

ABUNDANCE OF MICROPLASTICS IN SEDIMENTS FROM THE URBAN RIVER IN MONGOLIA

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Abstract Microplastics (MPs) are ubiquitous pollutants of the aquatic environment; however, the behavior of these materials in freshwater environments is largely unknown. The present study aimed to identify types of MPs and quantify their concentration in the river bottom sediment in the Tuul River, northern central part of Mongolia. The goal was also to analyze and evaluate the behavior of MPs. Six sampling plots were chosen as the research objectives, which were close to the junctions between tributaries and the main river of Tuul in the capital of Mongolia, Ulaanbaatar. All sediment samples contained MPs with an average concentration of 603 ± 251 items kg^{-1} . The major morphotype was synthetic fibers, which originated from polyester and polyamide polymers. While the size of MPs ranged between $28.4 \mu\text{m}$ and $3409.1 \mu\text{m}$, most of these materials varied between 100 and $200 \mu\text{m}$. Furthermore, the finer the particle sizes of sediments, the higher the number of detected MPs. Distribution of the MPs in the study area indicates that the point-source of MPs such as wastewater treatment plant strongly affects their concentration. However, domestic wastes (i.e., plastic litter) impact the distribution of MPs as non-point sources.

Keywords: Tuul River, sediment, microplastics, size distribution, micro-FTIR

1. Introduction

Plastic debris has been widely spread and recognized in aquatic environments for many years. This phenomenon is a consequence of the exponential increase of global plastic production and consumption since the 1950's (Barnes *et al.* 2009). Plastic abundance has been reported in the ocean (Lusher *et al.* 2014), sea (Jayasiri *et al.* 2013), and freshwater (Miller *et al.* 2017; Sadri and Thompson 2014) environments with different orders of magnitude across the worldwide sampling locations. Debris of artificial products from terrestrial environments affects the accumulation of plastic debris in the marine ecosystem through river flow. It has been estimated that between 4.8 and 12.7 million tons of plastics end up in the oceans as a result of river emissions (Jambeck *et al.* 2015). In addition, rivers are not only considered as major pathways of plastic transportation to the open water systems but also reservoirs of plastics on the shore (Battulga *et al.* 2019) and in the bottom sediments (Anderson *et al.* 2016) affected by seasonal changes and hydrological conditions.

Owing to the small sizes, large surface areas, resistance to degradation, and ubiquitous distribution in aquatic environments, microplastics (MPs, defined by size below 5 mm in any dimension) have been of great environmental issues. In recent years, the toxicity of MPs resulting

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from interactions with various pollutants (Jang *et al.* 2017) as well as ingestion by biota (Wesch *et al.* 2016) has been investigated in marine and freshwater ecosystems.

Plastic is one of the carriers of synthetic polymers, organic pollutants, and trace metals, which leads to transfer of these contaminants as vectors into the aquatic food web (Koelmans *et al.* 2013). Although ingestion of MPs by fish, macroinvertebrates, and zooplankton is evident (Vroom *et al.* 2017), influence of MPs on aquatic organisms is not thoroughly understood. It is important to assess the behavior of MPs relating to biotic as well as abiotic processes in the aquatic ecosystems. Although fine sediments can be nutrient reservoirs and bases of aquatic food systems, they concurrently collect various types of contaminants including heavy metals and persistent organic pollutants (Liu *et al.* 2016). Microplastics also accumulate in river bottom sediments during their fluvial transportation (Blašković *et al.* 2018), causing a significant threat to habitats. The precipitated MPs can act as food sources for a range of organisms and affect the health of the entire riverine ecosystem. In biological systems, MPs become sediments through biofouling and hydro morphological processes (Fazey and Ryan 2016) due to changes in the surface properties on plastic materials accompanied by their gradual sedimentation at the bottom of rivers (Kowalski *et al.* 2016; Vermeiren *et al.* 2016). Studies of coastal environments have demonstrated that urban environments with high population density are significant sources of MPs in river systems (Peng *et al.* 2018). Nevertheless, there are only few studies focusing on river shores where the dynamics of MPs can be affected by changes in the water level. Hence, it is necessary to understand the processes associated with MP sedimentation in rivers near areas with high population density.

Although the urban area of Ulaanbaatar, the capital city of Mongolia, occupies just 1% of the total area of the Tuul River watershed, the sewer water from the city greatly influences the mainstream. The major pollution of the river is a consequence of existence of sewage plants with low capacity, improper waste management, and inappropriate industrial and mining activities (Altansukh and Davaa 2011). This is despite the crucial role of the river water as the only drinking water source in Ulaanbaatar. High population density, rapid urbanization, and intense industrial activities have considerably impacted the basin, leading to severe environmental deterioration, including shortage and pollution of drinking water (Nadmitov *et al.* 2015; Batbayar *et al.* 2017). With the help of volunteers, the city municipal offices organize daily and seasonal clean-up activities; however, the waste dispersion in the river basin originating from the populated areas has been out of control (Battulga *et al.* 2019).

We have previously reported a study considering the distribution of plastics on river shores, focusing on the Selenga River basin in the central northern region of Mongolia (Battulga *et al.* 2019). However, investigation of MPs in river sediments remains unexplored. As a consequence, we decided to examine on the Tuul River basin located in the Ulaanbaatar region, which is highly contaminated by waste plastic loads from the city. Behavior and distribution of MPs in sediments allow us to gain better understanding of the dynamics of plastics in the freshwater system. Given the limited knowledge on the MPs in urban river sediments, the current study is designed with the following objectives: (1) identification and quantification of MPs in river bottom sediments, (2) investigation of behavior of MPs in river sediments, and (3) comparison of the distribution of MPs among the world rivers.

2. Materials and methods

Study area and site description

The Tuul River Basin located in the northern central part of Mongolia lies ranging between 48°30'N and 48°56'N, and from 104°48'E to 108°13'E (Fig. 1). The study area is located in Ulaanbaatar along the Tuul River (Fig. 1). The Tuul River in this region connects with four main tributaries, which are Gachuurt, Uliastai, Selbe, and Tolgoit streams located from east to west. These tributaries predominantly flow in the middle of Ulaanbaatar city. The Gachuurt and Uliastai tributaries flow in the less populated areas in the eastern districts. However, residents of the city enjoy camping and outdoor activities along the Tuul River in near the confluence point with Gachuurt tributary because of better water quality (Batbayar *et al.* 2017; Dolgorsuren *et al.* 2012). Herein, the upper region, located approximately 40 km from the city center, is comparatively less affected by various land use activities. Moreover, at the time of the study there were no significant industrial works along the Tuul River. However, one of the largest organic farming agricultures is located in this area.

The lower reach of the Uliastai tributary, a confluence to the Tuul River, has been protected by the government as the drinking water source. The Selbe tributary runs through the most populated area in Ulaanbaatar, collecting all drainage from residential and industrial areas. The watershed of the Tolgoit tributary has been occupied by traditional Mongolian yurts “Ger” as well as modern concrete apartment blocks. There is a social problem of management of the waste released from the Mongolian yurts (Byamba and Ishikawa 2017). Major industries such as textiles, tanning, food processing, and fire power stations are also located in this watershed.

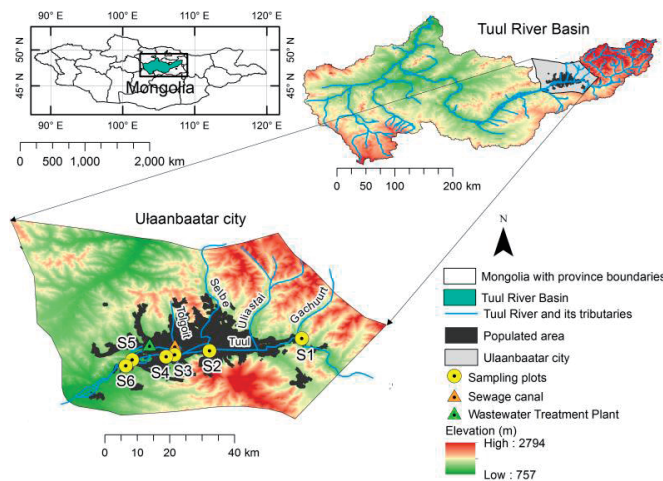


Fig. 1 Study area and research plots.

Six research plots were set along the Tuul River in March 2018 (Fig. 1). The area between Gachuurt and Selbe tributaries was omitted for sediment sampling because of less anthropogenic activities. The S1 plot was located at the upper stream of the Tuul River, close to the confluence point with the Gachuurt tributary (Fig. 1). The remaining S2, S3, and S4 plots were located downstream of the Tuul River between the confluence point with the Uliastai tributary and the wastewater treatment plant (WWTP). The S3 plot was the closest point from the mouth of Selbe,

to which the access was prohibited. S5 and S6 are the downstream plots of central WWTP, located just after the confluence of the Tolgoit tributary.

Sediment sample collection and preparation

At each plot, triplicate samples were randomly collected to the depth of 5 cm using a stainless-steel shovel. All sediment samples were dried at room temperature for over two weeks. Sediment samples were sieved into three fractions and separated by particle sizes. Gravels (>2 mm) were separated using a 2 mm sieve. The coarse sand fraction was collected from the sediment samples previously passed through a 2 mm sieve using a 0.5 mm sieve. The sediment sample, which passed through a 0.5 mm sieve, was collected as the fine particle fraction. Microplastics in the three particle size fractions were used for further analyses. To remove natural organic materials, the size fractions of sediment samples were digested with 30% hydrogen peroxide (H₂O₂) solution in the presence of an iron (II) catalyst solution utilizing the Wet Peroxide Oxidation method (Masura *et al.* 2015). The digestion was conducted on a hot plate (at approximately 75 °C) until no organic matter was visible in the mixture. Subsequently, the mixture was processed in an ultrasonic bath for several minutes to destroy aggregates of plastic and sediment particles. The extraction of MPs was then conducted using the density separation method (Ballent *et al.* 2016). Sodium polytungstate powder was added to the mixture to obtain a final density of 1.5 g cm⁻³ and stirred for 5–10 min using a magnetic stirrer. The mixture was allowed to settle for 20 min before being transferred to a centrifugation tube. Following centrifugation at 4000 rpm for 30 min, the supernatants were filtered through a 47 mm membrane filter with 0.2 μm pore sizes (Millipore, Co Ltd.). A complete recovery of total MPs was achieved by carefully rinsing the filter and the filter holder with deionized water during the filtration process. Finally, the obtained samples were dried.

Plastic quantification and identification

Morphotypes, colors, sizes, and abundance of MPs were examined under a digital microscope with 100× magnification (VH-7000, Keyence, Japan). The observed MPs were categorized into four morphotypes (foam, film, fragment, and fiber) and four color categories (translucent, colored, white, and black) according to Crawford and Quinn (2017) and Shabaka *et al.* (2019). The number of MPs was counted under a digital microscopic view with an area of 12 mm². An averaged number of MPs was obtained after six to ten replications of views, which were randomly selected in the filter. The average number was converted to the number of MP items in a theoretical area of the filter which collected whole of MPs from the applied sediment fractions. In addition, the sizes of MPs were determined using a scale with a 50 μm resolution in the microscopic view. The size distribution of the MPs was classified based on their sizes with 100 μm intervals under the digital microscopic view. Kernel density estimation (KDE) is commonly used to estimate the probability of density function of continuous random data (Rajagopalan *et al.* 1997). In this case, distribution of sizes of MPs is estimated employing the size data from the microscopic views. The KDE data were obtained utilizing the R program (R studio ver. 3.6.0) and using the following equation,

$$\hat{f}_n(x) = \sum_{i=1}^n \frac{1}{nh_i} K\left(\frac{x - x_i}{h_i}\right) \quad (1)$$

where K is the kernel function centered on the observation of i th targeted plastics (x_i), h_i is the

bandwidth of the size distribution or “scale” parameter of the kernel centered at x_i and n is the number of target MPs.

Infrared (IR) spectroscopic analysis was performed using micro-Fourier-transform infrared (FTIR) spectroscopy with reflectance mode (Shimadzu Co. Ltd., Kyoto Japan) to identify the origin of MP materials. Infrared spectra in the range of 400–4000 cm^{-1} were recorded after 100 scans of the MPs on the filter. The smallest microscopically analyzed area was a $10\ \mu\text{m} \times 10\ \mu\text{m}$ square. The conditions used for the FTIR analysis were analogous to our previous study (Battulga *et al.* 2019). The peaks and shoulders observed in the spectra were assigned according to the Aldrich library of FTIR spectra to determine the polymer types (Pouchert 1985).

3. Results and discussion

Occurrence and identification of MPs in sediment

The average number of MPs found in sediment samples was 603 ± 251 items kg^{-1} . The highest number of MPs was recorded at S6 and amounted to 998 items kg^{-1} , while the lowest was 312 items kg^{-1} at S2 (Fig. 2). The S6 plot was located downstream and was exposed to daily discharge of wastewater from the central WWTP. It was found that continuous deposition of plastic fragments from the city center through the river flow enhances the number of plastics at sites following S3. The lower number of MPs at S2 could be a consequence of the newly developed sewage system along the river coast, where new residential areas were constructed. Furthermore, sources of MPs were diverse due to various land uses in the Tuul River watershed in Ulaanbaatar. Various concentrations of MPs in the bottom sediments (Fig. 2) may be a result of the release of plastic waste, poorly controlled waste management (Batsaikhan *et al.* 2018) in the residential (mainly S1, S3, and S4), recreational (S1), agricultural (S2), and industrial areas (S3–S6) as well as plastic self-degradation into MPs.

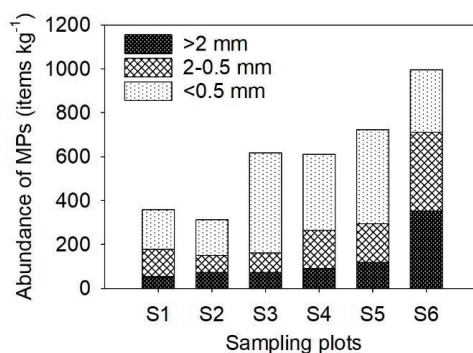


Fig. 2 Abundance of MPs in the three particle size sediment fractions at each sampling plot.

In the fine particle fraction, the abundance of MPs at plot S3 was 455 ± 96 items kg^{-1} , revealing that MP pollutants were heavily loaded to the Tuul River through the sewage discharge from the city center (Fig. 2). With the exception of plot S6, larger amounts of MPs were observed in the fine particle fraction at all sampling plots (Fig. 2). Moreover, MPs were continuously precipitated along the river as bottom sediments. MPs from the central WWTP increased the concentration of MPs in the coarse sediment fraction (Table 1). The differing abundance of MPs at each sampling

plot with varying MP composition in sediment size fractions (Fig. 2) might be explained by environmental factors, which affect the precipitation of minerals and MPs (e.g., water current and micro-topography on the shore), as well as aging (e.g., degradation, fragmentation, physical and chemical changes, and biofouling) (Ballent *et al.* 2016). Under environmental conditions, most plastic debris becomes brittle over time. The aged MPs form biofilms on rough surfaces and the increase of the specific density leads to precipitation. Additionally, during the transfer of plastic debris, the particles interact with other substances in the aquatic environment and form aggregates with natural and anthropogenic products. These results in an increase in the density of aggregates precipitated in the sediment (Koelmans *et al.* 2017). The aforementioned phenomena affect the MP deposition and distribution patterns in the river sediment.

Table 1 Average concentration of MPs by morphotypes in the studied plots ($n=3$; items $\text{kg}^{-1} \pm \text{Stdev}$)

Sampling plots	Foam	Film	Fragment	Fiber	Total
S1	84 \pm 13	72 \pm 9	74 \pm 18	129 \pm 24	358 \pm 41
S2 [†]	78	68	84	82	312
S3	125 \pm 28	96 \pm 15	153 \pm 57	245 \pm 51	619 \pm 126
S4	132 \pm 9	133 \pm 16	160 \pm 4	183 \pm 19	609 \pm 30
S5	142 \pm 19	162 \pm 33	185 \pm 47	234 \pm 24	723 \pm 70
S6	171 \pm 15	160 \pm 5	199 \pm 17	468 \pm 380	998 \pm 373

[†]no replicates

Among the different morphotypes (Rezania *et al.* 2018), fiber-type MPs account for 35% of the total MPs and dominated in the investigated sediments. The occurrence of the remaining three morphotypes showed similar distribution (Table 1). Fibers were previously confirmed as the dominant plastic type in the river and coastal sediments (Baptista *et al.* 2019). In the Chinese Wei River, the concentration of fibers accounted for 42.25–53.20%, which was the highest compared to other types of plastics (Ding *et al.* 2019). The prevalence of fibers in marine bottom sediments of the Southern Baltic Sea was described by Graca *et al.* (2017). Laundry wastewater is suspected to be a significant source of MP fibers, which can be transported to river environments by wastewater drainage (Haave *et al.* 2019). In fact, sampling plots S3 and S4 investigated in our study were located downstream of the city sewage canal. Moreover, plots S5 and S6 were also positioned just after the wastewater effluent discharge.

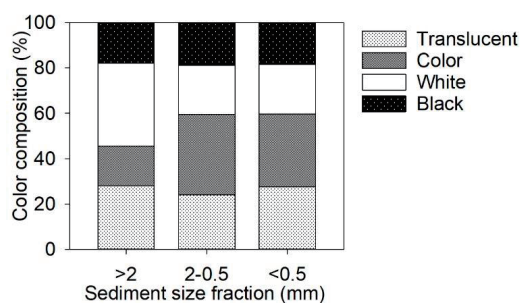


Fig. 3 Color composition of MPs extracted from sediment samples.

The drainage of wastewater enhances the micro-plastic fibers contamination in river bottom sediments. Although it was determined that the polystyrene foam (PSF) was dominant on the Tuul River shore (Battulga *et al.* 2019), it was not the main type of plastic in the bottom sediment due to its low density and light weight in comparison to fibers released from sewage and wastewater effluents. The distribution of MPs in the riverbed sediment can be attributed to closely located point-source locations. Water stagnation accompanied by low flow rate at S3 and S6 plots can promote sedimentation of MP fibers in the river sediments (Table 1). Similar processes of sedimentation of MPs have been described for the English Channel, UK (Browne *et al.* 2011). From the geographical viewpoint, proportionately more plastics were deposited downward of the river flow, with slow-moving waters along the estuarine shoreline.

Assessment of the morphotypes depending on colors (translucent, colored, white and black) revealed that the prevalent color category was white (36.6%) in the sediment fraction of >2 mm, whereas the colored MPs dominated in other size fractions (35.4% for 0.5–2 mm and 32.0% for <0.5 mm, Fig. 3). The diversity of color indicated various sources of MPs, while white color mostly originated from PSF debris in the Tuul River sediment (Battulga *et al.* 2019). Furthermore, the translucent MPs in the collected samples were mainly found in the films, whereas black and colored MPs were derived from the fibers and fragments. Overall, the examined MP samples were characterized by a variety of colors and types.

Size distribution of MPs and polymer characterization

The size of MPs ranged between 28.4 μm and 3409.1 μm . The majority of MPs in the sediments were below 1 mm (95.2%). In addition, over 70% of MPs were ranged between 100 and 400 μm in each sediment fraction (Figs. 4a–4c).

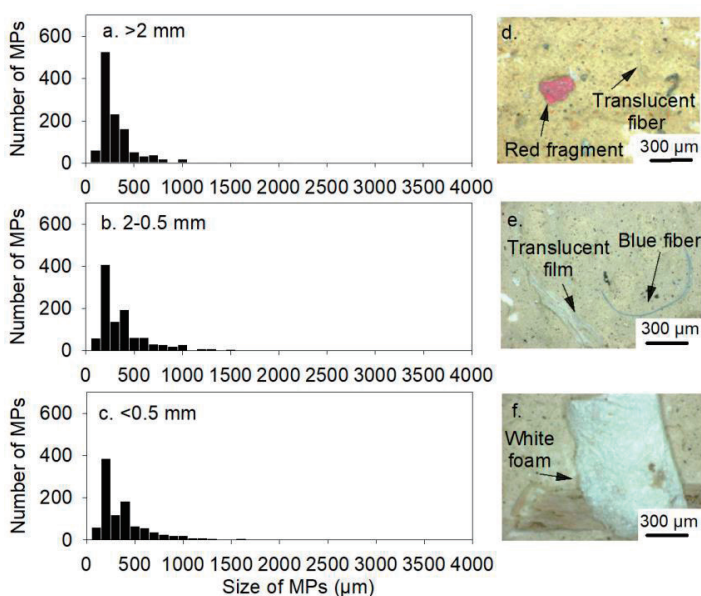


Fig. 4 Size distribution of MPs in (a) >2 mm, (b) 0.5–2 mm; and (c) <0.5 mm size fractions of sediment samples and microscopic images of morphotypes of plastics (d-f).

Microplastics with the size range between 0.5 and 0.99 mm occupied the largest proportion of precipitated materials of plastics from the Jiaozhou Bay, north China (Zheng *et al.* 2019). Moreover, Bergmann *et al.* (2017) reported that 99% of all particles in the Arctic deep-sea sediments were smaller than 0.15 mm. Another study, which focused on the St. Lawrence River (Canada) sediments, demonstrated that most (99.9%) of the microbeads were below 2 mm in diameter (Castañeda *et al.* 2014). Both the previously reported results and our study show that fragmentation leads to a large number of smaller sized MPs in the sediments. In the current study, various sizes and shapes of MPs were found on the filters (Figs. 4d–4f). Furthermore, a portion of MPs was regarded as degraded and they exhibited minute cracks and rough surfaces. The highest number of MPs was in the range of 100–200 μm in all sediments (Figs. 4a–4c). However, the second highest fraction was the 200–300 μm MPs in the sediments over 2 mm in size (Fig. 4a). In addition, the 300–400 μm fractions were higher than the 200–300 μm fractions in the sediments below 2 mm (Figs. 4b and 4c). Importantly, the size ranges observed during the microscopic analysis were different among the various MP morphotypes (Fig. 5). Half of foams and fragments were in the size range between 100 and 200 μm . On the other hand, films and fibers have a larger range of MP size distribution.

The size distribution of MPs expressed by KDE is different between morphotypes (Fig. 5). Size distributions of foam and fragment MPs are characterized by the sharp peak attributed to the highest proportion of the size at 150 μm . The observed distributions indicate dominance of smaller sized foams and fragments in the bottom sediments. In addition, film type MPs exhibits a slightly shifted peak in the larger size region over 150 μm , whereas the fiber-type MPs display a broad range of sizes with the broad peak at the larger size.

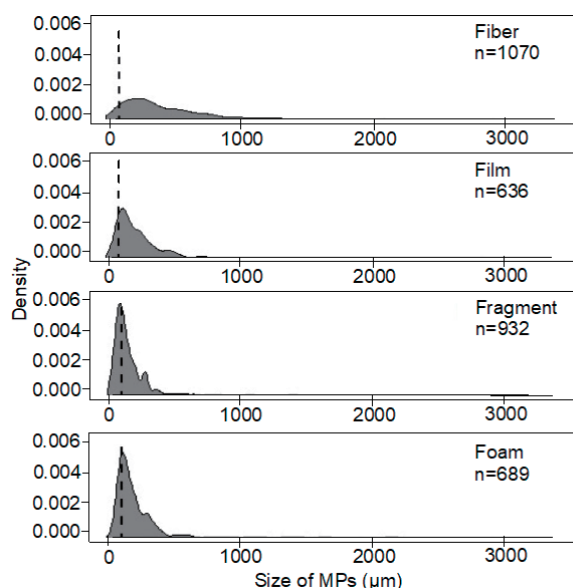


Fig. 5 Distribution of Kernel density calculated by R program with the dependency of particle sizes morphotypes of MPs in the river bottom sediments. The higher the Kernel density, apparent frequency of abundance is higher. The dashed line indicates 150 μm which was the highest density observed in the Foam and Fragment type of plastics.

Plastic bags used for packaging are the major source of micro-films in the environment due to their light weight as well as large amount of production for single-uses (Luijsterburg and Goossens 2014). Because of bulky plastic bags and their daily consumption, film type plastic litter is widely distributed in the environment, resulting in the wide distribution of their sizes indicated in our study. The wide distribution of sizes can be attributed to different fragmentation status during their travel in the environment. Moreover, synthetic fibers can be released from the discharge of WWTPs (Napper and Thompson 2016) and direct drain of sewage water from the surrounding households through inappropriate waste management (Lahens *et al.* 2018). A broad range of fiber sizes suggests that during the washing process, fibers of different sizes are released depending on the type of textile material and the duration of consumption (Hartline *et al.* 2016; Yang *et al.* 2019). Additionally, a recent estimation specifies the more than 6,000,000 micro-fibers are released from a 5 kg wash load of textile fabrics (De Falco *et al.* 2018). Undoubtedly, this phenomenon highly influences the sizes and variety of polymers of micro-fibers in the aquatic system.

The collected MPs were identified as polyester (PEs), polyethylene (PE), polystyrene (PS), acrylonitrile butadiene styrene (ABS), polyvinylchloride (PVC), and polyamide (PA) based on their corresponding FTIR spectra (Fig. 6). Within these typical MPs, PE, PVC, and PS are the most common materials in the world plastic market (PlasticsEurope 2015) and they were dominantly found in the Tuul River sediments. Furthermore, fibrous MPs made from PEs and PA polymers are also widespread in aquatic environments (Kanhai *et al.* 2017). Density of plastic materials and their surface roughness affects the retention of MPs in sediments. A floating PE plastic cannot sink to the bottom due to its low density ($<1 \text{ g cm}^{-3}$), while aged PE plastics lose their buoyancy as a result of biofouling and changes in the surface status, mainly by photo-oxidation (Kaiser *et al.* 2017). The non-buoyant polymers (PA, PS, PVC, and ABS) were also commonly detected from the IR spectra (Fig. 6).

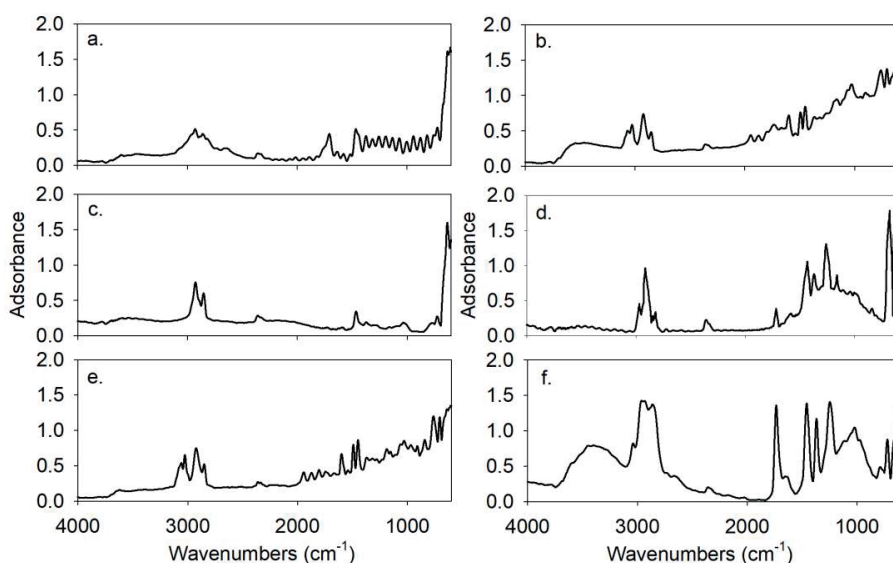


Fig. 6 FTIR spectra of MPs extracted from sediment samples. The spectra were identified as a. polyester, b. acrylonitrile butadiene styrene, c. polyethylene, d. polyvinylchloride, e. polystyrene, and f. polyamide polymers according to Aldrich library of FTIR spectra.

Overall, our results demonstrated that the detected MPs are largely influenced by environmental and anthropogenic factors, which lead to alteration of the surface properties of plastic materials. In addition, the reformed surface status affects the distribution pattern of MPs in the aquatic environment.

Comparison of MPs content with worldwide studies

The number and sizes of MPs in river bottom sediments reported in several other studies are listed in Table 2. It can be seen that the abundance of MPs in the Tuul River sediments was one or two orders of magnitude lower than in the Atoyac River in Mexico (Table 2). However, comparable amounts of MPs were determined in the River Thames (UK), Wei River (China), and rivers in Shanghai (China) (Ding *et al.* 2019; Horton *et al.* 2017; Peng *et al.* 2018). The differences in the data among the studies can be attributed to the background of land use as well as different sampling methods and techniques used for identification of MPs in the sediments (Eerkes-Medrano *et al.* 2015).

The results also indicate that local environmental parameters such as wind, precipitation, UV radiation, and river flow affect the MPs transportation and accumulation in the river environment (Nel *et al.* 2019; Piperagkas *et al.* 2019). For instance, snow-melting and strong winds in the spring season in Mongolia could increase the flow rate and lead to mobilization of sediments containing previously settled plastic particles. On the other hand, the degradation of larger plastics from landfill sites must be taken into account when considering further distribution of MPs to river environments. As previously mentioned, degraded plastics or aged MPs are ingested by various aquatic organisms (Vroom *et al.* 2017), which poses serious ecological risks. Smaller MPs can enter into the higher or lower trophic levels, which cause further adverse effects in aquatic ecosystems. Recently, ingestion of MPs by freshwater fish has been reported in the River Thames in the UK (McGoran *et al.* 2017). Other results show that 45% of sunfish are exposed to MPs in Brazos River Basin, USA (Peters and Bratton 2016). Microplastics are also detected in the gut of 11 different species of fish living in the Río de la Plata estuary in Argentina (Pazos *et al.* 2017). The toxic effects of MPs include primarily intestinal damage and digestive stress to aquatic biota (Lei *et al.* 2018).

Table 2 A comparison of abundance and size distribution of MPs in river sediments across the worldwide sampling locations

Country	Location	Abundance of MPs	Dominant size classes	References
UK	River Thames	18.5±4.2 to 66±7.7 particle/100 g	1 mm–2 mm size was higher than 2–4 mm size	Horton <i>et al.</i> (2017)
China	Rivers in Shanghai	802±594 items kg ⁻¹	<100 mm - 31.19%, 100–500 mm - 62.15%	Peng <i>et al.</i> (2018)
Indonesia	Ciwalengke River	3.03±1.59 MP particles/100 g	1000–2000 µm size was higher than 50–100 µm	Alam <i>et al.</i> (2019)
China	Wei River	360 to 1320 items kg ⁻¹	<0.5 mm is 40.8% to 68.8%	Ding <i>et al.</i> (2019)
Mexico	Atoyac River basin	4500±702.23 items kg ⁻¹	-	Shruti <i>et al.</i> (2019)
Mongolia	Tuul river	603±251 items kg ⁻¹	100–200 µm size	This study

In the case of Mongolia, although there is no official record of MP exposure in aquatic organisms, the MP pollution has been reported in the Selenga River system (Battulga *et al.* 2019) and the largest freshwater lake, Hovsgol (Free *et al.* 2014), where the probability of MPs exposure to fish is increased. For these reasons, comprehensive understanding of MP behavior in the river environments is necessary to focus the attention on the growing concerns of ecological consequences of MP pollution. Therefore, more studies covering spatial and temporal distribution as well as risk assessment of MPs are currently required to comprehend the interaction of these pollutants with other natural or anthropogenic substances in the aquatic environment.

4. Conclusion

In this study, the abundance, distribution pattern and behavior of MPs in the urban river sediment of Mongolia have been investigated. Although sediment samples from the Tuul River contain various size and shape of MPs, the dominance of smaller size (100–200 μm) and fibrous MPs illustrate that the studied urban river have received heavy load of released MP particles and fragmented MP debris. Moreover, most of the plastic debris appears to be smeared with organic substances such as humic substances or algae, resulting in their deposition as sediments. It is noteworthy that fine particle sediments ($<0.5\text{ mm}$) have a great potential to collect MPs in the river environment.

Regarding to environmental fate and behavior, hydrodynamic forces (turbulence, stratification and plume fronts) influence on plastic items to follow circulation, dispersion, suspension, and settling pathways, which are reason for hotspots distribution in the downstream. In addition, point and non-point sources of pollution and polymer characterization have been led to unexpected accumulation patterns of MPs in the fluvial ecosystem. Furthermore, present study, which highlighted the behavior and distribution of MPs in the river bottom sediment, illustrates the novel aspects of widespread aquatic plastic pollution in terrestrial inland river system. In future research work, MPs aggregation and their formation mechanisms, and aggregates ingestion by terrestrial aquatic biota are necessary to investigate in order to understand adverse effects of MPs on wildlife and ecosystem.

Acknowledgements

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