Environmental Science and Pollution Research (2025) 32:25603–25617
https://doi.org/10.1007/s11356-025-37130-8

RESEARCH ARTICLE

Microplastics in drinking water: quantitative analysis of microplastics from source to tap by pyrolysis–gas chromatography-mass spectrometry

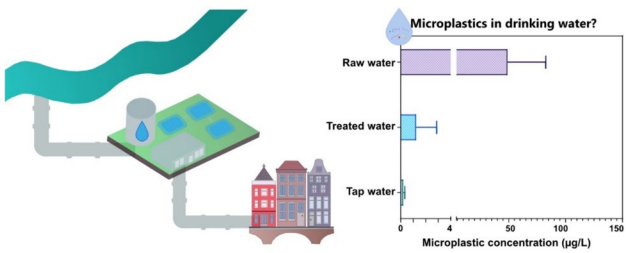
Feride Öykü Sefiloglu[1,5] · Marthinus Brits[1,6] · Martin J. M. van Velzen[1] · Eelco N. Pieke[2] · Onno J. I. Kramer[3,4] · Marja H. Lamoree[1]

Received: 2 May 2025 / Accepted: 21 October 2025 / Published online: 5 November 2025

Abstract

The widespread presence of microplastics (MPs) in fresh surface water has raised concerns about potential human exposure through drinking water sourced from these environments. While MP research is advancing to understand the occurrence and fate of MPs in drinking water production systems, data based on mass concentration is scarce. This study assesses MP concentrations in the drinking water supply system of Amsterdam (the Netherlands) from source to tap, analyzing raw water from two freshwater sources (Lek Canal and Bethune Polder), treated water from two drinking water treatment plants (DWTPs) (Leiduin and Weesperkarspel DWTPs), and household tap water samples from the Amsterdam distribution area. MPs ≥ 0.7 µm were identified and quantified using pyrolysis gas chromatography-mass spectrometry (Py-GC–MS) targeting 6 high production volume polymers: polyethylene (PE), polyethylene terephthalate (PET), polymethyl methacrylate (PMMA) polypropylene (PP), polystyrene (PS), and polyvinyl chloride (PVC). Average MP concentrations in raw water samples were 50.6 ± 34.7 µg/L ($n = 14$) and 47.5 ± 33.7 µg/L ($n = 14$), while treated water samples exhibited significantly lower levels of 0.80 ± 0.44 µg/L ($n = 12$) and 1.65 ± 2.19 µg/L ($n = 14$), demonstrating high removal efficiencies of 97–98%. PE, PVC, and PET were the most abundant polymer types detected. Household tap water samples showed lower concentrations with an average of 0.21 ± 0.12 µg/L ($n = 20$). These findings highlight the effective removal of MPs during drinking water treatment processes while emphasizing the need for further research to understand the factors influencing MP transport and fate within water distribution networks.

Graphical Abstract



Keywords Microplastics · Drinking water · DWTP · Surface water · Pyrolysis-GC–MS

Introduction

The contamination of freshwater sources with microplastics (MPs) has been documented globally, raising concerns about drinking water safety (Kurniawan et al. 2023; Sefiloglu et al. 2024). Drinking water has been increasingly studied in recent

Responsible Editor: Thomas D. Bucheli

Extended author information available on the last page of the article

years as a possible route of MP intake, providing evidence that both bottled (Kankanige and Babel 2020; Li et al. 2023) and tap water (Kirstein et al. 2021; Shruti et al. 2022; Sun et al. 2024) can contain MPs. Research generally reveals higher concentrations in bottled water than in tap water, suggesting that plastic packaging may contribute to increased MP levels found in bottled liquids (WHO 2022). Reported MP concentrations in tap water vary widely, from 0.0001 (Johnson et al. 2020) to 1247 MP/L (Tong et al. 2020), while the levels in single-use plastic bottles water were reported to be up to 5.4×10^7 MP/L (Zuccarello et al. 2019). Notably, MP levels in European potable waters are relatively lower compared to those in America and Asia, as reported by Belz et al. (2024). In 2020, the European Drinking Water Directive was updated to consider including MPs on the Watch List of emerging substances of concern (EU 2020). In 2024, the EU Commission advanced this effort by publishing a harmonized methodology for monitoring MPs in drinking water across the EU, with a view to their potential inclusion in the recast Watch List (EU 2024).

The drinking water source and water treatment efficiency affect MP levels in potable water. Studies analyzing raw water for drinking water production revealed higher MP concentrations in surface water than in groundwater sources (Mintenig et al. 2019; WHO 2022). The removal of MPs through drinking water treatment plants (DWTPs) has been investigated by several studies from different countries, and varying treatment efficiencies ranging from 80 to 99% have been reported (Johnson et al. 2020; Pivokonský et al. 2020; Bäuerlein et al. 2022; la Cecilia et al. 2024). Besides the incomplete removal of MPs in the treatment processes, the degradation of water tanks, plumbing fittings, and distribution pipes made from plastic material is considered another possible source of MPs in drinking water (Mintenig et al. 2019; Zhou et al. 2023). Conversely, distribution tanks and pipes were also identified as potential sinks for MPs due to settling and adsorption processes (Chu et al. 2022).

The reported concentrations of MPs in drinking water are also influenced by the methods used for sampling, sample treatment, and final analysis. Studies investigating MPs in drinking water show a general trend of increasing MP particle numbers with decreasing particle size, indicating the importance of the methodological size cut-off or the detection size limit of the analytical instruments on the reported MP levels (Koelmans et al. 2019; WHO 2022). Different analysis tools have varying strengths and limitations in terms of measured units (particle number or mass), detection size/mass limits, polymer identification capabilities, and sensitivity. Most studies on tap water focus on particle numbers within a certain size range (down to 1 µm), primarily using Fourier Transform Infrared (FTIR) spectroscopy and Raman spectroscopy (Mintenig et al. 2019; Pivokonský et al. 2020; Pittroff et al. 2021; Shruti et al. 2022; Taghipour et al. 2023). There is currently limited data on the mass concentrations of MPs in drinking water networks, particularly for particles < 10 µm, which are especially relevant for assessing human exposure (Belz et al. 2024).

Complementary to spectroscopic techniques, thermoanalytical techniques, such as pyrolysis gas chromatography-mass spectrometry (Py-GC–MS), offer rapid analysis and high sensitivity for the simultaneous identification and quantification of different plastic polymers. It provides information on MP mass concentrations, a valuable metric for mass balance calculations, assessing the treatment efficiency of potable water plants, and evaluating human exposure. Unlike spectroscopic techniques, analysis with Py-GC–MS is unrestricted by the size of the particle but by mass; hence, it has been utilized for the analysis of particles in nanometer ranges in water samples (Okoffo and Thomas 2024). So far, Py-GC–MS (or Py-GC–MS/MS) has been applied for the analysis of bottled water (Hermabessiere and Rochman 2021; Albignac et al. 2023; Okoffo and Thomas 2024; Huang et al. 2024) and water samples from drinking water supplies and distribution systems (Kirstein et al. 2021; Gomiero et al. 2021; Li et al. 2024; Xu et al. 2024; Dalmau-Soler et al. 2024), while the analysis of household tap water remains scarce.

To fill these knowledge gaps, this study provides a full-scale assessment of MPs in the drinking water production system from source to tap, employing Py-GC–MS for quantitative analysis for particles down to 0.7 µm. Six high-production volume polymers, including polyethylene (PE), polyethylene terephthalate (PET), polymethyl methacrylate (PMMA), polypropylene (PP), polystyrene (PS), and polyvinyl chloride (PVC), were identified and quantified in the water samples. Freshwater samples collected from the sources for drinking water production and treated water samples from two distinct drinking water production lines were analyzed to evaluate the efficiency of water treatment processes in removing MPs. Additionally, tap water samples from various household locations in Amsterdam were examined to assess potential MP exposure through tap water consumption. By providing valuable data on mass concentrations of MPs in the drinking water production network, this study contributes to the growing knowledge essential for developing strategies to mitigate microplastic contamination in drinking water systems.

Materials and methods

Sampling

Raw and treated water samples

Amsterdam's drinking water is supplied from two freshwater sources, the Lek Canal and the Bethune Polder,

with different treatment systems. Raw water from the Lek Canal undergoes initial treatment through ferric chloride for coagulation and then rapid sand filtration before being transported to the Amsterdamse Waterleidingduinen for natural dune infiltration. The recovered water is then sent to the Leiduin DWTPs for final treatment, which includes rapid sand filtration, ozonation, softening with calcite, carbon filtration, and slow sand filtration. Similarly, raw water from the Bethune Polder is initially treated with ferric chloride for coagulation and then transported to the Waterleidingplas for natural filtration. The recovered water is subsequently delivered to the Weesperkarspel DWTP, where it undergoes final treatment through rapid sand filtration, ozonation, softening with calcite, carbon filtration, and slow sand filtration. The whole treatment process can take 60 to 90 days.

In this study, four sampling points were selected from these two branches of the drinking water supply system to determine the MP levels before and after the treatment processes. Raw water samples were collected from Lek Canal and Bethune Polder, and treated water samples were collected from Leiduin and Weesperkarspel DWTPs (Fig. 1). Sampling was conducted over 5 months in 2022, starting with bi-weekly sampling in March and transitioning to bi-monthly sampling from April to July (sampling dates were provided in Supplementary Information Table S1). On each sampling day, 4 samples were taken from 4 locations, resulting in a total of 28 raw water and 26 treated water samples (2 samples from Leiduin DWTP could not be sampled at the scheduled date) (Fig. 1). At each sampling point, bulk samples of 2 L (both raw and treated water) are collected in pre-cleaned glass bottles. Samples from Lek Canal, Leiduin DWTP, and Weesperkarspel DWTP were collected from existing sampling taps following standard sampling protocols, including pre-rinsing the taps, and Bethune Polder samples were collected by extracting samples with a metal bucket at 50 cm below the water surface. The sampling bottles and the bucket are rinsed 3 times before the bottles are filled up to brim capacity.

Considering the sampling taps are in the in-door production line, we designed a field blank collection procedure for possible air contamination at the sampling points ($n = 54$). In parallel with the sample collection, 100 mL of prefiltered (with 0.7 µm glass filter) demineralized water was transferred to another pre-cleaned glass bottle at each sampling site to mimic the possible air contamination from the surroundings.

All sample bottles were immediately covered with aluminum foil (before and after the sampling) to reduce the exposure of the sample to air to a minimum and prevent contact of the sample with the plastic bottle lid. After each sampling day, the samples were stored at 4 °C until analysis.

Tap water samples

Samples from the household tap were collected directly with a closed in-line filtration system by placing a glass filter of 0.7 µm mesh size and 47 mm diameter (GF/F, Whatman, United Kingdom) in a stainless steel filter disk (Sterlitech Corporation, United States) as used by Kirstein et al., (2021). The sampling set-up is shown in Supplementary Information (Figure S1). The inlet of the original metal holder was modified to make it adaptable to different types of taps with other stainless steel components, whereas the outlet was attached to a flowmeter (Zenner International GmbH & Co. KG, Germany) to measure the water flow. The water flow rate was kept at 1.5 L/min. Tap water samples (n=20) were collected from 10 different locations in Amsterdam: 9 private houses (1 sample per location) and 1 kitchen tap of our department (11 samples from the same location on different days) (Fig. 1). At each sampling point, 100 L of tap water was filtered and the filter was then transferred to a pre-cleaned glass petri dish in the laboratory under restricted contamination conditions. Samples in the Petri dishes were dried and stored in an oven (Binder, Emergo, Landsmeer, the Netherlands) at 50 °C until analysis. Photos of samples are provided in the Supplementary Information (Figure S1).

Contamination control

Special attention was given to minimize the contamination within the laboratory. All the glass filters and Petri dishes used for sampling and sample preparation were pre-heated in a muffle oven at 500°C. The demineralized water used in the laboratory for cleaning purposes was prefiltered with an in-line closed filtration system by placing a 0.7 µm glass filter in a stainless steel filter disk (Sterlitech Corporation, USA). All solvents and reagents utilized in sample treatment and cleaning were first filtered through 0.7 µm glass (GF/F, Whatman, United Kingdom) filters to prevent procedural contamination. 100% cotton lab coats were worn, and the lab surfaces were cleaned daily with water and technical ethanol (70%). All equipment, including sampling bottles, filtration glassware, Petri dishes, and tweezers, was cleaned meticulously with pre-filtered demineralized water and immediately covered with aluminum foil to minimize air exposure. To remove possible contamination, the stainless steel pyrolysis cups (Eco-cup LF, Frontier LAB, Japan) were flamed with a torch before use to remove external contaminants.

Sample preparation

The raw and treated water samples were filtered through a 0.7 µm pore size pre-cleaned fiberglass filter (GF/F, 25 mm ⌀, Whatman, United Kingdom) using a custom-made filtration unit where the samples are concentrated on an 8 mm

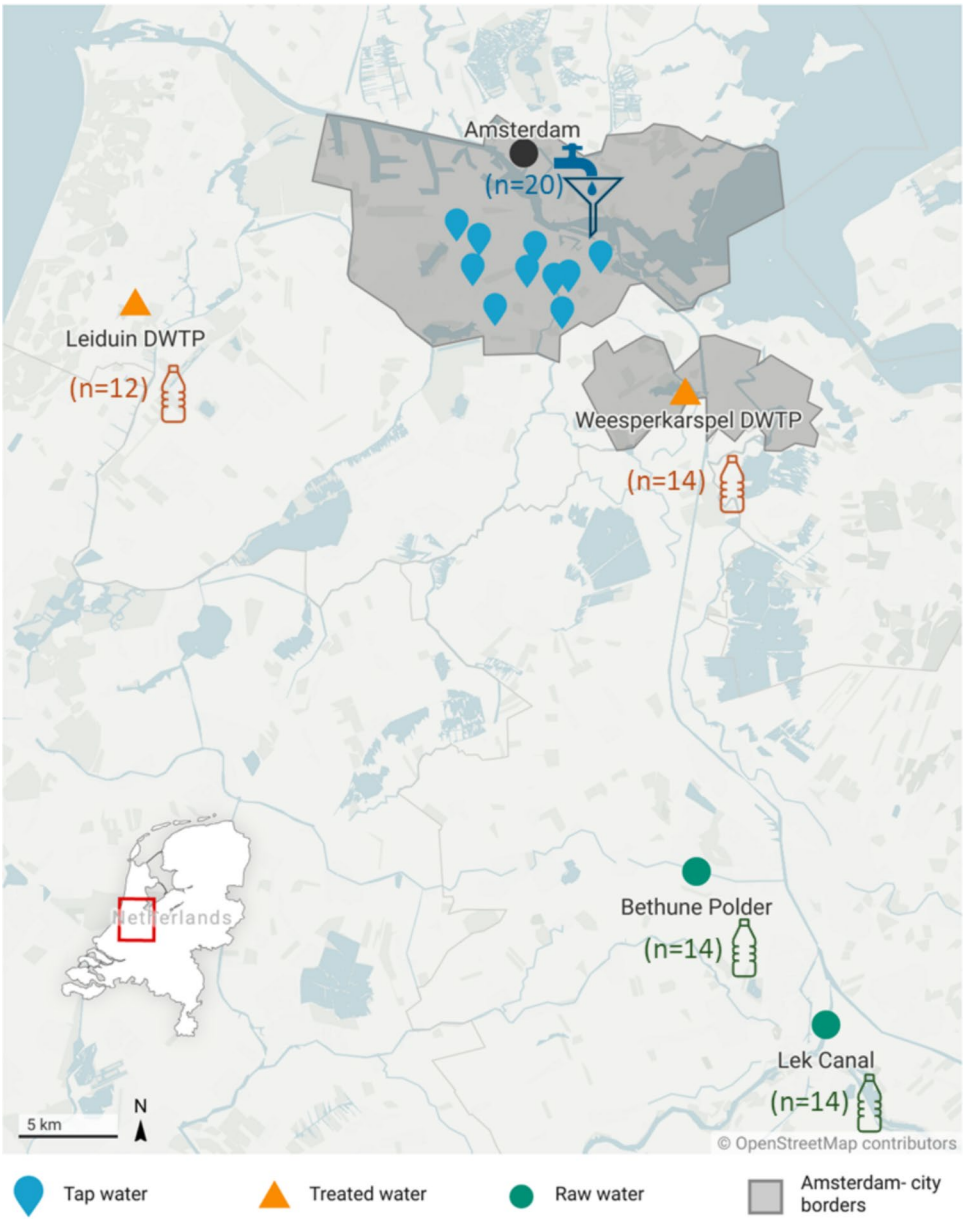


Fig. 1 Sampling locations and collection details. The number of samples collected from each location and the sampling method (bottle sampling or in-line filtration) are indicated. The map was generated using Datawrapper

diameter filter area. To remove organic matter, raw water samples (200 mL) were oxidized with 30% H_2O_2 (Merck, Darmstadt, German) overnight in a water bath at 50°C. The treated water samples coming from the DWTPs were directly filtered without pretreatment. After the filtration step, the filtration unit and the sample were rinsed with 10 mL of prefiltered water and 10 mL of ethanol (99.95%, Biosolve, France). The resulting sample, concentrated on the 8 mm filter, was cut out, folded, and placed in a pyrolysis cup. Field and laboratory blanks were prepared the same way as the real samples.

From each of the tap water samples, filtered on a 47 mm glass filter with the in-line filtration system, 2 random subsamples were cut out with an 8 mm punch, folded, and placed in a pyrolysis cup for the subsequent analysis. It has been suggested that MP quantification can vary depending on the punch location on the filter, introducing a degree of subsampling uncertainty. The homogeneity of particle distribution across the filters was not experimentally assessed in this study, and this potential source of uncertainty is acknowledged as a methodological limitation.

The pyrolysis cups were placed in a custom-made aluminum rack (VU workshop, Amsterdam, Netherlands) with an aluminum lid and the rack was kept in the oven at 50 °C before Py-GC-MS analysis to remove excess moisture from the samples.

Py-GC-MS analysis

Samples were analyzed using an Agilent 6890 GC and 5975 C MS (Santa Clara CA, USA) equipped with an Agilent DB-5HT column (30 m × 0.25 mm × 0.25 µm). The GC-MS was coupled with a multishot pyrolysis unit EGA/PY-3030D and autosampler AS-1020E (Frontier Laboratories, Saikon, Japan). Analyses were performed in double-shot mode, starting with thermal desorption (first shot) followed by pyrolysis (second shot) as previously developed and described by Brits et al. (2024). For thermal desorption, samples were first introduced to the pyrolysis unit at 100 °C and heated up to 300 °C at a rate of 50°C/min to separate the volatile and semi-volatile compounds, including additives, unpolymerized polymers, and other chemicals as a first clean-up step interferences. Then, pyrolysis (second shot) was performed at 600 °C for 18 seconds for the identification and quantification of the target polymers. GC oven program was started at 40 °C for 2 min and increased to 360 °C at a rate of 20°C/min, and 1 min hold at 360°C. Helium was used as the carrier gas at a flow rate of 3 mL/min (Brits et al. 2024). Measurements were done in selected ion monitoring (SIM) mode with a split ratio of 1:10. The inlet and transfer line temperatures were set at 300°C.

Six of the most commonly detected polymers in freshwater and drinking water were targeted including PE, PET, PMMA, PP, PS, and PVC (Koelmans et al. 2019). External calibration curves of the pyrolysis products of all target polymers were obtained by using a mixture of polymers at known concentrations obtained by pressurized liquid extraction (Thermo Scientific^{TM} ASE^{TM} 350 Accelerator Solvent Extractor 083146, Waltham, MA, USA) as described by (Sefiloglu et al. 2024). The polymer standards used for method development are listed in Table S2 in the Supplementary Information. Each indicator compound was confirmed based on the retention time (within 0.1 min of the reference standard's average) and the relative ratios of the characteristic fragment ions of the compound (within ±30% based on the reference standard). An example of this confirmation is provided for PET in Figure S2 in the Supplementary Information. Indicator compounds chosen to quantify the target polymers were 1-hexadecene (PE), benzoic acid (PET), methyl methacrylate (PMMA), styrene trimer (PS), 2,4-dimethyl-1-heptene (PP), and 2-methyl naphthalene (PVC). To monitor and compensate for instrumental variability, poly(4-fluorostyrene) was used as an injection standard, with 4-fluorostyrene selected as its characteristic pyrolysis product. Before the analysis of the samples, 0.25 µg of the injection standard was added to the pyrolysis cups, and the solvent was evaporated at 50 °C in the oven. The responses of the indicator compounds were normalized with respect to the injection standard's response. Data processing was performed using Agilent Mass Hunter software. The details of selected pyrolysis products of the target polymers were listed in Tables S3, the SIM method was summarized in Table S4 and examples of pyrograms of different sample types were shown in Figure S3 in the Supplementary Information.

Quality assurance and control

Procedural lab blanks and field blanks were prepared and analyzed identically to the samples. Procedural blanks (n=14) were processed on different days of sample filtration to account for the background laboratory contamination and instrumental variability by filtering 100 mL of pre-filtered demineralized water. The limit of detection (LOD) and limit of quantification (LOQ) for each indicator compound were determined as 3 and 10 times the standard deviation (SD) of the response of the quantifier ion in the procedural blanks, respectively. The average concentrations of the target polymers in the procedural lab blanks ranged from <LOD to 0.17 µg. All values detected in water samples above LOD were blank-corrected for individual polymers based on average procedural blank values. With this approach, the resulting effective detection and quantification thresholds are equivalent to those obtained using the stricter definitions of LOD = *mean* blank + 3*xSD of* blanks and LOQ = *mean* blank + 10*x SD* blank, as applied in recent literature (Le et al. 2025). LOD, LOQ, and average absolute amount of the polymers in blanks, and calibration curve information were listed in the Supplementary Information (Table S5). The same quantification parameters were applied to the tap water samples. Field blanks (n=54) were treated as samples and used to assess the statistical significance of differences in polymer concentrations between field samples and blanks. Raw and treated water samples were corrected for individual polymers with the corresponding field blank from the same date and location.

To evaluate the filtration efficiency (recovery), positive control samples were prepared by adding a known amount of polymers with known particle sizes to filtered tap water (filtered with 0.7 µm glass fiber filter). Polymer suspensions of the target plastics (excluding PET, as no standard was available) were purchased in the size range of 1–6 µm to closely match the filter mesh size and simulate the expected small size ranges present in drinking water. Detailed information on the standards is provided in the Supplementary Information (Table S6). The recovery experiments were conducted at 2 spiking levels (high and low) with three replicates, as recommended by (Koelmans et al. 2019). 100 mL of water was spiked with a mixture of polymers in two different concentration ranges in triplicate. The samples with higher concentrations contained 4 µg of PE, PVC, and PP, and 0.4 µg of PMMA and PS, while the samples with lower concentrations contained 0.5 µg of PE, PVC, and PP, and 0.1 µg of PMMA

and PS. After spiking, the spiked samples underwent the same filtration and rinsing process as the actual samples. For the control samples, unspiked pre-filtered tap water (n=3) was filtered, and the filters were transferred to the pyrolysis cups. The same amounts of polymers were then added directly to the top of the filters. Recoveries (%) were calculated based on the ratio between the response of the spiked samples and the control samples.

Statistical analysis

Statistical analysis and visualization were conducted using GraphPad Prism software v. 10.2.3. to determine the significance of differences in concentration levels in field blanks and samples. The Shapiro–Wilk test was used to assess the normality of the data for each polymer type in the samples. In the cases of normal and non-normal distribution, multiple unpaired t-tests and the Mann–Whitney U test were used for group comparison, respectively. To account for multiple comparisons, p-values were adjusted using the false discovery rate (FDR) method, with significance defined as $p < 0.05$ after FDR correction.

Results and discussion

Yields of recovery

The assessment of the MP particle recovery through filtration shows recoveries ranging from 52.9% to 118% at the lower spiking concentration and from 59.9% to 86.1% at the higher spiking concentration (Table 1). Higher variability in recoveries was observed for most analyzed compounds at the lower spiking concentration, with relative standard deviations (RSDs) ranging from 15.1% to 45.2%, while the RSDs at the higher spiking level ranged from 2.01% to 31.9%. This variability can be attributed to the greater impact of particle losses on recoveries at lower spiking levels. In addition, the lower spiking level is closer to the LOQ of the analysis, where higher variability is expected. However, even at higher spiking concentrations, RSD values for some polymers exceeded the generally acceptable threshold of ≤ 20% according to analytical method validation guidelines (European Commission 2021). This indicates that reproducibility remains a challenge when dealing with MP particles compared to homogeneously distributed chemical molecules in environmental samples. Particle-based analyses are particularly susceptible to losses during transfer of samples and vacuum filtration, which can result in losses and variability between replicates, as reported by other studies (Dimante-Deimantovica et al. 2022; Hagelskjær et al. 2023). In the present study, recoveries for both concentration ranges were determined in triplicate, following the minimum recommended number of replicates proposed by Koelmans et al. (2019) for MP quantification studies. However, we acknowledge that a larger number of replicates would improve statistical confidence and better characterize method variability in future work.

It is important to note that the use of virgin polymer standards may contribute to polymer losses due to their high static charges, which can cause adherence to glass surfaces and a tendency to roll off the filter during handling, such as cutting and folding (Wiggin and Holland 2019; Dimante-Deimantovica et al. 2022). Therefore, higher recoveries are anticipated for environmental samples, which are more likely to contain weathered particles with reduced static properties. We have previously observed particle adhesion to glass funnels when working with larger virgin particles; although the < 10 µm particles used in this recovery test were not visible, we assume that similar processes may have occurred. To mitigate this, additional rinsing with ethanol was employed; however, additional losses may still occur during filter handling steps, such as cutting and folding the filter prior to transfer into the pyrolysis cups. Further research is needed to evaluate the recovery of environmentally relevant particles using a glass filtration setup. The current lack of reference standards for weathered plastic particles in the necessary size ranges limits confirmation through representative spiking experiments.

Overall, existing literature on recovery experiments from water reports varying recovery yields mostly below 85%, highlighting a general tendency toward underestimation (Way et al. 2022; Hagelskjær et al. 2023). In this study, concentrations were not corrected for recovery, and therefore,

Table 1 Summary of recoveries of plastic particles in two spiking levels

	Size (µm)	Spiked amount (µg)	Recovery (Mean ± RSD)%, n = 3	Spiked amount (µg)	Recovery (Mean ± RSD)%, n = 3
PE	5–6	0.5	111 ± 33.8	4	77.4 ± 28.0
PMMA	4	0.1	52.9 ± 15.1	0.4	79.5 ± 24.5
PP	5	0.5	54.3 ± 33.7	4	62.1 ± 22.8
PS	1	0.1	59.4 ± 16.7	0.4	86.1 ± 2.01
PVC	5–5.5	0.5	118 ± 45.2	4	59.9 ± 31.9

the reported values may represent conservative estimates. While these recovery tests provide valuable insight into physical sample losses during filtration and subsequent sample transfer steps, this does not account for the chemical digestion treatment applied to raw water samples. This decision was based on findings of previous studies under comparable conditions, which reported minimal alterations in polymer mass and integrity following similar oxidative treatments (Pfeiffer and Fischer 2020; Schrank et al. 2022). However, we acknowledge that the influence of each applied step on mass recoveries should be evaluated experimentally, and we recognize this as a methodological limitation.

MPs in raw and treated water samples

Field blanks vs. real samples

In this study, field blanks ($n = 54$) were collected at each sampling date and location to monitor the possible background contamination during sampling. Polymer concentrations above the LOD in field blanks were statistically compared to those in water samples from each sampling location. The comparison per polymer type is in Fig. 2. Targeted MPs were detected (> LOD) in 92% of the field blanks collected from both DWTPs, while field blanks from Lek Canal and Bethune Polder locations contained the targeted compounds only in 50% and 43% of the field blank samples, respectively. These location-based differences in field blanks are likely related to the sampling environment: sampling at the DWTP locations was conducted inside the treatment plants, where higher levels of airborne contamination are anticipated. For some polymer types and locations, statistical comparisons were not feasible due to insufficient data, as polymers such as PMMA and PP were only detected in a few samples and field blanks. Notably, PET, PS, and PVC were abundantly detected in field blanks from the Leiduin DWTP and Weesperkarspel DWTP.

PVC and PE concentrations in water samples were significantly higher compared to the field blanks, except at the Lek Canal location, having only 3 blank samples > LOD, which does not provide a statistically meaningful comparison. PET levels in treated water samples from the DWTPs showed no significant differences compared to the blanks (Fig. 2)**.** Similarly, PS concentrations in samples from the Leiduin DWTP were not significantly different from field blanks, while samples from Weesperkarspel DWTP were significantly higher than blank values (Fig. 2). These findings emphasize the critical need to account for potential airborne contamination during sampling to ensure the reliability of MP measurements. Given the significant levels of PS and PET observed in the field blanks from the interior sampling locations, the field blank values for each polymer on each sampling date were subtracted from the corresponding sample. Based on these corrections, the total concentration of the targeted polymers per sampling location was reduced by 4.7 ± 9.5% for Lek Canal, 38 ± 24% for Leiduin DWTP, 4.4 ± 3.1% for Bethune Polder, and 35 ± 30% for Weesperkarspel DWTP samples.

MP concentrations

MPs were detected in all water samples collected from both the source and at the final stage of the water treatment process. Total measured MP concentrations (> LOD) for each sampling location are presented in Fig. 3. In raw water samples, concentrations ranged from 19.6 to 152 µg/L (median = 40.3 µg/L) and 11.2 to 108 µg/L (median = 33.5 µg/L) for Lek Canal and Bethune Polder, respectively. In treated water samples, MP concentrations ranged from 0.19 to 1.69 µg/L (median: 0.72 µg/L) for samples collected from Leiduin DWTP and from 0.18 to 8.62 µg/L (median: 1.20 µg/L) for samples collected from Weesperkarspel DWTP.

MP removal efficiency in DWTPs has been examined in several studies across different countries. Most of these studies use spectroscopic techniques, reporting concentrations in particles per water volume (Romphophak et al. 2024). In our study, based on the average MP concentrations in raw and treated water from two branches of drinking water production, MP removal efficiencies by mass were calculated as 98% and 97%, respectively. Similar removal rates have been reported in studies analyzing influent and effluent waters of DWTPs with comparable treatment processes, with values exceeding 90% for particles ranging from 20 µm to 5 mm (Dalmau-Soler et al. 2021; Negrete Velasco et al. 2023; Zhou et al. 2023). Bäuerlein et al. (2022) reported MP removal efficiencies between 80 and 99% in the Dutch drinking water production system, including the Lek Canal branch examined in our study. In that study, laser direct infrared spectroscopy (LDIR) and optical microscopy were employed for MP detection. MP removal in the Lek Canal branch was reported to be greater than 99% for particle sizes between 20 µm and 500 µm, with concentrations in intake and final drinking water recorded as > 400 MP/L and 0.241 MP/L, respectively. Another study investigating MPs in a Dutch drinking water system from groundwater sources reported significantly lower concentrations in the source and treated water with values ranging from 0 to 0.007 MP/L for the particles > 20 µm using µ-FTIR analysis (Mintenig et al. 2019).

Given the methodological differences and size ranges analyzed, a direct comparison of our mass-based results and the findings of Bäuerlein et al. (2022) is challenging. Studies using spectroscopic techniques potentially report lower MP particle counts due to particle size detection cut-off at the 5- 20 µm range, whereas our mass-based analysis with Py-GC–MS includes much smaller particles (≥ 0.7 µm). The MP removal efficiency calculated in our study was slightly

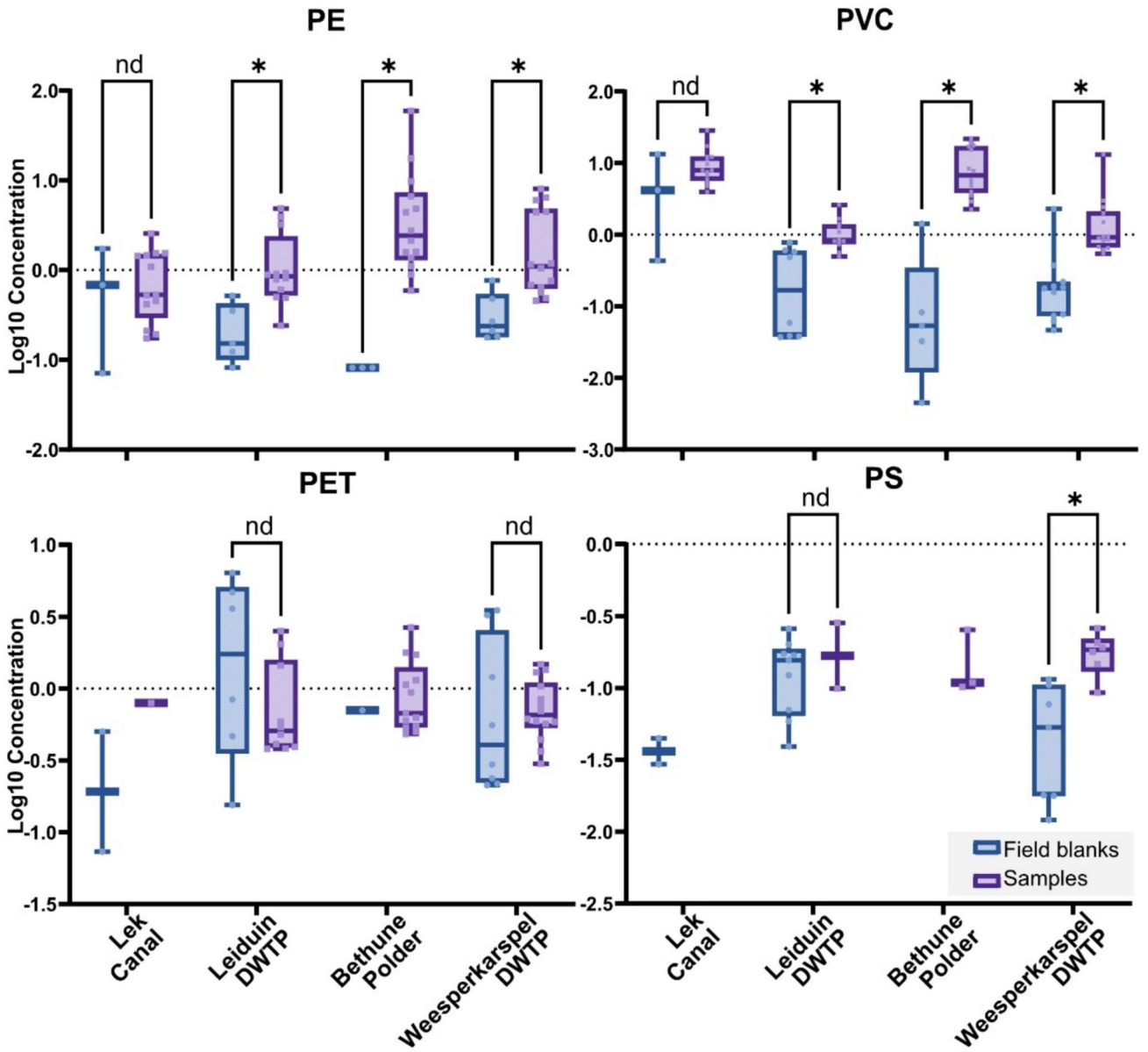


Fig. 2 Comparison of field blanks and samples from four locations. Box plots display the minimum, maximum, and median of the measured concentrations, while individual sample concentrations are represented by dots. Data < LOD were excluded from the analysis. Multiple comparisons were corrected using the false discovery rate (FDR) method. Asterisks (*) indicate statistically significant differences ($p < 0.05$, FDR-corrected), whereas "nd" indicates no significant difference after FDR correction. Results should be interpreted with caution, as at certain locations, such as the Lek Canal, only a few blank samples exhibited contamination (e.g., for PE and PVC)

lower, which may be attributed to the inclusion of smaller particles in our analysis. In another study, Maurizi et al. (2023) reported reduced MP removal rates with decreasing particle sizes in a Danish DWTP. Using μ-Raman analysis, they observed removal rates of 41.1% for particles in the 1–5 μm range, increasing to 74.6% for particles sized 5–10 μm and 79.6% for those larger than 10 μm. It should be noted that smaller particle sizes contribute less to total mass; therefore, mass-based removal efficiency must be considered with this factor in mind.

Mass concentrations of MPs in drinking water production processes were examined in only a few studies using Py-GC–MS. Kirstein et al. (2021) analyzed MPs (PE, PET, PS, PMMA, PP, PVC, polyamide (PA), polycarbonate (PC)) in the treated water samples taken from different stages of the drinking water distribution system in Sweden, reporting total concentrations from 0.00014 to 0.00543 μg/L (originally reported as 0.14–5.43 μg/m^3) for particles ≥ 5 μm. Gomiero et al. (2021) investigated mass concentrations of the same polymers in the drinking water supply system in

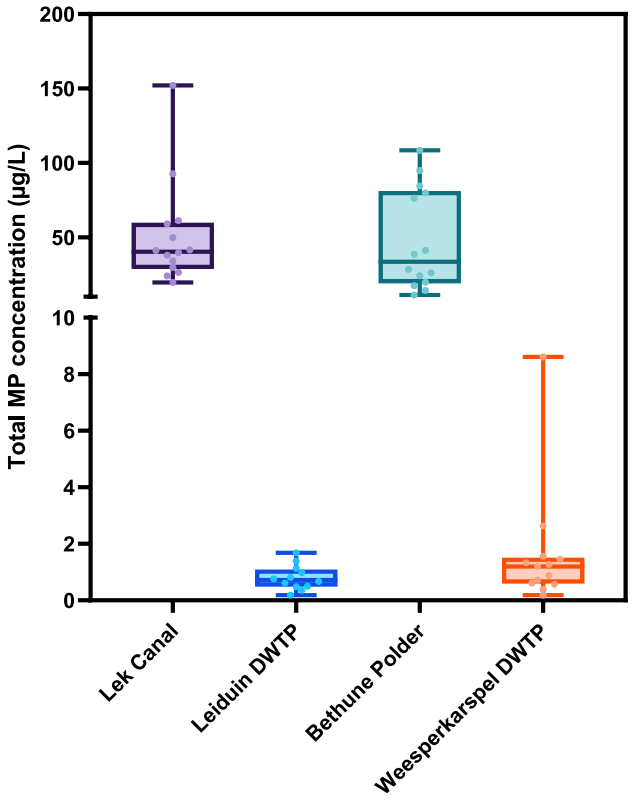


Fig. 3 Total MP concentrations based on 6 targeted polymers in the 2 branches of drinking water production. Lek Canal ($n=14$) and Bethune Polder ($n=14$) samples correspond to the raw water, and the samples taken from Leiduin ($n=12$) and Weesperkarspel ($n=14$) DWTP are treated water. Box plots show the min, max, and median of the measured concentrations, and the dots represent the individual sample concentrations

Norway with analysis of particles ≥ 1 µm revealing concentrations of 0.0931 µg/L (originally reported as 93.1 µg/m^3) in raw water and 0.0061–0.0128 µg/L (originally reported as 6.1–12.8 µg/m^3) in treated water. Xu et al. (2024) examined MPs (PMMA, PA, PS, PET, PS) in raw and treated water in China across size ranges from 0.01 µm to 1000 µm, reporting average concentrations of 9.63 µg/L and 0.77 µg/L in raw and treated water, respectively. Similarly, Dalmau-Soler et al. (2024) analyzed PMMA, PP, PE, PET, PVC, PS, and PC in raw and treated water samples from Barcelona, Spain for particles ≥ 1 µm, finding total MP concentrations of 77 µg/L in raw water and 11.3 µg/L in treated water. The MP concentrations observed in our study fall within the range of these findings (ranging from 0.18 to 8.7 µg/L). Variations in reported concentrations can be attributed to differences in MP levels in the source water, the efficiency of the treatment processes, and methodological variations in sampling and analysis.

Occurrence of polymers

Of the six polymers targeted in this study, PVC, PE, and PET were the most abundant types above the limit of detection (LOD) in both raw and treated water, with detection rates of 96%, 91%, and 57%, respectively. Following these, PS was detected in 15% of the water samples, while PP and PMMA were detected in only a few samples ($n=2$, each). The mass concentration ranges of these polymers across four sampling points are shown in Fig. 4, and the concentrations per polymer and sampling location are listed in the Supplementary Information (Table S7). For PVC, average concentrations of 48.5 ± 31.8 ($n=14$) and 47.5 ± 33.7 µg/L ($n=14$) were recorded from the raw water samples, while average concentrations after treatment were found to be 0.46 ± 0.29 ($n=11$) and 1.01 ± 1.65 µg/L ($n=13$), respectively. Although PE was frequently detected, its mass concentrations were lower than those of PVC, with mean concentrations of 2.37 ± 2.65 (n = 11) and 9.52 ± 8.56 µg/L ($n=14$) in raw water samples and 0.25 ± 0.13 ($n=12$) and 0.48 ± 0.55 µg/L (n = 12) in treated water. PET was most abundant in the Bethune Polder branch, averaging 3.99 ± 2.66 µ/L in raw water ($n=13$) and 0.21 ± 0.18 µg/L in treated water ($n=10$). In contrast, PET was detected in only one raw water sample from Lek Canal (3.13 µg/L), as the treated water from the same branch had an average concentration of 0.18 ± 0.24 µg/L ($n=7$). It should be noted that these samples represent snapshots from specific sampling days; the full treatment process can take up to three months, so the raw water samples do not correspond to the treated water from the same day.

Previous studies conducted in the Dutch drinking water network using untargeted spectroscopic techniques have reported up to 25 different polymer types. Mintenig et al. (2019) identified PET, PVC, PE, PA, and epoxy resin in groundwater-sourced drinking water. The presence of these polymers was attributed to their extensive use in DWTP systems, as they are commonly found in pipes and plumbing fittings. Bäuerlein et al. (2022) detected PA, PET, rubbers, PE, and chlorinated PE (CPE) as the most abundant polymers, with PVC also being present in both raw and treated water samples. The high abundance of CPE is particularly noteworthy, as it produces pyrolysis products similar to PVC (Cheng-Yu Wang and Smith 1997). This overlap may contribute to the elevated PVC concentrations observed in our study, highlighting the need for further investigation.

Studies from various other countries investigating the distribution of polymers in drinking water networks have also reported similar polymer-type distributions. Negrete Velasco et al. (2023) monitored MPs in waters at different stages of drinking water treatment in Geneva, Switzerland using µ-FTIR, identifying PVC, PE, and PET as the most abundant polymers in the treated water. The LDIR analysis employed by Sun et al. (2024) revealed PET, PE, PVC, and PP as the

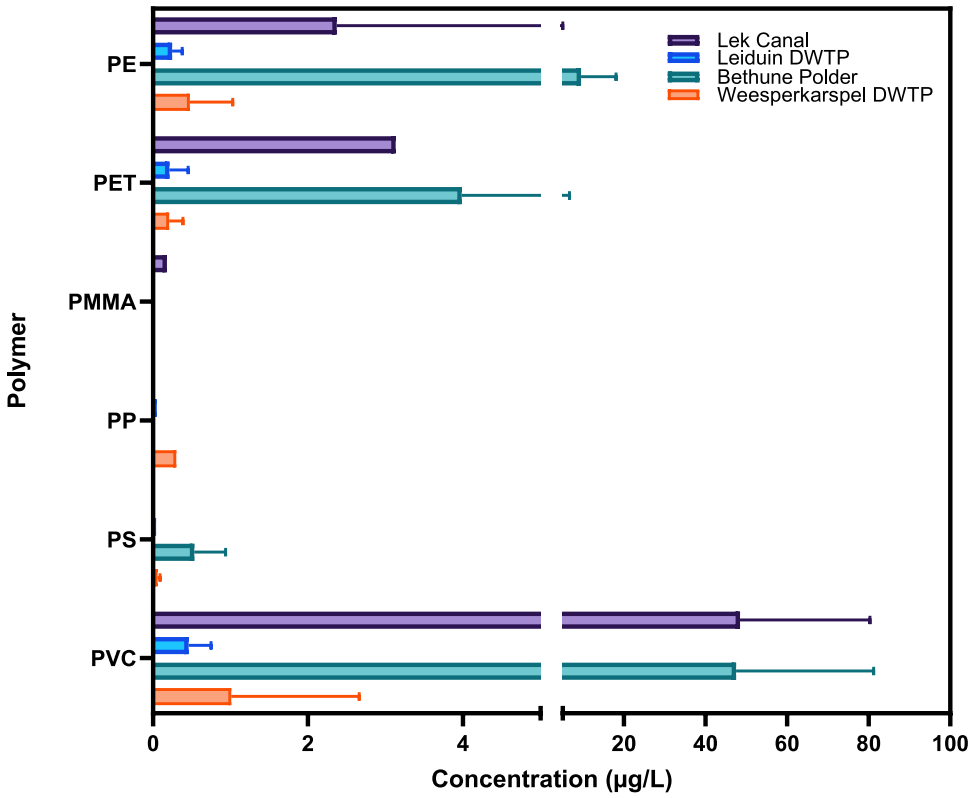


Fig. 4 Polymer concentrations per sampling location. Bar plots show mean mass concentrations (µg/L) with standard deviation

most abundant polymers in raw and treated water in China. In another study, Li et al. (2024) analyzed nanoplastics (20–1000 nm) using atomic force microscopy-based infrared spectroscopy (AFM-IR) and Py-GC–MS, reporting PE and PVC as the dominant polymers in influent and treated water from a DWTP in China, with PVC concentrations reaching 76.83 µg/L in treated water. Similarly, Dalmau-Soler et al., (2024) examined the Spanish drinking water network and found PVC and PP to be the most abundant polymers in raw and treated water, with PVC concentrations of 53.5 µg/L in raw water and 9.3 µg/L in treated water.

MPs in household tap water

The total polymer concentrations based on 6 targeted polymers in the tap water samples ranged from 0.09 to 0.57 µg/L (median 0.18 µg/L). The concentration ranges for each polymer are shown in Fig. 5, and the results of individual samples are listed in Supplementary Information (Table S8). Consistent with the findings in post-treatment water samples, PE and PVC were detected in all of the tap water samples in this study, while PS was only detected in 1 sample above the LOD. The average concentrations of PVC and PE were 0.11 and 0.09 µg/L, respectively.

Previous studies have reported varying MP concentrations, ranging from 0.00014 to 11.3 µg/L, in treated water samples from DWTP outlets and distribution networks, as discussed in Section "MP concentrations". However, data on MP mass concentrations in consumer tap water remains limited. In this study, the MP concentrations in household tap water samples were lower than those in the treated water samples. This difference should be interpreted in light of the sampling approaches: household tap water was collected using an in-line filtration system with minimized exposure to air, whereas treated water samples from the DWTPs were collected in bulk from interior sampling points within the plants. Filtering larger volumes of tap water (100 L) in a closed system likely reduced the potential for airborne contamination compared to the smaller volume (2 L) collected for DWTP samples (Belz et al. 2024). The significant presence of PET and PS in field blanks indicates that airborne MPs likely contributed to the samples, yet even after field blank correction, concentration differences persisted. Additionally, while bulk DWTP samples were filtered via vacuum filtration and analyzed in their entirety, tap water samples were subsampled from the filters for analysis. Another possible contributor to the lower concentrations measured in tap water is deterioration of the glass fiber filters during filtration of large volumes (100 L), which could reduce recovery efficiency. This possibility warrants further investigation but was beyond the scope of the present study. Harmonizing sampling volumes and assessing filter integrity could improve comparability in future work.

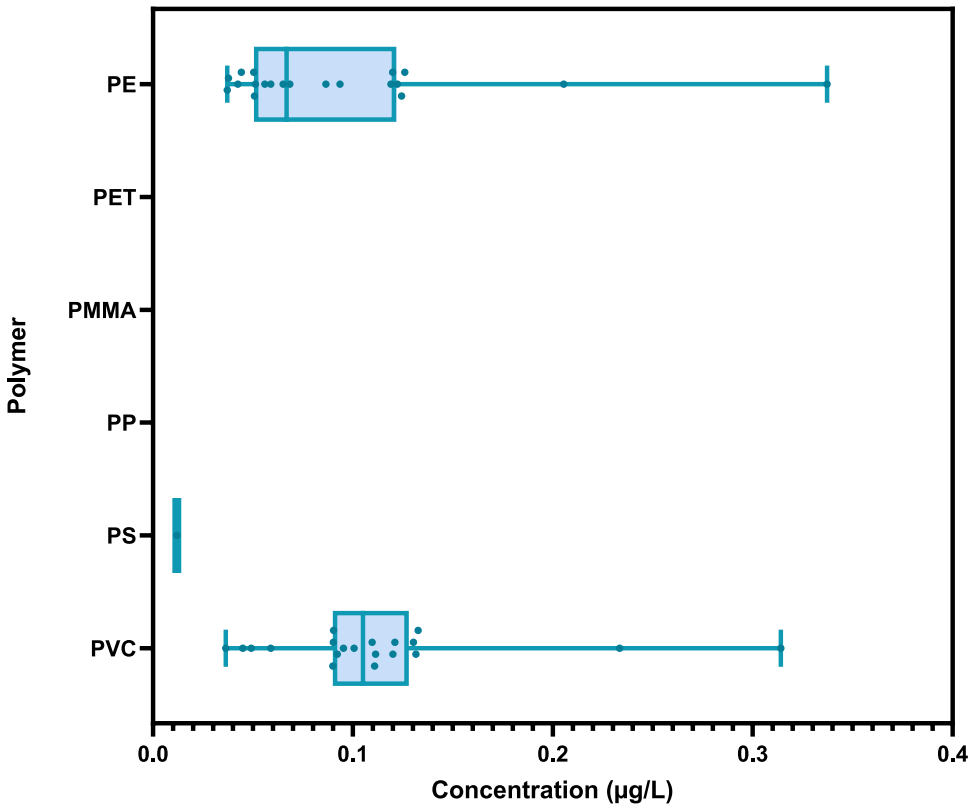


Fig. 5 Concentration of the polymers in tap water samples ($n = 20$) from 10 different locations. Box plots show the min, max, and median of the measured concentrations, and the dots represent the individual sample concentrations

Notably, PET was not detected above LOD in tap water samples, while PVC concentrations were also reduced compared to treated water. This reduction may be explained by the possible settling and adsorption of these polymers during storage in the tanks and distribution through the pipes, as the densities of these polymers are greater than 1 g/cm^3 (Chu et al. 2022; Yang et al. 2023). Similar trends have been observed in other studies, which reported higher MP concentrations in DWTP effluents than in household tap water (Chu et al. 2022; Sun et al. 2024). Chu et al. (2022) also identified polymers, such as PA and PVC, on pipe scale samples, while Yang et al. (2023) found PET to be the predominant polymer in pipe scales. Conversely, some studies have reported higher MP concentrations in household water than in DWTP effluent, associating the difference with the material abrasion in the distribution network (Taghipour et al. 2023; Zhou et al. 2023). In our study, the absence of PET and reduced PVC and PE concentrations in tap water suggest that abrasion likely has a minor effect, as any significant contribution from abrasion would have been evident in the data. Given that polymeric materials constitute much of the current piping infrastructure, understanding the impact of distribution networks on MP levels in tap water is crucial. These opposing findings suggest that further research is needed to better understand the dynamics of MP transport, accumulation, and potential release within distribution systems.

Comparison of tap water MP exposure with other dietary and inhalation sources

Based on the results of this study, assuming an average daily consumption of 1 L of tap water, the estimated daily MP intake is approximately 0.18 µg per person, corresponding to an annual intake of 6.57×10^{-5} g per capita. This value, derived from our mass spectrometry analysis, is compared below with reported MP exposures from other food, beverage, and air sources.

Human exposure to MPs has been estimated in various studies based on current knowledge of MPs in food, drinking water, beverages, and air (Mohamed Nor et al. 2021; WHO 2022). Among the various pathways, inhalation and shellfish consumption have been identified as major sources of MP intake, whereas tap water contributes relatively little (WHO 2022). Most of these studies rely on spectroscopic techniques that report particle counts per sample, which makes direct comparisons to our mass-based results challenging. While mass-based comparisons are possible, they should be interpreted with caution, as differences in size detection limits, analytical approaches, and targeted polymer types can significantly affect reported concentrations, potentially leading to discrepancies.

Senathirajah et al., (2021) estimated the mass of MP ingestion in the range of 0.1–5 g/week/capita. Based on their findings, the estimated yearly MP intake from shellfish, salt,

beer, and drinking water was 26.4 g, 7.4 g, 0.5 g, and 0.08 g per capita, respectively. As a comparison to these estimates and our results, we refer to available empirical data from other mass spectrometry-based analyses. In the study of Ribeiro et al. (2020), MPs in seafood were examined, and potential MP exposure via oysters, crabs, and sardines was reported as 0.7 mg, 3 mg, and 30 mg per serving. In another study, Kwon et al. (2025) investigated MPs in different edible salts with Py-GC–MS, revealing annual MP exposure from sea salt at 2.3 mg per person. Braun et al., (2021) analyzed beverages in plastic bottles using thermal extraction desorption-gas chromatography-mass spectrometry (TED-GC–MS), reporting PET concentrations ranging from 0.15–4.61 µg/L. Besides ingestion, exposure to MPs through inhalation is an important pathway. Hashemihedeshi et al. (2024) quantified MPs in indoor air at levels up to 24 µg/m^3, which corresponds to an estimated annual inhalation exposure of 88 mg per capita, assuming an average daily inhalation of 15 m^3/day.

Overall, our results confirm that tap water is a minor pathway for MP exposure relative to other known ingestion and inhalation sources. However, the available empirical data on human MP exposure is still incomplete, especially for fruits, vegetables, grains, and cereals, which constitute 50% of daily food consumption (WHO 2022). Further research is needed to address these knowledge gaps and to provide a more comprehensive understanding of human exposure to MPs via ingestion from various dietary sources.

Conclusions

In this study, the identification and quantification of MPs in the drinking water production system were investigated using Py-GC–MS. Six target polymers, including PE, PP, PS, PMMA, PET, and PVC, were analyzed in raw water, treated water, and household tap water. Samples were collected from two branches of drinking water production. The average mass concentrations in raw waters were 50.6 ± 34.7 ($n = 14$) and 47.5 ± 33.7 µg/L ($n = 14$), while the average concentrations in treated water from the two DWTP outlets were 0.80 ± 0.44 µg/L ($n = 12$) and 1.65 ± 2.19 µg/L ($n = 14$). The removal efficiencies of the DWTPs were 97–98% based on average mass concentrations. Household tap water samples ($n = 20$) exhibited total MP concentrations ranging from 0.09 to 0.57 µg/L. PVC and PE were identified as the most abundant polymers across all water types.

Field blanks collected during sampling highlighted the importance of accounting for background air contamination, particularly for bulk samples. Significant levels of PET and PS were observed in field blanks from interior sampling locations at the DWTPs, emphasizing the influence of airborne MPs in such environments. The difference between DWTP effluents and household tap water was attributed to the potential settling and adsorption of MPs within the water distribution network, as well as differences in sampling techniques. These processes, however, warrant further investigation. A comparison of our results with other available mass-based data revealed that the intake of MPs through investigated tap water is minor compared to MP intake through other food and inhalation.

This study provides valuable insights into the occurrence of MPs in drinking water supply systems, extending the analysis to particle sizes as small as 0.7 µm. The use of Py-GC–MS for MP analysis in water samples needs further research and application. Increasing the analyzed water volumes and expanding the target polymer list would enhance the reliability and comparability of the data. Furthermore, the application of complementary techniques alongside harmonized methodologies is essential to align findings from different studies and advance our understanding of microplastic contamination in drinking water systems.

Supplementary Information The online version contains supplementary material available at https://doi.org/10.1007/s11356-025-37130-8.

Acknowledgements We would like to thank Het Waterlaboratorium (HWL, the Netherlands) for the collaboration and for conducting the sampling. Furthermore, we thank Daniel Giesen for the coordination of the sampling. We thank Sara Evangelista for the help during the data interpretation.

A. Dick Vethaak passed away during the preparation of the manuscript on 1 June 2024, we are grateful for all his input into this work and his supervision throughout the project.

Authors' contributions Feride Öykü Sefiloglu: Conceptualization, Methodology, Software, Validation, Formal analysis, Investigation, Writing—Original Draft, Visualization; Marthinus Brits: Conceptualization, Methodology, Validation, Writing—Review & Editing, Supervision; Martin J.M. van Velzen: Methodology, Writing—Review & Editing; Eelco N. Pieke: Conceptualization, Resources, Writing—Review & Editing, Supervision; Onno J.I. Kramer: Writing—Review & Editing, Resources, Supervision; Marja Lamoree: Conceptualization, Writing—Review & Editing, Supervision, Project administration.

Funding This project has received funding from the European Union's Horizon 2020 research and innovation programme under the Marie Skłodowska-Curie grant agreement No 860720.

Data availability Data supporting the results of this study will be available on request.

Declarations

Ethics approval and consent to participate Not applicable.

Consent to participate Not applicable.

Consent to publish Not applicable.

Competing interests The authors declare no competing interests.

References

Albignac M, de Oliveira T, Landebrit L, Miquel S, Auguin B, Leroy E, Maria E, Mingotaud AF, ter Halle A (2023) Tandem mass spectrometry enhances the performances of pyrolysis-gas chromatography-mass spectrometry for microplastic quantification. J Anal Appl Pyrolysis. https://doi.org/10.1016/j.jaap.2023.105993

Bäuerlein PS, Hofman-Caris RCHM, Pieke EN, ter Laak TL (2022) Fate of microplastics in the drinking water production. Water Res. https://doi.org/10.1016/j.watres.2022.118790

Belz S, Cella C, Geiss O, Gilliland D, La Spina R, Méhn D, Sokull-Kluettgen B (2024) Analytical methods to measure microplastics in drinking water. Luxembourg

Braun U, Altmann K, Herper D, Knefel M, Bednarz M, Bannick CG (2021) Smart filters for the analysis of microplastic in beverages filled in plastic bottles. Food Addit Contam Part A Chem Anal Control Expo Risk Assess 38(4):691–700. https://doi.org/10.1080/19440049.2021.1889042

Brits M, van Velzen MJM, Sefiloglu FÖ, Scibetta L, Groenewoud Q, Garcia-Vallejo JJ, Vethaak AD, Brandsma SH, Lamoree MH (2024) Quantitation of micro and nanoplastics in human blood by pyrolysis-gas chromatography–mass spectrometry. Microplast Nanoplast 4(1):12. https://doi.org/10.1186/s43591-024-00090-w

Cheng-Yu Wang F, Smith PB (1997) Composition and microstructure analysis of chlorinated polyethylene by pyrolysis gas chromatography and pyrolysis gas chromatography/mass spectrometry. Anal Chem 7(69 (4)):618–622. https://doi.org/10.1021/ac960947c

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. https://doi.org/10.1016/j.scitotenv.2021.150004

Dalmau-Soler J, Ballesteros-Cano R, M[a], Boleda 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). https://doi.org/10.1007/s11356-021-13220-1/Published

Dalmau-Soler J, Boleda MR, Lacorte S (2024) Routine method for the analysis of microplastics in natural and drinking water by pyrolysis coupled to gas chromatography-mass spectrometry. J Chromatogr A. https://doi.org/10.1016/j.chroma.2024.465153

Dimante-Deimantovica I, Suhareva N, Barone M, Putna-Nimane I, Aigars J (2022) Hide-and-seek: threshold values and contribution towards better understanding of recovery rate in microplastic research. MethodsX. https://doi.org/10.1016/j.mex.2021.101603

EU (2020) Directive (EU) 2020/2184 of the European Parliament and of the Council of 16 December 2020 on the quality of water intended for human consumption (recast)(Text with EEA relevance). Off J Eur Union

EU (2024) Commission Delegated Decision (EU) 2024/1441 of 11 March 2024 supplementing Directive (EU) 2020/2184 of the European Parliament and of the Council by laying down a methodology to measure microplastics in water intended for human consumption (notified under document C(2024) 1459). Off J Eur Union

European Commission (2021) Analytical quality control and method validation procedures for pesticide residues analysis in food and feed (SANTE/11312/2021 rev. 2). Brussels, Belgium

Gomiero A, Øysæd KB, Palmas L, Skogerbø G (2021) Application of GCMS-pyrolysis to estimate the levels of microplastics in a drinking water supply system. J Hazard Mater. https://doi.org/10.1016/j.jhazmat.2021.125708

Hagelskjær O, Crézé A, Le Roux G, Sonke JE (2023) Investigating the correlation between morphological features of microplastics (5–500 µm) and their analytical recovery. Microplast Nanoplast. https://doi.org/10.1186/s43591-023-00071-5

Hashemihedeshi M, Haywood E, Gatch DC, Jantunen L, Helm PA, Diamond ML, Dorman FL, Cahill LS, Jobst KJ (2024) Size-resolved identification and quantification of micro/nanoplastics in indoor air using pyrolysis gas chromatography-ion mobility mass spectrometry. J Am Soc Mass Spectrom 35(2):275–284. https://doi.org/10.1021/jasms.3c00362

Hermabessiere L, Rochman CM (2021) Microwave-assisted extraction for quantification of microplastics using pyrolysis-gas chromatography/mass spectrometry. Environ Toxicol Chem 40(10):2733–2741. https://doi.org/10.1002/etc.5179

Huang Z, Wu E, Shi D, Zhang J, Shen Z, Zheng R, Shen J (2024) Six microplastics analysis in bottled water, purified tap water and branded table salt by double-shot pyrolysis–gas chromatography/mass spectrometry. Chromatographia. https://doi.org/10.1007/s10337-024-04359-3

Johnson AC, Ball H, Cross R, Horton AA, Jürgens MD, Read DS, 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(19):12326–12334. https://doi.org/10.1021/acs.est.0c03211

Kankanige D, Babel S (2020) Smaller-sized micro-plastics (MPs) contamination in single-use PET-bottled water in Thailand. Sci Total Environ. https://doi.org/10.1016/j.scitotenv.2020.137232

Kirstein IV, Hensel F, Gomiero A, Iordachescu L, Vianello A, Wittgren HB, Vollertsen J (2021) Drinking plastics? – quantification and qualification of microplastics in drinking water distribution systems by µFTIR and Py-GCMS. Water Res. https://doi.org/10.1016/j.watres.2020.116519

Koelmans AA, Mohamed Nor NH, Hermsen E, Kooi M, Mintenig SM, De France J (2019) Microplastics in freshwaters and drinking water: critical review and assessment of data quality. Water Res 155:410–422

Kurniawan TA, Haider A, Ahmad HM, Mohyuddin A, Umer Aslam HM, Nadeem S, Javed M, Othman MHD, Goh HH, Chew KW (2023) Source, occurrence, distribution, fate, and implications of microplastic pollutants in freshwater on environment: A critical review and way forward. Chemosphere 325. https://doi.org/10.1016/j.chemosphere.2023.138367

Kwon J, Kim H, Siddiqui Z, Kang S, Choi H, Kumagai S, Watanabe A, Teramae N, Kwon E, Kim M (2025) A comprehensive pyrolysis-gas chromatography/mass spectrometry analysis for the assessment of microplastics in various salts. Food Chem. https://doi.org/10.1016/j.foodchem.2024.142193

la 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. https://doi.org/10.1016/j.scitotenv.2023.168378
Le QNP, Halsall C, Peneva S, Wrigley O, Braun M, Amelung W, Ashton L, Surridge BWJ, Quinton J (2025) Towards quality-assured measurements of microplastics in soil using fluorescence microscopy. Anal Bioanal Chem 417(11):2225–2238. https://doi.org/10.1007/s00216-025-05810-6
Li H, Zhu L, Ma M, Wu H, An L, Yang Z (2023) Occurrence of microplastics in commercially sold bottled water. Sci Total Environ. https://doi.org/10.1016/j.scitotenv.2023.161553
Li Y, Zhang C, Tian Z, Cai X, Guan B (2024) Identification and quantification of nanoplastics (20–1000 nm) in a drinking water treatment plant using AFM-IR and Pyr-GC/MS. J Hazard Mater. https://doi.org/10.1016/j.jhazmat.2023.132933
Maurizi L, Iordachescu L, Kirstein IV, Nielsen AH, Vollertsen J (2023) Do drinking water plants retain microplastics? An exploratory study using Raman micro-spectroscopy. Heliyon. https://doi.org/10.1016/j.heliyon.2023.e17113
Mintenig SM, Löder MGJ, Primpke S, Gerdts G (2019) Low numbers of microplastics detected in drinking water from ground water sources. Sci Total Environ 648:631–635. https://doi.org/10.1016/j.scitotenv.2018.08.178
Mohamed Nor NH, Kooi M, Diepens NJ, Koelmans AA (2021) Lifetime accumulation of microplastic in children and adults. Environ Sci Technol 55(8):5084–5096. https://doi.org/10.1021/acs.est.0c07384
Negrete Velasco A, Ramseier Gentile S, Zimmermann S, Le Coustumer P, Stoll S (2023) Contamination and removal efficiency of microplastics and synthetic fibres in a conventional drinking water treatment plant in Geneva, Switzerland. Sci Total Environ. https://doi.org/10.1016/j.scitotenv.2023.163270
Okoffo ED, Thomas KV (2024) Quantitative analysis of nanoplastics in environmental and potable waters by pyrolysis-gas chromatography–mass spectrometry. J Hazard Mater. https://doi.org/10.1016/j.jhazmat.2023.133013
Pfeiffer F, Fischer EK (2020) Various digestion protocols within microplastic sample processing—evaluating the resistance of different synthetic polymers and the efficiency of biogenic organic matter destruction. Front Environ Sci. https://doi.org/10.3389/fenvs.2020.572424
Pittroff M, Müller YK, Witzig CS, Scheurer M, Storck FR, Zumbülte N (2021) Microplastic analysis in drinking water based on fractionated filtration sampling and Raman microspectroscopy. Environ Sci Pollut Res Int 28(42):59439–59451. https://doi.org/10.1007/s11356-021-12467-y
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. https://doi.org/10.1016/j.scitotenv.2020.140236
Ribeiro F, Okoffo ED, O'Brien JW, Fraissinet-Tachet S, O'Brien S, Gallen M, Samanipour S, Kaserzon S, Mueller JF, Galloway T, Thomas KV (2020) Quantitative analysis of selected plastics in high-commercial-value Australian seafood by pyrolysis gas chromatography mass spectrometry. Environ Sci Technol 54(15):9408–9417. https://doi.org/10.1021/acs.est.0c02337
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. https://doi.org/10.1016/j.jwpe.2024.105669
Schrank I, Möller JN, Imhof HK, Hauenstein O, Zielke F, Agarwal S, Löder MGJ, Greiner A, Laforsch C (2022) Microplastic sample purification methods - assessing detrimental effects of purification procedures on specific plastic types. Sci Total Environ. https://doi.org/10.1016/j.scitotenv.2022.154824
Sefiloglu FÖ, Stratmann CN, Brits M, van Velzen MJM, Groenewoud Q, Vethaak AD, Dris R, Gasperi J, Lamoree MH (2024) Comparative microplastic analysis in urban waters using μ-FTIR and Py-GC-MS: a case study in Amsterdam. Environ Pollut. https://doi.org/10.1016/j.envpol.2024.124088
Senathirajah K, Attwood S, Bhagwat G, Carbery M, Wilson S, Palanisami T (2021) Estimation of the mass of microplastics ingested – a pivotal first step towards human health risk assessment. J Hazard Mater. https://doi.org/10.1016/j.jhazmat.2020.124004
Shruti VC, Kutralam-Muniasamy G, Pérez-Guevara F, Roy PD, Elizalde-Martínez I (2022) Free, but not microplastic-free, drinking water from outdoor refill kiosks: a challenge and a wake-up call for urban management. Environ Pollut. https://doi.org/10.1016/j.envpol.2022.119800
Sun X, Zhu Y, An L, Liu Y, Zhuang Y, Wang Y, Sun M, Xu Q (2024) Microplastic transportation in a typical drinking water supply: from raw water to household water. Water (Basel) 16(11):1567. https://doi.org/10.3390/w16111567
Taghipour H, Ghayebzadeh M, Ganji F, Mousavi S, Azizi N (2023) Tracking microplastics contamination in drinking water in Zahedan, Iran: from source to consumption taps. Sci Total Environ. https://doi.org/10.1016/j.scitotenv.2023.162121
Tong H, Jiang Q, Hu X, Zhong X (2020) Occurrence and identification of microplastics in tap water from China. Chemosphere. https://doi.org/10.1016/j.chemosphere.2020.126493
Way C, Hudson MD, Williams ID, Langley GJ (2022) Evidence of underestimation in microplastic research: a meta-analysis of recovery rate studies. Sci Total Environ. https://doi.org/10.1016/j.scitotenv.2021.150227
WHO (2022) Dietary and inhalation exposure to nano-and microplastic particles and potential implications for human health. Geneva
Wiggin KJ, Holland EB (2019) Validation and application of cost and time effective methods for the detection of 3–500 μm sized microplastics in the urban marine and estuarine environments surrounding Long Beach, California. Mar Pollut Bull 143:152–162. https://doi.org/10.1016/j.marpolbul.2019.03.060
Xu Y, Ou Q, Wang X, van der Hoek JP, Liu G (2024) Mass concentration and removal characteristics of microplastics and nanoplastics in a drinking water treatment plant. ACS ES T Water 4(8):3348–3358. https://doi.org/10.1021/acsestwater.4c00222
Yang X, Xu X, Zhou Y, Yao Y, Shen C, Liu J (2023) Longitudinal and vertical distribution of microplastics in various pipe scales in an operating drinking water distribution system. J Hazard Mater. https://doi.org/10.1016/j.jhazmat.2023.132108
Zhou G, Wu Q, Wei XF, Chen C, Ma J, Crittenden JC, Liu B (2023) Tracing microplastics in rural drinking water in Chongqing, China: their presence and pathways from source to tap. J Hazard Mater. https://doi.org/10.1016/j.jhazmat.2023.132206
Zuccarello P, Ferrante M, Cristaldi A, Copat C, Grasso A, Sangregorio D, Fiore M, Oliveri Conti G (2019) Exposure to microplastics (<10 Mm) associated to plastic bottles mineral water consumption: the first quantitative study. Water Res 157:365–371. https://doi.org/10.1016/j.watres.2019.03.091

Authors and Affiliations

Feride Öykü Sefiloglu[1,5] · Marthinus Brits[1,6] · Martin J. M. van Velzen[1] · Eelco N. Pieke[2] · Onno J. I. Kramer[3,4] · Marja H. Lamoree[1]

✉ Feride Öykü Sefiloglu
f.o.sefiloglu@vu.nl

Marja H. Lamoree
marja.lamoree@vu.nl

[1] Amsterdam Institute for Life and Environment (A-LIFE), Vrije Universiteit Amsterdam, De Boelelaan 1085, Amsterdam 1081 HV, The Netherlands

[2] Het Waterlaboratorium N.V., Haarlem, The Netherlands

[3] Waternet, PO Box 94370, Amsterdam 1090 GJ, The Netherlands

[4] School of Engineering and Materials Science, Centre for Sustainable Engineering, Queen Mary University of London, London, UK

[5] Present Address: TNO Environmental Modelling, Sensing and Analysis, Princetonlaan 6-8, Utrecht 3584 CB, The Netherlands

[6] Present Address: The Southern African Grain Laboratory (SAGL), Grain Building-Agri Hub Office Park, 477 Witherite Street, The Willows, Pretoria 0040, South Africa