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Research Article

Unveiling Microplastic Removal and Characteristics in Wastewater from Two Municipal Wastewater Treatment Facilities in Indonesia

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Abstract

Wastewater treatment plants (WWTPs) are considered an entrance pathways for microplastic (MP) pollution in aquatic environments. This study reveals the removal and characteristics of MPs in wastewater from two municipal WWTPs in Indonesia. The influent contained 17.1 ± 5.65 particles L^{-1} (WWTP A) and 15.45 ± 4.31 particles L^{-1} (WWTP B), whereas the effluent contained 1.41 ± 0.01 and 1.5 ± 0.16 particles L^{-1}. The removal efficiency was 91.75% for WWTP A and 90.32% for WWTP B, with no statistically significant difference ($p > 0.05$). WWTP A employed advanced treatment units, whereas WWTP B used a conventional pond-based system. MPs were characterized via light microscopy, with most particles ranging from 100–300 µm and 1000–5,000 µm. Fibers and fragments were the dominant shapes, with transparent and black being the most common colors. ATR-FTIR analysis identified polymers such as polypropylene (PP), polyethylene (PE), polyethylene terephthalate (PET), polyester, and polystyrene (PS). These findings emphasize the important role of WWTPs in reducing MP pollution and highlight the need to improve treatment technologies to better protect aquatic ecosystems.

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Introduction

Microplastics (MPs) have become ubiquitous environmental contaminants that threaten aquatic ecosystems and human health (Haas et al., 2015; Rohaningsih et al., 2025). Defined as plastic particles smaller than 5 mm, these pollutants come from several sources, including synthetic textiles, personal care products (PCPs), and the breakdown of larger debris, eventually entering water systems via urban runoff, industrial discharge, and domestic wastewater (Browne et al., 2011). In this context, wastewater treatment plants (WWTPs) play a dual role as barriers and potential sources of MPs for the environment, especially in aquatic areas. While WWTPs are specifically designed to remove solids and organic matter, their efficiency in eliminating micropollutants such as MPs varies significantly depending on treatment processes (Iyare et al., 2020; Mahon et al., 2017; Sun et al., 2019). As a result, effluents in treated water often contain MPs that are released into aquatic environments (Murphy et al., 2016), such as rivers, lakes, and coastal water, and pose long-term ecological dangers.

As a global hotspot for plastic pollution, issues such as high plastic consumption, poor waste management, and rapid urbanization in Indonesia are especially pressing (Jambeck et al., 2015). Municipal WWTPs, which serve as essential infrastructure for wastewater management, may unintentionally contribute to MP pollution if removal mechanisms are inadequate. However, studies on MP abundance, characteristics, and removal efficiency in Indonesian WWTPs are still scarce. Previous studies have investigated mainly marine and freshwater MP pollution (Cordova et al., 2019; Sulistyowati et al., 2022; Suteja et al., 2021), whereas few studies have investigated critical reservoirs for MPs along the pollution transport pathway in WWTPs. Because Indonesia has a dense population and much of its population is highly dependent on aquatic ecosystems for food and livelihood (Napitupulu et al., 2022, understanding MP dynamics in wastewater systems is critical for reducing environmental and public health risks.

Research from other regions has shown that the ability of WWTPs to remove MPs largely depends on the treatment technologies applied (Iyare et al., 2020; Mahon et al., 2017; Sun et al., 2019). Typically, primary and secondary methods can remove approximately 50–90% of MPs (Dris et al., 2015; Magni et al., 2019; Magnusson and Norén, 2014; Maw et al., 2024; Parashar and Hait, 2022), whereas more advanced methods involving tertiary treatment can achieve higher efficiencies (Hidayaturrahman et al., 2019; Talvitie et al., 2017). However, these results might not apply directly to Indonesia. Differences in wastewater composition, climate, and how treatment plants operate may lead to different outcomes. Additionally, there is still limited information on the characteristics of MPs found in Indonesian wastewater, which makes it harder to develop effective, locally tailored strategies for dealing with them.

This study aims to fill some knowledge gaps by investigating how MPs are removed and what types of MPs are present in two municipal WWTPs in Indonesia. This research focuses explicitly on the main objectives: to examine the MP abundance and characterize it in both influent and treated water, to evaluate how effectively MPs are removed, and to compare the performance of the two plants in reducing MP pollution. This study contributes to the broader understanding of plastic pollution in tropical regions by providing data on MP pollution in Indonesia's wastewater systems. The insights gained can help guide improvements in wastewater treatment technologies and inform policies grounded in evidence. These findings will benefit environmental authorities, wastewater plant operators, and policymakers working to reduce MP emissions and safeguard aquatic ecosystems.

Materials and methods

1) Study sites

Two domestic wastewater treatment facilities, the Setiabudi WWTP (A) and the Bojongsoang WWTP (B), were the sites for sample collection. An overview of each study site is provided in Figure 1, with the treatment process schemes shown in Figure 2. Further details on the treatment capacity, unit processes, and technologies are summarized in Table 1. Both plants employ primary and secondary treatments with biological processes as the core method but differ in their treatment approaches. WWTP A uses a mechanical-biological system comprising a spiral sieve, dual moving bed biofilm reactor (MBBR) lines, coagulation–flocculation, clarification, and a feed well tank, with sludge managed through disposal and treatment. In contrast, WWTP B utilizes a conventional pond-based system involving bar screening, mechanical screening, grit removal, and sequential anaerobic, facultative, and maturation ponds, with solid and sludge waste handled through separation, disposal, and drying. The effluent from WWTP A is discharged into the Setiabudi Reservoir, whereas the effluent from WWTP B flows into a tributary of the Citarum River.

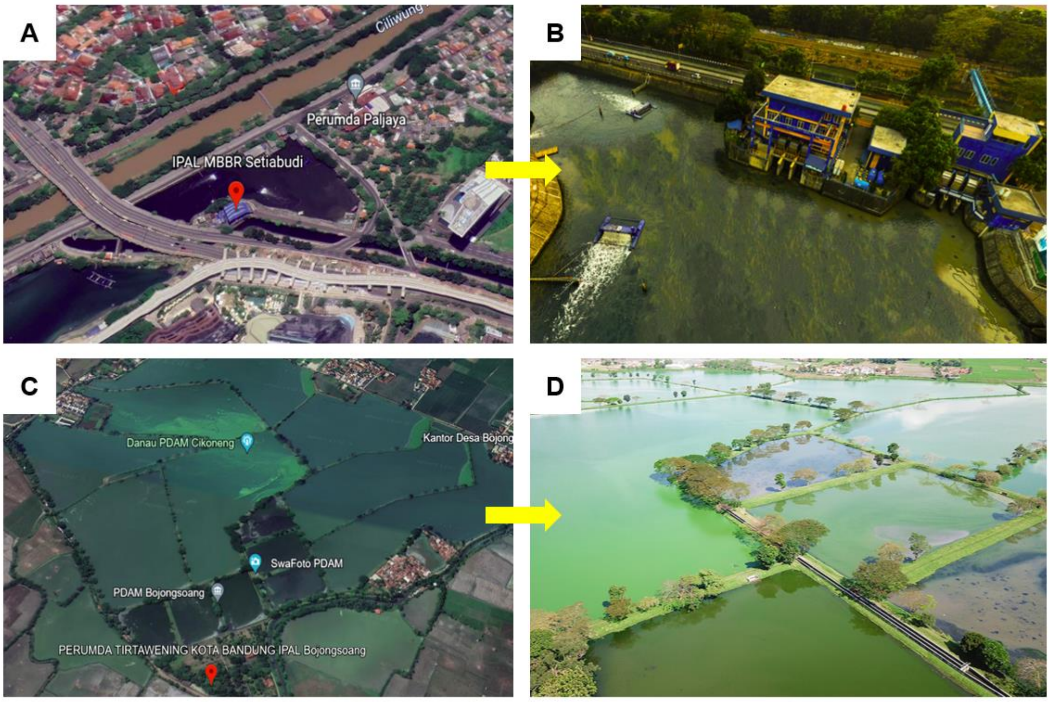


Figure 1 Locations and site views of the studied wastewater treatment plants;

(a–b) Setiabudi WWTP (MBBR-based system), showing satellite and facility views. (c–d) Bojongsoang WWTP (pond-based system), showing satellite and aerial views of the treatment area.

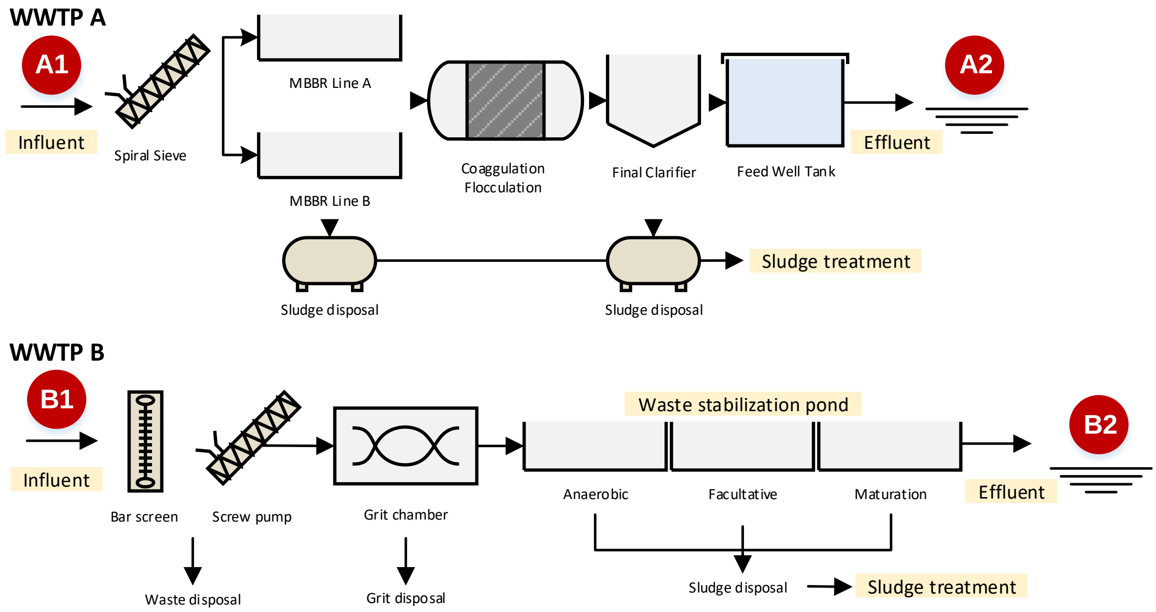


Figure 2 Treatment process for each WWTP studied.

Table 1 Detailed information of each WWTP

WWTP	Coordinates	Operations time	Treatment capacity (m^3 d^{-1})	Treatment process	Main treatment technology
Setiabudi (A)	6°12'18.0"S 106°49'44.2"E	Since 2019	250	Primary and secondary treatment	Moving bed biofilm reactor with a high rate clarifier
Bojongsoang (B)	6°59'35.3"S 107°39'14.8"E	Since 1992	80,000	Primary and secondary treatment	Waste stabilization ponds

2) Sample collections

Water samples were taken at sampling points A and B in February and March 2022, respectively. The technique employed was based on previous similar research, as there is currently no standardized procedure for sampling MPs in wastewater (Liu et al., 2022; Sun et al., 2018; Zhou et al., 2022). In this investigation, grab sampling, a widely used method for collecting MPs, was applied (Liu et al., 2021; Sun et al., 2018). Two sampling points were selected to collect wastewater: influent (A1 and B1) and effluent (A2 and B2). Importantly, the sludge samples were not collected or analyzed in this study because of resource and logistical constraints. The volume of collected water varies depending on water clarity, which differs between raw and treated wastewater (Liu et al., 2022; Murphy et al., 2016); 10 L of raw sewage (influent) and 50 L of treated water (effluent) were collected at each sampling point. The samples were collected in duplicate via a 5 L stainless steel bucket, gently poured and immediately filtered through a plankton net with mesh sizes of 300 µm and 100 µm. The net was rinsed at least three times after filtration to ensure thorough particle recovery. After collection, the samples were placed in sterile 250 mL bottles and transported to the laboratory for further processing and analysis. While no standard preservation method currently exists for MP samples, all collected samples were physically preserved by freezing at 3–5°C. Although the use of grab sampling and the absence of sludge analysis may limit the completeness of this investigation, the chosen methodology remains consistent with commonly accepted practices in MP research. These methodological constraints were acknowledged and considered when interpreting the results and drawing conclusions.

3) Sample processing

The pretreatment process consisted of three stages: organic matter digestion, density separation, and extraction (Tan et al., 2022). Digestion was performed using 30% hydrogen peroxide (H_2O_2) to remove organic matter without damaging plastic particles (Bakaraki et al., 2021). The samples were heated to 60°C on a hot plate for 30 min with continuous stirring to accelerate the reaction. Density separation was subsequently conducted using a saturated zinc chloride ($ZnCl_2$) solution with a density of 1.6 g cm^{-3}. The samples were gently stirred, covered with aluminum foil, and left undisturbed for 24 hours to allow the plastic particles to separate from the heavier materials.

For particle size fractionation, the supernatant containing floating residues was sequentially filtered through stainless steel sieves with mesh sizes of 100, 300, 500, and 1,000 µm. The particles retained on each sieve were collected and assigned to the corresponding size classes (101–300 µm, 301–500 µm, 501–1,000 µm, and 1,001–5,000 µm). Particles larger than 5,000 µm were visually excluded prior to analysis, in accordance with the commonly accepted upper size limit for microplastics (<5 mm). The filtrate passing through the 100 µm sieve was subsequently vacuum-filtered using a Whatman GF/C filter with a pore size of 1.2 µm (Maw et al., 2024). The particles retained on this filter were operationally defined as the 45–100 µm size class. Although smaller particles (<45 µm) may also be retained on the GF/C filter, they were not included in the size distribution analysis because of limitations in reliable visual identification and polymer characterization via optical microscopy. All the filters were placed in sealed glass Petri dishes to prevent contamination and dried in an oven at 70°C for 30–60 minutes prior to quantification and identification.

4) Quantification and identification

A stereomicroscope (Olympus SZ61, equipped with a 10× eyepiece and a zoom range of 6.7x to 45x) was used for visual identification and enumeration of suspected MPs. This magnification range allowed reliable observation and detailed characterization of particles ≥100 µm. In contrast, particles in the 45–100 µm size class were quantified, but their morphological and color features were interpreted cautiously due to limited visual resolution. MPs were manually counted and classified according to size, shape, and color (Zhou et al., 2022). The size classes included 45–100 µm, 101–300 µm, 301–500 µm, 501–1,000 µm, and 1,001–5,000 µm. The lower size class (45–100 µm) corresponds to particles retained on the GF/C filter after passing through the 100 µm stainless steel sieve, whereas the upper size limit reflects the exclusion of particles larger than 5 mm.

Particles within the 45–100 µm size class were quantified on the basis of their visibility under maximum magnification; however, classification by color and shape was performed only when these features could be clearly distinguished owing to optical limitations. Ambiguous particles were excluded from further morphological interpretation. For particles ≥100 µm, size, shape (fibers, fragments, films, beads, and foams), and color were classified following visual criteria commonly applied in previous MP studies (Hidalgo-Ruz et al., 2012; Sun et al., 2019). Chemical identification was conducted via attenuated total reflectance Fourier transform infrared (ATR-FTIR) spectroscopy (LUMOS II, Bruker). Owing to methodological limitations (Magnusson and Norén, 2014; Murphy et al., 2016), ATR-FTIR analysis was applied only to a randomly selected subset of visually identified particles, primarily those larger than 500 µm, for which the spectral quality is more reliable (Bretas et al., 2020). The polymer types were determined by comparing the obtained spectra with reference spectral libraries (Maw et al., 2024).

5) Data analysis

Data analysis was performed via Microsoft Excel. The MP concentration was expressed as the number of particles per liter of water sample (particles L^{-1}), representing the quantity of MPs detected in 1 L of the sampled water (Zhou et al., 2022). The MP removal efficiency (RE%) was calculated by comparing the number of MPs in the influent and effluent, as shown in Eq.1 (Maw et al., 2024; Murphy et al., 2016).

$$\mathrm{RE\%} = \frac{(\mathrm{MPs\ influent}) - (\mathrm{MPs\ effluent})\ \mathrm{x}\ 100\%}{(\mathrm{MPs\ influent})} \quad \text{(Eq.1)}$$

The daily quantities of MPs discharged into the environment were estimated by multiplying the MP concentration in the effluent (particles L^{-1}) by the treatment capacity of each WWTP ($L\ d^{-1}$) (Murphy et al., 2016). To evaluate whether the difference in MP removal efficiency between WWTP A and WWTP B was statistically significant, a Mann–Whitney U test was performed. This nonparametric test was selected because of the small sample size and the potential nonnormal distribution of the data. The test was conducted via simulated datasets derived from the reported mean and standard deviation values for each plant, with the significance level set at $p < 0.05$.

6) Quality assurance and quality control (QC/QA)

To ensure the reliability and validity of the MP data, several QA/QC measures were implemented throughout the sampling, processing, and analytical stages. All equipment used, including stainless steel buckets and glass petri dishes, was made of metal or glass to avoid plastic contamination (Liu et al., 2021; Sun et al., 2019). Laboratory personnel wore cotton laboratory coats to minimize synthetic fiber shedding (Lares et al., 2018; Pirc et al., 2016). Before and after use, all the materials (bottles, sieves, and filtration units) were rinsed three times with filtered distilled water and covered with aluminum foil to prevent airborne contamination (Murphy et al., 2016). Procedural blanks (filtered distilled water processed alongside real samples) were included in every batch to monitor background contamination. No significant contamination was detected in these blanks. Sample processing took place in a clean area, and the filters were always covered with aluminum foil or lids to further reduce airborne contamination (Browne et al., 2011). These procedures aim to minimize potential contamination and increase the accuracy of MP detection.

Results and discussion

1) MP concentration and removal efficiency

The abundance of MPs in the influent and effluent, as well as the removal efficiencies of both WWTPs, are summarized in Table 2. A comparison between the two plants is presented in Figure 3. The influent concentration of MPs was 17.1±5.65 particles L^{-1} at the Setiabudi WWTP (A) and 15.45±4.31 particles L^{-1} at the Bojongsoang WWTP (B). After treatment, the concentrations decreased to 1.41±0.01 and 1.5±0.32 L^{-1} particles, respectively. These results confirm that both WWTPs are effective at removing MPs, with removal efficiencies of 91.75% for WWTP A and 90.32% for WWTP B. Although WWTP A showed slightly higher efficiency, the difference was not statistically significant ($p > 0.05$), indicating comparable performance despite their different treatment designs. This lack of significance may be due to overlapping performance ranges and variability in influent MP characteristics and flow rates, as reported in previous studies (Liu et al., 2021; Sun et al., 2019).

The high removal observed is consistent with reports from other countries (Table 3), where WWTPs achieve efficiencies of 80–95% depending on the treatment type (Dris et al., 2015; Hidayaturrahman et al., 2019; Koyuncuoğlu and Erden, 2023; Murphy et al., 2016; Magni et al., 2019; Magnusson et al., 2014; Maw et al., 2024). Variations in influent concentrations are influenced by population density, wastewater composition, and human activities (Liu et al., 2022). The removal performance also reflects the treatment technologies applied, with studies showing that more advanced processes generally achieve greater removal than conventional systems do (Badawi et al., 2025; Hadi et al., 2024; Lapointe et al., 2020; Rajala et al., 2020). Although the relative efficiencies are high, the estimated daily MP discharges remain considerable due to the large treated volumes. This highlights the potential ecological risks of the release of MPs into receiving waters, including biodiversity loss and the disruption of aquatic ecosystems (Ziajahromi et al., 2016). These findings underscore the need to strengthen wastewater treatment practices and regulatory measures to further reduce MP emissions.

Table 2 Average number of potential MPs released per day and year

Site	Sampling point	MPs Concentration			%RE
		particles L^{-1}	particles d^{-1}	particles $year^{-1}$	
WWTP A (Setiabudi WWTP)	Influent (A_1)	17.1 ± 5.65	4.28×10^6	1.56×10^9	-
	Effluent (A_2)	1.41 ± 0.01	3.53×10^5	1.29×10^8	91.75
WWTP B (Bojongsoang WWTP)	Influent (B_1)	15.45 ± 4.31	1.24×10^9	4.51×10^{11}	-
	Effluent (B_2)	1.50 ± 0.32	1.20×10^8	4.38×10^{10}	90.32

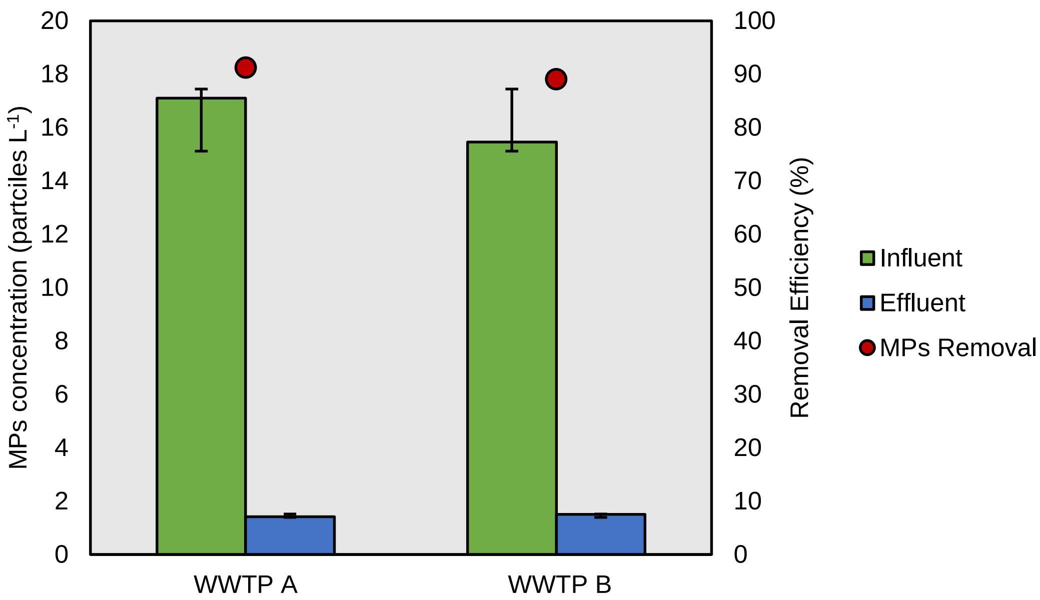


Figure 3 MP concentration and removal efficiency in the two WWTPs studied.

Table 3 Influent and effluent MP concentrations in several WWTPs in different countries

WWTP location	Capacity	MPs concentration (particles L^{-1}; particles g^{-1})			Potential discharge (particles d^{-1})	Treatment process	Removal rate (%)	Reference
		Influent	Effluent	Sludge				
Italy	400,000,000 L d^{-1}	2.5±0.3	0.4±0.1	113±57	160,000,000	Pre, primary, secondary (activity sludge) and tertiary (sand filter and disinfection)	84%	Magni et al., 2019
Scotland (River Clyde, Glasgow)	260,954 m^3 d^{-1}	15.7±5.2	0.25±0.4	19.67±4.51	65,238,500	Pre, primary, and secondary (aeration & clarifier)	98.4%	Murphy et al., 2016
France	2.4 x 10^5 m^3 d^{-1}	260–320	14–50	N/A	N/A	Sedimentation, biofilter	88.1%	Dris et al., 2015
South Korea	172,211.3 m^3 d^{-1}	4,200– 31,400	33–297	N/A	47.24 x 10^9	Primary, secondary (bioreactor), tertiary (coagulation, ozone, membrane, RSF)	98%	Hidayaturrahman et al., 2019
Sweden	5,160 m^3 d^{-1}	15.1±0.89	0.0082	8.36±0.98 x 10^3	4.25 x 10^4	Primary and secondary	99.9%	Magnusson et al., 2015
Vietnam	30,000 - 350,000 m^3 d^{-1}	4.3– 51.9	1.3–4.2	20–214	N/A	Primary and secondary (activated sludge)	50 - 96.8%	Maw et al., 2024
China	150,000 m^3 d^{-1}	288.5±32.8	22.9±7.2	128	3.4 x 10^9	Primary, secondary (bioselection tank, oxidation ditch), and tertiary (contact tank)	92.1%	Yang et al., 2021
India	131,400 m^3 $year^{-1}$ (Flow rate: 360 m^3 d^{-1})	64.3±4.89 – 47.66±4.71	24.33±2.16 – 28 ± 2.1	1.14±0.30 – 1.38±0.65	9360	Primary, secondary (aeration tank and tube settler)	37.30 - 41.46%	Parashar et al., 2022
Bandung, Indonesia (Communal scale)	6.08 - 11.88 particles L^{-1}	493.33–573.33	80–133.33	3.957±0.284	N/A	Primary and secondary (anaerobic system, sedimentation, and filtration)	76.74 - 83.78%	Fauzi et al., 2024
Jakarta, Indonesia (Centralized)	250 m^3 d^{-1}	17.1±5.65	1.41±0.01	N/A	3.53 x 10^5	Primary and secondary (MBBR with high-rate clarifier)	91.75%	This study
Bandung, Indonesia (Centralized)	80,000 m^3 d^{-1}	15.45±4.31	1.50±0.32	N/A	1.20 x 10^8	Primary and secondary (waste stabilization pond)	90.32%	This study

Note: N/A = not available (data not reported in the referenced study).

2) MP characteristics

2.1) MP sizes

Figure 4 shows the size distribution of MPs in the influent and effluent at each WWTP studied. Both WWTP A (Setiabudi) and WWTP B (Bojongsoang) displayed similar size distribution patterns, with the highest abundance observed in the 1000–5000 μm and 100–300 μm size ranges. In Setiabudi's influent, MPs with sizes of 1,000–5,000 μm and 100–300 μm accounted for 6.00 particles L^{-1} (35.08%) and 3.95 particles L^{-1} (23.09%), respectively. After treatment, these values decreased to 0.44 particles L^{-1} (31.21%) and 0.38 particles L^{-1} (26.95%) in the effluent. Similarly, in Bojongsoang, the influent contained 4.90 particles L^{-1} (31.71%) and 3.80 particles L^{-1} (24.56%) within the same size ranges, which decreased to 0.41 particles L^{-1} (27.42%) and 0.31 particles L^{-1} (20.54%) in the effluent. Overall, larger MPs were more efficiently removed during treatment, whereas smaller MPs were more frequently detected in the effluent.

The presence of smaller MPs in the effluent may result from the fragmentation of larger plastic particles through physical, chemical, and biological processes occurring during wastewater treatment (Magni et al., 2019). While smaller MPs can pass through treatment units and reach the effluent, their interpretation, particularly for particles less than 100 μm, should be treated with caution and is therefore not emphasized in detailed morphological or polymer-specific analyses. The frequent detection of MPs smaller than 500 μm in effluent samples is consistent with previous studies reporting reduced removal efficiency for smaller size fractions (Hidayaturrahman et al., 2019; Long et al., 2019; Sun et al., 2019).

2.2) MP shapes

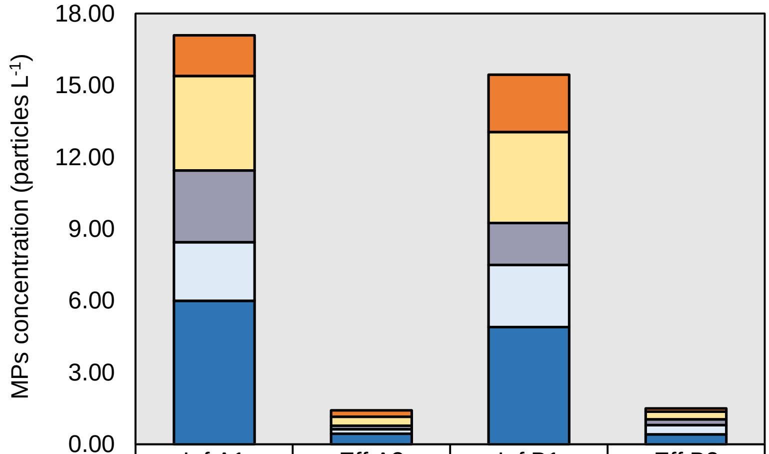
Figure 5 presents the distribution of different MP shapes in the influent and effluent samples from both WWTPs, as identified under a microscope following established criteria (Hidalgo-Ruz et al., 2012). Fibers and fragments were the most dominant shapes in both the influent and effluent samples. In the Setiabudi WWTP (A) influent, fibers accounted for 70.17% (12±7.21 particles L^{-1}), fragments accounted for 23.68% (4.05±1.2 particles L^{-1}), and other shapes (beads, film, foam) accounted for 6.14% (0.35±0.12 particles L^{-1}). The high proportion of fiber likely results from laundry wastewater, as supported by previous studies (Lares et al., 2018; Pirc et al., 2018). This abundance of fiber is concerning, as it can harm aquatic organisms and disrupt food chains (Maw et al., 2024). Fragments, with their irregular shapes, are typically produced by the breakdown of larger plastic items (Carr et al., 2016). Consistent with other studies, fibers and fragments are more prevalent in wastewater than other MP shapes are (Blair et al., 2019). Film and foam may originate from plastic packaging, whereas microbeads are commonly found in personal care products (Sun et al., 2019). In the effluent from the Setiabudi WWTP, the composition shifted slightly: fibers made up 68.08% (0.96±0.01 particles L^{-1}), fragments 26.24% (0.37±0.01 particles L^{-1}), and others 5.67% (0.02±0.01 particles L^{-1}). The persistence of microfibers in the effluent is likely due to their thin, elongated shape, which allows them to escape treatment processes, whereas irregular fragments are more easily retained (Wei et al., 2020). Bojongsoang WWTP (B) showed a different pattern in the influent, with a lower proportion of fibers (60.51%, 9.35±1.62 particles L^{-1}) and a higher proportion of fragments (36.89%, 5.7±2.26 particles L^{-1}), whereas other shapes made up 2.58% (0.13±0.14 particles L^{-1}). In the effluent, the fiber content increased to 72.88% (1.08±0.14 particles L^{-1}), the fragment percentage decreased to 23.33% (0.35±0.12 particles L^{-1}), and the other percentage rose slightly to 3.77% (0.03±0.01 particles L^{-1}). These differences may reflect variations in service area size and population between the two WWTPs. Overall, MPs, especially fibers and fragments, were not completely removed and remained at notable concentrations in the treated effluent, which is consistent with findings from other studies (Blair et al., 2019; Maw et al., 2024).

2.3) MP colors

Figure 5 shows the distribution of MPs by color in the two WWTPs studied. White or transparent MPs were the most dominant in both the influent and effluent samples, which is consistent with the findings of previous studies (Long et al., 2019; Zhou et al., 2022). In the Setiabudi WWTP, white or transparent MPs made up 35% of the total MPs (6.15±2.05 particles L^{-1}in the influent and 0.5 ±0.17 particles L^{-1} in the effluent). In the Bojongsoang WWTP, this proportion was even greater, at approximately 40% (6.2±0.98 particles L^{-1}in the influent and 0.54±0.12 particles L^{-1} in the effluent). Other observed colors included red, blue, green, brown, yellow, purple, and black. Across both WWTPs, red, black, and blue MPs accounted for 2.5–26.98% of the total, while the remaining colors were present in smaller amounts, each below 10%.

2.4) Polymer types for selected MPs

Figure 6 shows the FTIR spectra of the polymers recovered from the selected MP samples. The identified polymers, such as polypropylene (PP), polyethylene (PE), polyethylene terephthalate (PET), polyester (PES), and polystyrene (PS), are consistent with those commonly found in WWTPs (Sun et al., 2019; Maw et al., 2024; Ziajahromi et al., 2017), although their abundance can vary (Liu et al., 2021). While some studies have attempted to identify polymers on the basis of particle shape (Lares et al., 2018), this method does not accurately reflect the overall polymer distribution. In this study, PES and PET were found as fibers and fragments, typically originating from textiles such as yarn or clothing (Xu et al., 2021), although PET can also come from plastic bottles (Bretas et al., 2020). PEs mostly appear as fragments or filmlike flakes, whereas PPs are also found as fragments, likely from food packaging, beverage containers, or household plastics (Xu et al., 2021). PS have been identified as bead-like particles that are often associated with disposable food packaging (Bretas et al., 2020; Hidalgo-Ruz et al., 2012).



	Inf A1	Eff A2	Inf B1	Eff B2
45-100 µm	1.70	0.26	2.40	0.14
101-300 µm	3.95	0.38	3.80	0.31
301-500 µm	3.00	0.14	1.75	0.25
501-1000 µm	2.45	0.19	2.60	0.40
1001-5000 µm	6.00	0.44	4.90	0.41

Figure 4 Size distribution of MPs in influent and effluent from WWTPs A and B.

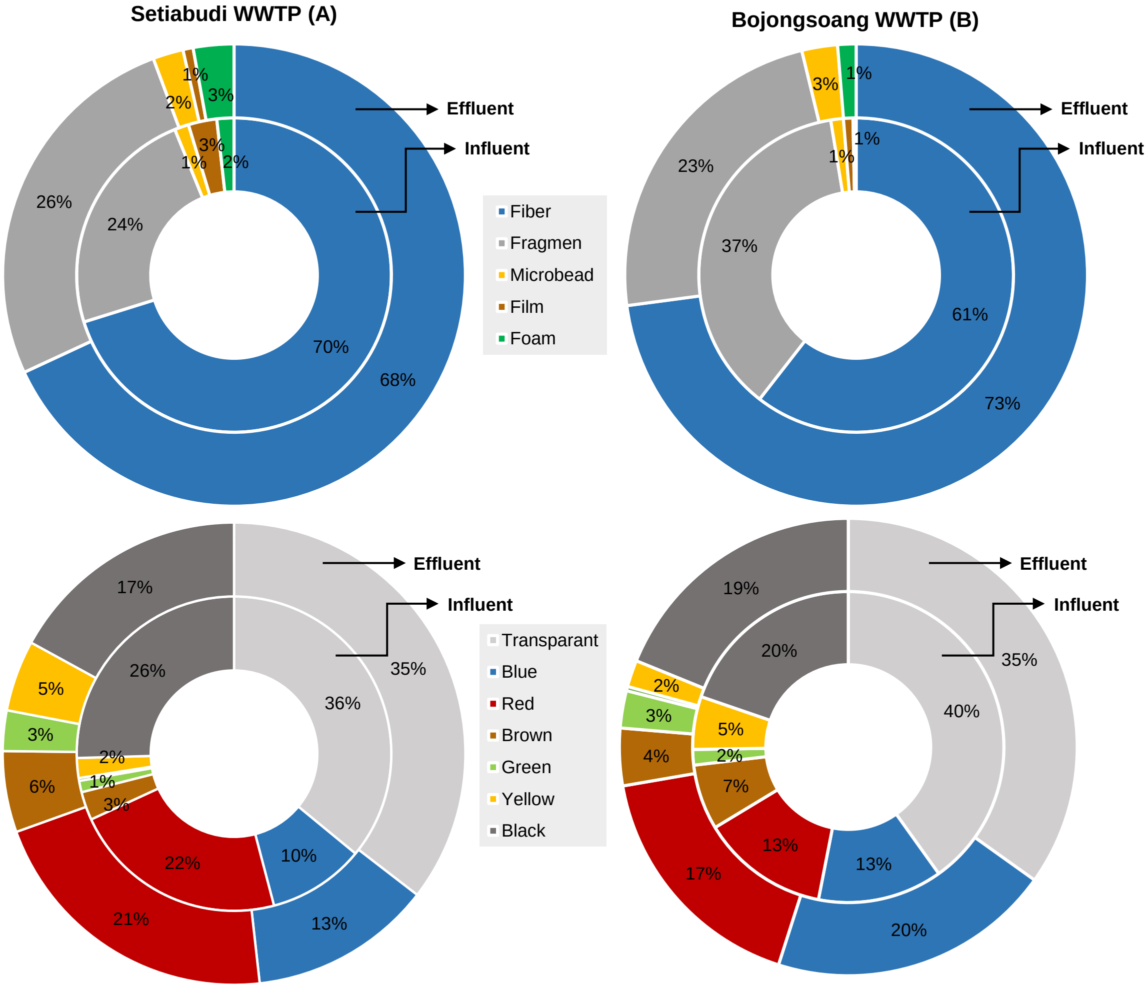


Figure 5 Shape and color composition of MPs identified in influent and effluent from WWTPs A and B. The inner ring indicates influent; the outer ring indicates effluent.

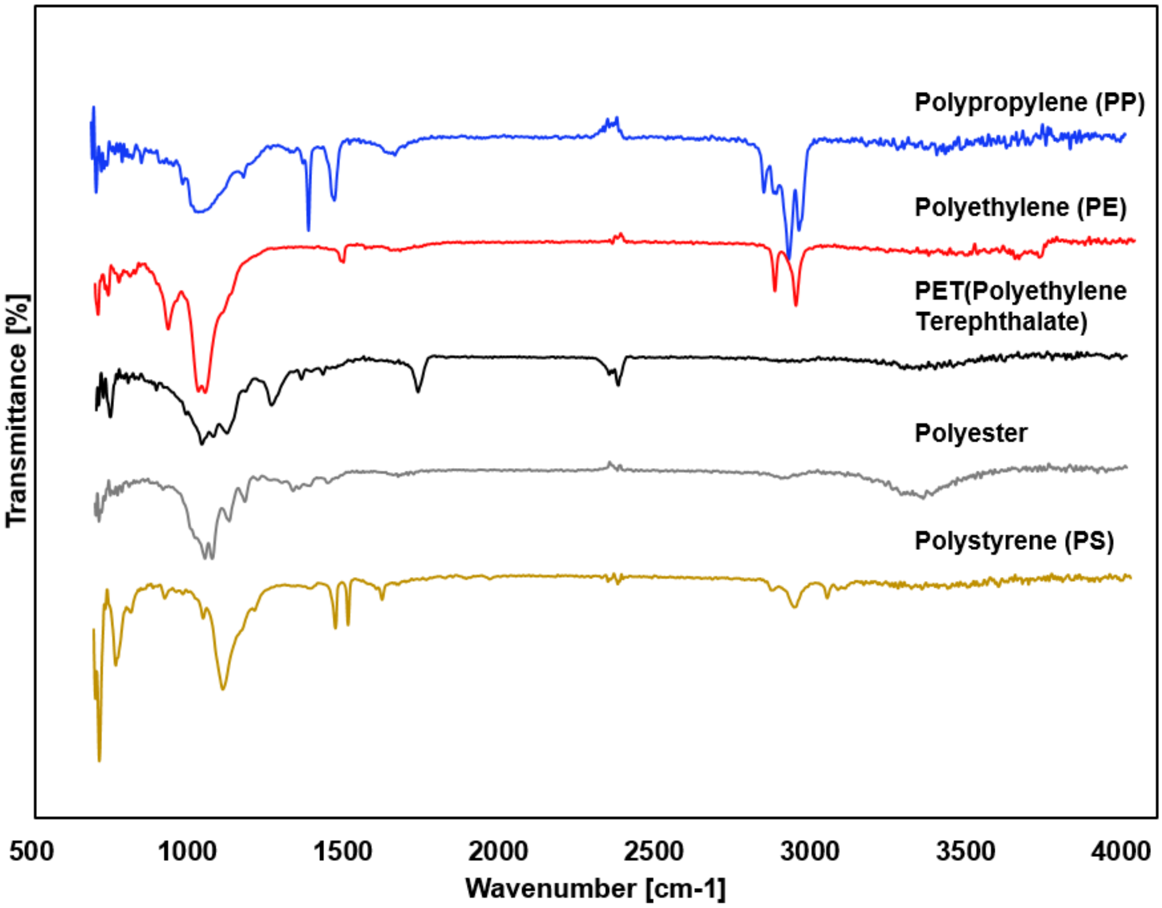


Figure 6 FTIR results for the detected polymers from the selected MP samples.

3) MP removal mechanisms in both WWTPs

The >90% MP removal observed in both WWTPs reflects the combined contribution of primary and secondary treatment processes. Primary units, such as bar screens, spiral sieves, and grit chambers, physically remove larger particles through skimming, trapping, and sedimentation. Previous studies reported that primary treatment alone can eliminate 25–45% of MPs (Talvitie et al., 2017; Xu et al., 2021; Ziajahromi et al., 2017). Secondary treatment provides an additional removal pathway, even though it is not specifically designed for MPs. In the Setiabudi WWTP (A), the MBBR likely contributed to MP reduction through biofilm adsorption, entrapment, and accumulation, which is consistent with findings from earlier research (Hidayaturrahman et al., 2019; Liu et al., 2021; Phu et al., 2022; Setiadewi et al., 2023). MPs can also act as carriers, influencing microbial communities and the fate of other contaminants (Amaral-Zettler et al., 2020; Huang et al., 2024). Coagulation–flocculation processes also increase MP removal through the formation of aggregates with other particulates, which subsequently settle during sedimentation (Xu et al., 2021). Previous studies have shown that optimized coagulation-flocculation can achieve up to 97–99% removal of diverse MP types under varying conditions (Badawi et al., 2025; Lapointe et al., 2020; Rajala et al., 2020), whereas biofilm-based systems such as MBBR ensure stable performance in handling fluctuating influent characteristics and heterogeneous MP compositions (Hadi et al., 2024). These findings highlight the operational advantages of advanced treatment technologies in maintaining consistent performance across different wastewater conditions.

In the Bojongsoang WWTP (B), the stabilization ponds rely on long hydraulic residence times (HRTs) that promote settling and biological activity (Kumar et al., 2020). Although these systems are designed mainly for organic matter and pathogen reduction, they can also achieve notable MP removal, with reported efficiencies of up to 90% (Mara et al., 1992). Another important factor is MP retention in sludge. More than 90% of removed MPs are known to accumulate in sludge (Alavian et al., 2021), raising concerns about sludge management as a potential secondary source of pollution if not properly handled. Although the exact mechanisms were not directly examined in this study, the observed reductions are likely associated with a combination of physical settling, adsorption to biofilms, and entrapment in sludge, as highlighted by previous research (Hidayaturrahman et al., 2019). Importantly, this study compared only influent and effluent concentrations and did not investigate unit-specific or stagewise removal processes. This study did not measure the removal efficiency at each treatment stage or assess MP accumulation in the sludge. Future research should therefore evaluate unit-level contributions and sludge pathways to fully understand MP removal mechanisms.

Conclusions

MPs were detected in wastewater samples from the studied WWTPs. The concentrations of MPs in the influent and effluent were similar between the two WWTPs, indicating comparable levels of contamination. However, a significant reduction in the MP concentration from influent to effluent demonstrated that the treatment processes were relatively effective at removing these micropollutants. Despite this overall decrease, differences in MP characteristics were observed between the two WWTPs, likely due to variations in design, operational conditions, and external environmental factors. Notably, WWTP A (Setiabudi), which uses more advanced treatment technology, achieved slightly higher MP removal efficiency than did WWTP B (Bojongsoang), which relies on a conventional natural treatment system. Nevertheless, even with high removal rates, WWTPs remain a significant source of MP pollution, as the large volume of treated wastewater still releases substantial amounts of MPs into receiving water bodies. To better understand the fate of MPs in WWTPs, further research is needed, particularly studies on MP distribution across different treatment processes, retention in sludge, and the effects of seasonal variations on removal efficiency.

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Data availability statement

Information and data used in the study will be disclosed upon request.

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Conflicts of interest

The authors declare that there are no conflicts of interest in competing financial or personal relationships that could have appeared to influence the work reported in this work.

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