Final Technical Report

for

Research Project

Assessment of Microplastic Pollution in different water sources in Thimphu, Bhutan

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Microplastic pollution and polymer-specific hazards in different water sources of Thimphu, Bhutan

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ABSTRACT

Microplastics (MPs) are an emerging contaminant of concern due to their persistence, polymer-specific toxicity, and potential human exposure through drinking water. This study provides the first systematic assessment of MPs in diverse water sources in Thimphu, Bhutan (n = 54), including streams, rivers, groundwaters, municipal taps, and bottled waters. Samples were analysed using pyrolysis–GC/MS and evaluated through Polymer Risk Index (PRI) and Estimated Daily Intake (EDI).

MPs concentrations varied widely, with streams (5.12 ± 0.70 μg/L) and groundwaters (2.70 ± 0.32 μg/L) showing the highest levels, followed by rivers (2.69 ± 0.26 μg/L). Treated waters, including municipal taps (1.05 ± 0.18 μg/L) and bottled waters (0.78 ± 0.10 μg/L) had comparatively lower concentrations. PET (52.5 %) was the dominant polymer, followed by PVC (14.6 %) and SBR (14.3 %), with upstream-to-downstream increases linked to urban pressures. While all sources showed overall moderate MPs concentrations, PRI classified every source as "high hazard" due to toxic polymers such as PVC and SBR. EDI values were highest in streams (0.17 ± 0.02 μg/kg/day) and lowest in bottled water (0.02 ± 0.00 μg/kg/day), comparable to intermediate global estimates.

Although concentrations were moderate, the predominance of hazardous polymers and chronic low-dose exposure highlight potential ecological and health risks. The presence of MPs in municipal tap and bottled water shows treatment and packaging does not fully eliminate contamination, highlighting the need for strengthening catchment protection, management, and long-term monitoring. Polymer-specific toxicological studies are crucial for risk-based policy under rapidly urbanizing and climate-sensitive context.

1. Introduction

Microplastics (MPs), defined as plastic particles less than 5 mm in size, including nano-sized particles [1], have emerged as a significant environmental concern worldwide. These tiny plastic particles are ubiquitous across various ecosystems, and their persistence and potential for bioaccumulation make them an increasing threat to aquatic life and human health [2]. A recent study has documented MPs, predominantly polypropylene fibres, in commercial fish species, demonstrating their entry into the aquatic food web [3]. Extensive studies have been conducted in coastal and marine environments, highlighting numerous detrimental effects on aquatic life, including impaired fertility, reduced growth, and oxidative stress [4,5]. Nevertheless, despite these environmental risks, plastics have become essential in modern life because of their durability, lightweight properties, and versatility in various applications.

With the continuous increase in global plastic production and contamination of aquatic environments, MPs have entered the food chain, posing an emerging food safety concern and hazard. In addition to their pervasive physical features, MPs can adsorb and accumulate a variety of chemical pollutants, including phthalates and bisphenol A, which can leach into the environment and potentially harm aquatic organisms. Further, MPs may act as a vector for the transport of persistent organic pollutants and other hazardous chemicals in the marine environment, escalating the concerns about their impact on ecosystems [6,7].

Drinking water sources are vulnerable to MPs contamination due to their mass, density, and size which favours their transportation via numerous carriers such as wind that eventually are deposited in water

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bodies. MPs from sources like footwear wear and tear, vehicle tires, and artificial turfs can enter the environment through atmospheric fallout [8]. In higher altitude regions, MPs due to their low density are spread into the upper layers of the atmosphere by wind and then deposited in high-altitude ecosystems through snowfall or rainfall, posing significant environmental risks [9,10]. Additionally, land-use practices, such as surface runoff from road paint and tire debris [11], synthetic fibres from washing machines and microbeads from personal care products, contribute to MPs contamination in aquatic environments [12,13]. These particles can also deposit in the soil, potentially contaminating groundwater sources [14].

Once in aquatic systems, MPs undergo fragmentation and degradation through physical, chemical, and biological processes [15], leading to toxicity in aquatic animals, disruption of the food chain, biodiversity loss, and ecosystem impacts. Bhutan, known for its rich biodiversity and strong environmental policies, has unique ecological and cultural landscape. Despite strong commitment from the government in conservation and sustainability of its ecosystem, the country still faces environmental challenges, including MPs. The reliance on surface water and groundwater as primary drinking water sources makes the Bhutanese population particularly susceptible to MPs exposure, especially due to the rapid urbanization and development activities taking place.

Thimphu, the capital city of Bhutan, exemplifies these changes, with urbanization contributing to increased plastic waste generation. The National Waste Inventory Survey, 2019 reported that plastic waste was second highest (17 %) solid waste generated after food waste (50 %) [16]. Inadequate waste management infrastructure exacerbates the problem, as improper disposal of plastics results in their breakdown into MPs, which then enter water bodies, soils, and the atmosphere. In Bhutan, waste is currently segregated into wet waste (biodegradable or organic wastes) and dry waste (free of organic material such as plastics, paper etc.). While a majority of households separate their waste at the source, 40.9 % of households dispose of waste in open pits, 34.9 % resort to open burning, and 2.0 % discard waste in open spaces due to a lack of collection services in certain areas [16].

The introduction of MPs into Bhutan's pristine environments could have negative consequences for local wildlife and aquatic organisms. Currently, there are limited evidence to assess the level of MPs contamination in water sources or their environmental impacts in Bhutan [17]. This study will attempt to address this gap by evaluating the presence and concentration of MPs in Thimphu's water systems, with a focus on developing strategies to mitigate their effects and protect the country's unique ecosystems in the future.

2. Objectives

The primary objectives of this research are:

1. To determine the concentration of MPs in different types of water sources including rivers, streams, springs and municipal water sources in Thimphu.
2. To identify different types of MPs (e.g., Polyethylene, Polyethylene terephthalate, Polypropylene etc.) present in these water bodies.
3. To evaluate the risk of MPs contamination on the local environment and public health.

3. Methodology

3.1. Sample collection

Water samples were collected from diverse freshwater sources in Thimphu, Bhutan, including rivers, streams, springs, and municipal tap water supplies. Sampling sites were selected to represent gradients of anthropogenic influence, including proximity to urban settlements, industrial zones, and waste disposal sites. A total of 54 samples were collected from 18 sampling sites during the winter season (December 2024 to January 2025). The details of sampling points are given in Table S1.

At each site, 2 L of water was collected in pre-cleaned borosilicate glass containers with the cap protected using aluminium foil to minimize plastic contamination. Three replicate samples were taken per site to ensure statistical reliability. Field blanks were prepared using Milli-Q water and processed along with environmental samples to monitor potential procedural contamination. All glassware were thoroughly rinsed with Milli-Q water before use. Samples were transported to the laboratory and processed within one week of collection, and stored at room temperature until filtration.

3.2. Sample processing and transportation

The GF fiberglass filters (0.6 μm pore size) with a diameter of 25 mm (4130025, Macherey-Nagel, Germany) were pre-heated for 5 h at 200 °C using hot air oven Gallenkamp (Sanyo) OMT150, UK) to ensure removal of plastic contamination. The filters were stored in a desiccator to prevent moisture absorption until used.

For each replicate, 2 L of water was vacuum-filtered through pre-heated 0.6 μm pore size glass fibre filter to capture MPs. The filtered samples were further heated at 200 °C for 2 h to eliminate non-plastic organic materials while preserving thermally stable polymers. All filtration steps were conducted in a clean-air workspace to minimize airborne fiber contamination. Processed filters were wrapped in aluminum foil, placed in sterile 47 mm glass Petri dishes, and sealed for shipment.

Prepared filter samples were taken under contamination-free conditions to the Environmental Toxicology Laboratory, Chulabhorn Research Institute (Bangkok, Thailand) for polymer identification and quantification. Transport complied with international packaging and transportation protocols for environmental MPs analysis.

3.3. MPs identification and quantification

MPs in the water samples were determined using Pyrolysis Gas Chromatography Mass Spectrometry (Py-GC/MS). The inner 13 mm circle of the 25 mm GF fiberglass filters holding the analyte residue was cut out using a sharp cylindrical metal hollow punch and placed into pre muffled Py-GC/MS tin cups (80 μl). MPs in these filter papers were analysed using Frontier Lab Multi-Shot Pyrolyzer™ (EGA/PY-3030D) with Auto-Shot Sampler™ (AS-1020E; Frontier Lab, Koriyama, Japan) coupled to Thermo Scientific™ TRACE™ 1310 Gas Chromatograph equipped with Ultra ALLOY capillary column 30 m x 0.25 mm I.D x 0.5 μm film (P/N: UAMP-K01, Frontier Lab, Koriyama, Japan), coupled with Thermo Scientific™ Orbitrap™ Exploris GC mass spectrometer. The pyrolyzer temperature protocols were developed and optimized in accordance with the manufacturer's recommendations. The specific instrumental settings and thermal program employed for Py-GC/MS characterization of MPs are summarized in Table S2.

Identification and quantification of each polymer in the filtered paper were determined against the external standards calibration curve of commercial mixture 12 polymers namely polyethylene (PE), polypropylene (PP), polystyrene (PS), acrylonitrile-butadiene-styrene (ABS), styrene butadiene rubber (SBR), polymethyl methacrylate (PMMA), polycarbonate (PC), polyvinylchloride (PVC), polyurethane (PU), polyethylene terephthalate (PET), nylon 6 (N6) and nylon 66 (N66). The total ion chromatogram (TIC) of the standard mixture of 12 plastic polymers is shown in Fig. 1. The *m/z* of each plastic polymer present in the commercial mixture supplied by Frontier Laboratories Ltd. is indicated in Table S3. The validation of the Py-GC/MS method was determined in terms of linearity (R^2), limit of detection (LOD), limit of quantification (LOQ) and coefficient of variance (%CV) (Table S4), aligning with established practices for Py-GC/MS analysis of MPs in en-

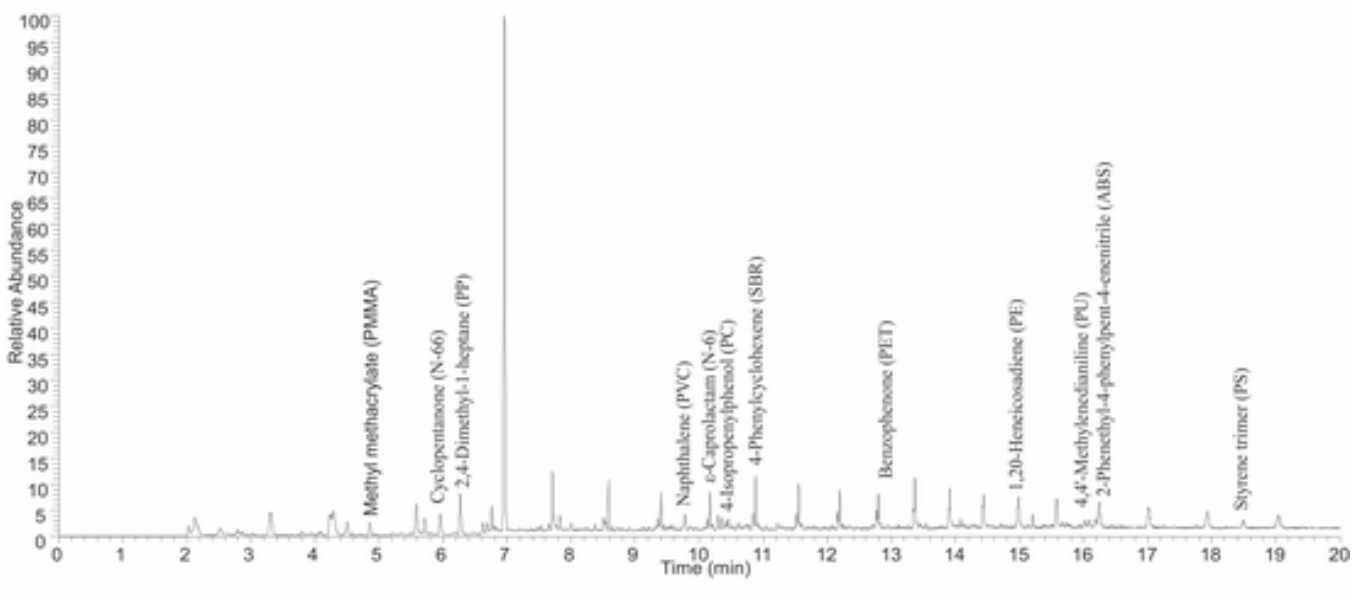

Fig. 1. Total ion chromatogram (TIC) of the standard mixture of 12 plastic polymers.

vironmental samples [18]. The F-Search MPs (Frontier Laboratories Ltd.) was used for quantifying the plastic polymers in the water samples.

3.4. Health risk assessment of MPs exposure from various water sources

Direct quantitative risk assessment of MPs in drinking water remains challenging due to the absence of reference dose values or established threshold limits. In this study, potential risks were assessed using Polymer Risk Index (PRI) and Estimated Daily Intake (EDI). The PRI estimates polymer-specific hazards based on intrinsic chemical properties, allowing prioritization of high-toxicity polymers [19]. The EDI was applied to approximate human exposure to MPs through drinking water consumption.

3.4.1. Polymer Risk Index (PRI)

The PRI was measured for MPs detected in different water sources by combining polymer concentrations and their corresponding hazard grades derived from published classifications [20]. The index was calculated using the following equation:

$$H = \sum P_n \times S_n$$

Where P_n is the proportion of each polymer in the water sample, and S_n represents the hazard score assigned to that polymer. The mean MPs concentrations are provided in Table 1, hazard category criteria in Table S6, and the hazard scores of specific polymers in Supplementary Table S7.

3.4.2. Estimated daily intake

Human exposure to MPs via drinking water was estimated using the following equation:

$$EDI = \frac{\rho \quad \times U \times EF \times ED}{BW \times AT}$$

Where:

Table 1
Concentration of different types of polymer identified using py-GC/MS.

Sampling point	Type of polymer (μg/L ± SEM)										
	ABS*	N6	N66	PC	PE	PET	PMMA	PP	PS	PU	PVC	SBR
Stream upstream (n = 6)	ND	0.05 ± 0.00	0.04 ± 0.00	0.02 ± 0.00	0.28 ± 0.18	1.89 ± 0.21	0.00 ± 0.00	0.16 ± 0.12	0.02 ± 0.01	0.01 ± 0.00	0.33 ± 0.05	0.39 ± 0.06
Stream downstream (n = 6)	ND	0.07 ± 0.01	0.07 ± 0.01	0.02 ± 0.00	2.34 ± 0.18	3.01 ± 0.45	0.01 ± 0.00	0.00 ± 0.00	0.04 ± 0.00	0.01 ± 0.00	0.68 ± 0.07	0.81 ± 0.14
River upstream (n = 6)	ND	0.01 ± 0.01	0.03 ± 0.00	0.01 ± 0.00	0.00 ± 0.00	0.90 ± 0.07	0.00 ± 0.00	0.06 ± 0.06	0.02 ± 0.00	0.01 ± 0.00	0.21 ± 0.03	0.18 ± 0.03
River midstream (n = 6)	ND	0.05 ± 0.00	0.06 ± 0.01	0.02 ± 0.01	0.00 ± 0.00	1.84 ± 0.12	0.01 ± 0.00	0.00 ± 0.00	0.03 ± 0.00	0.01 ± 0.00	0.57 ± 0.03	0.54 ± 0.10
River downstream (n = 6)	ND	0.06 ± 0.01	0.06 ± 0.01	0.02 ± 0.00	0.11 ± 0.11	1.95 ± 0.16	0.01 ± 0.00	0.00 ± 0.00	0.03 ± 0.01	0.01 ± 0.00	0.69 ± 0.06	0.57 ± 0.11
Groundwater (n = 12)	ND	0.03 ± 0.01	0.03 ± 0.01	0.01 ± 0.00	0.13 ± 0.10	1.39 ± 0.20	0.00 ± 0.00	0.09 ± 0.05	0.03 ± 0.02	0.01 ± 0.00	0.38 ± 0.06	0.60 ± 0.07
Tap water (n = 6)	ND	0.01 ± 0.00	0.02 ± 0.00	0.00 ± 0.00	0.05 ± 0.05	0.61 ± 0.11	0.00 ± 0.00	0.00 ± 0.00	0.01 ± 0.00	0.01 ± 0.00	0.21 ± 0.04	0.13 ± 0.03
Bottled water (n = 6)	ND	0.01 ± 0.01	0.02 ± 0.00	0.00 ± 0.00	0.04 ± 0.03	0.41 ± 0.06	0.00 ± 0.00	0.00 ± 0.00	0.01 ± 0.00	0.00 ± 0.00	0.26 ± 0.05	0.03 ± 0.01

* ABS was not detected (ND) in any of the samples

- ρ is the MPs concentration in each type of drinking water (Table 1).
- U is the per capita ingestion rate, considered as 2.0 L/day* for an average adult.
- EF is the exposure frequency in years (365 days).
- ED is the exposure duration for Bhutanese population (75 years) obtained from life expectancy at birth from WHO [21].
- BW represents body weight considered as 60 kg* and
- AT is the Average exposure time (ED x 365).

*These values are considered for this study, as region-specific data for Bhutan were not available.

3.5. Scope and limitations

This study provides a systematic assessment of MPs in multiple drinking water sources in Thimphu, Bhutan, using pyrolysis–GC/MS. Risk evaluation was based on polymer-specific hazard (PRI) and human exposure (EDI), following established approaches for polymer risk ranking [19,20]. Several limitations should be acknowledged. Sampling was conducted exclusively during the winter season. Consequently, the findings represent baseline winter conditions and do not capture seasonal dynamics. MPs concentrations, polymer profiles, and transport pathways particularly during the high-flow monsoon period may differ significantly. Therefore, conclusions regarding annual averages or trends are beyond the scope of this snapshot study, and the data should be interpreted as foundational winter-baseline information.

Particle size, a key determinant of bioavailability, could not be incorporated due to methodological constraints of Py-GC/MS. The PRI relies on hazard rankings reported in previous studies rather than site-specific toxicity data. While this provides a conservative and essential screening tool for prioritizing polymers of concern, it may not fully capture polymer-specific risks in the local context and therefore cannot be used to infer specific human health risks. Furthermore, while road networks and artificial turf were identified as key sources of PVC and SBR, respectively, the downstream urban setting suggests these are contributing rather than exclusive sources, with other urban activities likely adding to the polymer load.

3.6. Data analysis

Data were cleaned and organized using Microsoft Excel (Microsoft Office, 2021), and statistical analyses and graphical outputs were generated with GraphPad Prism (version 8.0.2; GraphPad Software Inc., San Diego, CA, USA). Differences among groups were evaluated using one-way analysis of variance (ANOVA), and when significant effects were detected, Tukey's multiple comparison test was applied for pairwise comparisons. Additional analyses, including linear regression for spatial relationships and unpaired *t*-tests for specific group comparisons were applied. All results are presented as mean ± standard error of the mean (SEM) based on replicate analyses. Statistical significance was considered at $p < 0.05$.

4. Results

4.1. Concentrations and polymer distribution

The concentration of MPs varied significantly among the different water sources (Fig. 2A). The highest mean MPs concentration were recorded in stream water (5.12 ± 0.70 μg/L, $n = 12$), followed by groundwater (2.70 ± 0.32 μg/L, $n = 12$) and river water (2.69 ± 0.26 μg/L, $n = 18$). Treated sources showed comparatively lower concentrations, with municipal tap water at 1.05 ± 0.18 μg/L, $n = 6$ and bottled water at 0.78 ± 0.10 μg/L, $n = 6$.

Across all samples, PET was the predominant polymer (52.5 %) (Fig. 3B), with highest concentrations in stream downstream (3.01 ± 0.45 μg/L) and river downstream (1.95 ± 0.16 μg/L) sites (Table 1). PVC (14.6 %) was the second most abundant polymer, with highest concentration in river downstream (0.69 ± 0.06 μg/L) and stream downstream (0.68 ± 0.07 μg/L). SBR was another common polymer (14.3 %) with highest level observed in streams downstream (0.81 ± 0.14 μg/L) and groundwaters (0.60 ± 0.07 μg/L). Some polymers, including ABS, PMMA, PU and PC were either absent or were detected only in trace concentrations across all sources.

Groundwater samples contained substantial amounts of PET 1.39 ± 0.20 μg/L, SBR 0.60 ± 0.07 μg/L and PVC 0.38 ± 0.06. Although MPs concentration were lower in municipal tap and bottled water than in surface sources, both contained detectable levels of multiple polymers. In municipal tap water, PET (0.61 ± 0.11 μg/L), PVC (0.21 ± 0.04 μg/L), and SBR (0.13 ± 0.03 μg/L) were present, while bottled water contained PET (0.41 ± 0.06 μg/L) and PVC (0.26 ± 0.05 μg/L).

4.2. Spatial patterns: upstream vs. downstream

Both stream and river systems showed clear downstream accumulation of MPs (Fig. 3B), particularly for PET and PVC. In streams, PET concentrations increased from 1.89 ± 0.21 μg/L at upstream sites to 3.01 ± 0.45 μg/L downstream, PVC rose from 0.33 ± 0.05 μg/L to 0.68 ± 0.07 μg/L and SBR from 0.39 ± 0.06 μg/L to 0.81 ± 0.14 μg/L over the same transect. Similarly, in rivers, PET increased from 0.90 ± 0.07 μg/L to 1.95 ± 0.16 μg/L, and PVC from 0.21 ± 0.03 μg/L to 0.69 ± 0.06 μg/L while SBR increased from 0.18 ± 0.03 μg/L to 0.57 ± 0.11 μg/L. Similar downstream increases were observed for PE and N6.

4.3. Spatial patterns and source apportionment

Comparison of sites downstream of artificial turf versus all other locations revealed highly significant differences in SBR contamination ($P = 0.0007$) (Fig. 4A). The median SBR concentration downstream of turf was 1.84 μg/L compared to 0.96 μg/L at other sites. In contrast, PVC concentrations showed no significant difference between these groups (median 1.72 μg/L vs. 1.50 μg/L; $P = 0.345$). Regression analysis further highlighted distinct source patterns for different polymers. A significant negative correlation was found between PVC concentrations and distance from major roads ($R^2 = 0.31$, $p = 0.039$), indicating declining PVC levels with increasing distance from roadways (Fig. 4B). Similarly, SBR exhibited a negative trend with road proximity ($R^2 = 0.28$, $p = 0.052$), though the relationship was marginally non-significant (Fig. 4C).

4.4. Chemical risk assessment

The hazard level assessment based on the Polymer Risk Index (PRI) indicated consistently high risk across all water sources (Fig. 5). Mean PRI values were highest in groundwater sources (3908 ± 253), followed by bottled water (3743 ± 439), river water (3377 ± 130), municipal tap water (3161 ± 263), and streams (2247 ± 109). All samples were classified in the "high hazard" category.

4.5. Estimated daily intake

EDI values suggested variable potential human exposure depending on water source (Fig. 6) Stream waters showed the highest mean EDI at 0.17 ± 0.02 μg/kg body weight/day. This was followed by groundwaters and rivers, with similar mean EDI values of 0.09 ± 0.01 μg/kg/day. On the other hand, drinking water samples from municipal tap demonstrated a lower mean EDI of 0.03 ± 0.01 μg/kg/day, while bottled water had the lowest value of 0.02 ± 0.00 μg/kg/day.

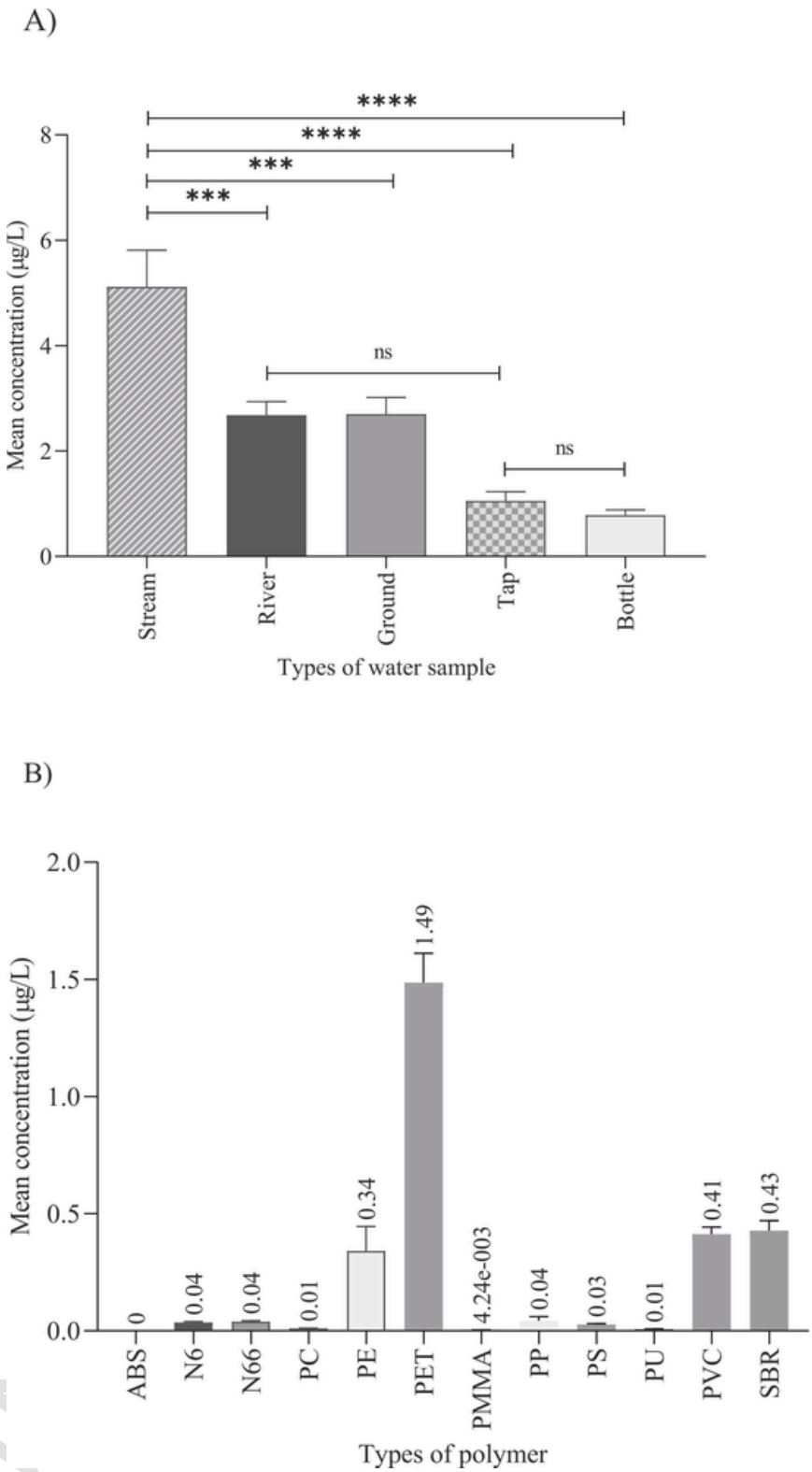

Fig. 2. MP concentration across different water sources in Thimphu. MP concentration in µg/L (A). Streams showed significantly higher abundances than other types of drinking water sources (one-way ANOVA, $p < 0.05$). MP concentration in µg/L, reflecting the mass load (B). All values represent mean concentration and error bars represent SEM.

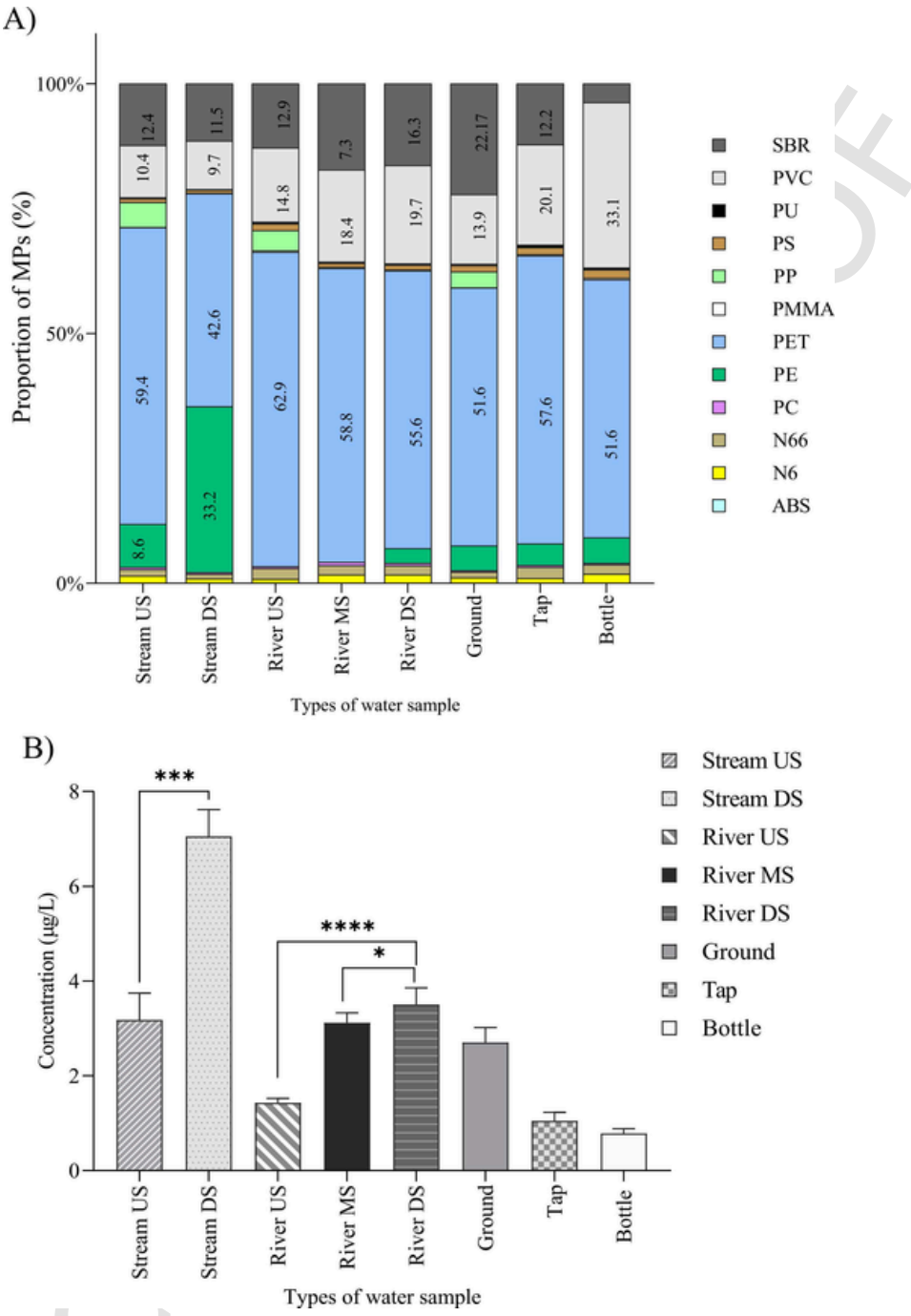

Fig. 3. Concentration of MPs (µg/L) in raw water, treated municipal water, and bottled water (A). Error bars represent standard error of mean. Relative abundance (%) of polymer types across water sources. legend include PET (polyethylene terephthalate), PE (polyethylene), PU (polyurethane), PP (polypropylene), PVC (polyvinyl chloride), PS (polystyrene), PMMA (polymethyl methacrylate), PC (polycarbonate), ABS (acrylonitrile butadiene styrene), N6 (nylon-6), N66 (nylon-66), and SBR (styrene–butadiene rubber) (B). Abbr: Stream US, stream upstream; Stream DS, Stream downstream; River US, river upstream; River MS, river midstream; River DS, river downstream.

5. Discussion

This study offers the first comprehensive assessment of MPs contamination across diverse water sources in Thimphu, including streams, rivers, groundwaters (spring source and infiltration borewells), and drinking waters (municipal tap and bottled). The results highlight polymer-specific distributions and sources, with clear spatial patterns that carry implications for both water management and public health.

Streams and groundwaters were the most contaminated sources, followed by rivers. The freshwater environments including rivers and streams act as dynamic sinks and transport vectors for MPs [22]. In Thimphu, this trend is strongly linked to rapid urbanization, road expansion and construction, where atmospheric deposition of synthetic fibers, stormwater runoff, and wastewater discharges contribute substantially [23]. The detection of MPs in groundwater suggests additional pathways, such as infiltration from surface water, agricultural leaching, or contact with surrounding plastic infrastructure [24]. How-

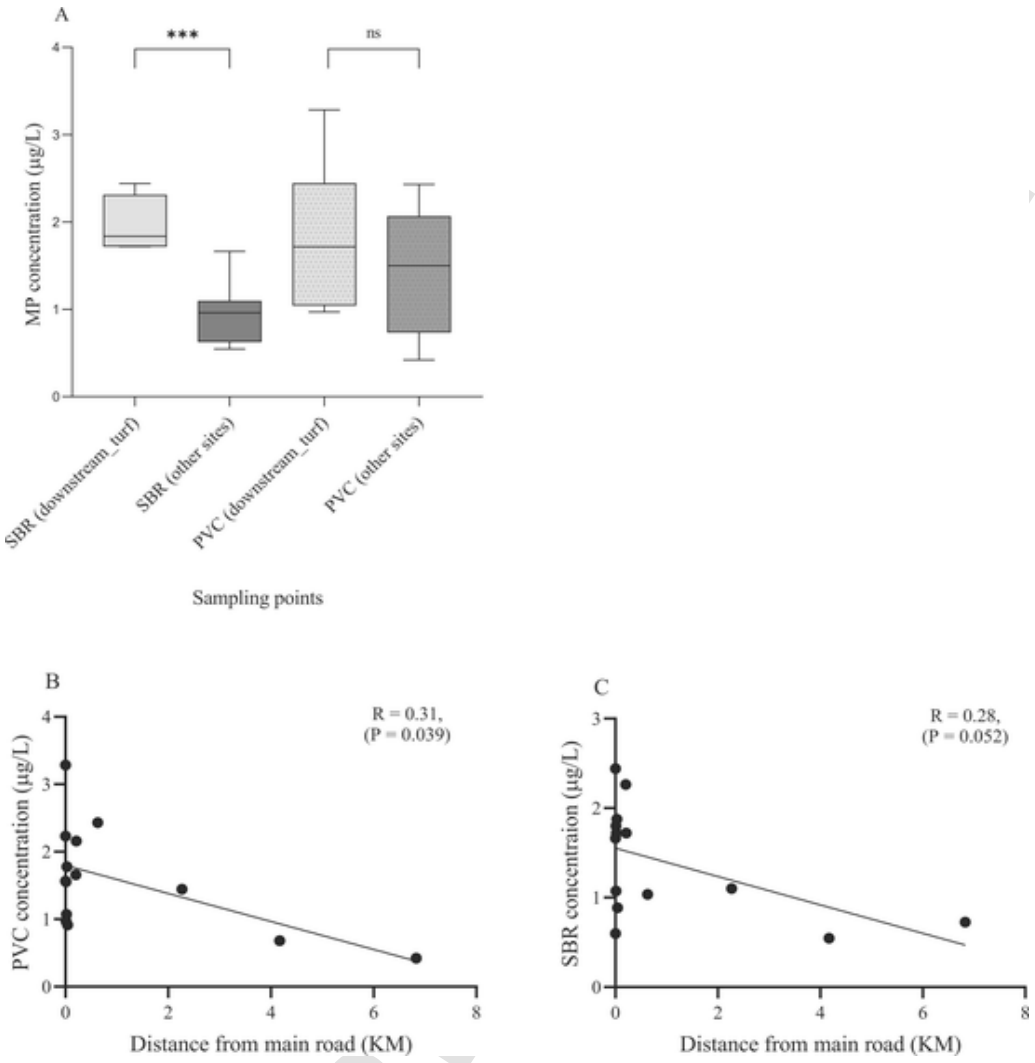

Fig. 4. Source analysis of SBR and PVC microplastics relative to artificial turf and road proximity. Box plots showing polymer concentration at sites located downstream of artificial turf fields versus all other sites (A). PVC concentration versus distance to major roads, showing a significant negative correlation ($p = 0.039$) (B). SBR concentration versus distance to major roads, showing a strong negative trend ($p = 0.052$) (C). Data in B are presented as box plots showing the median, quartiles, and range. Data in B and C are shown as scatter plots with regression lines and 95 % confidence intervals.

ever, the mechanisms of MPs entry into groundwater could not be directly evaluated, and without assessment of soil cores or surrounding infrastructure materials, they should be regarded as plausible hypotheses for future investigation rather than confirmed sources.

When compared globally, MPs concentration in Thimphu reflects an intermediate pollution scenario. Concentrations are lower than in highly urbanized catchments such as Barcelona (11.3–77.1 µg/L) [25] but higher than in Nordic countries such as Sweden (0.00014–0.00543 µg/L) [26] and Norway (0.0061–0.0931 µg/L) [27]. However, direct numerical comparisons of MPs concentration should be interpreted with caution due to methodological differences between studies. Particularly, 500-fold smaller sample volume (2 L vs. 1000 L in Nordic studies) significantly affects detection limits and reported concentrations. Moreover, the unique geophysical context of the Himalayas including glacial melt patterns, steep topographic gradients, and seasonal sediment mobilization creates distinct MPs transport and deposition dynamics not observed in comparative studies from other regions.

The detection of MPs in both municipal taps and bottled water is a direct concern for human health. Drinking water is a primary exposure pathway for MPs, which may carry toxic additives [28]. Studies have documented leaching of endocrine-disrupting chemicals such as Bisphenol A and phthalate esters from PET bottles [29,30]. Although concentrations were lower than in sources, PET and PVC were consistently detected in both treated samples. These may originate from insufficient water treatment, leaching during storage, or distribution network contamination. In Thimphu, slightly higher levels in municipal tap water likely reflect reliance on contaminated surface sources, while bottled water (largely groundwater-fed) carries lower loads but is not risk-free due to polymer leaching.

Polymer-specific analysis showed PET as the predominant polymer across all water sources (52.5 %), followed by PVC (14.6 %) and SBR (14.3 %). The predominance of PET is consistent with its extensive use in single-use beverage bottles and food packaging [31]. Its presence in both surface and treated waters underscores its environmental persistence and potential release from treatment plants and from PET bottles during storage [32]. The presence of PU and PMMA, although in trace amounts, likely originates from coatings, paints, and industrial activities [33]. PP was only detected in upstream sites (streams and rivers), which may suggest localized or point-source pollution, possibly from agricultural plastics used upstream.

Spatial analysis revealed clear source patterns and accumulation trends. For instance, PVC, detected at high levels downstream, is likely

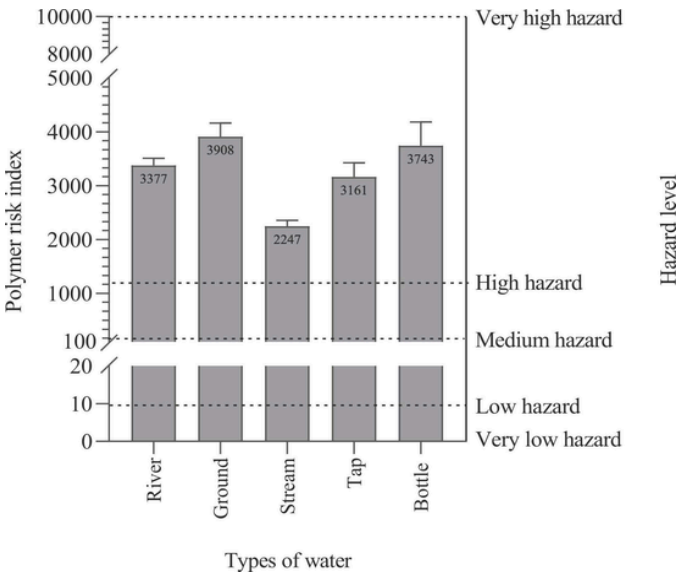

Fig. 5. Polymer Risk Index (PRI) for different water sources. The PRI evaluates the potential hazard of microplastics based on the relative toxicity of the detected polymer types. The numbers inside each bar represent the mean PRI for each sample type and error bars represent SEM.

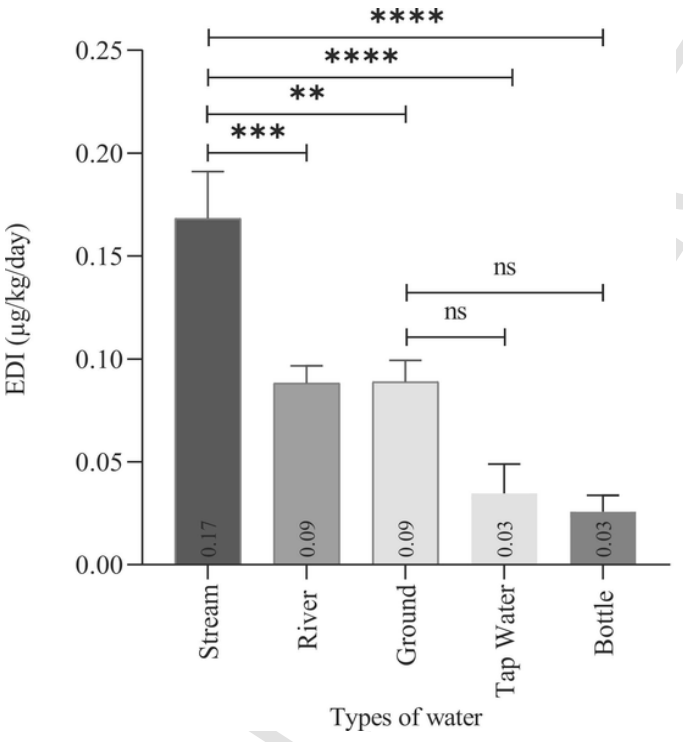

Fig. 6. Estimated Daily Intake (EDI) of microplastics via water consumption for different population groups in Thimphu, Bhutan. Streams showed significantly higher EDI than other types of water samples (one-way ANOVA, $p < 0.05$). Values are presented as mean ± SEM micrograms of microplastics ingested per kilogram of body weight per day (µg/kg bw/day).

derived from piping [34] and construction materials, as suggested by its significant positive correlation with road proximity ($p = 0.039$). Similarly, the notable abundance of SBR, particularly in streams downstream (0.81 ± 0.14 µg/L) and groundwaters (0.60 ± 0.07 µg/L), points towards multiple urban sources. Our data suggests that vehicular tire wear is a key contributor, as indicated by a negative correlation between SBR and distance from major roads ($p = 0.052$). Furthermore, artificial turf was identified as a significant source, with a strong elevation of SBR at sites downstream of synthetic playing fields ($P = 0.0007$). These observations are consistent with global studies identifying tire wear and construction-related plastics as dominant contributors to urban MPs pollution [35–37] and reinforces the role of river networks as key pathways for terrestrial plastics into aquatic systems [38].

While overall MPs concentration were low to moderate, the PRI classified all sources as "high hazard" category due to the presence of toxic polymers such as PVC and SBR [19,20]. To provide a quantitative measure of the disparity between total MPs concentration and polymer-specific hazards, a Hazard-to-Concentration Ratio (HCR) was calculated. The HCR values increased sharply from streams (439) to bottled water (4773), despite bottled water having the lowest MPs concentration (0.78 µg/L). This inverse relationship demonstrates that lower MPs abundance does not necessarily correspond to lower risk, as treated and packaged waters may contain a higher proportion of hazardous polymers. The results highlight that polymer composition, rather than total MPs mass, is the principal determinant of potential hazard [39,40]. Thus, a combined assessment of both concentration and PRI offers a more comprehensive evaluation of ecological and human health hazards associated with MPs.

Besides chemical hazards, MPs can adsorb heavy metals [41] and persistent organic pollutants [42], and can act as vectors for diverse microbial communities, including pathogenic species [43,44]. Given that nearly 47.2 % of urban drinking water in Bhutan showed microbial contamination [45], concerns regarding synergistic risks remain considerable.

EDI calculations further emphasize the human health implications. The EDI was observed in higher in untreated sources including stream waters, ground waters and river waters. Treated water showed lower EDI having 0.03 ± 0.01 µg/kg/day and 0.02 ± 0.00 µg/kg/day for municipal tap and bottled water respectively. It is critical to note that these EDI values, along with the PRI, are based on the MPs fraction ≥ 0.6 µm and represent a significant underestimate of the true exposure and risk, as the excluded nanoplastic fraction is potentially more bioavailable and toxic [46]. EDI values reported here align with intermediate global estimates ranging from 0.02 to 0.2 µg/kg/day [27] but fall below those reported in highly urbanized regions having EDI for adults (40.1 µg/kg/day) and children (87.8 µg/kg/day) [47]. Conversely, studies from regions with lower plastic pollution report EDI below the concentrations found in this study, ranging from 0.004 to 0.018 µg/kg/day [48]. Differences between studies likely reflect variability in regional pollution, treatment efficacy, and sampling methodology. Despite relatively low values, chronic low-dose ingestion of hazardous polymers remains a pressing ecotoxicological concern, particularly for vulnerable populations such as children or immunocompromised individuals.

While total MPs loads appear relatively low in Thimphu, the dominance of toxic polymers and direct exposure pathways highlight vulnerabilities that cannot be ignored. Reliance on bottled water as a "safer" option may inadvertently increase risk from leached polymers. Protecting Thimphu's source waters, particularly streams and rivers during monsoon runoff, is critical. Municipal and bottled water treatment must adapt to emerging contaminants like MPs, alongside microbial and chemical risks. Policy frameworks should evolve beyond quantifying total MPs concentrations to include polymer hazard weighting and compositional profiling to better inform risk-based regulation. To mitigate these risks, key actions include strengthening catchment protection, improving plastic waste management, and enforcing standards for drinking water. Further, the identification of artificial turf as a significant source of SBR calls for environmental impact assessments of synthetic playing fields and the promotion of safer, non-plastic alternatives

in public installations. Future studies should expand to long-term monitoring, seasonal variation, and polymer-specific toxicology to fully capture the risks in Thimphu's rapidly changing urban environment.

6. Conclusion

This study provides a systematic assessment of MPs contamination in diverse water sources in Thimphu, Bhutan, revealing polymer-specific risks that extend beyond overall particle loads. While total concentrations were relatively moderate compared to highly urbanized regions, the predominance of hazardous polymers such as PVC and SBR, alongside the ubiquity of PET across all sources, indicates significant ecotoxicological concern. Upstream-to-downstream increases highlighted cumulative urban pressures and runoff contributions.

The combined application of polymer quantification, PRI, and EDI highlights that although MPs abundance appears moderate, their composition and chronic low-dose ingestion pathways pose a considerable hazard to both ecosystems and human health. The presence of MPs in both municipal tap water and bottled water underscores that treatment and packaging processes do not eliminate contamination. These findings emphasize the critical need for polymer-focused regulatory measures, including source-targeted controls for construction materials and artificial turf. Source water protection, strengthened plastic waste management and optimized treatment systems remain fundamental to mitigating risks.

Given Bhutan's increasing reliance on bottled and piped water in urban centers, future research should focus on long-term and seasonal monitoring, polymer-specific toxicological impacts, and the interactions of MPs with co-contaminants such as heavy metals and persistent organic pollutants. Such evidence will be essential to inform risk-based policy and ensure safe drinking water under Bhutan's rapidly urbanizing and climate-sensitive context.

Ethical clearance

This study did not include human participants or animal experimentation; therefore, ethical approval was not required. An ethics waiver was granted by the Research Ethics Board of Health, Bhutan (Proposal No. 2025.38.NNW). In addition, administrative approval was obtained from the Ministry of Health prior to commencement of the study.

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CRediT authorship contribution statement

Chimmi Dorji: Writing – review & editing, Supervision, Conceptualization. **Amin Ngawang Tashi:** Validation, Resources, Investigation. **Rinzin Wangdi:** Validation, Investigation, Data curation. **Phub Zam:** Visualization, Validation, Data curation. **Pema Chophel:** Writing – original draft, Visualization, Project administration, Methodology, Investigation, Formal analysis, Data curation, Conceptualization.

Declaration of Competing Interest

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

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Data availability

Data will be made available on request.

Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.hazmp.2026.100044.

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