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Abundance, Characteristics, and Microplastics Load in Informal Urban Drainage System Carrying Intermixed Liquid Waste Streams

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ABSTRACT

This first-of-its-kind study systematically assesses the abundance and characteristics of Microplastics (MPs) in different categories of informal open drains (nallas) carrying different liquid waste streams from different functional areas of an Indian city. Such drains are part of the informal urban drainage system that carries wastewater, stormwater, industrial effluent, and rural runoff. Logistical and locational limitations of traditional wastewater (WW) sampling methods severely limit their application in open drains. To overcome sampling challenges owing to complex geography, vast drainage network spread across different functional areas of the entire city, and local challenges, appropriately modified sampling strategies were adopted to collect samples from 35 open WW drains (small/local, intermediatory, and large). MPs (50µm-5mm) were present in a bucket, and net samples obtained from all 35 WW drains. The average MP concentration in WW drains was 4.20 ± 1.40 particles/L (bucket samples) and 5.19 ± 1.32 particles/L (net samples). A declining trend of MPs abundance was observed from larger to smaller drains, confirming that smaller and intermediatory drains (carrying WW from different functional areas of the city) are discharging their MP loads into larger drains. Intermixing different WW streams (municipal WW, stormwater surface runoff, agricultural runoff, and industrial WW) increases MP levels in drains. The local riverine ecosystem is being put at risk by a daily MPs load of 12.6 x 10^8 particles discharged from 9 larger drains into the local river Kharun. To protect the riverine ecosystem, controlling the high daily MPs load from such drains is important. Diversion of WW drains through constructed wetlands built near river banks can be a cost-effective solution. Because the entire Indian subcontinent and parts of Africa rely mainly on such drains having similar characteristics and local conditions, the findings of this study reflect the status and pattern of MPs pollution in informal drains of the entire Indian subcontinent and can be used by stakeholders and governments to take mitigative and preventive measures to manage the MPs pollution and protect the local riverine ecosystem.

INTRODUCTION

Microplastics (MPs) Distribution in Different Types of Liquid Waste Streams

Microplastics (plastic particles with sizes between 1 µm-5 mm) are an emerging class of pollutants and multi-dimensional environmental stressors, capable of transboundary migration (Jong et al. 2022), threaten the local, regional, national, and global ecosystems due to their diverse characteristics, ubiquity, and ecotoxicological effects (Rochman 2019). Different types of liquid waste (LW) streams originate from different anthropogenic and natural activities in any semi-urban area or urban agglomerate. These include – municipal wastewater (MWW) from residential dwellings, institutions, and public places, stormwater (SW) runoff, industrial wastewater (IWW) (treated and untreated depending upon local regulations), and rural WW (agricultural wastewater (AWW) and rural runoffs from other non-point sources). Characteristics of these streams differ, and their management may be done separately or together. Wastewater (WW) management practices vary locally, regionally, and nationally, influencing the fate of WW-associated MPs and traditionally involve – the collection of liquid waste (LW), their conveyance through drainage systems, treatment, disposal, and reuse. Conveyance conduits are designed as a separate, partial, and combined system based on the provision of SW collection.

Wastewater treatment plants (WWTPs) are a major pathway of MPs to ecosystems where effluent and sludge are disposed of or reused. High MPs levels are present in WWTPs influents, effluents, sludge, and treatment units. MPs (10-5000 µm) concentration in effluent of 79

WWTPs varied between 0.004 to 450 particles/L with an average of around 6.4 particles/L (Schmidt et al. 2020). The observation that WWTPs are a pathway for MPs comes from studies conducted in developed nations. To our knowledge, their presence in WW conveyance systems has not been explored. Until 2019, MP levels have only been assessed in 121 WWTPs located in 17 nations spread across Europe (53%), USA and Canada (24%), Asia (18%), and Australia (5%). Among Asian WWTPs, most studies originate from China and Korea, and one from Iran, Thailand, and Turkey (Yaseen et al. 2022). No study (either in WWTPs or in WW conveyance systems) has been conducted within developing nations of the Indian subcontinent, one of the most densely populated regions with similar geographic, climatic, and social characteristics.

The type of sewerage system and functional areas of the city from where WW originated influence the distribution of MPs (Yang et al. 2022). In water conduits, those flowing through commercial and residential areas are more abundant with MPs than on campus and highways (Sang et al. 2021). SW conveyance conduits and management sites are abundant with MPs (Shruti et al. 2021). Combined sewerage systems and SW collection networks have a high fraction of larger MPs, including tire wear MPs mainly relating to road dust (RD)-associated MPs (Wang et al. 2022). In separate sewerage systems carrying MWW from residential areas, fibrous MPs from laundry activity and microbeads from personal care products (PCPs) are the most abundant shapes of MPs (Hamidian et al. 2021, Ziajahromi et al. 2016).

High levels of MPs are observed in IWW (Yuan et al. 2022), and a strong positive correlation exists between MPs in WWTPs influent and several plastic industries in the study area (Long et al. 2019). MPs concentration is reportedly twice as high in IWW than in MWW of urban habitat (Franco et al. 2020). WWTPs treating MWW & IWW in highly industrialized areas report higher levels of MPs than less industrialized areas (Wei et al. 2022). Elsewhere, MPs abundance in MWW, IWW, and AWW is of the same magnitude (Wang et al. 2020). Due to the extensive use of plastic products and MPs ladened biosolids as soil conditioners in agroecosystems, rural runoff from point (rural WW (RWW), and agricultural wastewater (AWW)) and non-point sources are also abundant with MPs (Yano et al. 2021).

Given the ubiquity of MPs in MWW, IWW, SW, and rural runoff as reported in the literature, we hypothesize that if such streams intermix together, it will significantly increase MPs load and change MPs composition in intermixed WW stream. A lack of study deals with intermixing four such LW streams, limiting the scientific understanding of the implications of intermixing. Before conducting any study, it's important to understand scenarios and reasons for intermixing LW streams.

Informal Urban Drainage Systems as a Carrier of MPs

In developing economies like India, habitats lacking centralized WW management systems (underground pipes, pumping stations, and treatment plants) and sewage collection systems typically use onsite sanitation systems (OSS) in the form of individual septic tanks (ST). Households have built-in STs to treat black water. The supernatant of ST and sullage from households is discharged into nearby WW surface drains or nallas (closest translation will be natural or man-made WW brook or WW drain or WW canal or WW channel), relying mainly on the natural drainage system. Such drains or nallas can be natural or man-made, and they are basically of two types- *Kucha* (unlined) and *Pucca* (lined) (CPCB 2020). A vast network of small and large earthen and man-made drains across different functional areas connects entire cities. Lager drains discharge WW to local aquatic sources or treatment facilities (Fig. 1).

IWW, particularly from the small industries or unregulated and informal industries, is also sometimes transported and discharged and mixed with MWW drains, despite strict governmental regulations forbidding intermixing of IWW and MWW or IWW discharge without treatment (Amerasinghe et al. 2013). Like the absence of the WW collection system, the SW collection network is also deficient; such WW drains collect both SW and WW, somewhat similar to the combined sewerage system. Intermixing of SW (Shukla et al. 2020) and IWW makes MWW more toxic with organic pollutants and heavy metals, thus threatening the ecosystems where they are being discharged (Vaid et al. 2022). Household gardens, public gardens, and agriculture farms in peri-urban areas discharge agricultural runoff, which also reaches these drains and mixes with the combined WW.

Most WW drains are not covered, and such natural drains or man-made kaccha nallas flow in the open and follow the natural gradient (Fig. 1). Only small portions of pucca nallas are covered, mostly in residential areas. However, even these covered stretches are opened at frequent intervals to facilitate cleaning. Planned residential colonies and gated communities may have underground sewerage, but they eventually meet with open drains. Certain kaccha nallas get lined, or sometimes they are merged with pucca nallas to prevent nallas from changing their course. Open drains flowing through the city become prone to anthropogenic pollution. Garbage and MSW disposal, littering, disposal of construction and demolition waste, and disposal of PW generated from food and catering activities, households, and public places, into the drains, cause obstruction and

Fig. 1: Different types of informal urban drains carrying liquid waste streams.

choking (Fig. 2). This, coupled with negligence in cleaning and maintenance hinders the natural flow of LW and is partly responsible for waterlogging and urban flooding in Indian cities.

Considering the huge gap in management infrastructure, intermixing of LW streams, susceptibility to receive MPs from atmospheric deposits, the volume of untreated WW discharged daily, encroachment, rampant dumping and littering of MSW and PW in the drains, and absence of preventive maintenance, informal urban open drains carrying intermixed WW can be a significant pathway of MPs pollution in an urban environment, which has been overlooked till date (Veerasingam et al. 2020). Though it's difficult to assess the quantity of MPs originating from separate LW streams (due to the mixed land use pattern of unplanned cities) and external inputs (MSW and PW disposal and atmospheric deposition of MPs), we can generate scientific understanding on levels of MPs pollution in open drains conveying intermixed LW streams (domestic WW, industrial, stormwater and agricultural runoff) which receives plastics/MPs from illegal dumping and littering. Such understanding will help plan mitigating measures for MPs and design future studies to quantify the contribution of separate streams and external inputs.

Additionally, scientific findings mainly attribute to WWTP effluent discharge points upstream of the waterbodies for the high levels of MPs observed in downstream sampling locations in waterbodies (Woodward et al. 2021). With small numbers of WWTP discharge points, the MPs' pollution can be addressed adequately. In the absence of a sewerage system and WWTPs, multiple WW drains discharge untreated WW (Intermixed with IWW, SW, and rural runoff and loaded with disposed plastic waste) at multiple locations in local waterbodies. This substantially

Fig. 2: Informal urban drains choked with solid waste.

impacts the quantity and distribution of MPs discharged into local waterbodies. So, knowledge of the quantity of MPs being released from such drains of a particular city is urgently needed to ascertain the risk to the receiving ecosystem.

To fill this knowledge gap regarding MPs pollution in the open, informal drainage system, this study addresses the following objectives – 1) propose sampling strategies to collect samples from different types of open drains with different geographical and channel characteristics, 2) assess the levels of MPs pollution (abundance and characteristics) in such drains, and 3) estimate the daily and annual MPs loads being discharged by these drains in the local river.

MATERIALS AND METHODS

Study Area

The study was conducted in the old residential-industrial capital of Chhattisgarh state of India, Raipur. Raipur is the most populous city in the state and one of central India’s important industrial hubs. According to the 2011 census, the population of Raipur City was 1048112. The revenue boundary of Raipur city lies between 22º33’ to 2114°’N latitude and 8206°’ to 81° 38’E longitudes. Raipur district is spread across a 490.43 km^2 geographical area, with Raipur municipality having an area of 226 km^2 with 70 revenue wards and 7 villages, and the Birgaon municipality governs the remaining area.

Description of Liquid Waste Management in Study Area and Sample Collection Strategy

Currently, the city lacks a planned sewerage system and WW treatment facilities. WW generated from households (1,77,334), institutions, public places, and industries is collected through natural and man-made local (small), intermediatory, and large drains, which are part of the natural drainage system. 144,882 households have OSS-based ST that collect black water. House drains are connected with local drains (RMC 2014).

According to RMC, the city's total water consumption is 246 MLD (Million Liter Per Day), and 150-170 MLD of WW is generated daily, collected, and transported through surface drains, locally known as nallas. Location, discharge data, and other relevant details were obtained from RMC. Sullage and supernatant of ST are discharged in local drains. A vast network of local nallas meets 11 intermediatory nallas and 9 large nallas that cover the entire city. These drains receive MWW, SW IWW, and RWW depending upon location, season, and functional areas from where LW streams originate. Nine large nallas discharge the entire LW of the city in the Kharun River at 9 different locations. Smaller nallas also emit their load in local urban ponds and agricultural lands.

To assess the influence and behavior of MPs from various functional areas of the city and to identify the influences of human activities, WW samples were obtained from 35 locations that covered local nallas, intermediatory nallas, and all larger nallas. Nine sampling stations (WW1-WW9) were located at outfalls of nine large nalla to the river, 11 sampling stations were selected from 11 intermediary nallas that flow through different functional areas of the city, and 14 sampling stations were selected at local nallas. A large

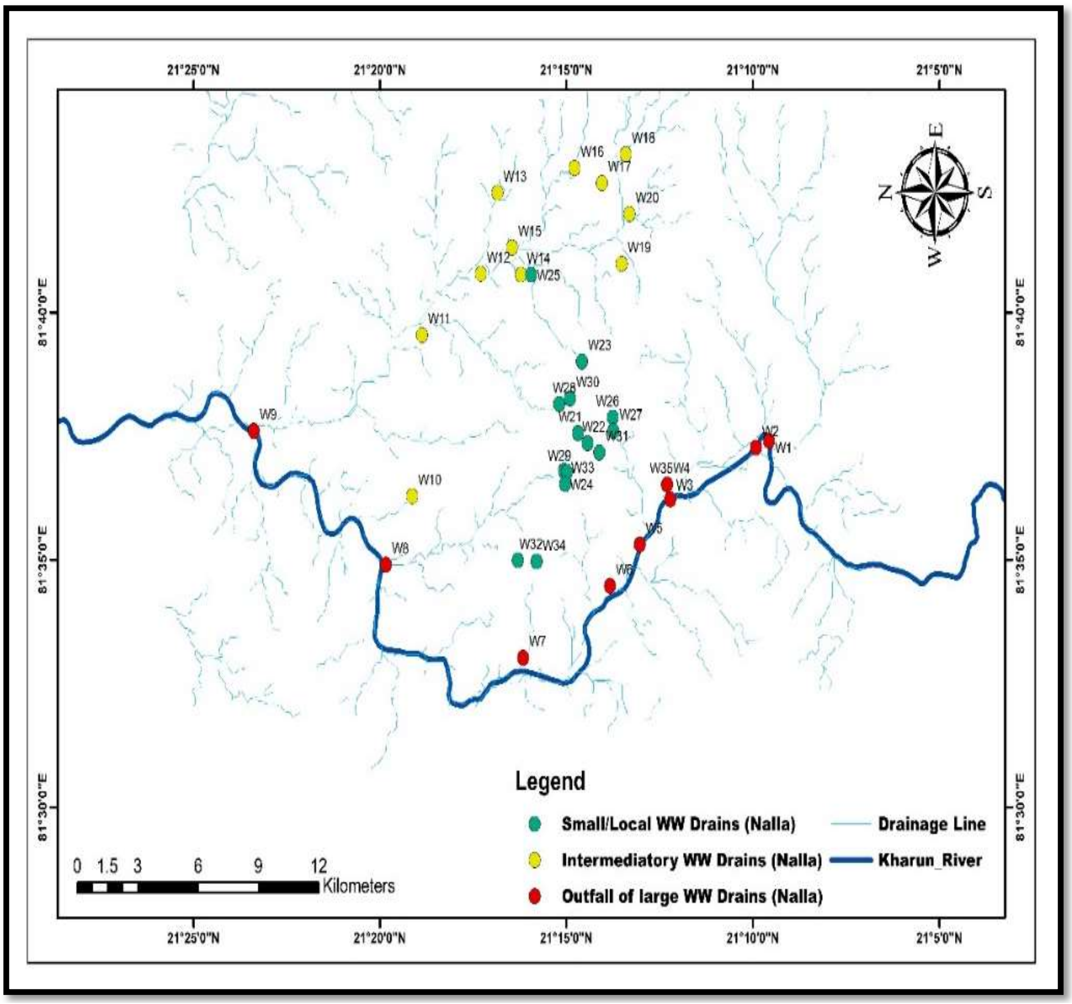

Fig. 3: Study area map depicting 35 nallas sampled in the present study.

Fig. 4: Images of some informal urban drains sampled in the present study.

nalla, the Govardhan drain, passes through a natural wetland spread over 500 m^2 before discharging into the river. To understand the removal efficiency of wetlands, samples were collected from the inflow area (WW35) of the wetland and wetland outfall (WW4) in the river. Samples were collected directly from three smaller nallas (WW32, WW33 & WW34) receiving the MWW from residential dwellings. The study area map is illustrated in Fig. 3, and Fig. 4 depicts images of some drains sampled here.

Sample Collection Methodology

The sample collection was performed using two strategies. In the bucket, rope, and sieve strategy, we used a 5 L steel bucket, 50 µm mesh steel sieve, and 20 m of rope. As the access to the sampling locations became challenging – drains below the roads, drain aqueducts crossing the underpass and roadside nallas, and drains through marshy terrain –buckets and rope were necessary.

Bucket sampling strategy: The bucket was used to collect (through hand or through cotton and jute rope to avoid contamination of samples from plastic ropes) samples from the flowing WW. 5 L of WW was collected from drains and sieved through the steel mesh by slowly pouring the WW from the top. The steel bucket was rinsed with prefiltered double distilled water (DDW) thrice and sieved through the same mesh, which ensured a complete transfer of the sample material. The same process was done 10 times at 15-minute intervals, and a total of 50 L of WW was collected and sieved at each sampling station. The time interval allowed us to address sample heterogeneity and temporal variability, and we obtained a homogeneous and composite sample representative of the WW flowing in the drain.

For roadside drains, samples were collected by drawing the bucket in the opposite direction of the WW flow. Many sampling stations were located near overhead bridges, aqueducts, and underpasses that were not directly or easily accessible. The elevation between the sampling surface and the nallas varied from 5 to 30 meters, so direct sampling was impossible. So, a sufficient length of rope was used to pull the bucket containing WW samples. The handle of the bucket was tied with rope, and the bucket was slowly lowered till it reached the surface of WW flow.

Using a high-weight steel bucket ensured that the bucket did not get carried with the flow. The WW sample was collected in the bucket by maneuvering the rope, and the bucket was lifted by drawing the rope using hands. Using motor pumps was impossible due to geographical, economic, and logistical constraints. The collected WW was sieved and rinsed using the procedure described earlier. The process was repeated 10 times at 15-minute intervals, and a total of 50 L of WW was collected and sieved at each sampling station.

At stations, WW32-WW34, the WW coming directly from the residential dwellings was collected by placing the bucket below the outfall of the plumbing system. The collected WW was sieved and rinsed using the procedure described earlier. The process was repeated 10 times at 15-minute intervals, and a total of 50 L of WW was collected and sieved at each sampling station.

Once the sampling was complete, the residue on the sieves was washed into a jar using DDW. The rinsing process was repeated three times, and finally, the DDW jet was sprayed from the bottom end, which allowed the capture of the MPs clogged in the sieve pores in the collection jar. To overcome clogging and subsequent retention of WW in the sieve due to high suspended impurities in the WW, the bottom portion of the sieve was gently tapped, and the bottom surface of the mesh was tapped using fingers. This unclogged the sieve and allowed WW to pass. Aluminum foil was placed on top of the jar, and the lid of the jar was tightly closed.

Net sampling strategy: The plankton net, rope, and rod strategy involved using a plankton net (mesh size – 50 µm, mouth opening – 13.5 cm, length – 100 cm including the cod end attached to a glass collection jar), tied with a rope and the rope tied to a stick. A mechanical flowmeter (Hydro-Bios) was attached to calculate the volume of WW passing through the net. The length of the rope varied depending on the difference in elevation between the ground surface and the flowing WW in the drain.

The roadside drains, having small level differences, were easily accessible. The net was placed in the drain opposite the WW flow. The flowing WW continued to pass through the net, and the residue was collected in a glass jar. In drains with a small flow, the net was dragged using the stick to sample a sufficient volume of WW. The net was lowered from the roads lying overhead, aqueducts, and underpasses, with high elevation differences, until it reached the surface of the WW flowing in the drain below. The net was suspended in a fixed position for the entire sampling duration at a particular station where the flow was steady. However, the flow was very low or very high at many stations. When there was practically little flow through the net due to the substantially low WW flow, the net was dragged in the transverse direction of the drain by moving the stick from left to right direction, right to left direction, and opposite to flow direction. This was repeated until a sufficient volume of WW passed through the net.

At stations with high WW flow, the net started to flow away due to high velocity. In such a situation, the net was allowed to move with the flow until the entire length of the rope was stretched. When the rope was completely stretched, we dragged the net against the flow using the stick and the rope. This was repeated until a sufficient volume of WW passed through the net. At station WW32-WW34, the WW coming from the residential dwellings was collected directly by placing the mouth of the net below the outfall of the house sewer system such that the WW fell directly into the net. The sample collection continued till a sufficient volume of WW passed through the net.

After sample collection, the net was carefully lifted and placed in a steel tub. The suspended solids in the jar attached to the cod end were transferred into a sample storage jar. The cod jar was rinsed three times in the storage jar using DDW. Then, the inside of the net was turned outward, which allowed the captured suspended solids to fall into the tub. The inside portion of the net was washed into the tub using DDW. The process was repeated three times. Lastly, we jet spray the outer portion of the net mesh using DDW to transfer the clogged impurities in the mesh to the tub.

The material collected in the tub was sieved through a 50-µm mesh sieve, and the residue captured on the sieve was transferred to the previously used sample collection jar using a DDW jet. The process was repeated until the entire residue was collected. Finally, the steel tub was washed three times using DDW and sieved through the same sieve to ensure no suspended solids remained attached to the tub's surface. To overcome the problem of a clogged sieve, we followed the strategy used in the case of bucket sampling. The top of the sample storage jar was covered with aluminum foil, and the lid was tightly closed. The duration of net sampling varied between 20 min and 60 min, depending on the flow velocity. The flowmeter reading at each station was noted to calculate the volume of WW sampled at each station. The WW samples were collected from all the sampling stations following the same procedure.

Microplastics Extraction Methodology

Samples stored in the freezer were brought back to room temperature, and wet sieving was performed to segregate MPs present in samples into 3 size groups – 5 mm to 1 mm, 1 mm to 500 µm, and 500 µm to 50 µm. The organic matter digestion protocol using Fenton's reagent (FR) recommended by MPs studies (Hurley et al. 2018, Tagg et al. 2017) was modified to neutralize the organic contaminants present in the sample matrix while ensuring no damage to MPs. FR is very effective for WW samples (Horton et al. 2021) which generates a strong oxidative and exothermic reaction using hydrogen peroxide (H_2O_2) in the presence of a strong ferrous catalyst (Bretas Alvim et al. 2020). Ideally, an exothermic reaction should not require external heating; however, many studies have employed additional external heating to increase efficacy and shorten digestion (Al-Azzawi et al. 2020). Studies report varying digestion temperatures, but recently it has been highlighted that certain polymers tend to exhibit alteration in morphological properties when reaction temperature (RT) reaches above 60°C (Munno et al. 2018).

So, to avoid any alteration, we ensured that the RT should be allowed to reach 60°C at no point during the extraction procedure. This was achieved by putting the reaction vessel in a cold water bath as soon as RT neared 60°C and, in extreme cases, adding cold double distilled water. Here we used multiple aliquots of 20 mL of acidified ferrous sulfate heptahydrate ($FeSO_4.7H_2O$) and 20 mL of 30% H_2O_2, and digestion was performed in multiple steps.

After organic matter digestion, vacuum filtration was performed to extract MPs in the digested suspension on filter paper (Whatman mixed cellulose ester, pore size - 0.45 µm, diameter - 50 mm, type – white/black grid). Filter papers were transferred into the labeled glass Petri dishes, covered with aluminum foil, and stored for further processing.

Morphological and Chemical Characterization of Extracted MPs

Filter papers were visually examined with the naked eye and stereomicroscope (Leica S9D) for morphological characterization. Images were acquired using a handheld digital microscope (QSCOPE). Besides particle count, the suspected MPs were morphologically categorized by shape and color. The criteria for ascribing any shape and color to the suspected MPs was based on the recommendations from the literature. The suspected MPs in the size range (50µm- 5mm) were categorized as beads, spheres, fragments, films, foams, lines, and fibers. We encountered black, white/transparent, green, yellow, pink, orange, red, purple, and blue-colored MPs. The remaining MPs' colors were grouped under the umbrella of 'others. The particle count shapes, and colors were recorded for each size class.

A subset of MPs (50) was pooled from the separated MPs and chemically analyzed through FTIR (Fourier Transform Infrared Spectroscopy)- Attenuated Total Reflectance (ATR) for polymeric identification of the MPs. The particles were randomly selected from all three-size classes to avoid selection bias. Due to the limited accessibility to the instrument, it was impossible to select particles from each filter. Hence the polymer distribution reflects the distribution of MPs in the entire drains of the city rather than reflecting the distribution for all individual drains as done in the case of MPs morphology distribution. The particles were picked manually, cleaned using ethanol, air-dried in a desiccator, and placed manually on the FTIR platform for analysis.

The FTIR spectra were generated by combining 64 scans in the mid-infrared range at a set resolution of 4 cm^{-1} by Bruker ALPHA-II spectrometer having ZnSe crystal. The spectra obtained were compared with the reference library of Openspecy (Cowger et al. 2021) and Bruker. The similarity cutoff was set at 70% to ascribe polymer ID to the spectra obtained from FTIR.

Contaminant Control, Quality Assurance, and Data Analysis

During the sample collection, storage, and analysis procedures for MPs, there is the possibility of sample contamination by the MPs from the atmosphere and consumables and instruments used, leading to an overestimation of MPs (Prata et al. 2021). To avoid this, strict contaminant control and quality assurance steps and precautions were meticulously followed. Field, procedural, and analytical blanks were used to produce reliable results by adopting the recommendations of previous studies (Horton et al. 2021, Miller et al. 2021). Many blanks were contaminated with MPs (1 to 5 particles), and the results reported here are based on the field data as the MPs present in the blank samples were negligible compared to the MPs abundance in the field samples.

During polymer identification, 3 particles were identified as organic matter, bringing down the efficiency of visual identification to 94%. We didn't use this to correct the final results of MPs, as the analysis was performed on the subsamples. The reader may like to consider this factor while interpreting the results.

The data analysis was conducted in Microsoft Excel-2019, plots were prepared in Origin-2023, and the study area map was prepared in ArcGIS. One-way ANOVA was performed to test the significance of the results.

RESULTS

Abundance of MPs

Samples obtained from all 35 drains were abundant with MPs (Fig. 5) which varied between smaller, intermediatory, and larger drains. The average MPs concentration from net samples (5.19 ± 1.32 particles/L) was greater than bucket samples (4.20 ± 1.40 particles/L). For bucket samples, maximum and minimum concentration was observed in W2 (6.68 particles/L) and W33 (0.72 particles/L). For net samples, maximum and minimum concentration was observed in W9 (7.53 particles/L) and W34 (1.92 particles/L).

Among different drain categories, the highest average abundance was in larger drains (B: 4.83±1.04 particles/L, N: 5.76±1.17 particles/L), and we observed (Fig. 6) a clear declining trend ($p < 0.05$) for average MPs concentration in intermediatory drains (B: 4.51±0.93 particles/L, N: 5.69±0.95 particles/L) and smaller drains (B: 3.56±1.68 particles/L, N: 4.43±1.35 particles/L) in both bucket samples and net samples. This trend is due to the basin characteristics, flow volume, intermixing of different LW streams, and functional areas where drains are located, which allows MPs to dilute or accumulate.

Smaller and intermediatory drains that carry WW generating from different functional areas of the city are discharging their MP loads into larger drains, leading to high MPs abundance. Higher MPs concentrations in intermediatory drains are due to incoming MP loads from households, intermixing of SW and IWW, and disposal and littering of MSW and PW into open drains. However, all drains are susceptible to disposal and littering of MSW and PW. It's the magnitude of the disposal that is affecting MP concentration. Within the same drain category, at a particular drain, we did not necessarily observe a similar trend of MPs concentration for both bucket and net sample, indicating that MPs captured are dependent on the sampling techniques. For instance, in smaller drains, maximum MPs concentration was observed in W31 for net samples and in W27 for bucket samples, while minimum MPs concentration was observed in W34 for net samples and in W33 for bucket samples.

WW from residential dwellings (smaller drains – WW32, WW33 & WW34) were least abundant. Drains (W18-W20)

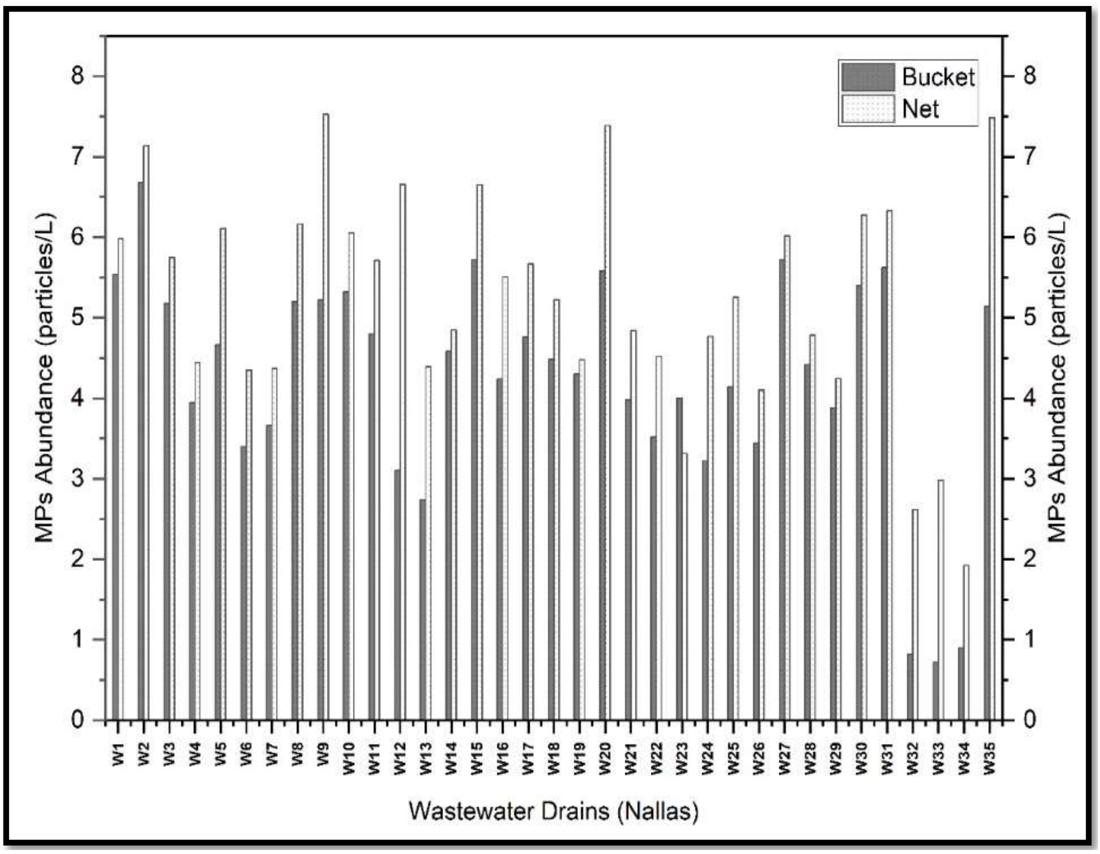

Fig. 5: Abundance of MPs in drains of study area sampled through bucket strategy and net strategy.

carrying agricultural runoff from semi-urbanized zones were comparatively higher than other drains. *Govardhan nalla* (W35) flows through a small natural wetland before being discharged into the river, and the wetland reduced MPs load in the drain by 40 %, thereby limiting the daily MPs load discharged into the river. Small wetland size, high WW volume, and disposal of MSW in the catchment areas of wetland could be the reason for lower MPs removal efficiency (Long et al. 2022). Nonetheless, even a 40 % reduction is significant considering the drain's higher average daily discharge capacity. Construction of wetlands between WW drains and local aquatic sources in locations where the commissioning of WWTPs is not feasible can significantly reduce WW-related MPs load to water bodies.

Only one study from Lahore, Pakistan (Irfan et al. 2020) reports MPs load (16.15 ±0.008 particles/L) in an untreated WW stream of open drain (*nalla*). This is the only reference available for directly comparing our results, as both study areas lie in the Indian subcontinent, having similar environmental, social, and basin characteristics. Drains in both cities are susceptible to disposal and littering of MSW and PW and are clogged in many locations with visible PW floating in drains. As sampling was conducted using a bucket, the MP concentration is twice higher than the maximum concentration observed in drain W2 while greatly higher than the average concentration in all drains. Nearshore sampling was performed in Lahore, which could be the reason for increased MPs levels, as dumping and littering majorly happen near shores that remain accessible to residents. Further, the WW drain was more polluted than freshwater streams, canals, and the *Ravi* River making open WW drains a significant source of terrestrial MP pollution.

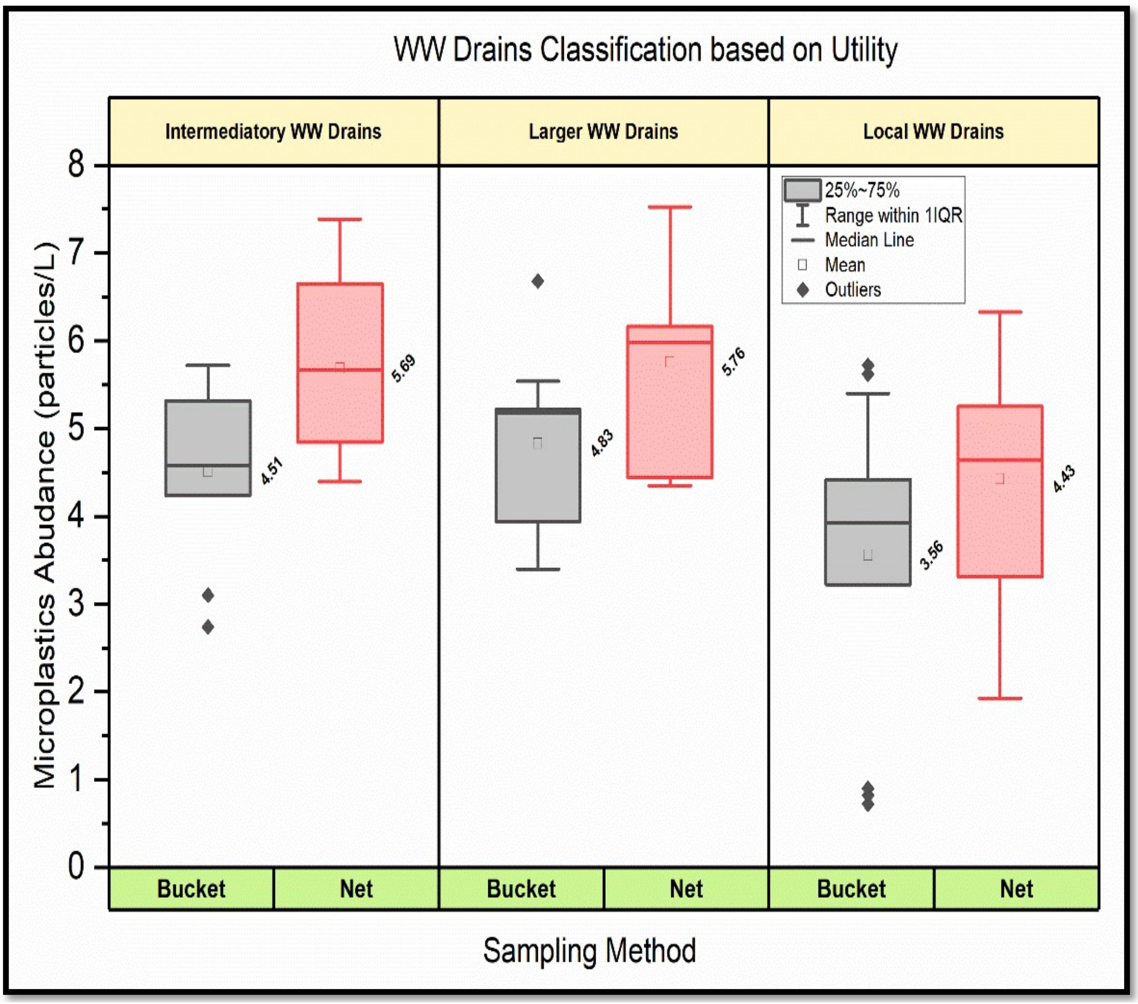

Fig. 6: MPs abundance (average) in different types of WW drains of the study area.

Another study that highlights the extent of MPs distribution in the urban drainage systems of developing areas similar to Raipur City was conducted in Da Nang City. In the drainage channel of the city, the reported high MPs concentration in surface water ((1.48 ± 1.06 particles/L) is indicative of the extraordinary impact of the discharge of urban LW streams on the MPs distribution as the channel receives MWW and landfill leachate from the city (Tran-Nguyen et al. 2022).

In the canal conveying IWW and effluent of Al-Hayer WWTP (Riyadh, Saudi Arabia) to an artificial pond, the average MPs abundance (3.2 ± 0.2 particles/L) is lower than Raipur, mainly due to the absence of intermixing of SW with WWTP's effluent in canal (Picó et al. 2021). In 4 WW ditches (Bahir Dar City, Ethiopia), MPs concentration (1670 580± to 4300 ± 1580 particles/L) is four magnitudes higher than Raipur (Mhiret Gela & Aragaw 2022). Annual discharge of 0.27 billion tons of municipal and industrial effluent from Yenagoa, Nigeria, into Ox-Bow Lake is the primary source of fibrous MPs in the lake (Oni et al. 2020).

Saigon River is connected to Ho Chi Minh City, Vietnam, by four large canals that carry WW of the city. MPs concentration in these canals varies from 270 to 519 particles/L, mostly comprising anthropogenic fibers originating from households WW (Lahens et al. 2018). Wet weather overflow (WWF) of urban agglomeration in Shanghai, China, mainly comprises MPs associated with MWW and road dust, discharges 130 ± 30 to 8500 ± 1241 MPs particles/L into local aquatic sources daily (Chen et al. 2020).

MPs abundance observed in drains of unsewered areas of the present study is lower than the influence of many WWTPs in Europe, the USA, and China (Wu et al. 2022). Moderately higher MPs concentration was observed in WW drains when

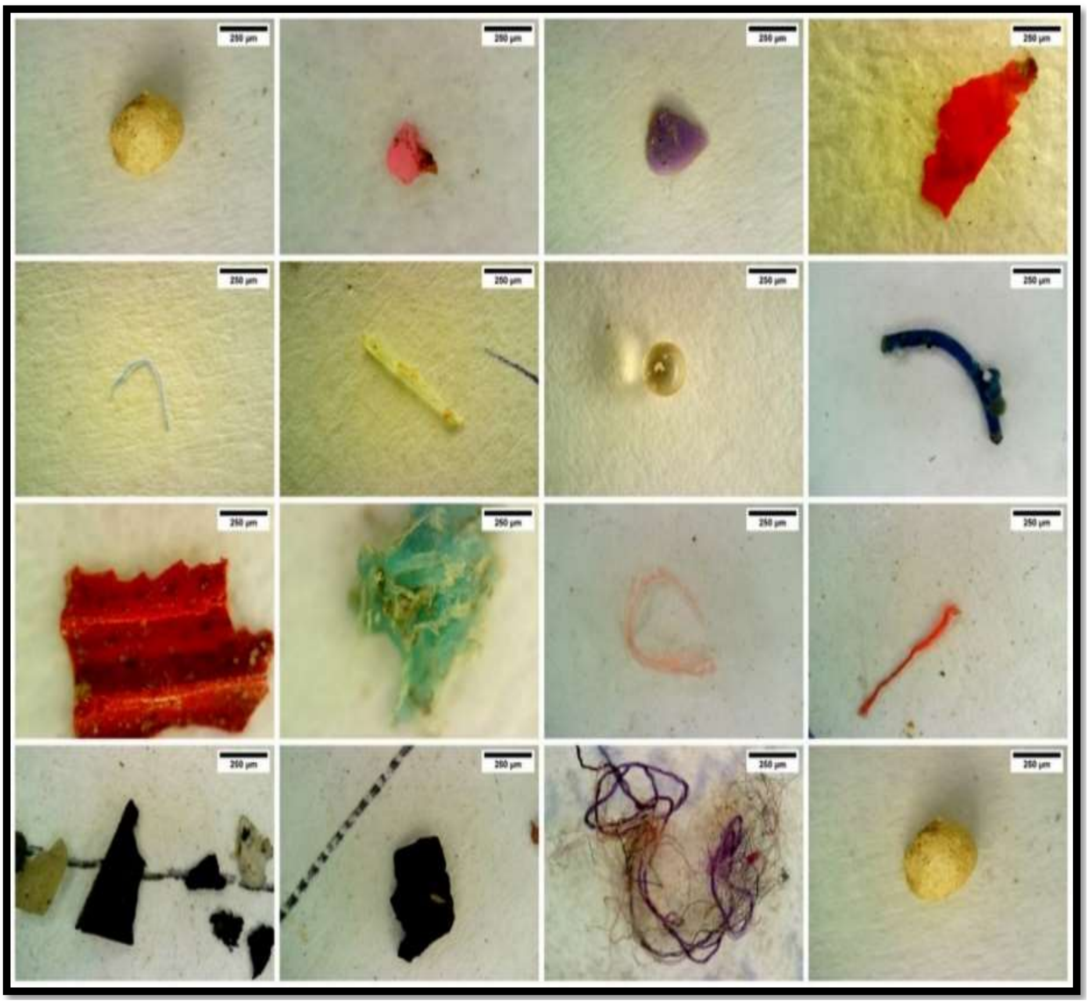

Fig. 7: Micrographs of different MPs extracted from drains in the present study.

compared with the effluent of WWTPs. As the WWTPs are highly effective in removing MPs loads from influent, lower reported MPs concentration is understandable. However, the WWTP effluent of many locations also reports higher levels of MPs, especially from Netherlands – 39 to 81 particles/L (Leslie et al. 2017), Finland - 100 – 13.50 particles/L (Lares et al. 2018), and Denmark – 19 to 447 particles/L (Simon et al. 2018). This suggests that despite WWTPs effectively removing MPs from influent, the lower MPs concentration in WW of Raipur in comparison to WWTPs effluent is mainly due to variation in volume of WW, MPs concentration in influent, difference in sampling and analysis strategy, and difference in the social, economic, and environmental factors.

Characteristics of MPs

Various microplastics were identified in the samples obtained from the drains of Raipur City, and micrographs of a few MPs are presented in Fig. 7. The different characteristics of MPs are discussed below.

MPs size distribution: The size distributions of MPs in 3 size classes – size class I (5 mm to 1 mm), size class II (1 mm to 500 µm), and size class III (500 µm to 50 µm) – in different drains sampled from the bucket and net method is shown in Fig. 8.

We observed that MPs are equally distributed across 3 size classes for both bucket and net samples suggesting input from multiple MPs sources. In MWW coming directly from households (W32. W33, & W34), large MPs prevailed followed by smaller sized MPs, while size class- II was found to be the least Fig. 9. As fibrous MPs from laundry activity were dominant in these drains, we can infer that release of MPs from laundry activity generates varying sized MPs. Additionally, floor wiping is done mostly using wet wipes or cloth rags, and wash water from wiping and washing of rags becomes part of sullage, adding MPs settled from indoor dust into the drains. Drains carrying agricultural runoff (W18-W20) were abundant with large-sized films and foams.

In choked drains, where the flow is either absent or very low, the smaller MPs will remain in suspension, while larger MPs will tend to settle (Jiang et al. 2020). The intermixing of WW streams and atmospheric deposition contributes to varying-sized MPs altering the size distribution in the WW

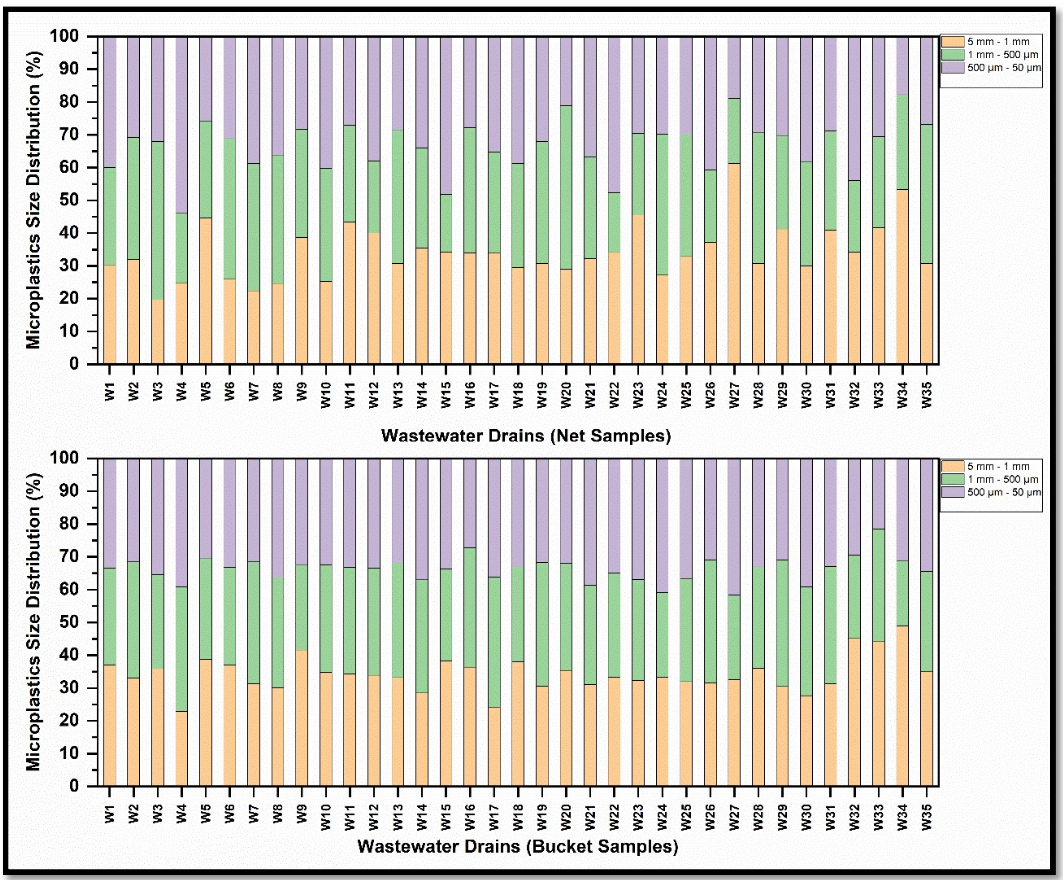

Fig. 8: Size distribution of MPs in the drains sampled in the present study.

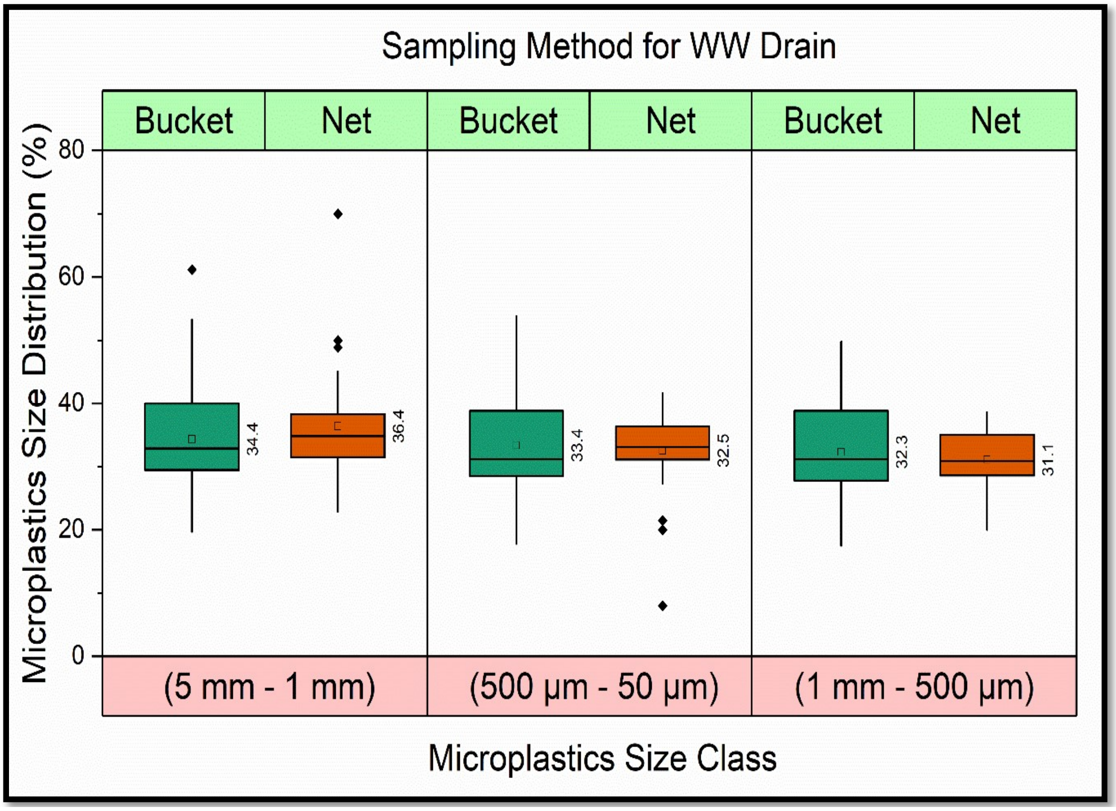

Fig. 9: Average MPs size distribution in the drains sampled in this study.

drains. For instance, larger-sized MPs associated with road dust dominate stormwater (Stang et al. 2022). Large MPs are abundant in agricultural soils and runoff (He et al. 2022). Small-sized (<100 µm) MPs prevail in atmospheric deposits (Sun et al. 2022). Laundry activity also generates smaller fibrous MPs, which mostly end up in the WW (Kelly et al. 2019). The size distribution of MPs is also governed by the shape of MPs which is discussed in the next section.

MPs shape distribution: MPs from both sampling techniques exhibited diverse shapes across all drains (Fig. 10) within the drain category and size class. Overall, fibrous MPs were dominant in all drains, followed by foams, films, and fragments in bucket samples; and fragments, films, and foams in net samples (Fig. 11). Collectively, these four shapes accounted for more than 96% of total MPs. Line-type MPs accounted for approximately 2 % of the MPs, while beads and spheres were observed at least (1% each).

Fibers were also dominant within all three-drain categories in bucket and net samples, followed by fragments in net and films in bucket samples. Interestingly, foams were observed in the highest numbers in larger drains, which confirms our hypothesis that smaller and intermediatory drains, which are more susceptible to obstructions, accumulate SUPs, where they start to fragment and release secondary MPs, which flow with the WW and mix with the larger drains. The distribution of fragments was somewhat uniform across all three-drain categories, understandable so, as all drain categories are susceptible to receiving stormwater. Fibers, films, foams, and fragments were distributed consistently in all three-size classes with small variation (less than 10%). For net samples - the fragments were observed highest in size class I, films in size class III, foams in size class III, lines in size class I, and fibers in size class III. For bucket samples, the fragments were observed highest in size class III, films in size II, foams in size III, lines in size 1, and fibers in size class I.

Our results are consistent with MPs shape distribution in WWTPs. In WW, fibers are the dominant MPs shape, followed by fragments (Yaseen et al. 2022). High variation in MPs shape distribution was observed among individual drains, though fibers prevailed in each drain, suggesting the inflow of fibers from laundry (Kelly et al. 2019a) and atmospheric fallout (Napper et al. 2023) as major contributors to MPs in drains. In addition, the fraction of fibrous MPs in the WW drains W32, W33, and W34 were as high as 90.00%. Since these drains receive sullage from residential dwellings and supernatant from ST, they are abundant with

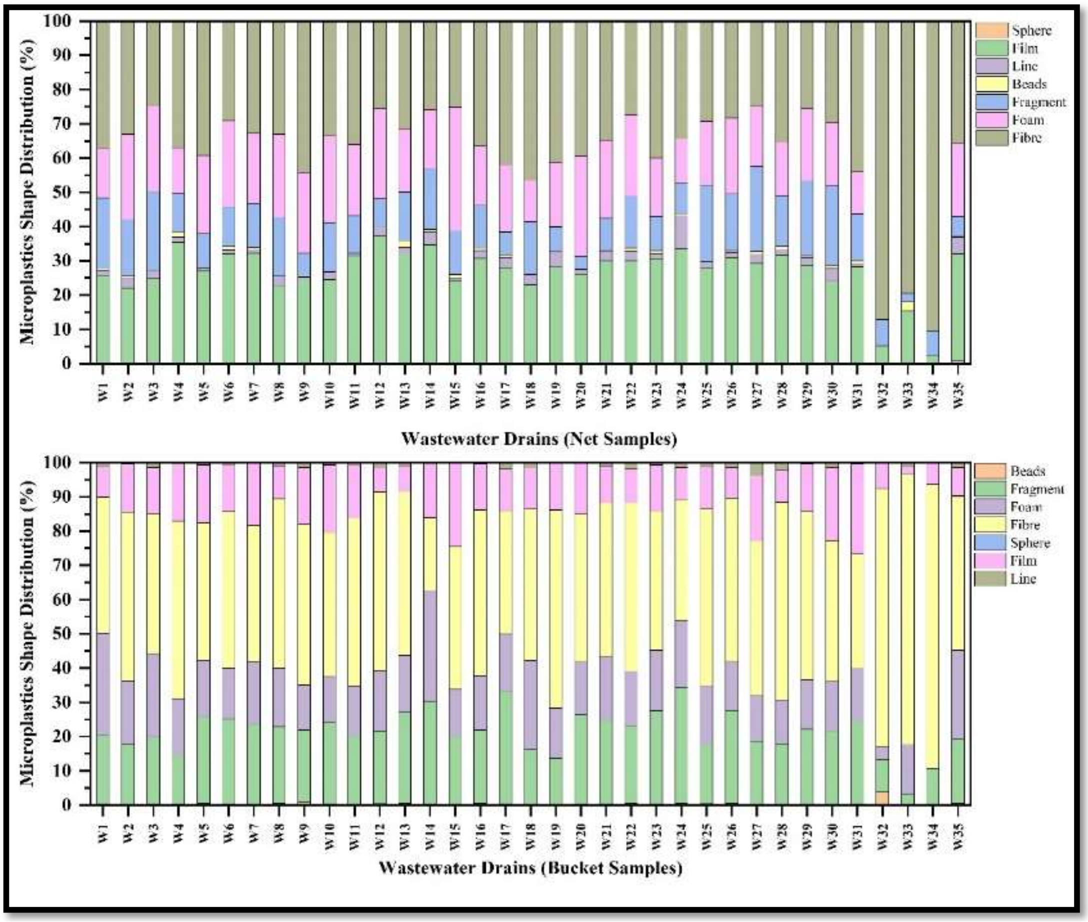

Fig. 10: Shape distribution (%) of MPs in the drains sampled in the present study.

textile fibers from laundry and indoor dust deposition. These drains are not susceptible to PW littering, MSW disposal and absence of SW intermixing, which is why there is no foam, line, and fragments.

With the inflow of MWW occurring in all drains, other varying factor influencing MPs distribution in drains is intermixing of other LW streams. This inference is supported by the observation that a substantial quantity of foam, line, and fragments are present in other drains. The consistent occurrence of rubbery fragments, such as tire wear across the remaining drains, leads to the logical conclusion that SW is transporting RD-associated MPs from the surface to the drains (Monira et al. 2021). The low distribution of beads and spheres highlights that variations in lifestyle and urbanization levels influence microbeads' release in MWW (Ziajahromi et al. 2021).

Foams were the second most abundant MPs shape in drains, followed by film. This observation helps us recognize a new source of MPs relating to WW, overlooked by previous WW studies. The vast network of open and mostly earthen natural drains runs through various functional areas throughout the city. Poor SWM practices, which vary among functional areas, lead to the disposal of SW & PW in these drains. Channel properties (depth, width, location, and elevation) vary throughout the entire course, leading to accumulation of the disposed SW & PW at various portions in the drains having narrow channels and low depth of flow with low elevation difference, especially near LIG residential areas and recreational areas of the city.

Disposed PW in drains were composed of SUPs and polyethylene bags. Overtime, accumulated PW in drains undergoes a fragmentation process (environmental, chemical, and biological processes) and starts generating secondary MPs in drains, which get mixed with LW. SUPs from the eateries, food joints, and party halls, along with foams used in packaging food items and consumer goods, decorative items, plastic bags, FMCG wraps, and other film covers used on beverage bottles, were observed floating in all the sampled drains.

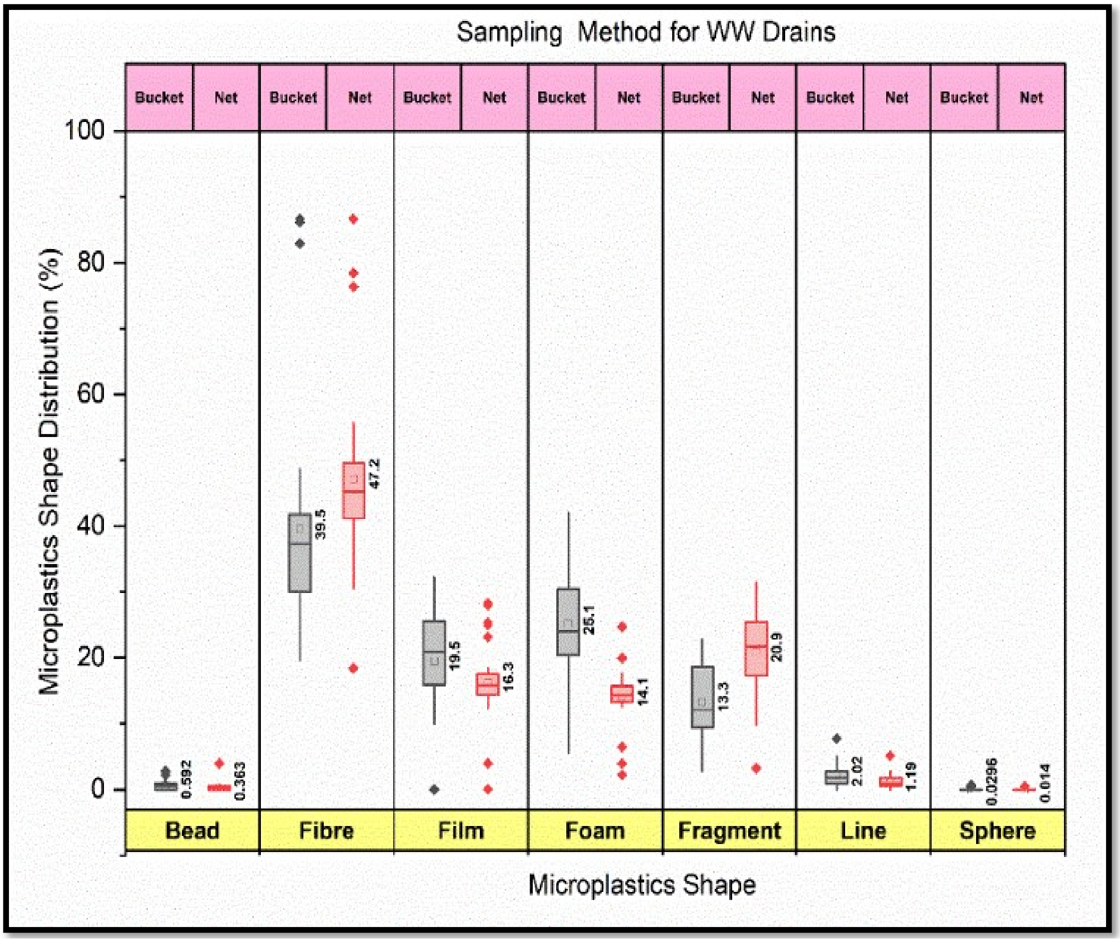

Fig. 11: Overall shape distribution (%) of MPs.

Even though the primary purpose of using two sampling strategies was to collect samples from different types of drains with different channel characteristics and logistical challenges, variation in shape distribution between the bucket and net samples highlights the importance of using at least two sampling strategies while planning any WW-related study. Regardless of the sampling technique, every sampling technique will have a bias that may not produce reliable results.

For instance, we observed that nets were not best suited to collect foams and films, which due to their low density, escaped the mouth of the net, while fibers and fragments were collected efficiently. So, if any WW drain appears to be abundant with floating foams and films during the preliminary survey, we recommend using the bucket sampling technique, and for other shapes, net sampling will work well. We understand the resources required for adopting two sampling strategies are quite higher, and the research laboratories should choose the strategy best suited for the drains of the study area. Another approach can be to conduct a pre-experiment by collecting samples from a few drains using both strategies and analyzing the MPs diversity in the samples. Based on this, an optimum sampling strategy can be adopted.

MPs Color Distribution: The extracted MPs from both sampling techniques exhibited diverse spectrum of colors with low diversity, as black and white colored MPs were prevalent in all samples (Fig. 12). Average distribution (%) of colored MPs across all the drains is shown in Fig. 13. White/transparent MPs accounted for 40 -50% of all MPs in both bucket and net samples, followed by black (25-30%) and green (4-7%) MPs. Collectively, these three-color categories accounted for 74.44% (B) and 82.84% (N) of all the colors.

Particular dominance of any color or color distribution is observed to be independent of the sampling method. Blue (B-4.32%, N-5.07%) colored MPs were the fourth most observed color of MPs, followed by yellow (B-4.68%, N-2.03%) colored MPs. The percentage distribution of pink, purple, red, and orange MPs were less than 2.5 % each.

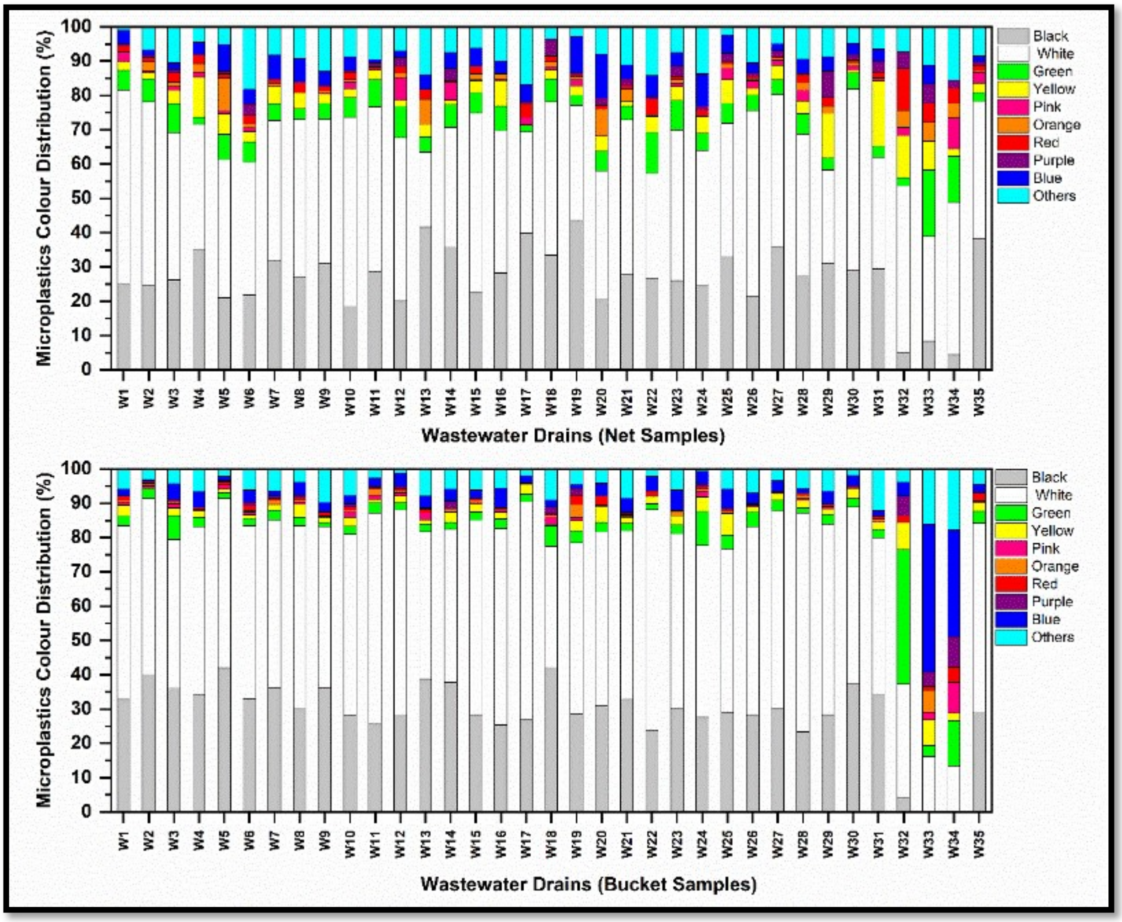

Fig. 12: Color distribution (%) of MPs in the drains sampled in the present study.

Color distribution trend (white/transparent>black> others>green>blue>yellow) with white/transparent, black, and green being dominant was similar in all three drain categories – larger WW drains (N -87.80%, B- 77.58%), intermediatory WW drains (N -86.24%, B- 76.85 %), and smaller WW drains (N -84.84%, B- 74.64%).

The remaining colors were less than 2.5% each. Color distribution trend and percentage distribution are similar in individual drains, within drain categories, and combinedly in all drains. This suggests that the sources of MPs in the drains do not vary, and different LW streams are intermixed in all the drains. However, the lower distribution of black fragments and the prevalence of fibers in drains (W32, W33 & W34) carrying MWW (coming directly from residential dwellings) suggests that intermixing other LW streams is absent. The higher distribution of black-colored MPs in other drains (where intermixing of SW with other LW streams is occurring) makes SW a major source of MPs in drains. The dominance of black-colored MPs in SW is mainly due to rubbery MPs generated from the wear and tear of the vehicular tire (Werbowski et al. 2021)

Comparatively, white MPs were prevalent in smaller and black MPs in larger drains. Smaller drains connected with households mostly receive MWW (mainly comprising of laundry, hygiene (wet wipes), sullage, and other WW from residential dwellings), and being narrow, open, and accessible, are more susceptible to receive SUPs, bags, FMCG scathes, and wraps, choking and accumulating PW which leads to their fragmentation into secondary fibrous, foam and film MPs.

In addition, these drains being open channels may also receive substantial amounts of white and transparent fibrous MPs from atmospheric fallouts (Sun et al. 2022). Further, the foam MPs generated from accumulated SUPs' breakdown are majorly white in color (Chen et al. 2020). Sources of colored MPs are laundry fibers, fibers from atmospheric deposits, films from the packaging of food items and consumer goods, decorative items, plastic bags, FMCG wraps, and other film

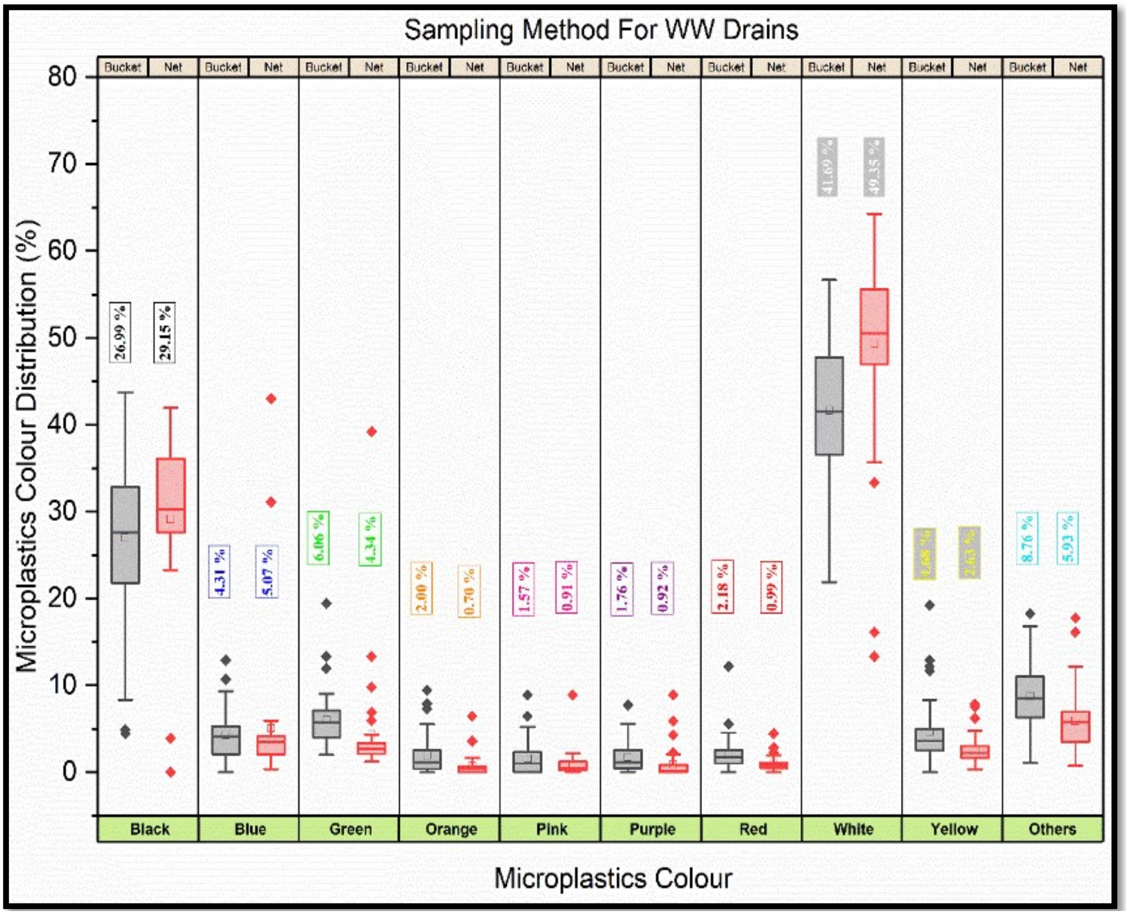

Fig. 13: Overall color distribution (%) of MPs.

covers used on beverage bottles, and colored fragments from disintegration and fragmentation of disposed PW

MPs Polymer Distribution

Out of 50 particles analyzed using FTIR-ATRA, only 3 were residues of undigested organic matter, and 11 types of polymers were identified. No identity could be assigned to rubbers due to the limitation of the FTIR-ATR in the analysis of black rubbery MPs. However, based on their appearance, they were categorized as rubber and not considered for assessing the polymer distribution.

The polymer distribution observed through FTIR analysis is depicted in Fig. 14, and FTIR spectra are presented in Fig. 15. Polyethylene (PE) dominated the polymeric distribution, followed by Polypropylene (PP) and Polystyrene (PS). Combinedly, these three accounted for 54% of all the particles analyzed. The distribution of other polymers was below 16 %, and their occurrence can be traced to the SW disposal in the drains. The polymer distribution in the urban drainage system of the current study follows the polymer demand trend and plastic waste generation observed globally (PlasticsEurope 2021) in India and Raipur (CPCB 2015).

Other freshwater studies conclude that PE, PP, and PS are the most produced and demanded single-use plastics, and their low density hinders settling in the water column (Irfan et al. 2020). The polymer distribution in WW samples can assist in identifying MPs sources in the drains. Since WW drains receiving different WW streams affected the MPs' shape, size, and color distribution, it's highly likely that polymer type and their distribution are also influenced by intermixing. This is the reason that we observed high polymer diversity in the samples.

Most of the PE material appeared to be film and foam. While the films from FMCG products, wraps, mulching sheets, and plastic bags are mainly made of PE, the observation that many foams are of PE origin was interesting. Contrary to popular belief that foams are mainly manufactured from PS, SUPs, which majorly contributed to the foam MPs, are also made from PE. Interestingly, few foams were composed of styrene allyl alcohol (4%) and

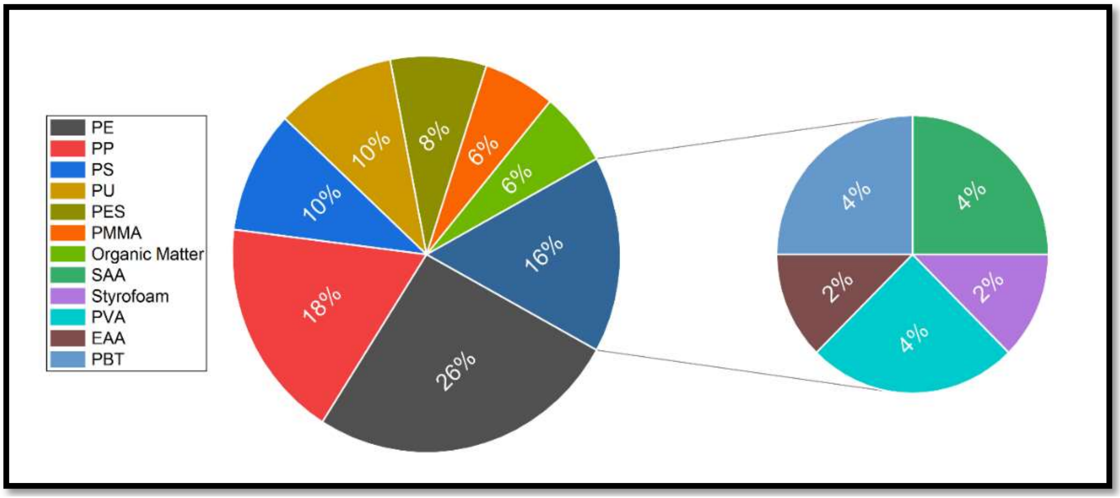

Fig. 14: Polymer distribution of the 50 MPs (subsample) analyzed through FTIR.

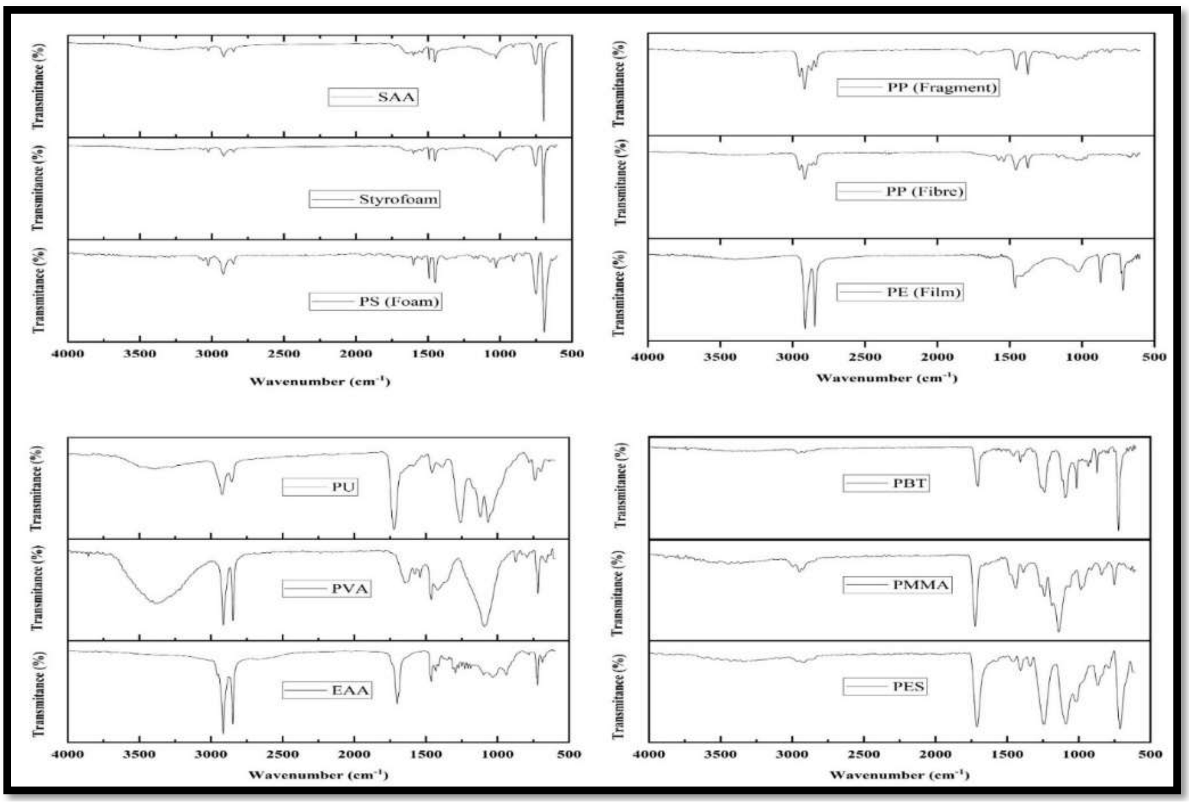

Fig. 15: FTIR spectra of MPs analyzed through FTIR-ATR.

styrofoam (2%), while polyurethane (PU) was observed to be 10%, highlighting that the disposal of PW into the drains or the MWW is contributing to the foam type of MPs in the drainage system. The disposal of electronic waste into the drains is primarily responsible for the occurrence of polybutylene terephthalate (PBT).

Synthetic fibers were mainly polyamide (PA) and polyester (PES), which also agrees with the literature and confirms the entry of fibrous MPs from laundry activity, those disintegrated from discarded textiles in the drains, and atmospheric deposits into the drains. Despite the fact that the fibers were the dominant shape category observed here, their polymeric composition is influenced by their distribution in the subsample used for FTIR analysis, and polymeric distribution may vary due to extrapolation error from lack of homogenization, highlighting the need for an appropriate subsampling strategy for stereoscopic analysis. Nonetheless, all the fibers subsampled for FTIR comprised PP, PA, and PES.

Due to the wide range of polymers currently being produced, their wide distribution in the environment as waste, and the complex nature of informal urban drainage systems carrying intermixed liquid, coupled with instrumental limitations and lack of enough resources to analyze plentiful particles, its challenging to assess the polymeric distribution in different liquid waste streams. However, given that the majority (72%) of MPs analyzed here were composed of PE, PP, PS, PES, and PU, it is possible to use them as an indicator for assessing the MPs' pollution in informal urban drainage systems as these polymers are associated with all for LW streams being intermixed and conveyed by the drains.

DISCUSSION

Total MPs Burden and Daily Emission Load from Urban Drains

The total daily microplastic load (L) being discharged into the local Kharun river through 9 larger drains is calculated by formulae below and expressed as the number of particles

$$L = \sum_{n=1}^{9} Cn * Qn$$

where Cn is the MP concentration being discharged by the WW drains, expressed as Particles/L, Qn is the daily discharge capacity of WW drains in liters per day, and n =1 to 9 for the nine wastewater drains discharging daily.

For net samples, TDML discharged by drains varied between 0.90×10^8 particles (W7) to 5.6×10^8 particles (W9), with combined TDML of 12.6×10^8 particles being discharged in the Kharun river from the entire city (all 9 larger drains) daily. The quantity of MPs emitted daily by any particular drain directly depends upon the discharge capacity of that drain. 75 MLD of liquid waste is emitted daily by drain W9 which is nearly ten times higher than drain W1 (8 MLD). Hence such a large variation in the TDML. The TDML of individual drains and discharge capacity are summarized in Table 1.

Due to limited studies conducted in the urban informal drainage systems, TDML being discharged from different WWTPs around the globe (Table 2) is used for comparative assessment. Studies report both lower and higher TDML than that observed in our study. Lower TDML in drains of Raipur in contrast with TDML of WWTPs effluent highlights the influence of variations in sources of MPs contributing to the LW, sampling, and extraction protocol, lowest size of the MPs studied, the efficacy of WWTPs, and other socioeconomic factors such as level of urbanization, and difference in lifestyle.

Higher TDML in untreated WW of the present study than TDML of WWTPs effluent is expected since WWTPs effectively reduce MPs load of the WW (Yaseen et al. 2022). Three important reasons for higher TDML from Raipur city that are absent elsewhere are intermixing of waste streams (MWW, SW runoff, agricultural runoff, industrial runoff, and secretly disposed of IWW), MSW and PW litter reaching the open drains, and atmospheric deposition in open drains.

Table 1: Total daily MPs load of nine larger drains emitting their load in the local river.

WW Drain	Discharge capacity (MLD)	Daily discharge capacity (Liters)	MPs Concentration Net (particles/L)	MPs Concentration Bucket (particles/L)	Daily MPs load in the WW drains (net)	Daily MPs load in the WW drains (bucket)	Daily MPs load in the WW drains (Average of bucket & Net)
W1	8	8000000	5.979	5.540	47835897.44	44320000	46077948.72
W2	12	12000000	7.139	6.680	85668016.19	80160000	82914008.1
W3	18.9	18900000	5.746	5.180	108597902.1	97902000	103249951
W4	32	32000000	4.441	3.940	142097902.1	126080000	134088951
W5	15	15000000	6.109	4.660	91628959.28	69900000	80764479.64
W6	9	9000000	4.347	3.400	39125874.13	30600000	34862937.06
W7	2.07	2070000	4.371	3.660	9048351.648	7576200	8312275.824
W8	28.43	28430000	6.167	5.200	175337516.9	147836000	161586758.4
W9	75	75000000	7.527	5.220	564560439.6	391500000	478030219.8
Total	200.4	200400000	-	-	1263900859	995874200	1129887530

Table 2: Total daily MPs load from WWTPs reported in the literature.

S. No.	Type of treatment facility	Country	TDML	Reference
1.	1 WWTP	China	9.1×10^{10}	(Tang et al. 2020)
2.	1 WWTP	USA	3.56×10^{10}	(Mason et al. 2016)
3.	1 WWTP	Germany	1.91×10^{10}	(Schmidt et al. 2020)
4.	7 WWTPs of Xiamen, a typical coastal city	China	0.065×10^{10}	(Long et al. 2019)
5.	1 WWTP near Vancouver, British Columbia	China	0.008×10^{10}	(Gies et al. 2018)
6.	1 WWTP of Sari	Iran	0.009×10^{10}	(Alavian Petroody et al. 2020),
7.	3 WWTPs in Mersin Bay	Turkey	0.018×10^{10}	(Akarsu et al. 2020)
8.	1 WWTP located in Wuxi	Eastern China	0.1×10^{10}	(Lv et al. 2019)
9.	1 tertiary WWTP in Gumi	South Korea	0.79×10^{10}	(Kim et al. 2022),
10.	Secondary WWTP in Istanbul	Turkey	0.0293×10^{10}	(Vardar et al. 2021)
11.	2 WWTPs in Adana city	Turkey	0.16×10^{8}	(Gündoğdu et al. 2018)
12.	1 WWTP of northern Italy	Italy	0.016×10^{10}	(Magni et al. 2019),
13.	Choneibe WWTP, located in Ahvaz City of Khuzestan province	Iran	0.24×10^{8}	(Takdastan et al. 2021),
14.	3 WWTPs in the Canterbury region	New Zealand	0.24×10^{6}	(Ruffell et al. 2021),
15.	12 WWTPs in Lower Saxony	Germany	0.24×10^{6} to 0.109×10^{8}	(Mintenig et al. 2017),
16.	Xi'an city	China	0.34×10^{10}	(Yang et al. 2021),
17.	WWTP of Tokyo	Japan	4.6×10^{10}	(Sugiura et al. 2021)
18.	3 WWTPs Charleston Harbor	USA	0.05×10^{10} to 01×10^{10}	(Conley et al. 2019)
19.	1 Conventional WWTP	Bangkok	$12.80\ 10^{8}$	(Tadsuwan & Babel 2022)
20.	1 WWTP of Helsinki	Finland	0.011×10^{10}	(Talvitie et al. 2017)
21.	1 WWTP of Glasgow	Scotland	0.006×10^{10}	(Murphy et al. 2016)

MPs originating from households are quite different in developed countries with higher living standards. Limited use of PCPs by residents of Raipur resulted in a negligible quantity of beads and microspheres in drains, while they remain abundant in WW of developed countries (Hu et al. 2022). Further, in ST-based OSS and Indian-style toilets with built-in laundry and bathing areas, a fraction of sullage gets disposed into the toilets and ends up in ST. A fraction of household-related MPs probably gets retained in the ST. Higher consumption and disposal of wet wipes, PCPs, and other hygiene-related products directly into the toilet (flushing) generates substantial MPs in Western countries (Ó Briain et al. 2020). Such practices are not observed in India, which explains lower MPs in the domestic WW directly emitted from households.

Globally, treated effluent disposal can add around 1.47×10^{15} MPs annually, whereas discharge of untreated effluent is likely to add a staggering 10×3.85^{16} MPs annually to aquatic environments (Uddin et al. 2020). Saigon River system disposes 1.1-1.6x10^{14} fibrous MPs annually into coastal zones, and besides atmospheric fallout, MWW and IWW discharge from the city into the river is mainly responsible for fibrous MPs (Strady et al. 2020). In Malaysia, 95% of untreated WW and effluent of the WWTPs treating 5% of the WW are discharging 5.45x10^{8} MPs particles daily in the aquatic environment (Praveena et al. 2018). The poor performance of WWTP located in the Bandar Abbas City of Iran led to the release of 1.2x$10^{8\ \text{MPs}}$ through 60 MLD of WW in the Persian Gulf (Naji et al. 2021). In the port of Durban, the highest MPs concentration was observed at the site located in in front of SW drains, and the authors report that due to frequent blockage occurring at the sewage pumping station, sewage overflows into SW drains result in higher MPs concentration at this site (Preston-Whyte et al. 2021). TDML from WWF to local aquatic sources was (2.32×10^{12}), six times higher than MPs loads discharged from local WWTP. This highlights the significance and influence of WWF (intermixing of MWW and SW) mainly on the distribution of MPs in drains and ecosystems (Chen et al. 2020).

A WW drain from Lahore, Pakistan, with characteristics exactly similar to the drains studied here, discharges 7 x 10^{9} MPs particles daily into the Ravi River (Irfan et al. 2020). In contrast, the collective TDML of nine larger drains of Raipur is one order less than a single drain of Lahore, suggesting that even in open drains of the Indian subcontinent with mostly the same characteristics and sources, variation in the distribution of MPs occurs, which is mainly due to different sampling strategies, extraction and analysis techniques, and LoD. The findings stress the urgency of standardization of MPs sampling and analysis protocols for WW. Untreated WW from Longyearbyen, a smaller town with approximately 2,500 inhabitants in Svalbard, a Norwegian archipelago, emits 0.33×10^8 fibrous MPs particles daily into the fjord system. This emphasizes the importance of commissioning WWTPs even at the community level, as evident even smaller communities with low populations generate a significant number of MPs (Herzke et al. 2021).

Laundry activity from washing machines releases a significant quantity (in order of 10^6) of microfibers (Kelly et al. 2019), and they are an important source of MPs. The primary laundry mode in Raipur is manual, unlike West, where laundry machines are prevalent. Hand washing of garments has not been studied in detail. However, hand washing is also done with detergents and fabric softeners like machine washing. Hand washing process involves constant abrasion, swishing, scrubbing, twisting, rubbing, and throwing the garment on the surface either through hand or some laundry scrub generating enough mechanical and chemical stress to release fibrous MPs from the garments, becoming part of the sullage and introduce fibrous MPs in WW drains of Raipur.

Atmospheric deposits are a significant source of fibrous MPs whose concentration (several orders higher than that emitted from WWTPs) varies between dry and wet periods (Napper et al. 2023). In Paris, the annual MPs load from atmospheric fallout is estimated to be between 3 to 10 tons, and the highest deposits occurred during the wet period in the urban zone, and atmospheric deposition in catchment areas increases the distribution of MPs (Dris et al. 2018). Hence atmospheric deposits can be a significant contributor of MPs in WW drains of Raipur for two reasons – first, unlike covered sewers limiting the entry of fallouts, WW drains, mostly natural and unlined channels, are open and prone to receive MPs from atmospheric fallout. Second, Raipur City has a tropical climate receiving a substantial portion of high annual precipitation, mostly during the three monsoon months, increasing the MPs deposition during this period. Additionally, MPs deposited from fallouts in catchment areas also enter the WW drains through urban runoff.

CONCLUSION

This study presents the first evidence of the presence and characteristics of MPs in urban informal (open and natural) drainage systems carrying intermixed liquid waste streams originating from a residential habitat. Urban informal drainage systems are a significant source of MPs pollution discharged into aquatic environments. MPs sampling techniques were appropriately modified for logistical and locational challenges. The sampling methodology adopted here can be employed to investigate the distribution of MPs in urban informal drainage systems across the Indian subcontinent and Africa. The bucket strategy best suits the local or smaller drains and drains on the roadside, while the Net strategy suits larger drains where flow is higher.

MPs concentration and characteristics vary in different types of drains carrying liquid waste streams from different city functional areas. Moreover, we also estimated the total daily MPs emission from intermixed liquid waste streams discharged to local rivers. To protect local aquatic ecosystems that receive liquid waste streams from urban areas, constructed wetlands can be a cost-effective solution to protect the local aquatic ecosystems.

Our results establish that besides MPs emitted from households, atmospheric deposition, and SW disposal are also contributing to MPs in urban open drainage systems and must be considered when adopting preventive measures. Such potential sources warrant detailed investigation into how they influence the overall distribution of MPs associated with liquid waste streams, as these sources are not recognized as associated with wastewater. Local environmental regulations on MSW, plastic waste, MPs and microbeads, etc., and their enforcement will govern the quantity of MPs in drainage systems. We observed that WW drains in LIG and slum areas with poor SWM practices were more abundant with floating PW than those near the HIG area. By providing better municipal services and conducting educational and awareness campaigns to communicate best practices to manage MSW and PW at the household level, the entry of SW & PW can be restricted.

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