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Microplastics in subsurface coastal waters along the southern coast of Viti Levu in Fiji, South Pacific

Jasha Dehm[a], Shubha Singh[a], Marta Ferreira[a], Susanna Piovano[a,*]

[a] School of Marine Studies, Faculty of Science Technology and Environment, The University of the South Pacific, Laucala Bay Road, Suva, Fiji

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ABSTRACT

Microplastics (MPs) remain largely understudied in Small Island Developing States. This study is aimed at comparing the abundance and characteristics of MPs in rural and urban marine coastal sites located along the southern coast of Fiji's main inhabited island, Viti Levu. Collection of subsurface waters (at depth of ca. 0.6 m) was performed at seven sites via Niskin bottle. Samples were filtered over a membrane filter to extract MPs and to allow visual analysis and polymer identification by using attenuated total reflectance in Fourier transform infrared spectroscopy. Findings from this study depict widespread presence of MPs in both urban and rural sites, and show no significant differences in the four parameters studied, i.e. abundance of MP pieces (2.0 vs 1.6 MP/L, respectively), form types (dominance of fibers), size (0.5–0.9 and 1.0–1.4 mm totaling 48% of the samples), and color (blue contributing 30%, and red and black contributing 25% each). These findings challenge the common expectation of a higher MPs pollution in urban areas compared to rural areas.

1. Introduction

Plastic pollution has been of increasing global concern in the last decades, so much in fact that the United Nations identified it as 'one of the biggest environmental challenges of this lifetime' (Schwarz et al., 2019; United Nation Environment Programme, 2018). The versatility and persistent nature of plastics, which are key drivers of global supply and demand, are also the key aspects that make them such a concern for the environment (Derraik, 2002; Schwarz et al., 2019). Among plastics, the ones with a size ranging 0.01–5 mm are categorized as "microplastics" (MPs) (Frias and Nash, 2019). Depending on origin, MPs are termed as either "primary MPs" when directly manufactured as small pieces of plastics, or "secondary MPs" when they result from weathering of larger pieces (GESAMP, 2016). MPs are often further classified based on form type, with the most widely used classification being pellets, fragments, films, and fibers (Guo and Wang, 2019). Pellets are primary MPs that often enter the marine environment through sewage and wastewater outfalls (Akdogan and Guven, 2019; Guo and Wang, 2019). They include subgroups such as microbeads and nurdles, which are manufactured for use in powders, personal care products, paints and solvents, and a variety of cleaning agents (United Nation Environment Programme, 2015). Fragments and films are secondary MPs that often originate from plastic bags, bottles, food containers and packaging, and vehicle tires (Akdogan and Guven, 2019; Guo and Wang, 2019). They reach the marine environment primarily via littering and urban runoff (Burns and Boxall, 2018). Fibers in the marine environment can have either a primary origin, such as from personal care products (Akdogan and Guven, 2019), or a secondary one, where abrasion from washing of clothes and textiles is often the most common source (Mishra et al., 2019).

While most documented deaths of marine species result from ingestion or entanglement with larger plastic debris (Derraik, 2002; Kühn and van Franeker, 2020), recent literature suggests that MPs have the potential to be harmful to the marine biota (Andrady, 2011; Chae and An, 2018; Silva et al., 2019; Sun et al., 2018). This is due to MPs acting as vectors for biological and chemical contamination (Caruso, 2019), whereby the relatively large surface to volume ratio allows for easy adsorption of other pollutants and of biofouling bacteria (Chae and An, 2018). MPs easily enter the food web at lower trophic levels, which facilitates the bioaccumulation of biological and chemical toxins as well as persistent organic pollutants in the organisms at each trophic level (Andrady, 2011; Johansen et al., 2018), and the subsequent biomagnification in successive trophic levels (Caruso, 2019; Saley et al., 2019). Ingestion of MPs has been documented for a large number of vertebrate and invertebrate species (Lusher, 2015).

Small pieces of plastic in the marine environment have been reported for the first time in 1971, in the sea surface waters of the Sargasso Sea in the North Atlantic Ocean, although at that time these

* Corresponding author.
E-mail address: susanna.piovano@usp.ac.fj (S. Piovano).

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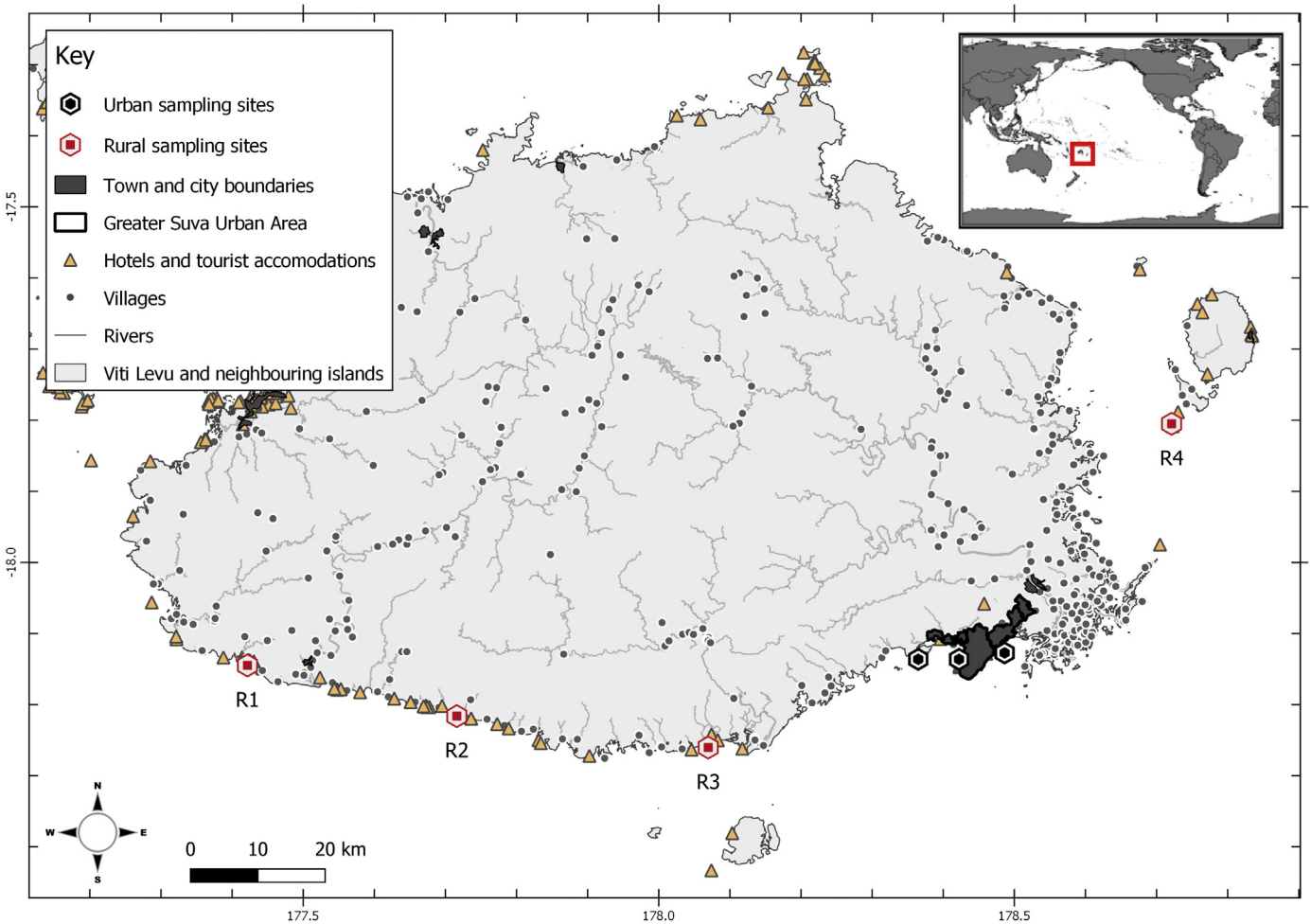

Fig. 1. Rural (red marker) and urban (black marker) sampling sites along the southern coast of Viti Levu, Fiji, South Pacific Ocean. Inset: location of Fiji within the Pacific Ocean. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

plastic fragments where not named as MPs (Carpenter and Smith, 1972). Since then, MPs have been found in virtually all marine geographic areas (Lusher, 2015); in the coastal waters of all major landmasses, such as Africa, the Americas, Asia, Australia, and Europe (Boucher and Friot, 2017; Chubarenko and Stepanova, 2017; Hedge et al., 2014; Phillips and Bonner, 2015; Zhang et al., 2019), in high seas, such as between Fiji and Australia (Reisser et al., 2013), and in the most remote parts of the oceans, such as in the depths of the Mariana Trench and in waters off Antarctica (Cincinelli et al., 2017; Peng et al., 2020; Rochman, 2018). In the marine environment, the low density of plastics results in far higher concentrations of MPs within the surface waters, where distribution is influenced by surface currents that promote accumulation in areas of convergence or in the global gyres, such as the South Pacific subgyre system (Cincinelli et al., 2017; Eriksen et al., 2013; Rochman, 2018). In addition, comparative studies between offshore and coastal waters have shown that concentration of MPs is generally higher within coastal systems, due to higher exposure to human-made debris (Cordova et al., 2019; Murphy et al., 2016). Studies of MPs have mostly targeted the surface coastal waters around the continents, especially in Europe and the Americas (Boucher and Friot, 2017; Browne et al., 2011; Frias et al., 2010; Isobe et al., 2014; Mohamed Nor and Obbard, 2014; Pauna et al., 2019), while a major knowledge gap on marine MPs has been identified in the Small Island Developing States (SIDS) in the South Pacific (Varea et al., 2019).

Marine habitats of coastal urban areas have been increasingly exposed to stressors, among which habitat loss, exploitation and pollution (Todd et al., 2019). While stressors can be complex and plentiful, they are primarily the result of human population-based factors, such as development and poor waste management (Burt, 2014). Within cities, dependency on plastic polymers is much higher than in rural areas (Boucher and Friot, 2017), which results in a greater release of MPs into urban environments (Shahul Hamid et al., 2018). For example, sediments from estuaries in urban areas had about five times the amount of MPs of the estuaries in rural areas (16 vs 3 pieces of MPs/250 g of sediment) along Singapore's coastline (Mohamed Nor and Obbard, 2014). Moreover, concentration of MPs in sea surface waters at inhabited islands was about twenty times the one recorded at uninhabited islands (0.00048 MP/L and 0.00002 MP/L, respectively) in Faafu Atoll in the Maldives (Saliu et al., 2018). MPs pollution has the potential to be particularly acute within urban centers of SIDS, where coastal marine resources are essential in sustaining the livelihoods of people, and where waste management practices are often absent or poorly implemented (Barnett and Adger, 2003; Todd et al., 2019). A recent urban drift within SIDS (United Nation Population Fund Pacific Sub-Regional Office, 2014), driven by expectations of economic benefits (Narsey et al., 2010), is likely to further increase the release of plastics and primary MPs in the coastal marine environments.

Fiji is among the largest countries within the South Pacific SIDS, and is considered to have a relatively high level of human development (United Nation Population Fund Pacific Sub-Regional Office, 2014). Although Fiji has > 300 islands and a larger total land surface area than most other South Pacific SIDS, it has been estimated that approximately 50% of the Fijian population lives within the coastal zone of the southern coast of the largest island, Viti Levu (Gonzalez et al., 2015).

This area includes the 'Greater Suva Urban Area', which extends from Lami town to the west and Nausori town to the east and includes the capital city, Suva. Hosting about 500 persons/km^2 (Fiji Bureau of Statistics, 2018), this is the most densely populated area in Fiji (United Nation Population Fund Pacific Sub-Regional Office, 2014). In addition, Suva houses the largest shipping port of the South Pacific SIDS (Asian Development Bank, 2007), as well as a small number of industries (Gonzalez et al., 2015). In comparison, the rest of the southern coast of Viti Levu has a population density below 40 persons/km^2, organized over a vast number of villages and two small towns (Fiji Bureau of Statistics, 2018), and it hosts a network of hotels which can cater for up to one third of Fiji's annual tourism (estimated at about 870,000 visitors/year by Fiji Bureau of Statistics, 2019). A low (0.00010 MP/L) concentration of MPs in subsurface waters has been reported for Suva inshore coastal waters (Ferreira et al., 2020).

The aim of the present study was to produce a broad screening of the abundance and characteristics of MPs in subsurface coastal waters in Fiji. Given that higher human density and higher level of urban and industrial development make an area more likely exposed to plastic pollution (Chubarenko and Stepanova, 2017), the southern coast of Viti Levu was chosen for this study. The baseline data obtained for seven sites across 200 km of coastline will allow future comparisons by monitoring programs, which can ultimately inform management decision in the region.

2. Materials and methods

Samples were collected during the tropical wet summer season of 2018, in January, February and March. The seven sampling sites (Table S1) are spread along the southern coast of Viti Levu in Fiji, South Pacific (Fig. 1), and consist of four rural agglomerations and three urban and peri-urban (i.e. within 3 km from the nearest urban area) agglomerations (hereafter collectively referred to as 'urban'). Subsurface coastal water samples were collected from shoreline at a depth of ca. 0.6 m (Fok et al., 2019) at slack water (± 60 min). On each sampling occasion, four replicates were taken within a radius of 200 m and at least 50 m apart. Water samples were collected via a polyvinyl chloride (PVC) Niskin bottle, and then transferred into new, sterile, 1 L polyethylene terephthalate (PET) bottles. They were kept frozen until filtration. Methods for extraction of MPs through filtration were adapted from Cincinelli et al. (2017) and Cordova et al. (2019), with the key difference being the type of filter membrane used, where we used a gridded 0.45 μm membrane (Millipore HAWG047S6). Before filtering, the volume of water sampled was measured by transferring the content of each bottle into previously flushed measuring cylinders. To avoid losing any MPs that may have been retained in the sampling bottles, each bottle was flushed three times directly into the filter apparatus. To reduce the risk of contamination, the equipment that came in contact with the water samples was flushed three times using filtered distilled water. Blank controls were carried out with filtered distilled water treated in the same manner as the water samples (including being frozen in sampling bottles). Immediately after filtration, filters were transferred into Petri dishes for visual analysis at a stereo-microscope (Olympus SZ51) fitted with a mobile camera (Samsung S5). MPs were visually classified as fragment, film, fiber and granule according to their form types (Cordova et al., 2019). Upon identification, each piece was photographed and its length (longest straight or curved edge, according to the shape of the piece) was measured using ImageJ v.1.52a (Rasband, 2019). To assess the polymer type of which the MPs were made, an attenuated total reflectance in Fourier transform infrared (ATR-FTIR) spectroscopy with a Spectrum Two Universal ATR (Perkin Elmer) and related Spectrum 10 (Perkin Elmer) software were used. ATR-FTIR analysis was based on spectra collected from 450 cm^{-1} to 4000 cm^{-1} at a resolution of 4 cm and a rate of 40 scans/sec (Jung et al., 2018). Identification of the polymers was done by comparing peak absorption bands with known absorption bands of plastic polymers (Jung et al., 2018). A minimum of four confirmed absorption bands were required to accept identification of the polymer. MP abundance was standardized as mean number of MP pieces per liter (MP/L). To rule out contamination from the Niskin bottle, color and form type of PVC MPs in the water samples were compared to the Niskin bottle for matching.

The software QGIS v.3.4.8 (QGIS Development Team, 2019) was used to map the distribution of MPs across the study area using spatial layers obtained from the Department of Land and Surveys of the Fiji Ministry for Lands and Mineral Resources. Differences in MP abundance among the seven sites were analyzed with Kruskal-Wallis test, while differences in form type, polymer type and length of MP pieces among sites were analyzed with Friedman test. Wilcoxon test was used to analyze the difference in form types, length classes and colors between urban and rural sites. Statistical analyses were performed using R v.3.6.1 (R Core Team, 2019).

3. Results

A total of 106 pieces of MPs were identified from across the seven sites sampled, with the highest and lowest abundance both recorded at two urban sites (mean ± S.D. = 2.9 ± 1.7 MP/L and 1.0 ± 0.7 MP/L at site U1 and U2, respectively; Fig. 2a), separated by approximately 5 km (straight line) of coastal water (Fig. 1). Differences in abundance of MP pieces were not statistically significant among the seven study sites (χ^2 (6, 7) = 10.9, p = 0.090), nor between urban and rural sites (mean ± S.D. = 2.0 ± 1.8 MP/L and 1.6 ± 1.7 MP/L, respectively; χ^2 (1, 2) = 3.0, p = 0.082). Controls for the sampling bottles yielded no MPs.

Abundance of MP form types (Fig. 2b) was significantly different (χ^2 (2, 3) = 12.1, p < 0.002), where the majority of the MPs were fibers (81%), followed by fragments (13%) and films (6%). On average, the smallest pieces were fragments (mean ± S.D. = 0.8 ± 0.4 mm), and the largest were films (2.3 ± 1.1 mm), while fibers had the broadest length range, ranging from 0.3 mm to 5.0 mm (1.6 ± 1.1 mm) (Figure S1:Figure S2:). No significant differences between urban and rural sites were found on abundance of fibers (mean ± S.D. = 1.6 ± 1.5 MP/L and 1.3 ± 1.4 MP/L, respectively; W = 9.5, p = 0.280), fragments (0.2 ± 0.4 MP/L in urban sites and 0.2 ± 0.5 MP/L in rural sites; W = 7.0, p = 0.843) or films (0.1 ± 0.3 MP/L at both urban and rural sites; W = 6.0, p = 1.000).

ATR-FTIR analysis was successfully performed on 101 samples, whereby a total of 14 polymers were identified (Table 1). Overall, PET, polyethylene (PE) and polypropylene (PP) were the most abundant polymers (27%, 23% and 20%, respectively), and were found across all seven sites. Nylon made up 7% of the samples and was observed in only three of the sites (U1, U3 and R2). The rest of the polymers were found scattered across the sites in residual amounts (< 5%). Different form types were composed of different polymers. Fibers were predominantly composed of PET (32%), PE (23%) and PP (21%), fragments were equally composed of polycarbonate (PC), PE, PP and polystyrene (PS) (15% each), while films were the least diverse, with ethylene vinyl acetate (EVA) and PE each contributing 33%.

Out of all 106 pieces of MPs analyzed for length, the smaller length classes, i.e. < 0.5 mm, 0.5–0.9 mm and 1.0–1.4 mm, were the most abundant making up 12%, 25%, and 23% of the total occurrence, respectively (Table S2), while the largest length classes (between 4 and 5 mm) were the least abundant (1% each). When comparing abundances of length classes between types of site, the length class 0.5–0.9 mm was the most abundant in the urban sites (mean ± S.D. = 0.5 ± 0.4 MP/L). Least abundant length class in these sites was 2.5–2.9 mm (0.0 ± 0.1 MP/L), while no piece was recorded in the class 4.5–4.9 mm. Within rural sites the length classes 0.5–0.9 mm and 1.0–1.4 mm were most abundant (0.4 ± 0.2 MP/L and 0.4 ± 0.3 MP/L, respectively), the length classes from 2.0–2.9 mm to 4.5–4.9 mm had the lowest abundance (0.1 ± 0.1 MP/L, each), while no piece was

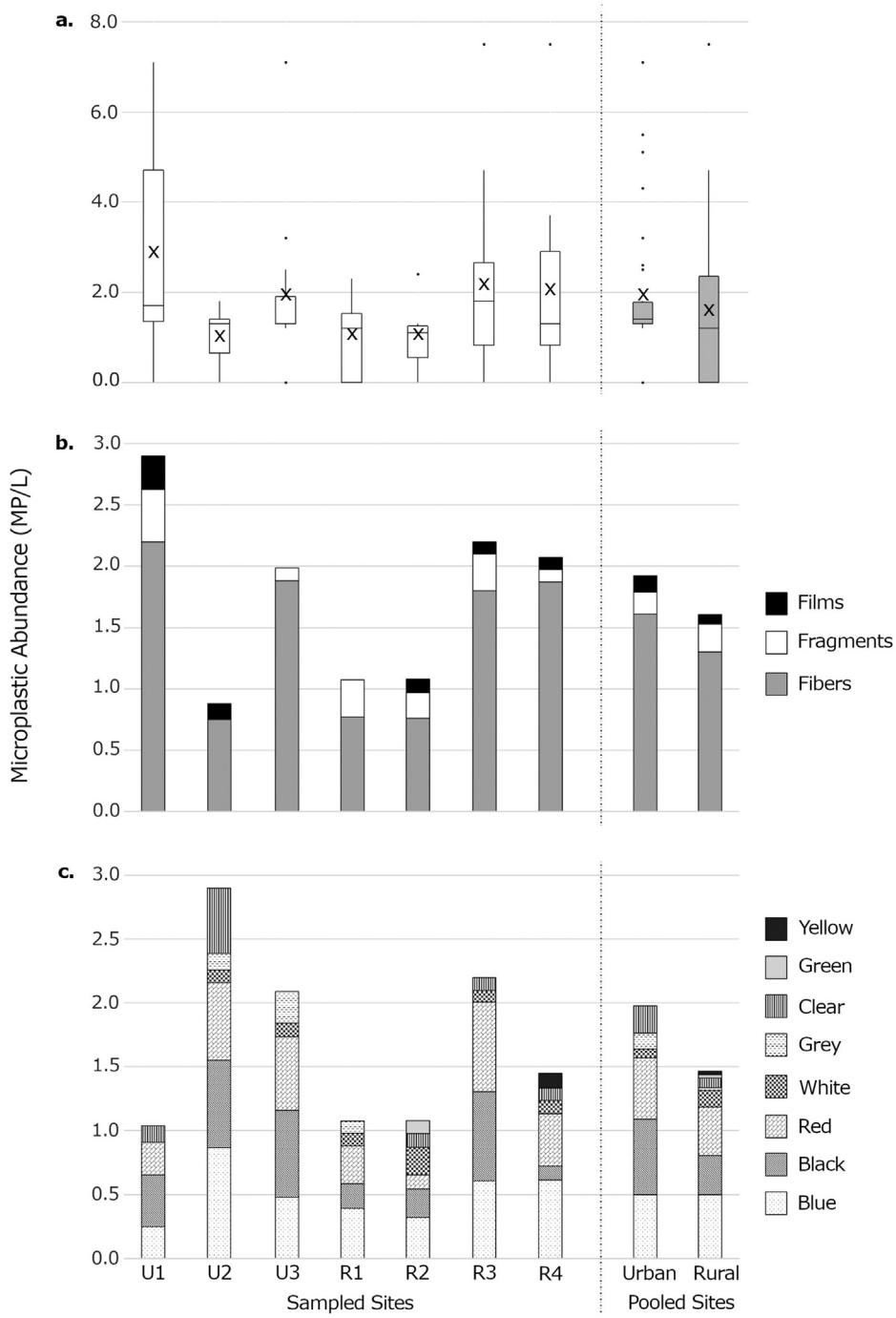

Fig. 2. Abundance of microplastics (MP/L) in subsurface inshore coastal waters across the seven sites along the southern coast of Viti Levu, Fiji, South Pacific. U = urban; R = Rural. (a) Abundances of pieces of MPs (mean is denoted by 'x') Same as 1.4.. (b) Abundances of form types of MPs. (c) Abundances of colors of MPs.

recorded in the class 5 mm. No statistical differences were observed when comparing MP length between rural and urban sites (W = 686.0, p = 0.669).

Among the different colors found (Figure S2:), blue pieces were the most abundant (30%), followed by red (25%) and black (25%) (Table S3, Fig. 2c). Other colors were found at residual amounts and combined made up 20% of the entire sample, with the least common being green, grey and yellow (4%, 3% and 3% respectively). Overall, no statistically significant difference in MPs abundance was found between the three main colors (χ^2 (2, 3) = 3.4, p = 0.338), however their abundances were found to be statistically higher than the other colors combined (χ^2 (7, 8) = 37.7, p < 0.001). Within pooled sites, blue MPs were equally abundant (mean ± S.D. = 0.5 ± 0.8 MP/L in urban sites and 0.5 ± 0.7 MP/L in rural sites). Black and red MPs were each found to be more abundant in urban sites (mean ± S.D. = 0.5 ± 0.7 MP/L and 0.6 ± 0.9 MP/L, respectively) than in rural sites (0.4 ± 0.6 MP/L and 0.3 ± 0.6 MP/L for black and red, respectively); the difference however was not statistically significant (black: W = 10.0, p = 0.197, red: W = 7.5, p = 0.718).

Analysis of the color, i.e. color matching between pieces of PVC and Niskin bottle, excluded that any of the PVC found (2% at both urban and rural sites; Table 1) originated from fragmentation of the equipment used.

Table 1
Composition of microplastics polymers according to their form type (fibers, fragments and films), sampling sites and type of site (urban and rural). List of polymer acronym (and name) is as follows: CA (cellulose acetate), EVA (ethylene vinyl acetate), latex, nitrile, nylon, PC (polycarbonate), PE (polyethylene), PET (polyethylene terephthalate), PMMA (poly(methyl methacrylate)), PP (polypropylene), PS (polystyrene), PU (polyurethane), PVA (polyvinyl acetate), and PVC (polyvinyl chloride).

Polymer	Form type (N (%))			Sampling sites (N (%))							Site types (N (%))		Overall (N (%))
	Fiber	Fragment	Film	U1	U2	U3	R1	R2	R3	R4	Urban	Rural	
CA	2 (2)	0 (0)	0 (0)	0 (0)	1 (11)	0 (0)	1 (9)	0 (0)	0 (0)	0 (0)	1 (2)	1 (2)	2 (2)
EVA	1 (1)	1 (8)	2 (33)	2 (10)	0 (0)	1 (6)	1 (9)	0 (0)	0 (0)	0 (0)	3 (7)	1 (2)	4 (4)
LATEX	2 (2)	0 (0)	0 (0)	0 (0)	1 (11)	1 (6)	0 (0)	0 (0)	0 (0)	0 (0)	2 (4)	0 (0)	2 (2)
Nitrile	0 (0)	1 (8)	1 (17)	1 (5)	0 (0)	0 (0)	0 (0)	0 (0)	1 (5)	0 (0)	1 (2)	1 (2)	2 (2)
Nylon	7 (9)	0 (0)	0 (0)	4 (19)	0 (0)	1 (6)	0 (0)	2 (22)	0 (0)	0 (0)	5 (11)	2 (4)	7 (7)
PC	0 (0)	2 (15)	0 (0)	1 (5)	0 (0)	0 (0)	0 (0)	0 (0)	1 (5)	0 (0)	1 (2)	1 (2)	2 (2)
PE	19 (23)	2 (15)	2 (33)	5 (24)	4 (45)	3 (19)	1 (9)	2 (22)	5 (22)	3 (23)	12 (26)	11 (20)	23 (23)
PET	26 (32)	1 (8)	0 (0)	5 (24)	1 (11)	6 (38)	1 (9)	3 (34)	7 (32)	4 (31)	12 (26)	15 (27)	27 (27)
PMMA	1 (1)	0 (0)	0 (0)	0 (0)	0(0)	0 (0)	1 (9)	0 (0)	0 (0)	0 (0)	0 (0)	1 (2)	1 (1)
PP	17 (21)	2 (15)	1 (17)	3 (13)	1 (11)	2 (13)	2 (18)	2 (22)	5 (22)	5 (38)	6 (13)	14 (25)	20 (20)
PS	2 (2)	2 (15)	0 (0)	0 (0)	0 (0)	1 (6)	3 (28)	0 (0)	0 (0)	0 (0)	1 (2)	3 (5)	4 (4)
PU	3 (4)	1 (8)	0 (0)	0 (0)	0 (0)	1 (6)	1 (9)	0 (0)	2 (9)	0 (0)	1 (2)	3 (5)	4 (4)
PVA	1 (1)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	1 (5)	0 (0)	0 (0)	1 (2)	1 (1)
PVC	1 (1)	1 (8)	0 (0)	0 (0)	1 (11)	0 (0)	0 (0)	0 (0)	0 (0)	1 (8)	1 (2)	1 (2)	1 (1)
Total N (%)	82 (81)	13 (13)	6 (6)	21 (21)	9 (9)	16 (16)	11 (11)	9 (9)	22 (22)	13 (12)	46 (46)	55 (54)	101 (100)

4. Discussion

This study presents a baseline and a comparison of abundance and characteristics of MPs in rural and urban sites along the same coastline within a SIDS. No significant differences between sites were found in the four parameters studied, i.e. presence and abundance of MPs, abundance of MPs form types, MPs size distribution and MPs color. These findings challenge the common expectation of a higher MPs pollution in urban areas compared to rural areas.

Pieces of MPs were found in all sites of this study, suggesting the likely presence of MPs along the whole 200 km of coastline between the eastern and western sites of this study. This finding reinforces that SIDS are not an exception to the widespread MPs pollution. Given that the majority of the population of SIDS live along the coastline exploiting coastal resources as source of food and livelihood (Barnett and Adger, 2003; Gonzalez et al., 2015; Narsey et al., 2010), and that MPs may introduce biological toxins and chemicals in the food chain (Andrady, 2011; Caruso, 2019; Johansen et al., 2018; Saley et al., 2019), results from this study point to both management implications and identification of ecological consequences of such a situation. Within the South Pacific, studies from a number of SIDS including French Polynesia, Samoa and Tahiti, showed that 8% to 25% of individual fish and 25% to 97% of examined species contained MPs within the gastrointestinal tracts (Forrest and Hindell, 2018; Markic et al., 2018). In the Greater Suva Urban Area in Fiji, 67% of the individual fish examined had ingested an average 5.5 ± 9.4 pieces of MP (Ferreira et al., 2020). The effects of MPs ingestion in fishes are not fully understood (Phillips and Bonner, 2015), but there is increasing evidence on MPs affecting fish molecular, biochemical and cellular pathways (Avio et al., 2017), possibly leading to oxidative damage, endocrine disruption or cell mutation (Ašmonaite et al., 2018).

The lack of a single common method for sampling MPs in surface and subsurface waters has resulted in an array of procedures being used, which makes difficult a direct comparison of findings among studies and across regions (Löder and Gerdts, 2015). For example, in the South China Sea, Cai et al. (2018) used a centrifugal pump to take subsurface waters at 0.5 m, which is similar to the depth of water sampling of the present study (0.6 m). Even though their MP abundance is only slightly higher than the findings from the present study (2.6 MP/L vs 1.8 MP/L, respectively), the different methodology used prevents a direct comparison between the two geographic areas. For the same reason, previous results from Reisser et al. (2013), which by using net tows estimated that the amount of MPs in surface offshore waters west of Viti Levu was of a similar magnitude to that of sites close to large Australian cities (i.e. > 15,500 MP/km^2), are not comparable with our findings.

The average global abundance of MPs in coastal waters was estimated by Barrows et al. (2018) as 5.9 ± 0.6 MP/L, whereby within the Pacific Ocean it ranged from an average 2.2 MP/L in pooled locations (which did not include Fiji) to an average 8.2 MP/L in Alaska. The average abundance of MPs found in this study (1.8 ± 1.8 MP/L) and in Vanuatu (ranging from 0 to 6.14 MP/L, in Barrows et al., 2018) suggest that SIDS within the South Pacific are currently at the lower end of MP pollution in coastal waters. However, despite their smaller population size and lower development, SIDS have MP abundances considerably higher than some highly populated and developed areas, such as San Francisco Bay in California (0.086 MP/L) and coastal waters of Surabaya in Indonesia (0.5 MP/L) (Cordova et al., 2019; Sutton et al., 2016). All these and the present study used the same type of methodology, known as "bulk sampling", whereby a volume of medium is collected before being processed for MP content (GESAMP, 2019). Bulk sampling is a low cost methodology that allows for easy and quick replication at relatively large scale (Barrows et al., 2018), and we recommend its adoption for monitoring programmes in SIDS as opposed to more expensive and less precise volume-reducing procedures (Barrows et al., 2017), where MPs are extracted and aggregated from the medium before analysis (Cutroneo et al., 2020). While volume-reducing methods such as net tow allow for a greater area and volume of water to be sampled, recent experiments have shown that they underestimate MP abundance when compared with bulk sampling (Barrows et al., 2017; Cutroneo et al., 2020; Fok et al., 2019). This is primarily due to the mesh of trawl-net, which sets a lower size limit and hence excludes smaller MPs; additionally, towing a trawl-net too fast or using a mesh size that is too small, has the potential to push MPs out of the way preventing their entrance in the net (Barrows et al., 2017; Cutroneo et al., 2020; Fok et al., 2019; Zhu et al., 2018).

MPs are characterized by patchy distribution, which has been attributed primarily to the dynamic nature of MPs pollution as well as to clustered MP abundance, which can vary drastically based on a number of factors including locations, coastal hydrodynamics, seasons and weather patterns (Avio et al., 2017). Abundances of MP in the Greater Suva Urban Area found in this study (2.0 MP/L) are several orders of magnitude higher than previously reported (0.00024 MP/L, in Ferreira et al., 2020). As the investigated areas roughly overlap, the differences are likely conducible to temporal variability (the two study periods are consecutive), depth of sampling and the methodology employed, even

though an increase in MP pollution cannot be excluded. Ferreira et al. (2020) sampled at a depth of 1–2 m and likely encountered MPs already exposed to biofouling, which alters the density of MPs causing them to sink to the subsurface (Kaiser et al., 2017). The previous study used a volume-reducing procedure which, although common (Löder and Gerdts, 2015), has been recently found to underestimate the extent of microplastic pollution (Barrows et al., 2017). Nevertheless, it is worth noting that both studies identified the same origin of MP producing pollutants, suggesting that since at least 2016 MP pollution in the Greater Suva Urban Area has been dominated by fibers, with minor contributions from fragments and films. Altogether, these findings point towards the need of a widespread screening of marine waters throughout the water column to efficiently gauge the extent of MP pollution in Fiji. The present study presents baseline data and suggests a low-cost standardized methodology that can be modelled into a regular monitoring program whereby samples can be collected continuously and regularly from along Fijian surface coastal waters.

Analysis of length ranges show that MPs with a length range of 0.5–0.9 mm were the most dominant pieces in both rural and urban sites. This is consistent with findings from Australia, China and Indonesia (Cordova et al., 2019; Reisser et al., 2013; Zhang et al., 2019), where MP pieces with lengths between 0.5 mm and 1 mm made up the majority of MPs. Progressive degradation of plastic particles into ever smaller sizes due to environmental weathering should be eminent, and unless regular sources of new, larger MPs exist, sizes < 0.5 mm should be dominant (Cordova et al., 2019; Cózar et al., 2014). For example, in more isolated areas such as Faafu Atoll in the Maldives, or in the remote waters of the north-western Pacific Ocean, between 50% and 64% of MPs pieces were smaller than 0.5 mm (Pan et al., 2019; Saliu et al., 2018). Therefore, given that in our study the majority of the MPs were larger than 0.5 mm and that both rural and urban sites showed no difference between the abundances of MPs across the different length ranges, it is possible that rural sites have contributed to MP pollution just as much as the urban sites.

The most common form type of MPs found in this study, fibers, is consistent with the global trend reported for coastal waters worldwide, whose source has been identified primarily in washing laundry (Boucher and Friot, 2017; Siegfried et al., 2017). In the present study, the widespread distribution of fibers in both rural and urban sites suggests that laundry is the most likely source of MPs along the southern coast of Viti Levu. This is further supported by the prevalent types of polymer that were found (i.e. PET, PE, PP and Nylon) which, although used for a variety of plastic-based objects, are common within the clothing industry and widely used to form a variety of clothing textiles (Browne et al., 2011; Siegfried et al., 2017). The most likely explanation for the characteristics of the observed dominant MPs is linked to wastewater treatment. Wastewaters are subject to different types and levels of treatment in wastewater treatment facilities, where sewage water treatment plants can remove up to 99% of MPs from wastewaters (Burns and Boxall, 2018). Within the urban sites sampled for this study, the Kinoya sewage treatment plant includes secondary treatment of black water and grey water collected from approximately 60% of the Greater Suva Urban Area (Campbell et al., 1982), and possibly traps a large portion of fibers from laundry effluents. However, the sewage treatment plant has been reported to contribute to MPs accumulation in sediments at Laucala Bay (Ferreira et al., 2020). Within rural sites, treatment effort is concentrated primarily on black water, mainly through septic systems, while grey water, which includes discharge from washing laundry, is often diverted into nearby bays, streams or rivers (Taloiburi, 2009). Sampling of grey water from the coastal areas, as well as from the freshwater were laundry is most commonly done, would allow to confirm the likely source of MPs suggested by the present findings from subsurface marine waters. Given that majority of the SIDS lack efficient, centralized wastewater treatment systems (World Health Organisation, 2008), it is possible that the findings from Fiji are representative of the majority of the SIDS. As such, it would be of interest for SIDS to increase efforts of small-scale grey water treatment devices such as grey water sumps, or implement requirements for MP filters to be used together with laundry machines (Napper and Thompson, 2016). Improvement of wastewater treatments and implementation of a monitoring program of marine coastal waters that would allow comparison with the baseline data of this study are highly recommended, as they will allow an overall evaluation of the effectiveness of any management measures (or lack thereof) implemented to reduce environmental MPs pollution.

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CRediT authorship contribution statement

Jasha Dehm: Formal analysis, Investigation, Data curation, Writing - original draft, Writing - review & editing, Visualization. **Shubha Singh**: Investigation, Data curation, Writing - review & editing. **Marta Ferreira**: Conceptualization, Writing - review & editing, Funding acquisition. **Susanna Piovano**: Conceptualization, Writing - original draft, Writing - review & editing, Supervision, Project administration, Funding acquisition.

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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References

Akdogan, Z., Guven, B., 2019. Microplastics in the environment: a critical review of current understanding and identification of future research needs. Environ. Pollut. 254, 113011. https://doi.org/10.1016/J.ENVPOL.2019.113011.

Andrady, A.L., 2011. Microplastics in the marine environment. Mar. Pollut. Bull. 62, 1596–1605. https://doi.org/10.1016/J.MARPOLBUL.2011.05.030.

Development Bank, Asian, 2007. Oceanic Voyages: Shipping in the Pacific. Pacific Studies. Asian Development Bank, Philippines.

Ašmonaite, G., Larsson, K., Undeland, I., Sturve, J., Carney Almroth, B., 2018. Size matters: ingestion of relatively large microplastics contaminated with environmental pollutants posed little risk for fish health and fillet quality. Environ. Sci. Technol. 52, 14381–14391. https://doi.org/10.1021/acs.est.8b04849.

Avio, C.G., Gorbi, S., Regoli, F., 2017. Plastics and microplastics in the oceans: from emerging pollutants to emerged threat. Mar. Environ. Res. 128, 2–11. https://doi.org/10.1016/J.MARENVRES.2016.05.012.

Barnett, J., Adger, W.N., 2003. Climate dangers and atoll countries. Clim. Chang. 61, 321–337.

Barrows, A.P., Neumann, C.A., Berger, M.L., Shaw, S.D., 2017. Grab *vs*. neuston tow net: a microplastic sampling performance comparison and possible advances in the field. Anal. Methods 9, 1446–1453. https://doi.org/10.1039/c6ay02387h.

Barrows, A.P.W., Cathey, S.E., Petersen, C.W., 2018. Marine environment microfiber contamination: global patterns and the diversity of microparticle origins. Environ. Pollut. 237, 275–284. https://doi.org/10.1016/j.envpol.2018.02.062.

Boucher, J., Friot, D., 2017. Primary Microplastics in the Oceans:A Global Evaluation of Sources, Primary Microplastics in the Oceans: A Global Evaluation of Sources. IUCN, Gland, Switzerland. https://doi.org/10.2305/iucn.ch.2017.01.en.

Browne, M.A., Crump, P., Niven, S.J., Teuten, E., Tonkin, A., Galloway, T., Thompson, R., 2011. Accumulation of microplastic on shorelines worldwide: sources and sinks. Environ. Sci. Technol. 45, 9175–9179. https://doi.org/10.1021/es201811s.

Burns, E.E., Boxall, A.B.A., 2018. Microplastics in the aquatic environment: evidence for or against adverse impacts and major knowledge gaps. Environ. Toxicol. Chem. 37, 2776–2796. https://doi.org/10.1002/etc.4268.

Burt, J.A., 2014. The environmental costs of coastal urbanization in the Arabian Gulf. City

18, 760–770. https://doi.org/10.1080/13604813.2014.962889.
Cai, M., He, H., Liu, M., Li, S., Tang, G., Wang, W., Huang, P., Wei, G., Lin, Y., Chen, B., Hu, J., Cen, Z., 2018. Lost but can't be neglected: huge quantities of small microplastics hide in the South China Sea. Sci. Total Environ. 633, 1206–1216. https://doi.org/10.1016/j.scitotenv.2018.03.197.
Campbell, L., Cock, R., Corris, R., 1982. Kinoya Sewerage Treatment Plant - Report on Receiving Water Study. Commonwealth Department of Transport and Construction & Caldwell Connell Engineers, Canberra, Australia.
Carpenter, E.J., Smith, K.L., 1972. Plastics on the Sargasso sea surface. Science 175, 1240–1241. https://doi.org/10.1126/science.175.4027.1240.
Caruso, G., 2019. Microplastics as vectors of contaminants. Mar. Pollut. Bull. 146, 921–924. https://doi.org/10.1016/J.MARPOLBUL.2019.07.052.
Chae, Y., An, Y.-J., 2018. Current research trends on plastic pollution and ecological impacts on the soil ecosystem: a review. Environ. Pollut. 240, 387–395. https://doi.org/10.1016/J.ENVPOL.2018.05.008.
Chubarenko, I., Stepanova, N., 2017. Microplastics in sea coastal zone: lessons learned from the Baltic amber. Environ. Pollut. 224, 243–254. https://doi.org/10.1016/J.ENVPOL.2017.01.085.
Cincinelli, A., Scopetani, C., Chelazzi, D., Lombardini, E., Martellini, T., Katsoyiannis, A., Fossi, M.C., Corsolini, S., 2017. Microplastic in the surface waters of the Ross Sea (Antarctica): occurrence, distribution and characterization by FTIR. Chemosphere 175, 391–400. https://doi.org/10.1016/j.chemosphere.2017.02.024.
Cordova, M.R., Purwiyanto, A.I.S., Suteja, Y., 2019. Abundance and characteristics of microplastics in the northern coastal waters of Surabaya, Indonesia. Mar. Pollut. Bull. 142, 183–188. https://doi.org/10.1016/J.MARPOLBUL.2019.03.040.
Cózar, A., Echevarría, F., González-Gordillo, J.I., Irigoien, X., Úbeda, B., Hernández-León, S., Palma, Á.T., Navarro, S., García-de-Lomas, J., Ruiz, A., Fernández-de-Puelles, M.L., Duarte, C.M., 2014. Plastic debris in the open ocean. Proc. Natl. Acad. Sci. 111, 10239–10244. https://doi.org/10.1073/PNAS.1314705111.
Cutroneo, L., Reboa, A., Besio, G., Borgogno, F., Canesi, L., Canuto, S., Dara, M., Enrile, F., Forioso, I., Greco, G., Lenoble, V., Malatesta, A., Mounier, S., Petrillo, M., Rovetta, R., Stocchino, A., Tesan, J., Vagge, G., Capello, M., 2020. Microplastics in seawater: sampling strategies, laboratory methodologies, and identification techniques applied to port environment. Environ. Sci. Pollut. Res. 27 (9), 8938–8952. https://doi.org/10.1007/s11356-020-07783-8.
Derraik, J.G.., 2002. The pollution of the marine environment by plastic debris: a review. Mar. Pollut. Bull. 44, 842–852. https://doi.org/10.1016/S0025-326X(02)00220-5.
Eriksen, M., Maximenko, N., Thiel, M., Cummins, A., Lattin, G., Wilson, S., Hafner, J., Zellers, A., Rifman, S., 2013. Plastic pollution in the South Pacific subtropical gyre. Mar. Pollut. Bull. 68, 71–76. https://doi.org/10.1016/J.MARPOLBUL.2012.12.021.
Ferreira, M., Thompson, J., Paris, A., Rohindra, D., Rico, C., 2020. Presence of microplastics in water, sediments and fish species in an urban coastal environment of Fiji, a Pacific small island developing state. Mar. Pollut. Bull. 153, 110991. https://doi.org/10.1016/j.marpolbul.2020.110991.
Fiji Bureau of Statistics, 2018. 2017 Fiji population and housing census: administration report and general tables. Fiji Bureau of Statistics, Suva, Fiji.
Fiji Bureau of Statistics, 2019. Provisional visitor arrival October 2019 (FBoS Release No. 67). Fiji Bureau of Statistics, Suva, Fiji.
Fok, L., Lam, T.W.L., Li, H.X., Xu, X.R., 2019. A meta-analysis of methodologies adopted by microplastic studies in China. Sci. Total Environ. 718, 135371. https://doi.org/10.1016/j.scitotenv.2019.135371.
Forrest, A.K., Hindell, M., 2018. Ingestion of plastic by fish destined for human consumption in remote South Pacific Islands. Aust. J. Marit. Ocean Aff. 10, 81–97. https://doi.org/10.1080/18366503.2018.1460945.
Frias, J.P.G.L., Nash, R., 2019. Microplastics: finding a consensus on the definition. Mar. Pollut. Bull. 138, 145–147. https://doi.org/10.1016/J.MARPOLBUL.2018.11.022.
Frias, J.P.G.L., Sobral, P., Ferreira, A.M., 2010. Organic pollutants in microplastics from two beaches of the Portuguese coast. Mar. Pollut. Bull. 60, 1988–1992. https://doi.org/10.1016/j.marpolbul.2010.07.030.
GESAMP, 2016. Sources, Fate and Effects of Microplastics in the Marine Environment: Part Two of a Global Assessment. GESAMP Reports and Studies 93 International Maritime Organization, London, UK.
GESAMP, 2019. Guidelines for the Monitoring and Assessment of Plastic Litter and Microplastics in the Ocean. GESAMP Reports and Studies, 99 . International Maritime Organization, London, UK.
Gonzalez, R., Ram-Bidesi, V., Leport, G., Pascal, N., Brander, L.M., Fernandes, L., Salcone, J., Seidl, A., 2015. National Marine Ecosystem Service Valuation: Fiji. MACBIO (GIZ/IUCN/SPREP), Suva, Fiji.
Guo, X., Wang, J., 2019. The chemical behaviors of microplastics in marine environment: a review. Mar. Pollut. Bull. 142, 1–14. https://doi.org/10.1016/J.MARPOLBUL.2019.03.019.
Hedge, L., Johnston, E., Ayoung, S.T., Birch, G.F., Booth, D.J., Creese, R.G., Doblin, M.A., Figueira, W.F., Gribben, P.E., Hutchings, P.A., Pinto, M., Marzinelli, E., Pritchard, T., Roughan, M., 2014. Sydney Harbour: A Systematic Review of the Science. Sydney Institute of Marine Science, Sydney, Australia.
Isobe, A., Kubo, K., Tamura, Y., Kako, S., Nakashima, E., Fujii, N., 2014. Selective transport of microplastics and mesoplastics by drifting in coastal waters. Mar. Pollut. Bull. 89, 324–330. https://doi.org/10.1016/J.MARPOLBUL.2014.09.041.
Johansen, M.P., Prentice, E., Cresswell, T., Howell, N., 2018. Initial data on adsorption of Cs and Sr to the surfaces of microplastics with biofilm. J. Environ. Radioact. 190–191, 130–133. https://doi.org/10.1016/j.jenvrad.2018.05.001.
Jung, M.R., Horgen, F.D., Orski, S.V., Rodriguez, C.V., Beers, K.L., Balazs, G.H., Jones, T.T., Work, T.M., Brignac, K.C., Royer, S.-J., Hyrenbach, K.D., Jensen, B.A., Lynch, J.M., 2018. Validation of ATR FT-IR to identify polymers of plastic marine debris, including those ingested by marine organisms. Mar. Pollut. Bull. 127, 704–716. https://doi.org/10.1016/J.MARPOLBUL.2017.12.061.
Kaiser, D., Kowalski, N., Waniek, J.J., 2017. Effects of biofouling on the sinking behavior of microplastics. Environ. Res. Lett. 12, 124003. https://doi.org/10.1088/1748-9326/aa8e8b.
Kühn, S., van Franeker, J.A., 2020. Quantitative overview of marine debris ingested by marine megafauna. Mar. Pollut. Bull. 151, 110858. https://doi.org/10.1016/J.MARPOLBUL.2019.110858.
Löder, M.G.J., Gerdts, G., 2015. Methodology used for the detection and identification of microplastics — a critical appraisal. In: Bergmann, M., Gutow, L., Klages, M. (Eds.), Marine Anthropogenic Litter. Springer International Publishing, Cham, pp. 201–227.
Lusher, A., 2015. Microplastics in the marine environment: distribution, interactions and effects. In: Bergmann, M., Gutow, L., Klages, M. (Eds.), Marine Anthropogenic Litter. Springer International Publishing, Cham, pp. 245–307.
Markic, A., Niemand, C., Bridson, J.H., Mazouni-Gaertner, N., Gaertner, J.-C., Eriksen, M., Bowen, M., 2018. Double trouble in the South Pacific subtropical gyre: increased plastic ingestion by fish in the oceanic accumulation zone. Mar. Pollut. Bull. 136, 547–564. https://doi.org/10.1016/J.MARPOLBUL.2018.09.031.
Mishra, S., Rath, C. charan, Das, A.P., 2019. Marine microfiber pollution: a review on present status and future challenges. Mar. Pollut. Bull. 140, 188–197. https://doi.org/10.1016/J.MARPOLBUL.2019.01.039.
Mohamed Nor, N.H., Obbard, J.P., 2014. Microplastics in Singapore's coastal mangrove ecosystems. Mar. Pollut. Bull. 79, 278–283. https://doi.org/10.1016/j.marpolbul.2013.11.025.
Murphy, F., Ewins, C., Carbonnier, F., Quinn, B., 2016. Wastewater treatment works (WwTW) as a source of microplastics in the aquatic environment. Environ. Sci. Technol. 50, 5800–5808. https://doi.org/10.1021/acs.est.5b05416.
Napper, I.E., Thompson, R.C., 2016. Release of synthetic microplastic plastic fibres from domestic washing machines: effects of fabric type and washing conditions. Mar. Pollut. Bull. 112, 39–45. https://doi.org/10.1016/J.MARPOLBUL.2016.09.025.
Prasad, B.C., Narsey, W., Robertson, A.S., Seniloli, K., Jongstra, E., Smiles, S., Hau'ofa, B., Pene, F., 2010. Population and Development in the Pacific Islands: Accelerating the ICPD Programme of Action at 15. Proc. Reg. Symp. held Univ. South Pacific 23–25 Novemb 2009. UNFPA Office for the Pacific : University of the South Pacific, Suva, Fiji.
Pan, Z., Guo, H., Chen, H., Wang, S., Sun, X., Zou, Q., Zhang, Y., Lin, H., Cai, S., Huang, J., 2019. Microplastics in the Northwestern Pacific: abundance, distribution, and characteristics. Sci. Total Environ. 650, 1913–1922. https://doi.org/10.1016/J.SCITOTENV.2018.09.244.
Pauna, V.H., Buonocore, E., Renzi, M., Russo, G.F., Franzese, P.P., 2019. The issue of microplastics in marine ecosystems: a bibliometric network analysis. Mar. Pollut. Bull. 149, 110612. https://doi.org/10.1016/J.MARPOLBUL.2019.110612.
Peng, G., Bellerby, R., Zhang, F., Sun, X., Li, D., 2020. The ocean's ultimate trashcan: Hadal trenches as major depositories for plastic pollution. Water Res. 168, 115121. https://doi.org/10.1016/J.WATRES.2019.115121.
Phillips, M.B., Bonner, T.H., 2015. Occurrence and amount of microplastic ingested by fishes in watersheds of the Gulf of Mexico. Mar. Pollut. Bull. 100, 264–269. https://doi.org/10.1016/J.MARPOLBUL.2015.08.041.
QGIS Development Team, 2019. QGIS Geographic Information System.
R Core Team, 2019. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing.
Rasband, W., 2019. ImageJ 1.52a.
Reisser, J., Shaw, J., Wilcox, C., Hardesty, B.D., Proietti, M., Thums, M., Pattiaratchi, C., 2013. Marine plastic pollution in waters around Australia: characteristics, concentrations, and pathways. PLoS One 8 (11), e80466. https://doi.org/10.1371/journal.pone.0080466.
Rochman, C.M., 2018. Microplastics research—from sink to source. Science 360 (6384), 28–29. https://doi.org/10.1126/science.aar7734.
Saley, A.M., Smart, A.C., Bezerra, M.F., Burnham, T.L.U., Capece, L.R., Lima, L.F.O., Carsh, A.C., Williams, S.L., Morgan, S.G., 2019. Microplastic accumulation and biomagnification in a coastal marine reserve situated in a sparsely populated area. Mar. Pollut. Bull. 146, 54–59. https://doi.org/10.1016/J.MARPOLBUL.2019.05.065.
Saliu, F., Montano, S., Garavaglia, M.G., Lasagni, M., Seveso, D., Galli, P., 2018. Microplastic and charred microplastic in the Faafu atoll, Maldives. Mar. Pollut. Bull. 136, 464–471. https://doi.org/10.1016/J.MARPOLBUL.2018.09.023.
Schwarz, A.E., Ligthart, T.N., Boukris, E., van Harmelen, T., 2019. Sources, transport, and accumulation of different types of plastic litter in aquatic environments: a review study. Mar. Pollut. Bull. 143, 92–100. https://doi.org/10.1016/J.MARPOLBUL.2019.04.029.
Shahul Hamid, F., Bhatti, M.S., Norkhairiyah, Anuar, Norkhairah, Anuar, Mohan, P., Periathamby, A., 2018. Worldwide distribution and abundance of microplastic: how dire is the situation? Waste Manag. Res. 36, 873–897. https://doi.org/10.1177/0734242X18785730.
Siegfried, M., Koelmans, A.A., Besseling, E., Kroeze, C., 2017. Export of microplastics from land to sea. A modelling approach. Water Res. 127, 249–257. https://doi.org/10.1016/J.WATRES.2017.10.011.
Silva, C.J.M., Silva, A.L.P., Gravato, C., Pestana, J.L.T., 2019. Ingestion of small-sized and irregularly shaped polyethylene microplastics affect *Chironomus riparius* life-history traits. Sci. Total Environ. 672, 862–868. https://doi.org/10.1016/J.SCITOTENV.2019.04.017.
Sun, X., Liang, J., Zhu, M., Zhao, Y., Zhang, B., 2018. Microplastics in seawater and zooplankton from the Yellow Sea. Environ. Pollut. 242, 585–595. https://doi.org/10.1016/J.ENVPOL.2018.07.014.
Sutton, R., Mason, S.A., Stanek, S.K., Willis-Norton, E., Wren, I.F., Box, C., 2016. Microplastic contamination in the San Francisco Bay, California, USA. Mar. Pollut. Bull. 109, 230–235. https://doi.org/10.1016/j.marpolbul.2016.05.077.
Taloiburi, E.J., 2009. An Evaluation of the Effects of Wastewater Treatment Initiatives on Water Quality in Coastal Waters along the Coral Coast, Southwest Viti Levu, Fiji

Islands. University of the South Pacific, Suva, Fiji, pp. 208.
Todd, P.A., Heery, E.C., Loke, L.H.L., Thurstan, R.H., Kotze, D.J., Swan, C., 2019. Towards an urban marine ecology: characterizing the drivers, patterns and processes of marine ecosystems in coastal cities. Oikos 128, 1215–1242. https://doi.org/10.1111/oik.05946.
United Nation Environment Programme, 2015. Plastic in cosmetics: are we polluting the environment through our personal care? UNEP, Nairobi, Kenya.
United Nation Environment Programme, 2018. World Environment Day Overview. United Nations Environment Programme.
United Nation Population Fund Pacific Sub-Regional Office, 2014. Population and Development Profiles : Pacific Island Countries. UNFPA Pacific, Suva, Fiji.
Varea, R.R., Piovano, S., Ferreira, M., 2019. Knowledge Gaps in Ecotoxicology Studies in Pacific Island Countries and Territories' Marine Environments – A Review. (submitted).
World Health Organisation, 2008. Sanitation, Hygiene and Drinking-Water in the Pacific Island Countries.
Zhang, J., Zhang, C., Deng, Y., Wang, R., Ma, E., Wang, J., Bai, J., Wu, J., Zhou, Y., 2019. Microplastics in the surface water of small-scale estuaries in Shanghai. Mar. Pollut. Bull. 149, 110569. https://doi.org/10.1016/J.MARPOLBUL.2019.110569.
Zhu, L., Bai, H., Chen, B., Sun, X., Qu, K., Xia, B., 2018. Microplastic pollution in North Yellow Sea, China: observations on occurrence, distribution and identification. Sci. Total Environ. 636, 20–29. https://doi.org/10.1016/j.scitotenv.2018.04.182.