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# Spatiotemporal patterns, loads, and risks of microplastics in the stormwater system of a subtropical coastal city

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A B S T R A C T

Urban stormwater runoff is a critical pathway for microplastics (MPs) to enter marine ecosystems, posing a significant threat. However, systematic studies on spatiotemporal patterns, loads, and risks of MPs in stormwater systems, particularly in subtropical coastal cities, remain limited. This study characterized MPs in Macao's stormwater drains across seasons, using density separation, $H_2O_2$ digestion, stereomicroscopy, and μ-FTIR analysis to investigate their abundance and distribution. MPs were ubiquitous across all sample sites, with abundance ranging from 0.86 to 53.03 n/L. Irregular shaped fragments (56.78 ± 0.13.80%) and the color black (38.44 ± 7.47%) were dominant, and the most common polymers were Polyethylene terephthalate (39.55 ± 16.63%), polypropylene (23.05 ± 18.15%), and polyethylene (9.52 ± 8.87%). While abundance of MPs was higher in the wet season, no significant seasonal variation was found. MP characteristics varied significantly with land use. The concentration range of MPs was estimated to be from 2.86 ± 0.38 to 3145.94 ± 766.74 μg/L, with an overall mean of 454.68 ± 572.20 μg/L through the mass concentration models. The daily MP emission load into the receiving water was estimated to range from $8.29 \times 10^3$ to $3.09 \times 10^8$ particles per day or 2.66 to 10,365.23 g/day. MP risk was high (Polymer Hazard Index/Potential Ecological Risk Index: III/V) due to hazardous polymers, with spatiotemporal variations further observed from fluctuating water flow erosion. Correlation analysis revealed that MP abundance showed a significant negative correlation with dissolved oxygen and a positive correlation with antecedent rainfall. These findings identify stormwater drains as a key MP conduit, influenced by rainfall and land use, calling for targeted management in urban coastal areas.



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Abbreviations

| Abbreviation | Full Name |
|---|---|
| MPs | Microplastics |
| PLI | Pollution load index |
| PHI | Polymer hazard index |
| PERI | Potential ecological risk index |
| T | Temperature |
| DO | Dissolved oxygen |
| EC | Electrical conductivity |
| S | Salinity |
| CA | Cellulose acetate |
| PET | Polyethylene terephthalate |
| PE | Polyethylene |
| PVC | Polyvinyl chloride |
| PP | Polypropylene |
| PAN | Polyacrylonitrile |
| EVA | Ethylene-vinyl acetate |
| PVA | Polyvinyl alcohol |
| BOD | Biochemical oxygen demand |
| ADD | Antecedent dry days |
| WWTPs | Wastewater treatment plants |

## 1. Introduction

Plastic particles with a size of less than 5 mm are referred to as microplastics (MPs) [1]. In recent decades, MPs have emerged as a widespread pollutant and a source of increasing global concern. In urban environments, major sources of MPs include municipal solid waste, textile abrasion, vehicle tire wear, and the aging and fragmentation of various plastic products [2,3]. Unlike point source pollution, urban non-point source runoff transports MPs accumulated on surfaces into rivers and coastal receiving waters via stormwater systems of coastal cities, constituting a critical pathway for MPs to enter the marine environment [4,5].

Beyond their physical impacts, MPs can act as carriers for co-occurring contaminants due to their large surface area and high adsorption capacity. Pollutants such as antibiotics and persistent organic pollutants may sorb onto MPs, potentially enhancing ecological toxicity (Franzellitti et al., 2019; Pan et al., 2023; Yu et al., 2020). In addition, tire wear particles, an important MP source, release 6PPD-quinone, which has been identified as a major cause of acute mortality in coho salmon (Tian et al., 2021). Because stormwater is often discharged directly into receiving waters without treatment, storm drainage systems represent an important yet underregulated pathway for MP transport.

Over the past decade, research on MPs has grown exponentially. However, studies specifically targeting stormwater MPs remain limited. While MP pollution in stormwater systems has begun to attract attention in recent years, details about their distribution and abundance within storm drains is still incomplete due to a lack of systematic studies [6,7]. The abundance and characteristics of MPs in stormwater systems are influenced by various factors, including catchment land use patterns and seasonal variations [4,8]. Moreover, MP concentrations exhibit significant temporal variability across individual rainfall events [9]. This variability introduces substantial uncertainty in characterizing the spatiotemporal distribution of MPs in storm drains and in making cross-regional comparisons. Therefore, investigating the occurrence, distribution characteristics, ecological risks, and flux of MPs from coastal urban stormwater systems into the marine environment is crucial for accurately assessing MP pollution levels and associated risks.

This study investigated the typical stormwater drainage system of Macao, a subtropical coastal city and explored the spatiotemporal distribution and pollution risks of MPs. For the first time, a mass concentration model was employed to estimate the load of MPs discharged from the urban island's storm drains into the surrounding marine environment from July 2024 to February 2025.

## 2. Materials and methods

### 2.1. Sampling locations and sampling

The study area was located on the islands of Taipa, Coloane, and the Cotai reclamation area in the Macao SAR (special administrative region). A total of six sampling sites were established and categorized into residential/commercial areas, industrial areas, and ecological areas based on the land use classification in the master plan for Macao (2020-2040). Detailed site information is provided in Table 1 and Fig. 1, while the rationale for the sampling-site layout is described in Text S1. Sampling was conducted from July 2024 to February 2025, with three sampling events in each of the dry and wet seasons. During each event, water quality parameters—including temperature (T), pH, dissolved oxygen (DO), electrical conductivity (EC), and salinity (S)—were measured in situ using a multi-parameter water quality meter (Hach HQ-4300, USA). Additionally, water depth, flow velocity, and channel width were determined at each sampling site using flow meter (LS1206B, China), measuring tape, and laser rangefinder. Water samples were collected following the field-sieving method described by Zhang et al. [10]. Briefly, a total of 50 L of water was collected using a stainless-steel bucket and sequentially passed through a 300 μm stainless steel sieve and a 20 μm plankton net. The retained particulate matter on both sieves was then rinsed into sample bottles using ultrapure water. Three parallel samples were collected during each sampling event. All samples were transported to the laboratory and stored at 4 °C until further processing.

### 2.2. MP extraction

An appropriate volume of $H_2O_2$ solution (30 wt%, v/v) was added to the filtered samples in a glass bottle, followed by heating in a water bath at 60 °C for 48 h. The reaction was considered complete when no further bubbling was observed. Subsequently, the supernatant was vacuum-filtered through 0.45 μm cellulose acetate (CA) membranes to collect floating MPs, and the MPs in the remaining sediment were collected through density separation. The sediment was mixed with a NaI solution (1.6 g/cm$^3$) and stirred thoroughly. After standing for layer separation, the supernatant was decanted off and filtered through a new 0.45 μm CA membrane, and the density separation step was repeated three times. The CA membrane was stored in the Petri dish.

### 2.3. MPs identification and classification

Each membrane was examined using a stereomicroscope and identified particulate material classified according to size, shape, and color, using a previously standardized classification system

**Table 1**
Land usage and geographical co-ordinates of sampling sites.

| Sampling site | Land use type | Location |
|---|---|---|
| S1 | Industrial area | 22°09′52.7″N 113°34′09.4″E |
| S2 | Residential/Commercial area | 22°09′53.6″N 113°33′19.8″E |
| S3 | Residential/Commercial area | 22°09′16.0″N 113°32′33.1″E |
| S4 | Ecological area | 22°08′28.2″N 113°33′06.8″E |
| S5 | Industrial area | 22°07′41.9″N 113°33′05.5″E |
| S6 | Residential/Commercial area | 22°06′56.2″N 113°33′05.9″E |

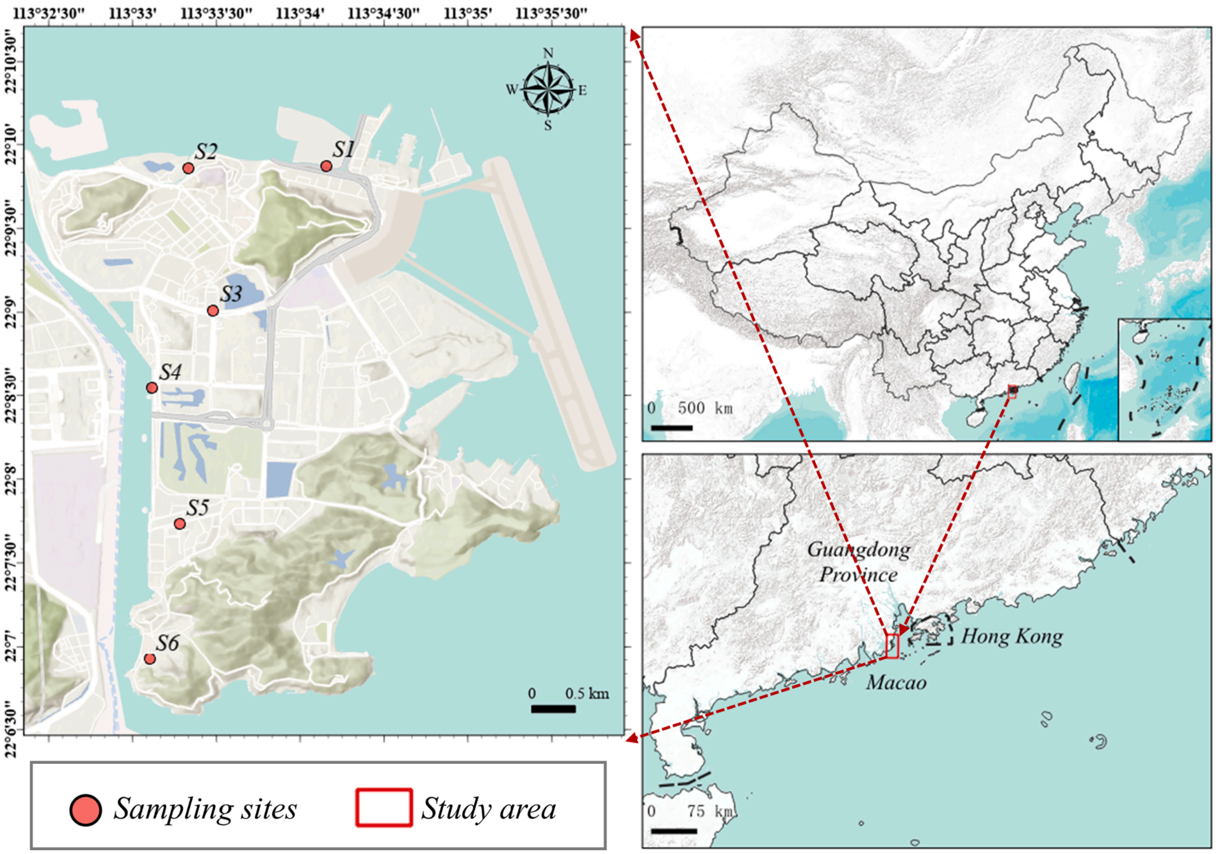


**Fig. 1.** Study area and sampling sites.

[10], the size categories of particulate material were divided into 20-100 μm, 100-200 μm, 200-300 μm, 300-500 μm, 500-1000 μm, and 1000-5000 μm; shapes were classified as fiber, film, fragment, and foam; and colors included white, black, blue, green, yellow, red, and transparent. 1868 particles were randomly identified using μ-FTIR (Nicolet iN10, Thermo Scientific), with the spectral range set at 4000 $cm^{-1}$ to 600 $cm^{-1}$ and using 16 scans per measurement. The obtained spectra for particles were compared with the reference spectra from the open-access databases, OMNIC Commercial Library.

### 2.4. Risk assessment of MP pollution

Risk assessment of MPs in stormwater drains was conducted using three established models: the Pollution Load Index (PLI), the Polymer Hazard Index (PHI), and the Potential Ecological Risk Index (PERI). The PLI primarily evaluates risk based on microplastic abundance [11]. In contrast, the Polymer Hazard Index (PHI) assesses potential ecological risks by quantifying the hazard indices of different polymer types [12]. The PERI is a more comprehensive model that integrates elements of both the PLI and PHI. The specific formula used for the calculations are provided below, and the corresponding risk classification criteria can be found in Table S1.

$$PLI_i = \sqrt{C_i/C_0}$$

$$PLI_{zone} = \sqrt[n]{PLI_1 \times PLI_2 \times \cdots \times PLI_n}$$

Where, $C_i$ is the MP abundance of the $i$ th sampling site. The background value ($C_0$) for MP abundance was set as the lowest value measured in this study; and $PLI_i$ is the pollution load index at the $i$ th sampling site. $PLI_{zone}$ is the pollution load index of MP in the whole region.

$$PHI = \sum P_n \times S_n$$

$P_n$ refers to the percentage of MP polymers identified for each sampling point, $S_n$ is the ecological risk score of the polymer [12].

$$PERI_i = PLI_i \times PHI_i$$

$$PERI_{zone} = \sqrt[n]{PERI_1 \times PERI_2 \times \cdots \times PERI_n}$$

$PERI_i$ is the potential ecological risk index of MP at the $i$ th sampling site. $PERI_{zone}$ is the potential ecological risk index in the whole region.

### 2.5. Data analysis

The homogeneity of variance for MP data was assessed using Levene's test, while normality was evaluated using the Shapiro–Wi; lk test and Q–Q plots. For datasets meeting parametric assumptions, one-way analysis of variance (one-way ANOVA) and Student's $t$-test were applied to examine differences in MP characteristics. For datasets that did not meet parametric assumptions, Spearman's rank correlation analysis, the Kruskal–Wallis test (K–W test), and the Mann–Whitney $U$ test were employed. A significance level of 0.05 was adopted for all statistical tests. Data analysis was performed using Excel 2019 (Microsoft Corp., USA) and SPSS 27.0 (IBM Corp., USA), while graphing and mapping were conducted using Origin 2024 (OriginLab Corp., USA), Python 3.11, and ArcGIS Pro 3.4 (Esri Inc., USA), respectively.

### 2.6. Estimation of MP mass concentration

In recent years, various conversion models have been developed to estimate microplastic mass concentrations from particle abundance data [13,14]. This study adopted the model proposed by Zhang et al. [15], which is based on the following core formula:

$$C_{shape} = \sum_i N_{i,shape} \times \left(\rho \times V_{i,shape}\right)$$

$$N_{i,shape} = A \times p_{shape} \times p_i$$

Where $A$ represents the total MP abundance; $p_{shape}$ denotes the proportion of a specific shape within the total MPs; $p_i$ indicates the proportion of the $i$ th size bin among MPs of the same shape; $\rho$ represents the plastic density; $V_{i,shape}$ denotes the volume of a single particle in the $i$ th size bin of a given shape; and $N_{i,shape}$ represents the abundance of the $i$ th size bin for that specific shape.

Fiber, fragment, and film were modeled as cylinders, while foam were approximated as sphere [13,15]. Polymer density was calculated using a weighted average method based on polymer type. The conversion formulas used for microplastics of different shapes were as follows:

$$C_{fiber} = \sum_{i=1}^{n} A \times p_{fiber} \times p_i \times \rho\pi\left(\frac{d_f}{2}\right)^2 L_i = A\rho\pi \cdot p_{fiber} \sum_{i=1}^{n} \frac{1}{4} d_f^2 p_i L_i$$

$$C_{fragment} = \sum_{i=1}^{n} A \times p_{fragment} \times p_i \times \rho\pi\left(\frac{L_i}{2}\right)^2 \times \frac{L_i}{10} = A\rho\pi \cdot p_{fragment} \sum_{i=1}^{n} \frac{1}{40} p_i L_i^3$$

$$C_{film} = \sum_{i=1}^{n} A \times p_{film} \times p_i \times \rho\pi\left(\frac{L_i}{2}\right)^2 h = A\rho\pi \cdot p_{film} \sum_{i=1}^{n} \frac{1}{4} h p_i L_i^2$$

$$C_{foam} = \sum_{i=1}^{n} A \times p_{foam} \times p_i \times \frac{4}{3}\pi\left(\frac{L_i}{2}\right)^3 = A\rho\pi \cdot p_{foam} \sum_{i=1}^{n} \frac{1}{6} p_i L_i^3$$

Where $d_f$ denotes the fiber diameter, $L_i$ is the median value of the $i$ th size bin, and $h$ represents film thickness.

The total MP mass concentrations:

$$C_{MP} = 10^9 \times \sum_{shapes} C_{shape}$$

The factor $10^9$ is used to convert the calculated mass concentration to μg/L.

The parameters used for calculating the mass concentration of MPs, along with their corresponding units, are summarized in Table 2.

**Table 2**
Parameters used in the MPs mass concentration model.

| Symbol | Definition | Unit |
| --- | --- | --- |
| $C_{MP}$ | Total modeled mass concentration of MPs | μg/L |
| $A$ | Total MP abundance | n/L |
| $\rho$ | Weighted average polymer density | kg/m$^3$ |
| $L_i$ | Median value of the $i$ th size bin | m |
| $d_f$ | Assumed fiber diameter | m |
| $h$ | Film thickness | m |
| $p_{shape}$ | Proportion of a specific MP shape | dimensionless |

### 2.7. MPs emission load estimation

MP load discharged through stormwater drains may be a key factor influencing the abundance and composition of MPs in receiving coastal waters and their subsequent deposition along adjacent shorelines [5]. The formula used to estimate the daily discharge of MPs in stormwater drains was as follows:

$$Load = v_i \times S \times C$$

Where $v_i$ represents the flow rate at the sampling site, $S$ denotes the cross-sectional area, and $C$ was the MP concentration, which was calculated using either MP abundance or mass concentration.

### 2.8. QA/QC

The sampling tools used in the study were constructed from non-plastic materials. Prior to use, they were thoroughly cleaned and rinsed at least three times with ultrapure water. To minimize potential contamination from the bottle cap, the interior of each sampling bottle cap was lined with tin foil. During both field sampling and laboratory procedures, cotton lab coats and nitrile gloves were worn. Field blanks were prepared alongside the samples, which were wrapped in tin foil and securely sealed to prevent contamination during transport. All sample processing was performed inside a fume hood. Procedure blanks using ultrapure water as a control were implemented to evaluate background contamination and ensure analytical accuracy. Additionally, to test the efficiency and reliability of the sample pretreatment process recovery tests were performed in duplicate by spiking into ultrapure water polypropylene (PP) and polylactic acid (PLA) particles. Both tests demonstrated recovery rates exceeding 90%.

## 3. Result and discussion

### 3.1. MP abundance

MPs were detected in all stormwater samples (S1 – S6), with abundance ranging from 0.86 ± 0.32 n/L to 53.03 ± 22.73 n/L and an average value of 11.53 ± 11.03 n/L (Fig. 2a). During the dry season, average MP abundance was 8.53 ± 6.14 n/L, the highest average MP abundance was observed at site S5 (20.08 ± 3.92 n/L), followed by S1 (13.10 ± 2.33 n/L), S3 (9.76 ± 8.05 n/L), S6 (4.79 ± 1.86 n/L), S2 (4.70 ± 0.50 n/L), and S4 (1.49 ± 0.55 n/L). Similarly, in the wet season, average MP abundance was 14.53 ± 13.93 n/L, the highest abundance was found at S5 (38.85 ± 16.54 n/L), followed by S1 (16.37 ± 0.80 n/L), S3 (13.92 ± 0.81 n/L), S2 (12.10 ± 6.14 n/L), S6 (6.03 ± 3.81 n/L), and S4 (2.01 ± 0.64 n/L).

In samples from different land type usage, the industrial areas (S1 and S5) exhibited the highest MP abundance (21.59 ± 12.88 n/L), followed by the commercial/residential areas (S2, S3, S6) (8.57 ± 4.81 n/L), and finally by the ecological area (S4) (1.75 ± 0.61 n/L). A Kruskal–Wallis test revealed significant differences in MP abundance among land-use types. Post hoc pairwise comparisons showed that MP abundance in the industrial areas was significantly higher than the commercial/residential areas ($p < 0.001$) and the ecological area ($p < 0.05$). Additionally, the MP abundance in the commercial/residential areas was also significantly higher than that in the ecological area ($p < 0.01$) (Fig. 3a).

Stormwater runoff may be an important pathway for MPs to enter aquatic systems [9,16]. Compared to other global studies on stormwater, the MP pollution level in a subtropical coastal city in Macao was moderate to low. For instance, MP abundance in

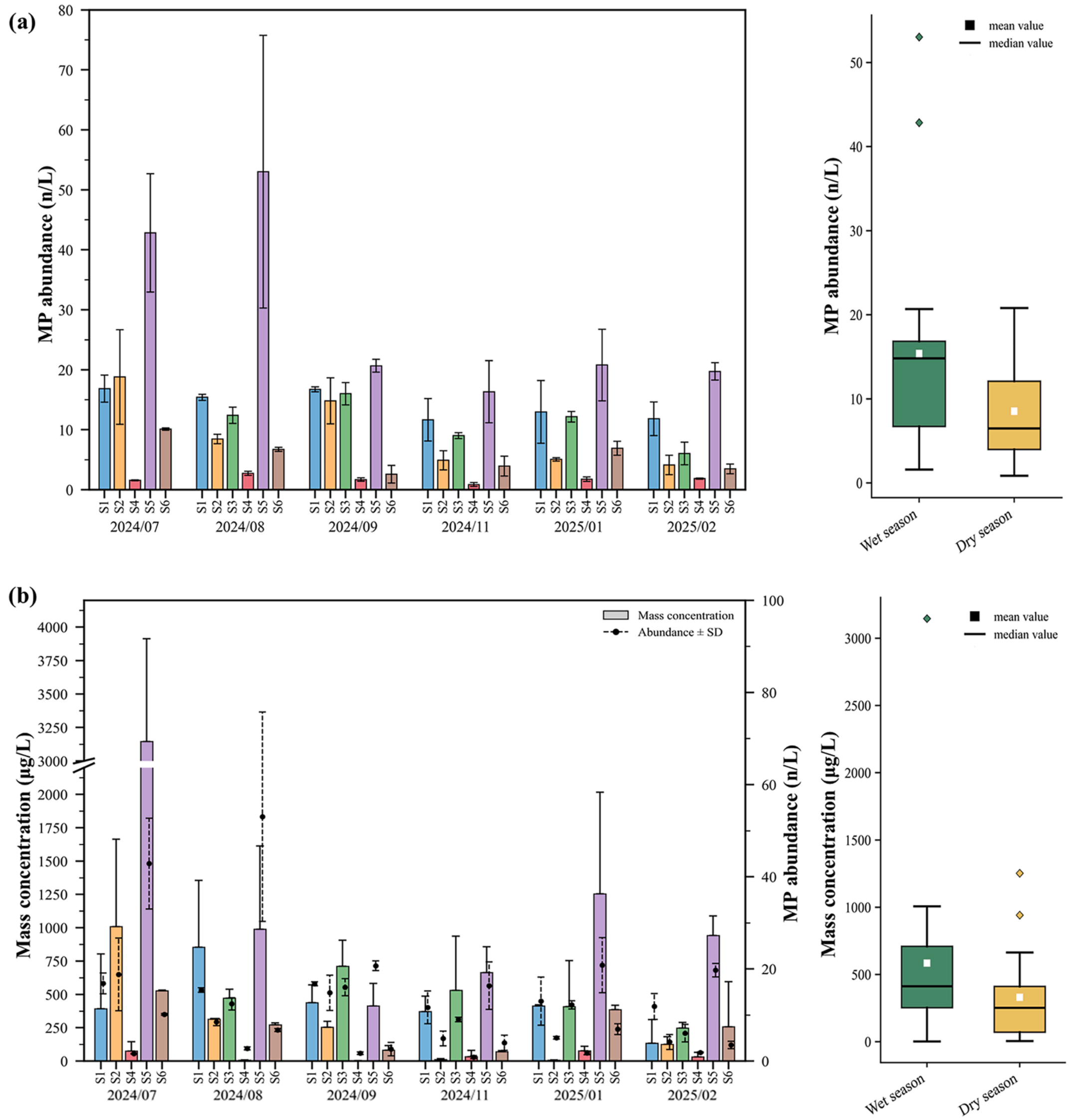


**Fig. 2.** Temporal and seasonal variations in (a) MP abundance and (b) mass concentration in Macao stormwater drains (n = 3).

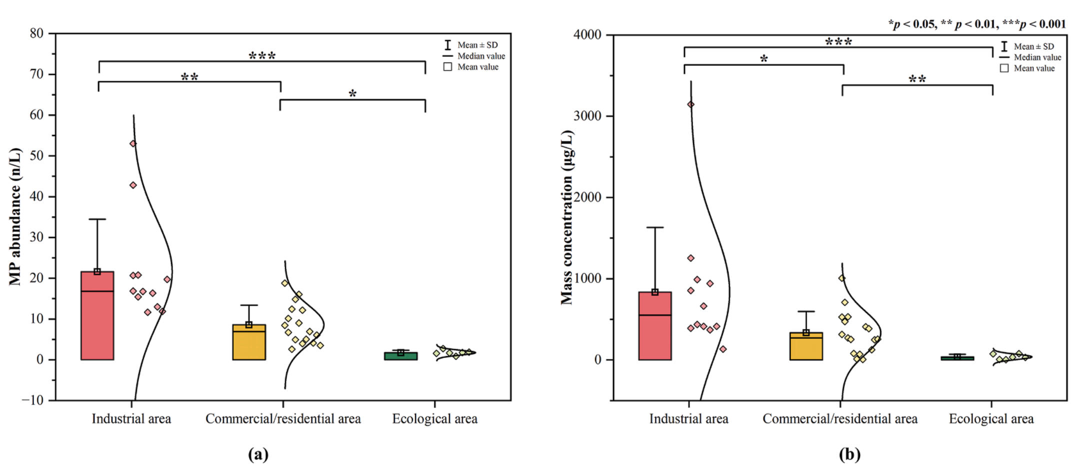


**Fig. 3.** Land-use differences in (a) MP abundance and (b) mass concentration in Macao stormwater drains.

stormwater from industrial and residential areas in Gumi, South Korea, ranged from 1 to 827 n/L and 2 to 1080 n/L, respectively [9]. Stormwater samples from Tijuana, Mexico, showed concentrations of 66 to 191 n/L [17]. Sang et al. [18] detected MP abundance between 2.75 ± 0.76 and 19.04 ± 2.96 n/L in stormwater drains from different areas of Wuhan, while Zhang et al. [10] reported a range of 0.4 to 36.48 n/L in stormwater samples from six major drains in Hong Kong.

### 3.2. Mass concentration of MPs

The modeled mass concentration of MPs in Macao stormwater drains ranged from 2.86 ± 0.38 to 3145.94 ± 766.74 μg/L, with an overall mean of 454.68 ± 572.20 μg/L, indicating substantial spatial and temporal variability (Figs. 2b and 3b). Among different land-use types, industrial areas exhibited the highest mass concentration (834.02 ± 797.60 μg/L), followed by commercial/residential areas (334.00 ± 263.68 μg/L), whereas the ecological area showed the lowest concentration (37.91 ± 32.30 μg/L). The mass concentration differed significantly among land-use types (Kruskal–Wallis test, $p < 0.001$). Pairwise comparisons further showed that industrial areas had significantly higher mass concentrations than both commercial/residential areas ($p < 0.05$) and the ecological area ($p < 0.001$), while commercial/residential areas also had significantly higher concentrations than the ecological area ($p < 0.01$) (Fig. 3b). The mean mass concentration was higher in the wet season (585.53 ± 731.49 μg/L) than in the dry season (331.10 ± 343.41 μg/L), although the seasonal difference was not statistically significant. It should be noted that the mass concentration was estimated using a geometric approximation model rather than direct particle weighing. This model provides a rapid and simple approach for converting MP abundance data into mass concentration estimates. In particular, irregular fragments were simplified into regular geometric forms, which may introduce uncertainty because fragment shapes are heterogeneous. Therefore, the modeled mass concentration should be interpreted as an approximation for comparing spatial and seasonal patterns and MPs emission load estimation, rather than an exact mass measurement.

In comparison with other studies, Rasmussen [19] estimated MP concentrations in stormwater retention ponds ranged from $0.2 \pm 0.2$ to $1.8 \times 10^4$ ($\pm 4.4 \times 10^3$) μg/L, which is consistent with the results of the present study. Current research on MP mass concentrations remains limited. Cavazzoli et al. [20] using TD-GC/MS reported effluent MP concentrations from five wastewater treatment plants (WWTPs) in the range of 215.0 to 761.5 μg/L, which aligns closely with the concentration range observed at site S1 in this study (134.13 to 855.07 μg/L). This similarity can be attributed to the fact that the storm drain at S1 received discharges from WWTP as well as pollution from other stormwater runoff sources.

### 3.3. Characteristics of MPs

#### 3.3.1. Color of MPs

MPs of various colors were detected across all sampling sites (S1 – S6), with black (38.44 ± 7.47%) and transparent (31.20 ± 7.80%) being the most common, followed by green (8.65 ± 4.92%), blue (7.10 ± 3.80%), red (6.83 ± 3.40%), white (4.45 ± 2.45%), yellow (1.93 ± 1.16%), and other colors (1.43 ± 1.00%) (Fig. S1c and Table S4). Black and transparent MPs were most prevalent in stormwater drains, a result consistent with previous studies. For instance, Ross et al. [21] found that black (33.5 ± 33.8%) and transparent (22.6 ± 31.3%) MPs dominated at fifteen stormwater sites in Calgary, Alberta, Canada. The high proportion of black MPs are mainly due to the presence of tire wear particles in most stormwater samples [8]. Ziajahromi et al. [22] noted that most of the black fragments in the sediment of stormwater wetlands were isobutylene (or butyl rubber) filled with synthetic rubber carbon, which can be introduced into water systems through surface runoff.

#### 3.3.2. Size of MPs

The size of MPs revealed a predominance of small particles, this finding aligns with the conclusions of previous studies [8,18]. The size distribution was as follows: 20–100 μm (32.79 ± 10.40%), 100–200 μm (15.58 ± 3.82%), 200–300 μm (11.30 ± 2.63%), 300–500 μm (11.51 ± 3.10%), 500–1000 μm (14.57 ± 4.32%), and 1000–5000 μm (14.25 ± 4.54%), as shown in Fig. S1b and Table S3. Notably, larger size fractions, especially 1–5 mm and 0.5–1 mm, were more abundant in the wet season ($t$-test, $p < 0.01$), whereas the smallest fraction, 0.02–0.1 mm, was markedly enriched in the dry season ($t$-test, $p < 0.001$).

#### 3.3.3. Shape of MPs

Fragments (56.78 ± 13.80%) and fibers (36.30 ± 13.37%) were the dominant MP shapes detected, while film (5.88 ± 4.00%) and foam (1.02 ± 1.43%) accounted for considerably lower proportions (Fig. S1a and Table S2). This distribution pattern is consistent with findings from previous studies [8,23]. The high proportion of fragments in the samples can be attributed to abrasion of larger plastics or incorrect disposal of plastic products such as food containers and beverage bottles. These items gradually break down due to weathering and mechanical wear after entering the environment, and eventually reach stormwater drains via runoff. Particles generated by wear of tires of urban vehicular traffic are another major source of fragment MPs [6]. On the other hand, clothing and textiles release a significant number of fiber-MPs due to wear of day-to-day usage and washing. The high population density and tourism-driven economy in Macao likely contributes to the elevated proportion of fiber-MPs. Seasonally, the proportion of fragments and fiber was significantly higher in the dry season than in the wet season (Mann–Whitney $U$ test, $p < 0.001$) presumably due to reduced rainfall and weaker hydrodynamic conditions.

#### 3.3.4. Chemical composition of MPs

A total of 1868 putative MP particles were randomly selected from the samples and identified via μ-FTIR analysis and confirmed 1267 were MPs, and consisted of 36 different polymer types. The most frequently detected polymers were Polyethylene terephthalate (PET, 39.55 ± 16.63%), Polypropylene (PP, 23.05 ± 18.15%), and Polyethylene (PE, 9.52 ± 8.87%); the polymer distribution is illustrated in Fig. S2 and Table S5. These polymers which are key components of textile fibers and packaging materials are a major source of environmental MP pollution [24].

Most previous studies have reported PE as the most common MPs in stormwater samples, with PP, PA, PS, and PET also frequently ranked among the top five polymer types [8]. While our findings were generally consistent with this pattern, some differences were observed. For example, Zhang et al. [10] reported that PP and PE collectively accounted for over 50% of MPs in six stormwater drains in Hong Kong, whereas in the present study, their combined proportion was somewhat lower, with a higher percentage identified for PET. The widespread occurrence of PET fibers is largely attributed to the shedding of textiles and a single wash of 5 kg of laundry is estimated to release up to 6 million microfibers [3,25]. Carney Almroth et al. [2] further highlighted that PET exhibits the highest fiber release rate among common synthetic textiles. Macao still has combined sewer systems, and possess the risk that fibrous microplastics in wastewater could overflow into the stormwater system. In summary, MPs generated by coastal urban activities can enter the marine environment through stormwater drains, and pose a potential risk to the marine environment including aquatic organisms.

### 3.4. The spatiotemporal distribution of MPs

#### 3.4.1. Spatial distribution

The composition and abundance of MPs in stormwater discharges may vary with land use types and the intensity of human activity [4,8]. Lutz et al. [4] reported that MP abundance increased with the intensity of land use and urbanization around urban storm drains. In this study, significant differences in MPs were observed among storm drains in different land use areas. Sampling sites in the industrial area, such as the S1 had sources of MPs since it contained multiple auto repair shops, factories, and a wastewater treatment plant in the proximity of open stormwater channels. In contrast, site S4 was characterized by low land use intensity, and had lower MP pollution. A similar relationship between land usage and MP abundance in storm drains of industrial versus commercial/residential areas, was previously reported in Melbourne, Australia (Kabir et al. [26].

Regarding the abundance of MP morphological characteristics, no significant differences were found between industrial and commercial/residential areas. In the ecological area (S4), the proportion of fibrous MPs was 41.64 ± 15.86%, higher than that in commercial/residential (34.62 ± 10.87%) and industrial areas (36.03 ± 15.16%). This suggests fewer sources of MPs contamination in this ecological area, and may results from atmospheric deposition of fibers [27]. Differences in MP colors may reflect varying sources, with the proportion of colored MPs being significantly higher in industrial and commercial/residential areas than in the ecological area (Mann–Whitney *U* test, $p < 0.05$).

The diversity of polymer types also varied, with the commercial/residential area containing the most types (33), followed by the industrial area (26), and the ecological area the fewest (14). This indicates more complex and diverse sources of MPs in the commercial/residential and industrial area storm drains. The abundances of PET and PE MPs were significantly higher in commercial/residential and industrial areas than in the ecological area (Mann–Whitney *U* test, $p < 0.05$). PET is commonly used in textiles such as clothing, carpets, winter apparel, and woolen products, as well as containers and packaging materials [28]. The high content of PET in the industrial area, particularly at site S5, might be attributed to the presence of multiple laundry facilities within its catchment, which could release substantial amounts of PET fibers during operation. Conversely, at site S3 in the commercial/residential area, despite its large catchment area, the dominance of public infrastructure and low resident population might explain its lower PET content. PE, one of the most common polymers in plastic products, is widely used in packaging and single-use items due to its versatility [18,29]. The presence of PP and PE MPs in storm drains could be related to the improper disposal of plastic waste [28].

#### 3.4.2. Temporal distribution

The wet season in Macao typically spans from April to September, while the dry season extends from October to March of the following year. Rainfall during the wet season accounts for over 90% of the annual total. Some studies have reported higher MP abundance during the wet season compared to the dry season [30,31]. However, other studies have suggested that once rainfall reaches a certain level, it may dilute MPs in stormwater drains. During the dry season, reduced rainfall and minimal flushing may allow MPs to accumulate, potentially leading to increased abundance [32].

Consistent with the findings of Zhang et al. [10] for storm drains in Hong Kong, no statistically significant difference was observed in the overall abundance of MPs between the wet and dry seasons in this study. However, the mean MP abundance was higher during the wet season, although this difference was not statistically significant. Given that only three sampling events were conducted per season at each site, and that MP concentrations in stormwater systems can vary substantially among individual rainfall events [9], the non-significant result may reflect either a true lack of seasonal variation or the limited ability of the current sampling design to detect such variation. The variability observed among samples within the same season may suggests that MP abundance in storm drains is influenced by intense rainfall fluctuations rather than stable seasonal controls.

Analysis of MP characteristics revealed that the proportion of fibrous MPs was significantly higher in the wet season than in the dry season (Mann–Whitney *U* test, $p < 0.05$). This may be attributed not only to the wash off of land-based fibers due to rainfall but also to the fact that the wet season coincides with Macao's tourism peak, where increased visitor numbers may contribute to elevated microfiber release. In terms of size distribution, the proportion of MPs larger than 0.3 mm was also significantly higher in the wet season (Mann–Whitney *U* test, $p < 0.05$). This is likely because rainfall introduces larger, newly transported particles into the drains via surface runoff. Moreover, higher flow velocities during the wet season may reduce the residence time of larger MPs in the drainage system, thereby limiting their fragmentation into smaller particles.

Regarding polymer types, Ethylene-Vinyl Acetate (EVA) and Polyvinyl Alcohol (PVA) being absent in the dry season. Significantly lower quantities of PET were detected in the dry season compared to the wet season, whereas PP and PE were significantly more abundant in the dry season (Mann–Whitney *U* test, $p < 0.05$). Most of the detected PET particles were fibrous, while PP-based MPs were predominantly fragments. Given that different MP shapes exhibit distinct transport behaviors in flowing water [33,34], it can be inferred that the lower flow velocities and weaker hydrodynamic conditions during the dry season may reduce the mobility of fragments, contributing to the observed seasonal variations in polymer abundance.

### 3.5. Relationship between environmental factors and MPs

#### 3.5.1. Relationship between water quality and MPs

The transport and sedimentation of MPs in aquatic environments are influenced by factors such as flow velocity and MP morphology [35], as well as environmental parameters including water temperature, salinity, and pH [36]. Tables S6 and S7 present the water quality and rainfall conditions. Spearman correlation analysis between MP characteristics and stormwater quality parameters (Fig. 4a and Fig. S3) revealed a significant negative correlation between MP abundance and dissolved oxygen (DO) ($p < 0.05$). In contrast, no statistically significant correlations were observed between MP abundance and salinity (S), pH, electrical conductivity (EC), or water temperature (T) (Fig. 5a). Furthermore, certain MP morphological characteristics showed specific correlations with water quality. The abundance of MP fragments had a significant negative correlation with water temperature ($p < 0.001$), while fiber abundance demonstrated a highly significant positive correlation with water temperature ($p < 0.001$) and a significant positive correlation with flow velocity ($p < 0.05$). Regarding size distribution, the abundance of large-sized MPs had a significant positive correlation with water temperature

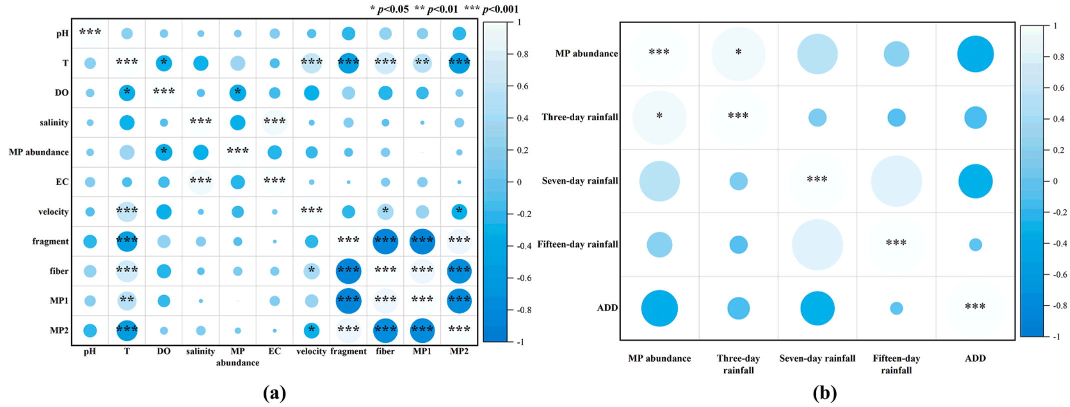


**Fig. 4.** (a) relationship between the water quality of Macao stormwater drains and MP size classes and abundance. MP1 - MPs with a size of 0.5 - 5 mm. MP2 - MPs with a size of less than 0.1 mm, (b) the relationship between rainfall conditions and MP abundance.

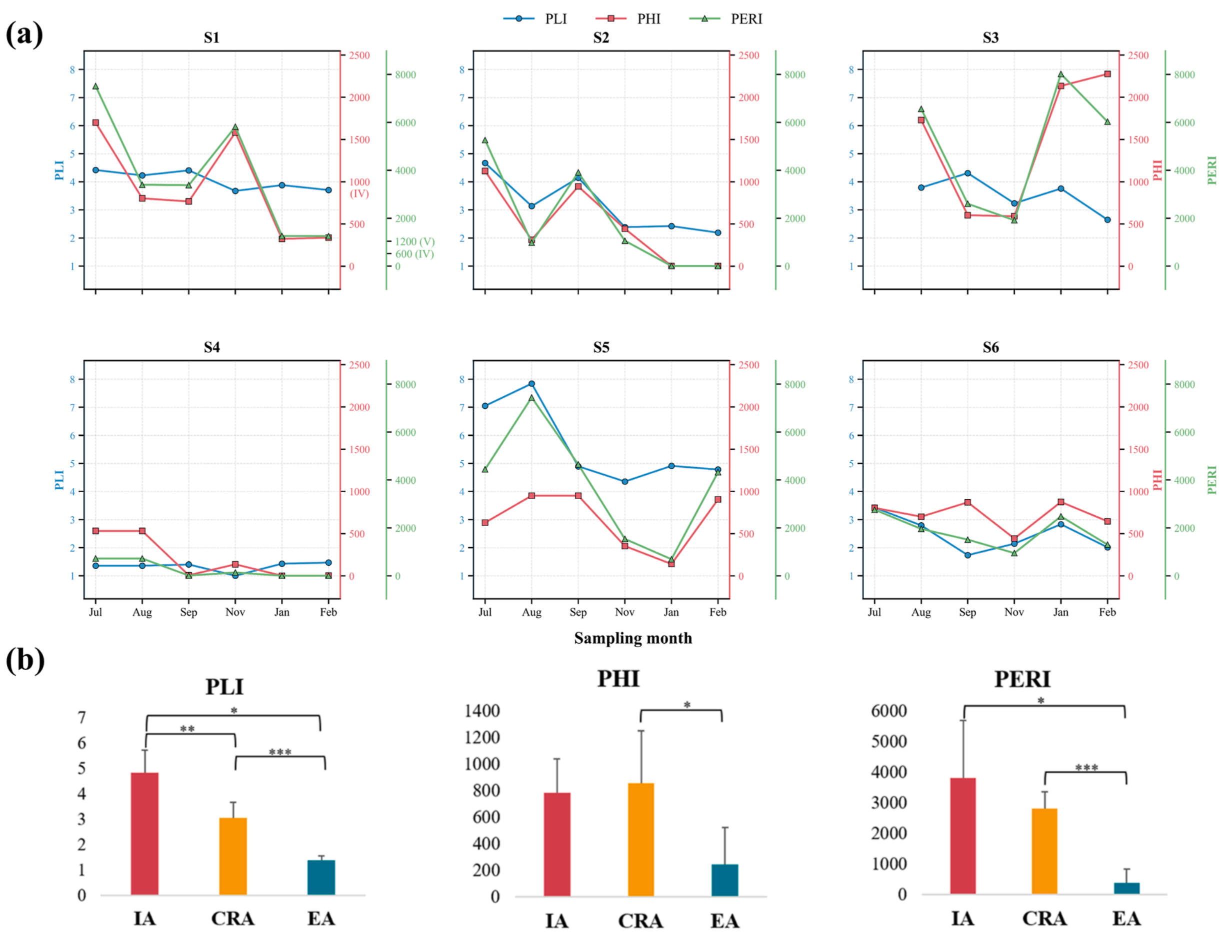


**Fig. 5.** Temporal dynamics and land-use differences of MPs ecological risk indices in Macao stormwater drains. (a) Monthly variations in PLI, PHI, and PERI at each sampling site. (b) Differences in PLI, PHI, and PERI among industrial areas (IA), commercial/residential areas (CRA), and ecological areas (EA).

($p < 0.01$), whereas small-sized MPs showed a highly significant negative correlation with water temperature ($p < 0.001$) and flow velocity ($p < 0.05$). Water temperature had distinct correlations with different MP characteristics. Correlation analysis revealed a significant positive relationship between water temperature and flow velocity ($p < 0.05$), suggesting that the observed correlations between MP morphological features and water temperature are likely driven by seasonal factors. Specifically, during the wet season, heavy rainfall washed terrestrial fibers into stormwater drains and introduced larger MPs. Conversely, during the dry season, lower flow velocities in the drains prolonged the retention time of MPs, increasing the likelihood of their fragmentation into smaller particles and thus elevating the abundance of smaller MPs. Kataoka et al.[37] reported a positive correlation between MP mass concentration and biochemical oxygen demand (BOD) in 29 Japanese rivers. Given that BOD is dynamically and inversely related to DO, this supports the DO depletion observed in the present study when MP loads were high. In agreement with this a study on MP contamination in groundwater of bedrock islands in the South China Sea found that the severity of MP pollution increased with decreasing DO levels [38]. Lestari et al. [39] demonstrated a marked negative correlation between DO and MP abundance in the Progo River.

### 3.5.2. The influence of rainfall conditions on MPs

It is generally recognized that MP abundance is influenced by antecedent dry days (ADD), rainfall intensity, and runoff rate [9,40,41]. Spearman correlation analysis was conducted to examine the relationship between MP abundance and rainfall conditions. The results indicated a significant positive correlation between MP abundance in storm drains in Macao and the cumulative rainfall recorded three days prior to sampling ($p < 0.05$). In contrast, no significant correlations were found between MP abundance in storm drains and the total rainfall over the preceding seven or fifteen days (Fig. 4b). The limited catchment area of the studied storm drains in Macao suggests a short hydrological transport pathway for MPs from land-based entry via runoff to marine discharge. This may explain why rainfall over seven and fifteen days had a minimal influence on MP abundance in stormwater drains.

ADD represents another key factor affecting MPs in stormwater [28]. Previous studies suggested a significant positive correlation exists between ADD and MP abundance in storm drains, implying that longer dry periods allowed more MPs to accumulate on surfaces (Vaze and Chiew, 2002), and then in subsequent rainfall events they were washed into drains. However, in the present study based in Macao, no significant relationship was observed between MP abundance and ADD. This lack of significance may be attributed to the relatively short ADD in the study period, since in previous studies significant correlations between MP abundance and dry periods typically involved much longer (months) antecedent dry periods [42,43].

## 3.6. MPs emission load

This study presents the first mass concentration-based estimation of MP emission load from stormwater drains in Macao. The results indicate that the daily emission loads from six typical stormwater systems in Macao ranged from 2.66 to 10,365.23 g/day (Table 3 and Table S8). The corresponding abundance-based emission loads ranged from $1.12 \times 10^5$ to $3.41 \times 10^8$ particles/day. Compared with previous research on six stormwater drains in Hong Kong, where Zhang et al. [10] reported MP emission loads ranging from $2.24 \times 10^8$ to $3.21 \times 10^9$ particles/day, the loads estimated in the present study were generally lower, indicating substantial variability among urban drainage catchments.

The variation in MP emission load was jointly controlled by MP concentration and hydrological conditions. Although the industrial site S5 recorded relatively high MP abundance and mass concentration, its emission load was markedly lower than that of the other industrial site, S1. This discrepancy was mainly attributable to the smaller cross-sectional area and lower flow velocity at S5, which limited stormwater discharge and consequently reduced the final MP load. Similarly, within commercial/residential areas, S2 generally showed higher emission loads than S3 and S6, likely reflecting differences in catchment characteristics, drainage capacity, and surrounding human activities. S2 drains a relatively densely populated commercial/residential area and is adjacent to a university campus, whereas S3, despite having a larger catchment area, is associated with a lower resident population and lower MP transport intensity.

Seasonal differences were also observed in the estimated emission loads. The wet-season loads were generally higher than the dry-season loads; however, the seasonal difference was not statistically significant for mass-based loads ($t$-test, p = 0.109) and was only marginal for abundance-based loads ($t$-test, p = 0.051). At the site level, wet-season mass loads were higher at S1, S2, S4, and S6, whereas S3 and S5 showed slightly higher dry-season values. For abundance-based loads, all sites except S3 showed higher wet-season values. The pronounced wet-season increases at S4 and S6 suggest that higher flow velocities and greater stormwater discharge during the wet season enhanced the mobilization and transport of MPs accumulated in drainage systems and surrounding catchments. Nevertheless, given the limited number of sampling events and the large variability among sites, the absence of statistically significant seasonal differences should not be interpreted as definitive evidence of no seasonal effect. Instead, it may reflect the limited statistical power of the current sampling design and the high event-to-event variability of stormwater MP loads.

## 3.7. Risk assessment of MP pollution

Currently, there is no unified risk assessment framework for environmental MPs. Due to differences in standards, different models may yield varying risk outcomes [44]. In the load index (PLI) model, the background level of MPs is unclear, so the minimum MP abundance value is used as the baseline ($C_0$) [45,46]. In the risk assessment of MPs in Macao's stormwater drains, the PLI model results showed a $PLI_{zone}$ of 3.11, with $PLI_{zone}$ values for each sampling site ranging from 1.00 to 5.64. This indicates that all sampling sites were within Risk Level I, reflecting a minor pollution level. In contrast, the polymer hazard index (PHI) and potential ecological risk index (PERI) results indicated that MPs in the stormwater drains posed a serious risk. The $PHI_{zone}$ was 754.13,

**Table 3**
Seasonal variation in MP particle and mass emission loads from stormwater drains at different sampling sites.

| Site | Season | n | Particle load mean ± SD (particles day$^{-1}$) | Mass load mean ± SD (g day$^{-1}$) |
|---|---|---|---|---|
| S1 | Wet season | 3 | $(1.44 \pm 0.40) \times 10^8$ | 5462.28 ± 4249.21 |
| | Dry season | 3 | $(1.49 \pm 0.64) \times 10^7$ | 336.86 ± 168.93 |
| S2 | Wet season | 3 | $(1.80 \pm 1.53) \times 10^8$ | 5294.36 ± 3723.40 |
| | Dry season | 3 | $(3.30 \pm 2.00) \times 10^7$ | 543.33 ± 880.90 |
| S3 | Wet season | 2 | $(2.38 \pm 0.43) \times 10^6$ | 99.03 ± 28.20 |
| | Dry season | 3 | $(3.11 \pm 1.39) \times 10^6$ | 139.32 ± 67.11 |
| S4 | Wet season | 3 | $(9.45 \pm 9.16) \times 10^7$ | 372.38 ± 203.43 |
| | Dry season | 3 | $(1.92 \pm 1.00) \times 10^5$ | 6.79 ± 5.76 |
| S5 | Wet season | 3 | $(9.22 \pm 5.90) \times 10^5$ | 28.26 ± 16.10 |
| | Dry season | 3 | $(5.65 \pm 2.98) \times 10^5$ | 29.22 ± 20.38 |
| S6 | Wet season | 3 | $(3.70 \pm 2.86) \times 10^7$ | 1324.52 ± 854.61 |
| | Dry season | 3 | $(1.63 \pm 1.30) \times 10^6$ | 87.44 ± 80.83 |

classified as Risk Level III. The PERI model categorized the MPs in Macao as Risk Level V, with a $PERI_{zone}$ of 1981.28, indicating an extremely hazardous risk level.

Temporal dynamics and land-use differences of MPs ecological risk indices in Macao stormwater drains were shown in Fig. 5. The risk was higher during the wet season, with both PHI and PERI being significantly higher than in the dry season ($p < 0.05$). Spatial differences were even more pronounced, with significant variations in MP risks among stormwater drains in different land use types. The PLI, PHI, and PERI values in industrial and commercial/residential areas were significantly higher than those in the ecological area. The PLI in the industrial area was significantly higher than in the commercial/residential areas, but the PHI and PERI between these two areas did not differ significantly. It is worth noting that S3 located in the stormwater pump station of the commercial/residential area was the only one where PHI and PERI were higher in the dry season than in the wet season. However, the abundance at S3 did not increase significantly during the dry season. Therefore, the increased risk may be related to the higher proportion of polyvinyl chloride (PVC) detected at S3 during the dry season. PVC is widely used in building materials and has a high hazard score of 10,551 [12]. As a stormwater pumping station, the proportion of PVC at S3 increased from 0% in the wet season to between 5.56% and 13.3% in the dry season. This increase was likely due to the release of microplastics from aging pipes and other equipment within the pumping station, leading to the accumulation of these high-risk MPs. Therefore, based on the results of the study it is recommended that future efforts should focus on mitigating the MP risks associated with PVC-based stormwater pipes and pumping stations.

## 4. Conclusion

This study provides a comprehensive assessment of MP pollution in the stormwater system of a subtropical city by integrating MP abundance, polymer composition, mass concentration, emission load estimation, and ecological risk assessment. This integrated approach provides a more complete understanding of stormwater MP pollution and its potential environmental implications. The results confirm that urban stormwater runoff was a significant vector for transporting MPs, with characteristics and concentrations strongly influenced by land use and seasonal hydrology.

A key advance of this study is the estimation of MP mass concentration and emission load from stormwater drains in Macao. Although the average MP abundance and mass concentration in stormwater drains was higher in the wet season than in the dry season, this seasonal difference was not significant. Land use type was one of the important factors affecting the abundance and characteristics of MPs in stormwater drains. The abundance of MPs in stormwater drains with different land use types had a distinct gradient. From the morphological characteristics of MPs, fragment ($56.78 \pm 0.13.80\%$) and black ($38.44 \pm 7.47\%$) were the dominant shape and color, with 20–100 μm ($32.79 \pm 10.40\%$) as the prevalent size. The predominant polymers identified were PET ($39.55 \pm 16.63\%$), PP ($23.05 \pm 18.15\%$), and PE ($9.52 \pm 8.87\%$).

Risk assessments using various models indicated that MPs in stormwater drains of Macao pose a serious pollution risk, with the PHI and PERI significantly higher than the PLI due to the content of high-risk polymers such as PVC. Stormwater drains exhibited a notable daily load of MPs and mass concentrations and although the ecological area had low MP abundance, its considerable flow rate resulted in a noteworthy emission load into the ocean.

Rainfall conditions were another key factor influencing MP pollution in the stormwater drains, a significant positive correlation was found with the cumulative rainfall from the three days preceding sampling ($p < 0.05$). However, likely due to the relatively short ADD during the study period, no significant correlation was observed between ADD and MP pollution. Additionally, MP abundance in stormwater drains was significantly and negatively correlated with DO ($p < 0.05$), while the correlations between MP characteristics and both water temperature and flow velocity further supported the influence of seasonal rainfall flushing on MP pollution. The case of the stormwater pump station highlighted how infrastructure itself is a potential source of high-risk MPs like PVC. Therefore, management strategies should consider the material composition of urban drainage infrastructures to reduce the risk posed by MPs in urban stormwater.

Overall, this study shows that stormwater drains are important pathways for MP transport in subtropical coastal cities. The results also demonstrate that mass concentration and emission load estimation can provide useful support for evaluating MP discharge from urban stormwater systems into coastal waters. The research results also emphasize that when managing urban microplastic pollution, attention needs to be paid to runoff control for specific land uses, monitoring of rainfall events, and the selection of materials for drainage infrastructure.

## CRediT authorship contribution statement

**Aolei Du:** Writing – original draft, Methodology, Investigation, Formal analysis, Data curation. **Youxiang Xu:** Investigation. **Yutang Luo:** Investigation. **Qiying Jian:** Investigation. **Yanjun Li:** Investigation. **Ziying Zhan:** Investigation. **Sizhe Qiu:** Investigation. **Chon Wa Kam:** Resources, Investigation. **Tin-Chak Chan:** Investigation. **Tsz-Yin Huang:** Investigation. **Debora M. Power:** Writing – review & editing. **Kai Zhang:** Writing – review & editing, Supervision, Methodology, Conceptualization.

## Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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## Appendix A. Supplementary data

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