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Microplastic pollution and risk assessment in surface water and sediments of the Zandvlei Catchment and estuary, Cape Town, South Africa☆

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ABSTRACT

Microplastic (MP) (plastic <5 mm) pollution in South Africa is widespread but few studies have been done in catchments and estuaries of the country. The aim of this study was to investigate the abundance, characteristics and risks posed by microplastics in the Zandvlei Catchment and Estuary in Cape Town, South Africa. Water and sediment were sampled between 2019 and 2021, during wet and dry seasons, MPs extracted and identified using microscopy and fourier-transformed infrared spectrophotometry (FTIR) analyses. MP abundances were 70.23 ± 7.36 (standard error) MPs/Kg dw in sediment and 2.62 ± 0.41 MP/L in water samples for the study period. Lower reaches of the catchment and upper reaches of the estuary can be considered sinks for MP contamination as these sites recorded higher MP abundances. MPs were mainly transparent fibres smaller than 0.5 mm. Polyethylene (46%) followed by polypropylene (16%) fibres were the most common polymers recorded. Pollution load indices in MPs were categorised as dangerous in both water and sediment. MP polymer risk indices ranged from moderate in catchment sediment to very high in catchment water. Pollution risk indices were categories as dangerous in water (catchment and sediment) and sediment estuary but low in catchment sediment. Ecological risk assessments hence indicated that polymers in water and sediment were mostly dangerous and poses a threat to the ecological health of both the catchment and estuary studied.

1. Introduction

Microplastic (MP) (particles between 1 μm and 5 mm in size) pollution has become a global concern as it is found in almost every ecosystem globally (Ivar Do Sul and Costa, 2014; Thompson et al., 2004). This is as result of the increased global demand for plastics (Jambeck et al., 2018) due to its versatile use in various applications of daily life (Andrady, 2011; Thompson et al., 2004). MPs are categorised based on their sources. Primary MPs are manufactured to be used in cosmetic products or abrasion applications (microbeads) or as raw materials used to produce plastic materials (nurdles). Secondary MPs result from the fragmentation and degradation of plastics due physical, chemical, photo- and microbial degradation (Barnes et al., 2009; Thompson et al., 2009). MPs may end up in rivers (Wang et al., 2021), estuaries (Gholizadeh and Cera, 2022) and other receiving bodies, thus causing environmental and health risks (Vilakati et al., 2020; Wright et al., 2013). MPs are comprised of polymers and monomers, including polyethylene, polypropylene, polyester and polystyrene that are chemical hazards (Li et al., 2021). The environmental risks of MPs are due to these particles being mistaken as prey and ingested by organisms (Zhu et al., 2018), with detrimental subsequent biological effects such as gut blockage and intestinal injury (Qiao et al., 2019), oxidative damage and reduced energy production (Silva et al., 2020) and impaired physiological functioning, growth and reproduction (Jiang et al., 2022).

The state of waste management in South Africa is considered to be poor, with significant leakages of plastic debris to the environment, largely because of inadequate waste collection and disposal (Arabi et al., 2020). The coastal ecosystems of South Africa support a high degree of biodiversity and can be regarded as moderately pristine (Wepener and Degger, 2012). However, the rapid increase of coastal population densities, urbanization and industrialization are placing strains on the environment and its resources. South Africa has an extensive plastics manufacturing industry, but recycling is restricted and inadequate, with a large fraction of unregulated waste entering the environment (Verster et al., 2020). The unregulated waste (including MPs) found in lentic (lakes and wetlands) environments have a longer residency time because

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of low circulation whereas in lotic systems (rivers), MPs would be dispersed easier and more widely, ultimately ending up in the marine environment (Blankson et al., 2022).

Amongst the first plastic pollution studies in South Africa was an investigation of the characteristics and abundances of open ocean litter sampled between 1977 and 1978, where 3640 particles per km^2 (mainly fragments and foam) were recorded (Ryan, 1988). Amongst the first studies relating to MPs was research on the presence of polyethylene pellets in coastal birds (Ryan, 1987). Research on MPs in South Africa have increased over the past few years, mainly focusing on the coastal environment (de Villiers, 2018; Naidoo and Glassom, 2019; Nel and Froneman, 2018, 2015; Ryan, 2020), with relatively fewer studies on MPs in estuaries (Boshoff et al., 2023; Govender et al., 2020; Naidoo et al., 2015). Given the few investigations on MPs in estuaries and none on MPs in estuaries and catchment areas in South Africa, the aim of the study was the measure the abundances, characteristics and ecological risks of MPs in the Zandvlei Estuary and catchment situated in Cape Town, South Africa.

2. Materials and methods

2.1. Study area

Zandvlei Estuary (and its catchment) is situated in Cape Town, South Africa, at the north-western shore of False Bay (Fig. 1). The estuary is a shallow coastal wetland, spanning 2.6 × 0.5 km that has been modified to maintain water levels in a marina (Quick and Harding, 1994). As it is ecologically significant in terms biodiversity, the estuary is ranked in the top 25% of ecologically important estuaries in South Africa (Hawley, 2017). Zandvlei Estuary provides a range of ecologically services, including nurseries for juvenile fish, feeding sites for resident and migratory birds, and biological filtration (McQuaid and Griffiths, 2014). Zandvlei is used for aesthetics, sports and cultural practices and is affected by urban development (McQuaid and Griffiths, 2014), invasive alien plants and dredging (River Health Programme, 2005). The mouth is opened artificially by the City of Cape Town's Catchment Management Department when a high spring tide is expected to improve water quality in the estuary (City of Cape Town, 2018). Zandvlei Estuary's catchment is approximately 92 km^2 and mainly comprises residential areas, few commercial zones and farming areas (Taljaard et al., 2000). Due to the Mediterranean climate, the seasonal streams flow in winter after the rainfall and often stops flowing in dry summer months. The key streams which drain into the estuary are the Westlake Stream, Keysers River, Langvlei Canal and the lower end of the Diep River, known as the Sand River Canal (Quick and Harding, 1994).

2.2. Sample collection

Samples were collected the Zandvlei Estuary and catchment areas during the wet (winter) season in 2019 and dry (summer) season in 2020/2021. The reaches of each area were categorised into upper (catchment site 1; estuary site 6–7), middle (catchment sites 2–3; estuary sites 8–9) and lower (catchment sites 4–5; estuary site 10) reaches (see Table S-1).

2.3. Water and sediment sample collection

Surface waters were sampled from the Zandvlei Estuary and catchment areas by collecting five replicates per site using a 20 L bucket (100 L per site). Surface water was collected approximately 1–2 m offshore of the shoreline. The water was poured through a 250 μm sieve, the retained particles rinsed with deionized water and the concentrates

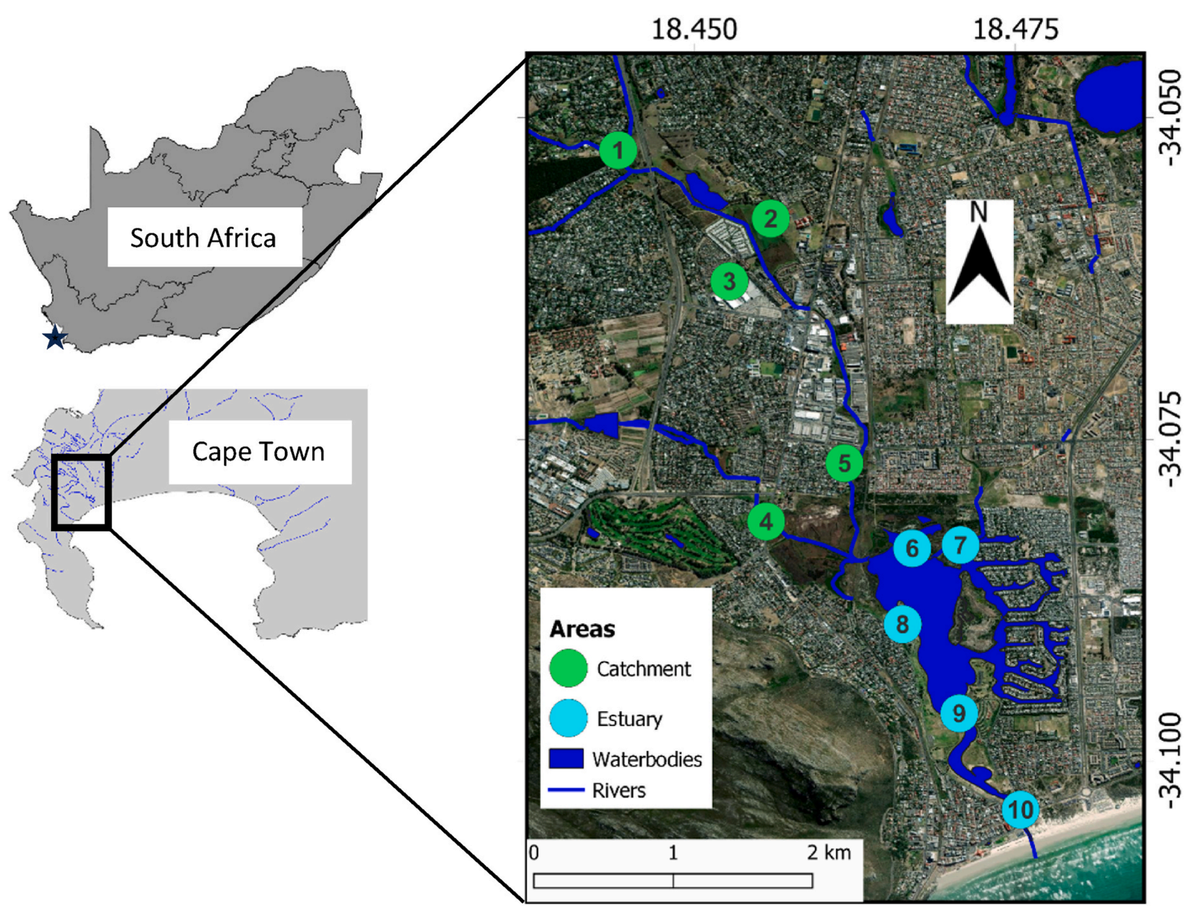

Fig. 1. Estuary and catchment sites sampled in Zandvlei, Cape Town, South Africa. Sites 1–5 were in the Zandvlei catchment and sites 6–10 in Zandvlei Estuary.

added to a falcon tube. Falcon tubes were stored in a freezer at −20 °C until processing. Sediment samples were within 1 m of the landward side of the shoreline and collected from the top 5 cm using a metal spatula. Sediment samples were collected in five 0.5 m × 0.5 m quadrats per site (Hidalgo-Ruz et al., 2012) and stored in pre-cleaned plastic bags.

2.4. Laboratory analyses: digestion and extraction

Prior to processing, surface water samples were removed from the freezer and defrosted at room temperature (±25 °C) (GESAMP, 2019). A KOH solution (10% w/v) was added to the sample (1:2 ratio) based on methods described previously (Su et al., 2016) and placed in a drying oven at 60 °C for 24 h. The sample was then manually agitated for 2 min and vacuum-filtered (model Rotary Vane VP 145 1/3 HP) onto pre-cleaned 20 μm nylon filters. Each filter was placed into a pre-cleaned (acid washed) Petri dish and allowed to dry before microscopic analysis.

Sediment samples were decanted into aluminium containers and covered with aluminium foil (GESAMP, 2019). Each aluminium container had a sample of 500 g and split into containers A and B. Sediment of containers A were used for grain size analysis and containers B for MP analysis. Sediment samples for MP analyses were placed into a drying oven (model DHG 9070A) 60 °C for 48 h, or until constant mass (Naidoo et al., 2015). Each of the sediment A samples were processed for grain size analysis using universal test sieves on a shaker. After shaking samples for 5 min, sediment in each sieve were weighed at intervals according to the Wentworth scale: 500, 250, 125, 63 and < 63 μm.

To remove biological material, 500 ml of 10% KOH (w:v) solution was added to sediment B samples (200 g) and placed in an oven for 48 h at 60 °C. Density separation was used to extract MPs from the sediment based on methods commonly reported in the literature (GESAMP, 2019; Hidalgo-Ruz et al., 2012). A fully saturated NaCl solution was prepared by adding 360 g NaCl to a beaker containing 1 L of MilliQ water. The hypersaline solution was mixed using a magnetic stirrer at a high speed at room temperature (25 °C) for 10 min or until the solution became saturated. Commercially available table salt have been found to contain MPs (Karami et al., 2017), therefore the NaCl solution was vacuum-filtered through a 10 μm mesh. The salt-sediment mixture was left to settle for 15 min, the supernatant filtered onto a 20 μm mesh and placed into pre-cleaned petri dishes. The extraction procedure was repeated three times per sediment sample to improve MP recovery rates. The petri dishes were stored until microscopic analyses.

2.5. Microscopic identification of MPs

Individual meshes containing water and sediment samples were visually examined under a dissecting microscope (Olympus SC30) equipped with a camera (DinoCapture Camera V1.5.39.C). MPs were identified based on the following characteristics: shape, color and size (Hidalgo-Ruz et al., 2012). MP shapes were categorised as: fibre, fragment, sphere, film and foam, size classes as: <0.5, 0.5–1, 1–2 and 2–5 mm (using 1 mm graph paper) and color as: white, black/grey, blue/-green, red/orange, yellow/brown and transparent.

2.6. Polymer identification using spectrophotometry

Putative plastic particles larger than 0.5 mm were removed from meshes using fine tweezers, for validation using a PerkinElmer Two FTIR-ATR Spectrophotometer. The MPs were compressed at a force of at least 80 N. The spectra were recorded in the wavenumber ranging from 4000 to 450 cm^{-1} with a resolution set to 4 cm^{-1}, a data interval of 1 cm^{-1} and scans set to 10. Before each particle was analysed, the ATR base was cleaned with 70% isopropanol and background scans done prior to each sample analysis. The ST Japan Polymer and Polymer Additives Library (supplied by Perkin) was used to verify polymer types and scans 70% and above were accepted. Abbreviations for polymers recorded are as follows: PA: polyamide; PE: polyethylene; PES: polyester; PMMA: polymethyl methacrylate; PP: polypropylene; PS: polystyrene; PU: polyurethane; PVA: polyvinyl acetate; PVC: polyvinyl chlorinate.

2.7. Ecological risk assessment

MPs indices were applied to provide comparative assessments of the potential effects of MPs, with risk categories presented in Table S-2. The MPs contamination factor (MPCF) assesses the abundance of MPs (C_{MP}) compared to background abundances using the following equation:

$$MPCF_i = \left(\frac{C_{MP}}{C_{baseline}}\right)$$

where the $C_{baseline}$ values selected were the average MP abundances in water and sediment for site 1 (control site) as there are no historic values for the region and this method is considered acceptable (Kabir et al., 2021). MP pollution index (MPPLI) calculations were using the following equation.

$$MPPLI_{site} = \sqrt[2]{MPCFrXMPCFi}$$

where MPCFr and MPCFi were for fragments and filaments, respectively. The chemical toxicity of polymers were analysed based on the method by (Lithner et al., 2011), where hazard scores are assigned to polymer types to assess the risk of polymers using the following equation:

$$H_i = \sum P_n \times S_n$$

where H_i is the calculated polymer risk index, P_n the ratio of a polymer type recorded at a site and S_n the polymer hazard score. The S_n scores for polyethylene (PE), polyester (PES), polymethyl methacrylate (PMMA), polypropylene (PP), polystyrene (PS) and polyvinyl chlorinate (PVC) were 11, 4, 1021, 1, 2788, 10599, respectively (Lithner et al., 2011). The pollution risk index (PRI) is calculated as follows:

$$PRI_i = \sum H_i \times MPPLI_{site}$$

where PRI_i indicates the ecological hazard of polymers when associated with the polymer risk index (H_i).

2.8. Quality control

Procedural blanks and contamination controls during the sampling and laboratory processing were used to minimise including potential MP contaminants. As far as possible, we tried to reduce the use of plastics and pre-cleaned containers before use by acid washing and rinsing with MilliQ ultra-pure water. In the field, samples were collected by standing downwind. In the lab, pre-clean petri dishes with filter paper were placed in workspaces and left exposed during filtration and microscope work, to determine if there were any airborne contamination. Procedural blanks for foil, falcon tubes and sediment bags, (n = 12), were taken by rinsing these items with MilliQ ultra-pure water and filtering it through a 20 μm mesh. Protocols observed in the laboratory included cleaning workbenches with ethanol before processing samples, wearing cotton lab coats, keeping doors and vents shut, using metal equipment (scoops, tweezers), covering samples with aluminium foil and opening samples only when strictly necessary (Prata et al., 2020). MilliQ water was used to make up all solutions used i.e hypersaline solutions and KOH solutions. Petri dishes were kept closed and only opened when being processed under the microscope to record MPs. Extraction efficiencies were done by filtering known abundances of fibres and fragments ranging from 0.5 to 5 mm.

Tests for contamination resulted in a total of 58 fibres recorded for the duration of the study and the data adjusted accordingly, based on where the contaminants were found. Blanks were included in all sample filtrations and no MP contamination reported. Extraction efficiencies were done by filtering known quantities of filaments and fragments. Extraction efficiencies for fibres recorded recovery rates of 83–85% for MPs between 250 and 5000 μm in size and 91–94% for fragments of the same size categories.

2.9. Data analysis

MP water data were recorded as MPs per litre (MP/L) and sediment abundances as MP per kg dry weight (MPs/Kg dw). Univariate statistics were conducted using IBM SPSS Statistics® (Version 28 for Microsoft® Windows® 10). Statistical tests for normality was done using the Kolmogorov-Smirnov test and variance using the Levene's test. Even after the data were log-transformed, it did not meet the assumptions to do parametric analyses. Non-parametric tests were subsequently done using the Kruskall Wallis (KW) test for multiple groups and the Mann Whitney (MW) test between two groups (eg. catchment vs estuary). Spearman rank correlations (r) were used to assess relationships between groups. Results are reported as means, variances as standard error of the mean (±SEM) and significances set at $p < 0.05$.

3. Results and discussion

3.1. MP abundances and distribution

MPs were recorded at all sampling stations of the estuary and catchment, but no MPs were recorded in water samples at site 1 during the dry season. A total of 360 samples were processed and 15273 MPs recorded. Based on FTIR polymer scans, 4.5% of putative particles (>0.5 mm) were identified as natural (fibres) and the final data values adjusted by 4.5% to account for natural fibres. The mean MP abundances recorded for all sites and seasons were 2.62 ± 0.41 standard error of the mean (SEM) MPs/L in water (n = 170) and 70.23 ± 7.36 MP/kg dry weight (dw) in sediment (n = 190).

MP abundances in water samples were significantly lower (MW = 4147, $p < 0.05$) in the catchment area (0.57 ± 0.09 MP/L) than estuary (4.12 ± 0.65 MP/L) during the wet season (Fig. 2a). Seasonally, MPs in water were significantly lower (MW = 5067, $p < 0.01$) in the dry season (0.43 ± 0.05 MP/L) than wet season (5.25 ± 0.78 MP/L), with MPs in the estuary increasing in the upper (site 6) to lower reaches (site 10) of the estuary. MPs extracted from sediment were consistently low in both the catchment and estuary during the dry season (30.24 ± 3.75 MP/kg dw) (Fig. 2b). During the wet season, sediment MPs increased from site 1 to site 5 (catchment), with an opposite trend recorded in the estuary, decreasing from site 6 (upper reaches) to the mouth (site 10). Sediment MPs differed significantly between seasons (MW = 6490, $p < 0.05$) (dry: 30.24 ± 3.75; wet: 103.56 ± 13.41 MP/kg dw), but not between areas (MW = 4557, p = 0.22) (catchment: 64.63 ± 11.94; estuary: 73.85 ± 7.36 MP/kg dw).

The increase in water MP abundances at the lower reaches of the estuary and sediment MPs at the lower reaches of the catchment during the wet season indicates that the wet season influences the movement and accumulation of MPs in catchment-estuarine systems. Similar patterns were recorded in KwaZulu Natal, South Africa (Naidoo et al., 2015), where higher MP abundances were reported in water close to the mouth and MPs in sediment were highest in the upper reaches of the estuary. Elevated water MP abundances closest to the mouth during the wet season (Fig. 2a) may be due to the influence of increased water flow, transporting MPs from the catchment to the estuary, as well bioturbation of sediment in the estuary during the wet season. Bioturbation has the potential to return buried or settled MPs in the sediment, to the sediment-water interface, resulting in the resuspension of MPs in the water column (Näkki et al., 2019, 2017). The increase in sediment MPs in the lower reaches of the catchment areas during the wet season (Fig. 2b) have been reported elsewhere (Gholizadeh and Cera, 2022), indicating that lower reaches of catchment systems may be areas of sinks for MPs (Eibes and Gabel, 2022). Seasonal variations of wind movement in the catchment and estuary may affect MP distribution. During the wet season, north-westerly winds are strongest in Cape Town (Taljaard et al., 2000), which could driving MP movements from the catchment, through the estuary and towards the open sea (Fig. 1). The increased rainfall also allows MPs from the catchment to be flushed through the catchment system into the estuary.

3.2. MP shape, color and size

MP fibres were the dominant shape recorded in water samples. MPs extracted from water samples were lowest at site 2 (catchment) and increased towards the lower reaches of the catchment area (site 5) during both the wet and dry seasons (Fig. 3a). During both the wet and season, water MP proportions did not show any patterns from the upper reaches to the mouth of the estuary. During the dry season, highest proportions of fragments (60%) were recorded at site 2, which decreased towards the site 5. During the wet season, higher proportions of film were recorded at site 1 (17%) and site 2 (49%), with the remainder of the sites dominated by fibres (sites 3 to 10). Like MPs in water, MP fibres were the dominant shape recorded in sediment (Fig. 3b). During the dry season, fibres from the catchment area were lowest at site 2 (61%) and increased towards the lower reaches of the catchment at site 5 (80%). In the dry season, fibre proportions decreased from the upper reaches of the estuary (96% at site 6) towards the mouth (33% at site 10). During the wet season, the pattern of MP sediment proportionality for shapes differed than the dry season, with fibres comprising the dominant shape for all catchment sites, ranging from 95% at site 3–99% at site 5. MP shape in the estuary during the wet season were lowest at the upper (57%) and lower reaches (54%), and highest in the middle reaches of the estuary (81% at site 8).

The increasing abundances of fibres from site 1 to 10 may be related

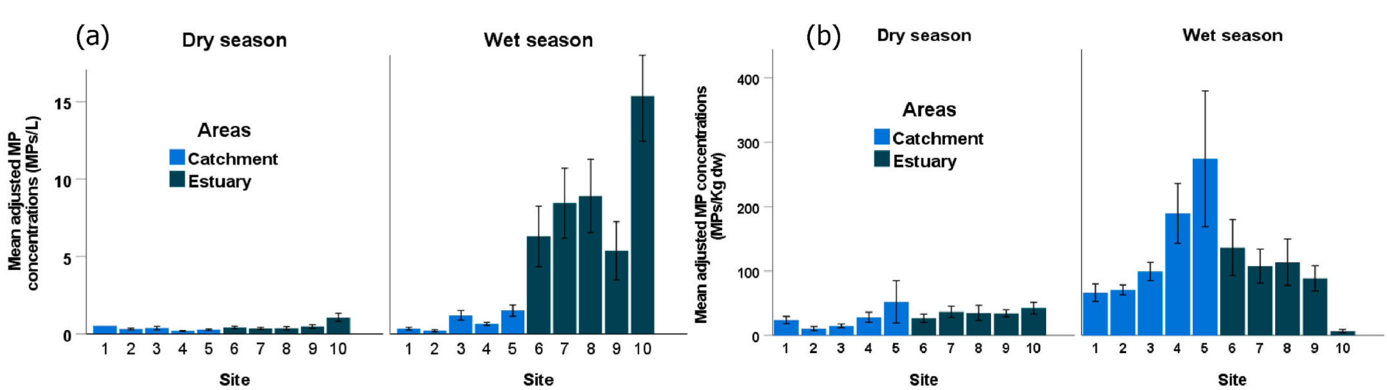

Fig. 2. MP abundances (mean ± SEM) in (a) water (MPs/L) (n = 5 per site) and (b) sediment (MPs/Kg dw) (n = 5 per site) sampled in summer (dry season) and winter (wet season). Sites 1–5 represent catchment area and 6–10 estuary sites.

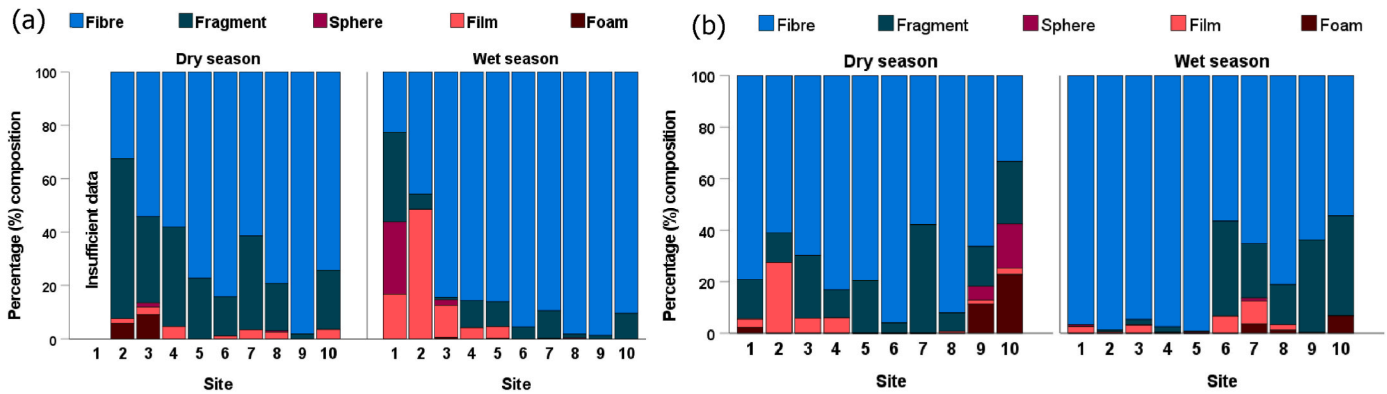

Fig. 3. Percentage composition of MP shape (fibre, fragment, sphere, film and foam) in (a) water and (b) sediment samples collected in wet and dry seasons as well as the catchment (sites 1–5) and estuary (sites 6–10).

to the proximity of residential areas where stormwater drainage systems may be conduits of laundry wastewater. Informal human settlements and residential areas in the catchment area and adjacent to the estuary may result in the release of synthetic textiles into the system due to the washing of clothes (de Villiers, 2019). Zandvlei Estuary is also an area well-known for tourism and recreational fishing, with commercial fishing taking place in False Bay. Illegal dumping, inadequate disposal of waste such as cigarettes and broken fishing gear i.e. wrapping bands and fish nets are primary sources of high fibre abundances (Wang et al., 2018).

MPs were categorised based on 6 colors. In water samples, the proportion of white MPs decreased from site 1 to site 10, with transparent MPs having the opposite trend, increasing from site 1 to 10 (Fig. 4a). Transparent MPs were the dominant MPs colors recorded in water samples during the wet season, similar to that reported in the Plankenberg river within the same province as our sampling sites (Apetogbor et al., 2023). Transparent MPs were also the dominant MP colour in sediment samples during the dry season (~50%) (Fig. 4b). Transparent MPs were higher in sediment of the catchment area (~61%) compared to the MPs sampled in the estuary (~29%), except at site 10, where no transparent MPs were recorded in sediment, where blue/green (57%) and yellow/brown (41%) were the dominant MP colors recorded.

MP were mainly smaller than 0.5 mm in both water (95%) and sediment (94%) samples (Fig. 5). More MPs greater than 0.5 mm were recorded in water from the catchment area during the wet season (12% in total all size categories) (Fig. 5a) and sediment samples in the estuary during the dry season (12% in total all size categories) (Fig. 5b). Correlations between MP abundances and various grain sizes indicate that MPs were negatively correlated to the smallest sediment processed (<0.063 mm) for all sites analysed (Table 1), with most significant correlations recorded in the estuary during the wet season.

3.3. Polymer composition

A total of 180 putative particles were selected to identify polymer type using FTIR analyses (Fig. S-1). Of these, 4.5% of particles were identified as natural particles (silk, wool and jute) (Fig. 6a) and were subsequently excluded from MPs data analyses. Proportion of polymer types differed between water (Fig. 6b) and sediment (Fig. 6c) samples. Polymers in water samples were mainly PE (33%) and PP (33%) in the catchment, with PE (43%) mainly recorded in the estuary of the dry season. During the wet season, polymers were less diverse, comprising 50% PS and 50% PVC in the catchment and 100% PE in the estuary. The proportion of polymers were less diverse in sediment samples than water samples (Fig. 6c). Polyethylene were the main polymer types recorded in the dry season (100% in catchment and 40% in the estuary). Similarly, PE were the main proportions in sediment of the wet season, comprising 67% of polymer types in the catchment and 82% in the estuary.

Polyethylene was the most prevalent polymer types across all seasons and areas sampled. Although PE is versatile, inexpensive, has a wide range of applications and the main types of plastic litter in South Africa (Ryan and Moloney, 1993), it is evident from this study that PE fibres are by far the most abundant MPs at the sites sampled. It is hence postulated that domestic laundry effluent is the main source of MP fibres in the Zandvlei Estuary and catchment. Research on MP fibres has reported that a 6 kg wash load can release 700,000 fibres during each laundry cycle (Napper and Thompson, 2016) and 5 kg of PE fabric can release up to 6 million fibres during a laundry cycle (De Falco et al.,

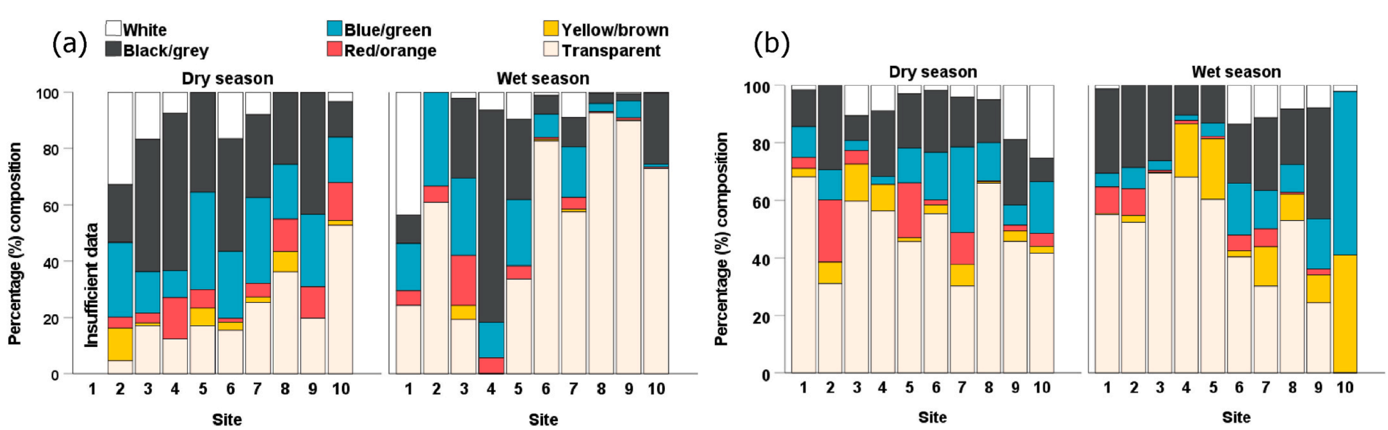

Fig. 4. Percentage composition of MP color in (a) water and (b) sediment samples collected in wet and dry seasons as well as the catchment (sites 1–5) and estuary (sites 6–10). (For interpretation of the references to color in this figure legend, the reader is referred to the Web version of this article.)

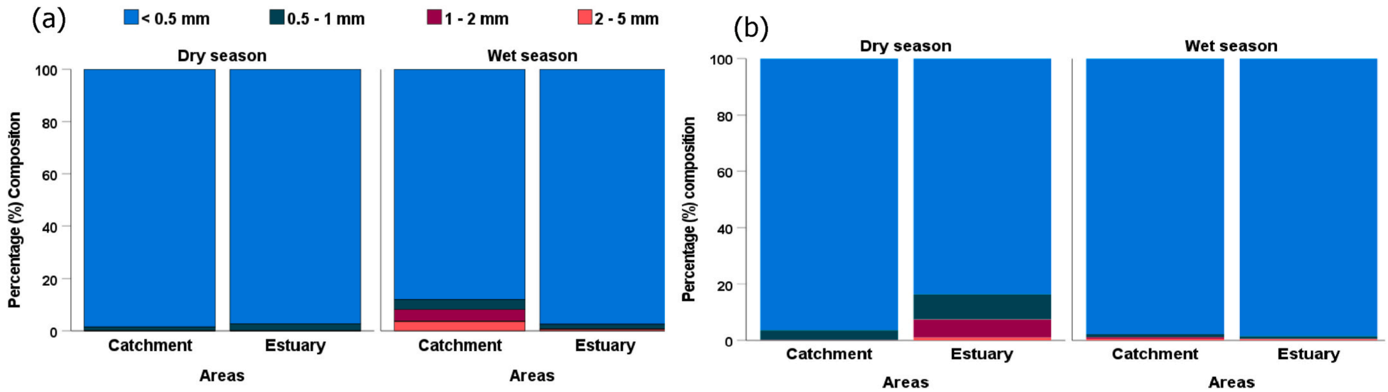

Fig. 5. Percentage composition of MP size categories in (a) water and (b) sediment samples collected in wet and dry seasons as well as the catchment (sites 1–5) and estuary (sites 6–10).

Table 1
Spearman rank correlations between adjusted MP abundances and percentage sediment grain size category (coarse sand: >0.5 mm; medium sand: 0.5–0.25 mm; fine sand: 0.25–0.125; coarse silt: 0.125–0.063 mm; fine silt: <0.063) and based on area (catchment and estuary) and season (wet and dry).

Area	Season	Coarse sand	Medium sand	Fine sand	Coarse silt	Fine silt
Catchment	Dry	0.31[a]	0.18	0.18	0.30[a]	−0.31[a]
Catchment	Wet	0.08	−0.35	0.10	0.08	−0.34
Estuary	Dry	0.06	0.13	0.40**	0.22	−0.25
Estuary	Wet	0.69**	0.25[a]	0.61**	0.57**	−0.65**

[a] Correlation is significant at 0.05; ** Correlation is highly significant at 0.01.

2018). In South Africa there is a positive correlation between river sediment MPs and households in catchment areas that do not have access to piped water (de Villiers, 2019). The increase in PE fibre abundances in water from catchment to estuary sites (Fig. 3a) indicates that the accumulation of PE fibres is likely related to rivers being vectors of MP fibre transport. This is particularly evident during the wet season where PE fibres accumulate in estuary sediment (Fig. 3b) and may potentially be an ecological hazard for organisms in the Zandvlei estuary.

3.4. *Risk assessment of polymers*

More than 50% of plastics are categorised as hazardous based on a hazard-ranking system of the United Nations' Globally Harmonized System of Classification and Labelling of Chemicals (Lithner et al., 2011). MPs in the aquatic environment are consumed by most aquatic organisms, including freshwater and estuarine invertebrates (Silva et al., 2021) and fish (Arias et al., 2019). Given that MPs are known to bioaccumulate in ecosystems (Goswami et al., 2020), the impacts of MPs have the potential to negatively affect food security and human health as well (De-la-Torre, 2020). Knowledge about the risks posed by MPs focuses on experiments that may expose organisms to abundances that are not environmentally relevant or do not mimic the natural environment (exposure to microbeads when fibres are the most prominent MPs in the environment) (Koelmans et al., 2022).

Based on the chemical composition of plastics and the chemical hazards of additives, monomers and polymers, a hazard ranking system was developed to assess the hazard effects of polymers (Lithner et al., 2011). The polymer hazard ranking system has subsequently adopted to develop an ecological risk assessment system that categorises risks posed by polymers based on a pollution load index (PLI), polymer risk index (H) and pollution risk index (PRI) that provides categories for ecological risks posed by polymers (Pan et al., 2021; Xu et al., 2018).

The PLI of water and sediment in the Zandvei Estuary and catchment were categorised as dangerous (>5) (Table 2). The MP PLI was three times higher in estuarine water than the catchment and four times higher in estuarine sediment than catchment sediment. This result was likely due to MP runoff accumulating in the estuary during the wet season, as evident in higher MP abundances during the wet season (Fig. 3). Wastewater treatment works (WWTWs) are major sources of MP fibres (Murphy et al., 2016). The higher PLI values in the estuary may also be due to WWTW failure, upstream of the Zandvlei Estuary, which occurs due to sewage spills and pump station failures (Matthews, 2023). The moderate to very high polymer risk index (H) values are based on the type of polymers recorded, mainly PE fibres (Fig. 6), which has a low

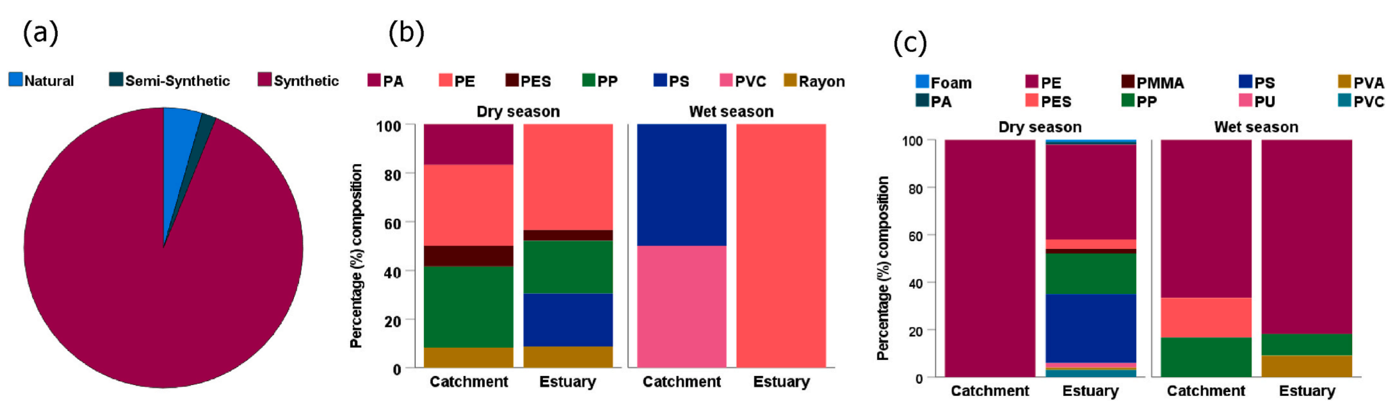

Fig. 6. Percentage composition of MP polymer categories in (a) all samples, (b) MPs in water and (c) MPs in sediment samples collected in wet and dry seasons as well as the catchment (sites 1–5) and estuary (sites 6–10) (PA: polyamide; PE: polyethylene; PES: polyester; PMMA: polymethyl methacrylate; PP: polypropylene; PS: polystyrene; PU: polyurethane; PVA: polyvinyl acetate; PVC: polyvinyl chlorinate).

Table 2
Risk assessment for water and sediment sampled in the catchment and estuary. Categories are colour-coded as low (green), moderate (blue), high (orange), very high (purple) and dangerous (red). See Table 2 for value ranges of categories.

	Catchment			Estuary		
	PLI	H	PRI	PLI	H	PRI
Water	2.93	2212	6213	10.18	537	5469
Sediment	3.79	8.57	2.7	12.21	865	10572

hazard score of 11 (Lithner et al., 2011). The H values (which reflects polymer toxicity) and MP abundances (PLI scores) provides an indication of MP pollution risk (PRI) (Wei et al., 2022).

The low PRI ranking for polymers in catchment sediment is expected, as MPs are not likely to settle in sediment of catchments, especially the upper reaches. However, this result requires further investigation as MP settlement rates are subject to a number of factors, including slope and riverine water flow rates (which were not measured in this study). The dangerous rankings for catchment and estuary water and sediment (see Table 2) poses a serious threat to the Zandvlei Estuary and catchment ecosystems and these results provide a valid method to assess the risks posed by polymers (Shirazi et al., 2023). The predominant MPs recorded in this study were PE fibres (See Figs. 3 and 6), suggesting that MPs from washing of clothes poses the greatest ecological risk for the area sampled. Cape Town has numerous informal human settlements that do not have access to municipal piped water. Wastewater from laundry activities adjacent to rivers and canals of the Zandvlei Catchment and Estuary is the probable cause for the high PRI values recorded.

4. Conclusion

To our knowledge, this is the first study in South Africa to investigate MPs in an estuary and catchment area simultaneously. MP abundances in water samples ranged from 0.57 MP/L in the catchment to 4.12 MP/L in the estuary and in sediment, ranged from 64.63 MP/kg dw in the catchment to 73.85 MP/kg dw in the estuary. The results clearly indicated that MP abundances are higher during the wet season and in the estuary. The predominant MP characteristics recorded in this study were transparent PE fibres. The high abundance of MP fibres were most likely due the high rates of residential laundry waste entering the catchment and upper reaches of the estuary and WWTWs failure. Our results show that MP fibres potentially pose a threat to the health of the catchment of estuary ecosystem, based on the dominance of PE fibres recorded. Hence, we have demonstrated that the rivers entering Zandvlei Estuary are vectors for MP transport and potentially pose an ecological risk to the functioning the Zandvlei Estuary and catchment. The elevated MP abundances reported here provides a baseline for future studies as it is evident that there is a need for investigations to focus on the effects of MPs on catchment and estuarine systems in South Africa.

Credit author statement

Whitney Samuels - Investigation, Sample collection, Data curation & Original draft preparation; Adetunji Awe - Data curation, Writing - Review & Editing; Conrad Sparks - Conceptualization, Supervision, Sample collection, Writing - Review & Editing, Project administration.

Credit author statement

Whitney Samuels - Investigation, Sample collection, Data curation & Original draft. preparation; Adetunji Awe - Data curation, Writing - Review & Editing; Conrad Sparks - Conceptualization, Supervision, Sample collection, Writing - Review & Editing, Project administration.

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.

Data availability

Data will be made available on request.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.envpol.2023.122987.

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