

Controls on Microplastics Accumulation in Stormwater Ponds

by

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Author's Declaration

This thesis consists of material all of which I authored or co-authored: see Statement of Contributions included in the thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners. I understand that my thesis may be made electronically available to the public.

Statement of Contributions

Chapter 2 of this thesis consists of a co-authored, manuscript-format paper. Chapter 3 of this thesis will be modified for submission. As the first author of both chapters, I was primarily responsible for the study design, execution, data collection and analysis, and writing. The contributions of the listed co-authors are as follows: I designed the experiment with the guidance of Fereidoun Rezanezhad, Stephanie Slowinski, and Philippe Van Cappellen. I performed the laboratory work with the assistance of Stephanie Slowinski. Amir Reshadi contributed to field sampling in this thesis as well as accumulation rates calculation in Chapter 2. Fereidoun Rezanezhad and Philippe Van Cappellen supervised the project and provided guidance on data interpretation. I performed the data analysis, visualization, and wrote the chapters. Chapter 2 was reviewed and approved by all co-authors, while Chapter 3 will be modified for journal submission and reviewed and approved by all co-authors.

Abstract

Microplastics (MPs), or plastic particles that are less than 5 mm in diameter, are an emerging threat to aquatic and terrestrial ecosystems because of their potential toxicity and their resistance to degradation. In urban watersheds, stormwater runoff is a major carrier of MPs to downstream water bodies, which often drains into green infrastructure such as stormwater ponds (SWPs). The existing evidence indicates that SWPs may be effective at reducing the export loads of MPs from urban areas. However, the effectiveness of SWPs in retaining MPs and the controls on their accumulation in SWP remain understudied. Hence, it is significant to investigate MP occurrence and factors controlling MP distribution in SWPs.

In this thesis, I aimed to (1) assess the variability in MP types, sizes, and abundances within SWP sediment and water samples, (2) determine the influence of sediment properties (*e.g.*, organic carbon concentration, particle size) on the types, shapes and sizes of MPs accumulating in sediments versus within and between SWPs, and (3) investigate the impact of SWP characteristics on MP accumulation, including land use and land cover (LULC). I addressed these research objectives by collecting sediment and water samples from five SWPs with different LULC (industrial, residential, and commercial) in the City of Kitchener, Ontario, Canada, extracting MPs from environmental samples, and characterizing MP particles using laser direct infrared (LDIR) imaging spectroscopy.

In Chapter 2, I extracted MPs from sediment cores in triplicates and determined MP counts and morphologies using the LDIR. I also analyzed sediment for grain size, mineralogy, and organic carbon (OC) content. The results revealed that the MPs accumulated in the sediments were predominantly fragments, with concentrations approximately 50 times higher than fibers, implying an important role of particle shape in controlling the accumulation of MP particles in

SWP sediments. The highest fragment concentrations (2.3×10^8 particles kg dw⁻¹) were found in the commercial SWP, while the highest fiber concentrations (4.5×10^6 particles kg dw⁻¹) were found in the industrial SWP. Surface area-normalized MP accumulation rates in the forebays were generally 2-5 times higher than in the main basins. Sediment grain size and catchment impervious cover were significantly correlated with MP accumulation rates, with increasing MP concentrations observed with finer sediment grain size and higher catchment imperviousness. Polyamide and polyethylene were the two most abundant polymers found in the pond sediment, along with an overwhelming dominance of MP particles less than 50 μ m. MP polymer composition and size distribution thus reflected the contribution of urban activities to MP pollution in a watershed. These findings indicate the important role of catchments' land cover in the build-up and wash-off of sediments and MPs to downstream areas such as SWPs.

In Chapter 3, I quantified and characterized MP shapes, types, and sizes in stormwater samples collected bi-monthly from 5 SWPs. In a one-year water sample collections, MPs appeared to fluctuate with significant seasonal variation throughout the year with the highest concentration recorded in a residential pond (up to 20,166 fragment L⁻¹ and 559 fiber L⁻¹). Polyamide and polyethylene accounted for approximately 80% of total MP in the pond water, while small-sized MPs make up 85% of the particles, highlighting the impact of catchment land use on MP occurrence in SWP. Precipitation, wind speed, and pond hydraulic loading were found to wash away surface MPs and dilute MP concentration in the water column. These findings demonstrate the diversity in MP profiles associated with climate factors, implying a need for long-term monitoring to address those spatial and temporal variability.

The results from Chapters 2 and 3 overall provided an insight into MPs' preferential partitioning between two different environmental matrices, which can be applied in future

research to assess the sources, transport, and fate of MPs in the freshwater ecosystem. Data from this research can be applied to generate MP accumulation rates across the Grand River watershed and eventually the Great Lakes. The outcomes from this study, moreover, actors influencing MP accumulation in urban catchments, and therefore, can support further studies on characterizing MP mass balance and budget for urban watersheds. Since SWPs are an effective indicator of local sources of MP pollution, understanding the MP from urban catchment can inform policymakers in a larger aquatic ecosystem to tailor management strategies accordingly. The research study presented in this thesis therefore contributes to the development of local policies and regulations, which not only address specific sources of MP pollution but also serve as models for larger-scale regulations aimed at protecting the freshwater environment.

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List of Abbreviations

Mt	Million tons
MP	Microplastic
SWP	Stormwater retention pond
OC	Organic carbon
LULC	Land use land cover
MDL	Method detection limit
ZnCl ₂	Zinc chloride
Fe(II)	Iron (II)
H ₂ O ₂	Hydrogen peroxide
LDIR	Laser direct infrared imaging
SA	Surface area
ABS	Acrylonitrile butadiene styrene
PA	Polyamide
PE	Polyethylene
PET	Polyethylene terephthalate
PLA	Polylactic acid
PMMA	Polymethyl methacrylate
POM	Polyoxymethylene
PP	Polypropylene
PTFE	Polytetrafluoroethylene
PU	Polyurethane
PVC	Polyvinyl chloride

1 Introduction

1.1 Mismanaged plastic waste and microplastics

Plastic, or synthetic polymers, is an ever-present material in every aspect of human lives owing to its durability, versatility, and cost-effective production (Tuladhar and Yin, 2019). Manufactured to be strong, lightweight and easily moldable for intended purposes, plastic has become prevalent across numerous sectors such as packaging, construction, and household appliances (Andrady and Neal, 2009; Zoeller, 2024). Fok *et al.* (2019) even emphasized that we currently live in the Plastic Age with the ubiquitous presence of plastic. First invented in the 1800s and developed for industry during the 1920s, the mass production of plastic started rocketing during the second industrial revolution in the 1950s (Andrady and Neal, 2009; Geyer, 2020). From 1.7 million tonnes (Mt) in 1950, global plastic production reached 350 Mt by 2017 with an average 8.7% growth per year (Geyer, 2020). Worldwide plastic production kept increasing to 400 Mt in 2022 with packaging accounting for 40% of the products and was projected to reach 1.1 billion tonnes in 2050 (PlasticEurope, 2023). The roaring production of plastic combined with inefficient strategies to handle end-of-life plastic products, nevertheless, has resulted in a continuously growing amount of plastic waste (Fok *et al.*, 2019; Jambeck *et al.*, 2015). According to Geyer *et al.* (2017), mismanaged plastic waste surpassed 6.3 billion tonnes in 2015, with only 9% of which was recycled, 12% was incinerated, and the other 79% was in landfills or released to the environment. Diggle and Walker (2022) later pointed out that packaging waste accounted for up to 35% of global plastic waste, while Jambeck *et al.* (2015) estimated that approximately 13 million metric tons of plastic were discarded into the ocean. MacLeod *et al.* (2021) also highlighted that global plastic emissions can continuously rise to

roughly 50 Mt in 2025, even if efforts to reduce plastic waste were made. Plastic waste in the environment is undoubtedly one of the most severe anthropogenic issues (Thushari *et al.*, 2020).

It is vital to note that after being exposed to the environment for a prolonged duration, plastics can be chemically and physically weathered and fragmented into smaller plastic debris (Zhang *et al.*, 2021). In fact, plastic particles that are less than 5 mm in size, or microplastics (MPs) have been an emerging contaminant in the environment (Ivleva *et al.*, 2017). MPs can be released directly into the environment as primary MPs from microfibers in textiles and microbeads in personal care products, or indirectly as secondary MPs from the breakdown of big plastic pieces (Bläsing and Amelung, 2018; Emenike *et al.*, 2023; Guo *et al.*, 2020). The sources of MPs, thus, are extremely diverse (Horton *et al.*, 2017). Small sizes of MP particles additionally facilitate wide-range transport in several environmental compartments such as water, atmosphere, and soil, resulting in a high abundance of MPs in the ecosystems (Chen *et al.*, 2020). Desforges *et al.* (2014) noted a density of up to 10,000 MP particles m⁻³ in seawater, while Eriksen *et al.* (2013) and Horton *et al.* (2017) reported MP contamination in freshwater with a similar concentration to those found in marine environments. Wong *et al.* (2020) further assessed that MPs abundance in terrestrial ecosystems can be at least 4 times more than MPs in aquatic environments. The numerous origins coupled with the ubiquitous presence of MP undeniably exacerbate MP pollution in the environment (Ghosh *et al.*, 2023).

1.2 Microplastics as an emerging contaminant

The degradation-resistant nature and prevalence abundance of MPs, moreover, posed a significant threat to aquatic organisms and ecosystem services (Ivleva *et al.*, 2017). In particular, ingestion of MPs can cause inflammation, induce metabolic disturbances, impair reproductive function, and increase mortality in marine organisms (Lu *et al.*, 2016; Oliveira *et al.*, 2013;

Rochman *et al.*, 2013; Von Moos *et al.*, 2012; Wang Xiao *et al.*, 2020). Anbumani and Kakkar (2018) also believed that at least 200 marine species are poisoned by plastic ingestion. Similarly, MP toxicity in freshwater biota has been widely documented with damaging impacts on species' survival, fitness, and growth (Au *et al.*, 2015; Blarer and Burkhardt-Holm, 2016). For instance, Greven *et al.* (2016) investigated polycarbonate and polystyrene particles in fathead minnow and indicated that MPs not only acted as a stressor on the immune system but also interfered with the species' disease resistance, raising serious health concerns for the freshwater fish population.

More significantly, plastic products are manufactured with chemical additives such as flame retardants and plasticizers, which can be released into the water bodies and accumulated in organisms throughout the aquatic food web (Gunaalan *et al.*, 2020; Hahladakis *et al.*, 2018). Uptakes of these toxic chemicals, especially hydrophobic persistent organic pollutants, can cause immobility, abnormal development, and neurological dysfunction in aquatic species (Hermabessiere *et al.*, 2017; Wang Xin *et al.*, 2020). Sanchez *et al.* (2014) and Wang *et al.* (2019) further reported large ingestions of small MPs particles by freshwater biota at different trophic levels, signalling bioaccumulation and biomagnification of MPs upward the food chain. Since humans are at the top of the food chain, it is no surprise that MP particles have been detected in human cells and blood, inducing damaging effects such as inflammatory reactions and cardiovascular disease. (Çobanoğlu *et al.*, 2021; Hwang *et al.*, 2019). The combined toxicity of MPs and chemical additives pose major risks to human health as well, including carcinogenesis, neurotoxicity and DNA damage (Haldar *et al.*, 2023; Kumar *et al.*, 2022). Even though the health impact of MPs on marine mammals and humans are not yet fully understood, the environmental abundance and potential toxicity of MPs and their associated hazardous substances are extremely alarming.

1.3 Microplastics in urban area

As MPs are interwoven in every aspect of anthropogenic activities, urban areas with high population density are anticipated to emit enormous amounts of MP (Qiu *et al.*, 2020; C. Wang *et al.*, 2021). In particular, MPs have been detected in various morphologies and quantities in air (Sun *et al.*, 2022; Wright *et al.*, 2020), soil (Rafique *et al.*, 2020; Zhou *et al.*, 2022), sediment (Townsend *et al.*, 2019; Turner *et al.*, 2019), and water (Tibbetts *et al.*, 2018; Xu *et al.*, 2021) of urban environments. Horton *et al.* (2017) and C. Wang *et al.* (2021), moreover, pointed out that there are relationships between MPs abundance in urban freshwater systems and proximity to the emitting source as well as the density of human activities nearby. Once MPs are released into the urban environment, they follow various pathways that facilitate their dispersion into other ecosystems, including stormwater runoff (Fältström and Anderberg, 2018; Österlund *et al.*, 2023). Stormwater runoff is excessive surface water overflowing the land covers during a rainstorm or snowmelt event, which is critical in transporting MPs from urban surfaces to water bodies (Werbowski *et al.*, 2021). de Jesus Piñon-Colin *et al.* (2020) moreover investigated MPs in stormwater from five urban catchments in Mexico and recorded up to 200 particles per L of runoff, while the review by Herath *et al.* (2022) indicated a significantly higher number of MPs in stormwater compared to the river and lake waters. As a result, de Jesus Piñon-Colin *et al.* (2020) and Herath *et al.* (2022) agreed that urban stormwater runoff is a critical source of MP pollution and emphasized the need for stormwater control measures.

To mitigate the non-point source pollution resulting from stormwater runoff, stormwater ponds (SWPs) have been designed to detain stormwater until they are overflowed by the upcoming runoffs (Tixier *et al.*, 2011). Moreover, contaminants associated with the runoff, including MPs, are deposited in the pond and removed via physical, chemical, and biological

processes such as sedimentation and plant uptake afterward (Flanagan *et al.*, 2021; Yazdi *et al.*, 2021). Studies have found that SWPs can accumulate significant quantities of MPs, with concentrations varying depending on the pond's catchment and surrounding land use (Brooks *et al.*, 2023; Liu *et al.*, 2019a; Liu *et al.*, 2019b; Rasmussen *et al.*, 2024). Stang *et al.* (2022) also performed a literature review on MP removal from stormwater and reported an 85% retention efficiency of MPs in SWPs. These findings thus suggest that SWPs can act as sinks for MPs, minimizing risks of contamination for downstream water bodies.

Having said that, factors controlling MP distribution and occurrence in SWPs are understudied due to a noticeable bias in MP research trends with dominance in the marine ecosystem. By surveying MP research, Blettler *et al.* (2018) and Horton and Dixon (2018) discovered that 80% of the studies were on marine environments, whereas investigations in freshwater, soil, and atmosphere accounted for only 20% of the papers. As a result, insights into MPs' occurrence in other environments are highly limited. Shruti *et al.* (2021) later performed literature searches for MPs in stormwater runoff and catchment and found only 11 studies available with few papers focusing on SWPs sediment. Stang *et al.*'s (2022) review also emphasized the limited number of available research on MP in stormwater and recommended future investigations into the efficiency of SWPs in reducing final MP concentration in outflowing water. Olesen *et al.* (2019), one of the few studies on MP in stormwater ponds, even stated that MP concentration in SWPs is poorly understood. Considering the damaging impacts and alarming abundance of MPs in the environment, the potential of SWPs as MPs sink, and the knowledge gap in MPs' behaviour from source to sink, research into MPs in SWPs, therefore, is much needed.

1.4 Thesis objectives

The main objective of this thesis was to investigate the variability in MP types, sizes, and abundances within SWP sediment and water samples, and impacts of catchment land use on MP accumulation in SWP. I pursued these investigations through field sampling, laboratory extraction of MPs and detection techniques using laser direct infrared (LDIR) imaging spectroscopy. The specific objectives of this thesis are:

- Assess the variability in MP shapes, types, sizes, and abundances within SWP sediment and water samples.
- Determine the influence of sediment properties (*e.g.*, organic carbon concentration, particle size) on MPs accumulating in sediments within and between SWPs
- Investigate the partitioning in MP distribution between water columns and sediment beds of SWPs, and the impact of catchment land use on MP composition and accumulation in SWPs

1.5 Thesis outlines

This thesis is composed of four chapters, including the current Chapter 1. Chapter 2 is a manuscript on which I am the first author and will be submitted as a standalone document for publication to the *Journal of Science of the Total Environment*. The chapter describes laboratory analyses of extracting and characterizing MPs in SWP sediment to investigate the influence of sediment properties on MPs occurrence in SWPs. Chapter 3 is also a manuscript that presents results regarding MP abundance in SWP water and compares with the results from Chapter 2 to investigate MP distribution between water and sediment of SWPs. Chapter 4 summarizes research findings from this thesis and proposes recommendations for future research. Following Chapter 4, Appendix A contain supplementary materials for Chapters 2 and 3.

Controls on Microplastics Accumulation in Stormwater Pond Sediment

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2 Controls on Microplastics Accumulation in Stormwater Pond Sediment

2.1 Introduction

There has been a recent consensus among the scientific community that microplastics (MPs), or plastic debris less than 5 millimetres in size, are an emerging pollutant in the aquatic and terrestrial environment (Cole *et al.*, 2011; Turner *et al.*, 2019). The mass production of plastics worldwide along with the mismanagement of plastic waste has introduced high amounts of MPs into the environment (Guo *et al.*, 2020). The small sizes of MP particles enable their long-range transport across environmental compartments, including oceans, freshwater systems, soil, sediments, and the atmosphere (Chen *et al.*, 2020). Most environmental MP particles originate from terrestrial sources, especially urban areas (Bradney *et al.*, 2019). MP particles may be carriers of various contaminants, such as polycyclic aromatic hydrocarbons and plastic additives, to downstream water bodies (Bradney *et al.*, 2019). The potential for MPs to release toxic additives or to carry contaminants, their resistance to degradation, and their bioaccumulation up the food chain makes MPs a critical threat to aquatic ecosystems (de Souza Machado *et al.*, 2018).

Urban areas are the major source of MPs to the environment and urban stormwater runoff is therefore a major pathway for the transport of land-derived MPs to downstream water bodies (Stang *et al.*, 2022; Saraswat *et al.*, 2016). To reduce both the large flow volume and large contaminant loads associated with stormwater runoff, it is often treated by green infrastructure such as stormwater ponds (SWPs), in which sediments and contaminants settle and accumulate in the SWP sediment (Österlund *et al.*, 2023). MP accumulation in SWP sediments has been

reported (Liu *et al.*, 2019a; Liu *et al.*, 2019b), indicating that SWPs can reduce the loads of MPs derived from catchments to downstream water bodies. However, the factors controlling MP accumulation in SWP sediments are not still well understood.

Once entering water bodies, the density of plastic type can determine the environmental fate of the MP particles (Kaiser *et al.*, 2017; Waldschläger *et al.*, 2020). Liu *et al.* (2019a) and Liu *et al.* (2019b) investigated MPs accumulation in water and sediments of seven stormwater ponds with different land use in Denmark, in which the results indicated low-density plastic tend to float in the water column while heavier plastic particles readily deposit into the sediment. He *et al.* (2021) later reported that the low-density polymers such as polyethylene might be captured by sediment due to suspension by clay particles. The pattern of MP distribution in sediment, therefore, remained unclear due to variability with MP particles' size, density, and type as well as the sediment properties. Considering MP particles' size and shape, Waldschläger *et al.* (2020) reported that larger particles can simply settle in sediments, whereas Ballent *et al.* (2016) found irregularly shaped fragments to be readily retained by sediment. Waldschläger and Schüttrumpf (2019) later showed that MP settling behaviour in sediment was vastly different among beads, fibers, and fragments.

Sediment properties such as organic carbon (OC) content and grain size are expected to influence MP distribution patterns as well. Enders *et al.* (2019) claimed that MP and sediment organic matter distribution were likewise owing to their similar physical properties such as MP density, leading to a hypothesis about the correlation between MP abundance and the OC content of the sediment. Previous studies have also investigated this hypothesis and the results were diverging, as only a few studies reported a positive correlation between percentage of OC and MP particle numbers (Jahan *et al.*, 2019; Li *et al.*, 2020; Ling *et al.*, 2017; Maes *et al.*, 2017;

Strand *et al.*, 2013). Harris (2020) further performed a comprehensive literature review of MP in marine environments and indicated that only 15 out of 80 examined studies measured OC in sediments, five of which reported no correlation between MP abundance and OC content. On the other hand, MP abundance has been reported to decrease with sediment grain size or increase with fine-grained sediment. Alomar *et al.* (2016) found that finer sediment such as clay or silt has a higher adsorption capacity for OC due to their large surface areas, implying a higher accumulation of MPs in fine-grained sediments. This implication has been investigated in some other studies with conflicting results, which again generated mixed findings. For instance, Falahudin *et al.* (2020) analyzed MP particles in coastal zones and reported a positive correlation with the proportion of silt and clay, in contrast with the negative correlation to the percentage of sand in sediment. Similar findings were also concluded in studies by Strand *et al.* (2013), Jahan *et al.* (2019), and Maes *et al.* (2017). In contrast, Harris (2020) literature review emphasized that half of the studies that performed grain size analyses found no correlation between MP abundance and grain size. Patterns of MP distribution and whether they are influenced by sediment characteristics or not, therefore, remain unanswered and intensify the gap in understanding MP behaviour in sediment.

It is critical to emphasize that catchment land use land cover (LULC) can influence MP prevalence in receiving water bodies, according to Chen *et al.* (2020) and Townsend *et al.* (2019). Specifically, Chen *et al.* (2020) conducted studies into MP distributions at the watershed scale and discovered the highest loads of MPs from the industrial areas, in contrast to the lowest MPs in the residential areas. Townsend *et al.* (2019) similarly noticed the abundance of plastic fragments decreased from industrial catchment to residential land use around urban wetlands. Nevertheless, there is no general agreement in the existing literature on the contribution of

different LULC types to MP concentrations in the aquatic environment, which requires further investigation (Chen *et al.*, 2020). Overall, there is limited research into the abundance and loading rates of MPs, and the impacts of LULC on them, in urban areas.

In this study, to investigate the role of SWPs as accumulators of MPs, we measured MP abundances, polymer compositions, and accumulation rates in the sediments of urban SWPs. We also analyzed the relative influences of catchment LULC, sediment properties, and the SWP's water flushing rate on MP abundances and accumulation rates in SWP sediments.

2.2 Materials and methods

2.2.1 Field sites, survey, and sediment sampling

Five SWPs with different catchment LULC were selected in the City of Kitchener, Ontario, Canada, for sediment sampling and analysis of MPs in the sediments. The information about the land use, design, and morphology of the five SWPs are provided in Table A-1. Four of the selected SWPs (Shirley, Wabanaki, Activa, and Zeller) were dredged within the 4 years prior to sediment sampling and the Bingemans pond was dredged 8 years prior (Table A-1). To collect the sediment samples, transects were formed within each pond from either the pond inlet or the pond outlet to the pond bank using wood posts, measuring tapes, and ropes (Figure 2-1). Along each transect, sediment depth measurements were performed with a measuring rod every 5 to 10 m.

At each transect within the SWPs, 2 to 3 sediment samples were collected using a Universal Corer (Aquatic Research Instruments) with clear polycarbonate core barrels (68 mm inner diameter). The cores were kept intact using extruding plugs for transferring them back to the laboratory and were stored at 5°C prior to analyses. The top 1 cm and bottom 1 cm sediment of each core were sliced and collected into 500 mL pre-weighed mason jars for bulk density

measurements (see 2.2). The rest of the core was collected into 1 L pre-weighed mason jars and homogenized for OC, grain size, and MP analyses (see 2.2, 2.3, and 2.4).

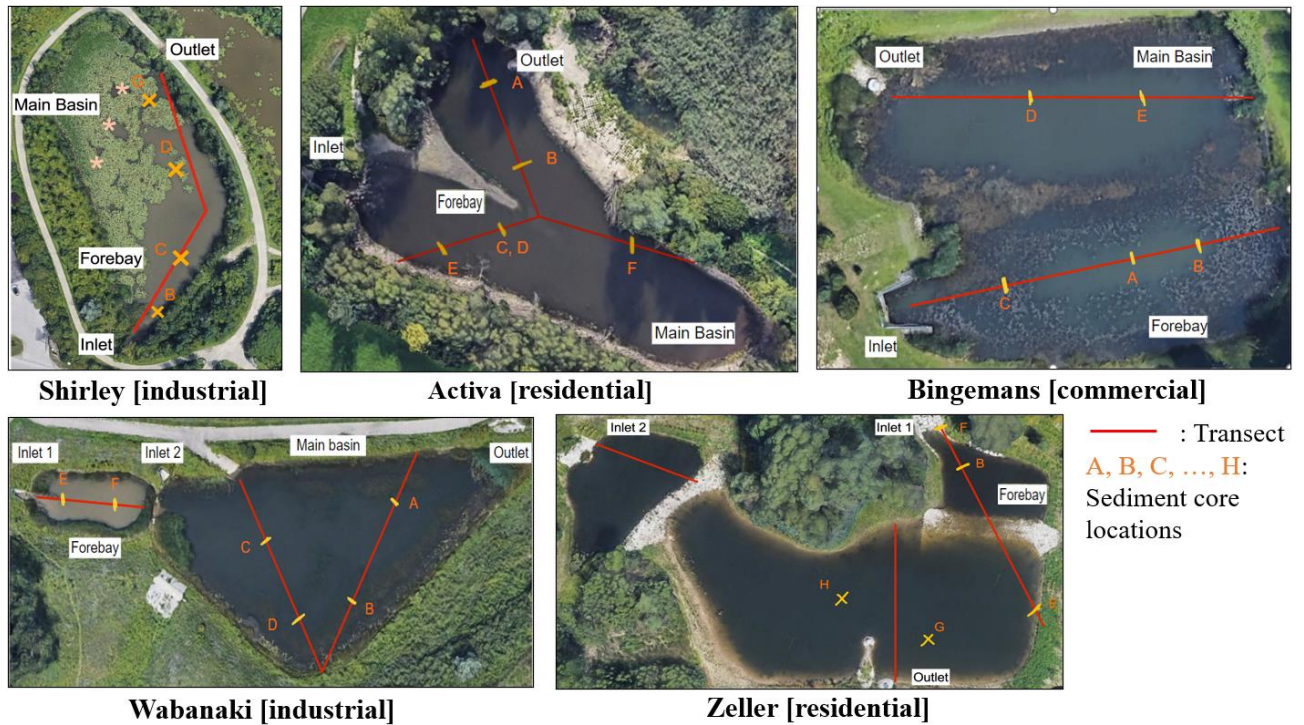


Figure 2-1: Layouts, transects, and relative locations of collected sediment cores at each pond's inlet forebay and main basin of five stormwater ponds in the City of Kitchener, Ontario, Canada, with industrial, commercial, and residential catchment land use.

2.2.2 Bulk density, organic carbon, and grain size analyses

The bulk density of the sliced top and bottom 1 cm of each pond's inlet forebay and main basin sediment samples were determined from the oven-dried mass (105°C for 24 h) and the original sliced sediment volume. Triplicate sediment aliquots from each pond's inlet forebay and main basin cores were used to determine OC using a CHNS Carlo Erba NA-1500 Elemental Analyzer (method detection limit, MDL: 1%). Sediment particle size distributions were analyzed using the pipette method (Gee and Bauder, 1986). The bulk density, average OC content, and

average particle size for each pond's inlet forebay and main basin sediments are summarized in Table 2-1.

Table 2-1: Average bulk density (g cm⁻³), OC (%), and grain size (μm) in each pond zone of each SWP.

Pond [Land Use]	Pond zone	Average bulk density (g cm ⁻³)	Average OC (%)	Average grain size (μm)
Shirley [Industrial]	Inlet forebay	1.2	3.27	361
	Main basin	0.8	2.49	230
Wabanaki [Industrial]	Inlet forebay	0.8	7.01	267
	Main basin	0.8	3.01	382
Activa [Residential]	Inlet forebay	0.7	5.23	469
	Main basin	0.5	4.67	158
Zeller [Residential]	Inlet forebay	0.3	1.54	176
	Main basin	1.7	2.35	308
Bingemans [Commercial]	Inlet forebay	0.6	4.47	204
	Main basin	0.4	5.60	246

2.2.3 Microplastic separation from sediment samples

The MP extraction process from sediment samples consisted of three steps: preliminary separation, density separation, and chemical digestion (Yu *et al.*, 2024; Prata *et al.*, 2019; Zhou *et al.*, 2020). Triplicate sediment aliquots of 2.0 g from each sediment core were weighed into 250 mL beakers and spiked with a known amount of MP beads of different sizes (Cospheric, polyethylene microspheres). Sediment samples were then oven-dried at 65°C for 72 hours. The

samples were later sieved through a 500 μm sieve and collected onto a 20 μm sieve during preliminary separation. The collected particles thus fell in the size range between 20 μm and 500 μm . The next step was density separation, a common technique used to separate MPs from sediment, where the sediment was soaked in a 60% ZnCl_2 solution with a density of 1.75 g cm^{-3} (denser than many polymers) and the MPs floated on the solution surface due to differences in density (Prata *et al.*, 2019; Uddin *et al.*, 2021). The density separation was repeated twice and then the collected particles were transferred into a 600 mL beaker with a thermometer and a stir bar for chemical digestion. At this step, wet peroxide oxidation in the presence of a Fe(II) catalyst was used to remove organic matter coatings from the particles following the procedure developed by the National Oceanic and Atmospheric Administration (NOAA) (Uddin *et al.*, 2021). For the wet peroxide oxidation, 20.0 mL of aqueous 0.05 M Fe(II) solution and 20.0 mL of 30% hydrogen peroxide (H_2O_2 , certified ACS grade) were added into the beakers and heated up on a hot plate. When the temperatures approached 75°C , the solution started bubbling, requiring the addition of Milli-Q water to avoid damage to the plastic particles. Afterward, the solution was left to cool down below 30°C and poured through a piece of nylon mesh (pore size 28 μm) for the collection of MP particles. The collected MP particles were then counted manually to assess the recovery of spiked standard particles using a stereomicroscope (Model: Leica EZ4 stereomicroscope). Recovery efficiency of the spiked standards ranged from 60%-100% and decreased with particle sizes (Figure A-1). Lastly, the MP particles collected on the nylon mesh were rinsed down a glass vial using ethanol (reagent grade, 70%) for storage.

2.2.4 Analysis of MP particle counts, shapes, sizes, and polymer types

Laser direct infrared (LDIR) imaging was used to analyse the polymer type, particle dimensions, and particle count in sub-samples of the extracted samples. LDIR imaging was

performed using a 8700 LDIR Chemical Imaging System (Agilent Technologies California, USA). For LDIR analysis, the extracted samples stored in the glass vials were dried in the incubator until all the ethanol was evaporated and a constant weight was achieved. A total of 5.0 mL isopropyl alcohol (reagent grade, 70%) was added to the vials to rinse down any particles on the walls of the vials. Particles in the sample were subsequently mixed and suspended, and 100 μ L of the sample was pipetted onto a clean Kevley slide for LDIR analysis. Analysis of MP compositions and size ranges was conducted by scanning the Kevley slide under the LDIR using the automated Agilent Clarity software (Cheng *et al.*, 2022). A hit quality index (HQI) of 0.7 was applied to classify whether particles were identifiable or not based on the matching of the particle's spectra with the library (Wright *et al.*, 2021; Whiting *et al.*, 2022) (Figure A-2). This HQI, which is derived from the first derivative algorithm, represents the match acceptance between particles spectra and library spectra (Ourgaud *et al.*, 2022). The higher the HQI is, the more reliable the particle identification is. In addition, the spectra of potential field and lab contaminants were also included in the LDIR spectra library and excluded during data analysis, which help ensuring that particles identified in the samples originated from the environment.

With the particles' dimensions provided by the LDIR, the particle size was determined by the average dimension between width and height, whilst the particle shape was classified using the aspect ratio (*i.e.*, the width/height ratio). Particles with circularity above 0.9 were classified as beads, particles with aspect ratios less than 4 were classified as fragments, and particles with aspect ratios of at least 4 were classified as fibers (Cheng *et al.*, 2022). Although Cheng *et al.* (2022) used a length:width ratio threshold of 3 to classify fragments versus fibers, we selected a ratio of 4 based on our own analysis of particle images (Figure A-3). The abundance of bead-shaped particles was close to 0 across the sediment samples in all five SWPs, and therefore, only

fiber and fragment concentrations were calculated and reported in this study. After characterization, the LDIR particle count results were corrected using the dilution factor and spike recovery to calculate the MP concentration (particles g⁻¹) in the SWP sediment samples.

2.2.5 Accumulation rates calculation

The sediment depth survey and information about when the ponds were last dredged were used to calculate the discrete linear sediment accumulation rates (cm year⁻¹) within the different sub-zones (*i.e.*, areal boxes) of the SWPs corresponding to different sediment depth measurement locations. The surface areas of the sub-zones were calculated using Google Earth Pro 7.3 (Google Inc., 2022). Each SWP's surface area was separated into a forebay and main basin zones by consulting engineering drawings and aerial photos of the SWPs (see an example pond in Figure A-4). The sub-zone surface area was multiplied by the linear sediment accumulation rate to calculate the volumetric sediment accumulation rate in each sub-zone. Then, the mean dry bulk density for each zone was used to calculate the sediment mass accumulation rates (kg year⁻¹). The sediment accumulation rates were then multiplied by the mean concentration of MP particles measured for that pond zone using the LDIR analysis (particles per gram sediment dry weight (dw); particles g dw⁻¹) to estimate the MP particle accumulation rates (for both fragment and fiber morphologies) (particle m⁻² year⁻¹). The sediment accumulation rates were also multiplied by the OC concentration for the cores collected in that pond zone to estimate the OC accumulation rates (kg m⁻² year⁻¹).

2.2.6 Statistical analysis

The relationships between MP concentrations and other sediment variables across all collected sediment cores in five SWPs were analysed using a Pearson correlation analysis. Correlations were considered significant if the ρ -value was less than 0.1. The variables

considered in the correlation analysis were including: pond morphology, sediment accumulation rate, OC content, grain size, imperviousness, impervious catchment-to-pond surface area (SA) ratio, MP fiber and fragment concentrations, and polymer type concentrations. The correlation analysis was performed and visualized in R 4.1.2 (R Core Team, 2020) using the factoextra package (Kassambara and Mundt, 2020).

2.3 Results

2.3.1 MP concentrations in SWP sediments

The commercial SWP Bingemans had the highest fragment concentration with $2.3 \times 10^8 \pm 5.5 \times 10^7$ particles kg dw⁻¹ ($\pm$ SD; standard deviation), followed by Wabanaki pond with $2.0 \times 10^8 \pm 3.5 \times 10^7$ particles kg dw⁻¹ (Figure 2-2a). The two residential SWPs Zeller and Activa had fragment concentrations of $1.9 \times 10^8 \pm 3.1 \times 10^7$ and $1.6 \times 10^8 \pm 5.3 \times 10^7$ particles kg dw⁻¹, respectively, whilst Shirley had the lowest fragment concentration among 5 SWPs with $8.8 \times 10^7 \pm 1.4 \times 10^7$ particles kg dw⁻¹. It is interesting to note that between the two industrial ponds, fragment concentration in Wabanaki pond was approximately 2 times higher than that in Shirley pond (Figure 2-2a). Meanwhile, the highest fiber concentration was also found in Wabanaki industrial pond with $4.5 \times 10^6 \pm 1.4 \times 10^6$ particles kg dw⁻¹ (Figure 2-2b). The Zeller pond had the second-highest fiber concentration among five SWPs, while fiber concentration in the only commercial SWP were relatively low compared to others (Figure 2-2b). Interestingly, fragment and fiber concentrations varied 3 to 4 times among five SWPs, and fragment concentrations were approximately 50 times higher than fiber concentrations for all ponds (Figure 2-2a,b).

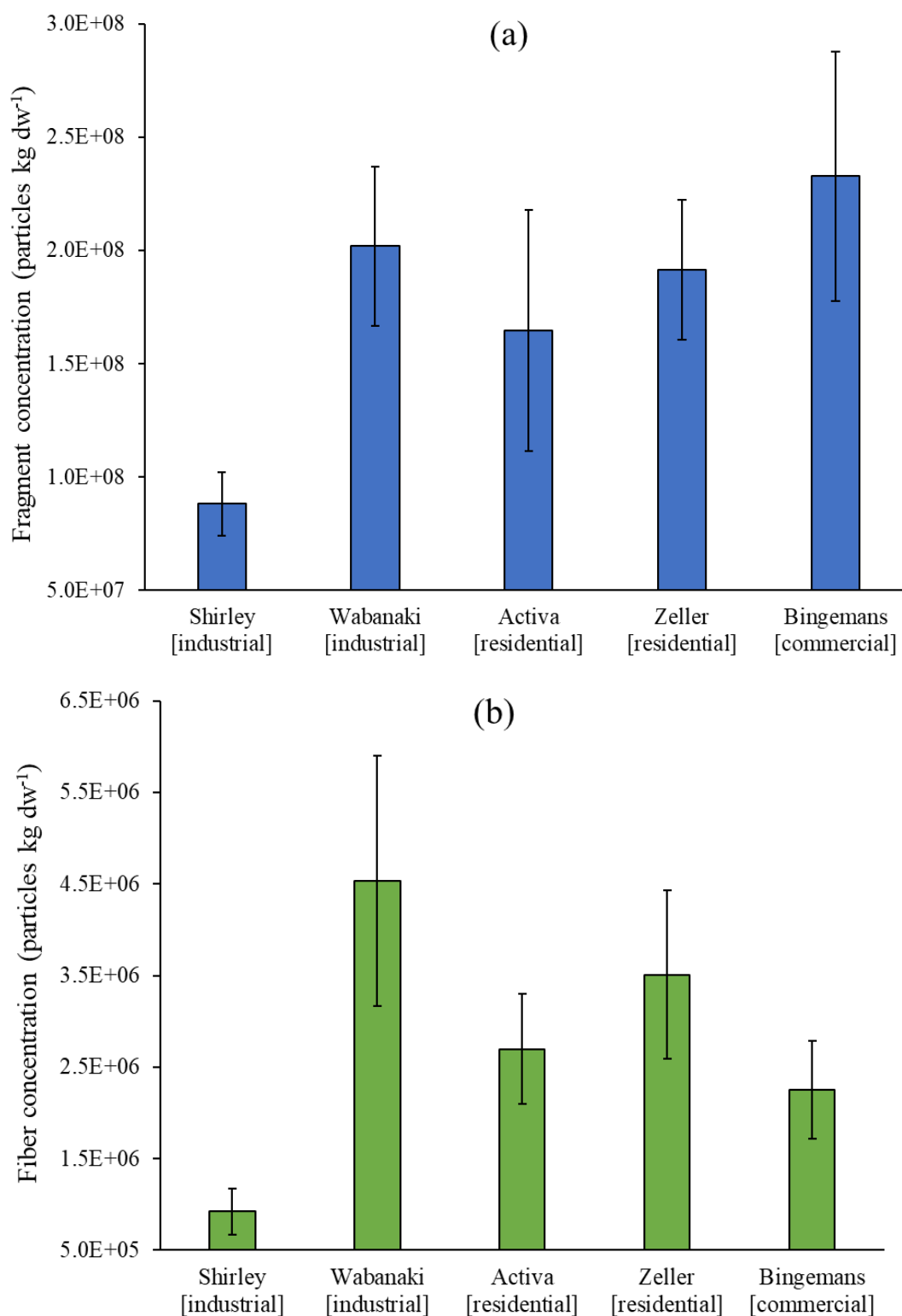


Figure 2-2: Microplastics fragment (a) and fiber (b) concentrations (particles per gram sediment dry weight (dw); particles g dw⁻¹) in five SWPs from LDIR analysis.

2.3.2 Sediment and MP accumulation rates

The area-normalized sediment and OC accumulation rates were substantially higher in the inlet forebay than in the main basin for most of the five SWPs (Figure 2-3a,b). The only exception was the residential Zeller SWP, with sediment accumulation rates of $13.9 \text{ kg m}^{-2} \text{ year}^{-1}$ in the inlet forebay and $31.3 \text{ kg m}^{-2} \text{ year}^{-1}$ in the main basin. The Zeller SWP also had the lowest sediment accumulation rate in both the forebay and the main basin among all of the five SWPs (Figure 2-3a). The two industrial SWPs had the highest sediment accumulation rates, with 241.0 and $298.9 \text{ kg m}^{-2} \text{ year}^{-1}$ in the inlet forebay for the Shirley and Wabanaki industrial SWPs, respectively (Figure 2-3a). The OC accumulation rates trends among the SWPs matched those of the sediment accumulation rates, with the Wabanaki SWP having the highest accumulation rates of $20.8 \text{ kg m}^{-2} \text{ year}^{-1}$ in the inlet forebay, followed by the Shirley SWP with $7.8 \text{ kg m}^{-2} \text{ year}^{-1}$ (Figure 2-3b). The residential Zeller SWP's OC accumulation rates were the lowest among all five SWPs and was the only SWP with less OC accumulated in its inlet forebay ($0.2 \text{ kg m}^{-2} \text{ year}^{-1}$) than in the main basin ($0.7 \text{ kg m}^{-2} \text{ year}^{-1}$) (Figure 2-3b). Both Aactiva and Bingemans SWPs had the approximately similar OC accumulation rates of $\sim 1.3 \text{ kg m}^{-2} \text{ year}^{-1}$ in the main basin (Figure 2-3b).

The fragment and fiber accumulation rates were considerably higher in the inlet forebay than in the main basin for four of the five SWPs (Figure 2-4a,b). The industrial Wabanaki SWP had the highest fragment accumulation rates in both the inlet forebay and main basin among the five SWPs, with a total annual fragment accumulation rate of $2.5 \times 10^{10} \pm 7.0 \times 10^9 \text{ particles m}^{-2} \text{ year}^{-1}$ for the whole SWP (Figure 2-4a). The Zeller residential SWP was the only SWP with lower fragment accumulation rates in the inlet forebay than in the main basin, with $1.1 \times 10^9 \pm 5.1 \times 10^8$ and $3.6 \times 10^9 \pm 6.8 \times 10^8 \text{ particles m}^{-2} \text{ year}^{-1}$, respectively. It is noteworthy that fragment

accumulation rates in the inlet forebay were 2 to 3 times higher than fragment accumulation rates in the main basin in both industrial SWPs (Figure 2-4a). Meanwhile, fragment accumulation rates in the main basin of both the Aactiva and Bingemans SWPs were relatively similar at around $3.2 \times 10^9 \pm 1.7 \times 10^9$ particles $\text{m}^{-2} \text{year}^{-1}$ (Figure 2-4a). A similar pattern was observed in the fiber accumulation rates, with the industrial Wabanaki SWP having the highest rates of $4.2 \times 10^8 \pm 1.6 \times 10^7$ particles $\text{m}^{-2} \text{year}^{-1}$ in the inlet forebay, followed by the Aactiva and Shirley SWPs with accumulation rates of $9.5 \times 10^7 \pm 2.5 \times 10^7$ particles $\text{m}^{-2} \text{year}^{-1}$ and $8.1 \times 10^7 \pm 5.3 \times 10^7$ particles $\text{m}^{-2} \text{year}^{-1}$, respectively (Figure 2-4b). Fiber accumulation rates in the inlet forebay of the residential Zeller SWP were four-times lower than those in the main basin, which was the opposite pattern than in the other SWPs. The Zeller SWP also had the lowest total accumulation rates of fibers among all five SWPs with only $8.5 \times 10^7 \pm 4.1 \times 10^7$ particles $\text{m}^{-2} \text{year}^{-1}$ (Figure 2-4b).

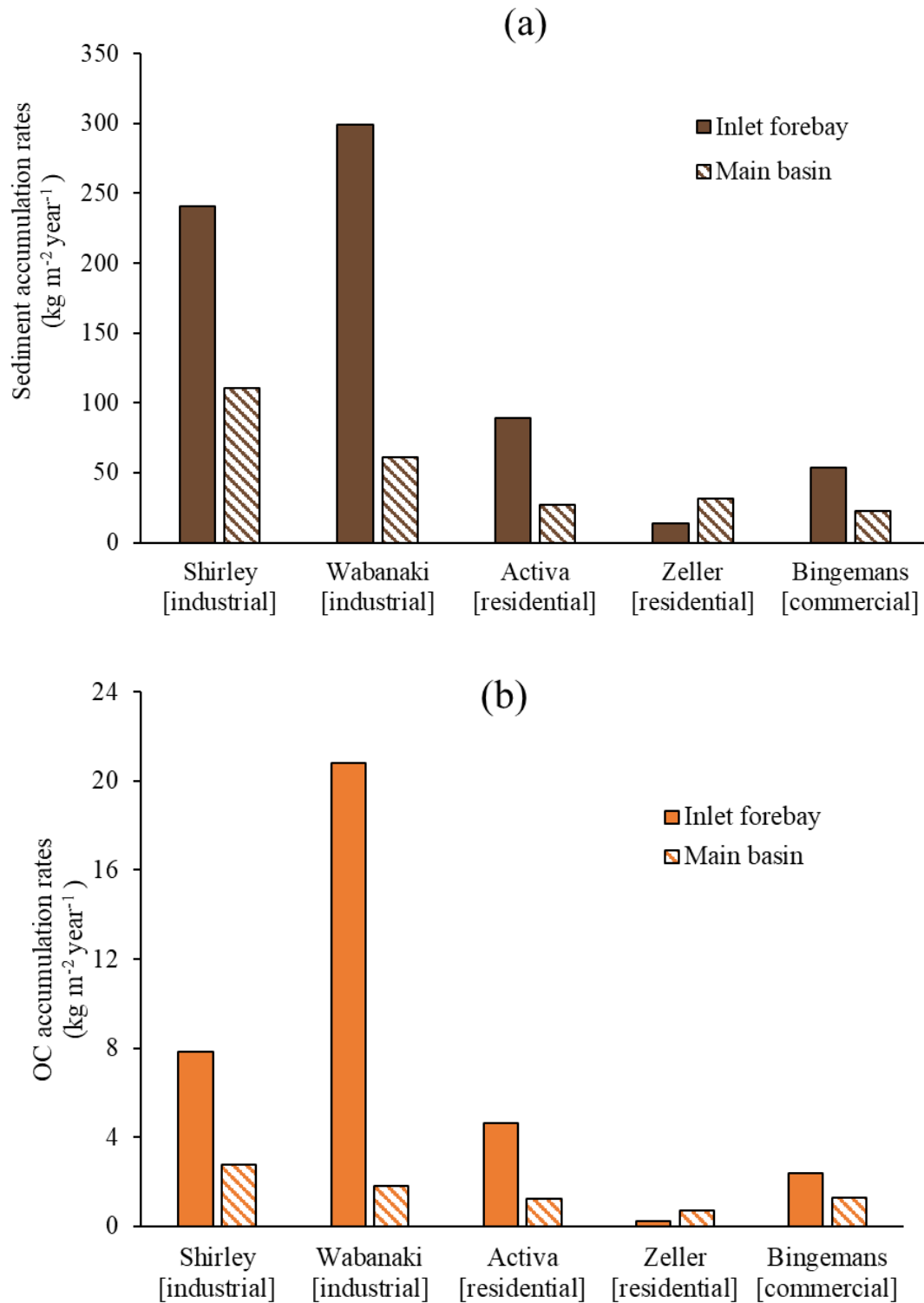


Figure 2-3: Sediment (a) and organic carbon (b) accumulation rates in the inlet forebay and main basin of the five SWPs

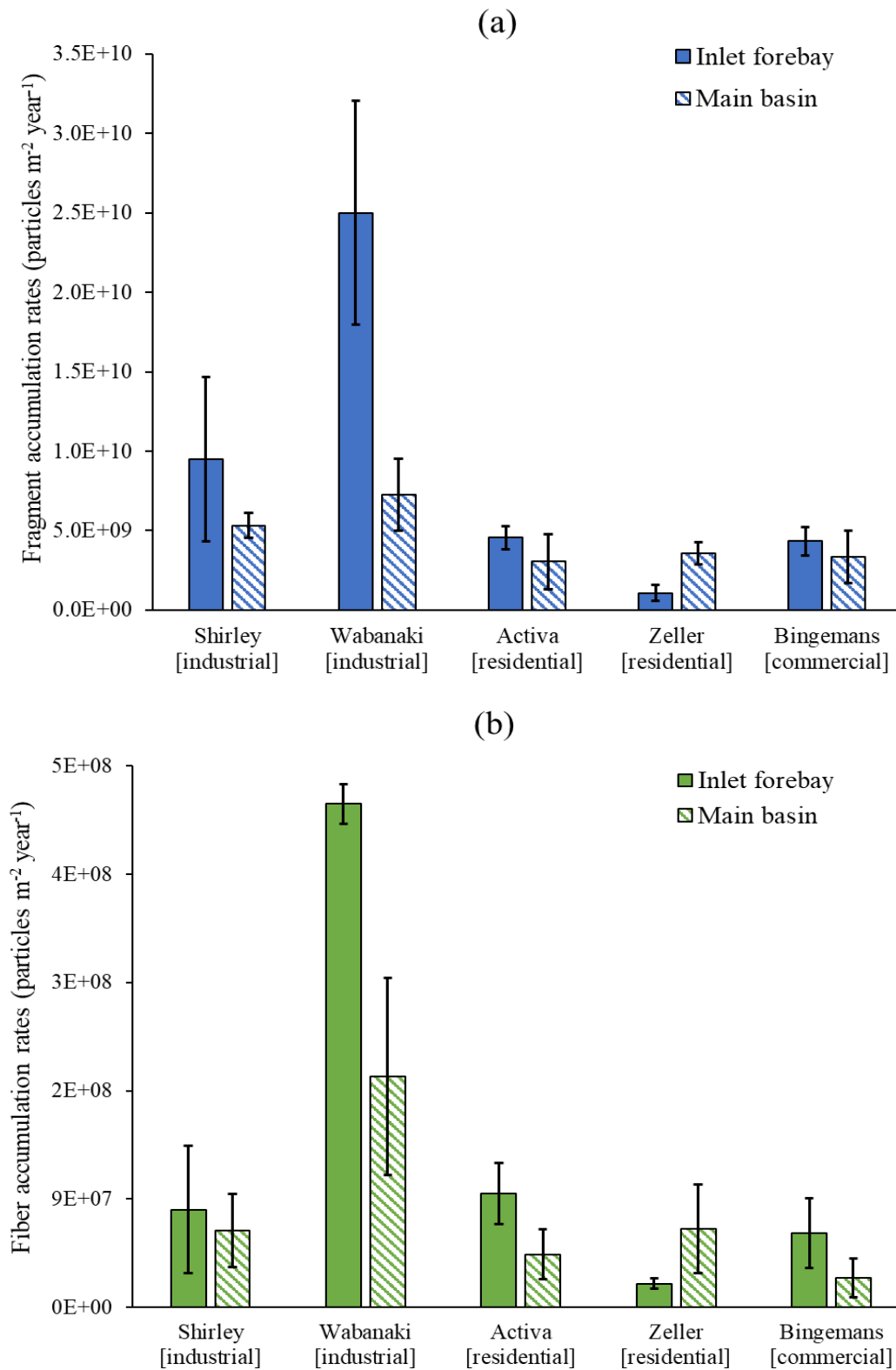


Figure 2-4: Fragment (a) and fiber (b) accumulation rates in the inlet forebay and main basin of five SWPs from LDIR analysis.

2.3.3 Controls on SWP sediment MP abundances and accumulation rates

The Wabanaki industrial SWP had the highest fragment and fiber accumulation rates with the smallest ratios between impervious catchment-to-pond surface area (SA) among all five SWPs (Figure 2-5a,b). Despite having a similar catchment-to-pond SA ratio to the Shirley pond, the residential pond Zeller had 3.2-times lower fragment accumulation rates and 1.7-times lower fiber accumulation rates. The other residential pond Activa with the highest catchment-to-pond SA ratio only has the third-highest fragment and fiber accumulation rates compared to other SWPs (Figure 2-5a,b). Meanwhile, Bingeman pond – the only commercial SWPs – had the second highest catchment-to-pond SA ratio of about 74.6 and considerably low fragment and fiber accumulation rates among five SWPs, with $7.7 \times 10^9 \text{ kg m}^{-2} \text{ year}^{-1}$ and $8.6 \times 10^7 \text{ kg m}^{-2} \text{ year}^{-1}$, respectively (Figure 2-5a,b). There were negative and non-significant relationship between fragment and fiber accumulation rates versus the ponds' impervious catchment-to-pond SA in most of the SWPs ($p > 0.05$) (Figure 2-5a,b).

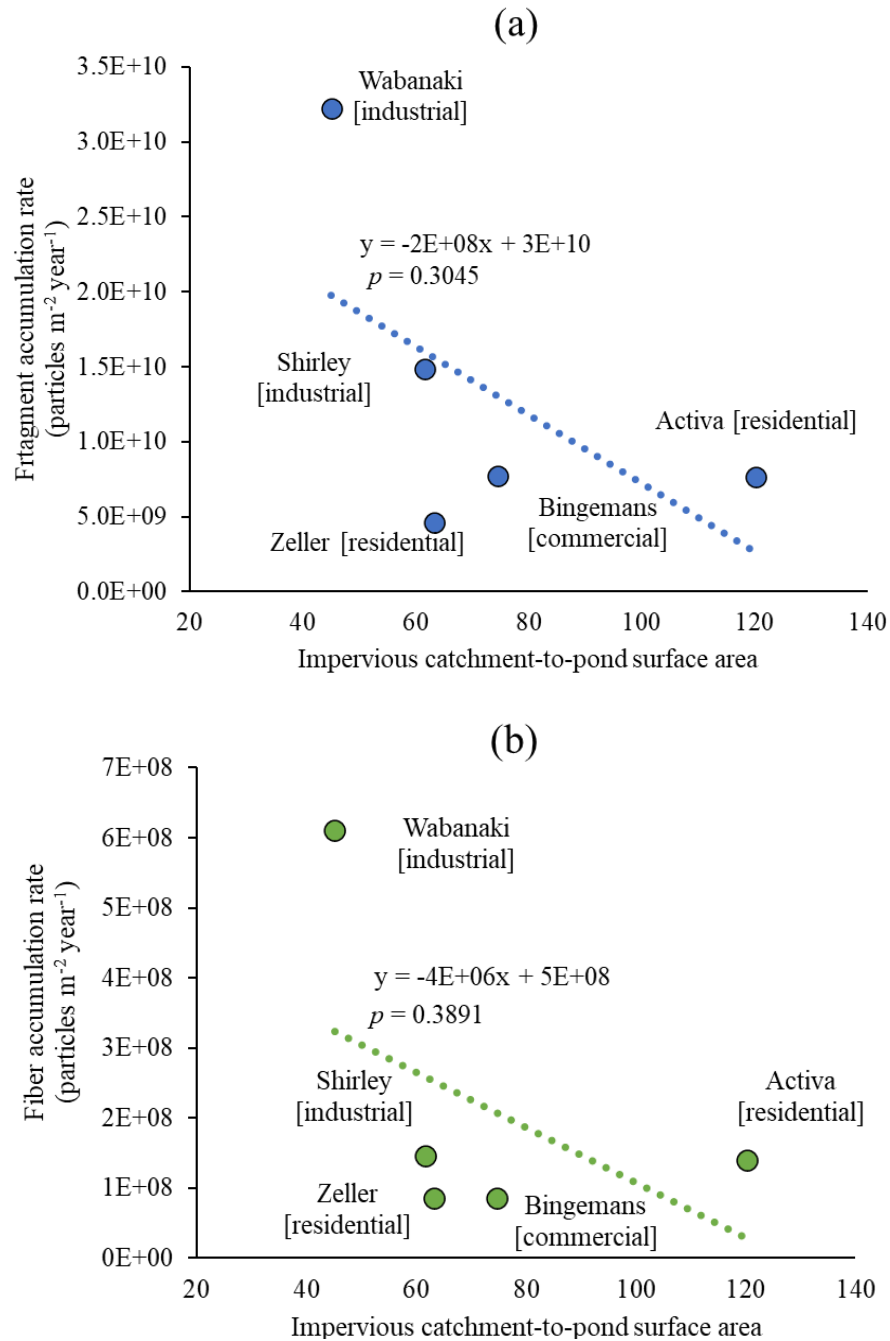


Figure 2-5: Fragment (a) and fiber (b) accumulation rates from LDIR analysis versus the ratio of impervious catchment-to-pond surface area of the five SWPs. The p -value for the regression coefficient indicates that the relationship is not statistically significant.

2.3.4 Polymer composition and abundance of MPs in 5 SWPs sediment

LDIR analysis of sediment samples across the 5 SWPs detected 8 major polymer groups: acrylonitrile butadiene styrene (ABS), polyamide (PA), polyethylene (PE), polyethylene terephthalate (PET), polypropylene (PP), polytetrafluoroethylene (PTFE), polyurethane (PU), and polyvinyl chloride (PVC). The polymer composition of MPs detected in sediment samples of five SWPs are presented in Figure 2-6 and the concentration of each polymer type with their density values are presented in Table A-2. Detected polymers with abundance less than 1% across all 5 SWPs were grouped into “Other” group (Figure 2-6). PA was the most abundant polymer, followed by PE and PTFE (Figure 6). PA concentrations were approximately 10-times higher than those of PE, the second-most abundant polymer, ranging from 10,365 particles g⁻¹ to 38,663 particles g⁻¹ (Table A-2). Meanwhile, polylactic acid (PLA), polymethyl methacrylate (PMMA), and polyoxymethylene (POM) were the least abundant polymers across the SWPs of all the polymers detected, with no particles detected at some SWPs (Figure 2-6, Table A-2).

Looking closely at the relative abundances of the 3 most abundant polymers - PA, PE, and PTFE - in each pond, it is noticeable that PA accounted for at least half of the polymer’s abundance detected in both the inlet forebay and the main basin of all of the SWPs, with the highest abundances observed in the two residential SWPs (Zeller and Active) and the industrial Wabanaki SWP (Figure 2-7). PA abundance also increased from the inlet forebay to the main basin for all 5 SWPs (Figure 2-7). Furthermore, PE had the highest relative abundance in the inlet forebay of Shirley and Bingemans ponds with 37% and 18%, respectively (Figure 2-7). The PE abundance then decreased to 22% and 13% in the Shirley and Bingemans’ SWPs’ main basins, respectively. PTFE, the polymer with the highest density detected in the SWP sediments,

was found in abundances of less than 10% across the SWPs. The only exception was in the Bingemans pond, where PTFE's relative abundance was 19% (Figure 2-7).

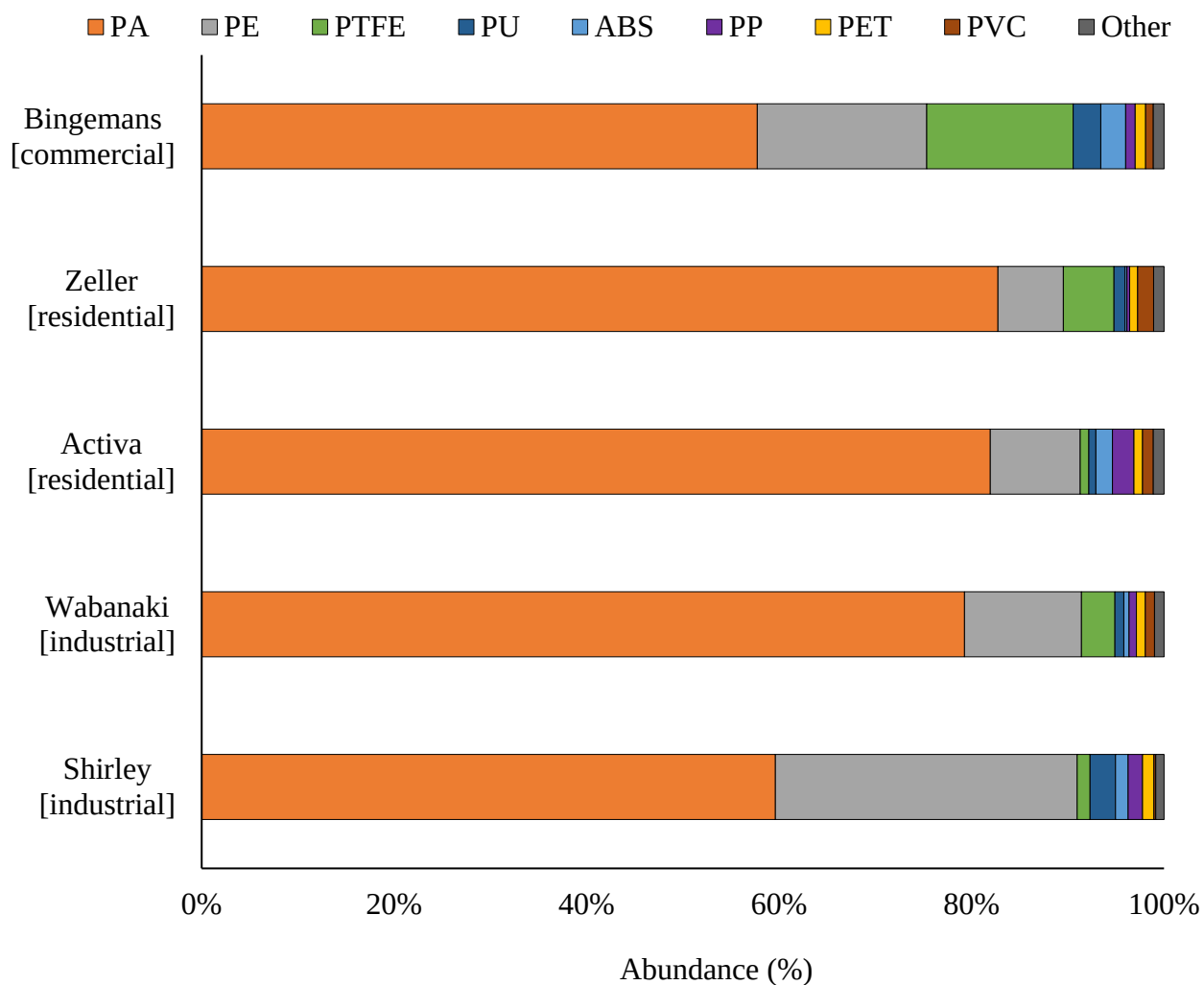


Figure 2-6: The polymer composition of MPs detected in sediment samples of five SWPs from LDIR analysis. Detected polymers with abundance less than 1% across all 5 SWPs are grouped into “Other” group.

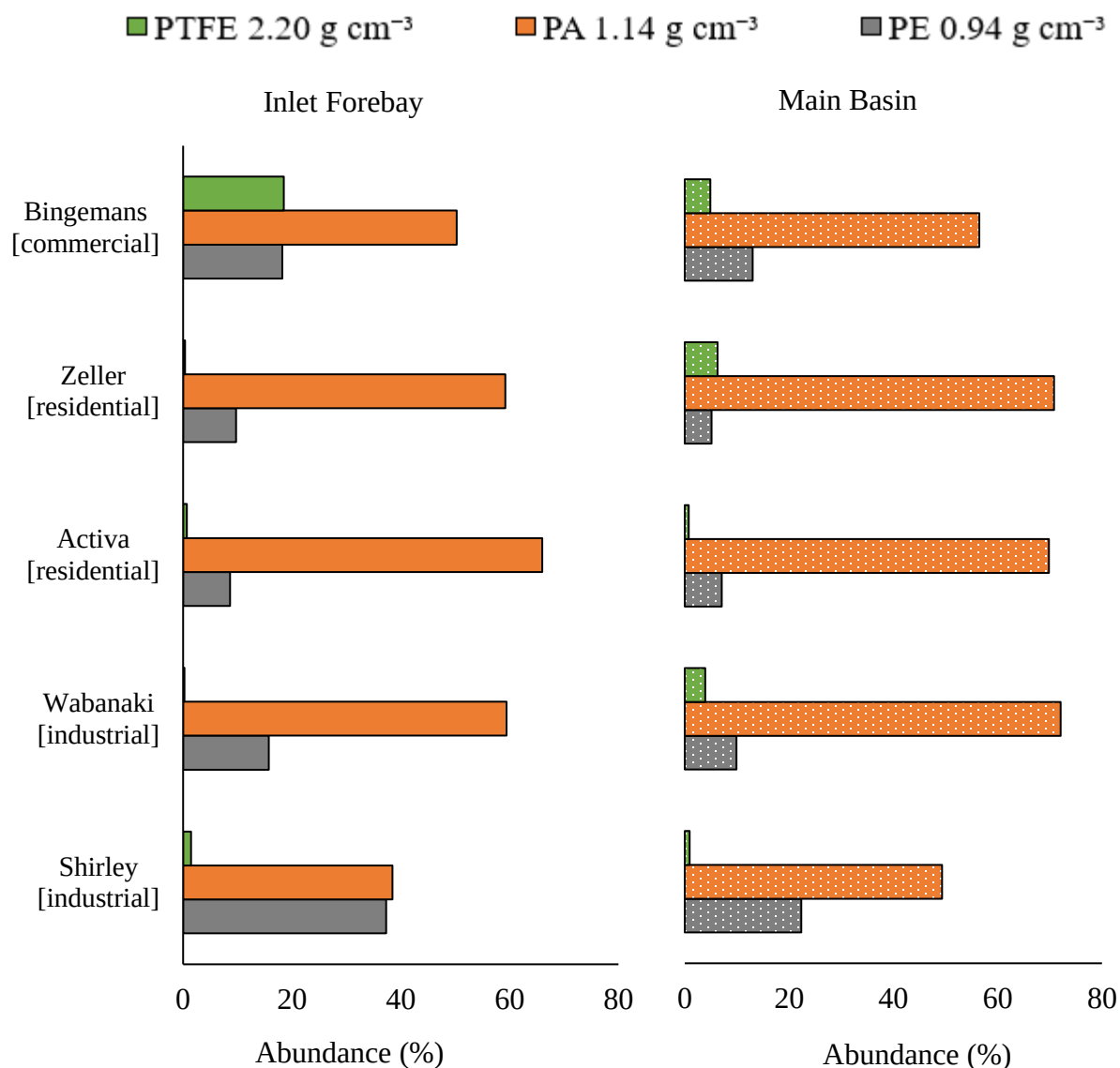


Figure 2-7: The abundance (%) of three polymer type: polytetrafluoroethylene (PTFE), polyamide (PA), and polyethylene (PE) in the inlet forebay and main basin of five SWPs from LDIR analysis.

2.3.5 MP particles size distribution in 5 SWPs sediment

MP particle sizes detected by the LDIR in sediment samples across 5 SWPS were categorized into 6 size ranges: <10 μm , 10-25 μm , 25-50 μm , 50-100 μm , 100-500 μm , and > 500 μm (Figure 2-8). It appeared that particles of 10-25 μm size class dominated in sediment

with abundance ranging from 35% to 50%, followed by 25-50 μm particles and <10 μm particles (Figure 2-8). MP particles that are larger than 100 μm were the least abundant in the pond sediment, accounting for only 6% of particle numbers, and no particles larger than 500 μm were detected (Figure 2-8). Moreover, particles that are smaller than 25 μm were higher in the inlet forebay compared to the main basin for most SWPs, with Activa SWP being the only exception (Figure 2-8). Activa pond's inlet forebay also has the highest proportion of particles in 50-100 μm and 100-500 μm size classes among all 5 ponds, with 14% and 6% abundance respectively. Meanwhile, Bingemans pond has the highest share of particles less than 25 μm in both the inlet forebay and main basin among all 5 SWPs (Figure 2-8).

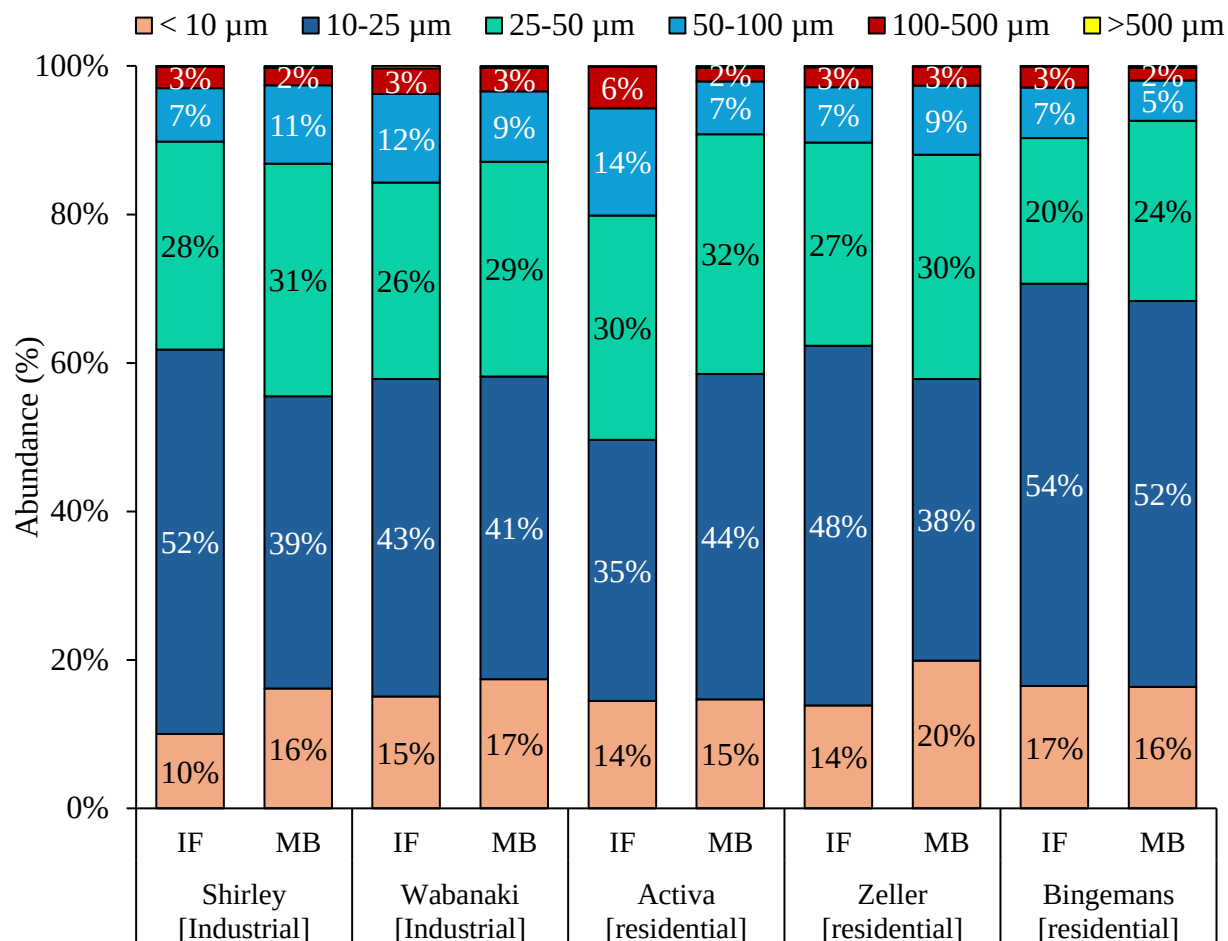


Figure 2-8: The abundance (%) of MP particles in different size classes in the inlet forebay (IF) and main basin (MB) of five SWPs from LDIR analysis.

2.3.6 Spatial variation of sediment, OC, and MP properties

Pearson correlation analysis between sediment and MP variables across collected sediment cores in five SWPs was performed and the results are presented in Figure 2-9. A statistically significant correlation ($\rho < 0.1$) between fragment, fiber, and PA concentration was noted. Furthermore, the ponds' imperviousness was positively correlated with OC content, fragment, and fiber concentration ($\rho < 0.1$). There was a positive correlation between imperviousness and the three polymer types. PTFE was statistically correlated to % clay of sediment, while PE concentration had a positive relationship with fragment and fiber concentration. Interestingly,

there was little to no correlation between grain size and other OC and MP characteristics (Figure 2-9).

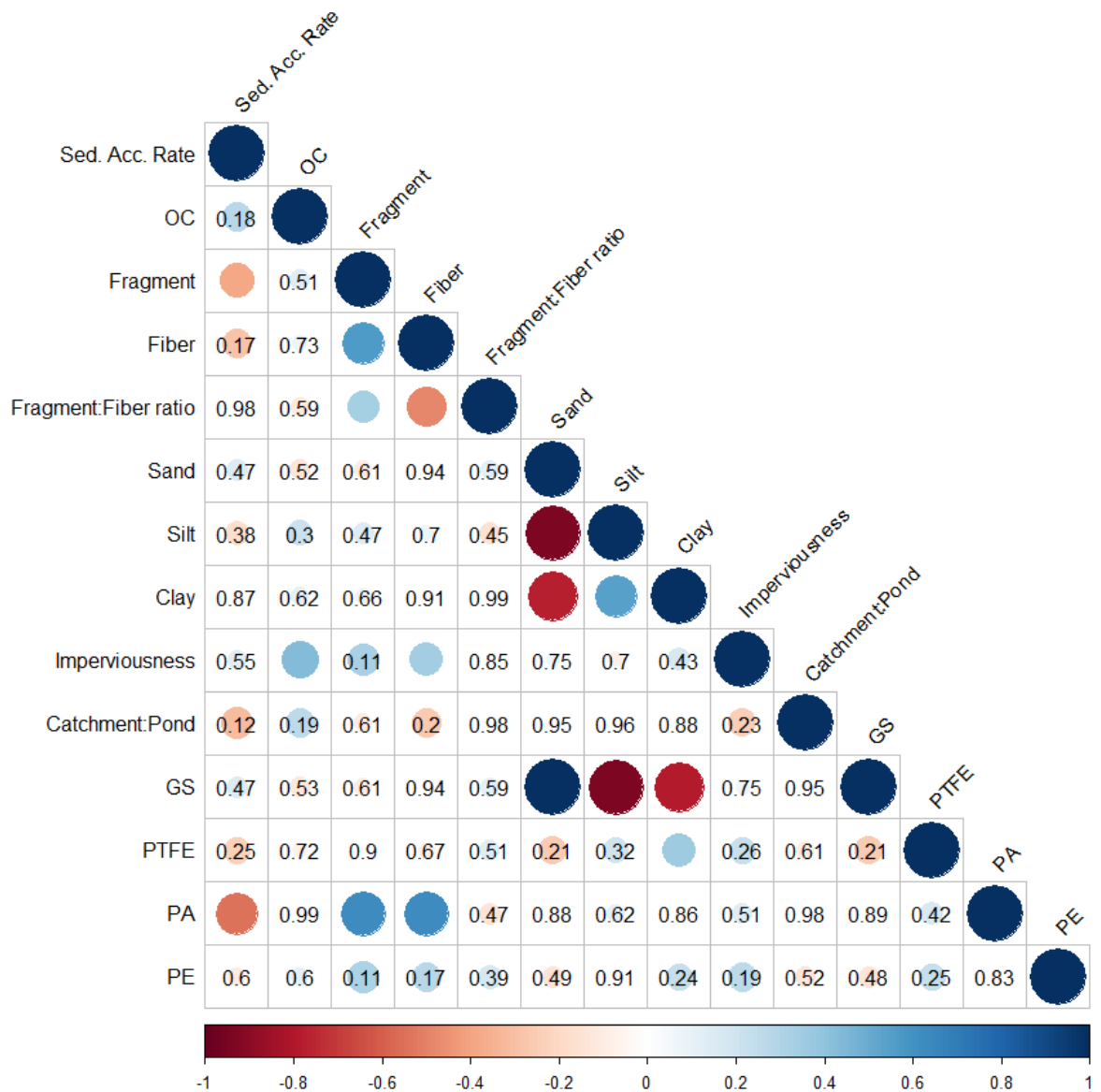


Figure 2-9: Pearson's correlation analysis between sediment and MP variables ($\rho < 0.1$). Size and color of a circle indicate the strength and direction (blue is positive, red is negative) of the correlation, respectively. For non-significant relationship, the ρ -value is displayed.

2.4 Discussion

2.4.1 MP concentrations and accumulation rates in SWPs sediments

The MP concentrations and accumulation rates observed in this study are in a substantially higher range than the concentrations reported for SWP sediments in previous studies (Brooks *et al.*, 2023; Liu *et al.*, 2019b; Olesen *et al.*, 2019). Both Liu *et al.* (2019b) and Olesen *et al.* (2019) reported MP concentrations of up to 1.3×10^6 particles kg^{-1} for SWPs draining industrial catchments, which was two orders-of-magnitude lower than MP concentrations observed in our industrial SWPs ($\sim 2.1 \times 10^8$ particles kg^{-1} ; sum of fragment and fiber concentrations converted to particles kg^{-1} ; Figure 2-2a,b). In addition, the highest fragment concentration found in sediment of the commercial Bingemans SWP (Figure 2-2a) can be the outcome of long-term sediment, and subsequently microplastics, accumulation, as that pond has not been dredged for 8 years. The relatively high MP concentrations were highest in the two residential SWPs (Activa and Zeller; Figure 2-2a,b) agreed with the findings in other studies. For example, Dikareva and Simon (2019) showed that river sediment MP concentrations increased with increasing residential land use.

The MP concentrations and accumulation rates observed for SWP sediments in this study are much higher than those reported for sediments in more natural systems (*e.g.*, rivers, lakes), which is expected given that SWPs are designed to remove pollutants such as MPs from urban runoff, which is enriched in MPs. For instance, 832 particles kg^{-1} were observed in sediments of the St Lawrence River in Canada (Crew *et al.*, 2020), and 360 to 1,320 particles kg^{-1} were observed in sediments collected from the Wei River in China (Ding *et al.*, 2019). In comparison, our average SWP sediment MP concentration across all five SWPs was 1.7×10^8 particles kg^{-1} , which is several order of magnitudes higher than that observed for these river sediments. The

several-magnitude difference in MP abundances in SWP than in the sediments in rivers and other natural systems can thus be ascribed to the higher catchment loads from urban catchments, highlighting the extent to which urban catchments outsize other land use types in contributing to MP pollution (Liu *et al.*, 2019b). It is also important to note that accumulation rates of MPs in freshwater environment has rarely been reported in existing literature, and hence, the values measured in this study reflected the earliest quantification of MPs in SWPs (Shruti *et al.*, 2021; Harris, 2020).

The higher fragment concentrations compared to fiber concentrations that is observed (50 times higher, on average, across all 5 SWPs; Figure 2-2a,b), has also been observed in studies by Moruzzi *et al.* (2020) and Liu *et al.* (2019b) in SWP systems. The higher abundance of fragments over other shapes of MPs suggests that particle shape is an important control on the distribution of MPs in sediment versus in the water column. In contrasts, studies that have analysed MPs in stormwater have found that fibers are the dominant shape (Smyth *et al.*, 2021), suggesting that fibers partition more into the water column whereas fragments partition more into sediments (Patterson *et al.*, 2020). Following Stoke's law, it has been suggested that MP shapes impact the drag force upon the particles, and consequently, the settling velocity of the particles into the sediment (Dietrich, 1982; Zhiyao *et al.*, 2008). As a result, MP shape can greatly influence particle mobility and dispersal behaviour in soils and sediments (Khatmullina and Isachenko, 2017; Shamskhany *et al.*, 2021).

The significant positive correlation that we observed between OC and fragment concentrations ($\rho < 0.1$, Figure 2-9) suggests that MP particles might be associated with OC, either physically or chemically, when being deposited into the sediment. This potential association between OC and MPs has also been observed in other studies (Enders *et al.*, 2019;

Ianuzzi *et al.*, 2024; Ling *et al.*, 2017; Strand *et al.*, 2013;). Additionally, both OC concentration and fragment concentration negatively related to sediment grain size, negatively related to % sand, and positively related to % silt and fiber concentration ($\rho < 0.1$; Figure 2-9), which suggests that fine-grained sediment may be facilitating MP sedimentation in these SWPs, as has been also suggested by other studies (Li *et al.*, 2020; He *et al.*, 2020; Sun *et al.*, 2021; Vermeiren *et al.*, 2021). Hence, it was not surprising that depositional areas with finer sediment and higher OC content favoured MPs accumulation, as Vianello *et al.* (2013) and Willis *et al.* (2017) also reported. Interestingly, fiber concentration exhibited no correlation to sediment grain size in this study, which is also supported by Ling *et al.* (2017)'s study as they found correlation between sediment grain size with MP particles concentration but not MP fibers. This pattern, therefore, refine the crucial impact of MP shapes on the particles' distribution pathway.

2.4.2 Sediment, MP, and OC accumulation rates in SWPs

Sediment and MP accumulation rates were considerably higher in the inlet forebay than in the main basin for most SWPs (Figures 2-3a and 2-4a,b). This trend was expected given that the forebays of SWPs are designed to accumulate most of the inflowing sediments and other particle-associated pollutants sourced from the catchment (Winston *et al.*, 2017). Other studies have observed similar decreasing trends in MP concentrations and/or accumulation rates with the distance from the inlet in similar systems such as detention ponds, treatment wetlands, and bioretention cells (Mbachu *et al.*, 2023; Ziajahromi *et al.*, 2020; Ashiq *et al.*, 2023). The one exception to this trend in our studied SWPs was the Zeller SWP, where lower sediment accumulation rates were observed in the forebay basin than in the main basin. The Zeller SWP's proximity to a busy road is a likely explanation for this difference. MPs and sediments produced by vehicle wear on the road are likely directly reaching the SWP's main basin via atmospheric

deposition and direct road wash off in this SWP whereas for the other SWPs, the inlet sewer is the major source of MPs, leading to higher accumulation rates in the forebay. These results highlight the need to measure MP concentrations in both the forebay and main basin of SWPs to be able to reconstruct total MP and sediment accumulation rates in SWPs.

Despite fragment and fiber concentrations varying by 3 to 4 times across the five SWPs and being highest in the commercial SWPs (Figure 2-2a,b), both fragment and fiber accumulation rates were highest in the Wabanaki industrial SWP (Figures 2-4a,b), followed by the 3 residential and commercial SWPs, which were similar. This trend matched the trend in the sediment accumulation rates between the SWPs (Figure 2-3a), highlighting that the sediment accumulation rate is the dominant variable driving the MP accumulation rates in all five SWPs. The negative relationship between fragment and fiber concentrations and sediment accumulation rates (Figure 2-9) further reflects the impact of the sediment accumulation rates, indicating that MP concentrations are being diluted by the accumulation of sediments in the SWPs with higher sediment accumulation rates. Meanwhile, the matching pattern between sediment and MP accumulation rates suggests that sediment and MP loads to SWPs from catchments are also closely correlated, and further, that catchment sediment loads, and the factors driving sediment loads, are important drivers of catchment MP loads. This close association between catchment and MP loads, as well as SWP sediment and MP accumulation rates, has also been observed by Saarni *et al.* (2023).

The SWP sediment accumulation rates observed were substantially higher (Figure 2-3) compared to those observed for other freshwater environments such as lake or river sediments. For instance, Turner *et al.* (2019) reported an average sediment accumulation rate of only 1.6 kg m⁻² year⁻¹ in an urban lake in the UK. Meanwhile, Hwang and Foster (2006) calculated a 32 kg

$\text{m}^{-2} \text{ year}^{-1}$ sediment deposition rate of rivers draining from a small urban catchment in the US, which was relatively similar to the sediment accumulation rate in Zeller SWP in this study, implying influence of urban population on catchment loading. Catchment LULC, hence, must be taken into consideration when estimating sediment as well as microplastic accumulation rates. Having said that, data on sediment and MPs accumulation rates in urban watersheds remain limited, and if documented, the inconsistent units reported for accumulation rates pose a challenge for results comparison across the literature (Stang *et al.*, 2022). Both studies by Harris (2020) and Uddin *et al.* (2021), thus, emphasized the need for microplastic quantitative data in $\text{g m}^{-2} \text{ year}^{-1}$ for a comprehensive review of MPs in sedimentary environments.

2.4.3 Influence of imperviousness and catchment-to-pond surface area ratio

Catchment characteristics are another factor affecting the delivery of MPs to SWPs, and therefore also the accumulation of MPs in SWP sediments. Schroer *et al.* (2019) moreover emphasized that the sediment accumulation rate was driven by impervious surface areas. In particular, the smaller the ratio of impervious catchment to SWP surface area (SA), the longer the water residence time (Egemose *et al.*, 2015). As a result, SWPs with a smaller catchment-to-pond SA ratio will have a smaller flushing rate and thus, more time for settling and partitioning between the water column and sediments for suspended sediments and MPs. For the same catchment sediment loading, for example, a SWP with a longer water residence time would be expected to have a higher area-normalized accumulation rate of bottom sediments, as illustrated by the negative Pearson correlation (Figure 8). Among the five SWPs studied here, the industrial Wabanaki pond had the smallest catchment-to-pond SA ratio and thus, it was expected that more MPs accumulated in the pond (Figure 5). Meanwhile, Brooks *et al.* (2023) and Liu *et al.* (2019b) also found that the catchment-to-pond SA ratio had a positive correlation with MP or other

contaminant concentrations in SWP sediments, and they also considered the catchment-to-pond SA ratio to be a proxy for the hydraulic loading to the SWP, and therefore also for the water residence time. These contrasting findings hence suggested that other catchment and pond characteristics might affect the MP deposition in the pond. For instance, industrial land use produces the highest sediment loading, leading to the highest MP accumulation rate, which was consistent with results reported by Townsend *et al.* (2019) and Bond *et al.* (2022). Chiandet and Xenopoulos (2016) also identified that pond morphological characteristics such as length: width and surface area: perimeter ratios are key controls on the ability of SWPs to trap suspended sediments and other contaminants found in particulate forms.

Meanwhile, for the two industrial SWPs and two residential SWPs with similar catchment-to-pond SA ratios, the sediment and MP accumulation rates were substantially higher in the industrial SWPs, highlighting the impact of catchment LULC on sediment and MP accumulation rates (Figure 2-5). This suggests that industrial LULC produces higher sediment and MP emissions, resulting in the higher loads to the SWPs and higher accumulation rates in the SWP sediments, which is consistent with the results of Townsend *et al.* (2020). This finding indicates that it will be important to design SWPs that are sufficiently large (*i.e.*, have moderate to low catchment-to-pond SA ratios) for SWPs draining industrial catchments to ensure that the high MP emissions from the industrial LULC are being controlled using SWPs to retain MPs in their sediments, thereby mitigating the emission of MPs to downstream systems.

The correlation analysis indicated that catchment imperviousness also exerted a major control on the sediment, MP and OC concentrations and accumulation rates (Figure 2-9). Imperviousness was found to be significantly positively correlated with OC and fiber concentrations, while having non-significant but positive relationships with sediment

accumulation rate, fragment concentrations, and the concentrations of the dominant polymers (Figure 2-9). Stormwater runoff can wash materials, including MPs and sediments, off impervious surfaces and carry urban-derived MPs into water bodies (Goehler *et al.*, 2022). As a result, SWPs draining catchments with higher imperviousness are expected to accumulate more MP particles. Strong positive correlations between sediment accumulation rates and imperviousness have also been observed in other studies (Schroer *et al.*, 2019), emphasizing the significant role of imperviousness in driving sediment wash off. Catchment land use and land cover, hence, must be taken into consideration when estimating sediment as well as MP accumulation.

2.4.4 MPs polymer composition and abundance in 5 SWPs sediment

Previous studies of MP occurrence in SWP sediments have often detected PP as the most abundant polymer, with relative abundances of around 50%-60% reported (Brooks *et al.*, 2023; Liu *et al.*, 2019a; Olesen *et al.*, 2019). In this study, PP appeared in the SWP sediments, yet its concentration was relatively low compared to that of the other polymers (Table A-2). Meanwhile, the overwhelming abundance of PA found in the five SWPs could be a signal of the impacts of anthropogenic disturbance to the ponds (Sait *et al.*, 2021). The appearance of PTFE in sediment has been documented previously in marine environments but rarely in urban areas (Uddin *et al.*, 2021). The considerable abundance of PTFE - a polymer used for coating materials - in the Bingemans SWP can likely be ascribed to the use of PTFE in the nearby water park and amusement centre. These findings point to the important role that surrounding human activities and land use in determining the occurrence of different polymer types in SWPs.

While the typical paradigm for particle settling in SWPs is that denser particles, including denser MPs, should settle out first and accumulate in the sediments closer to the inlet, we

observed the opposite: lighter polymers such as PE decreased moving away from the pond inlet whereas heavier polymers such as PA increased towards the pond outlet (Figure 2-7). This result suggests that there are other mechanisms influencing the transport, deposition, and accumulation of MPs in the SWP sediments. For instance, the correlation between all polymer concentrations and the sediment grain size fractions (Figure 2-9) suggests that attachment to fine-grained sediments could be contributing to MP particle accumulation. Other studies have also found surprisingly high abundances of low-density polymers in sediments of SWPs and other water bodies and have suggested attachment to fine-grained sediments or biofilm growth and aggregation as possible mechanisms (Liu and Vianello *et al.*, 2019; Hale *et al.*, 2020). These results highlight how multiple variables, including MP particle morphology, MP polymer type, and sediment characteristics play critical roles in shaping the fate of MPs in SWPs (Waldschläger and Schüttrumpf, 2019).

2.4.5 MP particles size distribution in 5 SWPs sediment

MP particles have been detected in freshwater sediment in various size ranges with widespread prevalence. In Liu *et al.*'s (2019b) investigation of MPs in sediment in Danish SWPs, the researchers chose 2 mm as an upper limit for particle sizes and reported uneven distribution of particle numbers in different size ranges. For instance, Liu *et al.* (2019b) noted only 5.3% of their particles were larger than 250 μm , while smaller particles presented in higher concentrations in those SWPs. These findings therefore supported the dominance of particles less than 25 μm in this study, as well as the absence of particles larger than 500 μm due to the upper cut-off size in our extraction method (See 2.2.3; Yu *et al.*, 2024). Similarly, Mani *et al.* (2019) expected the size distribution to shift toward the small size class owing to the breakdown of plastic pieces in the environment, resulting in the 96.3% abundance of particles smaller than 75

µm in the sediment of the Rhine river. Microplastic abundance, hence, tends to decrease with increasing particle size, and MP particles smaller than 100 µm are predominant in freshwater sediment (Corcoran *et al.*, 2015; Yang *et al.*, 2021). The irregular proportion of particles in each size class between 5 SWPs further signalled the potential impact of surrounding land use on MP occurrence in the pond sediment. Previous research has found a relationship between catchment characteristics and the median size of MP particles in freshwater sediment as well (Iannuzzi *et al.*, 2024; T. Wang *et al.*, 2021). MPs size distribution found in this study thus emphasized the striking prevalence of small-sized MPs and underlined potential influence of nearby anthropogenic activities on MP occurrence in SWP sediment.

Furthermore, it is essential to keep in mind that we expected particles larger than 20 µm only due to the lower cut-off size in our extraction technique (See 2.2.3; Yu *et al.*, 2024), yet there is an overwhelming abundance of particles in <10 µm and 10-25 µm size class. This pattern could be explained by MP aggregation being broken down during organic matter removal, resulting in particles smaller than the size limit (T. Wang *et al.*, 2021). Mani *et al.* (2019) also ascribed the dominance of small MP particles in sediment bed to the biofouling of MPs, as MP aggregates not only increased particles' density and sedimentation probability but also enhanced attachment between MPs and organic matters. Having said that, the diversity in sampling and extracting techniques, as well as detection methods, have given rise to a wide range of MP size distribution with varying abundance, challenging comparison across existing literature (Liu and Vianello *et al.*, 2019; Yang *et al.*, 2021). As Lu *et al.* (2021) indicated, the current smallest size of detected MPs can be even smaller with technological innovation, signifying an urgent need for standardizing methods and reporting MP particle sizes.

2.4.6 LDIR and microscope counting comparison

A popular approach to the physical characterization and counting of MP particles is visual identification by microscope, as it is a convenient and low-cost method (Shim *et al.*, 2017). This method, however, has several drawbacks, including a high possibility of false positives and missing small-sized particles (Hidalgo-Ruz *et al.*, 2012). Yang *et al.* (2021) emphasized that the size limitation of microscopy challenge visual detection of MP particles, especially for particles smaller than 100 μ m, while Lenz *et al.*'s (2015) study of visual detection indicated that false positive identification increases with decreasing MP particle sizes. In this study, we compared the total MP particle count using a microscope and LDIR imaging. Under the microscope, fragments were identified as any shiny, transparent, or irregular-shaped particles, while fibers were identified as particles with elongated and thread-like shapes. Criteria for MPs' physical characterization using the LDIR were discussed above (See 2.2.4). From the stereomicroscope, it appeared that there was an overwhelming abundance of fragments, while fibers were more difficult to identify due to their transparent nature (Figure A-5a,b). It was also impossible to account for small particles even when the microscope lens reached a maximum of 35X magnification (Figure A-5a,b). As a result, total MP counts from the microscope were only 1% of LDIR particle counts (Figure A-6a). Furthermore, Isobe *et al.* (2019) performed an interlaboratory comparison and emphasized that visual detection without any spectroscopic technique was only accurate for MPs larger than 2mm. Kotar *et al.* (2022) and Lusher *et al.* (2020) later suggested that the acceptable lower size limit for particle detection with high accuracy using microscopy and visual detection is 200 μ m. We subsequently summarized the LDIR particle count for particles with either width or height above 200 μ m and compared them with microscope counting. Afterwards, the particle counts between the two methods became

comparable at a 1:2 ratio for five SWPs (Figure A-6b). This comparison thus has reinforced that visual identification using stereo microscopy can be biased and significantly underestimate the number of microplastics in a real environment (Zarfl *et al.*, 2019).

2.5 Conclusions

In this study, MP concentrations in sediment of SWPs with different land use varied 3 to 4 times with fragments being the dominant particles. OC content and sediment grain size were found to correlate with MP concentrations of fragment and fiber in SWP. Meanwhile, the matching pattern between sediment, OC, and MP accumulation rates suggested that sediment accumulation rate is the driving force of MP deposition in SWPs. Industrials SWPs with the highest sediment burial rates hence had the highest MP accumulations. Ponds' imperviousness and catchment-to-pond SA ratio significantly impacted sediment loadings as well as MPs concentration in the SWP sediments. Polymer type and size distribution data further implied the influence of catchment land use on MP accumulation, and therefore, it is highly recommended that catchment, sediment, and MP particles' characteristics should be examined together on how they influence MP deposition mechanism.

Controls on Microplastics Accumulation in Stormwater Runoff

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3 Controls on Microplastics Accumulation in Stormwater Pond Runoff

3.1 Introduction

Microplastics (MPs), or plastic particles smaller than 5 mm, have been a pervasive pollutant in the aquatic environment due to their small size, degradation-resistant nature, and potential to release additives and contaminants (Cole *et al.*, 2011; Liu and Vianello *et al.*, 2019; Turner *et al.*, 2019). With high durability, MPs can remain in the ecosystem for thousands of years, and thus, pose a significant threat to aquatic organisms (Tibbetts *et al.*, 2018). It is further important to note that the majority of MPs in the marine environment originated from terrestrial sources, especially from urban areas with high population density (Bläsing and Amelung, 2018; de Jesus Piñon-Colin *et al.*, 2020; Guo *et al.*, 2020; Li *et al.*, 2016). Generated from anthropogenic activities, land-based MPs can enter other environmental compartments via several pathways such as atmospheric deposition and wastewater (Cho *et al.*, 2023). Among these pathways, stormwater runoff has been a major carrier of MPs from urban areas to downstream water bodies. In particular, stormwater runoff is excessive surface water overflowing the land covers during a rainstorm or snowmelt event, which can transport MPs from the terrestrial environment and introduce them to the aquatic ecosystem (Saraswat *et al.*, 2016; Werbowski *et al.*, 2021; Yang and Lusk, 2018). The concerns for MP pollution in stormwater thus grow exponentially (Shruti *et al.*, 2021). For instance, Boucher *et al.* (2019) estimated MPs discharged into the Lake Geneva basin and pointed out that stormwater runoff contributed much more MPs than other transport pathways. Likewise, Cho *et al.* (2023) found a significantly higher MPs concentration

in stormwater runoff than in nearby wastewater effluent, emphasizing the dominant role of stormwater in facilitating MPs transportation into the aquatic environment.

To minimize MPs input into downstream water bodies, stormwater runoff is generally drained into stormwater retention ponds (SWPs), where the particles can either float in the water column or deposit into sediment depending on their properties (Kaiser *et al.*, 2017; Waldschläger *et al.*, 2020). More specifically, the polymer density can determine the environmental fate of the MP particles, as polymers less dense than water ($<1 \text{ g cm}^{-3}$) (*e.g.*, polyethylene (PE), polypropylene (PP)) might float on the water surface. For instance, Liu and Olesen *et al.* (2019) and Liu and Vinello *et al.* (2019) investigated MP accumulation in water and sediments of seven stormwater ponds with different land use in Denmark, in which the results indicated low-density plastic tends to float in the water column while heavier plastic particles readily deposit into the sediment. In addition, MPs' shape can influence their settling behaviour, and thus, their fate and transport in the environment (Waldschläger and Schüttrumpf, 2019). In particular, Goral *et al.* (2023) tracked the settling velocities of MP particles with various shapes and highlighted that particles tend to settle in a direction that is perpendicular to their surface area to minimize resistance, which supported Ballent *et al.*'s (2016) expectation of irregularly shaped fragments being readily retained by sediment. Nguyen *et al.*'s (2022) experiment further document that fibers settle considerably slower than fragments and pellets and the settling velocity varies with the fibers' length. As a result, MP fiber predominantly remains in the water column while MP fragments tend to deposit into the sediment bed in stormwater ponds (Ashiq *et al.*, 2023; Dai *et al.*, 2018).

It is important to note that the partitioning of MPs distribution between the water column and sediment bed in the freshwater environment remained unresolved, as the transport

mechanism of MPs can be affected by several other factors (Lu *et al.*, 2023; Rezaei *et al.*, 2019). According to Darabi *et al.* (2021), wind forces enable long-distance transport for buoyant MP while water movements can influence MPs floating in the water column. Xia *et al.* (2021), moreover, emphasized that disturbance induced by water inputs can resuspend the sediment bed, resulting in the redistribution of MP particles. This vertical mixing within a water body thus complicates the distribution mechanism of MP, as it is influenced by particle size, shape and density (Constant *et al.*, 2023; He *et al.*, 2021). Additionally, MPs with high residence time in the aquatic environment are particularly subjected to biofouling, aggregation, and fragmentation due to weathering, all of which can impact the particles' morphology, and subsequently, their fate in the environment (Hidalgo-Ruz *et al.*, 2012; Li *et al.*, 2020).

Besides MP particles' properties, catchment land use and land cover can substantially influence MP accumulation in urban watersheds (Lenaker *et al.*, 2019). By investigating microplastics in stormwater ponds in Denmark, Liu and Olesen *et al.* (2019) noticed that commercial and industrial ponds released more MPs than residential catchments. Similarly, Chen *et al.* (2020) conducted studies into MP distributions at the watershed scale and discovered the highest loads of MPs from the industrial areas, in contrast to the lowest MPs in the residential areas. Moreover, Tanentzap *et al.* (2021) document a two-fold increase in MP concentration in catchments with decreased forest covers, which supported Townsend *et al.*'s (2019) findings. Having said that, there is no general agreement in the existing literature on the contribution of different land use types to MP concentrations in the aquatic environment (Chen *et al.*, 2020). Considering the negative damaging impacts of MPs in urban areas, their alarming abundance in the environment, and the knowledge gap in MPs' behaviour from source to sink, research into MPs occurrence in urban areas with different land use and land cover, therefore, is much needed.

In this study, we assessed the variability in MP types, sizes, and abundances within SWP water and the impacts of land use type on MP occurrence. We also investigated the partitioning in MP morphology, polymer composition between sediment and water within and between SWPs.

3.2 Materials and methods

3.2.1 Field sites, survey, and water sampling

The same five SWPs that were selected for sediment sampling were also sampling sites for water sampling and analysis of MPs in the water column (See 2.2.1 in Chapter 2; Table A-1). The grab water samples were collected every two months near each pond's outlet from February 2023 to December 2023 using 2×1.9 L glass mason jars. The bottles are rinsed three times with the pond water, filled up with samples, and capped underwater to avoid risks of contamination from the atmosphere (Barrows *et al.*, 2017; Olesen *et al.*, 2019). Afterwards, the bottles are transferred back to the lab and stored at 5°C prior to extraction and analysis.

3.2.2 Microplastic separation from water samples

The MP extraction process from water samples consisted of three steps: preliminary separation, density separation, and chemical digestion (Prata *et al.*, 2019; Zhou *et al.*, 2020). The samples were spiked with a known amount of MP beads of different sizes (Cospheric, polyethylene microspheres) and later filtered through a 500 μm sieve stacked on a 20 μm sieve during preliminary separation. The collected particles thus fell in the size range between 20 μm and 500 μm . The exact water volume of the sample was recorded using a graduated cylinder. The following steps, including density separation and organic matter removal, were performed using the same procedure as for sediment samples (see 2.2.3 in Chapter 2; Yu *et al.*, 2024). The collected MP particles were then counted manually to assess the recovery efficiency of spiking

method using a stereomicroscope (Model: Leica EZ4 stereomicroscope) (Figure A-7). In general, spiked particles recovery decreased with particles sizes (Figure A-7). Lastly, the MP particles collected on the nylon mesh were rinsed down a glass vial using ethanol (reagent grade, 70%) for storage.

3.2.3 Analysis of MP particle counts, shapes, and polymer types

Laser direct infrared (LDIR) imaging spectroscopy was used to analyze the polymer composition, particle dimensions, and particle count in the extracted samples. For water samples, the sample preparation as well as criteria selected for data analysis using LDIR imaging followed the similar protocols as for sediment samples (See 2.2.4 in Chapter 2). The abundance of bead-shaped particles was close to 0 across the water samples in all 5 SWPs, and therefore, only fiber and fragment concentrations were calculated and reported in this study. After characterization, the LDIR particle count results were corrected using the dilution factor and spike recovery to calculate the MP concentration (particles L^{-1}) in the SWP water samples. Polymer composition and size distribution results for SWP water samples were then plotted against the SWP sediment samples' results (See 2.2.4 and 2.2.5 in Chapter 2) for more detailed comparison.

3.2.4 Statistical analysis

The relationships between MP concentrations throughout the whole year 2023 and other pond variables were analyzed using a correlation analysis. Precipitation and wind speed data was also retrieved from Waterloo weather station and included in the correlation analysis (Environment and Climate Change Canada, 2024). Correlations were considered significant if the p -value was less than 0.1. The variables considered in the correlation analysis were: catchment imperviousness, impervious catchment-to-pond surface area (SA) ratio, MP fiber and fragment concentrations, MP fragment to fiber ratio, polymer type concentrations, amount of

precipitation during the month, the week before sampling and on sampling day, and wind speed the week before sampling day and on sampling day. The correlation analysis was performed and visualized in *R* 4.1.2 (*R* Core Team, 2020) using the *factoextra* package (Kassambara and Mundt, 2020).

3.3 Results

3.3.1 MP concentrations in SWPs water

The highest fragment concentration in SWP water in 2023 was mostly recorded in Zeller residential pond with 20,166 particles L⁻¹ in June, 19,846 particles L⁻¹ in October, and 18,098 particles L⁻¹ in December. Both Aactiva and Bingemans ponds had its highest fragment concentration in August of 17,639 particles L⁻¹ and 16,796 particles L⁻¹, respectively (Figure 3-1a). Meanwhile, Shirley pond had the lowest fragment concentration among 5 SWPs throughout the year, ranging from 3,479 particles L⁻¹ in February to 4,196 particles L⁻¹ in December (Figure 3-1a). It is interesting to note that fragment concentrations least varied across 5 SWPs in April 2023, with only 1 to 1.5 times variation, and the highest fragment concentration for each pond was recorded in either June or August of the year (Figure 3-1a). On the other hand, the highest fiber concentration was found in Zeller residential pond with 559 particles L⁻¹ in August, decreasing to 507 particles L⁻¹ in Bingemans pond in April and 399 particles L⁻¹ in Wabanaki pond in December. (Figure 3-1b). Both Shirley pond and Aactiva pond had relatively low fiber concentrations among 5 SWPs with less than 200 particles L⁻¹ throughout the year (Figure 3-1b). In contrast, both Zeller pond and Bingemans pond experiences a decreasing trend in fiber concentrations toward the end of the year (Figure 3-1b). Interestingly, fiber concentrations across all 5 SWPs were relatively similar in October, and fragment concentrations were approximately 50 times higher than fiber concentrations for all ponds (Figure 3-1a,b).

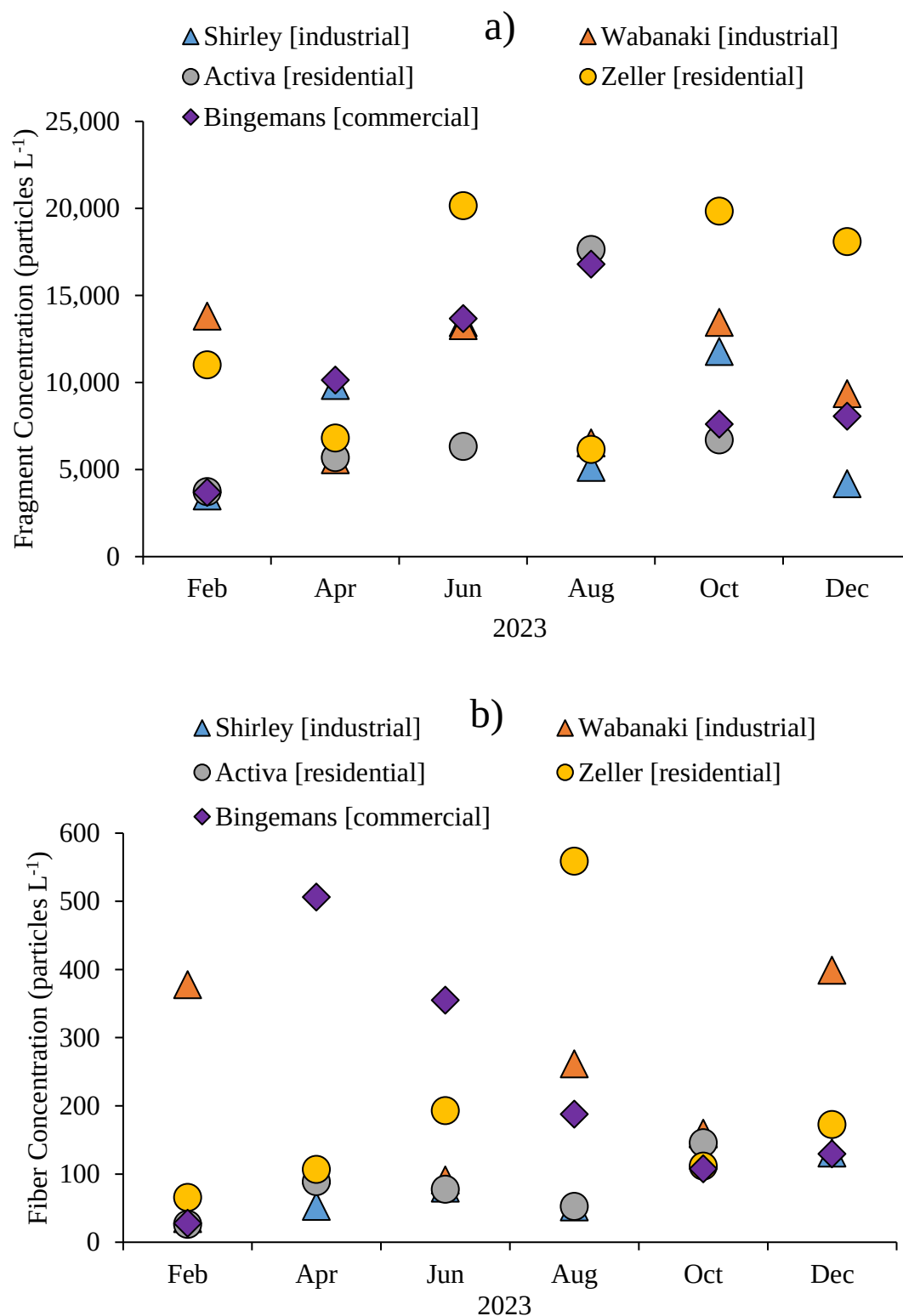


Figure 3-1: Microplastics fragment (a) and fiber (b) concentrations from LDIR analysis (particles per litre (L); particles L⁻¹) in water column in five SWPs collected in 2023.

3.3.2 MPs partitioning between water column and sediment bed in 5 SWPs

In SWP's sediment, fragment-to-fiber ratio was as high as 103.3 in Bingemans pond, followed by 95.6 in Shirley pond, and decreasing to the lowest ratio of 45.6 in Wabanaki pond (Figure 3-2). The two residential ponds Activa and Zeller had relatively close fragment-to-fiber ratios in sediment, with 61.0 and 54.6, respectively (Figure 3-2). Meanwhile, fragment-to-fiber ratio varied considerably in SWP's water throughout the year, with the highest ratio of 332.8 observed in Activa pond in August (Figure 3-2). The second and third highest ratio was 185.5 in Shirley pond in April and 177.2 in Zeller pond in October (Figure 3-2). Zeller pond also had the lowest fragment-to-fiber ratio among 5 ponds, with 11.0 in August (Figure 3-2). More significantly, fragment-to-fiber ratio was generally higher in water than in sediment throughout the year in most of the pond (Figure 3-2). The two industrial ponds both have higher ratio in April and June water compared to sediment, while the two residential ponds only have lower ratio in water than in sediment at one point later in the year (Figure 3-2). Bingemans pond was the only pond with most of fragment-to-fiber ratio in water lower than in sediment, except for February ratio (Figure 3-2).

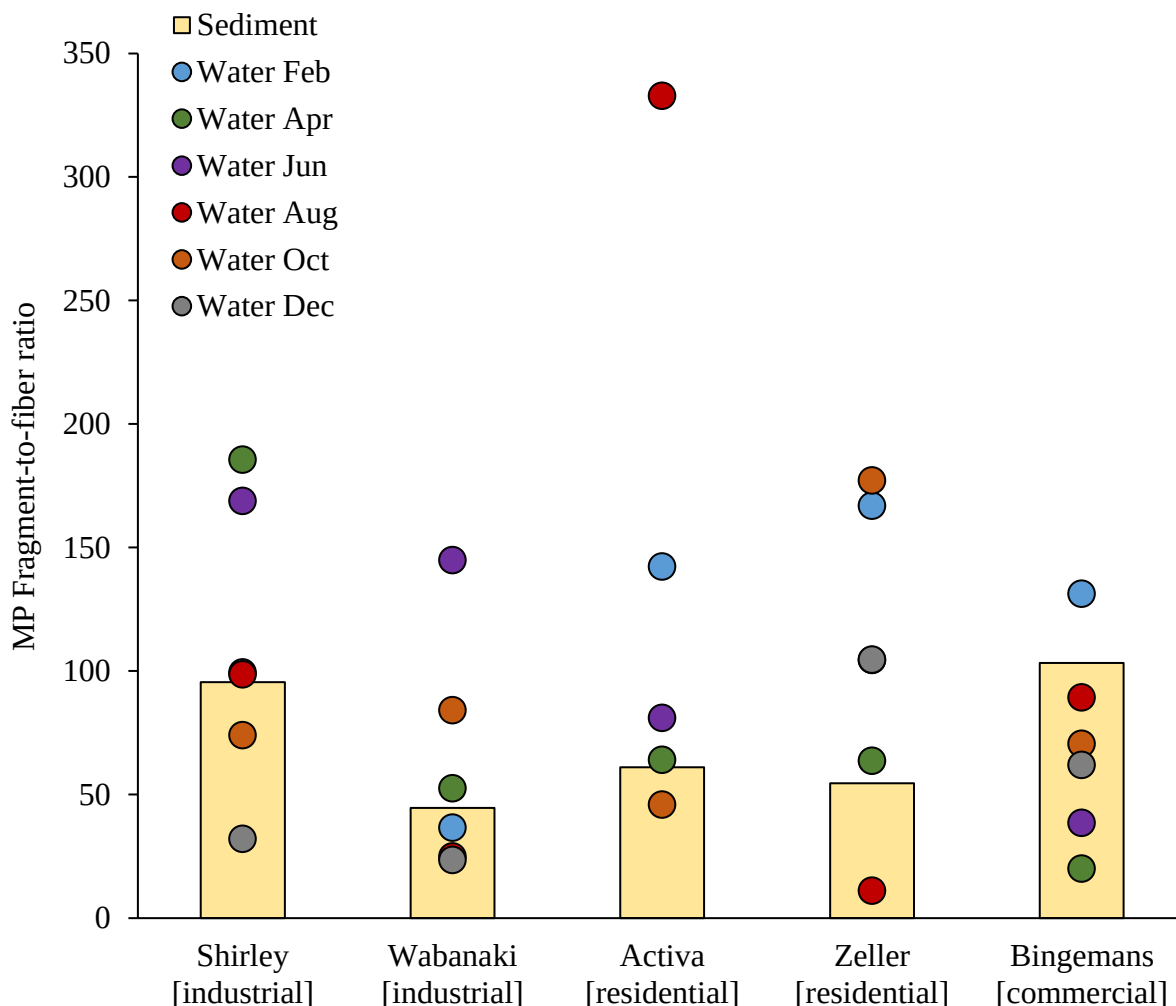


Figure 3-2: The MP fragment-to-fiber ratio from LDIR analysis in sediment and water samples of SWPs.

3.3.3 Polymer composition and abundance of MP in 5 SWPs water

LDIR analysis of water samples across the 5 SWPs detected 8 major polymer groups: acrylonitrile butadiene styrene (ABS), polyamide (PA), polyethylene (PE), polyethylene terephthalate (PET), polypropylene (PP), polytetrafluoroethylene (PTFE), polyurethane (PU), and polyvinyl chloride (PVC) (Figure 3-3). Detected polymers with abundance less than 1% across all 5 SWPs were grouped into “Other” group (Figure 3-3). PA was the most abundant polymer, followed by PE and PU (Figure 3-3). Both PA and PE accounted for approximately 70-

90% MPs detected in water across all 5 SWPs (Figure 3-3). Meanwhile, PET, ABS, PVC, and PP were identified in relatively small abundance compared to other polymers (Figure 3-3). Particularly, PET's abundance was relatively constant at 1-2% across all 5 ponds, whereas both ABS and PP had the highest abundance of 3% and 2%, respectively, in the Wabanaki pond (Figure 3-3). PTFE and PU were significantly more abundant in Bingemans pond than in other ponds, accounting for 6% of MPs particles each (Figure 3-3).

In comparison with SWP sediment's polymer composition, it appeared that PA in all 5 ponds was more prevalent in sediment with close to 80% abundance, whereas in water, PA only accounted for less than 70% of MP particles (Figure 3-4). PE particles, on the other hand, made up approximately 25% of MPs in water samples on average but only 15% of MPs in sediment across 5 SWPs (Figure 3-4). PTFE's abundance was the highest in Bingemans pond for both sediment and water samples, whilst PU was detected more frequently in water samples with up to 6% abundance (Figure 3-4).

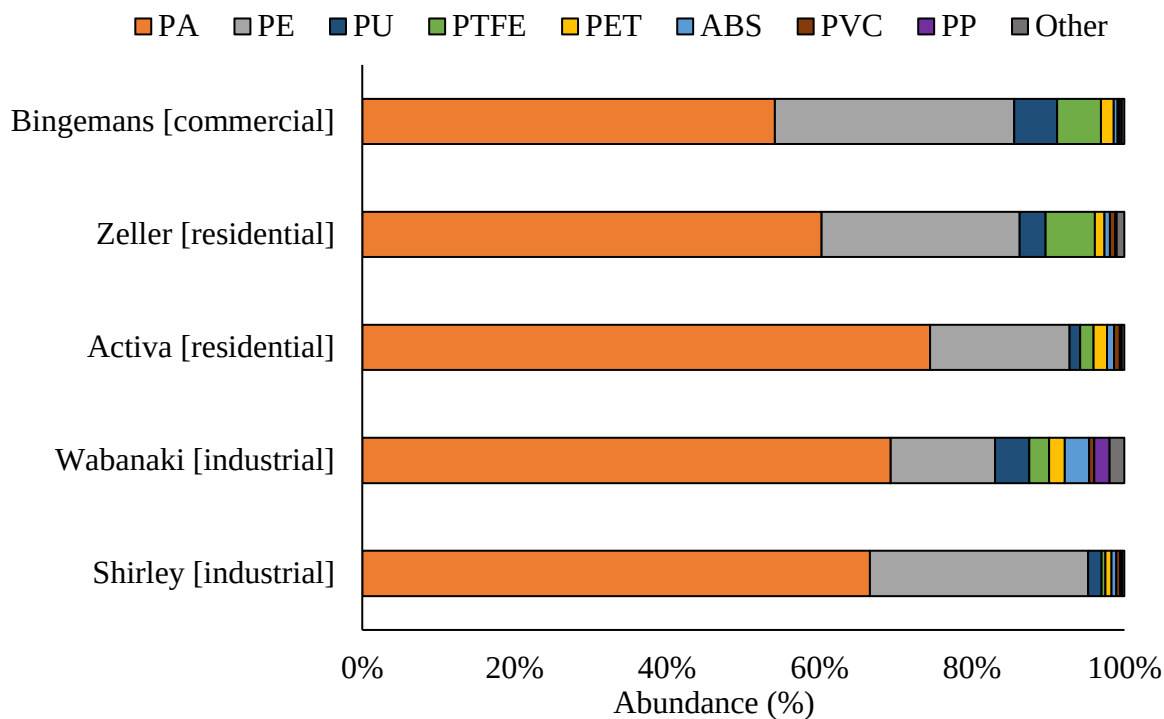


Figure 3-3: The average polymer composition of MPs detected in water column of five SWPs from LDIR analysis.

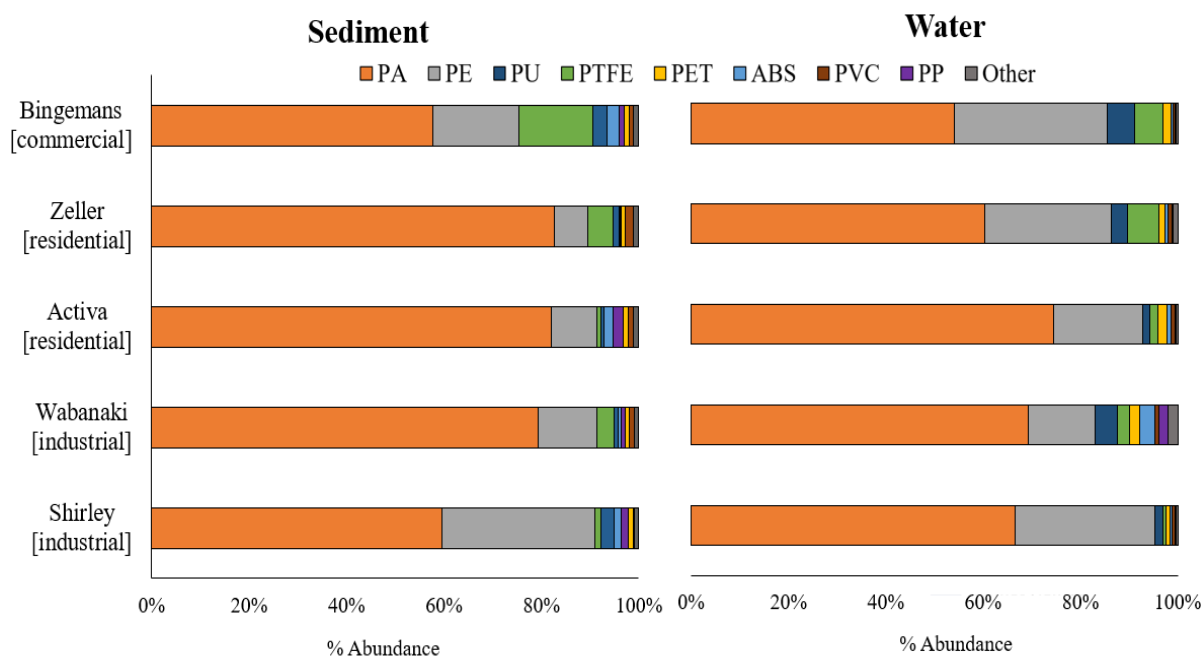


Figure 3-4: The average polymer composition of MPs from LDIR analysis detected in sediment versus in water of five SWPs.

3.3.4 MP particles size distribution in water samples of SWPs

MP particle sizes detected by the LDIR imaging spectroscopy in water samples across 5 SWPs were categorized into 6 size ranges: <10 μm , 10-25 μm , 25-50 μm , 50-100 μm , 100-500 μm , and > 500 μm (Figure 3-5). It was noticeable that particles in 10-25 μm size class dominated in water with relatively similar abundance of 42% across all 5 ponds (Figure 3-5). The second and third most-abundant size ranges are 25-50 μm particles and <10 μm particles, accounting for 30% and 15% of MPs in 5 SWPs, respectively (Figure 3-5). MP particles that are larger than 500 μm were the least abundant in the pond water with less than 0.5% prevalence across all ponds (Figure 3-5). Moreover, it was apparent that particles smaller than 25 μm were more likely to be found in sediment than in water, as its abundance could be as high as 55% in sediment inlet forebay (Figure 3-5). Similarly, particles less than 10 μm were also more abundant in sediment main basin with close to 20% prevalence (Figure 3-5). In contrast, particles larger than 25 μm were detected more frequently in water across all 5 SWPs, with roughly 31% of total MPs particles in 25-50 μm size class, 11% particles in 50-100 μm size range, and 3-4% abundance for 100-500 μm particles (Figure 3-5). Specifically, particles larger than 500 μm was identified in small quantity in water column whereas none of those particles were found in sediment of 5 SWPs (Figure 3-5).

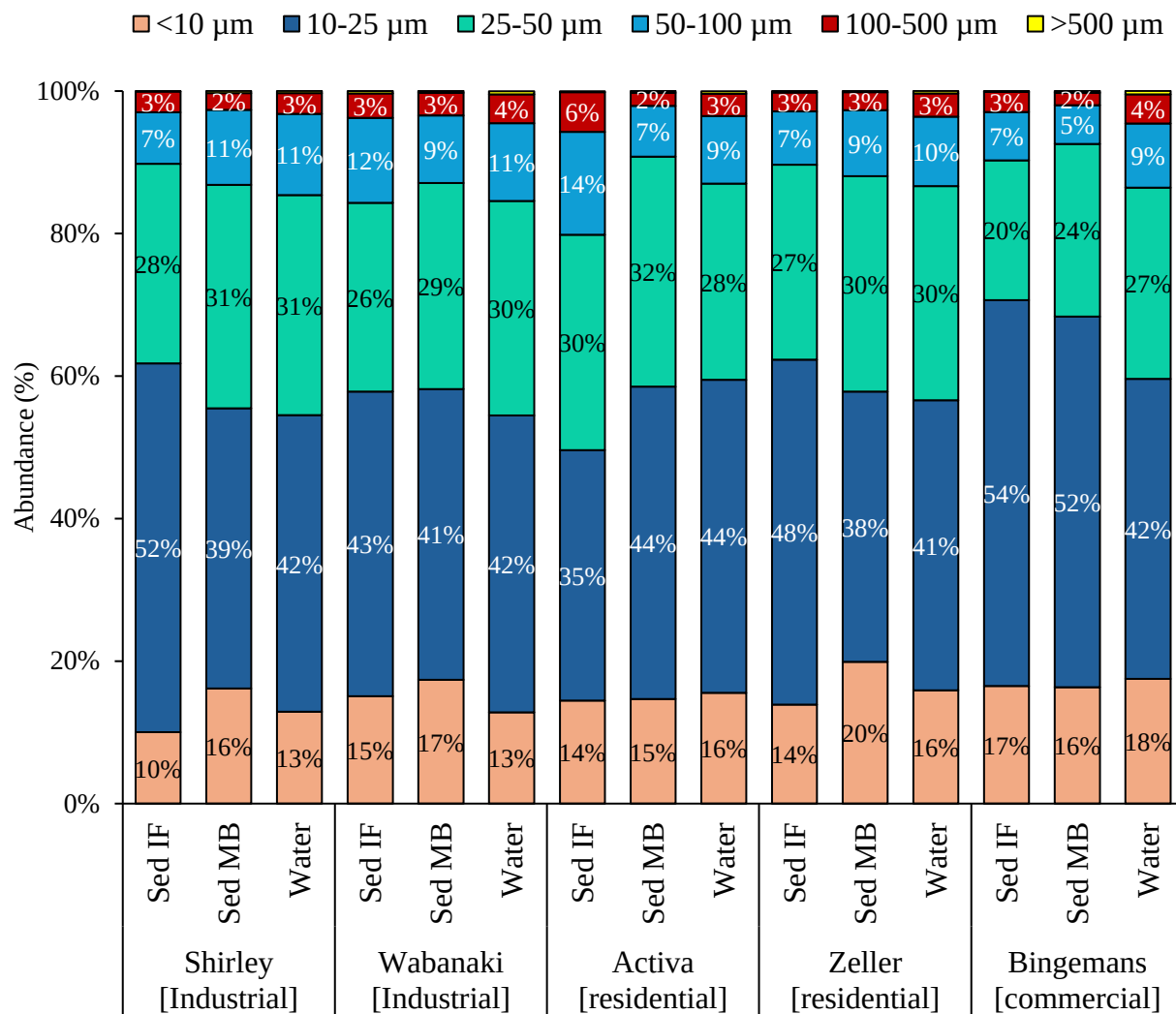


Figure 3-5: The abundance (%) of MP particles in different size classes in the sediment inlet forebay (Sed IF), sediment main basin (Sed MB), and water of five SWPs from LDIR analysis.

3.3.5 Correlation analysis: Controls on SWP water MP abundances

Pearson correlation analysis between pond morphology, climate data, and MP variables was performed and the results are presented in Figure 3-6. A statistically significant correlation ($\rho < 0.1$) between fragment concentration and polymer concentration was found (Figure 3-6).

Furthermore, both fragment and fiber concentrations negatively correlated to the catchment-to-pond surface area (SA) ratio while the pond's imperviousness positively correlated to fiber concentration (Figure 3-6). Specifically, MPs concentration statistically negatively correlated with the amount of precipitation the week before sampling, whereas only PE and PU concentration significantly positively correlated with the amount of precipitation on the sampling day ($\rho < 0.1$) (Figure 3-6). In addition, total precipitation the month before sampling, wind speed during the week before sampling, and wind speed on sampling day were all negatively correlated with fragment, fiber, and polymer concentration in general (Figure 3-6).

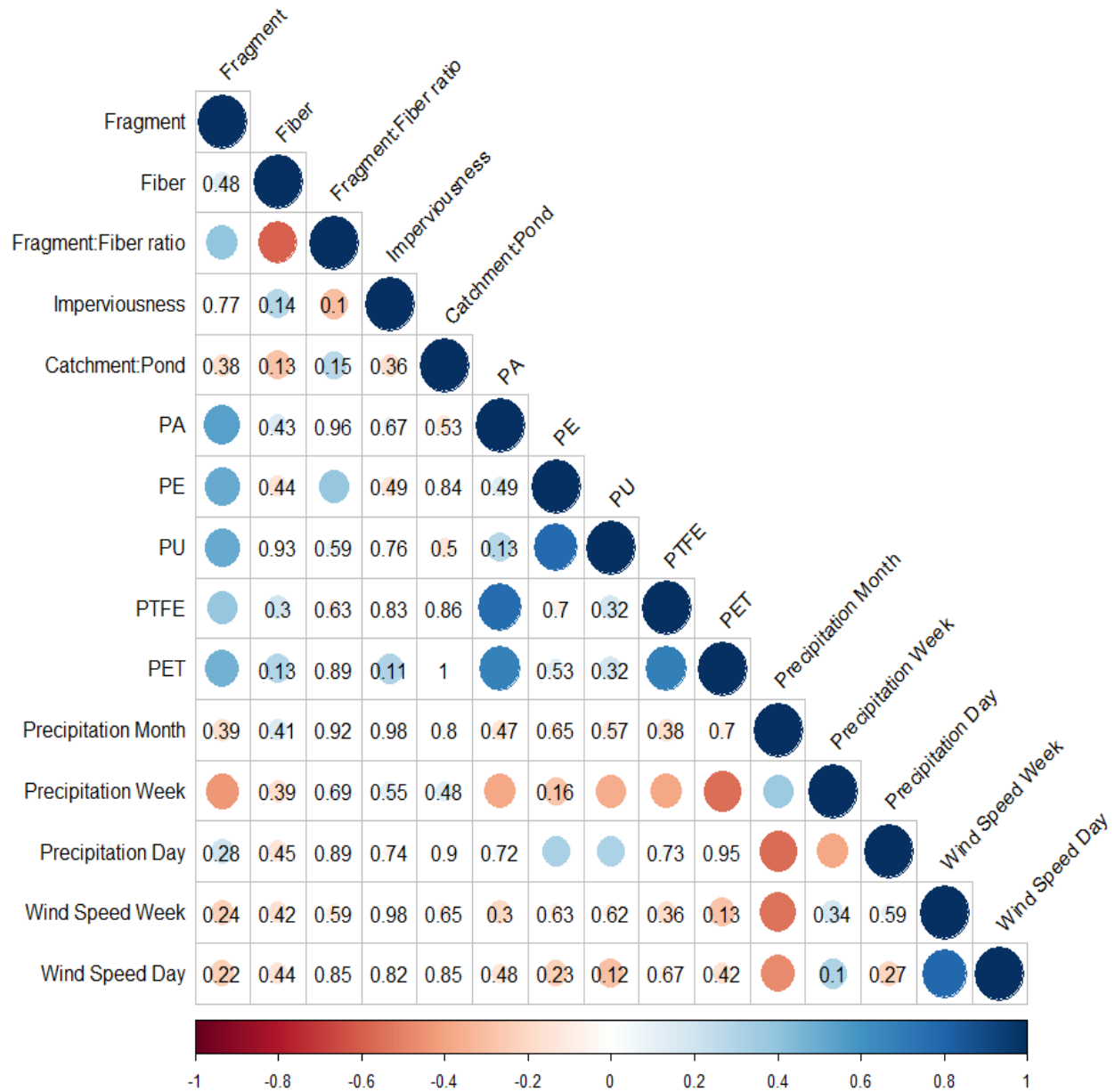


Figure 3-6: Pearson's correlation analysis between pond characteristics, climate data, and MP variables ($\rho < 0.1$). Size and color of a circle indicate the strength and direction (blue is positive, red is negative) of the correlation, respectively. For non-significant relationship, the ρ -value is displayed.

3.4 Discussion

3.4.1 MP concentrations in water samples of SWPs

The MP concentrations in SWP water reported in this study are considerably higher than the values documented in previous research (Brooks *et al.*, 2023; Liu *et al.*, 2019a; Olesen *et al.*, 2019). Both Olesen *et al.* (2019) and Liu *et al.* (2019a) estimated the highest MP abundance in SWPs in Denmark to be 270 items L⁻¹ and 230 items L⁻¹, respectively, which were in the range of our fiber concentration (26 particles L⁻¹ to 559 particles L⁻¹; Figure 3-1b) yet at least 10 times lower than our fragment concentration across 5 SWPs (3,479 particles L⁻¹ to 20,166 particles L⁻¹; Figure 3-1a). Järnskog *et al.* (2020) later document 5,900 MPs particles L⁻¹ in stormwater in Sweden, which was mostly lower than our fragment concentration. Similarly, MP concentrations in stormwater in other studies were approximately two orders-of-magnitude lower than our results (Brooks *et al.*, 2023; Ramussen *et al.*, 2024; Werbowski *et al.*, 2021). The noteworthy difference in MP abundance in stormwater across existing literature, hence, can be ascribed to the diversity in MP cut-off size as well as sampling techniques between studies (Shruti *et al.*, 2021; Stang *et al.*, 2022). For instance, Brooks *et al.* (2023) chose 300µm as their cut-off size while Werbowski *et al.*'s (2021) lower size limit was 125µm, which was substantially bigger than our targeted MP particle size of less than 500µm and larger than 20µm (see 3.2.2; Yu *et al.*, 2024). Meanwhile, both Ramussen *et al.* (2024) and Liu *et al.* (2019a) only collected the water samples during dry weather, which might not reflect the variability in MP concentration associated with runoff events and seasonal changes. Österlund *et al.* (2023) also emphasized that single grab samples are insufficient to characterize MP discharge loads from urban catchment and are not indicative of MP environmental fate in SWPs. By collecting samples every two months from the ponds' water column for a whole year, this research thus represents an initial

short-term monitoring of MP occurrence in SWPs. The seasonal variation in both fragment and fiber concentrations observed in all 5 ponds throughout the year in our study (Figure 3-1a,b) further supported Brooks *et al.*'s (2023) outcome, in which they recorded substantial differences in MP concentration between wet and dry seasons. G. Wang *et al.* (2021) additionally demonstrated that the seasonality of MPs prevalence and properties must be taken into account for an accurate quantification of MPs in the environment.

MP concentrations from the water column in SWPs found in this study are substantially higher than in other freshwater bodies such as rivers and lakes, which is anticipated as stormwater runoff is a major pathway for MP mobilization in the environment (Cho *et al.*, 2023; de Jesus Piñon-Colin *et al.*, 2020). For example, only 0.1 fragments L⁻¹ on average were recorded in the surface water of Ottawa River and its tributaries in Canada (Vermaire *et al.*, 2017), whilst Yuan *et al.* (2019) estimated 34 items L⁻¹ for Poyang Lake in China. In comparison, our average stormwater MP concentration of 10070 particles L⁻¹ was at least 1000 times higher than those reported for surface water from rivers and lakes, illustrating urban areas as a crucial contributor to MPs ubiquitous prevalence in the environment (Grbić *et al.*, 2020). Moreover, the dominance of MP fragments abundance over fibers in stormwater observed in this study (50 times higher, on average, across 5 SWPs, Figure 3-1a,b) agreed with Cho *et al.*'s (2023), Sang *et al.*'s (2021), and Ziajahromi *et al.*'s (2020) findings as they all recorded a higher fraction of fragments over other particles shapes with approximately 70% prevalence. Z. Wang *et al.* (2021) further conducted a settling experiment of MP particles following Stokes's Law and claimed that particles with irregular shapes had significantly slower settling velocity, supporting our findings of fragments floating in the water column of SWPs. Particle shapes, as a result, are a major control of MP distribution and mobilization in SWPs.

3.4.2 MPs partitioning between water column and sediment bed in SWPs

The diverse fragment-to-fiber ratio between the sediment bed and water column found in this study (Figure 3-2), moreover, demonstrated that MP particles could vertically partition within a water body, supporting Lenaker *et al.*'s (2019) findings. Brooks *et al.* (2023) additionally observed MP shapes in water to be more variable than those in sediment, highlighting that MP settling velocity, and subsequently particle distribution between different environmental matrices, are shape-dependent. The dominance of fragments in both water and sediment in our SWPs (Figure 3-2), nonetheless, signalled that besides particle characteristics, MPs partitioning was also under the influence of other environmental factors such as precipitation and re-suspension of sediment (Darabi *et al.*, 2021; Xia *et al.*, 2021). In particular, Xia *et al.* (2021) noticed a season-driven distribution of MPs as rainy seasons facilitated MPs accumulation in river sediment, while Darabi *et al.* (2021) found a significant correlation between MP loads in stormwater runoff and rainfall intensity. These results were thus applicable to this study as our fragment-to-fiber ratio in the water column varied remarkably throughout the year across 5 SWPs (Figure 3-2). Both Lenaker *et al.* (2019) and Xia *et al.* (2021) later pointed out that disturbance to the pond induced by wind forces and precipitation can resuspend the sediment bed, causing MPs to move to overlying water or deeper sediment. As a result, seasonality and other hydrologic factors must be considered to avoid under- or over-estimation of MP loads, according to Lenaker *et al.* (2019).

More significantly, the fragment-to-fiber ratio in the water versus in the sediment was different between 5 SWPs with different land use (Figure 3-2), signifying the impacts of catchment LULC on MP distribution in the ponds (Liu *et al.* 2019a; Liu *et al.*, 2019b; Townsend *et al.*, 2019). Specifically, this ratio was higher in water than in sediment for most of the year in

both residential ponds (Figure 3-2), which could be ascribed to frequent MP discharges from nearby residences, resulting in shorter residence time for MPs to settle into the pond sediment (Stang *et al.*, 2022). Meanwhile, Bingemans pond - our only commercial pond - had a fragment-to-fiber ratio in sediment higher than in water as a result of an 8-year MPs accumulation in the pond sediment (Figure 3-2). In agreement with these findings, previous research suggested that MP abundance and characteristics are subjected to catchment LULC, since populated areas with a higher density of anthropogenic activities exhibit higher concentrations of MPs with diverse morphologies (de Jesus Piñon-Colin *et al.*, 2020; Sun *et al.*, 2023; Xue *et al.*, 2023). Land use patterns, therefore, exert critical controls on MP occurrence and distribution in a water body (Iannuzzi *et al.*, 2024). Interestingly, when comparing MP characteristics in water and sediment, both Zhang *et al.* (2022) and Brooks *et al.* (2023) indicated that sediment samples provided more reliable correlations as well as better insights into MPs' environmental behaviors. Wei *et al.* (2023) later explained that MPs in the surface water reflect a short-term input from runoff whereas sediment is more of a MP sink with particles accumulated over the years. These fragment-to-fiber ratios presented in this study, thus, provide an insight into microplastics distribution between different environmental matrices within the same water body.

3.4.3 MPs polymer composition and abundance in water samples of SWPs

The polymers that have been detected frequently in stormwater are mostly light-density polymers with various commercial purposes such as PE and PP (Boni *et al.*, 2022; Stang *et al.*, 2022). In particular, Liu *et al.* (2019a) sampled urban stormwater and discovered that 72% of MP particles were PP, whilst Parmar *et al.* (2023) detected PE in all sampling sites with an abundance as high as 50% of total particles. In this study, PP were found in relatively small proportions whereas PA and PE were the most prominent polymer types, accounting for 80% of

MP particles in the SWP water (Figure 3-3). Since PA is the core material in the textile industry and PE is a versatile polymer for packaging, our polymer composition hence illustrated the influence of anthropogenic activities on MP occurrence in the pond (Shruti *et al.*, 2021). In other words, the polymer profiles of MPs in the SWP water column are an aftermath of the widespread polymer application in the urban catchment (Liu *et al.*, 2019b). Additionally, the difference in polymer abundance between ponds as well as the noticeable appearance of PU - a polymer commonly used in insulation materials- in the stormwater signalled the vital control that catchment land use has on polymer occurrence in SWPs (Gama *et al.*, 2018; Liu *et al.*, 2019a). Both Jayalakshamma *et al.* (2023) and Wei *et al.* (2023) also anticipated that MP polymer composition varies significantly with land use patterns, supporting this study's outcome.

It was noteworthy that light-density polymers (*e.g.* PE) were more prevalent in the water column, whereas heavier polymers such as PA and PTFE were detected more frequently in sediment in this study (Figure 3-4). The difference in polymer abundance between water and sediment in SWP therefore suggested polymer density as an important factor in the MPs distribution pattern between different environmental matrices, which has been observed in previous research (Auta *et al.*, 2017; de Jesus Piñon-Colin *et al.*, 2020; Mak *et al.*, 2020). Dhivert *et al.* (2022) even emphasized that particle density induced a vertical MPs partitioning between water and sediment under low flow conditions. By characterizing plastics in SWPs, Brooks *et al.* (2023) further identified a significant portion of plastics in sediment were polymers denser than water (*e.g.*, PET, PU), yet the high proportion of lighter polymer overall implied that polymer density was not the sole determinant of MP environmental fate. Other factors such as particle aggregation and biofilm growth can significantly alter MP particles' physicochemical properties and henceforth, their fates in the environment (Yan *et al.*, 2021; Wei *et al.*, 2023). In

particular, MP particles can get attached to natural colloids during runoff or to organisms in stormwater ponds to form aggregates, which subsequently change their distribution, mobilization, and toxicity in the water bodies (Koelmans *et al.*, 2022; Liu *et al.*, 2021; X. Wang *et al.*, 2021; Yan *et al.*, 2021). Lagarde *et al.* (2016) later conducted an experiment on MP aggregates and discovered that microalgae rapidly adhered to PP fragments, forming hetero-aggregates denser than water. As a result, other co-existing constituents in the environment can modify MP settling behaviour and thus must be considered in characterizing MP dispersion (Duan *et al.*, 2021).

3.4.4 MP particle size distribution in 5 SWPs water

MP abundance in this study appeared to decrease with increasing particle size (Figure 3-5), which has been documented in previous studies (Jayalakshamma *et al.*, 2023). For instance, Grbić *et al.*'s (2020) result exhibited an exponentially increasing trend in MP particle numbers with decreasing size, while Cho *et al.* (2023) mainly found MP particles in the 20-100 µm size class in stormwater runoff in Gumi, South Korea. Ziajahromi *et al.* (2020) further detected more microplastics in the 25-100 µm size range in stormwater outlet samples, which applied to this study (Figure 3-5). The dominance of small-size MPs, specifically particles less than 50 µm that make up 80% of particles in our SWPs water (Figure 3-5), thus agreed with other research's findings (Cho *et al.*, 2023). Having said that, the difference in selecting MP cut-off sizes between investigations poses a challenge to understanding MP dispersal patterns with various size ranges thoroughly. Jayalakshamma *et al.* (2023) specifically emphasized that there are major gaps in investigating MP particles that are smaller than 100 µm despite their overwhelming prevalence and their impacts on final MP concentration.

In comparison with sediment, it however appeared that there are more smaller particles in sediment whereas the proportion of bigger particles is higher in water (Figure 3-5), indicating that particle size essentially influenced MP partitioning in different environmental matrices (Wang *et al.*, 2018; Yan *et al.*, 2021). Khatmullina and Isachenko (2017) also examined settlements of MP particles in multiple sizes and shapes and claimed that particle shapes only influenced MP settling behavior once they reached a certain particle size. Another possible explanation for a higher fraction of small-sized MPs in sediment than in water is MPs fragmentation in the environment along with other hydrologic factors, according to Scherer *et al.* (2020). In particular, MP fragmentation arose from being consumed by microorganisms or UV exposure, which produced significantly smaller particles that transported differently spatially and temporally than larger MPs (Gerritse *et al.*, 2020; Song *et al.*, 2017; Ter Halle *et al.*, 2016). Onink *et al.* (2022) later pointed out that smaller, more mobile MPs also get mixed deeper, resulting in higher concentrations of small-sized MPs in sediment beds than in subsurface water. Besides, Xia *et al.* (2021) suggested that MPs buried in sediment are subjected to redistribution under disturbance-induced resuspension of sediment bed, which undoubtedly complicated MPs transportation and deposition mechanism in a water body.

3.4.5 Influence of environmental factors on MP concentrations in SWPs water

Hydraulic loading represented by the catchment-to-pond impervious surface area (SA) ratio is another factor influencing MP discharge in SWPs (Brooks *et al.*, 2023; Liu *et al.*, 2019b). According to S nderup *et al.* (2016), a higher catchment-to-pond SA ratio would result in a higher water flowing rate in the pond and less residence time for MPs to disperse between the water column and the sediment bed. SWPs with higher SA ratio, therefore, would have lower MP concentration due to increased flush rate, as demonstrated by the negative Pearson correlation

(Figure 3-6) (Anderson *et al.*, 2002). Other catchment characteristics such as land use and morphology can impact MPs occurrence and settlement in the pond as well (Shafi *et al.*, 2024). For instance, Cho *et al.* (2023) pointed out that industrial catchments with a higher degree of imperviousness tend to discharge a higher load of MPs, whereas Gu *et al.* (2017) and Istenič *et al.* (2012) suggested pond length-to-width ratio as the primary parameter influencing SWPs' performance in retaining pollutants.

More significantly, the negative correlation between climate factors (*e.g.*, precipitation, wind speed) and MP concentration found in this study (Figure 3-6) suggested that weather considerably affects MP deposition in SWPs (Österlund *et al.*, 2023). In line with this outcome, Boni *et al.* (2022) also reported a significant negative correlation between rainfall intensity and MP concentration, whilst Bullard *et al.* (2021) indicated that wind transport can mobilize MPs on the water surface, impacting MPs' final concentration and distribution in the pond. Furthermore, the finding that the polymer concentration negatively correlated with total precipitation during the week before sampling yet positively correlated with the amount of precipitation on sampling day (Figure 3-6) demonstrated a first flush phenomenon occurred at the ponds (Shafi *et al.*, 2024). According to Cho *et al.* (2023) and Cheung *et al.* (2019), MP input into the ponds peaked within the initial phase of a storm event and gradually reduced as rainfall continued. Lower MP abundance, therefore, is expected since MPs floating on the water surface can be washed away during the first flush and stormwater runoff further dilutes MP concentration in the water (Ross *et al.*, 2023; Sun *et al.*, 2023; Wang *et al.*, 2022). As a result, climatic variation of MP particles must be incorporated into investigating MP dynamics within a water body (Sun *et al.*, 2023).

3.5 Conclusion

In this study, MP concentration in the water of SWPs was higher than values reported in the existing literature and varied significantly with seasonal changes. The diverse fragment-to-fiber ratio between the sediment bed and water column, as well as fragments being the dominant shape in both environmental matrices, further implied that the particle morphology essentially influenced MP distribution in SWPs. Meanwhile, polymer composition and size distribution results not only emphasized the major control catchment land use had on MP occurrence but also signalled the preferential partitioning of MP particles in SWPs. Other factors such as precipitation and hydraulic loading affect MP abundance in SWPs as well, and hence, it is strongly recommended that catchment and climate factors should be taken into account when investigating MP accumulation in a water body.

4 Conclusion

4.1 Summary of key findings

The objectives of this thesis were to quantify and characterize microplastics (MPs) in stormwater pond (SWP), as well as investigate factors controlling MP accumulation and distribution in stormwater ponds. By collecting sediment and water samples from stormwater ponds with different land uses in their catchment area, I was able to assess the variability in MPs abundances, shapes, types, and sizes. Other sediment properties, catchment characteristics, and environmental conditions were evaluated to determine their impacts on MPs occurrence in SWPs, which advances the understanding of MPs deposition mechanism in the aquatic environment.

In Chapter 2, sediment cores were collected along the transect in the inlet forebay and main basin of 5 SWPs. Sediment cores were sectioned into sub-samples afterwards to analyze sediment properties such as bulk density, grain size and organic carbon (OC) content, while MPs in each sediment core were extracted in triplicates in a similar procedure reported by Yu *et al.* (2024) and characterized using LDIR imaging spectroscopy. The result revealed that MP concentration varied 2-3 times across 5 SWPs, and fragment concentration was 50 times higher than fiber concentration in the pond sediment. Moreover, sediment, OC, and MP accumulation rates were higher in the inlet forebay than in the main basin for most SWPs, with the highest accumulation rates recorded in industrial ponds, followed by residential and commercial ponds. These matching patterns therefore demonstrated that there were considerable variabilities in MPs

characteristics in ponds with different sediment properties. Catchment land use was another critical factor that impacted MP accumulation in the ponds, as illustrated by MP properties data, including accumulation rates, polymer composition, and size distribution. Findings from Chapter 2 hence narrow the gap in understanding MP occurrence and distribution in sediment, while providing insights into the contribution of urban catchments to MP pollution.

In Chapter 3, the same 5 SWPs were adapted to be the sampling sites for stormwater samples, where samples were collected in the water column near the pond outlet every two months throughout 2023. Water samples were extracted in a similar experimental design as in Chapter 2, analyzed, and correlated with catchment characteristics and climate data. Outcomes from Chapter 3 indicated that MP concentration fluctuated from month to month, with fragments being the dominant shape throughout the year. MP polymer types and size distribution reflected the influence of catchment land use, as MP profiles appeared to be the aftermath of nearby anthropogenic activities. Precipitation and wind speed data furthermore revealed a first flush phenomenon that occurred in the pond during a storm event, in which initial rainfall can wash away surface MPs and dilute MPs in the water column. Ponds' hydraulic loading was noted to be negatively correlated with MP concentration in the stormwater as well, confirming the dilution effect. Future research, therefore, must account for hydrologic conditions when exploring MP occurrence in the aquatic environment.

Interestingly, the comparison between MP profiles in sediment (Chapter 2) and in water (Chapter 3) shed light on the preferential partitioning of MPs between two different environmental matrices, which bridged the gap in existing literatures. In particular, the varying fragment-to-fiber ratio between sediment and water can be ascribed to seasonality and catchment land use. Polymer density also appeared to impact MP deposition mechanisms as well, since

heavier polymers such as PA and PTFE were more abundant in sediment and lighter polymers such as PE tend to float in the water column. The diverse size distribution between sediment and water, and the overwhelming prevalence of small-sized, fragment-shaped MPs further highlighted the influence of particle morphologies on MP partitioning. Findings from both chapters indicated a significantly higher MP concentration in SWPs than reported in previous research, confirming SWPs as an effective sink of MPs. Besides MP morphologies, other elements including land use, sediment properties, and climatic factors complicated MPs distribution pattern and subsequently alter MP environmental fate, which requires further investigation to examine these factors together on how they influence MPs transportation and deposition.

4.2 Recommendation for future research

Results from this thesis can be applied in further investigations to generate MP accumulation rates across the Grand River watershed and eventually the Great Lakes ecosystem. The outcomes from this study, moreover, reflect MP preferential partitioning and settlement behaviour in the aquatic environment, and thus, it can support future research on characterizing MP mass balance and budget for urban watersheds, help estimating the quantity of MPs reaching downstream water bodies. Given the control of several factors on MP occurrence and distribution found in this study, future investigations can establish source apportionment for MPs in urban area and advance toward managing MPs pollution. Findings from this study regarding MP characteristics can help policymakers to tailor management strategies accordingly as well.

Number of challenges were identified in this study, which can be noteworthy for future research. In Chapter 2, a major difficulty in comparing my results with previous studies was the inconsistency in reporting units for MP accumulation rates, which hindered the progress of

quantifying MPs in the environment. Given how limited data on MPs in stormwater are, uniform reporting units for comprehensive quantitative MPs data across existing literature are urgently needed. Furthermore, the variety in sampling, extracting, and detecting methods for MPs gave rise to a wide range of uncertainty in quantifying and characterizing MPs, and hence, establishing standardized criteria in methodology might be beneficial for future research into MPs. Additionally, the several order-of-magnitude differences in MP particle counts between the LDIR technique and stereoscopic method later reflect how easily MP abundance in the environment can be underestimated, recommending other studies to select their approach carefully. In Chapter 3, seasonality of MP concentration highlighted that a long-term monitoring of MP input into SWPs is essential to capture the variability in MP profiles as well as depict MP loads more accurately. The ubiquitous prevalence of small-sized MPs should be the topic of interest for future research as well, as smaller MP particles are more mobile with a bigger threat of toxicity due to their bioavailability. Overall, a multidisciplinary approach that combines MP properties, environmental factors, and modelling efforts will be essential for advancing our understanding of MPs in stormwater and developing effective mitigating strategies.

Data Availability

The experimental data reported on in Chapters 2 and 3 are openly available in the Federated Research Data Repository (FRDR) at [https://doi.org/ 10.20383/103.0984](https://doi.org/10.20383/103.0984) (Chapter 2) and [https://doi.org/ 10.20383/103.0985](https://doi.org/10.20383/103.0985) (Chapter 3).

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Appendix

Appendix A: Supplementary Materials

Table A-1: The land use type and information of the five stormwater ponds selected for sampling. All below design details and pond morphology discussed herein are provided by the City of Kitchener.

Pond (Coordinates)	Land Use Type	Catchment Surface Area (ha)	Pond Surface Area (m²)	Impervious Catchment Surface Area: Pond Surface Area ratio	Imperviousness (%)	Years since dredged	Years since installation
Shirley (43°28'47.3"N, 80°25'33.2"W)	Industrial	60	4,412	62	46	3	34
Wabanaki (43°24'43.7"N, 80°25'59.9"W)	Industrial	25	3,637	45	65	2	12
Activa (43°24'27.0"N, 80°30'54.4"W)	Residential	34	1,469	120	52	4	23
Zeller (43°26'57.5"N 80°25'19.9"W)	Residential	48	3,364	63	45	4	31
Bingemans (43°28'32.9"N, 80°26'57.2"W)	Commercial	18	1,478	75	61	8	32

Table A-2: Estimated MPs concentration (particles g⁻¹) of each polymer type and its density in the sediments of five SWPs. Density data are taken from Hildago-Ruz et al. (2012).

Polymer type	Polymer density (g cm ⁻³)	Shirley [industrial]	Wabanaki [industrial]	Activa [residential]	Zeller [residential]	Bingemans [commercial]
Acrylonitrile butadiene styrene (ABS)	1.06-1.08	228	250	672	92	1,203
Polyamide (PA)	1.02-1.05	10,365	35,726	32,014	38,663	26,648
Polyethylene (PE)	0.91-0.97	5,446	5,473	3,649	3,181	8,134
Polyethylene terephthalate (PET)	1.37-1.45	207	408	361	380	509
Polylactic acid (PLA)	1.25	49	118	0	133	47
Polymethyl methacrylate (PMMA)	1.17-1.20	30	63	0	0	94
Polyoxymethylene (POM)	1.41-1.61	73	260	437	370	370
Polypropylene (PP)	0.9-0.91	259	346	872	139	451
Polytetrafluoro-ethylene (PTFE)	2.2	233	1,566	349	2,450	7,022
Polyurethane (PU)	1.2	464	416	286	537	1320
Polyvinyl chloride (PVC)	1.16-1.58	29	440	428	782	365

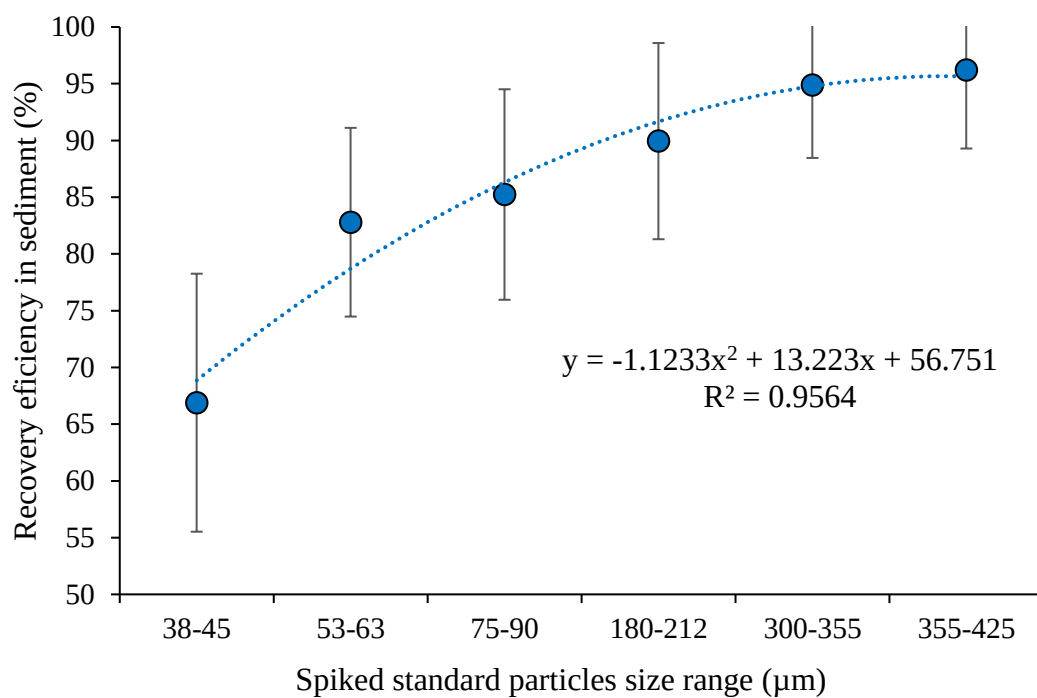


Figure A-1: The recovery efficiency (%) of spiked standard particles in various size ranges in sediment samples. A polynomial regression and its R^2 value are presented.

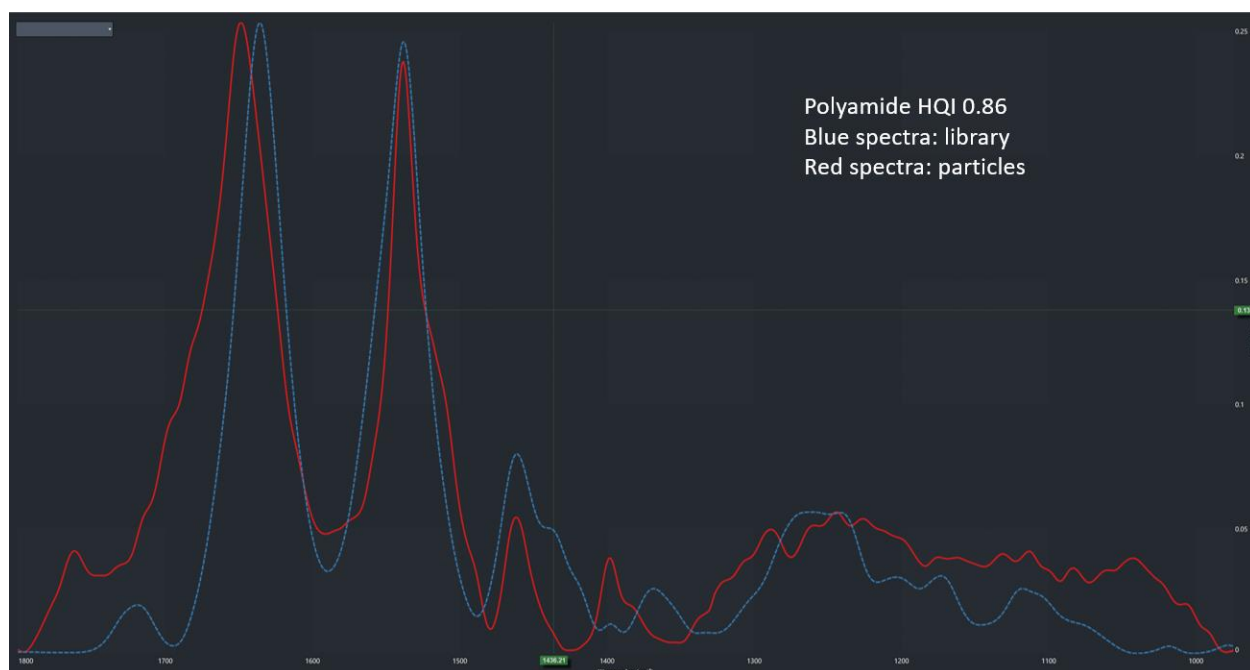


Figure A-2: Example of spectra matching for particle identification by the LDIR. The hit quality index (HQI) of the match is displayed.

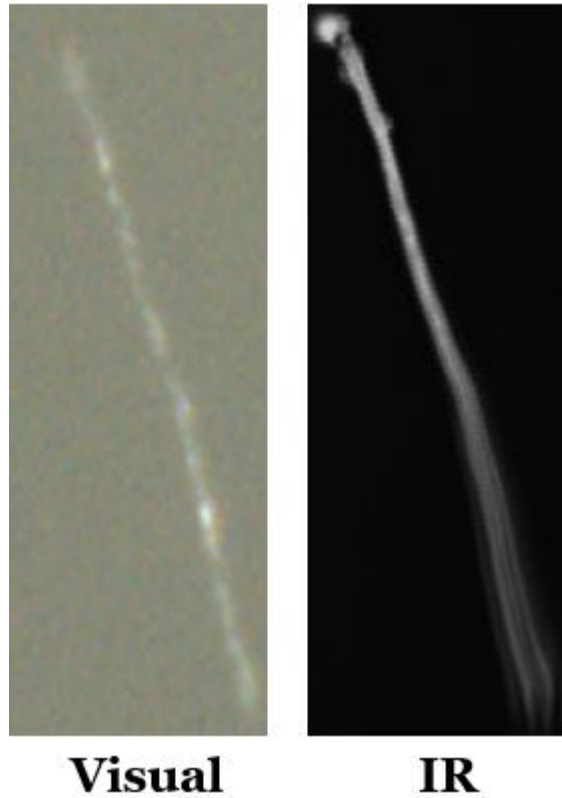


Figure A-3: Visual and IR images of particles with aspect ratio of 4:1 detected by the LDIR

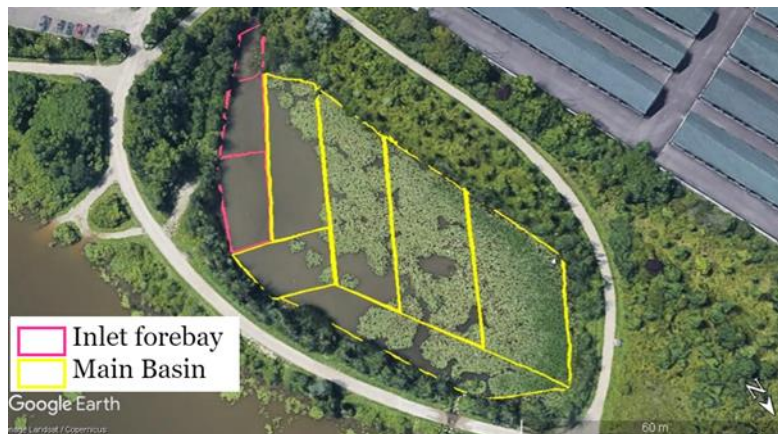


Figure A-4: The surface areas of Shirley stormwater pond sub-zones separated into a forebay and main basin zones based on consulting engineering drawings and aerial photos of the pond and estimated using Google Earth.



a) Sample at 8x magnification



b) Sample at 35x magnification

Figure A-5: MP particles viewed under the microscope at a) 8x magnification and b) 35x magnification, after extracted from sediment sample. Examples of particles being visually characterized as fragments and fibers, as well as spiked particles for recovery check are illustrated.

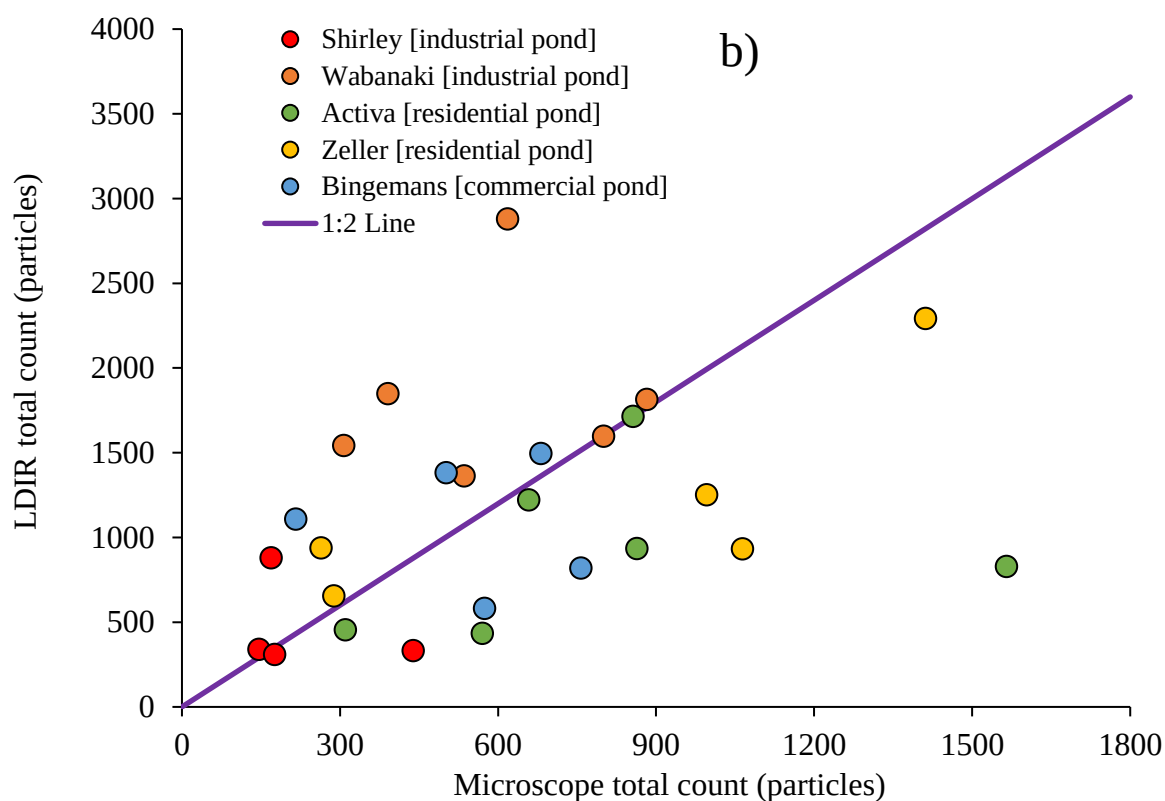
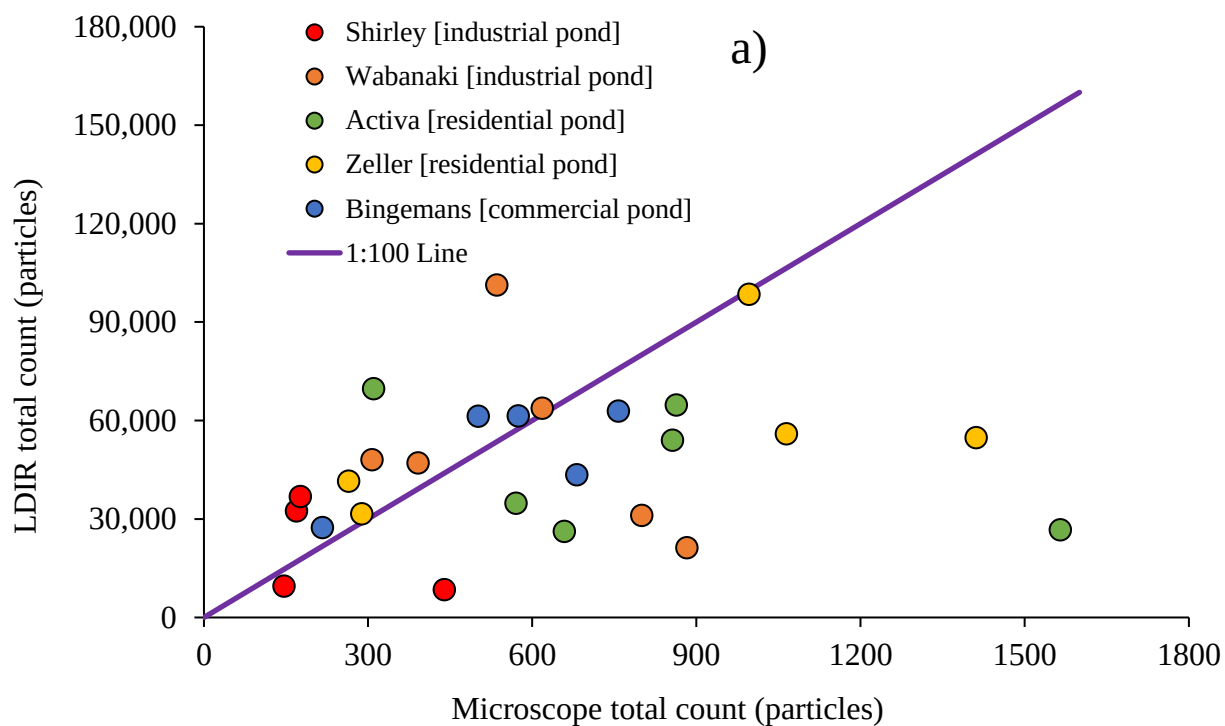


Figure A-6: Comparison of total MP particles count between microscope and a) all particles detected by LDIR and b) only particles larger than 200 μm detected by LDIR.

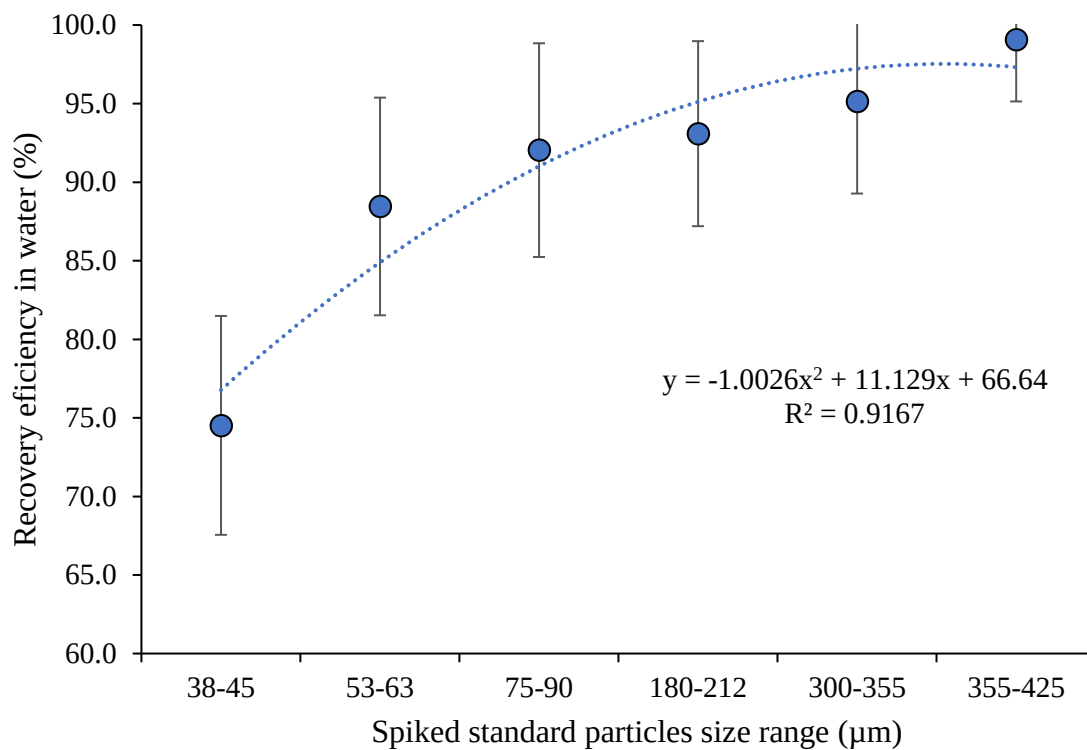


Figure A-7: The recovery efficiency (%) of spiked standard particles in various size ranges in water samples. A polynomial regression and its R^2 value are presented.