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Assessing microplastic contamination in Icelandic soils: Insights from remote, agricultural, and urban environments

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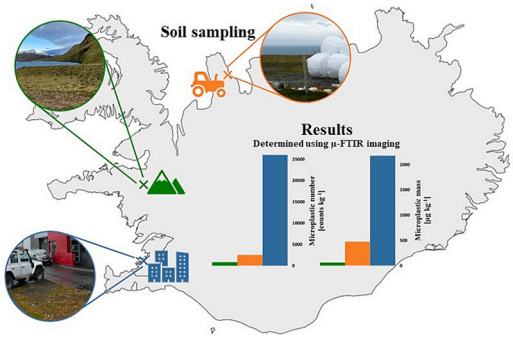
HIGHLIGHTS

- Soil with different land uses was investigated for MPs using μ-FTIR imaging.
- MP was present in both remote, agricultural, and urban soils.
- Increase in both MP count and mass with increasing anthropogenic activity.
- Predominantly medium-sized lightweight fragment presence in remote soils.

GRAPHICAL ABSTRACT



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ABSTRACT

Microplastic (MP) contamination is a growing concern across many environments. However, research on MP accumulation in remote soils remains limited. This study investigated MP abundance in remote soil in western Iceland, with agricultural and urban soils as references. The MP was extracted and measured using state-of-the-art methods, capturing types, sizes, and shapes. Results showed that MP was present in all samples, with the lowest average count and mass observed in the remote samples (857 (±561) counts kg^{-1}, corresponding to 64.37 (±47.96) μg kg^{-1}) and the highest in the urban samples (26,206 (±25,345) counts kg^{-1}, corresponding to 2175 (±1385) μg kg^{-1}), showing that increase in anthropogenic activity gives increasing MP concentrations. In the remote samples, the particles were primarily medium-sized (median = 67 μm), lightweight polyester fragments. This study also investigated the influence of soil parameters such as water content, bulk density, and particle size on MP retention in remote soils. However, no other significant correlations were found when relating the count and mass to soil parameters.

1. Introduction

Plastic, including microplastic (MP), is present in all environmental matrices (Barnes et al., 2009), from being trapped in marine sediments (Abel et al., 2021; Simon-Sánchez et al., 2022; Woodall et al., 2014) to being present in a diverse range of terrestrial environments, including

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agricultural (Klemmensen et al., 2024; Zubris and Richards, 2005) and urban soils (Leitão et al., 2023). Research concerning MP in the atmosphere is still limited. However, awareness of MP presence in the atmosphere and its potential environmental and human health risks has increased in recent years. This includes research on the possibility of MP reaching human lungs (Amato-Lourenço et al., 2021; Gasperi et al., 2018; Prata, 2018) and possible accumulation in terrestrial and aquatic environments (Mbachu et al., 2020; Prata, 2018).

Despite the limited literature, some studies have been published investigating both urban (Amato-Lourenço et al., 2021; Dris et al., 2015; Dris and Gasperi, 2016; Liu et al., 2019c; Wright et al., 2020) and atmospheres in remote areas (Allen et al., 2019; Brahney et al., 2020). All MP present in the air results from anthropogenic activities (Mbachu et al., 2020), such as the release of fibres from synthetic textiles (Liu et al., 2019c), tyre and traffic emission (Evangeliou et al., 2020; Kole et al., 2017), and agricultural activities such as mulch films (Mbachu et al., 2020). Once released into the atmosphere, MP might be transferred over long distances, potentially reaching and accumulating in even the world's most remote regions (Allen et al., 2019; Evangeliou et al., 2020; Wright et al., 2020).

Many published studies on MP in the atmosphere focus on deposition rates but do not investigate whether MP accumulates in the soil after being deposited in remote areas. Some studies address MP in soils sampled in remote areas, with one of the most extensively studied regions being the Tibetan Plateau (Feng et al., 2023; Liu et al., 2023; Li et al., 2024; Yang et al., 2022). These studies report varying concentrations of MP. For example, Liu et al. (2023) reported concentrations ranging from 380 to 13,330 MP kg^{-1} dry weight, while Feng et al. (2023) reported an average concentration of 49.2 MP kg^{-1} and 61.7 MP kg^{-1} in bare land and grassland, respectively. In addition to studies on the Tibetan Plateau, MP concentrations have also been investigated in more remote areas of France, with reported concentrations of 34 ± 116 MP kg^{-1} in forest soil and 640 ± 580 MP kg^{-1} in grassland soil (Palazot et al., 2024).

It is evident that the current knowledge of MP in remote soils is very limited, with most studies focusing on agricultural soils. For a comprehensive understanding of MP's environmental fate in remote soils, it is essential to determine how much MP accumulates in remote soils, as

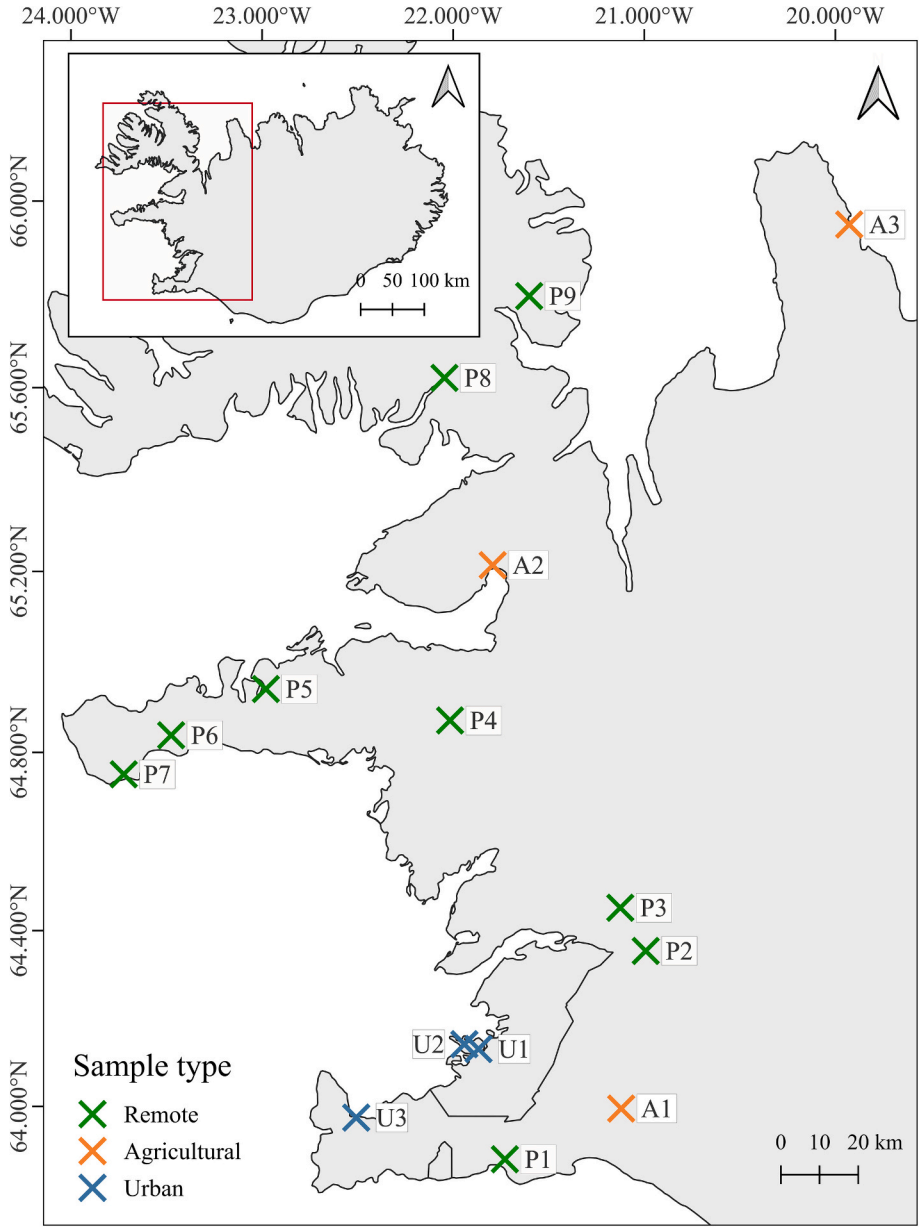

Fig. 1. Locations of the 15 samples taken in Iceland with the indication of whether they were taken in remote (green), agricultural (orange), or urban areas (blue).

well as the size and type of these particles, to fully understand if long-range transport and deposition of MP may represent an environmental concern. Furthermore, to better understand the factors possibly influencing the accumulation, there is a need to investigate whether soil parameters such as water content, dry bulk density, and soil particle size affect accumulation in these remote soils.

The primary aim of this study was to investigate the abundance of MP in remote soils and assess the extent to which MP fallout accumulates in these environments. This was achieved by using agricultural and urban soils as reference points to see the general contamination level across the western part of Iceland. State-of-the-art extraction and analytical methods were employed. While it is assumed that some MP will accumulate in remote soil, the degree of accumulation remains uncertain. Therefore, this study aimed to identify MP types, sizes, and shapes that persist in these soils. Additionally, various soil parameters, including water content, bulk density, and soil particle size, were examined to determine their potential influence on MP retention in soil. It was hypothesized that these parameters may affect the ability of MP to remain in the soil over time, especially in the more remote areas where anthropogenic activity is less pronounced.

2. Method and materials

2.1. Sampling area and sampling

Fifteen samples were collected in the western part of Iceland (Fig. 1) - nine from remote areas, three from agricultural areas, and three from urban areas in Reykjavík. The samples were collected in early October 2021. To ensure the pristineness of the remote samples, they were gathered at locations as distant as feasible from anthropogenic sources. The agricultural fields were either used for producing hay or for grazing livestock, here mainly sheep. Pictures of all sampling locations are given in Fig. S1 in Supplementary Materials.

The samples were taken with a headwind to minimize possible contamination from the clothing worn during sampling. Approximately 1 kg of soil was collected for each sample with a metal shovel to a depth of approximately 20 cm. Before sampling, vegetation, large gravel, and stones were cleared from the area. The soil was transferred into pre-washed aluminium bottles using a metal funnel; the bottle openings were sealed with aluminium foil before securing the lids to prevent contamination. Upon arrival at the lab, the samples were stored at 5 °C until further processing.

Adjacent to each sampling site, an quasi-undisturbed soil sample was collected to assess various soil parameters. The intact samples were taken with a 100 cm^3 sampling ring (d = 5 cm, h = 5.1 cm). After removing the top layer of roots, larger gravel, and stones, the ring was inserted into the soil as vertically as possible, ensuring minimal disturbance. Subsequently, the ring with the soil was carefully extracted, and the lids put on. Upon arrival at the lab, the samples were stored at 5 °C until further processing.

2.2. Sample preparation for MP extraction

MP was extracted from the collected soil using a protocol described by Klemmensen et al. (2024) and Molazadeh et al. (2023), with slight changes. The extraction comprised six main steps: pre-oxidation, wet-sieving, drying, density separation, chemical treatment, and a second density separation (Fig. S2).

During pre-oxidation, on average, 266 g of dry soil (Table S1) was added to a 5 L beaker containing filtered milli-Q water (GF, 0.7 μm). Hydrogen peroxide (H_2O_2) was gradually added over 7 days while continuously aerating the suspension using a special designed coil, to ensure mixing of the sample. It was ensured that the H_2O_2 concentration in the sample never exceeded 10 %. After seven days, the samples were wet-sieved through a 1 mm sieve and left to settle for at least 48 h. The supernatant was filtered off using a 10 μm stainless steel filter. The remaining soil and soil residue on the filter were combined and frozen for 48 h before freeze-drying for another 48 h.

The dried soil was transferred to a 2 L separation funnel, to which a $ZnCl_2$ solution ($\rho = 1.75$ g cm^{-3}) was added. The mixture was aerated with filtered compressed air for 30 min, sides flushed and left to settle overnight. Subsequently, the sediment was drained out of the funnel while the top three centimetres of liquid were retained for further treatment. The sediment was returned to the separation funnel, and the procedure was repeated. The collected liquid was filtered using the same 10 μm stainless-steel filter. Following this, the particles on the filter underwent multiple chemical treatments, including SDS, enzymatic treatment, and a Fenton oxidation (the specifics of each treatment can be found in Fig. S2 Supplementary Material).

After the chemical treatment, the samples were sieved through a 500 μm sieve and transferred to a $ZnCl_2$ solution ($\rho = 1.75$ g cm^{-3}) for a second density separation, following the same procedure as described earlier, albeit using a smaller funnel (either 100 or 250 mL) and without reintroducing the sediment for a second separation. Following the density separation, the particles were transferred to 50 % ethanol, and the sample was evaporated into a 10 mL vial using an evaporation bath (TurboVap® LV, Biotage) at 50 °C. Finally, the volume was adjusted to either 5 or 8 mL with 50 % HPLC grade ethanol, depending on the cleanliness of the sample (Table S1).

2.3. MP identification and quantification

During the extraction protocol, larger particles were first removed using a 1 mm sieve, followed by a 500 μm sieve. The two fractions were combined, and all particles larger than 500 μm were cleaned and manually sorted to identify potential MP. Suspected MP was then measured using ATR-FTIR (Agilent Cary 630 FTIR with a diamond ATR) and imaged using a stereomicroscope (ZEISS, SteREO Discovery.V8, OberKochen Germany).

The MP <500 μm was identified and quantified following the approach described in Klemmensen et al. (2024). The sample extracts were sonicated, and subsamples were deposited utilizing 100 μL capillary glass tubes onto a zinc selenide (ZnSe) window (Ø13 × 2 mm, Chrystan, UK) held in a compression cell (Pike technologies, Ø10). Three windows were prepared per sample to ensure an adequate sample fraction was scanned (Table S1).

The deposited samples were scanned following a procedure similar to that described by Liu et al. (2019a), Liu et al. (2019b), and Rasmussen et al. (2021). Each window was scanned using μ-FTIR imaging at a pixel resolution of 5.5 μm in transmission mode (Agilent Technologies Cary 670 FTIR spectrometer, combined with a Cary 620 FTIR microscope, 15× Cassegrain objective, and 128 × 128 pixel Focal Plane Mercury-Cadmium-Telluride detector). The wavenumber range was set to 850–3750 cm^{-1}, and the spectral resolution to 8 cm^{-1}. For the samples, 30 scans were co-added, while 120 scans were co-added for the background.

Data were processed with the freeware siMPle (Primpke et al., 2020b), employing a reference library comprising 512 reference spectra from 79 material groups, including plastics and natural materials. siMPle provides information on MP counts, size, shape, type, and estimated mass. The latter is calculated based on the method outlined by Simon et al. (2018). After scanning and processing each window, the data from the three windows for each sample were combined to create a single comprehensive dataset, which was then extrapolated to represent the entire sample volume.

2.4. Sample preparation soil characteristics

Each soil sample collected with the rings was transferred to an aluminium tray of known weight and weighed (Table S2). The soil was dried at 105 °C for approximately 48 h and weighed again. The water content and dry bulk density were determined based on the obtained

weights.

The particle size was determined by placing a known amount of dry soil in a sieving tower (Table S2) consisting of four sieves (63, 125, 250, and 2000 μm). Larger plant roots were removed during this process. The sieving tower was placed on an automatic sieve shaker for 20 min at an amplitude of 0.7 mm. Afterwards, the soil on the sieves was carefully transferred to aluminium trays and the fractions were weighed.

2.5. Contamination and recovery

Three field blanks were collected during sampling, one for each soil environment. This was done by placing an open pre-muffled petri dish next to the sampling spot. Upon returning to the lab, the petri dishes were flushed with 50 % HPLC grade ethanol, and the samples were concentrated into 10 mL vials. The volume was adjusted to 5 or 8 mL with 50 % HPLC grade ethanol (Table S1), and the samples were scanned as described in section 2.3.

The analytic method, analytical instrument, and techniques were identical to those described in Klemmensen et al. (2024). In addition, the same people conducted the analysis simultaneously. Hence, that study's procedural blanks and recovery study are also valid for the current study.

2.6. Statistical analysis

The statistical analysis was conducted using R (version 4.2.1). Shapiro-Wilk's test was employed to test normality, and Kendall's rank correlation coefficient was utilized to evaluate the correlation between counts and mass and the various soil parameters. A non-parametric approach was chosen due to non-normally distributed data. All statistical analyses were conducted at a confidence level of 95 %. The corresponding *p*-values are provided in supplementary material.

3. Results and discussion

3.1. Contamination and recovery

A sub-sample of 1 mL out of 5 mL concentrate was scanned for each of the field blanks covering the three soil environments, detecting only one PP particle in the agricultural field blank. The contamination during sampling was consequently low (below the detection limit).

Applying a similar analytical approach, Klemmensen et al. (2024) reported an average of 107 (±12) counts per procedural blank, corresponding to 8.3 (±4.0) μg per blank. Applying the procedural blank results from that study and considering that 266 g of soil was, on average, processed in the current study (Table S1), the calculated contamination in the samples was approximately 402 counts kg^{-1}, corresponding to 31.2 μg kg^{-1}. Klemmensen et al. (2024) also noted that 78 (±5)% of the total polymer count consisted of polyester, likely originating from polyester fibres being released into the air from clothing. Blank values were not subtracted from the results due to difficulty determining which specific particles to remove from the dataset (Dawson et al., 2023; Shruti and Kutralam-Muniasamy, 2023). It is, hence, essential to acknowledge that some of the MP found in the soil may result from contamination during analysis.

Klemmensen et al. (2024) also evaluated the MP recovery rate during the sample processing. They found a recovery of 48 (±8)% and 53 (±4)% for PS beads with a diameter of 52 μm and 106 μm, respectively, using clean sand as the matrix. The recovery rate was not factored into the results, meaning that the actual MP concentration might be slightly higher when accounting for losses during the analysis. The concentrations where not adjusted based on the recovery rate due to uncertainties associated with the limited number of polymers tested and differences in matrix composition. These uncertainties make it impossible to accurately determine the type, size, and shape of the polymers lost, and therefore, adjusting the concentration appropriately was not feasible. This aligns with HELCOM (2022), which advises against correcting water samples for recovery rates.

3.2. MP abundance in remote soils

MP was detected in all 15 analyzed samples (Fig. 2, Table 1). With the exception of one sample (P7), all MP counts exceeded the procedural blank value of 402 counts kg^{-1}. Consequently, it is uncertain whether the measured MP count in this remote sample originated from the sample or resulted from contamination during sample processing. Therefore, MP counts were detectable in 14 out of the 15 samples. No MP was found in the fraction above 500 μm, meaning that all reported MPs were found in the fraction below 500 μm.

When considering the mass of MP (Fig. 2), it becomes evident that among the remote samples, only five (P2, P4, P6, P8, and P9) were above the contamination level limit discussed in section 3.1, while one (P3) was at the limit. The remaining two remote samples (P1 and P7) were below the expected contamination level of 31.2 μg kg^{-1}. Hence, based on mass, MP presence was confirmed in 12 out of 15 samples. It is crucial to note that MP mass is highly dependent on the size of the particles found. Therefore, it was likely that one or more larger MPs were present in the procedural blank. This was evident in one of the blanks, where a larger PP particle contributed approximately 41 % of the total mass in the blank. This means that samples lacking large, heavy particles are likely to fall below the contamination level when considering mass. This is exemplified by sample P1, which exceeds the contamination level in terms of count with a factor of 2.5 but falls below when considering the mass.

The lowest average concentration was found across the nine remote samples with 857 (±561) counts kg^{-1}, where the highest concentration was 2190 counts kg^{-1} and the lowest concentration was approximately six times lower. The average concentration in terms of estimated mass was 64.37 (±47.96) μg kg^{-1}. Sample P6 held a relatively high concentration compared to the others, namely 176.87 μg kg^{-1} (Fig. 2), due to a single large, and hence heavy, PP particle. Had this particle been removed from the dataset as an outlier, the mass concentration would have decreased to 24.84 μg kg^{-1}, i.e., within the range of the other samples. For all subsequent data processing, this specific particle was included since no indication was found to exclude it.

Limited literature exists on MP contamination in remote terrestrial environments, with most studies focusing on the daily deposition rates of MP potentially reaching the soil without verifying how much remains there. Yang et al. (2022) investigated MP concentrations in remote soil across the Tibetan Plateau, reporting concentrations ranging from 5 to 340 counts kg^{-1} dry weight, with an average of 47.12 counts kg^{-1} dry weight, with the highest concentrations found near cities (Yang et al., 2022). Similarly, Wang et al. (2021) and Abbasi et al. (2021) studied MP in remote desert environments, with Wang et al. (2021) reporting concentrations as low as 0.9 (±1.6) counts kg^{-1} in remote non-touristic areas, while Abbasi et al. (2021) reported an average concentration of 20 counts kg^{-1}. Although generally lower concentrations than those observed in this study, their concentration falls within the range of some of the current study's samples, especially when considering background contamination.

Comparing quantitative data across studies is challenging due to variations in extraction methods and analytic techniques. Many studies, including Wang et al. (2021) and Abbasi et al. (2021), rely on manual and visual sorting and detection of MP. This technique often overlooks smaller particles (van den Berg et al., 2020; Primpke et al., 2020a), leading to underestimation compared to more automated methods such as μ-FTIR imaging and Raman spectrometry, which typically have lower detection limits (Primpke et al., 2020a). Differences in extraction methods, including the solution used for density separation, can affect the types of polymers extracted, further complicating the comparison between studies.

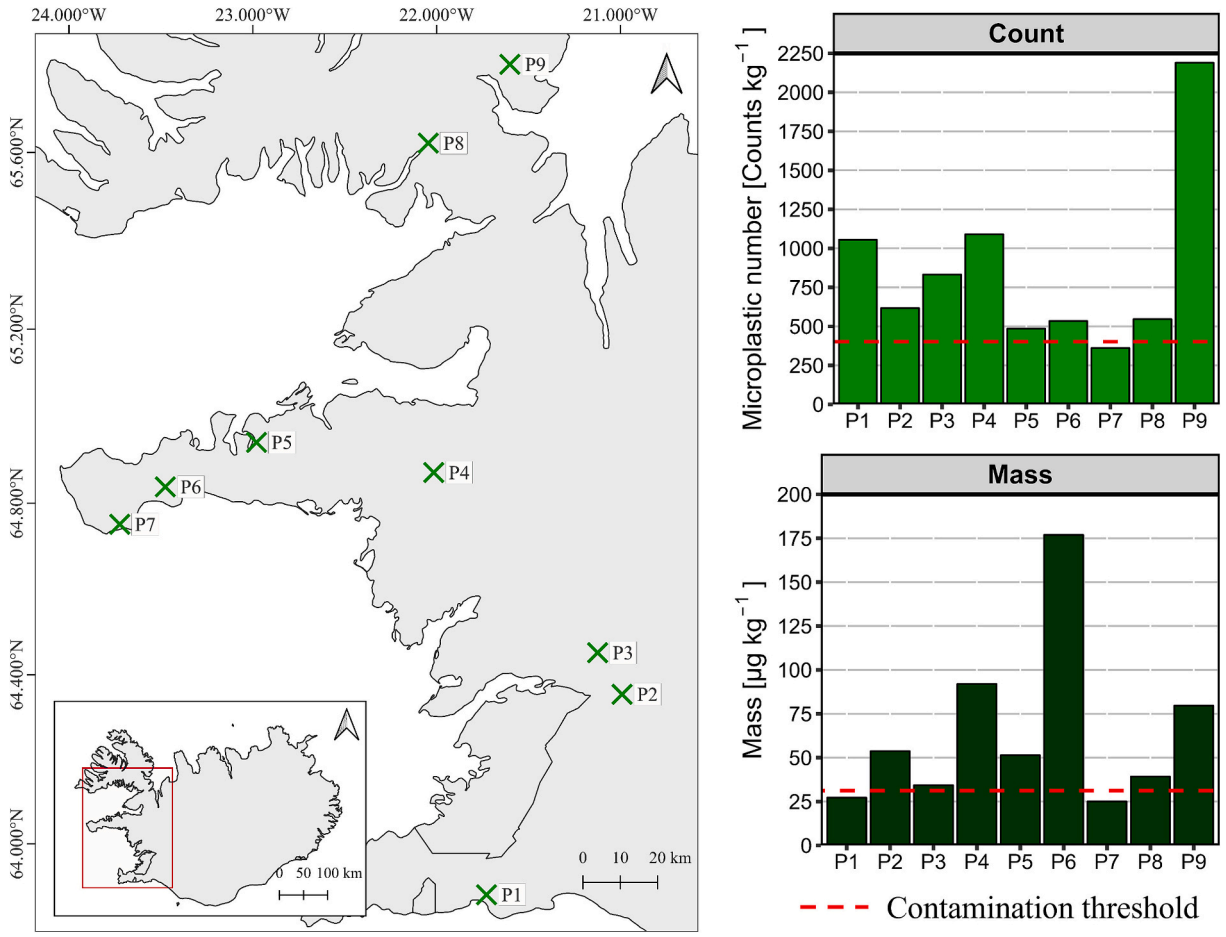

Fig. 2. The MP concentration in terms of particle count and mass for the nine remote sample locations, accompanied by a map showing the locations. The red dotted line represents the contamination threshold for particle count, and mass based on the procedural blanks discussed in section 3.1.

Table 1
MP count, mass, average, and standard deviation for the three agricultural and three urban samples, along with the average mass and count concentration for the remote samples.

		MP count [count kg^{-1}]	MP mass [µg kg^{-1}]
Remote	Average	857 (±561)	64 (±48)
Agricultural	A1	1151	602
	A2	1231	68
	A3	5348	760
	Average	2563 (±2412)	476 (±363)
Urban	U1	54,774	3525
	U2	17,422	2241
	U3	6422	758
	Average	26,206 (±25,345)	2175 (±1385)

3.3. Comparison to other soil environments

Comparing the remote soil data with samples from agricultural and urban areas (Table 1), reveals significant differences in MP levels. On average, the MP count for agricultural samples is approximately three times higher than in remote areas. The urban samples exhibit an even higher average count, approximately 30 times greater than the remote samples. This trend is even more pronounced when considering MP mass, with agricultural samples averaging seven times higher mass and urban samples approximately 34 times higher. These findings suggest a correlation between anthropogenic activity and MP contamination levels, which is also suggested in the literature (Chen et al., 2022; Yang et al., 2022). Moreover, the lower MP concentration observed in remote areas compared to regions with higher human activity supports the assumption that MP contamination in remote areas may primarily result from atmospheric deposition.

Analysis of polymer types across the three soil types (Table S6, S7) revealed variation in distribution, likely influenced by a relatively low particle detection rate, with only 31 (±9) counts detected in each remote sample. Consequently, a few particles of a specific type in a single sample can significantly impact the overall composition. Combining all particles in a given soil environment, differences in the most abundant polymer were observed: polyester dominated in remote samples (57 %), PP in agricultural samples (43 %), and acrylic in urban samples (28 %). The differences are most likely due to land use and the corresponding sources of MP that are transferred to the soil. For remote soils, it is suspected that all MP results from atmospheric deposition. Multiple studies have reported that polyester is a common polymer found in outdoor air (Amato-Lourenço et al., 2022; Wright et al., 2020), making polyester a candidate for deposition in these remote soils. In contrast, the prevalence of PP in agricultural samples is likely due to its common use in agriculture, which increases the potential for contamination. Urban samples show a more diverse composition of polymers likely due to a higher level of anthropogenic activity. This includes sources such as car tyre dust (Iordachescu et al., 2024; Rasmussen et al., 2023), litter, and the release of fibres from clothing (De Falco et al., 2020; Vianello et al., 2019), resulting in a more variation in the polymer composition compared to the remote samples.

The significance of proximity to larger cities and anthropogenic activity is also apparent when taking urban and remote samples. In urban areas, sample U1 exhibited the highest concentration and was collected

from a heavily populated and trafficked parking lot in the centre of Reykjavík. Sample U2 was collected from a park in downtown Reykjavík, while sample U3 was collected from a parking lot approximately 44 km outside Reykjavík in the smaller town of Njarðvík near the airport.

For remote samples, the distance from larger cities, which in this study was limited to Reykjavík, seems to be a factor that might influence MP concentration (Yang et al., 2022). Fig. 2 shows that samples with the lowest concentrations (P5, P6, P7, P8) were located farther from Reykjavík than those with medium MP concentrations (P1, P2, P3, P4). Interestingly, the sample with the highest measured concentration (P9) was also the furthest from Reykjavík. There are no clear environmental factors to explain why this samples had such a high concentration, except that, as shown in Fig. 3, the water content was higher compared to the other samples located further from Reykjavík. It is important to note, however, that this is purely speculative, since many environmental factors, such as wind conditions (which were not addressed in this paper), contribute to MP contamination. Nonetheless, the general trend suggests that MP content might be influenced by proximity to major cities, with urban areas experiencing a more pronounced effect due to higher anthropogenic activity. As the remote sample with the highest concentration was located the furthest from Reykjavík, other environmental factors may also affect the MP content of the soil in these remote areas, where the primary contamination source is atmospheric deposition.

3.4. Soil parameters' effect on MP concentration

Considerable variation was seen in the MP count and mass among the samples, even from samples within the same environment, as indicated by the large standard deviation. Besides natural variability, factors such as distance to the nearest major city and anthropogenic activity have been shown to influence MP concentration. Another factor that may influence the MP concentration in soil is soil characteristics, such as dry bulk density, water content, and silt and clay content (detailed in Table S3). It is assumed that these factors, especially in remote areas with less anthropogenic activity, might affect MP retention in soil. Upon entering the soil, MP is expected to interact with natural colloids, such as clay, which could affect their retention.

An investigation into the relationship between MP count and mass and soil parameters was conducted for the remote samples. Only the remote samples were chosen due to the small sample size for both the urban and agricultural samples, and they were not included in the dataset of the remote samples due to differences in anthropogenic activity across environmental soil areas. Fig. 3 shows that the relationship between the three soil factors and the MP count appears random. This observation was further confirmed by a Kendall correlation analysis, which no significant correlation observed between MP count and the soil parameters, as all showed *p*-values above 0.05.

Similarly, the effect of soil parameters on MP mass was analyzed (Fig. S3), along with verification through a Kendall correlation analysis (Table S5). However, no clear patterns or correlations were observed. This lack of correlation may be attributed to the mass estimation being primarily influenced by the size and type of polymer present in the samples. This factor can cause considerable variation, especially with smaller datasets.

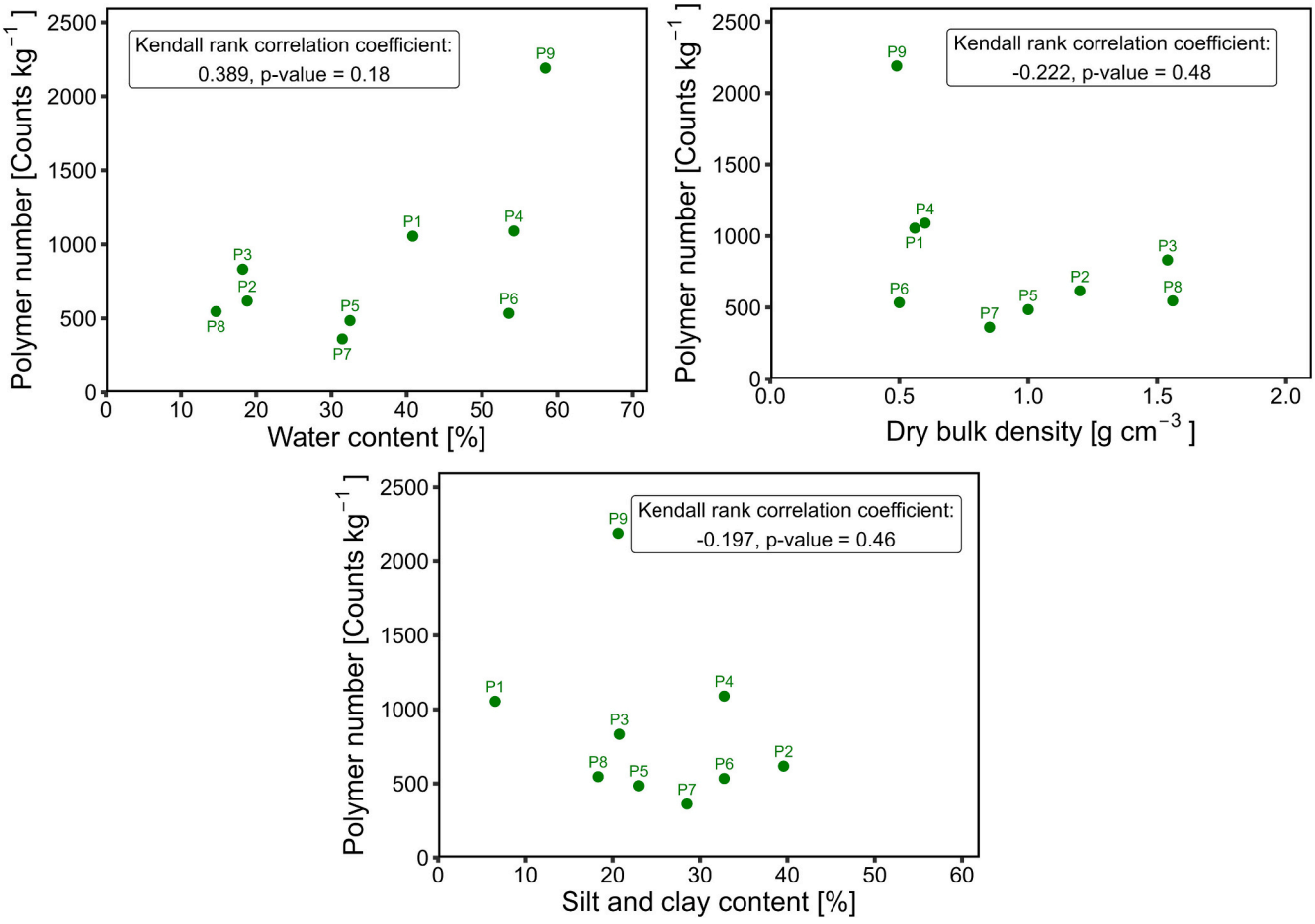

Fig. 3. Graphs illustrating the polymer number in relation to three distinct soil parameters: water content, dry bulk density, and silt and clay content. Each data point is labelled with the corresponding sample location.

3.5. Accumulation of MP in remote soils

When examining MP contamination in remote locations, most studies focus on atmospheric deposition rates (Abbasi and Turner, 2021; Allen et al., 2019; Brahney et al., 2020; Welsh et al., 2022), with limited data on actual concentrations in remote soil. Literature reports average deposition rates ranging from 12 (±5) counts m^{-2} day^{-1} (Abbasi and Turner, 2021) to 365 (±69) counts m^{-2} day^{-1} (Allen et al., 2019). Consequently, if all MP potentially deposited on the surface were to accumulate, the concentrations observed in this study would be expected to be higher than the numbers observed.

Literature investigating deposition rates reveals variability over different months (Abbasi and Turner, 2021; Allen et al., 2019) and dependence on weather conditions such as rain (Abbasi and Turner, 2021; Brahney et al., 2020). This variability suggests that MP concentration in a location might not be constant due to the varying deposition rates, as well as factors like wind (Rezaei et al., 2019), soil erosion (Rehm et al., 2021), and vertical movement of MP to deeper soil layers (Qiu et al., 2023) influencing MP concentration and distribution. However, more research is needed to investigate the fate of MP once deposited in remote soils.

3.6. Size and shape of MP in remote areas

As mentioned earlier, the MP counts were above the contaminant threshold in 8 out of the 9 remote samples, indicating that some MP accumulation occurred even in soil in remote places. However, limited literature is available on the MP types, shapes, and sizes present in remote soil.

An important consideration when investigating MP deposition and possible accumulation in remote soil is the size distribution of polymers, which was studied for the remote samples, using agricultural and urban samples as reference points. The size distributions, based on both count and mass, for the three soil environments are illustrated in Fig. 4. It can be seen that, in terms of count, 50 % of the identified particles measured below 67 μm for the remote samples, but these particles only contribute to around 7 % of the total mass. Interestingly, the urban samples contained more smaller, lightweight particles than the remote samples, with a median size of only 60.9 μm. Although the medium-sized fraction (50–200 μm) appears to have similar percentages of particles across both sample types, the remote samples seem to contain more medium-sized (50–200 μm), lightweight polymers that collectively contribute minimally to the total mass. Specifically, only 36 % of the total mass in the remote samples fell within this fraction, compared to 48 % for the urban samples.

It is also worth noting that a few particles with a major dimension above 500 μm were found. This was because these particles had a minor dimension below 500 μm, allowing them to pass through the sieve. As shown in Fig. 4, the remote samples contain more large MPs (>200 μm) compared to the urban samples. This was due to the presence of larger fibres in the remote samples, with 91 % of the MP being fibres, compared to approximately 69 % in the same fraction for the urban samples.

The detection of medium-sized (50–200 μm) MP in remote soil may be attributed to the behaviour of particles in the air. Here, small and light particles may be more likely to remain airborne due to lower settling velocities. In contrast, larger, heavier non-fibre particles may not be picked up and released into the air in the first place, thereby not reaching remote areas like the highlands of Iceland.

A total of 29 % of the detected polymers in the remote samples were identified as fibres based on the principle that the ratio between the major and minor dimensions exceeds three (Cole, 2016; Vianello et al., 2019). Among these fibres, 25 % were acrylic and polyester, commonly associated with clothing. Previous studies (Abbasi and Turner, 2021; Brahney et al., 2020; Welsh et al., 2022) have reported higher fibre percentages than this study. This disparity may be attributed to variations in sampling methodologies, as atmospheric deposition is often measured using passive sampling methods that capture MP in containers. Therefore, MP fibres might easily be re-dispersed or stay in the air. This phenomenon may be explained by the lower settling velocities of fibres compared to spherical particles (Saxby et al., 2018), enabling

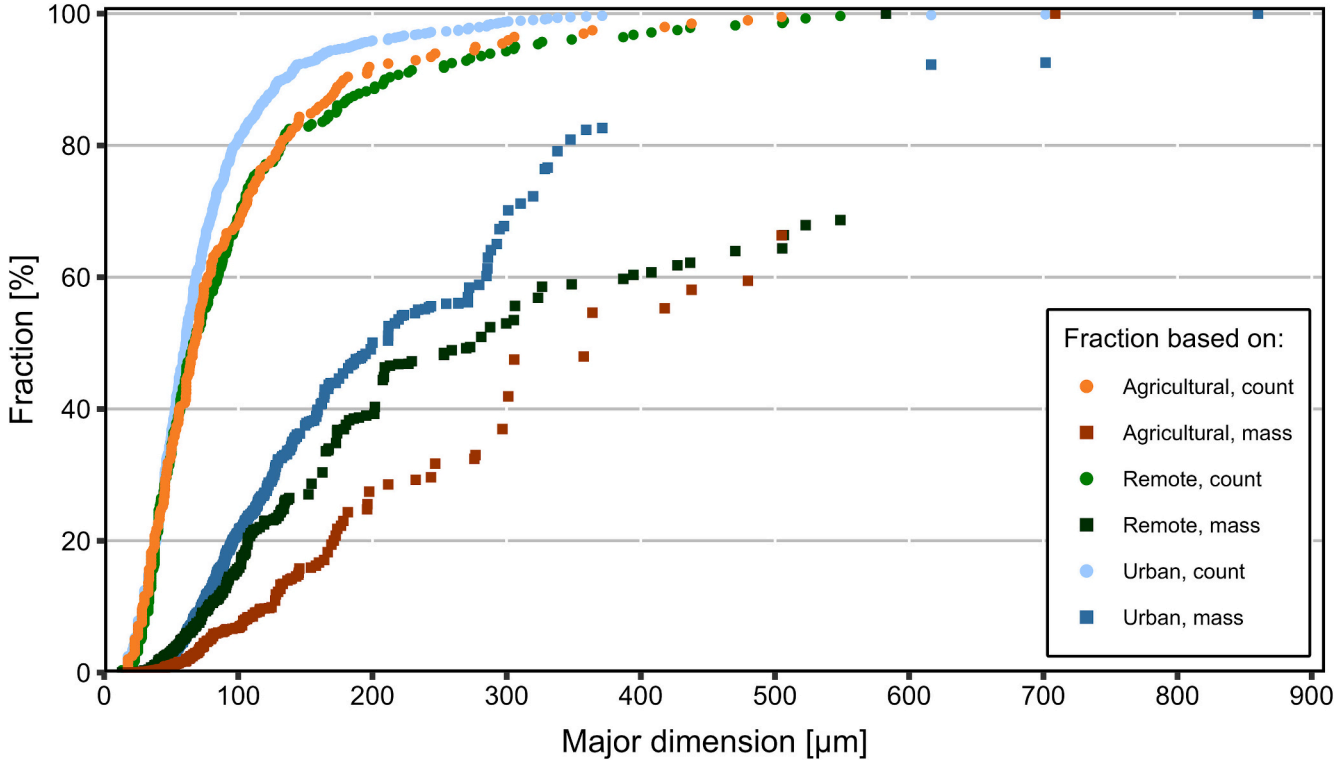

Fig. 4. Size distribution based on the count (round) and mass (square) for both agricultural (orange), remote (green), and urban samples (blue).

them to travel further and be more easily re-dispersed into the air. Consequently, it appears that fragments are more likely to accumulate in the soil compared to fibres.

This suggests that only a small fraction of the potential MP in the atmosphere actually accumulates in the remote soils. However, this small portion primarily comprises common lightweight polymers such as polyester. Furthermore, the majority of the detected particles in the remote samples were medium-sized fragments (50–200 μm) or larger fibres.

4. Conclusion

MP was detected in all 15 samples, with average counts of 857 (±561), 2563 (±2412), and 26,206 (±25,345) counts kg^{-1} for remote, agricultural, and urban samples, respectively. Of these, one sample held MP counts below the procedural contamination.

When comparing the three different soil environments, it is evident that the MP contamination increased with increasing anthropogenic activity, a trend observed in both the remote and urban datasets. In addition to anthropogenic activity, it appeared that soil parameters such as water content, dry bulk density, and silt and clay content did not seem to influence MP levels.

Moreover, when comparing the concentrations found in this study to deposition rates reported in the literature, the observed concentrations suggest that not all MP were to accumulate permanently in the soil. This indicate that some MP may be subjected to removal processes, such as soil erosion and migration to deeper soil layers after deposition. However, as the soil samples held more MP than the procedural blanks, some MP accumulated in the soil. The MP most likely to accumulate was medium-sized (50–200 μm), relatively lightweight fragments of commonly found polymers such as polyester.

CRediT authorship contribution statement

Nanna D.R. Klemmensen: Writing – original draft, Visualization, Project administration, Data curation, Conceptualization. **María Sobrino Blanco:** Writing – review & editing, Methodology, Data curation, Conceptualization. **Jes Vollertsen:** Writing – review & editing, Resources, Funding acquisition, 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.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2024.177215.

Data availability

Data will be made available upon request.

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