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Composition and concentrations of microplastics including tyre wear particles in stormwater retention pond sediments

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ABSTRACT

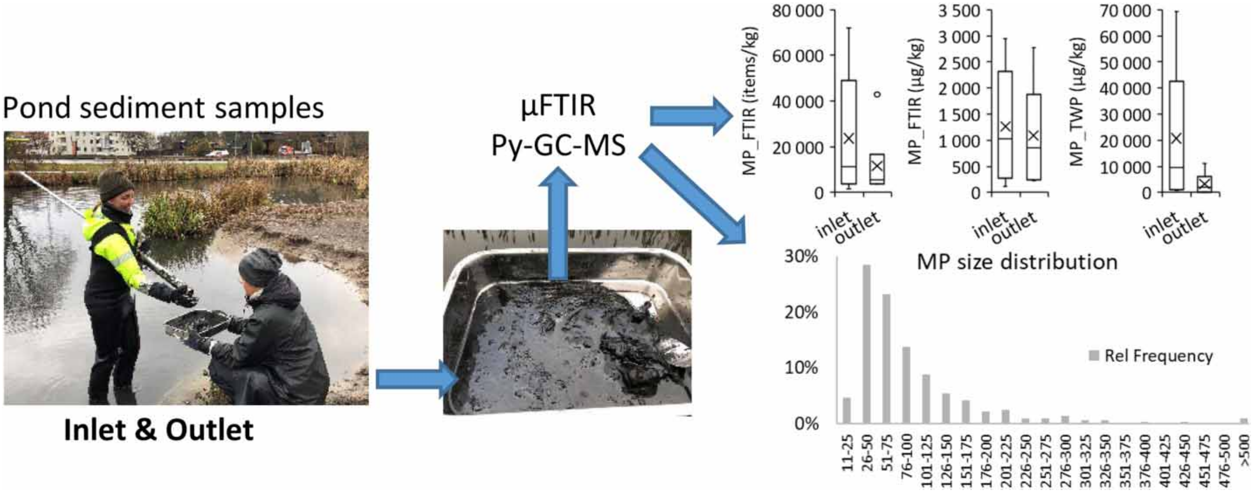
Stormwater is recognised as a vector for microplastics (MPs), including tyre wear particles (TWPs) from land-based sources to receiving waterbodies. Before reaching the waterbodies, the stormwater may be treated. In this study, sediments from six treatment facilities (five retention ponds and a subsurface sedimentation tank) were analysed to understand MP occurrence, concentrations, sizes, polymer types and distribution between inlet and outlet. The concentrations of MPs showed large variations between and within different facilities with MP concentrations of 1,440–72,209 items/kg (analysed by μFTIR) corresponding to 120–2,950 μg/kg and TWP concentrations from <DL up to 69,300 μg/kg (analysed by pyrolysis–GC–MS), with significantly higher concentrations at the inlet compared to the outlet. Polypropylene (PP) was the predominant MP type in terms of number in all samples. TWPs were dominant by mass in most (nine) samples. The relatively low density of PP polymers implies that density might not be the sole factor influencing particle settlement behaviour. Small particles occurred more frequently than large ones; around 70% of the particles detected in the samples were 100 μm or smaller. In summary, this study highlights the occurrence of MPs, including TWPs, in stormwater facilities and demonstrates variations in concentrations depending on sites and locations within the facility.

Key words: FTIR imaging, MP, pyrolysis–GC–MS, stormwater management, TWP, urban runoff

HIGHLIGHTS

- MPs including TWPs in the size range 20–450 μm were extracted from pond sediments.
- Pond sediments analysed by μ-FTIR held 1,440–72,209 MP items/kg and 120–2950 μg/kg.
- TWPs were analysed by Py–GC–MS with concentrations ranging up to 69,300 μg/kg.
- Concentrations of MPs showed high variability between stormwater pond sediments.
- Polypropylene (PP) and TWPs were the most common MP types by number and mass.

GRAPHICAL ABSTRACT



1. INTRODUCTION

The subject of microplastics (MPs) in urban stormwater systems has gained significant attention in recent years. This focus is due to their ubiquity and potential negative impacts, posing a multifaceted challenge across environmental protection, public health and regulatory domains. MPs are commonly defined as small pieces of plastic materials smaller than 5,000 µm as was suggested by Arthur *et al.* (2009). The term 'microplastics' in this study includes tyre wear particles (TWPs) following the definition by Hartmann *et al.* (2019). The effects of MPs in urban environments are far-reaching and complex. These small plastic particles do not merely act as particulate pollutants by themselves, as they may also contain hazardous substances, such as additives, thus also acting as vectors transporting persistent pollutants (Li *et al.* 2018). Additionally, MPs can also transport pollutants absorbed to their surfaces (Chen *et al.* 2024). MPs are most abundant in urban areas, e.g. an increased population density also increases litter including plastics as well as traffic and release of related MPs (Österlund *et al.* 2023). Moreover, MPs may negatively affect terrestrial and aquatic environments and humans (GESAMP 2015; Li *et al.* 2018).

Stormwater is recognised as a pathway for MPs from land-based sources to receiving water bodies (Werbowski *et al.* 2021; Österlund *et al.* 2023). MPs originate from various sources, including the fragmentation of larger plastic items, e.g. litter, shedding from synthetic textiles, runoff from traffic and atmospheric fallout (Dris *et al.* 2016; Liu *et al.* 2019a; Järlskog *et al.* 2020; Lange *et al.* 2022; Öborn *et al.* 2022). Among these traffic-related pollution, TWPs have been recognised as a significant contributor to the release of MP particles into the environment (Järlskog *et al.* 2020; Goßmann *et al.* 2021; Rødland *et al.* 2022). Airborne TWPs are ubiquitous in urban areas, e.g. as illustrated in model simulations of TWP-PM10 in the air across Stockholm (Svensson *et al.* 2023) and analysis of MPs including TWPs in spider webs from urban areas (Goßmann *et al.* 2022). Once in the urban environment, the MPs can accumulate in various locations, including streets, parks and water bodies.

In many urban areas, stormwater is inadequately treated or sometimes not treated at all before being discharged into receiving water bodies. Currently, MP concentrations in stormwater are not regulated (Stang *et al.* 2022). In many newly developed areas, it is common to have specific requirements for sustainable stormwater management to minimise the adverse impacts of stormwater on receiving waters. Common stormwater treatment systems, such as biofilters, swales, detention basins, wetlands and ponds, are not primarily designed to separate MPs but are used for flow control and retention of other types of pollutants, primarily particles and particle-bound pollutants (Österlund *et al.* 2023). Nevertheless, there are indications that some of the facility types also remove MPs to a fairly large extent (Stang *et al.* 2022; García-Haba *et al.* 2023; Rasmussen *et al.* 2024). In a literature review on MP removal from urban stormwater, Stang *et al.* (2022) suggest that retention ponds may potentially remove MPs in the range of 85–99%. However, these numbers were considered highly uncertain due to the limited number of studies conducted and included in the literature review. Also, stormwater treatment in sedimentation facilities, such as ponds, generates sediment that needs to be treated, especially concerning their pollutant content.

The most common types of MPs in the environment are TWPs and polyethylene (PE), polypropylene (PP), polystyrene (PS), polyvinyl chloride (PVC), polyamide (PA) and polyethylene terephthalate (PET) (GESAMP 2015; Plastics Europe 2019;

Järlskog *et al.* 2021). MPs composed of PP and PE have been categorised as buoyant MPs and are more likely to float considering their densities of around 0.910 and 0.935 g/cm^3, respectively; MPs composed of PS, PVC, PA and PET categorised as non-buoyant with densities ranging from 1.05 to 1.38 g/cm^3, and thus, are more likely to sink (Simon *et al.* 2018 and Molazadeh *et al.* 2023a). TWPs are among the denser MPs, they are also known to incorporate mineral particles further increasing their density. Jung & Choi (2022) found the densities of tyre and road wear particles were in the range of 1.20–1.70 g/cm^3. However, in previous research, PP and PE were commonly found in stormwater sediments, indicating that the removal of MPs by sedimentation also depends on other factors such as size, shape, composition and biofilm growth (Liu *et al.* 2019a; Lutz *et al.* 2021; Stang *et al.* 2022; Mendrik *et al.* 2023; Molazadeh *et al.* 2023a). There are still challenges, and more information about MPs accumulated in common stormwater treatment facilities such as ponds is needed (Stang *et al.* 2022; Wang *et al.* 2022; Österlund *et al.* 2023). This study aims to advance our understanding of the fate of these emerging pollutants in urban stormwater facilities by assessing the presence of MPs, including TWPs in the bottom sediment near the inlet and outlet of five stormwater ponds and a subsurface sedimentation tank. Specifically, we determine MP and TWP concentrations, MPs' polymer composition and size distribution.

2. MATERIALS AND METHODS

2.1. Study sites

Sediments were collected from six stormwater management facilities, five ponds and one subsurface sedimentation tank, previously described by Flanagan *et al.* (2021). A summary of facility characteristics and a map with their locations are found in Table 1 and Figure S1, Supplementary information, respectively. The sampled facilities were located in three Swedish cities, Östersund (Ös), Stockholm (S) and Växjö (V). The stormwater management facilities had different types of land use within their catchments: one pond with residential (S2), one with a combination of residential and industrial/commercial (S1) and two with industrial/commercial (Ös1, V1) land use. Two facilities (S5 and S6) had highway/road catchments; however, they also differed from the other four in how the stormwater was conveyed to the facility. The stormwater runoff was transported to pond S5 through a swale and to the underground facility, S6, through stormwater pipes. Setup S5 constitutes an example of a site with multiple consecutive measures for stormwater treatment, while in comparison S6, consists of an underground tank, i.e. treatment in a single step through sedimentation.

2.2. Sampling method

A total of 12 bottom sediment samples were collected from October to December 2019. Sediment cores were taken near the inlet and outlet of each stormwater management facility using a Kayak sediment core sampler (KC Denmark). The sampler was lined with a stainless-steel tube to avoid plastic contamination. Core samplers keep the integrity of a sediment sample with minimum losses of the smaller particles. Two facilities Ös-1 and V-1 had two inlets. For these facilities, sediments were collected from both inlets and pooled in proportions corresponding with the respective inlet's approximate size to

Table 1 | Summary of facility characteristics and locations

Facility ID	**Facility type**	**City**	**Catchment land use**	**Inlet**	**Year of construction**	**Surface area (ha)**	
						Catchment	**Facility**
Ös-1	Pond	Östersund	Industrial/commercial	Pipe/open channel	2000	91	0.45
S-1	Pond	Stockholm	Industrial/commercial residential	Open channel	2009	78	0.17
S-2	Pond	Stockholm	Residential	Open channel	2007	304	0.05
S-5	Pond	Stockholm	Road/highway	Swale	No data	No data	0.06
S-6	Subsurface sedimentation tank	Stockholm	Road/highway	Pipe	1997	1.14	0.006
V-1	Pond	Växjö	Industrial/commercial	Pipe/open channel	1989	519	0.36

one composite sample. Before sampling commenced at a new site, the equipment in contact with samples underwent a thorough rinsing process with water from the respective facility.

At each site, several sediment cores were needed to retrieve sufficient sample amounts for the analyses (further analysis were carried out in other studies by Gavrić *et al.* 2022 and Flanagan *et al.* 2021). To obtain representative composite samples, all the sediment cores from each site were placed in stainless-steel trays and, when feasible, homogenised through a coning and quartering technique using a stainless-steel spoon. For samples where quartering could not be performed due to very liquid sediments, samples were systematically mixed between trays using a spooning technique, as described by Flanagan *et al.* (2021), and carefully distributed into jars by alternating between jars. This sampling/mixing approach ensures that the sample's MP particle composition and concentration represent the average composition and concentrations in the pond sediments.

Empty glass jars were used to collect field blank samples, e.g. to account for MP atmospheric deposition, as described in Section 2.5 (Quality assurance and control).

2.3. Analytical procedures

2.3.1. Sample preparation

Sediment samples underwent pre-treatment in an external commercial laboratory (ALS Scandinavia AB) to remove inorganic and organic material interfering with the analysis and concentrate the samples prior to MP analysis. In the pre-treatment steps, MPs including TWPs in the size range 20–450 μm were extracted from pond sediments (described in detail in Supplementary Information) were similar to those described by Liu *et al.* (2019a) and Molazadeh *et al.* (2023a). Briefly, density separation using a $ZnCl_2$ solution with a density of 1.82 g/cm^3 was added to a 100 g of homogenised wet sediment sub-sample. The mixture was filtered through a 0.45-mm sieve to separate particles of larger dimensions. The mixture of the sample and $ZnCl_2$ (350 mL used for each sample) was then transferred into a separation funnel and left for approximately 20 days. Throughout this period, the settled material was periodically emptied from the funnels, and the funnels were gently agitated during these days to minimise the risk of particles adhering to the funnel walls. The density of the $ZnCl_2$ solution was selected to enable the removal of heavier particles which may otherwise interfere with the analysis, but to retain the road and tyre wear incorporating styrene-butadiene rubber (SBR) fragments from tyres. TWPs are known to have densities in the range of 1.20–1.70 g/cm (Jung & Choi 2022). After removing the heavier particles, the samples were filtered through a 20-μm mesh steel filter. Particles collected on the filters were remobilised and flushed into glass beakers in ultrasonic baths.

The sample collected after density separation underwent a Fenton reaction to remove organic matter, adding H_2O_2 (50%), 0.1 M $FeSO_4$ and 5 M NaOH (Simon *et al.* 2018). The samples were left in this treatment for 7 days. In a subsequent step, the sample solutions were filtered through the 20-μm steel filter, and then the procedure for the Fenton reaction was repeated to further oxidise the organic material; the sample was left undisturbed for another 7-day period. Finally, the processed samples were evaporated under a stream of nitrogen gas until dryness and fixed to a final volume of 5 mL with 50% ethanol in glass vials.

In parallel, sub-samples from the same batch were dried at 105°C to determine dry matter content, and these values were used to relate all concentrations reported to dry matter.

2.3.2. Analysis of MPs using μFTIR

The samples were deposited onto zinc selenide transmission windows (13 × 2 mm; active diameter of 10 mm, active area 78.5 mm^2 – Crystran, UK) and analysed for MPs (excluding TWPs) using μFTIR imaging (Cary 670 FTIR spectrometer coupled with an Agilent 620 FTIR microscope and 128 × 128 Focal Plane Array detector from Agilent Technologies, Santa Clara, Ca, USA) at 5.5 μm pixel resolution in the spectral range 850 to 3,750 cm^{-1}. The signal was collected in a transmission mode by co-adding 30 scans for each tile and scanning 14 × 14 tiles to analyse the whole active area of the ZnSe window. A background obtained on a single tile by co-adding 120 scans was used to subtract the contribution of CO_2 and water vapour (Simon *et al.* 2018; Vianello *et al.* 2019). The μFTIR imaging data were processed using siMPle$^{®}$ software (Systematic Identification of Micro Plastics in the Environment) to characterise particles, including their abundance reported in items/kg, polymer composition and size (Primpke *et al.* 2020). The siMPle$^{®}$ software was also used to estimate the mass of MP particles reported in μg/kg, considering both the particle's dimension and the density of the respective polymers. In these calculations, the particles were assumed to have an ellipsoidal shape, and their thickness was estimated to be 67% of the particle's smallest dimension, a method previously detailed by Liu *et al.* (2019a) and Simon *et al.* (2018).

2.3.3. TWP analysis using Py–GC–MS

μFTIR cannot identify TWPs as they contain carbon black, which absorbs IR light totally. TWP was, instead, analysed by Py–GC–MS. The equipment comprised a micro furnace pyrolyser (EGA/Py-3030D FrontierLab, Japan) coupled with an auto-shot sampler (AS-1020E, FrontierLab, Japan), gas chromatograph and mass spectrometer (Thermo Scientific TRACE 1310 GC, and ISQ™ single quadrupole GC–MS with helium as the carrier gas). For the identification and quantification of TWP, different pyrolysis products have been used in previous studies as markers of the most common rubbers in tyre production, e.g. natural rubber, butadiene rubber and SBR (Rosso *et al.* 2022). In the present study, dipentene, 2,4-dimethyl-4-vinylcyclohexene and cyclohexenylbenzene were used as markers for SBR and 4-vinylcyclohexene was used for quantification. Based on these results, TWP concentrations were determined using an external calibration curve and normalised to an internal standard, deuterated anthracene (A-d10) (Rasmussen *et al.* 2023).

TWP is a subset of the total MP in the sample, while MPs identified by μFTIR are another subset. The two subsets were analysed by two fundamentally different analytical techniques, which, in the current study, were used to complement each other. The two methods provided different types of data: Py–GC–MS gave a TWP mass quantification, while μFTIR yielded particle counts, sizes and mass estimates for each MP particle. To distinguish between the two types of data, the term MP_{TWP} is used to indicate TWPs, while the term MP_{FTIR} is used to indicate MPs determined by μFTIR, hence excluding TWP.

2.4. Data analysis

Descriptive statistics were calculated for MP_{FTIR} and MP_{TWP} concentrations at inlet and outlet in the ponds. The non-parametrical statistical hypothesis test, Wilcoxon signed-rank test, was used for pairwise comparison of inlet and outlet concentrations of MP_{FTIR} and MP_{TWP}. The test was performed at 90% significance level ($\alpha = 0.1$), which was necessary due to the limited number of sampled facilities ($N = 6$).

2.5. Quality assurance and control

Two air blanks were collected in the field in empty glass jars (described in Section 2.2.) to evaluate possible MPs' atmospheric deposition during sampling, homogenisation and sample preparation. Potential sources of MPs included high-visibility clothing (containing cotton, polyester, Cordura™ and PVC) and waders (containing PVC, polyester and neoprene/nylon). After sampling, the blanks were sealed with aluminium foil, handled, stored and transported in parallel with the other samples. Results from blank sample analysis are presented in Supplementary information (section Blank samples). In summary, the blank samples indicate that there may be contamination of polyester. However, the low numbers of other MPs detected indicated a low degree of contamination, therefore, abundance of MP reported were not corrected for blank values, in line with HELCOMs recommendations for monitoring of microlitter (Putna-Nimane & Barone 2022).

3. RESULTS AND DISCUSSION

3.1. MP abundance

MP_{FTIR} were abundant in all pond sediment samples. The total concentration of detected MP_{FTIR} ranged from 1,440 to 77,200 items/kg (average: 17,600 items/kg, median: 7,070 μg/kg) corresponding to 120–2,950 μg/kg (average: 1,180 μg/kg, median: 858 μg/kg), Figure 1.

In comparison to previous studies, our findings showed concentrations by number in the same order of magnitude as those reported in stormwater pond sediments in studies from Denmark by Liu *et al.* (2019a) and Molazadeh *et al.* (2023a). Liu *et al.* reported values in the range 1,510–128,000 items/kg and Molazadeh *et al.* reported an average of 44,400 items/kg. Furthermore, Moruzzi *et al.* (2020) measured 57,500 items/kg in sediments of reservoirs. Ashiq *et al.* (2023) reported average concentrations of 37,500 and 54,100 items/kg in ponds. However, some other studies show considerably higher and lower concentrations. For example, Olesen *et al.* (2019) reported concentrations of 950,000 items/kg in stormwater pond sediments. In line with this, a study of MP concentrations in sediment from a stormwater floating treatment wetland reported average concentrations of 320 and 595 items/kg at the inlet and outlet, respectively (Ziajahromi *et al.* 2020).

In terms of mass, there were larger variations. Mass concentrations reported in this study are within the range reported by Liu *et al.* (2019a) of 115–28,700 μg/kg. However, it is worth noting that the study by Liu *et al.* (2019a) included particles in a larger size range of 10–2,000 μm, which may affect the measured mass concentration, resulting in higher ones than if a smaller size range is applied, as in our case in the present study. A study by Brooks *et al.*, (2023) can illustrate the impact the analysed size ranges may have on the mass as they reported lower concentration in terms of number (2.5–203 items/kg)

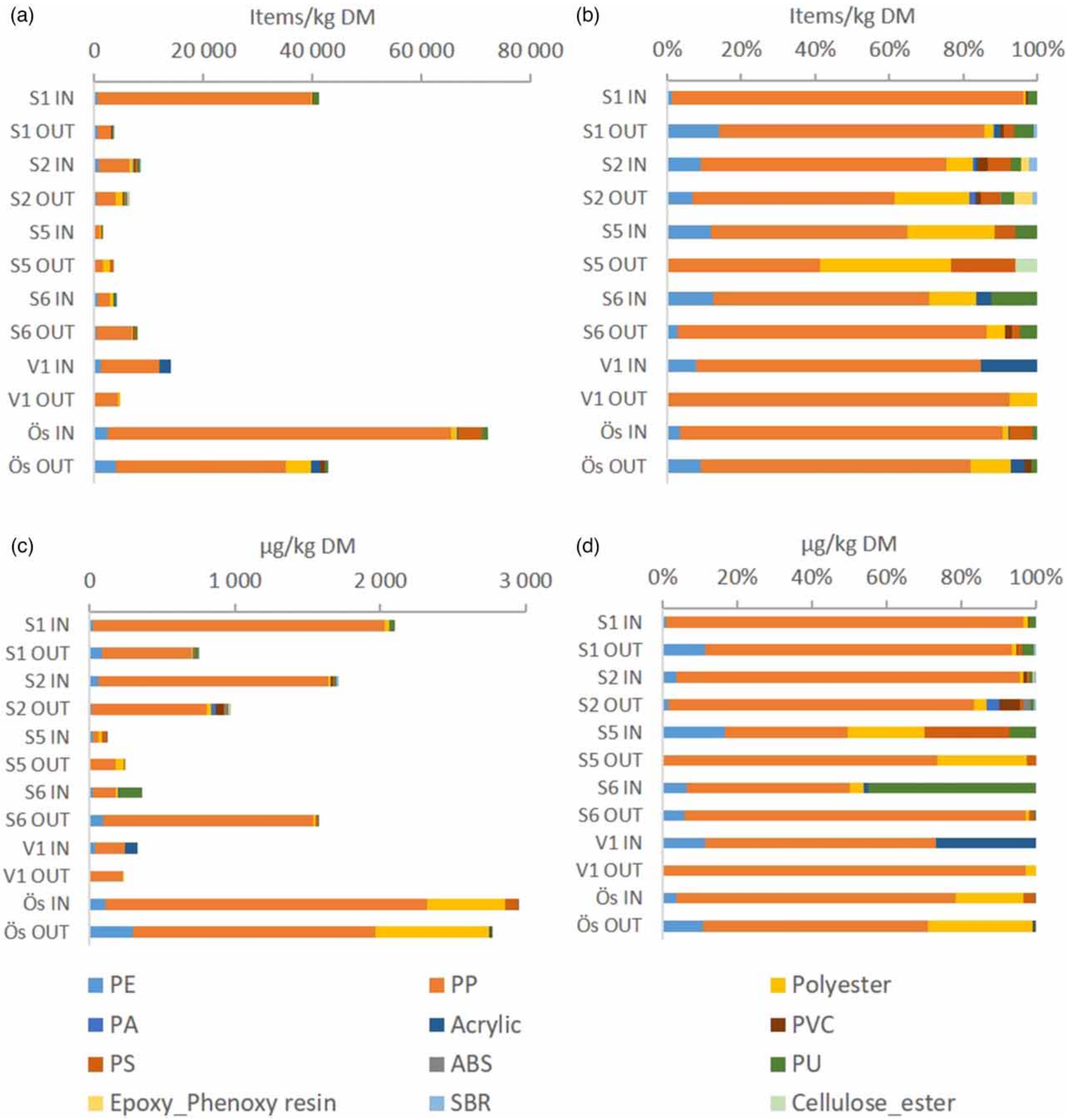


Figure 1 | MP concentrations by polymer type, illustrating (a) number of microplastics in items/kg and (b) percentage of all items by polymer type, (c) mass of microplastics in µg/kg and (d) percentage by polymer type. PE, polyethylene; PP, polypropylene; PA, polyamide; PVC, polyvinyl chloride; PS, polystyrene; ABS, acrylonitrile butadiene styrene; PU, polyurethane; SBR, styrene-butadiene rubber. All concentrations are referring to the dry mass of the sediment.

but reported mass concentration ranging up to 5,313,000 µg/kg. Note that they included MP particles up to 5 mm. Even when comparing studies with similar size ranges as in the present study, variations in mass concentration are noted. Ashiq *et al.* (2023) analysed particles in the size range 10–750 µm and reported higher mass concentrations for their two studied ponds, with average concentrations of 109,900 and 167,200 µg/kg, respectively. Moreover, Olesen *et al.* (2019), who analysed particles in the 10–500 µm size range, reported a higher mass concentration of 401,500 µg/kg. However, lower concentrations compared to those in the present study have also been reported. In a study by Molazadeh *et al.* (2023a), where sediment samples were taken at different locations within a stormwater pond, an average concentration of 11.8 µg/kg was reported, which is lower in comparison to this study. Noteworthy, in three of the above-mentioned previous studies (Liu *et al.* 2019a; Olesen *et al.* 2019; Molazadeh *et al.* 2023a), similar sample preparation and analytical methods were used, i.e. density separation, removal of organic material and identification and quantification using µFTIR. $\mathrm{MP_{FTIR}}$ concentrations in sediments show wide variability, and data from different studies are affected by site and source-specific differences (e.g. type of land use within the catchment). Moreover, using different methods for sample preparation and analysis, as well as analysed size ranges can significantly affect results and comparability between studies. Particularly, when comparing studies with methodological variations, using a denser solution in density separation, broader size ranges and including both smaller and larger particles can result in reporting higher MP concentrations.

3.2. Polymer type composition

In total, 12 different polymer types were identified with μFTIR in the analysed samples. The most commonly detected polymer types were (in decreasing order): PP (12), polyester (11), PE (10), polyurethane (PU) (9), PS (7), acrylic (7), PVC (6), PA (3), epoxy/phenoxy resins (3), SBR (3) acrylonitrile butadiene styrene (ABS) (1) and cellulose esters (1). The number of samples where each polymer type was detected is given within parenthesis. However, the results for polyester should be interpreted with caution since particles and fibres of this polymer type were present in the blank samples, see Supplementary information (section Blank samples). In addition to the PP ubiquity in the analysed samples, this polymer was also the dominating type in terms of number of particles with concentrations in the range 760–63,000 PP items/kg and mass concentrations in the range of 39.7–2,210 μg PP/kg. This accounted on average for 71% of the detected $\mathrm{MP_{FTIR}}$ by number and 74% by mass (Figure S2, Supplementary information). Accordingly, PP was reported as one of the dominant polymer types in sediments from a variety of stormwater treatment facilities such as drains, pipelines, gully pots, biofilters and in sediments from stormwater ponds (Liu *et al.* 2019a; Lutz *et al.* 2021; Sang *et al.* 2021; Lange *et al.*, 2023; Molazadeh *et al.* 2023a; Öborn 2024). Considering that PP, which has a relatively low density, was dominating among the settled particles, this indicates that additional factors to density affect whether or not the particles settle (Molazadeh *et al.* 2023a). The dominant prevalence of PP in the sediments could be due to these particles being so prevalent in the incoming stormwater making this polymer type dominant even if PP does not have the largest percentage settling in relation to what reaches the pond via the stormwater. The widespread of PP in the stormwater is also reflected in previous studies such as Liu *et al.* (2019b) and Rasmussen *et al.* (2024).

3.3. Particle size distribution

The size distribution of MP particles by major dimensions, presented as the average of all 12 samples included in this study, shows that 70% of all particles were 100 μm or smaller (size range 11–100 μm), Figure 2. This is in line with results from previous studies of stormwater pond sediments by Liu *et al.* (2019a) and Molazadeh *et al.* (2023a). Liu *et al.* (2019a) detected the majority of particles in the size range 10–50 μm and that 94.7% of the detected particles were smaller than 250 μm. However, particle sizes were only divided into five unevenly distributed size fractions, of which the two smallest fractions include particles 10–50 and 50–250 μm. Molazadeh *et al.* (2023a) also reported particle size distribution in pond sediment samples, and

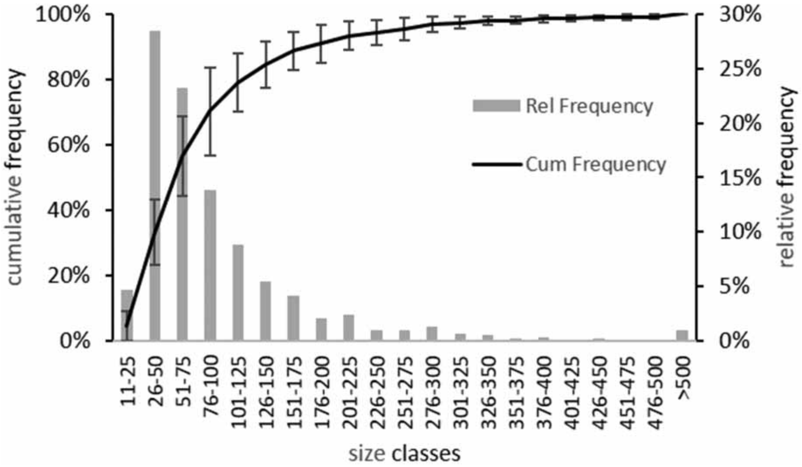


Figure 2 | Size distribution (major dimensions) shown as relative frequency and cumulative frequency (in percentage of number of microplastics). Values shown are an average of the 12 samples with error bars representing the standard deviation. Note that the first size class is smaller (11–25 μm), particles of larger size (500 μm) are reported as one group.

the absolute majority of the particles (in terms of number) have been reported in the size range 10–100 µm, followed by 100–200 µm. Hence, these results also align with the present study, even though larger size ranges of 100 µm were used, except for the smallest and largest classes of 10–100 µm and 500–1,000 µm, respectively.

Prior research on MPs within urban stormwater has shown a trend of increasing quantities with decreasing particle size. The predominant size range for MPs in stormwater, as observed in studies by Järlskog *et al.* (2021, 2020) and Lange *et al.* (2022), falls within the 20–100 µm range, compared to particles larger than 100 µm. In contrast, MPs accumulated in gully pot sediments were mainly particles larger than 125 µm, illustrated in Figure S3, Supplementary information (Öborn 2024). Based on this, it is reasonable to assume that the larger MP particles tend to settle early in the urban drainage system, for example, in gully pots, while a higher concentration (by number) of smaller particles tends to be transported further downstream via stormwater (Järlskog *et al.* 2020, 2021; Lange *et al.* 2022). This enables them to reach stormwater ponds and settle so they can be found to a relatively large extent also in pond sediments.

3.4. TWP abundance

TWPs were detected in 11 out of 12 analysed samples, and the results showed that concentrations in the sediments vary greatly across positions in the ponds and sites. Concentrations ranged from below detection limit up to 69,300 µg/kg (Figure 3) with an average concentration of 12,000 µg/kg (median: 3,530 µg/kg). Pond S5, which receives runoff from a heavily trafficked highway (AADT 93,900 vehicles/day, two directions), showed relatively low TWP concentration (840 µg/kg) in the inlet sediment, and no detected TWPs in the outlet sediment. This pond receives stormwater transported through a drained swale, allowing for detention of particles, including TWPs, and this is consequently a possible explanation to the unexpectedly low concentrations. The underground sedimentation tank S6, which also receives runoff from a road catchment (AADT 60,000 vehicles/day, two directions) similarly showed relatively lower TWP concentrations of 1,130 and 910 µg/kg at inlet and outlet, respectively. One possible explanation could be that runoff from this surface type contains relatively higher amounts of other particles, such as mineral particles, which could dilute the TWP content.

The results show variations in TWP concentrations between the different ponds and within ponds when comparing inlet and outlet concentrations, and a high variability was observed among the analysed triplicates, illustrated in Figure 3. A potential reason for the variations among triplicates can be that small volumes are analysed in the Py–GC–MS, and one or a few larger particles are likely to have a relatively large impact on the results.

Three previous studies utilised similar analytical techniques using Py–GC–MS to analyse the content of TWP (size range 10–500 µm) within sediment samples. They reported TWP concentrations in different environments and matrixes: stormwater pond sediments with concentrations between 18,000 and 860,000 µg/kg (Rasmussen *et al.* 2024), sediments from an urban lake in Denmark with an average TWP concentration of 19,000 µg/kg (Molazadeh *et al.* 2023b) and particles in dust from permeable road surfaces and parking lots in residential, industrial and commercial areas with an average TWP concentrations 894,800 µg/kg (Rasmussen *et al.* 2023).

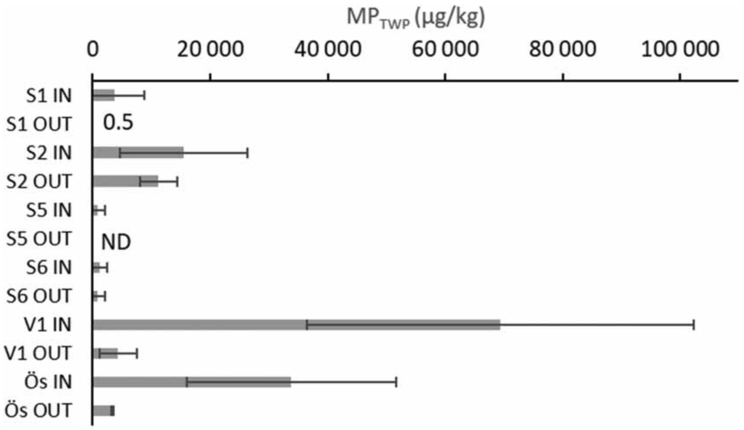


Figure 3 | TWP mass concentrations in µg/kg, showing the mean concentration for each sample (triplicates analysed) with error bars illustrating standard deviation. All concentrations are referring to the dry mass of the sediment.

The concentrations in the present study, ranged up to 69,300 µg/kg with an average of 12,000 µg/kg, are within the range of those previously reported by Rasmussen *et al.* (2024) in stormwater pond sediments and also in the same order of magnitude as those in the urban lake studied by Molazadeh *et al.* (2023b). Additionally, a study on TWP in sediment from the southern German Bight and river Weser showed lower TWP concentrations in sediments compared to the present study, with 72 µg/kg in the most contaminated sediments (Goßmann *et al.* 2021). TWP concentrations reported in sediments from gully pots were considerably higher in comparison to the results in the present study. Concentrations ranging from less than 1,000,000 up to 150,000,000 µg/kg (i.e. 0.1–15% of the sediment mass) were found in gully pot in southern Norway (Mengistu *et al.* 2021), and from 2,000,000 to 26,400,000 µg/kg (i.e. 0.2–2.6%) in roadside soils from low traffic areas (650–14,250 vehicles per day) (Rødland *et al.* 2023). Gully pots with a sediment trap/sump can be considered sedimentation facilities similar to ponds; however, they are at quite a different scale.

TWPs are relatively dense; therefore, sample preparation (specifically density separation) is an important aspect to consider when comparing results from different studies. Interestingly, Rødland *et al.* (2023) showed that it was feasible to analyse TWPs in soil using Py–GC–MS without prior removal of the sample matrix, whereas in the present study, density separation with $ZnCl_2$ (density 1.82 g/cm^3) was applied to remove dense material. Additionally, the selection of specific markers for TWPs and variations in tyre composition across countries and seasons also contribute to differences in the results, making the comparison between different studies even more challenging.

3.5. MP including TWP distribution between inlet and outlet

High variability of MP_{FTIR} and MP_{TWP} in inlet and outlet concentrations was observed both by number and mass as well as between the facilities (descriptive statistics of concentrations at in- and outlet in Table S2, Supplementary information). The abundance of MPs varied with an average concentration of 23,600 items/kg, corresponding to 1,260 µg/kg at inlets, and 11,500 items/kg, corresponding to 1,090 µg/kg at outlets (Figure 4(a) and 4(b)). When comparing corresponding concentrations for MP_{TWP}, higher values can be observed in inlet sediment compared to the outlet with average concentrations of 20,700 µg/kg and 3,300 µg/kg, respectively (Figure 4(c)). This was also confirmed by pairwise comparison ($p < 0.1$ Wilcoxon signed-rank test), potentially indicating that MPs_{TWP} tend to settle earlier in the pond, i.e. near inlets. This may be due to the relatively higher densities of this type of particles. In terms of MP concentrations by number, a tendency towards higher concentrations at inlet compared to outlet was noticeable, although it was not statistically significant ($p > 0.1$ Wilcoxon signed-rank test). This may be due to the small sample size ($N = 6$) in combination with the large variation in concentration between sites. In terms of MP_{FTIR} mass, no clear difference in concentrations was observed, which was also confirmed by pairwise comparison ($p > 0.1$ Wilcoxon signed-rank test). When comparing concentrations, one thing to keep in mind is where, in the system, other particles, such as mineral particles, tend to settle. The internal relationship between the sediment and MP content in the stormwater phase may not be conserved when the particles are settled.

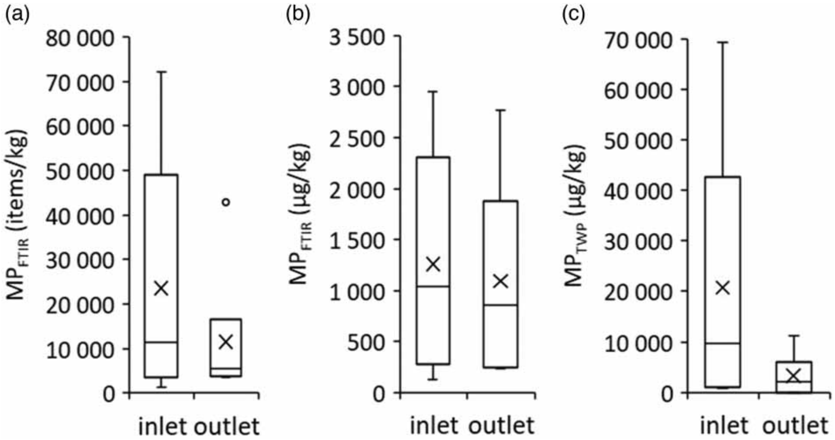


Figure 4 | Boxplots illustrating concentrations at inlet and outlet: (a) number of MP_{FTIR} /kg, (b) mass in µg of MP_{FTIR}/kg and (c) mass in µg of MP_{TWP}/kg.

Consequently, a similar concentration of MPs in inlet and outlet samples is not necessarily due to a similar settling rate at the two locations.

In general, pond sediment near the inlet has coarser particle sizes than near the outlet (Al-Rubaei *et al.* 2016), which is also in line with the measured particle size distribution in the ponds included in this study, previously presented by Gavrić *et al.* (2022). However, no clear trends were observed when comparing MP particle size distribution at inlet and outlet (Figure S4, Supplementary information). Fluid mechanics modelling of MPs in stormwater ponds found mixing to be an important factor for transporting buoyant MPs (e.g. PP) to the pond sediments (Molazadeh *et al.* 2023a). The same study found no systematic relationship between the location of sediment in a stormwater pond and the characteristics of MPs, such as their mass, size and polymer composition. Ziajahromi *et al.* (2023) found higher MP concentrations in sediment at the inlet compared to the outlet when sampling sediments in constructed wetlands. Similarly to stormwater ponds, constructed wetlands allow for the sedimentation of particles, including MPs, which is why the results can be considered in line with the present study. Furthermore, García-Haba *et al.* (2023) stressed in their review that the majority of descriptive statistics show a decline in MP concentrations from the system's inlet to its outlet. Urban drainage systems could play a substantial role in controlling MP pollution, for example, ponds where potential MP removal efficiencies of 85–99% were shown (Stang *et al.* 2022).

4. CONCLUSIONS

This study has increased the knowledge about MPs including TWPs in stormwater pond sediment. By combining μFTIR and Py–GC–MS analyses, MP_{FTIR} and MP_{TWP} concentrations were simultaneously determined. MP_{FTIR} concentrations were reported in both number and mass, with concentrations of 1,440–72,200 items/kg and 120–2,950 μg/kg, respectively. MP_{TWP} concentrations by mass were generally higher, ranging from below detection limit up to 69,300 μg/kg.

In general, in sediment samples from inlet and outlet alike, smaller particles, particularly measuring 100 μm and smaller, were more frequently occurring and accounted for on average approximately 70% of the particles.

In total, 12 different polymer types were detected using μFTIR. PP, detected in all sediment samples, was the predominant MP_{FTIR} type in terms of particle number and mass accounting on average for 71% of the detected MPs by number and 74% by mass. This despite the relatively low density (approximately 0.905 g/cm^3), which suggests that the polymer density might not be the sole factor influencing the settlement of PP particles. $MP_{TWP,}$ detected in 11 of 12 samples, was the dominating polymer type, in terms of mass, in nine samples.

The MP results showed great variation between concentrations in different ponds and also location within the pond (inlet and outlet). MP_{TWP} concentrations showed a trend of higher concentrations in sediments sampled at the inlet, compared to outlet, when pairwise comparing the concentrations. There were some indications of a similar trend concerning MP_{FTIR} concentrations to the number of particles; however, not statistically significant, whereas for the MP_{FTIR} concentrations of mass no such trend was observed at all.

The results from this study imply that ponds can act as sinks for MPs released in the urban environment since MPs were present in all pond sediment samples. In future research, a setup where stormwater and sediment are sampled in parallel (e.g. at inlet and outlet) over time would be valuable for assessing the treatment performance.

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DATA AVAILABILITY STATEMENT

All relevant data are included in the paper or its Supplementary Information.

CONFLICT OF INTEREST

The authors declare there is no conflict.

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