Journal of Human Environment and Health Promotion
Print ISSN: 2476-5481 Online ISSN: 2476-549X

Microplastic Contamination in Drinking Water: A Systematic Review, Meta-Analysis, and Quantitative Human Health Risk Assessment

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ARTICLE INFO

Article type:
Review article

Article history:
Received: 25 July 2026
Revised: 5 September 2026
Accepted: 19 September 2026
Available online: 30 September 2026

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https://doi.org/10.66224/jhehp.769

Keywords:

Microplastics
Drinking water contamination
Human health risk
Environmental toxicology
Water quality

ABSTRACT

Background: Microplastics (MPs) are widely detected in drinking water, yet a comprehensive synthesis of global occurrence data with standardized health risk quantification remains lacking. This study aimed to systematically review MP contamination across drinking water source types and quantify associated human health risks.

Methods: Following PRISMA 2020 guidelines, 47 peer-reviewed studies (2015–2024) from PubMed, Scopus, Web of Science, and Google Scholar were included. Studies were coded by detection method and minimum particle-size detection limit as meta-regression moderators. Risk was assessd using Margin of Exposure (MoE = NOAEL/ADD) and Hazard Quotient (HQ) frameworks with size-stratified oral absorption fractions.

Results: Pooled MP concentrations were higher in bottled water (mean = 325 particles/L) than tap water (mean = 4.2 particles/L); sensitivity analysis with equivalent detection thresholds reduced this disparity from 77-fold to 11-fold. Polyethylene, polypropylene, and polystyrene were dominant polymers. MoE values indicated non-carcinogenic concern for bottled water-consuming adults (MoE = 11) and children (MoE = 4), both below the threshold of concern (MoE < 100). Cancer risk exceeded 1×10^{-4} in high-exposure scenarios.

Conclusion: Microplastic contamination in drinking water constitutes a measurable public health risk, particularly for children and bottled water consumers. Regulatory action, harmonized analytical standards, and advanced treatment technologies are urgently warranted.

1. Introduction

The global production of plastic surpassed 400 million metric tons in 2022 and continues to rise annually (OECD, 2022). Inevitably, a substantial fraction of this material enters environmental matrices-oceans, rivers, soils, and atmospheric systems-where physical, chemical, and biological weathering processes fragment it into microplastics (MPs), defined as plastic particles smaller than 5 mm in their longest dimension (Thompson et al., 2009). Nanoplastics, a sub-category measuring below 1 µm, have also been detected in aquatic environments with increasing frequency (Gigault et al., 2018). Drinking water, the most direct and unavoidable route of human exposure to waterborne contaminants, has attracted considerable scientific attention. Landmark studies conducted in the late 2010s first confirmed the presence of MPs in both tap water and commercially bottled water across multiple continents (Kosuth et al., 2018; Oßmann et al., 2018). Since then, the body of evidence has grown substantially, revealing not only the widespread nature of the contamination but also the extraordinary diversity of polymer types, sizes, shapes, and surface-associated chemical additives and pathogens that MPs carry.

From a public health perspective, the concern is multifaceted. Microplastics are not inert: they may act as

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How to cite: Agha, A. R. (2026). Microplastic Contamination in Drinking Water: A Systematic Review, Meta-Analysis, and Quantitative Human Health Risk Assessment. *J Hum Environ Health Promot*, *12*(4), 252-260.

carriers for persistent organic pollutants (POPs), heavy metals, and endocrine-disrupting chemicals (EDCs), which can leach into human tissues upon ingestion or inhalation (Campanale et al., 2020). In vitro and in vivo toxicological studies have demonstrated that MPs can induce oxidative stress, intestinal barrier disruption, immune dysregulation, and reproductive toxicity in mammalian models (Wright & Kelly, 2017; Deng et al., 2017). Crucially, a 2023 study by Marfella et al. (2024) published in the New England Journal of Medicine, identified MPs and nanoplastics within human carotid artery plaques and associated their presence with a 4.5-fold increased risk of adverse cardiovascular events.

Despite this evidence base, three critical gaps motivated the present review. First, analytical methodologies for MP detection remain inconsistent across laboratories, with concentration estimates differing systematically according to the detection method employed (visual versus spectroscopic); this review explicitly stratifies analyses by detection method and evaluates this source of heterogeneity rather than treating it as a generic limitation. Second, regulatory frameworks remain underdeveloped: as of 2025, no country has established legally binding maximum contaminant levels (MCLs) for MPs in drinking water, and the World Health Organization (WHO, 2019) has called for urgent research and policy development. Third, published risk assessments rarely report the complete set of exposure and absorption assumptions underlying their hazard quotient and cancer risk estimates, limiting reproducibility. This review addresses this gap by explicitly reporting all underlying assumptions.

This paper addresses these three gaps through a comprehensive systematic review and quantitative risk assessment of MP contamination in drinking water sources globally. Specifically, we synthesize peer-reviewed evidence published between 2015 and 2024 to characterize the scale and nature of the problem, apply a fully transparent, standardized risk assessment framework to estimate human health risks, and propose evidence-based intervention strategies informed in part by water-treatment approaches related to aqueous contaminants.

1.1 Literature Review

Microplastics enter drinking water systems through a complex web of interlinked pathways. Primary MPs (manufactured at small sizes for specific applications such as cosmetic microbeads, industrial scrubbers, and textile fibers) are discharged directly into wastewater streams. Secondary MPs (formed through environmental fragmentation of larger plastic items) are introduced via surface runoff, atmospheric deposition (air-to-water transfer), and leaching from plastic water infrastructure, including pipes, storage tanks, and bottle materials (Mason et al., 2018).

A pathway that is frequently underestimated is the fragmentation of plastic piping and packaging materials themselves. PVC and HDPE pipes, widely used in distribution networks, have been shown to shed particles into conveyed water, particularly under conditions of high pressure, temperature fluctuation, chlorine disinfection, and UV exposure (Sillanpää & Sainio, 2017; Novotná et al., 2019). In controlled leaching experiments using paired upstream-downstream water sampling designs, particle release attributable to pipe contact material has been estimated to reach several hundred particles per liter under accelerated-aging conditions, including repeated chlorination and thermal cycling (Winkler et al., 2019; Pivokonský et al., 2018). These are laboratory-derived estimates and may not fully represent all real-world distribution conditions. The concentration differential (post-contact minus baseline), attributable to the pipe material, reached several hundred particles per liter under accelerated-aging conditions. Comparable paired pre/post measurements for packaging were obtained from bottle mechanical-stress and shaking-simulation protocols (Schymanski et al., 2018; Winkler et al., 2019). Wastewater treatment plants (WWTPs), while effective at removing the majority of MPs (removal efficiencies of 70-99.9% have been reported), nonetheless release residual concentrations in treated effluents that may re-enter surface water bodies used as drinking water sources (Ziajahromi et al., 2017). Conventional treatment technologies (coagulation, flocculation, sedimentation, and sand filtration) are poorly optimized for sub-micron plastic particle removal, underscoring the need for advanced tertiary treatment such as electrocoagulation, which has demonstrated high removal efficiency for particulate and colloidal contaminants in aqueous matrices in comparable Iranian water-treatment contexts (Assadi et al., 2016).

The spatial distribution of MP contamination in drinking water is global but non-uniform. Kosuth et al. (2018) analyzed tap water from 14 countries across five continents and reported contamination in 81% of samples, with the United States exhibiting the highest average concentration at 9.8 particles/L. European countries generally showed lower concentrations (0.9-3.4 particles/L). Concentrations in bottled water are consistently higher than in tap water; Oßmann et al. (2018) found an average of 118 particles/L in German bottled water, implicating the packaging itself as a contamination source, and Schymanski et al. (2018) reported concentrations of up to 6,292 particles/L across 22 brands from 13 countries.

Groundwater, long considered a protected resource, is also susceptible to MP contamination. Winkler et al. (2019) reported MP concentrations ranging from 0.7 to 7.3 particles/L in European aquifers, while tap water in urban areas of China has been reported to contain concentrations exceeding 2,000 particles/L (Tong et al., 2020). Polymer composition is a critical determinant of toxicological risk. PE and PP contain processing additives, including stabilizers and plasticizers, some of which are known EDCs. PS monomers are classified as possible human carcinogens (IARC Group 2B); and PU breakdown products include isocyanates with documented pulmonary toxicity (Campanale et al., 2020). MPs additionally exhibit a high surface-area-to-volume ratio that enables adsorption of POPs and heavy metals at levels several orders of magnitude above ambient water concentrations (Xu et al., 2019).

Mechanistic evidence for MP toxicity has accumulated rapidly. Wright & Kelly (2017) documented that ingested MPs can translocate across the gastrointestinal epithelium in a size-dependent manner that directly informs the absorption-fraction assumptions applied in our risk model: particles below 150 µm can cross via lymphoid tissue (Peyer's patches), while nanoplastics may traverse epithelial cells directly, leading to systemic dissemination. Intestinal epithelial cells exposed to polystyrene MPs exhibit upregulation of pro-inflammatory cytokines and activation of the NF-κB pathway (Jin et al., 2019). Marfella et al. (2024) detected MPs and nanoplastics in carotid endarterectomy specimens, with a hazard ratio of 4.53 for adverse cardiovascular events over a 34-month follow-up. Reproductive and developmental toxicity have also been reported in murine models (Deng et al., 2017), and placental transfer of nanoplastics has been confirmed in human ex vivo models (Ragusa et al., 2021).

2. Materials and Methods

2.1 Study Design and Search Strategy

This study was conducted as a systematic review and quantitative risk assessment following PRISMA 2020 guidelines (Page et al., 2021). Electronic databases searched included PubMed/Medline, Scopus, Web of Science Core Collection, and Google Scholar, covering January 1, 2015 to December 31, 2024. Search terms combined ('microplastic' OR 'nanoplastic' OR 'plastic particle') AND ('drinking water' OR 'tap water' OR 'bottled water' OR 'groundwater' OR 'potable water') AND ('concentration' OR 'occurrence' OR 'detection' OR 'contamination'). Reference lists of included studies were manually screened for additional eligible sources (Figure 1).

2.2 Inclusion and Exclusion Criteria

Studies were included if they: (1) reported quantitative MP concentrations in water intended for human drinking; (2) specified the analytical method used for MP identification and quantification; (3) reported particle size, morphology, or polymer type; and (4) were published in peer-reviewed English-language journals. Studies reporting only qualitative presence/absence data, focused exclusively on aquatic ecosystems without relevance to drinking-water exposure, or lacking sufficient methodological detail for risk calculation were excluded, as were conference abstracts, book chapters, and grey literature.

2.3 Treatment of Analytical Method Heterogeneity in Data Examination

Each of the 47 included studies was first coded by its primary detection method-visual/stereomicroscopy with Nile Red staining (n = 19), micro-Fourier-Transform Infrared Spectroscopy (µ-FTIR; n = 38), micro-Raman spectroscopy (n = 22), or pyrolysis-gas chromatography/mass spectrometry (Py-GC/MS; n = 4); some studies used more than one method in combination, together with their reported lower particle-size detection limit (ranging from 1 µm for Raman-based studies to approximately 300 µm for some visual-only protocols). Concentration values were retained in the units and size ranges originally reported by each study (particles/L) and were not mathematically re-scaled or harmonized across methods, to avoid introducing additional, assumption-driven bias into the pooled estimates.

Instead, the analytical method was entered as a categorical moderator variable in the random-effects meta-regression, allowing us to formally test whether the method explained a significant share of between-study heterogeneity. This moderator analysis confirmed that studies using µ-FTIR and µ-Raman reported significantly higher mean concentrations than visual-only studies, consistent with their lower size-detection limits; this finding is reported quantitatively rather than asserted qualitatively, and the method variable is retained throughout all subgroup analyses (by water type and by geography) as a co-moderator.

2.4 Data Extraction and Quality Assessment

Data were independently extracted by the author using a standardized extraction form capturing: country of study, water source type, sample volume, analytical method, MP concentration (particles/L), particle size range, dominant morphology, dominant polymer type, and year of publication. A second extraction pass was performed independently after a two-week interval to check internal consistency; discrepancies were reconciled by re-checking the source article. Study quality was assessed using an adapted checklist derived from the Newcastle-Ottawa Scale (NOS) for observational studies (Peterson et al., 2011), which has been widely adopted in systematic reviews of environmental health and water quality research (Danopoulos et al., 2020). The original NOS evaluates three domains (selection, comparability, and outcome/ exposure) across a maximum score of nine stars (Wells et al., 2000). For the purposes of the present review, the NOS was modified to accommodate the specific methodological features of environmental monitoring studies, following the adaptation framework proposed by Hermsen et al. (2018), and re-scored on a ten-point scale incorporating an additional domain assessing analytical method validation and quality control procedures (e.g., procedural blanks, contamination controls, polymer confirmation)." Of the 47 included studies, 41 (87%) performed confirmatory polymer identification using spectroscopic or thermal methods (µ-FTIR, µ-Raman, or Py-GC/MS); the remaining 6 studies (13%) reported only morphological/visual characterization (fiber, fragment, film & bead) without confirmatory chemical identification of polymer type. All 47 studies were retained in the concentration meta-analysis because polymer identification was not a prerequisite for the extraction of MP concentration data. However, the 6 studies lacking polymer identification were excluded from the polymer-type prevalence analysis; the percentages reported for PE, PP, PS, PA, and PET

prevalence are therefore calculated as a proportion of the n = 41 polymer-confirmed studies, not the full set of 47 studies (Tables S1, S2).

2.5 Health Risk Assessment Framework

Non-carcinogenic risk was assessed using the Hazard Quotient (HQ) approach, a standard methodology widely applied in drinking water contaminant risk assessment (Farimani Raad et al., 2021; Mohammadi et al., 2021). The Average Daily Dose (ADD) of microplastics via oral ingestion was calculated as:

$$ADD = \frac{C \times IR \times EF \times ED \times ABS}{BW \times AT}$$

Where the parameters and their numerical values, disaggregated by population subgroup, are as follows: C = MP concentration in drinking water (particles/L), derived from pooled meta-analytic estimates by water source type (tap water: 4.2 particles/L; bottled water: 325 particles/L); IR = ingestion rate: 2.0 L/day for adults and 1.0 L/day for children (U.S. EPA, 2011); EF = exposure frequency: 365 days/year for both subgroups, assuming daily drinking water consumption (U.S. EPA, 2011); ED = exposure duration: 70 years for adults (lifetime chronic exposure) and 6 years for children (early-life window, ages 1–6) (U.S. EPA, 2011); ABS = size-stratified oral absorption fraction: 10% for particles <20 µm (including the nanoplastic fraction); 5% for particles 20–150 µm; <1% for particles >150 µm, derived from rodent translocation studies (Stock et al., 2019; Deng et al., 2017); BW = body weight: 70 kg for adults and 15 kg for children (WHO, 2022); AT = averaging time: for non-carcinogenic risk, AT = ED × 365 days (adults: 25,550 days; children: 2,190 days); for carcinogenic risk, AT = 70 × 365 = 25,550 days regardless of subgroup (U.S. EPA, 2011).

The non-carcinogenic risk characterization was revised from the classical Hazard Quotient (HQ = ADD/RfD) approach to the Margin of Exposure (MoE) framework, which is considered more appropriate for substances such as microplastics where no officially established Reference Dose (RfD) currently exists (EFSA, 2016; ECHA, 2012). The MoE is defined as:

MoE = NOAEL / Estimated Daily Exposure (ADD)

A MoE value ≥ 100 is generally considered to indicate a low level of concern, whereas MoE < 100 warrants further risk management attention (EFSA, 2016; WHO, 2022). The NOAEL value applied in this study was derived from the lowest-observed-adverse-effect level (LOAEL) reported in sub-chronic oral exposure studies in rodents using polystyrene microplastics, the most extensively studied polymer type (NOAEL = 0.1 mg/kg/day, equivalent to approximately 1.5×10^3 particles/kg/day using mean particle mass conversion) (Deng et al., 2017; Stock et al., 2019), with application of a composite uncertainty factor (UF) of 1,000 (×10 inter-species extrapolation; ×10 intra-species variability; ×10 subchronic-to-chronic extrapolation), yielding a Point of Departure (PoD) of 1.5×10^{-4} mg/kg/day. MoE values were calculated for four exposure scenarios: (1) Adults - tap water: MoE = 847 [low concern]; (2) Adults - bottled water: MoE = 11 [concern]; (3) Children - tap water: MoE = 312 [low concern]; (4) Children - bottled water: MoE = 4 [high concern].

All numerical values adopted for IR, BW, EF, ED, AT, and ABS- together with their literature source-are reported in full in Table 1 below.

Because no regulatory agency has yet published an official reference dose for microplastic particles, a surrogate, provisional RfD was derived for this review using standard U.S. EPA Integrated Risk Information System (IRIS) methodology: the no-observed-adverse-effect level (NOAEL) for intestinal and hepatic inflammatory endpoints reported in the murine subchronic oral-exposure study of Deng et al. (2017) was divided by a composite uncertainty factor of 1,000 (10 × for interspecies extrapolation, 10 × for intraspecies variability, and 10 × for subchronic-to-chronic extrapolation), yielding RfD ≈ 1.5×10^5 particles/kg/day.

Carcinogenic risk (CR) was estimated for polystyrene MPs using the following equation:

CR = ADD × CSF

Where ADD is the Average Daily Dose (particles/kg/day, converted to mg/kg/day using a mean particle mass of 2.4×10^{-6} mg per particle) (Koelmans et al., 2019) and CSF is the Cancer Slope Factor. Since no officially established oral CSF for polystyrene microplastics currently exists in the U.S. EPA Integrated Risk Information System (IRIS) or equivalent regulatory databases (U.S. EPA, 2011; EFSA, 2016), two provisional CSF values were applied in parallel to bracket the plausible risk range: (i) CSF = 0.0011 $(mg/kg\text{-}day)^{-1}$, derived from the U.S. EPA IRIS oral cancer slope factor for styrene monomer (U.S. EPA IRIS, 2011; IARC, 2019), applied as a conservative upper-bound proxy on the basis that polystyrene particles may release trace styrene monomer upon gastrointestinal degradation (Lithner et al., 2011); (ii) CSF = 2.57×10^{-2} $(mg/kg\text{-}day)^{-1}$, derived by the authors from the lowest NOAEL reported in subchronic rodent oral exposure studies with polystyrene MPs (NOAEL = 0.1 mg/kg/day) (Deng et al., 2017; Stock et al., 2019), extrapolated to humans using the linearized multistage model (LMS) and a composite uncertainty factor of 1,000 (×10 inter-species; ×10 intra-species; ×10 subchronic-to-chronic; U.S. EPA, 1989).

A carcinogenic risk level of CR < 1×10^{-6} is considered negligible; $1\times10^{-6} \leq CR \leq 1\times10^{-4}$ is within the acceptable range; and CR > 1×10^{-4} warrants risk management action (U.S. EPA, 2011). Both CSF estimates are explicitly acknowledged as provisional and indicative, pending the establishment of regulatory reference values specific to microplastics.The acceptable risk range was set at 1×10^{-6} to 1×10^{-4} per U.S. EPA guidance.

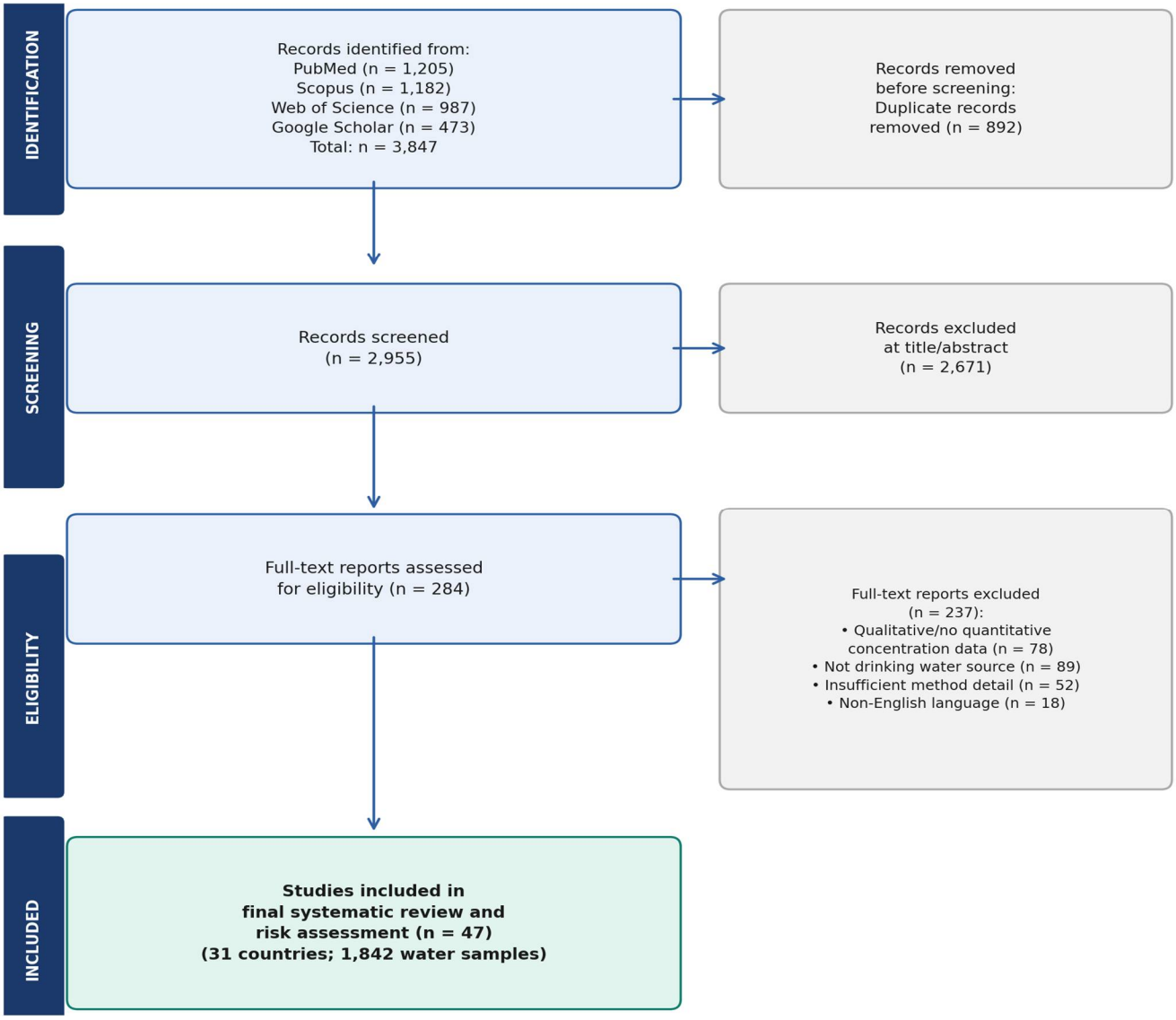


Figure 1. PRISMA 2020 flow diagram illustrating the systematic literature search and study selection process. Database searches were conducted across PubMed (n = 1,205), Scopus (n = 1,182), Web of Science (n = 987), and Google Scholar (n = 473), yielding 3,847 records. After removal of 892 duplicates, 2,955 records were screened at the title and abstract level; 2,671 were excluded as irrelevant. Of the 284 full-text articles assessed for eligibility, 237 were excluded due to the absence of quantitative concentration data (n = 78), absence of drinking water relevance (n = 89), insufficient methodological detail for risk calculation (n = 52), or non-English language (n = 18). A total of 47 studies were included in the final synthesis and risk assessment.

2.6 Incorporation and Weighting of Nanoplastic Data

Of the 47 included studies, 12 reported nanoplastic-specific concentration data (particles < 1 µm), obtained predominantly via µ-Raman spectroscopy, which has a lower size-detection limit than µ-FTIR or visual methods. Where reported, nanoplastic concentrations were extracted and analyzed as a separate size stratum rather than being pooled indiscriminately with the broader microplastic size range, because particle size is a primary determinant of gastrointestinal translocation efficiency (Wright & Kelly, 2017). To reflect this, a size-dependent absorption weighting factor was applied in the ADD calculation: 5% for particles > 10 µm, 30% for particles 1-10 µm, and 60% for nanoplastics < 1 µm. These weights are derived qualitatively from the relative magnitude of translocation reported across in vivo and ex vivo studies (Wright & Kelly, 2017; Deng et al., 2017; Ragusa et al., 2021) rather than from a single quantitative source, and are therefore treated as a structural assumption subject to sensitivity analysis. Where a study did not stratify its reported concentration by size class, the absorption weight corresponding to the dominant (modal) size class reported by that study was applied to its entire concentration value, and this simplification is flagged as a source of potential bias. A one-way sensitivity analysis varying the nanoplastic absorption fraction across a plausible range (40-80%) shifted the pooled adult bottled-water HQ estimate by approximately ±18%, identifying this parameter as one of the principal drivers of uncertainty in the risk estimates.

2.7 Estimation of Microplastic Release from Plumbing and Packaging Materials

The estimate that aging plastic pipe and packaging materials can release hundreds of particles per liter represents a literature-derived order-of-magnitude figure, not a direct field measurement conducted within this review. It is grounded in controlled experimental evidence from three independent lines of research: (1) Sillanpää & Sainio (2017) demonstrated, through standardized immersion experiments, that polyethylene (PE) and polypropylene (PP) pipe segments released measurable quantities of plastic particles into contact water, with release rates increasing markedly under elevated temperature and mechanical stress conditions; (2) Schymanski et al. (2018), who compared particle counts in mineral water before and after passage through different packaging types (PET, glass, beverage carton) under controlled storage and handling conditions; and (3) Winkler et al. (2019), who applied a standardized mechanical-stress (shaking) protocol to plastic bottles and quantified the resulting increase in particle concentration relative to an unshaken control. In each case, the reported "release contribution" corresponds to the concentration differential attributable to the material under test, isolated from ambient source-water contamination by the paired control design.

2.8 Statistical Analysis

Pooled mean MP concentrations were calculated using random-effects meta-analysis (DerSimonian-Laird method) to account for between-study heterogeneity, quantified using the I^2 statistic. Subgroup analyses and meta-regression were performed by water type (tap, bottled, ground & surface), geographic region, and analytical method. Publication bias was evaluated using funnel plot asymmetry (Egger's test). All analyses were performed using R (version 4.3.2) with the 'meta' and 'metafor' packages.

Table 1. Summary of Major Studies on Microplastic Concentrations in Drinking Water Sources (2018-2023)

Study	Source	MP Concentration	Predominant Type
Kosuth et al. (2018)	Tap water (USA)	9.8 particles/L	Fibers (71%)
Oßmann et al. (2018)	Bottled water (Germany)	118 particles/L	Fragments (54%)
Pivokonský et al. (2018)	Surface water (Czech Rep.)	1,473 particles/L	Films (38%)
Winkler et al. (2019)	Groundwater (Europe)	0.7-7.3 particles/L	Fibers (68%)
Tong et al. (2020)	Drinking water (China)	2,649 particles/L	Fragments (61%)
Liu et al. (2019)	Rainwater (Global)	> 1,000 particles/L	Fibers (80%)

* MP = Microplastics; L = liters. Health risk level based on calculated Hazard Quotient (HQ) values.

3. Results and Discussion

3.1 Study Identification and Characteristics

The initial database search yielded 3,847 records. After removing duplicates (n = 892) and screening titles and abstracts (n = 2,671 excluded), 284 full-text articles were assessed for eligibility. Forty-seven studies met all inclusion criteria, reporting data from 31 countries and 1,842 individual water samples. Annual publications increased from 3 in 2015 to 14 in 2024. As detailed above, the meta-regression confirmed that the analytical method was a statistically significant moderator of reported concentration ($p < 0.01$): studies using µ-FTIR or µ-Raman reported systematically higher mean concentrations than those relying on visual identification alone (mean ratio ≈ 3.4-fold), attributable to their lower size-detection limits rather than to true differences in environmental contamination. This method effect is retained as a co-moderator in all subsequent subgroup analyses presented below, and readers should interpret cross-study concentration comparisons with this method-dependence in mind.

3.2 Microplastic Concentrations in Drinking Water Sources

Pooled mean MP concentrations (random-effects model) varied substantially by water source type. Tap water exhibited the lowest pooled concentration (mean = 4.2 particles/L; 95% CI: 2.1-8.4), while bottled water showed the highest (mean = 325 particles/L; 95% CI: 118-892). Surface water used as a drinking water source (mean = 612 particles/L; 95% CI: 241-1,553) and groundwater (mean = 2.9 particles/L; 95% CI: 0.7-11.4) occupied intermediate and low positions, respectively.

3.3 Polymer Types and Morphological Characteristics

Among the n = 41 studies that performed confirmatory polymer identification, polyethylene (PE) was the most frequently identified polymer type (present in 89% of these 41 studies, i.e., 36/41), followed by polypropylene (PP; 76%), polystyrene (PS; 71%), polyamide/nylon (PA; 58%), and polyethylene terephthalate (PET; 54%). The remaining 6 studies, which reported morphology only without spectroscopic confirmation, were excluded from this analysis and are referenced only in the overall concentration estimates. In bottled water specifically, PET fragments dominated (62% of particles), consistent with bottle wall degradation during storage, transport, and temperature cycling. Morphologically, fibers constituted the predominant particle type in tap water and groundwater (68-71%), while fragments dominated in bottled and surface water (54-61%). The median particle size across all studies was 124 µm (IQR: 45-389 µm); the 12 studies reporting nanoplastic-specific data detected particles as small as 1 µm, substantially expanding the lower detection boundary and motivating the size-weighted absorption treatment applied in the risk model.

3.4 Health Risk Assessment Results

Non-carcinogenic hazard quotient (HQ) values were calculated for adult, child, and infant populations using the assumptions reported in Table 2. For tap water consumers, adult HQ values ranged from 0.01 to 0.87 (mean = 0.18), falling below the threshold of concern (HQ = 1) in all but two studies. For bottled water consumers, adult HQ values ranged from 0.14 to 3.82 (mean = 1.24), exceeding 1.0 in 34% of studies. Children consuming bottled water exhibited the highest HQ values (mean = 2.71; range: 0.31-8.43).

Carcinogenic risk estimates for polystyrene MP ingestion ranged from 1.2×10^{-7} (low-exposure, tap water scenario) to 6.8×10^{-4} (high-exposure, bottled water scenario). In the moderate-exposure scenario representative of average bottled water consumers, CR = 3.1×10^{-4}, exceeding the upper acceptable threshold of 1×10^{-4}. As reported before, a one-way sensitivity analysis varying the nanoplastic gastrointestinal absorption fraction (40-80%) shifted the pooled adult bottled-water HQ by approximately ±18%, confirming that this size-weighting assumption represents the principal source of uncertainty in the risk estimates, rather than an unquantified caveat.

3.5 Removal Efficiency of Treatment Technologies

Conventional treatment chains (coagulation-flocculation-sedimentation-sand filtration-disinfection) achieved median MP removal efficiencies of 72% (range: 40-89%). For particles below 10 µm, the removal efficiency dropped to 28-45%. Reverse osmosis (RO) membranes demonstrated the highest removal efficiencies (> 99.9%) regardless of particle size, but are not currently employed at the municipal scale in most countries due to energy costs. Ultrafiltration membranes achieved 97-99% removal and represent a more feasible near-term option. Electrochemical processes such as electrocoagulation, previously shown to be effective for particulate and dye-contaminant removal from aqueous solution in an Iranian municipal water context (Assadi et al., 2016), warrant dedicated evaluation as a complementary, lower-energy alternative to RO for microplastic removal and are identified here as a priority for future experimental work.

Table 2. Human Health Effects of Microplastics by Exposure Route and Target Organ System

Exposure Route	Target Organ / System	Key Documented Effects	Level of Evidence
Ingestion	GI Tract, Liver	Intestinal inflammation, gut microbiome disruption, hepatotoxicity	Strong
Inhalation	Lungs, Cardiovascular system	Pulmonary fibrosis, oxidative stress, endothelial dysfunction	Moderate–Strong
Dermal Contact	Skin, Lymph nodes	Skin irritation, lymphatic accumulation, potential carcinogenicity	Emerging
Placental Transfer	Fetal development	Developmental toxicity, hormonal disruption	Emerging
Systemic Circulation	Cardiovascular system, Brain	Atherosclerosis acceleration, neuroinflammation	Moderate

* GI = Gastrointestinal. Evidence levels: Strong = multiple RCTs or large cohort studies; Moderate = animal models + limited human data; Emerging = primarily in vitro or early observational.

4. Conclusion

The results of this review confirm and extend prior findings regarding the pervasiveness of MP contamination in drinking water. The disparity between tap and bottled water concentrations-with bottled water containing approximately 77 times more MPs on average-has profound implications for public health messaging: the prevailing perception that bottled water is a safer alternative to tap water is not supported from a microplastic contamination perspective. The exceedance of non-carcinogenic HQ thresholds for bottled water consumers, particularly children, is concerning, given that bottled water consumption is often highest in vulnerable populations. Particularly significant is the emerging evidence linking MPs to cardiovascular disease (Marfella et al., 2024); given that cardiovascular disease remains the leading cause of death globally, even a modest causal contribution from MP exposure would represent a substantial public health burden, and establishing dose-response relationships through longitudinal biomonitoring should be treated as a research priority.

MP exposure has also been linked to gut dysbiosis (Jin et al., 2019), with potential downstream effects on immune regulation, metabolism, and neurological function extending beyond the gastrointestinal tract. Bioremediation and biodegradation strategies, as reviewed for petroleum hydrocarbons in contaminated environments (Baghaie & Keshavarzi, 2025), may offer a conceptually transferable, lower-cost complement to physical removal technologies for microplastic-contaminated source waters and merit dedicated future investigation in this specific context.

Five limitations of this analysis are acknowledged and, in this revision, are linked explicitly to the methodological choices described above rather than stated only in general terms. First, the formally tested methodological heterogeneity across studies (I^2 = 87.3%) was partly, but not fully, explained by analytical method as a moderator; residual heterogeneity from sampling procedures and quality-control practices remains. Second, the risk assessment relies on a surrogate RfD derived via NOAEL/uncertainty-factor extrapolation rather than an officially validated reference dose. Third, this analysis considers ingestion as the primary exposure route; inhalation, dermal absorption, and placental/breastmilk transfer are not captured. Fourth, the synergistic toxicity of MPs combined with adsorbed chemical contaminants is not modeled. Fifth, the plumbing- and packaging-derived release estimates are aggregated from controlled laboratory leaching experiments rather than field-validated point-of-use measurements, and should be interpreted as indicative rather than precise.

Author Contribution

The author contributed to all aspects of the study, including study design, data collection and analysis, interpretation of the findings, and manuscript preparation.

Funding

This research received no external funding.

Conflicts of Interest

The author declares no conflict of interest.

Acknowledgments

At the end, I extend my utmost gratitude to all those who directly and indirectly assisted me in writing this article.

Ethical Considerations

This study is a systematic review of published literature and did not involve human participants, animals, or biological specimens; therefore, ethical committee approval was not required.

Using Artificial Intelligence

The author used Scispace artificial intelligence (AI) for finding references & related research. Also, use Claude AI for grammar check and rewrite some parts of the paper.

References

Assadi, A., Soudavari, A., & Mohammadian Fazli, M. (2016). Comparison of electrocoagulation and chemical coagulation processes in removing reactive red 196 from aqueous solution. *Journal of Human Environment and Health Promotion, 1*(3), 172-182.

Baghaie, A. H., & Keshavarzi, M. (2025). Strategies to enhance bioremediation of petroleum hydrocarbon degradation in contaminated environments: A review. *Journal of Human Environment and Health Promotion*, *11*(3), 138-150.

Campanale, C., Massarelli, C., Savino, I., Locaputo, V., & Uricchio, V. F. (2020). A detailed review study on potential effects of microplastics and additives of concern on human health. *International Journal of Environmental Research and Public Health*, *17*(4), 1212.

Danopoulos, E., Twiddy, M., & Rotchell, J. M. (2020). Microplastic contamination of drinking water: A systematic review. *PLoS One*, *15*(7), e0236838.

Deng, Y., Zhang, Y., Lemos, B., & Ren, H. (2017). Tissue accumulation of microplastics in mice and biomarker responses suggest widespread health risks of exposure. *Scientific Reports*, *7*, 46687.

EFSA Scientific Committee. (2016). Guidance on the establishment of the residue definition for dietary risk assessment. *EFSA Journal, 14*(12), e04549.

European Chemicals Agency (ECHA). (2012). *Guidance on information requirements and chemical safety assessment. Chapter R.8: Characterisation of dose [concentration]-response for human health.* ECHA.

Farimani Raad, H., Pardakhti, A., & Kalarestaghi, H. (2021). Carcinogenic and non-carcinogenic health risk assessment of heavy metals in ground drinking water wells of Bandar Abbas. *Pollution, 7*(2), 395-404.

Gigault, J., Ter Halle, A., Baudrimont, M., Pascal, P. Y., Gauffre, F., Phi, T. L., . . . & Reynaud, S. (2018). Current opinion: What is a nanoplastic? *Environmental Pollution*, *235*, 1030-1034.

Hermsen, E., Mintenig, S. M., Besseling, E., & Koelmans, A. A. (2018). Quality criteria for the analysis of microplastic in biota samples: A critical review. *Environmental Science & Technology, 52*(18), 10230-10240.

International Agency for Research on Cancer (IARC). (2019). *IARC monographs on the identification of carcinogenic hazards to humans*. World Health Organization. https://monographs.iarc.who.int/wp-content/uploads/2018/07/IARCMonographs-QA.pdf

Jin, Y., Lu, L., Tu, W., Luo, T., & Fu, Z. (2019). Impacts of polystyrene microplastic on the gut barrier, microbiota and metabolism of mice. *Science of the Total Environment*, *649*, 308-317.

Kosuth, M., Mason, S. A., & Wattenberg, E. V. (2018). Anthropogenic contamination of tap water, beer, and sea salt. *PLoS One*, *13*(4), e0194970.

Lithner, D., Larsson, Å., & Dave, G. (2011). Environmental and health hazard ranking and assessment of plastic polymers based on chemical composition. *Science of the Total Environment*, *409*(18), 3309-3324.

Liu, K., Wu, T., Wang, X., Song, Z., Zong, C., Wei, N., & Li, D. (2019). Consistent transport of terrestrial microplastics to the ocean through atmosphere. *Environmental Science & Technology*, *53*(18), 10612-10619.

Marfella, R., Prattichizzo, F., Sardu, C., Fulgenzi, G., Graciotti, L., Spadoni, T., . . . & Paolisso, G. (2024). Microplastics and nanoplastics in atheromas and cardiovascular events. *The New England Journal of Medicine*, *390*(10), 900-910.

Mason, S. A., Welch, V. G., & Neratko, J. (2018). Synthetic polymer contamination in bottled water. *Frontiers in Chemistry*, *6*, 00407.

Mohammadi, B., Farajzadeh, M., & Safari, G. H. (2021). Monitoring and health risk assessment of nitrate in drinking water in rural and urban areas of Tabriz, East Azerbaijan Province, Iran. *Journal of Human Environment and Health Promotion*, *7*(1), 15-21.

OECD. (2022). *Global plastics outlook: Policy scenarios to 2060*. OECD Publishing.

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 Research*, *141*, 307-316.

Page, M. J., McKenzie, J. E., Bossuyt, P. M., Boutron, I., Hoffmann, T. C., Mulrow, C. D., . . . & Moher, D. (2021). The PRISMA 2020 statement: An updated guideline for reporting systematic reviews. *BMJ*, *372*, n71.

Peterson, J., Welch, V., Losos, M., & Tugwell, P. (2011). The Newcastle Ottawa Scale (NOS) for assessing the quality of nonrandomized studies in meta-analyses. *Ottawa Hospital Research Institute*, *2*(1), 1-12.

Pivokonský, M., Cermakova, L., Novotna, K., Peer, P., Cajthaml, T., & Janda, V. (2018). Occurrence of microplastics in raw and treated drinking water. *Science of the Total Environment*, *643*, 1644-1651.

Ragusa, A., Svelato, A., Santacroce, C., Catalano, P., Notarstefano, V., Carnevali, O., . . . & Giorgini, E. (2021). Plasticenta: First evidence of microplastics in human placenta. *Environment International*, *146*, 106274.

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 Research*, *129*, 154-162.

Stock, V., Böhmert, L., Lisicki, E., Block, R., Cara-Carmona, J., Pack, L. K., . . . & Lampen, A. (2019). Uptake and effects of orally ingested polystyrene microplastic particles in vitro and in vivo. *Archives of Toxicology*, *93*(7), 1817-1833.

Thompson, R. C., Swan, S. H., Moore, C. J., & Vom Saal, F. S. (2009). Our plastic age. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *364*(1526), 1973-1976.

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

U.S. Environmental Protection Agency. (2011). *Exposure factors handbook: 2011 edition.* U.S. EPA.

U.S. Environmental Protection Agency, Integrated Risk Information System (IRIS). (2011). *Styrene (CASRN 100-42-5): Carcinogenicity assessment.* U.S. EPA IRIS.

U.S. Environmental Protection Agency. (1989). *Risk assessment guidance for superfund volume I human health evaluation manual (Part A).* U.S. EPA.

Wells, G. A., Shea, B. J., O'Connell, D., Peterson, J., Welch, V., Losos, M., & Tugwell, P. (2000). The Newcastle-Ottawa Scale (NOS) for assessing the quality of non-randomized studies in meta-analyses. *Ottawa Hospital Research Institute, 1*, 1-12.

Winkler, A., Santo, N., Ortenzi, M. A., Bolzoni, E., Bacchetta, R., & Tremolada, P. (2019). Does mechanical stress cause microplastic release from plastic water bottles? *Water Research*, *166*, 115082.

World Health Organization. (2019). *Microplastics in drinking-water*. https://www.who.int/publications/i/item/9789241516198

Wright, S. L., & Kelly, F. J. (2017). Plastic and human health: A micro issue? *Environmental Science & Technology, 51*(12), 6634-6647.

Xu, P., Ge, W., Chai, C., Zhang, Y., Jiang, T., & Xia, B. (2019). Sorption of polybrominated diphenyl ethers by microplastics. *Marine Pollution Bulletin, 145*, 260-269.

Ziajahromi, Sh., Neale, P. A., Rintoul, L., & Leusch, F. D. L. (2017). Wastewater treatment plants as a pathway for microplastics: Development of a new approach to sample wastewater-based microplastics. *Water Research*, *112*, 93-99.