower osmotic pressure (water). The RO process retains low-molecular-weight molecules, including monovalent salts, undissociated acids, and organic compounds. The separation mechanism in the RO process is described by the dissolution-diffusion model, which assumes that the transport of specific components through compact polymer membranes is determined by their dissolution in the polymer and diffusion. The TMP pressure used in RO must exceed the osmotic pressure of the raw water and is typically in the range of 1.0–8.0 MPa (Bodzek 2019, Malankowska 2021). RO is most commonly used for MNP removal, mainly as a third or even fourth stage of wastewater treatment (Xu and Wang 2020) and partly in municipal and industrial water treatment systems for the removal of salts, heavy metals, and other contaminants. Recently, very few studies have focused on the effectiveness of removing MNPs from drinking water using RO (Koelmans et al. 2019). Fouling of RO membranes by MNPs in surface water is one of the most important factors and a significant obstacle to the wider use of RO in drinking water treatment. The prerequisite for reducing fouling is intensive pre-treatment, which can extend membrane lifetime and reduce the need for cleaning during the process.

Adsorption

Adsorption in drinking water treatment aims to remove organic contaminants such as heavy metals or organic compounds that may be hazardous to health. The process is characterized by simple operation, relatively short adsorption equilibrium times, and the possibility of using recyclable adsorbents. Studies have shown that adsorption can also be effective in removing MNPs using activated carbon, biochar, metal oxides and hydroxides, magnetic nanoparticles, carbon nanotubes, and other materials (Liu and Khor 2024, Ali et al. 2023, Arenas et al. 2021, Ali et al. 2024, Tiwari et al. 2020, Sun et al. 2022, Shi et al. 2022, Tang et al. 2021, El Hadri et al. 2021, Yen et al. 2022). Functional groups (-COOH, -OH, -N(CH_3), -NH_2, -Si-O, Fe-OH) present on adsorbents play a key role in the removal of MNPs (Batool and Valiyaveettil 2021, Rong et al. 2022). The mechanism of MNP removal depends on their properties and those of the adsorbents as well as environmental conditions, i.e., pH and temperature, the presence of natural organic matter (NOM), ions, etc. (Shi et al. 2022, Huang et al. 2022) and involves hydrophobic, electrostatic, and intermolecular interactions, hydrogen bonding, π-π interactions, and intramolecular diffusion (Peng et al. 2022, Zheng et al. 2022).

Biochar produced by pyrolysis, carbonization, and impregnation of various types of biomass (sugarcane bagasse, corn cobs, etc.) is used for MNP adsorption (Singh et al. 2021, Wang et al. 2021c, Rosales et al. 2017). Biochar-based adsorbents can remove >75% of MNPs, with removal efficiencies reaching up to 99% for modified biochar types (Liu and Khor 2024, Arenas et al. 2021). Using a filtration column system with biochar and sand, the removal efficiency of MPs can reach 90% for particles larger than 10 μm (Rosales et al. 2017). Contaminants are removed mainly through a combination of biodegradation and adsorption (electrostatic attraction or complexation). Studies have shown that NPs are also effectively adsorbed using biochar from corn cobs and coffee grounds (74% efficiency) (Magid et al. 2021, Yen et al. 2022). The removal mechanism is based primarily on the formation of hydrogen bonds with functional groups of adsorbents.

In order to increase sorption capacity and recyclability, **adsorbents are modified by** adding **metals** (Mg, Zn, Fe, Cu, Co, Mn, Pt), metal oxides (e.g., Fe_3O_4), strong oxidants (HNO_3, H_2SO_4), and minerals (e.g., kaolin) (Batool and Valiyaveettil 2021, Huang et al. 2022, Singh et al. 2021, Wang et al. 2021c, Magid et al. 2021, Grbic et al. 2019). Adsorbents modified with iron or other metals are of particular interest because they can be easily separated using an external magnetic field (Shi et al. 2022, Grbic et al. 2019).

Grbic et al. (2019) investigated the effectiveness of magnetic extraction in isolating MNPs from seawater, freshwater, and sediments using hydrophobic Fe nanoparticles coated with hexadecyltrimethoxysilane (HDTMS), achieving 92% and 93% removal of PE and PS MNPs from seawater, and 84% and 78% from freshwater and sediments, respectively. Singh et al. (2021) studied iron-modified biochar for the elimination of MNPs (PS-COOH and PS-NH_2), achieving complete removal of nanoparticles of approximately 1 μm in size and enabling magnetic separation from the aquatic environment. Magnetic biochar (MBC) modified with Mg (Mg-MBC) and Zn (Zn-MBC) was also used for the magnetic removal of MNPs from PS with a size of approximately 1 μm (Wang et al. 2021c).

The removal efficiencies in all three cases were 98.75% and 99.46%, respectively. In this case, Mg- and Zn-modified magnetic biochar proved more effective than iron-modified magnetic biochar because the adsorption process is the result of electrostatic interactions and chemical bonding between magnetic particles and biochar (Wang et al. 2021c). Tang et al. (2021) used magnetic carbon nanotubes (M-CNT) as adsorbents to remove MNPs from PE, PET, and PA, with the possibility of easily separating the formed MNP/M-CNT composites from aqueous solutions in a magnetic field. The maximum adsorption capacities of PE, PET, and PA were 1650, 1400, and 1100 mg/g, respectively. The adsorbed M-CNT can be recycled by thermal treatment (600 °C), and after four cycles, the removal efficiency remains around 80%. Magnetic extraction is particularly suitable for removing MNPs from drinking water (Grbic et al. 2019).

Other metal-based adsorbents are also effective for MNP removal. For example, hydrophobic CuFeCo alloy powder can remove more than 83% of MNP-PE from water (Grbic et al. 2019), and CuNi carbon material (CuNi@C) completely eliminate MNPs-PS (100 nm). The mechanism of MNP removal involves physical adsorption, electrostatic attraction, and monolayer coating (Zhou et al. 2022). Co/Mn-kaolin and Fe-kaolin adsorbents are effective in removing MNPs from PVC, PS, PET, and acrylonitrile-butadiene-styrene terpolymer (ABS) through electrostatic interactions (Huang et al. 2022). Layered double hydroxides of Zn-Al (LDH) can also completely remove MNPs from freshwater at pH 4, although their effectiveness decreases significantly under alkaline conditions due to LDH destabilization (Tiwari et al. 2020). Sun et al. (2022) combined cellulose with Mg-Al LDH to obtain a cellulose/Mg-Al LDH composite with an MNP-PS (>100 nm) removal efficiency of 97%, attributed to a combination of pore diffusion, hydrogen bonding, and electrostatic interactions.

In summary, metal-based adsorbents are characterized by high removal efficiency, and their magnetic properties provide a significant operational advantage. The adsorption mechanism is mainly based on interactions between adsorbents and MPs/NPs and depends primarily on particle size and adsorbent morphology.

Aerogel adsorbents produced from biomass are characterized by high specific surface area and mechanical strength, porous structure, biodegradability, low cost, and the presence of many active functional groups capable of adsorbing various pollutants, including MNPs (Rong et al. 2022, Zheng et al. 2022, Sun et al. 2020, 2021, Wang et al. 2021b). The efficiency of removing MNPs from PS using aerogels/sponges is 89.8%, 72.4%, and 88.9% for PS, PS-COOH, and PS-NH_2 particles, respectively (Sun et al. 2020). The adsorption mechanism of MNPs involves hydrophobic, electrostatic, and van der Waals interactions, hydrogen and π-π bonds, as well as ion exchange and intramolecular diffusion. Oxygen-doped carbon nitride (O-C_3N_4) introduced into chitin-based sponges enables over 90% removal efficiency for MNPs from PS, PS-COOH, and PS-NH_2 (~1 μm) (Sun et al. 2021). Similarly, plant-based sponges can adsorb up to 81.2% of MNPs from PS due to their high porosity (83%), while hydrophobic groups in protein side chains can form hydrophobic interactions with the benzene rings present in PS microplastic particles (Wang et al. 2021b). Cellulose aerogel exhibits a high adsorption capacity (146.4 mg/g) for MNPs-PS due to its unique porous morphology and active sites that bind MNPs through electrostatic interactions and ion exchange (Zhuang et al. 2022). Similarly, magnetic chitosan aerogels containing polydopamine (PDA-MCS) bind up to 91.6% of PET magnetic particles through electrostatic interactions, physical adhesion, hydrogen bonding, and π-π interactions (Zheng et al. 2022). Studies have shown that a Fe_3O_4 particle-based sponge grafted with polydimethylsiloxane (PDMS) can remove up to 100% of MNPs (PP, PVC, and PE) through a combination of intramolecular diffusion, electrostatic interaction, hydrogen bonding, and σ-p (or p-p) conjugation mechanism (Rong et al. 2022).

Chemical oxidation and disinfection

Chemical oxidation is used to remove organic compounds that are difficult to remove in other unit processes. Studies have described the removal of MNPs using various oxidants such as ozone, hydrogen peroxide, oxidizing acids, as well as advanced oxidation processes (AOPs). The most commonly used and effective AOP methods are the photo-Fenton method, the electro-Fenton method, and photocatalytic oxidation (Hübner et al. 2024, Miao et al. 2020, Feng et al. 2011). Chemical oxidation causes the mineralization of polymers and their conversion into CO_2, water, and other substances. To increase the efficiency of these processes, radiation from various sources (e.g., UV-vis radiation and solar energy), electric current, and ultrasound are used (Feng et al. 2011).

Disinfection involves removing contaminants from water, mainly microorganisms (bacteria, viruses, protozoa, fungi). The process improves water quality, thereby reducing the risk of disease and health problems. Chemical disinfection involves the use of chlorine, chloramines, chlorine dioxide, ozone and physical methods including UV radiation and high temperatures (Kumar 2021). Disinfection is usually the final stage of drinking water treatment; therefore, MNPs present in drinking water may interact with microorganisms found in drinking water. For example, biofilm formation on the surface of MNPs may reduce the effectiveness of disinfection (Enfrin et al. 2019).

The Fenton process is one of the most commonly used advanced oxidation processes (AOPs). The reaction between hydrogen peroxide (H_2O_2) and heterogeneous catalysts containing Fe^{2+} ions produces highly reactive hydroxyl radicals, which then oxidize organic pollutants to CO_2, water, and mineral products. MNPs are difficult to oxidize using the classical Fenton process (Tagg et al. 2017, Ortiz et al. 2022). In practice, methods are used in which the main oxidant (H_2O_2) is produced electrochemically, or UV radiation is used to produce hydroxyl radicals from H_2O_2 molecules in the presence of an Fe(II) catalyst (Hübner et al. 2024, Miao et al. 2020). Miao et al. (2020) used the electro-Fenton method to degrade PVC MNPs and achieved 75% dechlorination efficiency and 56% mass loss within 6 hours of the experiment at 100°C, using a TiO_2/graphite cathode to generate H_2O_2. The method is also effective for removing MNPs from PVC, PS, PP, and PE (Miao et al. 2020). UV radiation also increases the rate of oxidative degradation of MNPs from PS to over 99% (Feng et al. 2011). In conclusion, further research is needed to enable the effective application of the Fenton process for the removal of MNPs.

The **photocatalysis process** involves exciting a photocatalyst by absorbing an appropriate amount of energy after UV irradiation or exposure to visible light. Photoexcitation results in the generation of photogenerated electron-hole pairs eCB^-/hVB^+ (CB - conduction band, VB - valence band) (Ouyang et al. 2021). The hVB^+ valence-band holes react with adsorbed water molecules and hydroxyl groups on the surface of the heterogeneous photocatalyst, producing hydroxyl radicals (•HO). At the same time, eCB^- electrons reduce O_2 in solution to form superoxide anion radicals (O_2^-), which react with water to form hydroperoxide radicals (HOO•). These radicals effectively oxidize organic pollutants, including polymers (Ouyang et al. 2021).

Various nanostructured semiconductors are used in photocatalysis, among which metal oxide nanomaterials (ZnO, TiO_2) are the most widely used due to their high redox potential, non-toxicity, adequate electron mobility, and flexibility in terms of achievable particle sizes and morphologies (Tofa et al. 2019). Tofa et al. (2019) studied the photocatalytic degradation of low-density PE-based MNPs using heterogeneous ZnO rod-shaped nanocatalysts. Based on microscopic images, morphological changes in the MNPs were observed, including the appearance of irregularities, brittleness, cracks, and stains on the irradiated surfaces. Nitrogen-containing TiO_2 photocatalysts (N-TiO_2) also have the ability to degrade MNPs in both aqueous and solid phases (Tofa et al. 2019). During irradiation with visible light, a 6.40% mass loss of high-density PE MNPs was achieved within 18 hours. Ali et al. (2016) demonstrated the photocatalytic degradation of composite PE films with the addition of a blue-green dye sensitive to visible light. Dye sensitization of nanostructures increased the degradation rate of low-density PE films under visible light, including sunlight. The formation of volatile organic products during photodegradation causes defects in the films, as confirmed by FTIR, SEM and tensile strength measurements. Thus, polyethylene films with 10% dye-colored titanium nanotubes showed approximately 50% degradation under visible light exposure over a short period of 45 days. In both cases, the surface area of the catalyst, as well as the extent and nature of interactions between MNPs molecules and the catalyst surface, significantly influenced removal efficiency.

Ozonation causes oxidation of polymeric substances containing unsaturated bonds as well as aromatic rings (Chen et al. 2018). Studies have confirmed the effectiveness of ozone in the degradation of polymers (Chen et al. 2018, Hidayaturrahman and Lee 2019, Tian et al. 2017) by creating highly reactive radicals (mainly hydroxyl radicals). The process is used as a direct method for removing MNPs or as a means of increasing the effectiveness of other purification methods, e.g., biological ones. Hidayaturrahman and Lee (2019) achieved a high degree of MNPs degradation (>90%) as a result of ozonation, compared to other methods such as membrane filtration (79.4%), RSF, and GAC filtration (73.8%). The main challenges associated with ozonation are the high cost of ozone production and environmental concerns (Hidayaturrahman and Lee 2019, Tian et al. 2017). The combination of ozonation with GAC filtration has shown an increase in the elimination of small MNPs in the size range of 1-5 μm, 10-50 μm and fibers, by up to 22% (Wang et al. 2020a, Pivokonsky et al. 2020).

Ozonation is also an effective method of disinfection. Ozone reacts quickly with microorganisms, destroying and deactivating their cell walls. However, it requires specialized equipment and careful control of the ozone dose. The oxidation potential of ozone is 2.08 eV, so the process can kill chlorine-tolerant microorganisms by attacking their cell membranes

(Ding et al. 2019). However, MNPs can interact with ozone, reducing the number of ozone molecules available to react with bacteria and leaving non-disinfected pathogens in the water (Enfrin et al. 2019). Ozonation provides relatively short-term protection and is therefore used as a complementary method (Kumar et al. 2023).

Chlorination is one of the most commonly used disinfection methods in drinking water treatment. Chlorine or its compounds kill bacteria by inhibiting the activity of their enzymes (Mahmood et al. 2018). The method is low-cost and effective, but micron-sized suspended solids in water, including MNPs, can hinder the action of chlorine on microorganisms (Kernegger et al. 2009). Therefore, these particles can act as protective substrates, causing bacteria to become resistant to the disinfection process (Enfrin et al. 2019). Chlorination can also lead to the formation of disinfection by-products, which require careful monitoring and control (Kumar 2021). Chlorine concentration, contact time, MNP abundance and properties, water quality, and environmental factors (pH, temperature, etc.) can influence the interaction of MNPs with chlorine and the effectiveness of their removal. The impact of disinfection on MNP removal in DWTPs has not yet been fully understood. Some studies conducted in WWTPs have shown that the contribution of disinfection to MNP removal ranges from 15.8% to 67.6% (Galafassi et al. 2022) and even up to 71% (Liu et al. 2019), while other studies report that chlorination reduces MNPs by 28% (Kankanige and Babel 2021) and even up to 7.1% (Radityaningrum et al. 2021).

UV disinfection involves neutralizing microorganisms by damaging their genetic material, preventing them from replicating and rendering them harmless (Carre et al. 2018). The process is effective in combating pathogens, does not produce chemical by-products, and does not change the taste or smell of water. However, it requires careful consideration of the appropriate UV dose and exposure time. Suspended solids, including MNPs, can protect microorganisms from UV radiation, reducing the effectiveness of MNP removal, especially for small particles ranging in size from 0.5 to 20 µm (Cherniak et al. 2022). There are also studies suggesting that UV disinfection contributes to the removal of MNPs (Galafassi et al. 2022). The observed increase in MNP numbers is likely due to the fact that the larger MNPs are first weakened by oxidation and then fragmented by external forces such as the shear force of water flow and aeration. Na et al. (2021) compared the degradation of MNPs by UV photolysis with and without the addition of H_2O_2 (UV and UV/H_2O_2 options). The results showed that UV/H_2O_2 treatment promoted fragmentation of MNPs and changes in their physical properties (i.e., size and morphology). Both UV and UV/H_2O_2 treatments resulted in a significant increase in smaller MNPs and total particle concentration. To date, there is still a lack of knowledge regarding the removal mechanisms of MNPs.

Despite limited information on the presence and behavior of plastics in drinking water, the impact of MNPs on disinfection process is harmful (Shen et al. 2020), and the removal of MNPs remains a controversial issue. Therefore, the impact of disinfection conditions (type and concentration of disinfectant, contact time, pH, temperature, etc.), the abundance and properties of MNPs in water on their removal in DWTPs should be investigated in more detail.

MNPs in drinking water

Recently, there have been increasing studies on the presence of MNPs in drinking water and other food products (Yousefi et al. 2025, Acarer 2023b). The presence of MNPs in surface water, groundwater, and wastewater (Bodzek et al. 2024) contributes to the contamination of drinking water. They can be found in these elements of the system as a result of physical, chemical, and biological degradation of plastic waste, surface runoff, industrial and municipal wastewater, and atmospheric deposition. They can also originate from landfills, which are considered to be the main source of MNPs in groundwater (Chia et al. 2021). In addition, plastic equipment used for drinking water treatment or distribution may become damaged or destroyed, thus constituting a source of MNPs in drinking water (Oßmann et al. 2018). MNPs contained in raw water are not completely removed during raw water treatment in DWTPs, and therefore the resulting drinking water contains MNPs. So far, there is little literature data on the presence of MNPs in water for human consumption, but MNPs have been detected in tap water and bottled water, which has an adverse effect on human health (Acarer 2023b, Zuccarello et al. 2019, Kirstein et al. 2021, Mason et al. 2018, Kosuth et al. 2018, Mintening et al. 2019, 2020). Ensuring safe drinking water is part of the Sustainable Development Goals on access to safe and affordable drinking water (Campos et al. 2022). Given the daily consumption of drinking water by humans, this could be a significant source of MNP exposure for humans.

MNPs in tap water

The presence of MNPs in tap water has been confirmed in studies conducted in various parts of the world, with concentrations ranging from 394 particles/L to as low as 0.01 particles/L (Chowdhury et al. 2023, Cecilia et al. 2024, Almaiman et al. 2021, Feld et al. 2021, Lam et al. 2020, Zhang et al. 2020b). The test results vary greatly, depending on the analytical methods used and geographical location, as well as differences in water management practices, infrastructure condition, and environmental pollution levels. MNP contamination in tap water comes mainly from three sources: contamination of the water source, the performance of DWTPs, and water transport pipelines.

As a result of MNPs analysis in 159 tap water samples collected from different parts of the globe, it was found that 81% of water samples had an average concentration of 5.45 particles/L, with the highest average number of particles detected in tap water in the United States (9.18 particles/L), and the lowest in the EU (France, Germany, Ireland, Italy, Slovakia) and England, Switzerland (3.60 particles/L)) (Kosuth et al. 2018). In addition, it was found that the average number of particles in tap water samples taken from more developed countries (US, EU, and others; 6.85 particles/L)) was higher than in samples taken from less developed countries (Cuba, Ecuador, India, Indonesia, Uganda; 4.26 particles/L)) (Kosuth et al. 2018). For example, in tap water in Akureyri, Iceland, only 2 µg/L of PE and 1.5-5.2 µg/L of PVC were detected (Asmundsdottir et al. 2019). Relatively low values were also found in drinking water fountains in Mexico, Tianjin in China, and 12 cities in Japan, the US, and Europe (Shahi et al. 2020, Mukotaka et al. 2021). Mintenig et al. (2020),

Table 5: Concentration, polymer types, shapes, and sizes of MPs in tap water from different countries

Country	Concentration, particles/L	Detected polymers	Shape	Particle size	References
USA	9.24	-	fibers/fragments	0.1–5 mm	Kosuth et al. 2018
	46	PS, PEST, PP, PVP	Fibers>fragments> balls	19-100 µm	Asmundsdottir et al. 2019
	182	-	fibers/fragments/pellets	2.5-300 µm	
Czech	443 338 and 628	PET, PEST, PE, PP, PVC	Fibers>fragments> balls	1-20 µm 1-50 µm	Pivokonsky et al. 2018
Sweden	0.174	PE, PA, PET, PP	fibers/fragments	<150 and <20µm	Kirstein et al. 2021
Ecuador	4.02	-	fibers/fragments	0.1–5 mm	Kosuth et al. 2018
UK	7.73	-	fibers/fragments	0.1–5 mm	Kosuth et al. 2018
France	1.82	-	fibers/fragments	0.1–5 mm	Kosuth et al. 2018
	97	PEST, PP, PS, PE, PVC	Fibers>fragments> balls	19-100 µm	Asmundsdottir et al. 2019
Germany	> 1	-	-	50-100 µm	Mintenig et al. 2019
	0.91	-	fibers/fragments	0.1–5 mm	Kosuth et al. 2018
	53	PS, PP, PEST, PVC, PE	Fibers>fragments> balls	19-100 µm	Asmundsdottir et al. 2019
India	6.94	-	fibers/fragments	0.1–5 mm	Kosuth et al. 2018
Indonesia	3.23	-	fibers/fragments	0.1–5 mm	Kosuth et al. 2018
Lebanon	6.64	-	fibers/fragments	0.1–5 mm	Kosuth et al. 2018
Slovakia	3.83	-	fibers/fragments	0.1–5 mm	Kosuth et al. 2018
Switzerland	2.74	-	fibers/fragments	0.1–5 mm	Kosuth et al. 2018
Uganda	3.92	-	fibers/fragments	0.1–5 mm	Kosuth et al. 2018
Belgium	0.01	PET, PP, PS, PE, PVC	-	214 µm	Semmouri et al. 2022 Fajar et al.2022
Denmark	0.31	-	fibers > fragments>films	-	Feld et al. 2021
Finland	47	PP, PS, PE, PEST, PVC	Fibers>fragments> balls	19-100	Sembiring et al.2021
Iceland	1.5-5.2 µg/L 2 µg/L	PVC PE			Asmundsdottir et al. 2019
China	440	PE, PP, PS, PET	Fibers>fragments> balls	1-5000 µm	Tong et al. 2020
	0.3-1.6	PET and others	fibers/fragment	10-5000 µm	Zhang et al. 2020b
Hong Kong	1753	-	fragments/ fibers	1–10 > 10–50 >50 µm	Tse et al. 2022
		-	fibers, films	150-5000 µm	Lam et al. 2020
Japan	7-65	PS, PEST, PVC, PP, PP	Fibers>fragments> balls	19-100 µm	Asmundsdottir et al. 2019
Saudi Arabia	1.9	PE, PS, PET, PA, PU	fragments	25–500 µm	Almaiman et al. 2021

studying groundwater and drinking water in Germany for the presence of MNPs, observed only a few polymer particles in both raw and drinking water, with MNP levels ranging from 0 to 0.007 particles/L. However, other studies have shown higher concentrations of MNPs in tap water (Tong et al. 2020, Pivokonsky et al. 2018, Shen et al. 2021). Studies of MNPs content in 38 tap water samples collected from households in China found concentrations ranging from 0 to 1,247 particles/L (with an average value of 440 particles/L) (Tong et al. 2020, Shen et al. 2021). A study conducted at three DWTPs in the Czech Republic determined the number and nature of MNPs (approximately 1 µm) in treated water supplied from surface water reservoirs (Pivokonsky et al. 2018). MNPs were detected in all samples, with average values of 443 particles/L, 338 particles/L, and 628 particles/L depending on the raw water source. The differences between these studies may be due to different water sources, DWTPs performance, and differences in size, shape, and color of MNPS particles, and other factors.

The main types of plastics detected in tap water MNPs are PP, PE, PS, PET (29.6%), and PVC, which essentially corresponds to the types of MNPs found in raw water (Shen et al. 2021, Romphophak et al. 2024, Chowdhury et al. 2023). The results showed that the size of MNPs ranged from <1-5000 μm, with the most frequently detected particles being 83-400 μm in size (Ramphophak et al. 2024). It was found that small MNPs sizes (<100 μm) are particularly dominant in tap water (Tong et al. 2020, Mukotaka et al. 2021). Similar to raw water, fragments, fibers, beads, granules, and membranes were observed in tap water, with fibers and fragments being the most common shapes, accounting for >70% (Shen et al. 2021, Chowdhury et al. 2023), and often even 99.2% of the identified MNPs (Babel and Dork 2021, Lam et al. 2020, Zhang et al. 2020b, Asmundsdottir et al. 2019, Tse et al. 2022).

Table 5 summarizes studies conducted on tap water samples collected in various countries worldwide.

MNPs in bottled water

Another way of introducing MNPs into the human body is by drinking bottled water (Zuccarello et al. 2019, Oßmann et al. 2018, Altunışık et al. 2023, Gálvez-Blanca et al. 2024, Hossain et al. 2023). MNPs in bottled water come from both treated water and packaging. Plastic (PET) and glass bottles are used to package drinking water, and caps are mainly made of PP, PE, and other plastics. MNPs are therefore transferred from the packaging to the water, as the types of polymers in bottles or caps correspond to the polymers found in MNPs contained in bottled water (Mason et al. 2018, Schymanski et al. 2018, Oßmann et al. 2018). The concentration of MNPs in bottled water varies greatly depending on the brand and country, ranging from 0.73 to as high as 54.2 million particles/L (Zhang et al. 2024), but most values are low, at less than 400 particles/L (Almaiman et al. 2021, Kankanige and Babel 2020, Li et al. 2023). Mason et al. (2018) analyzed the occurrence of MNPs in bottled water and found that concentrations ranged from a few to several thousand particles per liter, with an average of 10.4 particles/L, and that the most common form was fragments (66%), followed by fibers. In a study conducted in Bangladesh, mineral water from ten different brands also showed low levels of MNP contamination, ranging from 14 to 56 particles/L, with an average of about 35 particles/L (Hossain et al. 2023).

In many countries, table water and mineral water also contain high concentrations of MNPs, ranging from 1.9 to $5.42 \cdot 10^7$ particles/L (Zuccarello et al. 2019, Oßmann et al. 2018, Zhang et al. 2024). For example, in Italy, concentrations of over $5 \cdot 10^7$ particles/L were found in mineral water (Zuccarello et al. 2019), and in bottled water from Germany, over 35,000 particles/L were reported (Oßmann et al. 2018). These differences result from geographical location, but also from the origin of the water and the length of time it is stored in bottles, the materials from which the bottles are made, the use of different analytical methods for determining MNPs, the level of purification and conditioning of the water in the bottles, and the conditions of transport and storage of the water bottles (Zhang et al. 2024). The concentration of MNPs in bottled water also depends on the nature of the plastic bottles, i.e., whether the bottles are disposable or reusable. Schymanski et al. (2018) found that the concentration of MNPs in water in 15 types of reusable plastic bottles (118 particles/L) was 8 times higher than the amount of MNPs in disposable plastic bottles. This is due to greater material abrasion in reusable bottles than in disposable bottles. Similar observations are reported in a study by Oßmann et al. (2018), which found that water in disposable PET bottles contained fewer MNPs than water in reusable PET bottles.

Currently, plastic water bottles are often being replaced with glass bottles, which should reduce the release of chemicals from plastics, most commonly PET. A study conducted in Germany compared the concentration of solid particles in water from plastic and glass bottles (Oßmann et al. 2018). Water from glass bottles generally had slightly lower concentrations of MNPs, but these values were similar to those in single-use PET bottles and, in both cases, lower than in reusable PET bottles (Oßmann et al. 2018). However, there was a difference in the types of MNPs found, as PET particles dominated in water from both single-use and reusable PET bottles (over 70%), while glass bottles had a higher proportion of PE (46%) and PP (23%) (Oßmann et al. 2018). These results suggest that the type of packaging material plays an important role in determining the composition of MNPs in bottled water, potentially due to the degradation of these materials into smaller particles over time.

In general, MNP concentrations in reusable plastic bottles are higher than in single-use and glass bottles (Di et al. 2019). The levels found in bottled water from Bangladesh were lower than those found in water in glass bottles from Italy, Thailand and Germany (Shen et al. 2020, Sol et al. 2021, Silva et al. 2018). It can therefore be concluded that the reuse of plastic bottles contributes to an increase in the content of MNPs in bottled water. According to various studies, bottled water contains approximately 50% more MNPs than tap water (Rist et al. 2018). This suggests that water preparation for bottling and polymer leaching may contribute to the accumulation and increase of MNPs in drinking water. Similar results have been obtained in other studies. As consumers continue to rely on bottled water as a convenient source of drinking water, it is important to understand the factors that contribute to MNP contamination and to work to minimize their presence in bottled water products. Table 6 shows the abundance of MNPs in bottled water in different countries.

The impact of MNPs in drinking water on human health

MNPs enter the human body through ingestion, inhalation, and skin contact (Schwabl et al. 2019). This applies not only to drinking water containing MNPs, but also to other products such as seafood, honey and sugar, commercially available table salt, and other plastic-packaged products (Rist et al. 2018, Kosuth et al. 2018, Conti et al. 2020, Smith et al. 2018, Peixoto et al. 2019). Cox et al. (2019) estimated that, depending on age and gender, total human intake of MNPs through food and beverages (mainly from seafood and bottled water) in the U.S. ranges from 39,000 to 52,000 particles per year, while 35,000 to 62,000 MNPs per year are inhaled from the atmosphere. Most MNPs remain in the lungs or enter the digestive system, and some are excreted through coughing or sneezing, as well as via urine and feces, and other metabolic pathways due to their resistance to degradation (Wright and Kelly 2017, Cox et al. 2019).

The average daily intake of MNPs with drinking water varies by country (Hossain et al. 2023), ranging from 0 in

Table 6: MNPs concentration in bottled water in different countries

Country	Type of packaging	Concentration, particles/L	References
Italy	Plastic bottle – mineral water	$5.42 \cdot 10^7$	Yuan et al. 2022
Hong Kong	PET Reusable PET bottle	6827 19	Plastic the Fast Facts – Plastic Europe, 2025
Germany	PET Single-use plastic bottle Glass bottle Glass bottle Reusable PET bottle Reusable PET bottle Reusable plastic bottle Cardboard packaging	2649 118 50 3074 4889 242 14 11	Di et al. 2019 Shen et al. 2020 Shen et al. 2020 Di et al. 2019 Di et al. 2019 Rozman and Kalcíkova 2022 Shen et al. 2020 Shen et al. 2020
Italy	PET PET Glass bottle	315 148 195	Silva et al. 2018 Li et al. 2023 Silva et al. 2018
Thailand	Single-use plastic bottle Glass bottle	140 52	Sol et al. 2021 Sol et al. 2021
Australia	PET	13	Uoginte et al. 2022
Malaysia	Single-use plastic bottle	11	Chanpiwat and Damrongsiri 2021
China	Single-use plastic bottle	10	Shruti et al. 2020
Iran	Plastic bottle – mineral water	8.5	Lam et al. 2020
Saudi Arabia	Plastic bottle – mineral water	1.9 1-26 LQ	Hamm and Lenz 2021 Almaiman et al. 2021 Hamm and Lenz 2021
Bangladesh	Single-use PET bottle	35	Plastic the Fast Facts – Plastic Europe, 2025

LQ – below the detection limit

Italy to 61 particles per liter in the United States. On the other hand, in China and the UK, intake ranges from just 2.8 to 1,787 particles and from 0.068 to 20.8 particles per day, respectively (Kosuth et al. 2018, Han et al. 2024, Adediran et al. 2024). Most studies base their estimates of water intake on an average of 2 to 3 liters per day, but some report values well beyond this range (Paray et al. 2024), which influences the different assessment of human exposure to MNPs via tap water.

MNPs entering the human body can mainly cause oxidative stress, tissue damage, and chronic inflammation (Schirinzi et al. 2017), as well as gastrointestinal disorders (Wright and Kelly 2017, Tamargo et al. 2022, Zhang et al. 2022). In addition, autoimmune diseases, cytotoxicity, carcinogenicity, cardiovascular disease, skin irritation and human respiratory disorders have been reported (Smith et al. 2018, Lombardi et al. 2022, Kurniawan et al. 2021, Rai et al. 2021, Sana et al. 2020). Other studies have also detected MNPs in the human reproductive system (Contino et al. 2023, Enyoh et al. 2023, Liu et al. 2023, Zhao et al. 2023), contributing to hormonal disruption and potentially causing infertility in women. A recent study by Contino et al. (2023) confirmed the toxicity of PS-based MNPs to human sperm. MNPs have also been found to have negative effects on the nervous system (Schirinzi et al.2017, Prüst et al. 2020). Hua et al. (2022) reported reduced cell viability and changes in gene expression of DNA and neural tissues as a result of long-term exposure to PS-based MNPs in the human brain. A recent study found that MNPs can interact with proteins in the brain, potentially contributing to dementia and Parkinson's disease (Liu et al. 2023). MNPs such as PS, PET, and PE (>700 nm) have been detected in human blood at a total concentration of 1.6 μg/mL in samples of healthy volunteers, indicating the bioavailability of plastic particles in the bloodstream (Leslie et al. 2022), reaching distant organs in the human body (Smith et al. 2018).

Research into the potential impact of MNPs on human health has only just begun, and the toxic effects require further confirmatory studies. Detailed information on the impact of MNPs on human health is presented in review papers by Wright and Kelly (2017), Barboza et al. (2018), Rist et al. (2018), Schirinzi et al. (2017), Triebskorn et al. (2019), Wang et al. (2018b), and other studies. A summary of the various effects of MNPs on human health is presented in Table 7.

When considering the potential health risks of MNPs in drinking water, the following should be considered in particular: (1) the physical hazards associated with particulate matter, (2) the complexity of their chemical composition, which includes monomeric components, additives and various contaminants adsorbed on the surface of particles, and (3) the possibility of biofilm formation on MNPs (Kutralam-Muniasamy et al. 2023, Nirmala et al. 2023). The transfer of MNPs into the body depends on a number of factors, including their shape, composition, size, and volume, challenging traditional human health risk assessment strategies. For example, smaller particles are transported more easily, while larger ones (>2 μm) usually remain in the intestinal tract. Râpă et al. (2023) indicated that microplastic particles larger than 10 μm are unlikely to be transported across an intact intestinal barrier. As the size of MNPs decreases, penetration into

Table 7: Detection of MNPs in human cells and their toxic effects

MPs type	MPs size (μm)	Shape	Type of toxic effect	Reference
PVC	No data available	Irregular	Hepatic hemangiosarcoma	Bolt et al. 2005
PS	0.04, 0.25, 10	Spherical	Oxidative stress and cytotoxicity	Schirinzi et al. 2017
PE	0.1–16	Spherical	Cytotoxicity	Schirinzi et al. 2017
polycarbonate, epoxy resins	No data available	Irregular	Adverse effects on brain development, hormonal disorders	Rist et al. 2018
PS	0.1 and 5	Spherical	Oxidative stress, cytotoxicity, poor immune response, paracellular and transmembrane transport	Wu et al. 2019, Liu et al. 2020
PS	0.460, 1, 3, 10, 40 and 100	Spherical	Cytotoxicity, absorption challenges and poor immune response, cell proliferation and internalisation	Hwang et al. 2020, Goodman et al. 2021
PS	0.3, 0.5, 1, 3, 6 and 1.72	Spherical	Cytotoxicity, oxidative stress, immune response, barrier integrity, and predictive biomarker for COPD	Dong et al. 2020, Wang et al. 2020c
PVC and ABS	25–75 and 75–200	Irregular	Cytotoxicity and poor immune function	Han et al. 2020
HDPE	1–10, 50 (45–53) and 100 (90–106)	Spherical	Poor immune function and cytotoxicity	Choi et al. 2021
LDPE	25–75 and 75–200	Random	Causes oxidative stress, poor immune response and cytotoxicity	Choi et al. 2021
PS	≤ 0.45	Irregular	Toxicity to macrophages (the major components of the immune system) - decreased the viability of human macrophages, oxidative stress, and decreased cell proliferation	Koner et al. 2023
PS	0.025 and 0.050	no data	Respiratory cell toxicity (bronchial and airway epithelial cells; mitochondrial dysfunction, impaired bioenergetics)	Winiarska et al. 2024
PS	1 and 0.05	no data	Impact on the heart - indirect toxicity through mitochondrial dysfunction and oxidative stress	Ma et al. 2025
PE and PET	1 and 2,6	no data	Potential cytotoxic effects on intestinal epithelial cells - oxidative-stress-induced cytotoxicity	Najahi et al. 2025

PS − polystyrene, PE − polyethylene, PVC − polyvinyl chloride, ABS − acrylonitrile butadiene styrene, HDPE − high-density polyethylene, LDPE − low-density polyethylene, PET − polyethylene terephthalate,

peripheral tissues and the circulatory system through lymphatic aggregation becomes easier, leading to systemic exposure. Although low-concentration MNPs can enter the bloodstream, they have difficulty penetrating deep tissues through the cell membrane due to their size, and they are removed by the spleen (Bouwmeester et al. 2015). Ji et al. (2020) showed that the size of PET nanoparticles (20 nm, 60–80 nm, and 800 nm) plays an important role in in vivo toxicity studies. On the other hand, Banerjee et al. (2022) studied cellular responses to PS particles of different sizes (50–5000 nm), finding that the uptake of smaller nanoparticles (50 or 100 nm) by cells was higher than that of 1000 nm particles. These examples show that, in the context of the potential impacts on human health, studies on NPs are of greater importance compared to studies on MPs, especially since nanoparticles can penetrate the cell membranes of living organisms.

In addition to MNPs, chemical additives used in plastic production can cause diseases leading to cancer, affect immune response and enzyme activity, and cause reproductive disorders, as well as other disorders (Rist et al. 2018, Ziccardi et al. 2016) (Table 8). Examples of additives include PAHs, polychlorinated biphenyls (PCBs), brominated flame retardants, bisphenols (BPA), triclosan, heavy metals, octadecyl trichlorosilane (ODT), phthalates, and others that can cause physiological changes (Okoye et al. 2022). When MNPs are in an aqueous environment, additives are flushed out at a rate that depends on temperature, pH, pressure, etc. Many studies have shown the presence of plasticizers such as phthalates and bisphenol A in natural and treated water in DWTPs (Li et al. 2024c). These compounds have been associated with hormonal changes, negative effects at the onset of puberty, toxicity and effects on the gut microbiota and the nervous, reproductive, and immune systems (Li et al. 2018, Sun and Wang 2023). In addition, MNPs can adsorb various chemical compounds on their surfaces, mainly metals/metalloids such as cadmium, manganese, lead, arsenic, copper, zinc, chromium, etc. (Selvam et al. 2021, Gao et al. 2019). The adverse health effects

Table 8: Selected additives in MNP and their health risks

Additives	Application	Effect	References
Phthalates	Plasticizers	May cause dysregulation of thyroid function, reproduction, and metabolism	Wang and Qian 2021
Triclosan	Antibacterial agent	May cause decreased immune function and metabolic activity, as well as endocrine disruption, liver tumor promotion, and antibiotic resistance	Browne et al. 2013 Marazuela et al. 2022
Antimony	Flame retardants and biocides	May damage the liver and digestive system. May cause cardiovascular disease. Possibly carcinogenic to humans.	Byrne et al. 2013, Hahladakis et al. 2018 Wu et al. 2025
Heavy metals e.g. Ba, Cd, Ti, Pb, Cr	Inorganic pigments, UV and thermal stabilisers	It may cause cardiovascular and kidney diseases; mental disorders; breast cancer and neurological disorders, cell apoptosis, anaemia, hypertension, oxidative stress, brain and cell damage, and infertility.	Byrne et al. 2013, Jan et al. 2015, Gandamalla et al. 2019, Massons and Turner 2017, Abahussain et al. 2025

associated with metals are well known (Table 8), and changes in the endocrine system and sudden hormonal responses induced by metal/metalloid-laden MNPs have been confirmed in studies (Rehman et al. 2017). In addition, ingested MNPs can also serve as vectors for harmful bacteria adsorbed on their surface, such as *Vibrio spp.* (Kirstein et al. 2016).

Final conclusions and future research directions

This article describes research on the presence and nature of MNPs in drinking water and water used for its production, as well as the removal of MNPs in unit processes used in DWTPs. Previous studies have focused more on WWTPs, and there is a lack of research on the removal of MNPs in drinking water treatment processes.

1. MNPs of various types, sizes, and shapes present in **raw drinking water** reach DWTPs. Fragments, fibers, films, foams, and granules are the most commonly found shapes of these contaminants in such water samples , including seawater, river water, and even groundwater.
2. Studies have shown the release of MNPs into treated water during **transport of raw water and its purification**. Preliminary results indicate that conventional DWTPs treatment sequences are only partially effective in reducing the concentration of MNPs in drinking water, as DWTPs are not designed exclusively and specifically for the removal of MNPs. Due to the high diversity of MNPs in water and their incomplete removal, DWTPs with more stages are more effective than simpler treatment systems in terms of MNPs removal, particularly membrane processes or advanced oxidation processes.
3. The presence of MNPs in **tap water, bottled water, and beverages** has been confirmed in numerous studies conducted in various parts of the world. Number of MNPs varies greatly, ranging from a few to several thousand particles per liter of water. Such significant variation in test results is due to the analytical methods used, geographical location, but also differences in water management practices, infrastructure condition, and environmental pollution levels. A comparison of the amount of MNPs in plastic and glass water bottles shows that the use of the former contributes to an increase in their content in drinking water. In addition, MNPs in drinking water vary greatly not only in concentration but also in size, shape, and material. Commonly known MNP materials in drinking water include PE, PP, PS, PET, PVC, etc., and the most common shapes are fragments, spheres, and fibers.
4. Drinking water treatment plants must address the presence of MNPs in treated water, as they pose a new threat to **human health**. In addition to inhalation and ingestion of MNPs through contaminated air and food, drinking water is one of the main routes of exposure to MNPs. While data on actual adverse effects in humans remain scarce, preliminary studies have revealed a number of harmful health effects caused by MNPs. Furthermore, another challenge is the health effects of chemicals added to plastics or adsorbed on MNPs surface.

In summary, further research is needed on the following issues:

- Quantitative determination and characterization (size, shape, polymer composition) of MNPs in surface and groundwater, as well as drinking water, should become an important part of various water quality monitoring programs.
- In-depth research on the removal of MNPs by various unit processes that can be used in water treatment, including the optimization of current treatment technologies or the development of new processes aimed at removing MNPs.
- Investigate the possibility of secondary contamination of tap water with MNPs during water treatment and transport processes to consumers.
- Undertake joint efforts to eliminate single-use plastic products, develop appropriate rules for plastic recycling, increase efforts to develop biodegradable materials to replace plastics, and adopt the concept of a circular economy in this area.

References

Abahussain, A.A.M., Nasr, F.A., bin Jumah, A., Saravanan, P., Kumar, N.S., Al-zharani, M., Guganathan, L., Sasikumar, G., Alsalamah, S.A., Qurtam, A.A., Senthilkumar, J., Tamizhdurai, P. (2025). Toxic threats from plastic waste: human health impacts, challenges, and policy solutions, *RSC Adv*, 15(48), pp. 40761–40788. DOI:10.1039/d5ra05845g.

Acarer, S. (2023a). Microplastics in wastewater treatment plants: Sources, properties, removal efficiency, removal mechanisms, and interactions with pollutants, *Water Sci. Technol*., 87, pp. 685–710. DOI:10.2166/wst.2023.022.

Acarer, S. (2023b). Abundance and characteristics of microplastics in drinking water treatment plants, distribution systems, water from refill kiosks, tap waters and bottled waters, *Sci. Total Environ*., 884, 163866. DOI: 0.1016/j.scitotenv.2023.163866.

Adediran, G.A., Cox, R., Jürgens, M.D., Morel, E., Cross, R., Carter, H., Pereira, M.G., Read, D.S. & Johnson, A.C. (2024). Fate and behaviour of Microplastics (> 25μm) within the water distribution network, from water treatment works to service reservoirs and customer taps, *Water Res*., 255, 121508. DOI: 10.1016/j.watres.2024.121508.

Adib, D., Mafigholami, R. & Tabeshkia, H. (2021). Identification of microplastics in conventional drinking water treatment plants in Tehran, Iran, *J. Environ. Health Sci. Eng*., 19, pp. 1817–1826. DOI: 10.1007/s40201-021-00737-3.

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. DOI: 10.1016/j. envpol.2019.113011.

Ali, S.S., Qazi, I.A., Arshad, M., Khan, Z., Voice, T.C. & Mehmood, C.T. (2016). Photocatalytic degradation of low density polyethylene (LDPE) films using titania nanotubes, *Environ. Nanotechnol. Monit. Manag*. , 5, pp. 44–53. DOI: 10.1016/j.enmm.2016.01.001.

Ali, I., Tan, X., Li, J., Peng, C., Wan, P., Naz, I., Duan, Z. & Ruan, Y. (2023). Innovations in the development of promising adsorbents for the remediation of microplastics and nanoplastics – A critical review, *Water Res*., 230, 119526. DOI: 10.1016/j.watres.2022.119526.

Ali, I., Tan, X., Mustafa, G., Gao, J., Peng, C., Naz, I., Duan, Z., Zhu, R. & Ruan, Y. (2024). Removal of micro- and nanoplastics by filtration technology: Performance and obstructions to market penetrations, *J. Cleaner Prod*., 470, 143305. DOI: 10.1016/j. jclepro.2024.143305.

Alimi, O.S., Budarz, J.F., Hernandez L.M. & Tufenkji N. (2018). Microplastics and nanoplastics in aquatic environments: aggregation, deposition, and enhanced contaminant transport, *Environ. Sci. Technol*., 52, pp. 1704-1724. DOI: 10.1021/acs.est.7b05559.

Almaiman, L., Aljomah, A., Bineid, M., Aljeldah, F.M., Aldawsari, F., Liebmann, B., Lomako, I., Sexlinger, K. & Alarfaj, R. (2021). The occurrence and dietary intake related to the presence of microplastics in drinking water in Saudi Arabia, *Environ. Monit. Assess*., 193**,** 390. DOI: 10.1007/s10661-021-09132-9.

Altunışık, A. (2023). Microplastic pollution and human risk assessment in Turkish bottled natural and mineral waters, *Environ. Sci. Pollut. Res. Int*., 30, pp. 39815-25. DOI: 10.1007/ s11356-022-25054-6.

Arenas, L.R., Gentile, S.R., Zimmermann, S. & Stoll, S. (2021). Nanoplastics adsorption and removal efficiency by granular activated carbon used in drinking water treatment process, *Sci. Total Environ*., 791, 148175. DOI: 10.1016/j. scitotenv.2021.148175.

Arvaniti, O.S., Antonopoulou, G., Tsagkogianni, D. & Stasinakis, A.S., (2021). Screening on the sorption of emerging contaminants to polystyrene and polyethylene and use of coagulation–flocculation process for microplastics' removal, *Glob. Nest J.,* 23, pp. 303–308. DOI: 10.30955/gnj.003854.

Asmundsdottir, A.M., Gomiero, A. & Oysaed, K.B. (2019). Microplastics and nanoplastics occurrence and composition in drinking water from Akureyri urban area, Iceland, *2nd International Conference on microplastic pollution in the Mediterranean sea. Capri, ITALY,* pp. 106–111. DOI: 10.1007/978-3-030-45909-3_18.

Babel, S. & Dork, H. (2021). Identification of micro-plastic contamination in drinking water treatment plants in Phnom Penh, Cambodia, *J. Eng. Technol. Sci*., 53, 210307. DOI: 10.5614/j. eng.technol.sci.2021.53.3.7.

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, pp. 10377-10385. DOI: 10.1021/acs.est.6b02917.

Banerjee, A., Billey, L.O., McGarvey, A.M. & Shelver, W.L. (2022). Effects of polystyrene micro/nanoplastics on liver cells based on particle size, surface functionalization, concentration and exposure period. *Sci. Total Environ*., 836, 155621.

Barbier, J.S., Dris, R., Lecarpentier, C., Raymond, V., Delabre, K., Thibert, S., Tassin, B. & Gasperi, J. (2022). Microplastic occurrence after conventional and nanofiltration processes at drinking water treatment plants: preliminary results, *Front. Water,* 4, 886703. DOI: 10.3389/frwa.2022.886703.

Barboza, L.G.A., Vethaak, A.D., Lavorante, B.R.B.O., Lundebye, A.K. & Guihermino, L. (2018). Marine microplastic debris: an emerging issue for food security, food safety, and human health, *Marine Pollution Bulletin,* 133, pp. 336–348. DOI: 10.1016/j. marpolbul.2018.05.047.

Batool, A. & Valiyaveettil, S. (2021). Surface functionalized cellulose fibers–A renewable adsorbent for removal of plastic nanoparticles from water, *J. Hazard. Mater*., 413, 125301. DOI: 10.1016/j. jhazmat.2021.125301.

Bergmann, M., Gutow, L. & Klages, M. (2015). Marine Anthropogenic Litter. Springer.

Bodzek, M. (2019). Membrane separation techniques – removal of inorganic and organic admixtures and impurities from water environment – review, *Archives Environ. Prot*. 45(4), pp. 4–19. DOI: 10.24425/aep.2019.130237.

Bodzek, M. & Pohl, A. (2022). Removal of microplastics in unit processes used in water and wastewater treatment: a review, *Archives Environ. Prot*., 48(4). pp. 102–128. DOI: 10.24425/ aep.2022.143713.

Bodzek, M., Pohl, A. & Rosik-Dulewska, C. (2024). Microplastics in wastewater treatment plants: characteristics, occurrence and removal technologies, *Water*, 16, 3574. DOI: 10.3390/w16243574.

Bodzek, M. & Bodzek, P. (2025). Remediation of micro- and nanoplastics by membrane technologies. *Membranes*, 15, 82. DOI: 10.3390/membranes15030082.

Bolt, H.M. (2005). Vinyl chloride – a classical industrial toxicant of new interest, *Critical Reviews in Toxicology,* 35, pp. 307–323. DOI: 10.1080/10408440490915975.

Bordos, G., Urbanyi, B., Micsinai, A., Kriszt, B., Palotai, Z., Szabo, I., Hantosi Z. & Szoboszlay S. (2019). Identification of microplastics in fish ponds and natural freshwater environments of the Carpathian basin, Europe, *Chemosphere* 216, pp. 110-116. DOI: 10.1016/j.chemosphere.2018.10.110.

Bouwmeester, H., Hollman, P.C. & Peters, R.J. (2015). Potential health impact of environmentally released micro- and nanoplastics in the human food production chain: experiences from nanotoxicology, *Environ. Sci. Technol*., 49, 8932. DOI: 10.1021/acs.est.5b01090.

Browne, M.A., Niven, S.J., Galloway, T.S., Rowland, S.J. & Thompson, R.C. (2013). Microplastic moves pollutants and additives to worms, reducing functions linked to health and biodiversity, *Current Biol*., 23, pp. 2388–2392. DOI: 10.1016/j.cub.2013.10.012.

Byrne, C., Divekar, S.D., Storchan, G.B., Parodi, D.A. & Martin, M.B. (2013). Metals and breast cancer, *J. Mammary. Gland. Biol. Neoplasia*, 18, pp. 63–73. DOI: 10.1007/s10911-013-9273-9.

Cai, Y., Yang, T., Mitrano, D.M., Heuberger, M., Hufenus, R. & Nowack, B. (2020). Systematic study of microplastic fiber release from 12 different polyester textiles during washing, *Environ. Sci. Technol*., 54. pp. 4847–4855. DOI: 10.1021/acs.est.9b07395.

Campos, L. C., Olago, D. & Osborn, D. (2022). Water and the UN sustainable development goals. *UCL Open Environment*, 4(1). DOI: 10.14324/111.444/ucloe.000029.

Carre, E., Perot, J., Jauzein, V. & Lopez-Ferber, M., (2018). Impact of suspended particles on UV disinfection of activated-sludge effluent with the aim of reclamation, *J. Water Process Eng*., 22, pp. 87-93. DOI: 10.1016/j.jwpe.2018.01.016.

Casella, C., Sol, D., Laca, A. & Díaz, M. (2025). Investigating the Potential of Coagulants to Improve Microplastics Removal in Wastewater and Tap Water. *Microplastics*, *4*, 89. DOI: 10.3390/microplastics4040089.

Cecilia, D, Philipp, M, Kaegi, R, Schirmer, M. & Moeck, C. (2024). Microplastics attenuation from surface water to drinking water: Impact of treatment and managed aquifer recharge - and identification uncertainties, *Sci. Total Environ*., 908, 168378. DOI: 10.1016/j.scitotenv.2023.168378.

Chanpiwat, P. & Damrongsiri, S. (2021). Abundance and characteristics of microplastics in freshwater and treated tap water in Bangkok, Thailand, *Environ Monit Assess*., 193, 258. DOI: 10.1007/s10661-021-09012-2.

Chen, R., Qi, M., Zhang, G. & Yi, C. (2018). Comparative experiments on polymer degradation technique of produced water of polymer flooding oilfield, *IOP Conference Series: Earth and Environmental Science*, 113, 012208. DOI: 10.1088/1755-1315/113/1/012208.

Cherniak, S.L., Almuhtaram, H., McKie, M.J., Hermabessiere, L., Yuan, C., Rochman, C.M. & Andrews, R.C. (2022). Conventional and biological treatment for the removal of microplastics from drinking water, *Chemosphere* 288, 32587. DOI: 10.1016/j.chemosphere.2021.132587.

Chia, R.W., Lee, J.Y., Kim, H. & Jang, J. (2021). Microplastic pollution in soil and groundwater: a review, *Environ. Chem. Lett*., 19**,** pp. 4211–4224. DOI: 10.1007/s10311-021-01297-6.

Choi, D., Hwang, J., Bang, J., Han, S., Kim, T., Oh, Y., Hwang, Y., Choi, J. & Hong, J. (2021). In vitro toxicity from a physical perspective of polyethylene microplastics based on statistical curvature change analysis, *Sci. Total Environ*., 752, 142242. DOI: 10.1016/j.scitotenv.2020.142242.

Chowdhury, S.R., Razzak, S.A., Hassan, I., Zakir Hossain S.M. & Hossain M.M. (2023). Microplastics in freshwater and drinking water: sources, impacts, detection, and removal strategies, *Water Air Soil Pollut*., 234, 673, DOI: 10.1007/s11270-023-06677-y.

Chu, X., Zheng, B., Li, Z., Cai, C., Peng, Z., Zhao, P. & Tian, Y. (2022). Occurrence and distribution of microplastics in water supply systems: in water and pipe scales, *Sci. Total Environ*., 803, 150004. DOI: 10.1016/j.scitotenv.2021.150004.

Conti, G.O., Ferrante, M., Banni, M., Favara, C., Nicolosi, I., Cristaldi, A., Fiore, M. & Zuccarello, P. (2020). Micro- and nano-plastics in edible fruits and vegetables. The first diet risks assessment for the general population, *Environmental Research* 187**.** 109677. DOI: 10.1016/j.envres.2020.109677.

Contino, M., Ferruggia G., Indelicato S., Pecoraro R., Scalisi E.M., Bracchitta G., Dragotto J., Salvaggio A. & Brundo M.V. (2023). In vitro nano-polystyrene toxicity: metabolic dysfunctions and cytoprotective responses of human spermatozoa, *Biology*, 12, 624. DOI: 10.3390/biology12040624

Cox, K.D., Covernton, G.A., Davies, H.L., Dower, J.F., Juanes, F. & Dudas, S.E., (2019). Human consumption of microplastics, *Environ. Sci. Technol.,* 53, pp. 7068–7074. DOI: 10.1021/acs.est.9b01517.

Crittenden, J.C, Trussell, R.R, Hand, D.W., Howe K.J. & Tchobanoglous G. (2012). MWH's Water Treatment: Principles and Design. Hoboken: John Wiley & Sons.

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*., 28, pp. 59462–59472. DOI: 10.1007/s11356-021-13220-1.

Di, M. & Wang, J. (2018). Microplastics in surface waters and sediments of the three Gorges reservoir, China, *Sci. Total Environ*., 616, pp. 1620-1627, DOI: 10.1016/j.scitotenv.2017.10.150.

Di, M., Liu, X., Wang, W. & Wang, J. (2019). Pollution in drinking water source areas: microplastics in the Danjiangkou reservoir, *China. Environ. Toxicol. Pharmacol*., 65, 82–89. DOI: 10.1016/j.etap.2018.12.009.

Ding, W., Jin, W., Cao, S., Zhou, X., Wang, C., Jiang, Q., Huang, H., Tu, R., Han, S.-F. & Wang Q. (2019). Ozone disinfection of chlorine-resistant bacteria in drinking water, *Water Res*. 160, pp. 339-349. DOI: 10.1016/j.watres.2019.05.014.

Dong, C.-D., Chen, C.-W., Chen, Y.-C., Chen, H.-H., Lee, J.-S. & Lin, C.-H. (2020). Polystyrene microplastic particles: in vitro pulmonary toxicity assessment, *J. Hazard. Mater*., 385, 121575. DOI: 10.1016/j.jhazmat.2019.121575.

Dris, R., Gasperi, J., Rocher, V., Saad, M., Renault, N. & Tassin, B. (2015). Microplastic contamination in an urban area: a case study in greater Paris, *Environ. Chem*., 12, pp. 592–599. DOI: 10.1071/EN14167.

Dronjak, L., Exposito, N., Rovira, J., Florencio, K., Emiliano, P., Corzo, B., Schuhmacher, M., Valero, F. & Sierra, J. (2022). Screening of microplastics in water and sludge lines of a drinking water treatment plant in Catalonia, Spain, *Water Res.,* 225, 119185. DOI: 10.1016/j.watres.2022.119185.

El Hadri, H., Lisa, J.M., Gigault, J., Reynaud, S. & Grassl, B. (2021). Fate of nanoplastics in the environment: Implication of the cigarette butts, *Environ. Poll*., 268, 115170. DOI: 10.1016/j.envpol.2020.115170.

Enfrin, M., Ludovic, F., Dumee. & Lee, J. (2019). Nano/microplastics in water and wastewater treatment processes - origin, impact and potential solutions, *Water Res.,* 161. pp. 621-638. DOI: 10.1016/j.watres.2019.06.049.

Enfrin, M., Lee, J., Le-Clech, P. & Dumée, L.F. (2020). Kinetic and mechanistic aspects of ultrafiltration membrane fouling by nano- and microplastics, *J. Membr. Sci.,* 601, 117890. DOI: 10.1016/j.memsci.2020.117890.

Enyoh, C.E., Duru, C.E., Ovuoraye, P.E., & Wang, Q. (2023). Evaluation of nanoplastics toxicity to the human placenta in systems, *J. Hazard Mater*., 446, 130600. DOI: 10.1016/j.jhazmat.2022.130600.

Ezugbe, E.O. & Rathilal, S. (2020). Membrane Technologies in Wastewater Treatment: A Review, *Membranes*, 10, 89. DOI: 10.3390/membranes10050089.

Fajar, M., Sembiring, E. & Handajani, M. (2022). The effect of filter media size and loading rate to filter performance of removing microplastics using rapid sand filter, J. *Eng. Technol. Sci*., 54, 220512. DOI: 10.5614/j.eng.technol.sci.2022.54.5.12.

Feld, L., da Silva, V.H., Murphy, F., Hartmann, N.B. & Strand, J. (2021). A study of microplastic particles in Danish tap water, *Water*, 13, 2097. DOI: 10.3390/w13152097w13152097.

Feng, H.-M., Zheng, J.-C., Lei, N.-Y., Yu, L., Kong, K.H.-K., Yu, H.-Q., Lau, T.-C. & Lam, M.H.W. (2011). Photo-assisted Fenton degradation of polystyrene, *Environ. Sci. Technol*., 45, pp. 744–750. DOI: 10.1021/es102182g.

Frias, J.P.G.L. & Nash, R. (2019). Microplastics: Finding a consensus on the definition, *Marine Pollution Buulletin*, 138, pp. 145–147. DOI: 10. 1016/j. marpo lbul.2018. 11. 022.

Galafassi, S., Cesare, A.Di, Nardo, L.Di, Sabatino, R., Valsesia, A., Fumagalli, F.S., Corno, G. & Volta, P. (2022). Microplastic retention in small and medium municipal wastewater treatment plants and the role of the disinfection, *Environ. Sci. Pollut. Res*., 29, pp. 10535–10546. DOI: 10.1007/s11356-021-16453-2.

Gálvez-Blanca, V., Edo, C., González-Pleiter, M., Fernández-Piñas, F., Leganés, F. & Rosal R. (2024). Microplastics and non-natural cellulosic particles in Spanish bottled drinking water, *Sci. Rep*., 14, 11089. DOI: 10.1038/s41598-024-62075-2.

Gandamalla, D., Lingabathula, H. & Yellu, N., (2019). Nano titanium exposure induces doseand size-dependent cytotoxicity on human epithelial lung and colon cells, *Drug Chem. Toxicol*., 42, pp. 24–34. DOI: 10.1080/01480545.2018.1452930.

Gao, F., Li, J., Sun, C., Zhang, L., Jiang, F., Cao,W. & Zheng, L. (2019). Study on the capability and characteristics of heavy metals enriched on microplastics in marine environment, *Marine Pollution Bulletin*, 144, pp. 61–67. DOI: 10.1016/j. marpolbul.2019.04.039.

Godoy, L.G.R., Moruzzi, R.B., Odjegba, E.E., Sharifi, S. & dos Reis A.G. (2025). Enhanced removal of microplastics from drinking water using ballasted flocculation: a comparative study with conventional methods using non-intrusive image analysis. *Braz. J. Chem. Eng.,* 42, pp. 1363–1378. DOI: 10.1007/s43153-024-00530-3.

Goodman, K.E., Hare, J.T., Khamis, Z.I., Hua, T. & Sang, Q.-X.A. (2021). Exposure of human lung cells to polystyrene microplastics significantly retards cell proliferation and triggers morphological changes, *Chem. Res. Toxicol*. 34, 1069–1081. DOI: 10.1021/acs. chemrestox.0c00486.

Grbic, J., Nguyen, B., Guo, E., You, J.B., Sinton, D. & Rochman, C.M. (2019). Magnetic extraction of microplastics from environmental samples, *Environ. Sci. Technol. Letters*, 6, pp. 68–72. DOI: 1021/ acs.estlett.8b00671.

Hahladakis, J.N., Velis, C.A., Weber, R., Iacovidou, E. & Purnell, P. (2018). An overview of chemical additives present in plastics: migration, release, fate and environmental impact during their use, disposal and recycling, *J. Hazard. Mater*., 344, pp. 179–199. DOI: 10.1016/j.jhazmat.2017.10.014.

Hamm, T. & Lenz, M. (2021). Negative impacts of realistic doses of spherical and irregular microplastics emerged late during a 42 weeks-long exposure experiment with blue mussels, *Sci. Total Environ*., 778, 146088. DOI: 10.1016/j.scitotenv.2021.146088.

Han, S., Bang, J., Choi, D., Hwang, J., Kim, T., Oh, Y., Hwang, Y., Choi, J. & Hong, J. (2020). Surface pattern analysis of microplastics and their impact on human-derived cells, *ACS. Appl. Polym. Mater*, 2, pp. 4541–4550. DOI: 10.1021/acsapm.0c00645.

Han, Z, Jiang, J, Xia, J, Yan, C. & Cui, C. (2024). Occurrence and fate of microplastics from a water source to two different drinking water treatment plants in a megacity in eastern China, *Environ. Pollut*., 346,123546. DOI: 10.1016/j.envpol.2024.123546.

Hart, M. & Lenhart, J.J. (2026). What's in your water? A comparative analysis of micro- and nanoplastics in treated drinking water and bottled water, *Sci. Total Environ*., 1011, 181148. DOI: 10.1016/j. scitotenv.2025.181148.

Hendrickson, E., Minor, E.C. & Schreiner, K. (2018). Microplastic abundance and composition in western Lake Superior as determined via microscopy, Pyr-GC/MS, and FTIR, *Environ. Sci. Technol.,* 52, pp. 1787–1796. DOI: 10.1021/acs.est.7b05829.

Hidayaturrahman, H. & Lee, T.-G. (2019). A study on characteristics of microplastic in wastewater of South Korea: Identification, quantification, and fate of microplastics during treatment process, *Marine Poll. Bull*., 146, pp. 696–702. DOI: 10.1016/j. marpolbul.2019.06.071.

Hossain, M.B., Yu, J., Banik, P., Noman, M.A., Nur, A.-A.U., Haque, M.R., Rahman, M.M., Albeshr, M.F. & Arai, T. (2023). First evidence of microplastics and their characterization in bottled drinking water from a developing country, *Front. Environ. Sci.,* 11, 1232931. DOI: 10.3389/fenvs.2023.1232931.

Hu, P., Su, K., Sun, Y., Li, P., Cai, J. & Yang, H. (2022). Efficient removal of nano- and micro-sized plastics using a starch-based coagulant in conjunction with polysilicic acid, *Sci. Total Environ*., 850, 157829. DOI: 10.1016/j.scitotenv.2022.157829.

Hua, T., Kiran, S., Li, Y. & Sang, Q.X.A. (2022). Microplastics exposure affects neural development of human pluripotent stem cell-derived cortical spheroids, *J. Hazard Mater*., 435, 128884. DOI: 10.1016/j.jhazmat.2022.128884.

Huang, Z., Bu, J. & Wang, H. (2022). Application of two modified kaolin materials in removing micro-plastics from water, *J. Mater. Cycles Waste Manag*., 24, pp. 1460–1475. DOI: 10.1007/s10163-022-01393-x.

Hübner, U., Spahr, S., Lutz,e H., Wieland, A., Rüting, S., Gernjak, W. & Wenk, J. (2024). Advanced oxidation processes for water and wastewater treatment – Guidance for systematic future research, 10, e30402. DOI: 10.1016/j.heliyon.2024.e30402.

Hwang, J., Choi, D., Han, S., Jung, S.Y., Choi, J. & Hong, J. (2020). Potential toxicity of polystyrene microplastic particles, *Sci. Rep.,* 10, 7391. DOI: 10.1038/s41598-020-64464-9.

Jan, A.T., Azam, M., Siddiqui, K., Ali, A., Choi, I. & Haq, Q.M.R. (2015). Heavy metals and human health: mechanistic insight into toxicity and counter defense system of antioxidants, *Int. J. Mol. Sci*., 16, pp. 29592–29630. DOI: 10.3390/ijms161226183.

Ji, Y., Wang, C., Wang, Y., Fu, L., Man, M. & Chen, L. (2020). Realistic polyethylene terephthalate nanoplastics and the size- and surface coating-dependent toxicological impacts on zebrafish embryos. *Environ. Sci.-Nano*, 7, 2313–2324. DOI: 10.1039/D0EN00464B.

Johnson, A.C., Ball, H., Cross, R., Horton, A. A., Jürgens, M.D., Read, D.S., Vollertsen, J. & Svendsen, C. (2020). Identification and quantification of microplastics in potable water and their sources within water treatment works in England and Wales, *Environ. Sci. Technol*., 54, pp. 12326–12334. DOI: 10.1021/acs.est.0c03211.

Jung, J., Kim, S., Kim, Y., Jeong, S. & Lee, J. (2022). Tracing microplastics from raw water to drinking water treatment plants in Busan, South Korea, *Sci. Total Environ*., 825, 154015. DOI: 10.1016/j.scitotenv.2022.154015.

Kankanige, D. & Babel, S. (2020). Smaller-sized micro-plastics (MPs) contamination in single-use PET-bottled water in

Thailand, *Sci. Total Environ*., 717, 137232, DOI: 10.1016/j.scitotenv.2020.137232.

Kankanige, D. & Babel, S. (2021). Contamination by ≥6.5 μm-sized microplastics and their removability in a conventional water treatment plant (WTP) in Thailand, *Journal of Water. Process. Eng*., 40, 101765. DOI: 10.1016/j.jwpe.2020.101765.

Katrivesis, F.K., Karela, A.D., Papadakis, V.G. & Paraskeva, C.A., (2019). Journal of water process engineering revisiting of coagulation-flocculation processes in the production of potable water, *J. Water Process Eng*., 27, pp. 193–204. DOI: 10.1016/j.jwpe.2018.12.007.

Kernegger, L., Zweimüller, I. & Peduzzi, P. (2009). Effects of suspended matter quality and virus abundance on microbial parameters: experimental evidence from a large European river, Aquat Microb Ecol., 57(2), pp. 161-173. DOI: 10.3354/ame01341.O.

Kim, K.T. & Park, S. (2021). Enhancing microplastics removal from wastewater using electro-coagulation and granule-activated carbon with thermal regeneration, *Processes*, 9, 617. DOI: 10.3390/ pr904 0617.

Kirstein, I.V., Gomiero, A. & Vollertsen, J. (2021). Microplastic pollution in drinking water, *Current Opinion in Toxicology,* 28, pp. 70-75. DOI: 10.1016/j.cotox.2021.09.003.

Kirstein, I.V., Kirmizi, S.,Wichels, A.,Garin-Fernandez, A., Erler,R., Loder, M. & Gerdts, G. (2016). Dangerous hitchhikers? evidence for potentially pathogenic *Vibrio spp*. on microplastic particles, *Marine Environmental Research*, 120, pp. 1–8. DOI: 10.1016/j.marenvres.2016.07.004.

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, pp. 410-422. DOI:10.1016/j.watres.2019.02.054.

Koner, S., Florance, I., Mukherjee, A. & Chandrasekaran, N. (2023): Cellular response of THP-1 macrophages to polystyrene microplastics exposure, *Toxicology*, 483, 153385. DOI: 10.1016/j.tox.2022.153385.

Kosuth, M., Mason, S.A. & Wattenberg, E.V. (2018). Anthropogenic contamination of tap water, beer, and sea salt, *PLoS One*, 13, 0194970. DOI: 10.1371/journal.pone.0194970.

Kumar, M.V. (2021). Drinking water treatment with disinfection method: a review. *Int. J. Eng. Sci. Technol*.**,** 8(4), pp. 1-9. ISSN (online): 2349-784X.

Kumar, P., Sharma, Y., Singh, S. & Sharma, S. (2023). Microplastics in drinking water: assessing occurrence and potential risks, *J. Res. Appl. Sci. Biotechnol*., *2*(3), pp. 189–197. DOI: 10.55544/jrasb.2.3.25.

Kundu, A., Shetti, N.P., Basu, S., Raghava, K., Nadagouda, M.N. & Aminabhavi, T.M. (2021). Identification and removal of micro- and nano-plastics: efficient and cost-effective methods, *Chem. Eng. J.,* 421**,** 129816. DOI: 10.1016/j.cej.2021.129816.

Kurniawan, S.B., Said, N.S.M., Imron, M.F. & Abdullah, S.R.S. (2021). Microplastic pollution in the environment: Insights into emerging sources and potential threats, *Environ. Technol. Innov*., 23, 101790. DOI: 10.1016/j.eti.2021.101790.

Kutralam-Muniasamy, G., Shruti, V.C., Perez-Guevara, F. & Roy, P.D. (2023). Microplastic diagnostics in humans: "The 3Ps" Progress, problems, and prospects, *Sci. Total Environ*., 856, 159164. DOI: 10.1016/j.scitotenv.2022.159164.

Lam, T.W.L., Ho, H.T., Ma, A.T.H. & Fok, L., (2020). Microplastic contamination of surface water-sourced tap water in Hong Kong. A Preliminary study, *Applied Sci*., 10, 3463. DOI: 10.3390/app10103463.

Lapointe, M., Farner, J.M., Hernandez, L.M. & Tufenkji, N. (2020). Understanding and improving microplastic removal during water treatment: impact of coagulation and flocculation, *Environ. Sci. Technol*., 54, pp. 8719–8727. DOI: 10.1021/acs.est.0c00712.

Lee, J., Wang, J., Oh, Y. & Jeong, S. (2023). Highly efficient microplastics removal from water using in-situ ferrate coagulation: Performance evaluation by micro-Fourier-transformed infrared spectroscopy and coagulation mechanism. *Chem. Eng. J.*, 451, 2, 138556. DOI: 10.1016/j.cej.2022.138556.

Lenaker, P.L., Corsi, S.R. & Mason, S.A. (2020). Spatial distribution of microplastics in surficial benthic sediment of Lake Michigan and Lake Erie, *Environ. Sci. Technol*., 55, pp. 373– 384. DOI: 10.5066/P9WJUODZ.

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, pp. 133–142. DOI: 10.1016/j.envint.2017.01.018

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. DOI: 10.1016/j.envint.2022.107199.

Li, N., Liu, T., Guo, K., Zhu, J., Yu, G., Wang, S. & Ye, L. (2018). Effect of mono-(2-ethylhexyl) phthalate (MEHP) on proliferation of and steroid hormone synthesis in rat ovarian granulosa cells in vitro, *J. Cell. Physiol*., 233, pp. 3629–3637. DOI: 10.1002/jcp.26224.

Li, C., Busquets, R. & Campos, L.C. (2020a). Assessment of microplastics in freshwater systems: A review, *Sci, Total Environ*., 707, 135578. DOI: 10.1016/j.scito tenv.2019.135578.

Li, L, Liu, D, Song, K. & Zhou, Y. (2020b). Performance evaluation of MBR in treating microplastics polyvinylchloride contaminated polluted surface water, *Mar. Pollut. Bull*., 150, 110724. DOI: 10.1016/j.marpolbul.2019.110724.

Li, J., Wang, B., Chen, Z., Ma, B. & Chen, J.P. (2021). Ultrafiltration membrane fouling by microplastics with raw water: Behaviors and alleviation methods, *Chem. Eng. J.,* 410, 128174. DOI: 10.1016/j.cej.2020.128174

Li, H., Zhu, L., Ma, M., Wu, H., An, L. & Yang, Z., (2023). Occurrence of microplastics in commercially sold bottled water, *Sci. Total Environ*., 867, 161553. DOI: 10.1016/j.scitotenv.2023.161553.

Li, Y., Meng, Y., Qin, L., Shen, M.; Qin, T., Chen, X., Chai, B., Liu, Y., Dou, Y. & Duan, X. (2024a). Occurrence and removal efficiency of microplastics in four drinking water treatment plants in Zhengzhou, China, *Water*, 16, 131. DOI: 10.3390/w16010131.

Li, C., Busquets, R. & Campos L.C. (2024b). Enhancing microplastic removal from natural water using coagulant aids. *Chemosphere*, 364, 143145. DOI: 10.1016/j.chemosphere.2024.143145

Li, Y., Liu, C., Yang, H., He, W., Li, B., Zhu, X., Liu, S., Jia, S., Li, R. & Tang, K.H.D. (2024c). Leaching of chemicals from microplastics: a review of chemical types, leaching mechanisms and influencing factors, *Sci. Total Environ*., 906, 167666. DOI: 10.1016/j.scitotenv.2023.167666.

Lim, C., Kim, N., Lee, J. & Yoon, Y. (2022). Potential of adsorption of diverse environmental contaminants onto microplastics, *Water*, 14, 4086. DOI: 10.3390/w14244086.

Liu, Q. & Khor, S.M. (2024). Emerging absorption-based techniques for removing microplastics and nanoplastics from actual water bodies, *Trends Anal. Chem*., 170, 117465. DOI: 10.1016/j.trac.2023.117465.

Liu, S., Wu, X., Gu, W., Yu, J. & Wu, B., (2020). Influence of the digestive process on intestinal toxicity of polystyrene microplastics as determined by in vitro Caco-2 models, *Chemosphere,* 256, 127204. DOI: 10.1016/j.chemosphere.2020.127204.

Liu, X., Yuan, W., Di, M., Li, Z. & Wang, J., (2019). Transfer and fate of microplastics during the conventional activated sludge process in one wastewater treatment plant of China, *Chem. Eng. J*., 362, pp. 176–182. DOI: 10.1016/j.cej.2019.01.033.

Liu, Z., Sokratian, A., Duda, A.M., Xu, E., Stanhope, C., Fu, A., Strader, S., Li, H., Yuan, Y., Bobay, B.G. Sipe J., Bai K., Lundgaard I., Liu N., Hernandez B., Bowes Rickman C., Miller S.E. & West A.B. (2023). Anionic nanoplastic contaminants promote Parkinson's disease–associated α-synuclein aggregation, *Sci. Adv*., 9, eadi8716. DOI: 0.1126/sciadv.adi8716.

Lombardi G., Di Russo M., Zjalic D., Lanza T., Simmons M., Moscato U., Ricciardi W. & Chiara C. (2022). Microplastics inhalation and their effects on human health: a systematic review, *Eur. J. Public Health*, 32, 3. DOI: 10.1093/eurpub/ckac131.152.

Lu, S., Liu, L., Yang, Q., Demissie, H., Jiao, R., An, G. & Wang, D. (2021). Removal characteristics and mechanism of microplastics and tetracycline composite pollutants by coagulation process, *Sci.Total Environ*., 786, 147508. DOI: 10.1016/j.scitotenv.2021.147508.

Luo, W., Su, L., Craig, N.J., Du, F., Wu, C. & Shi, H. (2019). Comparison of microplastic pollution in different water bodies from urban creeks to coastal waters, *Environ. Pollut*., 246, pp. 174-182. DOI: 10.1016/j.envpol.2018.11.081.

Luogo, B.D.P., Salim, T., Zhang, W., Hartmann, N.B., Malpei, F. & Candelario V.M. (2022). Reuse of water in laundry applications with micro and ultrafiltration ceramic membrane, *Membranes*, 12, 223. DOI: 10.3390/membranes12020223.

Lv, L., Yan, X., Feng, .L, Jiang, S., Lu, Z., Xie, H., Sun, S., Chen, J. & Li, C. (2019). Challenge for the detection of microplastics in the environment, *Water Environ.Res*., 93. pp. 5–15. DOI: 10.1002/wer.128.

Ma, B., Xue, W., Ding, Y., Hu, C., Liu, H. & Qu, J. (2019a). Removal characteristics of microplastics by Fe-based coagulants during drinking water treatment, *J. Environ. Sci*., 78, pp. 267–275. DOI:10.1016/j.jes.2018.10.006.

Ma, B., Xue, W., Hu, C., Liu, H., Qu, J. & Li, L. (2019b). Characteristics of microplastic removal via coagulation and ultrafiltration during drinking water treatment, *Chem. Eng. J*., 359, pp. 159–167. DOI: 10.1016/j.cej.2018.11.155.

Ma, J., Ladd, D.M, Kaval, N. & Wang, H.-S. (2025). Toxicity of long term exposure to low dose polystyrene microplastics and nanoplastics in human iPSC-derived cardiomyocytes, *Food Chem. Toxico*l., 202, 115489. DOI: 10.1016/j.fct.2025.115489.

Ma, Y. & You, X. (2021). Modelling the accumulation of microplastics through food webs with the example Baiyangdian Lake. China, Sci.Total Environ., 762, 144110. DOI: 10.1016/j.scitotenv.2020.144110.

Magid, A.S.I.A., Islam, M.S., Chen, Y., Weng, L., Li, J., Ma, J. & Li, Y. (2021). Enhanced adsorption of polystyrene nanoplastics (PSNPs) onto oxidized corncob biochar with high pyrolysis temperature, *Sci. Total Environ*., 784, 147115. DOI: 10.1016/j.scitotenv.2021.147115.

Mahmood, Y.H., Al-hilali, B.M.I. & Kalaf, A.T. (2018). Concentration of residual chlorine and its health effects on the drinking water of the Kirkuk city, *Kafkas University Institute of Natural and Applied Science Journal,* 11. pp. 29–37, https://www.researchgate.net/publication/338459371

Mai, L., Bao, LJ., Shi, L. Wong, C.S. & Zeng E.Y. (2018). A review of methods for measuring microplastics in aquatic environments, *Environ. Sci. Pollut. Res*., 25, pp. 11319–11332. DOI: 10.1007/s11356-018-1692-0.

Malankowska, M., Echaide-Gorriz,l C. & Coronas, J. (2021). Microplastics in marine environment – sources, classification, and potential remediation by membrane technology – A review, *Environ. Sci., Water Res. Technol.,* **7**, pp. 243-258. DOI: 10.1039/D0EW00802H.

Marazuela, M.D., Klaiber, M., Moreno-Gordaliza, E., Barata, A., Gómez-Gómez M.M. (2022). Safety assessment of commercial antimicrobial food packaging: Triclosan and microplastics, a closer look. *Food Packag. Shelf Life,* 31, 100780. DOI: 10.1016/j.fpsl.2021.100780

Marsono, B.D., Yuniarto, A., Purnomo, A. & Soedjono, E.S. (2022). Comparison performances of microfiltration andrapid sand filter operated in water treatment plant, *IOP Conf. Ser.: Earth Environ. Sci*., 1111, 012048. DOI:10.1088/1755-1315/1111/1/012048.

Mason, S.A., Welch, V.G. & Neratko, J. (2018), Synthetic polymer contamination in bottled water, *Front. Chem*., 6, 407. DOI: 10.3389/fchem.2018.00407.

Massos, A. & Turner, A. (2017). Cadmium, lead and bromine in beached microplastics, *Environ. Pollution*, 227, pp. 139–145. DOI: 10.1016/j.envpol.2017.04.034.

Miao, F., Liu, Y., Gao, M., Yu, X., Xiao, P., Wang, M., Wang, S. & Wang, X. (2020). Degradation of polyvinyl chloride microplastics via an electro-Fenton-like system with a TiO2/graphite cathode, *J. Hazard. Mater*., 399, 123023. DOI: 10.1016/j.jhazmat.2020.123023.

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, pp. 631-635. DOI: 10.1016/j.scitotenv.2018.08.178.

Mintenig, S.M., Kooi, M., Erich, M., Redondo-Hasselerharm, P.E., Dekker, S.C., Koelmans, A.A. & van Wezel, A.P. (2020). A systems approach to understand microplastics measured in riverine surface waters and sediments, *Water Res*., 176, 115723. DOI: 10.1016/j.watres.2020.115723.

Moussa, D.T., El-Naas, M.H., Nasser, M. & Al-Marri, M.J. (2017). A comprehensive review of electrocoagulation for water treatment: potentials and challenges, *J. Environ. Manag*., 186, pp. 24-41. DOI: 10.1016/j.jenvman.2016.10.032.

Mukotaka, A., Kataoka, T. & Nihei, Y. (2021). Rapid analytical method for characterization and quantification of microplastics in tap water using a Fourier-transform infrared microscope, *Sci. Total Environ*., 790, 148231. DOI: 10.1016/j.scitotenv.2021.148231

Murphy, F., Ewins, C., Carbonnier, F. & Quinn, B. (2016). Wastewater treatment works (WwTW) as a source of microplastics in the aquatic environment, *Environ. Sci. Technol*., 50, pp. 5800–5808. DOI: 10.1021/acs.est.5b05416.

Na, S., Kim, M., Kim, J., Jeong, S., Lee, S., Chung, J. & Kim, E., (2021). Microplastic removal in conventional drinking water treatment processes: performance, mechanism, and potential risk, *Water Res*., 202, 117417. DOI: 10.1016/j.watres.2021.117417.

Najahi, H., Alessio, N., Venditti, M., Conti, G.O., Ferrante, M., Bernardo, G.D., Galderisi, U., Minucci, S. & Banni, M. (2025). Impact of Environmental Microplastic Exposure on Caco-2 Cells: Unraveling Proliferation, Apoptosis, and Autophagy Activation, *Res. Public Health*, 22, 922. DOI: 10.3390/ijerph2206092.

Namla, D., Oves, M., Alshaeri, M.A., Al-Maaqar, S M., Issa, H.N.Y. & Mangse, G. (2025). Nanofiltration as an advanced wastewater treatment technique: a comprehensive review, *Discover Applied Sciences*, 7, 355. DOI: 10.1007/s42452-025-06576-3.

Nirmala, K., Rangasamy, G., Ramya, M., Shankar, V.U. & Rajesh, G. (2023). A critical review on recent research progress on microplastic pollutants in drinking water, *Environ. Res.,* 222, 115312. DOI: 10.1016/j.envres.2023.115312.

Okoye, C.O., Addey, C.I., Oderinde, O., Okoro, J.O., Uwamungu, J.Y., Ikechukwu, C.K., Okeke, E.S., Ejeromedoghene, O. & Odii, E.C. (2022). Toxic chemicals and persistent organic pollutants associated with micro-and nanoplastics pollution, *Chem. Eng. J. Adv*., 11, 100310. DOI.:10.1016/j.ceja.2022.100310.

Ortiz, D., Munoz, M., Nieto-Sandoval, J., Romera-Castillo, C., de Pedro, Z.M. & Casas J.A. (2022). Insights into the degradation of microplastics by Fenton oxidation: From surface modification to mineralization, *Chemosphere*, 309, 136809. DOI: 10.1016/j.chemosphere.2022.136809.

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**,** pp. 307-316. DOI: 10.1016/j.watres.2018.05.027.

Ouyang, Z., Yang, Y., Zhang, C., Zhu, S., Qin, L., Wang, W., He, D., Zhou, Y., Luo, H. & Qin F. (2021). Recent advances in photocatalytic degradation of plastics and plastic-derived chemicals, *J. Mater. Chem. A,* 9, pp. 13402−13441. DOI: 10.1039/d0ta12465f.

Paray, B.A., Yu, J., Sultana,,S., Banik P., . Nur A.-A.U.,.Haque R. Rahman M., Arai T., Yan L. & Hossain M.B. (2024). Contamination, morphological and chemical characterization, and hazard risk analyses of microplastics in drinking water sourced from groundwater in a developing nation, *Front. Environ. Sci*., 12, 1379311. DOI: 10.3389/fenvs.2024.1379311.

Park, J.W., Lee, S., Hwang, D.Y. & Seo, S. (2021). Removal of microplastics via tannic acid-mediated coagulation and in vitro impact assessment, *RSC Advances*, 11, pp. 3556–3566. DOI: 10.1039/D0RA09645H.

Paliulis, D., Pranskevicius, M. & Taraseviciute D. (2025). Evaluation of microplastic pollution in the Vilnelė River, Archives Environ. Prot., 51(3), pp. 120–137. DOI:10.24425/aep.2025.156014.

Peixoto, D., Pinheiro, C., Amorim, J., Oliva-Teles, L., Guilhermino, L. & Vieira, M.N. (2019). Microplastic pollution in commercial salt for human consumption: a review, Estuarine. *Coastal and Shelf Science*, 219, pp. 161–168. DOI: 10.1016/j.ecss.2019.02.018.

Peng, G., Xiang, M., Wang, W., Su, Z., Liu, H., Mao, Y., Chen, Y. & Zhang, P. (2022). Engineering 3D graphene-like carbon-assembled layered double oxide for efficient microplastic removal in a wide pH range, *J. Hazard. Mater*., 433, 128672. DOI: 10.1016/j.jhazmat.2022.128672.

Perren, W., Wojtasik, A. & Cai, Q. (2018). Removal of microbeads from wastewater using electrocoagulation, *ACS Omega*, 3, pp. 3357-3364. DOI: 10.1021/acsomega.7b02037.

Peydayesh, M., Suta, T., Usuelli, M., Handschin, S., Canelli, G., Bagnani, M. & Mezzenga, R. (2021). Sustainable removal of microplastics and natural organic matter from water by coagulation-flocculation with protein amyloid fibrils, *Environ. Sci. Technol*., 55, pp. 8848–8858. DOI: 10.1021/acs.est.1c01918.

Pilco-Nuñez, A., Hinostroza-Antonio, E., Diaz-Bravo, P., Palacios-Salvador W., Solis-Toledo R. & Baldeon-Romero J. (2024). Removal of microplastics by electrocoagulation, Archives Environ. Prot., 50(4), pp. 64–71. DOI: 10.24425/aep.2024.152896.

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, pp. 1644–1651. DOI: 10.1016/j.scitotenv.2018.08.102.

Pivokonský, M., Pivokonská, L., Novotná, K., Čermáková, L. & Klimtová, M. (2020). Occurrence and fate of microplastics at two different drinking water treatment plants within a river catchment, *Sci. Total Environ*., 741, 140236. DOI: 10.1016/j.scitotenv.2020.140236.

Pivokonsky, M., Novotna, K., Petricek, R., Cermakova, L., Prokopova, M. & Naceradska, J. (2024). Fundamental chemical aspects of coagulation in drinking water treatment – Back to basics, *Journal of Water Process Engineering*, 57, 104660. DOI: 10.1016/j.jwpe.2023.104660.

Pizzichetti, A.R.P., Pablos, C., Álvarez-Fernández, C., Reynolds, K., Stanley, S. & Marugán, J. (2021). Evaluation of membranes performance for microplastic removal in a simple and low-cost filtration system, *Case Studies in Chem. Environ. Eng.,* 3, 100075. DOI: 10.1016/j.cscee.2020.100075.

Plastic the Fast Facts – Plastic Europe, (2025), https://plasticseurope.org/wp-content/uploads/2025/09/PE_TheFacts_25_digital-1pager-scrollable.pdf (10.03.2026).

Poleneni, S.R., Inniss, E., Shi, H., Yang, J., Huang, B. & Clamp, J. (2019). Enhanced flocculation using drinking water treatment plant sedimentation residual solids, *Water*, 11, 1821. DOI: 10.3390/w11091821.

Prüst, M., Meijer, J. & Westerink, R.H.S. (2020). The plastic brain: neurotoxicity of micro-and nanoplastics, *Part. Fibre Toxicol*., 17, 24. DOI: 10.1186/ s12989-020-00358-y.

Radityaningrum, A.D., Trihadiningrum, Y., Mar'atusholihah Soedjono E.S. & Herumurti W. (2021). Microplastic contamination in water supply and the removal efficiencies of the treatment plants: a case of Surabaya City, Indonesia, *J. Water Process Eng*., 43**,** 102195. DOI: 10.1016/j.jwpe.2021.102195.

Rahman, S., Saha, W., Maysha, T.I., Sarker, P., Datta, T.R., Rahman, S. & Chacrabati, R. (2026). Prevalence and health risks of microplastics in bottled water and beverages: A food safety concern, *J. Hazard. Mater. Plast.*, 2, 100024. DOI: 10.1016/j.hazmp.2025.100024.

Rai, P.K., Lee, J., Brown, R.J. & Kim, K.H. (2021). Environmental fate, ecotoxicity biomarkers, and potential health effects of micro- and nano-scale plastic contamination, *J. Hazard. Mater.,* 403, 123910. DOI: 10.1016/j.jhazmat.2020.123910.

Rajmohan, K.V.S., Ramya, C., Viswanathan, M.R. & Varjani, S. (2019). Plastic pollutants: effective waste management for pollution control and abatement, *Curr. Opin. Environ. Sci. Health*, 12, pp. 72-84. DOI: 10.1016/j.coesh.2019.08.006

Râpă, M., Darie-Niță, R.N., Matei, E., Predescu, A.-M., Berbecaru, A.-C. & Predescu, C. (2023). Insights into Anthropogenic Micro- and Nanoplastic Accumulation in Drinking Water Sources and Their Potential Effects on Human Health. *Polymers 15*, 2425. DOI: 10.3390/polym15112425

Rehman, K., Fatima, F.,Waheed, I. & Akash, M. S.H. (2017). Prevalence of exposure of heavy metals and their impact on health consequences, *Journal of Cellular Biochemistry*, 119, pp. 157–184. DOI: 10.1002/jcb.26234.

Rist, S., Almroth, B.C., Hartmann, N.B. & Karlsson, T.M. (2018). A critical perspective on early communications concerning human health aspects of microplastics, *Sci. Total Environ*., 626, pp. 720-726. DOI: 10.1016/j.scitotenv.2018.01.092.

Romphophak P., Faikhaw O., Sairiam S., Thuptimdang P. & Coufort-Saudejaud C. (2024). Removal of microplastics and nanoplastics in water treatment processes: A systematic literature review, J. *Water Process Eng*., 64, 105669. DOI: 10.1016/j.jwpe.2024.105669.

Rong, X., Chen, X., Li, P., Zhao, C., Peng, S., Ma, H. & Qu, H. (2022). Mechanically durable anti-bacteria non-fluorinated superhydrophobic sponge for highly efficient and fast microplastic and oil removal, *Chemosphere,* 299, 134493. DOI: 10.1016/j.chemosphere.2022.134493.

Rosales E., Meijide J., Pazos M. & Sanromán M.A. (2017). Challenges and recent advances in biochar as low-cost biosorbent: from batch assays to continuous-flow systems, *Bioresour. Technol*., 246, pp. 176-192. DOI: 10.1016/j.biortech.2017.06.084.

Rozman, U. & Kalcíkova, G., (2022). Seeking for a perfect (non-spherical) microplastic particle – the most comprehensive review on microplastic laboratory research, *J. Hazard. Mater*., 424, 27529. DOI: 10.1016/j.jhazmat.2021.127529

Sana, S.S., Dogiparthi, L.K., Gangadhar, L., Chakravorty, A. & Abhishek, N. (2020). Effects of microplastics and nanoplastics on marine environment and human health, *Environ. Sci. Pollut. Res*., 27, pp. 44743–44756. DOI. 10.1007/s11356-020-10573-x.

Sarkar, D.J, Das Sarkar, S., Das, B.K., Praharaj, J.K., Mahajan, D.K., Purokait, B., Mohanty, T.R., Mohanty D., Gogoi, P., Kumar, V.S., Behera, B.K., Manna, R.K. & Samanta S. (2021). Microplastics removal efficiency of drinking water treatment plant with pulse clarifier, *J. Hazard. Mat.,* 413, 125347. DOI: 10.1016/j.jhazmat.2021.125347.

Schirinzi, G.F., Perez-Pomeda, I., Sanchís, J., Rossini, C., Farre, M. & Barcelo, D., (2017). Cytotoxic effects of commonly used nanomaterials and microplastics on cerebral and epithelial human cells, *Environ. Res*., 159, 579. DOI: 10.1016/j.envres.2017.08.043.

Schwabl, P., Köppel, S., Königshofer, P., Bucsics, T., Trauner, M., Reiberger, T. & Liebmann, B. (2019). Detection of various microplastics in human stool: a prospective case series, *Ann. Intern. Med*., 171, pp. 453–457. DOI: 10.7326/M19-0618.

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. DOI: 10.1016/j.watres.2017.11.011.

Selvam, S., Jesuraja, K., Venkatramanan, S., Roy, P.D. & Jeyanthi Kumari, V. (2021). Hazardous microplastic characteristics and its role as a vector of heavy metal in groundwater and surface water of coastal South India, J. *Hazard. Mater*., 402, 123786. DOI: 10.1016/j.jhazmat.2020.123786.

Sembiring, E., Fajar, M. & Handajani, M. (2021). Performance of rapid sand filter – Single media to remove microplastics, *Water Supply*, 21, pp. 2273–2284. DOI: 10.2166/ ws. 2021. 060.

Semmouri, I., Vercauteren, M., Van Acker, E., Pequeur, E., Asselman, J. & Janssen, C., (2022). Presence of microplastics in drinking water from different freshwater sources in Flanders (Belgium), an urbanized region in Europe, *Int. J. Food Contam*., 9: 6. DOI: 10.1186/s40550-022-00091-8.

Shahi, N. K., Maeng, M., Kim, D. & Dockko, S. (2020). Removal behavior of microplastics using alum coagulant and its enhancement using polyamine-coated sand, *Process Safety and Environmental Protection*, 141, pp. 9–17. DOI: 10.1016/j.psep.2020.05.020.

Shen, M., Song, B., Zhu, Y., Zeng, G., Zhang, Y., Yang, Y., Wen, X., Chen, M. & Yi, H. (2020). Removal of microplastics via drinking water treatment: Current knowledge and future directions, *Chemosphere*, 251, 126612. DOI: 10.1016/j.chemosphere.2020.126612.

Shen, M., Zeng, Z., Wen, X., Ren, X., Zeng, G., Zhang, Y. & Xiao, R. (2021). Presence of microplastics in drinking water from freshwater sources: The investigation in Changsha, China, *Environ. Sci. Pollut. Res. Int*., 28, pp. 42313–42324. DOI:10.1007/s11356-021-13769-x.

Shen, M., Zhang, Y., Almatrafi, E., Hu, T., Zhou, C., Song, B., Zheng Z. & Zheng G. (2022). Efficient removal of microplastics from wastewater by an electrocoagulation process, *Chem. Eng. J*., 428, 131161. DOI: 10. 1016/j. cej. 2021. 131161.

Sheng, D.P.W., Bilad, M.R. & Shamsuddin, N., (2022). Assessment and optimization of coagulation process in water treatment plant: a review, *ASEAN J. Sci. Eng.,* 3, pp. 79–100. DOI: 10.17509/ajse.v3i1.45035.

Shi, X., Zhang, X., Gao, W., Zhang, Y. & He, D. (2022). Removal of microplastics from water by magnetic nano-Fe_3O_4, *Sci. Total Environ*., 802, 149838. DOI: 10.1016/j.scitotenv.2021.149838.

Shruti, V.C., Perez-Guevara, F. & Kutralam-Muniasamy, G. (2020). Metro station free drinking water fountain- A potential “microplastics hotspot” for human consumption, *Environ. Pollut*., 261, 114227. DOI: 10.1016/j.envpol.2020.114227.

Siegel H., Fischer F., Lenz R., Fischer D., Jekel M. & Labrenz M. (2021). Identification and quantification of microplastic particles in drinking water treatment sludge as an integrative approach to determine microplastic abundance in a freshwater river, *Environ. Pollut*., 286, 117524. DOI: 10.1016/j.envpol.2021.117524.

Silva, A.B., Bastos, A.S., Justino, C.I.L., da Costa, J.P., Duarte, A.C. & Rocha-Santos, T.A.P. (2018). Microplastics in the environment: challenges in analytical chemistry - a review, *Anal. Chim. Acta*., 9, 1017. DOI: 10.1016/j.aca.2018.02.043.

Singh, N., Khandelwal, N., Ganie, Z.A., Tiwari, E. & Darbha, G.K. (2021). Eco-friendly magnetic biochar: An effective trap for nanoplastics of varying surface functionality and size in the aqueous environment, *Chem. Eng. J*., 418,129405. DOI: 10.1016/j.cej.2021.129405.

Skaf, D.W., Punzi, V.L., Rolle, J.T. & Kleinberg, K.A. (2020). Removal of micron-sized microplastic particles from simulated drinking water via alum coagulation, *Chem. Eng. J.,* 386, 123807. DOI: 10.1016/j.cej.2019.123807.

Smith, M., Love, D.C., Rochman, C.M. & Neff, R.A. (2018). Microplastics in seafood and the implications for human health, *Current Environmental Health Reports,* 5, pp. 375–386. DOI: 10.1007/s40572-018-0206-z.

Sol, D., Laca, A.; Laca, A. & Díaz, M. (2021). Microplastics in Wastewater and Drinking Water Treatment Plants: Occurrence and Removal of Microfibres, *Appl. Sci*., 11, 10109. DOI: 10.3390/app112110109.

Su, L., Xue, Y., Li, L., Yang, D., Kolandhasamy, P., Li, D. & Shi, H. (2016). Microplastics in Taihu Lake, China, *Environ. Pollut*., 216, pp. 711-719. DOI: 10.1016/j.envpol.2016.06.036.

Su, L., Cai, H., Kolandhasamy, P., Wu, C., Rochman, C.M. & Shi, H. (2018). Using the Asian clam as an indicator of microplastic pollution in freshwater ecosystems, *Environ. Pollut*., 234, pp. 347-355. DOI: 10.1016/j.envpol.2017.11.075.

Sun, A. & Wang, W.-X. (2023). Human exposure to microplastics and its associated health risks, *Environ. Health,* 1, pp. 139–149. DOI: 10.1021/envhealth.3c00053.

Sun, C., Wang, Z., Chen, L. & Li, F. (2020). Fabrication of robust and compressive chitin and graphene oxide sponges for removal of microplastics with different functional groups, *Chem. Eng. J*., 393, 124796. DOI: 10.1016/j.cej.2020.124796.

Sun, C., Wang, Z., Zheng, H., Chen, L. & Li, F. (2021). Biodegradable and re-usable sponge materials made from chitin for efficient removal of microplastics, *J. Hazard. Mater*., 420, 126599. DOI: 10.1016/j.jhazmat.2021.126599.

Sun, J., Wang, Y., He, Y., Liu, J., Xu, L., Zeng, Z., Song, Y., Qiu, J., Huang, Z. & Cui, L. (2022). Effective removal of nanoplastics from water by cellulose/MgAl layered double hydroxides composite beads, *Carbohyd. Polym.,* 298, 120059. DOI: 10.1016/j.carbpol.2022.120059.

Tabatabaei, F., Mafigholami, R., Moghimi, H. & Khoramipoor, S. (2022). Effect of Fe and Al based coagulants and disinfectants on polyethylene microplastics removal in coagulation process through response surface methodology, *Water Sci. Technol*., 87**,** pp. 99–114. DOI: 10.2166/wst.2022.393.

Tagg, A., Harrison, J.P., Ju-Nam, Y., Sapp, M., Bradley, E.L., Sinclair, C.J. & Ojeda, J.J. (2017). Fenton's reagent for the rapid and efficient isolation of microplastics from wastewater, *Chem. Communications*, 53, pp. 372–375. DOI: 10.1039/c6cc08798a.

Talvitie, J., Mikola, A., Koistinen, A. & Setälä, O. (2017). Solutions to microplastic pollution – removal of microplastics from wastewater effluent with advanced wastewater treatment technologies, *Water Res*., 123, pp. 401–407. DOI: 10.1016/j.watres.2017.07.005.

Tamargo, A., Molinero, N., Reinosa, J.J., Reinosa J.J., Alcolea-Rodriguez V., Portela ., Bañares M.A., Fernández J.F. & Moreno-Arribas M.V. (2022). PET microplastics affect human gut microbiota communities during simulated gastrointestinal digestion, first evidence of plausible polymer biodegradation during human digestion, *Sci. Rep*., 12, 528, DOI: 10.1038/s41598-021-04489-w.

Tang, W., Li, H., Fei, L, Wei, B., Zhou, T. & Zhang, H. (2022). The removal of microplastics from water by coagulation: a comprehensive review, *Sci. Total Environ*., 851, 158224. DOI: 10.1016/j.scitotenv.2022.158224.

Tang, Y., Zhang, S., Su, Y., Wu, D., Zhao, Y. & Xie, B. (2021). Removal of microplastics from aqueous solutions by magnetic carbon nanotubes, *Chem. Eng. J*., 406, 126804. DOI: 10.1016/j.cej.2020.126804.

Tian, L., Kolvenbach, B., Corvini, N., Wang, S., Tavanaie, N., Wang, L., Ma, Y., Scheu, S., Corvini, P.F.X. & Ji, R. (2017). Mineralisation of 14C-labelled polystyrene plastics by Penicillium variabile after ozonation pre-treatment, *New Biotechnology*, 38(B), pp. 101-105. DOI: 0.1016/j.nbt.2016.07.008.

Tiwari, E., Singh, N., Khandelwal, N., Monikh, F.A. & Darbha, G.K. (2020). Application of Zn/Al layered double hydroxides for the removal of nano-scale plastic debris from aqueous systems, *J. Hazard. Mater*., 397, 122769. DOI: 10.1016/j.jhazmat.2020.122769.

Tofa, T.S., Kunjali, K.L., Paul, S. & Dutta, J. (2019). Visible light photocatalytic degradation of microplastic residues with zinc oxide nanorods, *Environ. Chem. Letters*, 17, pp. 1341–1346. DOI: 10.1007/s10311-019-00859-z.

Tong, H., Jiang, Q., Hu, X. & Zhong, X. (2020). Occurrence and identification of microplastics in tap water from China, *Chemosphere,* 252, 126493. DOI: 10.1016/j.chemosphere.2020.126493.

Triebskorn, R., Braunbeck, T., Grummt, T., Hanslik, L., Huppertsberg, S., Jekel, M., et al. (2019). Relevance of nano- and microplastics for freshwater ecosystems: a critical review, *Trac. Trends Anal. Chem*., 110, pp. 375-392. DOI:10.1016/j.trac.2018.11.023.

Tse, Y., Chan, S.M. & Sze, E.T., (2022). Quantitative assessment of full size microplastics in bottled and tap water samples in Hong Kong, *Int. J. Environ. Res. Public Health,* 19, 13432. DOI: 10.3390/ijerph192013432.

Uoginte, I., Pleskytė, S., Pauraitė, J. & Lujanienė, G. (2022). Seasonal variation and complex analysis of microplastic distribution in different WWTP treatment stages in Lithuania, *Environ. Monit. Assess*., 194, 829. DOI: 10.1007/s10661-022-10478-x.

Velasco, A.N., Gentile, S.R., Zimmermann, S. & Stoll, S. (2022). Contamination and removal efficiency of microplastics and synthetic fibres in a conventional drinking water treatment plant, *Front. Water* 4, 835451. DOI: 10.3389/FRWA.2022.835451.

Wang, W., Ndungu, A.W., Li, Z. & Wang, J. (2016). Microplastics pollution in inland freshwaters of China: A case study in urban surface waters of Wuhan, China, *Sci. Total Environ*., 575, pp. 1369-1374. DOI: 10.1016/j.scitotenv.2016.09.213

Wang, W., Yuan, W., Chen, Y. & Wang, J. (2018a). Microplastics in surface waters of dongting lake and hong lake, China, *Sci. Total Environ*., 633, pp. 539-545. DOI:10.1016/j.scitotenv.2018.03.211.

Wang, F., Wong, C.S., Da, C., Lu, X., Fei, W. & Zeng, E.Y. (2018b). Interaction of toxic chemicals with microplastics: a critical review, *Water Res*., 139, 208. DOI: 10.1016/j.watres.2018.04.003.

Wang, Y. & Qian, H. (2021). Phthalates and Their Impacts on Human Health, *Human Health. Healthcare*, 9, 603. DOI:10.3390/healthcare9050603.

Wang, Z., Lin, T. & Chen, W. (2020a). Occurrence and removal of microplastics in an advanced drinking water treatment plant (ADWTP), *Sci. Total Environ*., 700, 134520. DOI: 10.1016/j.scitotenv.2019.134520.

Wang, Z., Sedighi, M. & Lea-Langton, A. (2020b). Filtration of microplastic spheres by biochar: Removal efficiency and immobilisation mechanisms, *Water Res*., 184, 116165. DOI: 10.1016/j.watres.2020.116165.

Wang, Q., Bai, J., Ning, B., Fan, L., Sun, T., Fang, Y., Wu, J., Li, S., Duan, C. & Zhang, Y. (2020c). Effects of bisphenol A and nanoscale and microscale polystyrene plastic exposure on particle uptake and toxicity in human Caco-2 cells, *Chemosphere,* 254, 126788. DOI: 10.1016/j.chemosphere.2020.126788.

Wang. Y., Li Y., Tian. L., Ju. L. & Liu. Y. (2021a). The removal efficiency and mechanism of microplastic enhancement by positive modification dissolved air flotation, *Water Environ. Res*., 93. pp. 693–702, DOI: 10.1002/wer.1352.

Wang, Z., Sun, C., Li, F. & Chen, L. (2021b). Fatigue resistance, re-usable and biodegradable sponge materials from plant protein with rapid water adsorption capacity for microplastics removal, *Chem. Eng. J*., 415, 129006. DOI: 10.1016/j.cej.2021.129006.

Wang, J., Sun, C., Huang, Q.-X., Chi, Y. & Yan, J.-H. (2021c). Adsorption and thermal degradation of microplastics from aqueous solutions by Mg/Zn modified magnetic biochars, *J. Hazard. Mater*., 419, 126486. DOI: 10.1016/j.jhazmat.2021.126486.

Winiarska, E., Chaszczewska-Markowska, M., Ghete, D., Jutel, M. & Zemelka-Wiacek, M. (2024). Nanoplastics penetrate human bronchial smooth muscle and small airway epithelial cells and affect mitochondrial metabolism, *Int. J. Mol. Sci.*, 25, 4724. DOI: 10.3390/ijms25094724.

Wright, S.L. & Kelly, F.J. (2017). Plastic and human health: a micro issue? *Sci. Total Environ.*, 51, pp. 6634–6647. DOI: 10.1021/acs.est.7b00423.

Wu, B., Wu, X., L,u, S., Wang, Z. & Chen, L. (2019). Size-dependent effects of polystyrene microplastics on cytotoxicity and efflux pump inhibition in human Caco-2 cells, *Chemosphere*, 221, pp. 333–341. DOI: 10.1016/j.chemosphere.2019.01.056.

Wu, L., Zhong, Z., Wang, Z., Du, X., Tao, X., Zhou, J., Dang, Z. & Lu G. (2025). Antimony release from e-waste-derived microplastics in aqueous environments: Effect of plastic properties and environmental factors, *Environ. Pollut.*, 368, 125774. DOI: 10.1016/j.envpol.2025.125774.

Xia. Y., Xiang. X.M., Dong. K.Y., Gong. Y.Y. & Li. Z.J. (2020). Surfactant stealth effect of microplastics in traditional coagulation process observed via 3-D fluorescence imaging, *Sci. Total Environ*., 729, 138783. DOI: 10.1016/j.scitotenv.2020.138783.

Xu, L. & Wang, Z.F. (2020). Occurrence and removal of microplastics in a water treatment plant, *Water Purif. Technol*., 39, pp. 109–113. 120. DOI: 10.1016/j.scitotenv.2019.134520.

Xue, J., Peldszus, S., Van Dyke, M.I. & Huck, P.M. (2021). Removal of polystyrene microplastic spheres by alum-based coagulation-flocculation-sedimentation (CFS) treatment of surface waters, *Chem. Eng. J*., 422, 130023. DOI: 1016/j.cej.2021.130023.

Xue, J., Samaei, S.H.-A., Chen, J., Doucet, A. & Ng, K.T.W. (2022). What have we known so far about microplastics in drinking water treatment? A timely review, *Front. Environ. Sci. Eng*., 16, 58. DOI: 10.1007/s11783-021-1492-5.

Yen, P.-L., Hsu, C.-H., Huang, M.-L. & Liao, V.H.-C. (2022). Removal of nano-sized polystyrene plastic from aqueous solutions using untreated coffee grounds, *Chemosphere*, 286, 131863. DOI: 10.1016/j.chemosphere.2021.131863.

Yuan, W., Liu, X., Wang, W., Di, M. & Wang, J. (2019). Microplastic abundance, distribution and composition in water, sediments, and wild fish from Poyang Lake, China, *Ecotoxicol. Environ. Safe*, 170. pp. 180-187. DOI: 10.1016/j.ecoenv.2018.11.126.

Yousefi, M., Mazaheri, Y., Soltani, M., Rezagholizade-shirvan, A., Sadighara, P., Askari, E., Salehi, A., Alikord, M. & Shokri, S. (2025). Microplastics as emerging contaminants in the food chain: Assessing exposure and threats to consumers, *Food Bioscience*, 74, 108035. DOI: 10.1016/j.fbio.2025.108035.

Yuan, Z., Nag, R. & Cummins, E. (2022). Human health concerns regarding microplastics in the aquatic environment - from marine to food systems, *Sci. Total Environ*., 823, 153730. DOI: 10.1016/j.scitotenv.2022.153730.

Zhang, Y., Diehl, A., Lewandowski, A., Gopalakrishnan, K. & Baker, T. (2020a). Removal efficiency of micro- and nanoplastics (180 nm–125 µm) during drinking water treatment, *Sci. Total Environ*., 720, 137383. DOI: 10.1016/j.scitotenv.2020.137383.

Zhang, M., Li, J., Ding, H., Ding, J., Jiang, F., Ding, N.X. & Sun, C. (2020b). Distribution characteristics and influencing factors of microplastics in urban tap water and water sources in Qingdao, China, *Anal. Lett*., 53, pp. 1312–1327. DOI: 10.1080/00032719.2019.1705476.

Zhang, Y., Zhou, G., Yue, J., Xing, X., Yang, Z., Wang, X., Wang, Q. & Zhang, J. (2021). Enhanced removal of polyethylene terephthalate microplastics through polyaluminum chloride coagulation with three typical coagulant aids, *Sci. Total Environ*., 800, 149589. DOI: 10.1016/j.scitotenv.2021.149589.

Zhang, Y., Wang, S., Olga, V., Xue, Y., Lv, S., Diao, X., Zhang, Y., Han, Q. & Zhou, H. (2022). The potential effects of microplastic pollution on human digestive tract cells, *Chemosphere*, 291, 132714. DOI: 10.1016/j.chemosphere.2021.132714.

Zhang, J., Liu, Y., Zhao, L., Peng, C. & Wang, L. (2024). Microplastics and nanoplastics in drinking water and beverages: occurrence and human exposure, *J. Environ. Expo. Assess*., 3, 24. DOI: 10.20517/jeea.2024.37.

Zhao, Q., Zhu L., Weng J., Jin Z., Cao Y., Jiang H. & Zhang Z. (2023). Detection and characterization of microplastics in the human testis and semen, *Sci. Total Environ*., 877, 162713. DOI: 10.1016/j. scitotenv.2023.162713.

Zheng, B., Li, B., Wan, H., Lin, X. & Cai, Y. (2022). Coral-inspired environmental durability aerogels for micron-size plastic particles removal in the aquatic environment, *J. Hazard. Mater*., 431, 128611. DOI: 10.1016/j.jhazmat.2022.128611.

Zhou, G., Huang X., Xu H., Wang Q., Wang M., Wang Y., Li Q., Zhang Y., Ye Q. & Zhang J. (2022). Removal of polystyrene nanoplastics from water by CuNi carbon material: The role of adsorption, *Sci. Pollut. Res*., 820, 153190. DOI: 10.1016/j. scitotenv.2022.153190.

Zhuang, J., Rong, N., Wang, X., Chen, C. & Xu, Z. (2022). Adsorption of small size microplastics based on cellulose nanofiber aerogel modified by quaternary ammonium salt in water, *Sep. Purif. Technol*., 293, 121133. DOI:10.1016/j.seppur.2022.121133.

Ziccardi, L.M., Edgington, A., Hentz, K., Kulacki, K.J. & Driscoll, S.K. (2016). Microplastics as vectors for bioaccumulation of hydrophobic organic chemicals in the marine environment: a state of the science review, *Environmental Toxicology Chemistry,* 35, pp. 1667–1676. DOI: 10.1002/etc.3461.

Zuccarello, P., Ferrante, M., Cristaldi, A., Copat, C., Grasso, A., Sangregorio, D., Fiore, M. & Conti, O.G. (2019). Exposure to microplastics (10 µm) associated to plastic bottles mineral water consumption the first quantitative study, *Water Research,* 157, pp. 365-371. DOI: 10.1016/j.watres.2019.03.091.

Mikro- i nanoplastiki w wodzie do picia: przegląd literatury

Streszczenie. Zanieczyszczenie wody tworzywami sztucznymi staje się coraz bardziej niepokojące ze względu na coroczny wzrost globalnej produkcji tworzyw sztucznych. Mikro- i nanoplastiki (MNP) wykryto w źródłach wody pitnej i wodzie z kranu, co budzi obawy dotyczące skuteczności stacji uzdatniania wody pitnej oraz zdrowia publicznego osób spożywających wodę z kranu lub butelkowaną. Niniejszy artykuł przedstawia systematyczny przegląd dostępnej wiedzy na temat globalnego występowania MNP w źródłach wody pitnej i wodzie uzdatnionej, wodzie z kranu i/lub wodzie butelkowanej. Przedstawiono dane dotyczące występowania i właściwości MNPs (rodzaj polimeru, kształt i rozmiar). Następnie omówiono wiedzę na temat skuteczności usuwania MNPs w różnych procesach jednostkowych stosowanych do uzdatniania wody pitnej, w tym koagulacji, flokulacji, sedymentacji, filtracji piaskowej i membranowej, adsorpcji, zaawansowanych procesach utleniania i dezynfekcji. Zwraca się również uwagę na zagrożenia dla zdrowia, jakie mogą stwarzać MNP obecne w wodzie pitnej. Obecnie nadal wydają się istnieć luki w badaniach nad usuwaniem MNP w procesie uzdatniania wody pitnej. Ponadto, omówiono potencjalne wyzwania, strategie i potrzeby badawcze związane z występowaniem i usuwaniem MNP w procesach jednostkowych stosowanych w produkcji wody pitnej.