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Occurrence and abundance of *meso* and microplastics in sediment, surface waters, and marine biota from the South Pacific region

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ABSTRACT

Data on the occurrence and abundance of *meso* and microplastics for the South Pacific are limited and there is urgent need to fill this knowledge gap. The main aim of the study was to apply a rapid screening method, based on the fluorescence tagging of polymers using Nile red, to determine the concentration of *meso* and microplastics in biota, sediment and surface waters near the capital cities of Vanuatu and Solomon Islands. A spatial investigation was carried out for sediment, biota and water as well as a temporal assessment for sediment for two consecutive years (2017 and 2018). Accumulation zones for microplastics were identified supported by previous hydrodynamic models. Microplastics were detected for all environmental compartments investigated indicating their widespread presence for Vanuatu and Solomons Islands. This method was in alignment with previous recommendations that the Nile red method is a promising approach for the largescale mapping of microplastics in a monitoring context.

1. Introduction

World plastics production has almost doubled in the last twenty years to around 400 million tons per year (Geyer et al., 2017). It has been estimated that between 4.8 and 12.7 million tons of plastics enter the marine environment annually from land with rivers as main pathways (Jambeck et al., 2015; Lebreton et al., 2017), causing plastics to form a large proportion of marine litter. Additional sea-based sources of plastic marine litter include fishing industry, aquaculture, shipping and maritime activities (GESAMP, 2016). A large part of this plastic litter consists of microplastics which are plastic particles below 5 mm in diameter (Arthur et al., 2008). Microplastics can be defined as either primary microplastics such as pre-production pellets from manufacturing processes or microbeads from cleaning and personal care products or secondary microplastics originating from the degradation process of larger debris invertebrates (GESAMP, 2015). Microplastics have been found in every marine niche investigated, from coastal zones to the open ocean and the deep-sea (Kane and Clare, 2019; Taylor et al., 2016). Deep-sea sediments have been suggested as a likely final sink for microplastics (Woodall et al., 2014). High concentrations have now even been reported from remote locations within the Arctic circle (Lusher et al., 2015c; Bergmann et al., 2017; Kanhai et al., 2018). Field and laboratory studies have demonstrated the ingestion of microplastics by a large range of marine organisms representing various trophic levels including seabirds, marine mammals, fish and invertebrates (GESAMP, 2015). Detrimental physical effects of microplastics have been reported following ingestion (Wright et al., 2013). There is evidence that microplastics can act as carriers for harmful contaminants (i.e. hydrophobic organic compounds, additives, pathogens) with the potential for transfer to biota following ingestion (Rochman et al., 2013; Tanaka et al., 2013; Bakir et al., 2014). However, the transfer of sorbed co-contaminants from microplastics to biota may well be negligible compared to other routes of exposure (Bakir et al., 2016; Herzke et al., 2016; Koelmans et al., 2016; Lohmann, 2017).

The main aim of this study was to apply a rapid screening method, based on the fluorescence tagging of polymers using Nile red, to determine the concentration of *meso* and microplastics in marine environmental samples and to define accumulation zones for microplastics supported by previous hydrodynamic models (Graham et al., 2020). The main objectives were to investigate the occurrence and abundance of meso and microplastics in i) surface water from Vila Bay and Mele Bay (Vanuatu) trawled in 2018, ii) sediment samples from the

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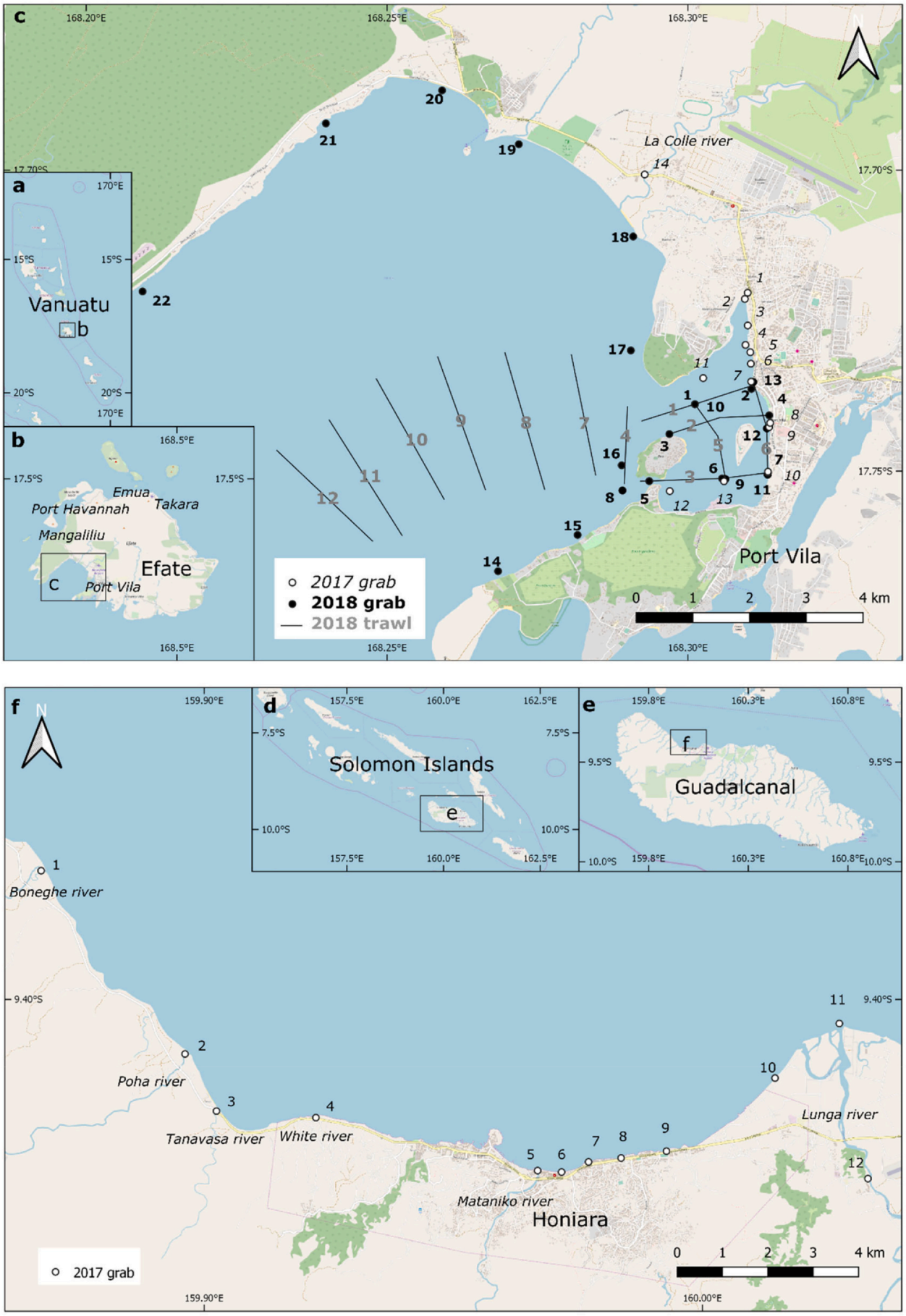

Fig. 1. Overview of sample sites in Vanuatu (a, b, c) and Solomon Islands (d, e, f). Numbers in c and f are the identifiers of the sample codes used in the text.

Table 1
List of chemicals, manufacturers and suppliers.

Chemicals	Molecular formula	Manufacturer/supplier	Purity (%)
Potassium hydroxide	KOH	South Pacific suppliers, Port Vila, Vanuatu	–
Sodium hypochlorite	NaClO	South Pacific suppliers, Port Vila, Vanuatu	12.5% active chlorine
Ethanol	C_2H_6O	Acros organics/Thermo Fisher scientific	95% purity
Nile red	$C_{20}H_{18}N_2O_2$	Acros organics/Thermo Fisher scientific	99% purity
Sodium chloride	NaCl	VWR/VWR	Technical grade

coastal region of Port Vila (Vanuatu) and Honiara (the Solomon Islands) grabbed in 2017 and 2018 and iii) a range of biota from around Vanuatu collected in 2018.

2. Materials and methods

2.1. Sampling locations

2.1.1. Surface water samples

In 2018, the surface waters of Vila Bay and Mele Bay were trawled with a manta trawl in a grid covering Vila Bay, and 6 trawls in Mele Bay towards the open ocean (Fig. 1). The manta trawl (5Gyres) has a mouth of 60 × 18 cm and a net with a mesh size of 335 μm. At each location, it was trawled for 30 min at the side of the boat at speeds below 3 knots and perpendicular on the currents to avoid collision with the vessel itself. A General Oceanics mechanical flowmeter (one-way clutch) was attached to the trawl to measure the sample volume. At the end of every trawl, the trawl frame and net were brought on board and the contents of the net were flushed with a high-pressure washer from the outside down to the cod end. The sample was then rinsed into a clean 315 μm test sieve and transferred into a 500 mL glass sample jar which had been cleaned beforehand using reverse osmosis (RO) water and closed with aluminium foil between lid and jar. To assess contamination by air, a second jar was opened and closed simultaneously with the sample jar. To assess potential contamination from the high-pressure washer, water was sprayed in a glass jar and analysed for microplastics. The samples were stored and frozen (−18 °C) until further analysis. In total, 12 surface water samples were collected. The distance measured by the flow meter was combined with the submerged surface of the trawl mouth to calculate plastic particles km^{-2} or m^{-3}, respectively. A schematic diagram of the protocol is presented in Fig. S1.

2.1.2. Sediments

The sediments of Vila Bay and Mele Bay (Vanuatu) were collected with a small Van Veen grab (Duncan and Associates, UK, sampled area 0.025 m^2) in 2017 and 2018. In 2017, samples were taken from the shores of Vila Bay and its northern tip, Fatumaru Bay. In 2018, the grab was deployed off a vessel by hand at points along the grid within Vila bay and at 9 locations spread evenly along the outside of Mele bay (Fig. 1a-c). Sediment grabs were subsampled with a pre-rinsed steel spoon and stored in glass jars. The jars were pre-rinsed with RO water in the laboratory and covered with aluminium foil, wedged between plastic lid and jar edges. In 2018, on each location, two grabs were collected, one for microplastics analysis and one for particle size analysis (PSA). No PSA grab samples were collected in 2017. To investigate background contamination during sampling, a collecting pot was left open during the time of sampling. The samples were stored and frozen (−18 °C) until further analysis. In total, 20 sediment grabs from 2018 and 14 from 2017 from Vanuatu were analysed for microplastics. The same method was used in 2017 to collect grab samples from 6 coastal stations on the north side of Guadalcanal within or nearby the Honiara urban area (Solomon Islands). Additional samples were taken from 6 rivers (Lunga, Poha, Mataniko, Tavasa, White and Boneghi) stations (Fig. 1e-f).

2.1.3. Biota

36 fishes were collected from four main locations on Efate Island in 2018 (Fig. 1). Six stomachs from pelagic yellowfin tuna (*Thunnus albacares*) were received from local (hook and line) game fishing. Additionally, 25 reef fish, one (1) Poulet or “chicken fish” (deep-water bottom- snapper: Etelis and *Pristipomoides* spp.) and another three (3) yellowfin tunas were collected with gillnets from local fishermen in Mangaliliu, Havannah bay, Emua and Takara wharf. All fish were returned and dissected in the lab at the Bureau of Standards. The shallow water reef fish collected consisted of a mix of species including parrotfish, triggerfish, trevally and surgeonfish. As a reference, 14 land crabs (*Cardisoma* spp.) captured for human consumption were also bought from the main food market in Port Vila (Fig. 1). A schematic diagram of the protocol is presented in Figs. S4 & S5.

2.2. Chemicals

The chemicals used in this study are listed in Table 1.

2.3. Quantification of contamination and quality control

Prior to use, all glassware was cleaned using a laboratory detergent and rinsed using reverse osmosis (RO) water. All chemical solutions used in this study were previously filtered using a 47 mm diameter, 0.2 μm regenerated cellulose membrane filter. Field blanks (empty, pre-rinsed collecting jars) were also carried out to compensate for field collection contamination. For the water samples, the analysis from the high-pressure washer was found to be minimal and negligible. Contamination monitoring within the laboratory was carried out by using blank filters processed in the same way as environmental samples for each batch of samples processed. For the water samples, the number of microplastics quantified on blank filters were then deducted from the total number of microplastics quantified in the environmental samples to compensate for background contamination. Control filters were also used alongside the preparation of the sediment samples. For time efficiency, a procedural control was prepared with every batch of samples and subjected to the same preparation steps (Fig. S6). As a validation step, each type of filter membrane used in this study was spiked with a known number of plastic particles to investigate recovery rates using both a visual (digital imaging and microscopy) and an automated particle counting method developed at Cefas based on ArcGIS.

2.4. Quality control and polymer identification using Fourier-transform infrared spectroscopy

Polymer identification of particles was carried out using attenuated total reflection Fourier Transform infrared spectroscopy (ATR-FT-IR) using a Thermo Fisher Scientific Nicolet iS5 ATR-FTIR with OMNIC software (version 9.9.473) and by comparison of their IR spectra to polymer libraries (HR Nicolet Sampler Library, HR Spectra IR Demo & Hummel Polymer Sample Library). ATR-FT-IR has been shown to be a fast and effective tool for the identification of polymers of plastic marine debris, including those ingested by marine organisms (Jung et al., 2018). Due to size limitations of ATR-FT-IR, only particles above ~250 μm could be analysed, therefore, in the best case, 10% of microplastics were validated by ATR-FTIR. Spectra were collected in the

range 4000–650 cm^{-1} at a resolution of 4 cm^{-1}. Polymer identification was accepted based on a 70% or higher match against a polymer library. Quality control was carried out by analysing a polystyrene reference material before each batch (Table S1 and Fig. S7).

2.5. Microplastics in surface waters

Surface water samples were processed and analysed as described in Fig. S1. The main steps included visual inspection, separation and rinsing with RO water of the larger items (mesoplastics and plant materials) using a 5 mm sieve placed on top of a pre-rinse 2 L beaker for the removal of mesoplastics (particles between 5 and 10 mm in size). Extracted mesoplastics were manually removed from the sieve with tweezers and oven dried at 50 °C for 24 to 48 h before polymer characterisation with ATR-FTIR. The remaining fraction (i.e. rinse water) was then topped up with 400 mL of a 30% KOH:NaClO solution (Enders et al., 2017) and the sample was incubated at 40 °C for 24 h on a heating plate under constant agitation using a glass covered magnetic stirrer. Digests were then filtered using a 47 mm diameter Whatman GF/D filter with a 2.7 μm pore size and stained with Nile Red (Maes et al., 2017a). A summary of the method is shown in Fig. S1.

2.6. Microplastics in sediments

Collected sediment samples were homogenised and dried at 50 °C for a minimum of 3 days until a stable weight was achieved. Three subsamples of 5 g sediment were placed into three 50 mL polypropylene centrifuge tubes in a fume cupboard. Density separation was carried out by adding a 1.2 g mL^{-1} solution of saturated sodium chloride (NaCl). The saturated sodium chloride solution was previously filtered using a 47 mm diameter regenerated cellulose filter with a 0.2 μm pore size. Each tube was shaken by hand for 1 min and centrifugated at 3900 $\times g$ for 5 min. The supernatant of individual subsamples was transferred to a previously cleaned filtration unit and filtered using a 0.2 μm porosity Whatman cellulose nitrate membrane. The whole process was repeated two more times and the supernatants combined on the respective filters. Glass beakers, funnels and filters were rinsed with 100 mL RO water. Each filter was then carefully transferred to previously cleaned 100 mL glass beakers covered with glass lids for the alkaline and digested, using 30 mL a 30% KOH:NaClO solution (Enders et al., 2017). After 72 h, digests were filtered on a 47 mm diameter regenerated cellulose filter with a 0.2 μm pore size, rinsed through with 100 mL RO water and stained with Nile Red. The method is summarised in Fig. S2. For each sediment sample, a particle size analysis (PSA) was carried-out to relate abundance of microplastics and sediment type. For 2018 sediment samples, a sub-sample was kept for PSA analysis in a 120 mL collecting glass pot. The sample was kept in a freezer at −18 °C until ready for analysis. PSA was carried out using a modified Cefas' protocol from Mason (2011) for fast PSA screening based on wet splitting into silt/clay (< 63 μm), sand (including very fine gravel up to 4 mm) (63 μm – 4 mm) and gravel (> 4 mm) fractions only. The method is summarised in Fig. S3.

2.7. Microplastics in biota

For crabs and fish samples, the gastrointestinal tracts (GITs) were removed in the laboratory while respecting the integrity of the gut content. Each GIT was transferred to a 120 mL glass beaker, previously cleaned with RO water, and the wet weight of the tissue was recorded. Each sample was sonicated using a VWR ultrasonic cleaning bath for 15 min. and 5 mL of a 30% KOH:NaClO solution was added per gram of tissue wet weight. Each sample was then incubated at 40 °C for 3 days before filtration using a pre-rinsed Whatman GF/D filter (2.7 μm porosity). For larger fish, like the yellow fin tunas, the stomachs were cut open and the content rinsed using RO water through a 5 mm metal sieve on top of a pre-rinsed 1 L beaker to collect the rinse solution. Mesoplastics were manually removed from the filter, RO rinsed above the sieve and placed in an oven at 50 °C. The remaining solution was digested at 40 °C for 24 h with a 30% KOH:NaClO mixture at 2/3rd of the solution volume. After digestion, the solution was filtered using a Whatman 47 mm diameter GF/D filter (2.7 μm porosity) and stained with Nile Red. Identification of the extracted microplastics was carried out using the fluorescence tagging of polymers using Nile Red published elsewhere. A summary of the methods is shown in Figs. S4 and S5.

2.8. Statistical analysis

To study variation among and between groups, an ANOVA test was used to analyse the differences among group means, followed by Tukey HSD post hoc test as the multiple comparison procedure using R (RStudio Version 1.2.5019, RStudio, Inc.) with $\alpha = 0.05$. Both ANOVA and Tukey's test assume independence of samples, homogeneity of variance and normality of residuals (Zar, 1999). Normal distribution was tested with Shapiro-Wilk test and both ANOVA and Tukey's test assumed approximately similar population sizes, even though both tests seem to be relatively robust against deviations from the assumptions (Norwegian Environment Agency, 2018; Osborne, 2010; Zar, 1999). As an additional precaution step for any type I errors, a more conservative p value was selected ($p < 0.01$) where heterogeneous variances remained after transformation. Analysis was conducted as ANOVA is considered robust to such departure from normality where large data sets are employed (Underwood et al., 1997). Statistical outputs are shown in the supporting information.

3. Results

3.1. Microplastics in surface waters

The concentration of microplastics in surface waters in Vanuatu ranged from 9779 to 101,700 particles km^{-2} with a mean concentration of 51,144 items km^{-2}. This corresponded to a range of 0.09 to 0.57 items m^{-3} with a mean concentration of 0.28 items m^{-3} (Fig. 2). The concentration of floating microplastics was significantly higher ($p = 0.029$) for Vila Bay compared to Mele Bay, indicating a decrease in concentration gradient from coastal to more offshore sampling sites. Plastic particles in surface waters were composed of both *meso* and microplastics defined as particles above 5–10 mm and below 5 mm in size. A total of 17 mesoplastics were extracted from the water samples and analysed using ATR-FTIR (Fig. S8). All the mesoplastics were analysed, the main polymers identified were polyethylene (41%), followed by polystyrene (23%) and polypropylene (24%). 12% of the collected items above 5 mm were not successfully identified or were of biological origin. A total of 672 microplastics were recorded from surface water samples. A total of 42 items below 5 mm were analysed using ATR-FTIR corresponding to about 6% of the extracted particles. 98% were confirmed to be manmade polymers. The main polymers identified using ATR-FT-IR were polystyrene (PS) (72%), followed by polyethylene (PE) (14%), polypropylene (PP) (10%) and polyvinyl acetate ethylene (PEVA) (2%) (Fig. S9). 2% of the items below 5 mm were not successfully identified as plastics or were of biological nature.

3.2. Microplastics in sediments

Occurrence and abundance of microplastics in sediment were investigated for Vanuatu and the Solomon Islands (Fig. 1). Spatial and temporal variation in the occurrence and abundance of microplastics in sediment were investigated for Vanuatu for 2017 and 2018 (Figs. 3-5) while only sediments collected from 2017 were processed for Solomon Islands (Fig. 6).

3.2.1. Vanuatu

The concentration of microplastics ranged from 833 ± 333 to

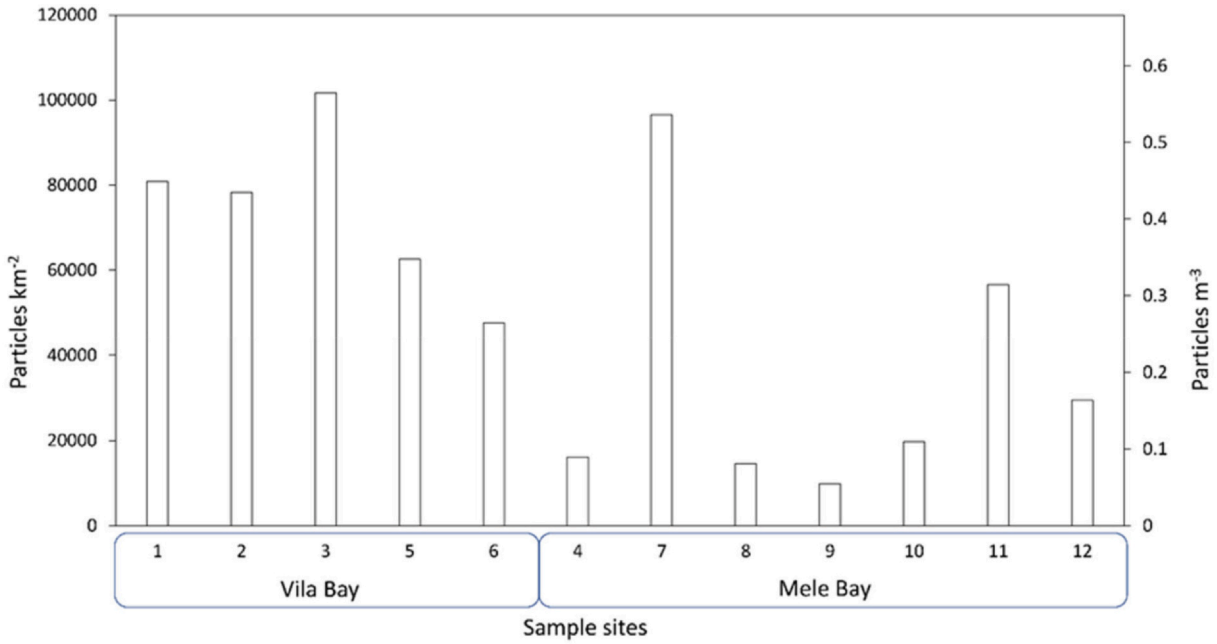

Fig. 2. (a) Number of items from surface water samples m^{-3} sampled water and Number of items km^{-2} sampled area ($n = 12$).

19,167 ± 5085 particles kg^{-1} dry weight sediment for samples collected for Vanuatu for 2017 and ranged from 333 ± 115 to 33,300 ± 7300 particles kg^{-1} dry weight sediment for 2018 (Figs. 3 & 4). The highest abundance of microplastics was recorded for sample 14 collected in 2017 corresponding to a grab sample from La Colle River with a concentration of 19,167 ± 5085 particles kg^{-1} dry weight sediment (Fig. 3). Concentration of microplastics in sediment for 2018 varied greatly according to locations and ranged from 333 ± 115 to 33,300 ± 7300 particles kg^{-1} dry weight sediment for sites 17 (Mele Bay) and 6 (Vila Bay) respectively. No significant differences were observed for the overlapping sites 2018 1 & 10 ($p = 0.148$) and 2018 6 & 9 ($p = 0.673$) indicating good reproducibility of the analytical technique. Interestingly, a gradual increase of the abundance of microplastics from 400 ± 200 to 6800 ± 3341 particles kg^{-1} dry weight sediment was observed from sampling sites 16 to 22 respectively and was strongly correlated with an increase in anthropogenic land use, shifting from high vegetation coverage to more densely populated areas including hotels and resorts with a closer proximity to road infrastructures (Fig. 4).

A temporal comparison was carried out for sediment samples collected from the same sites in Vanuatu for 2017 and 2018. Four sampling sites were consistent for the two consecutive years, namely 7/13, 9/12,

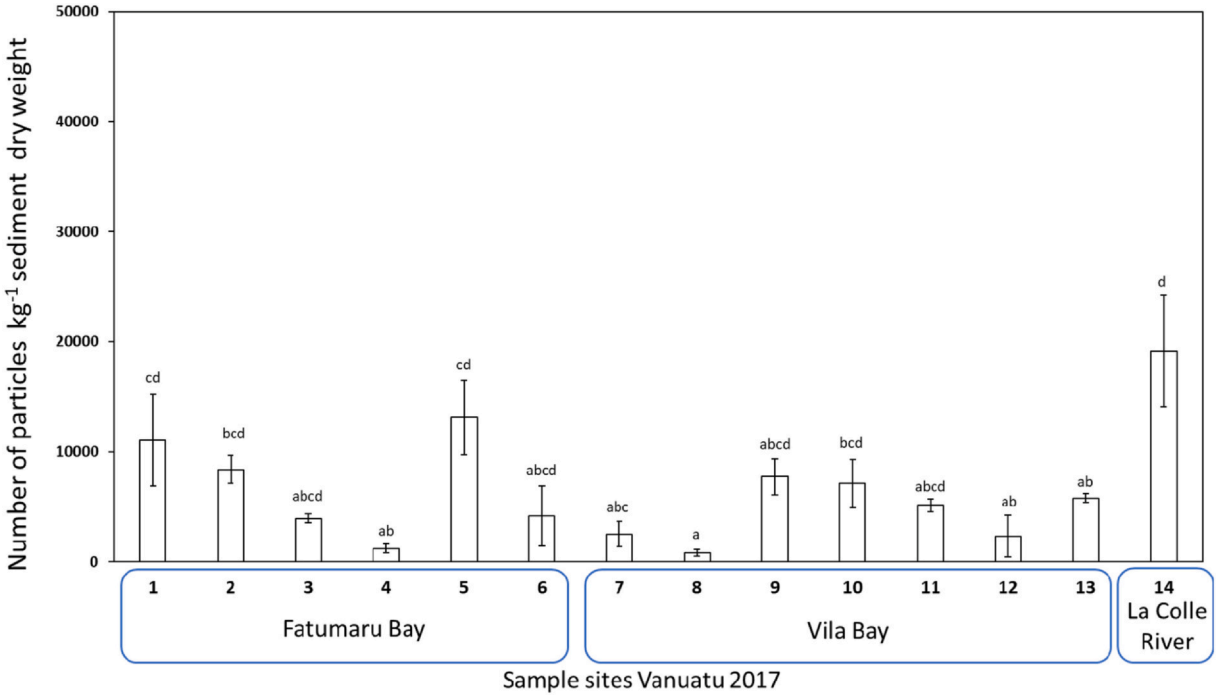

Fig. 3. Number of particles kg^{-1} dry weight sediment for Fatumaru Bay and Vila Bay collected in 2017 ($n = 3$, average ± SD) Letters refer to grouping following a one-way ANOVA using a Tukey (HSD) post hoc test. Means that do not share a letter are significantly different ($p = 0.01$).

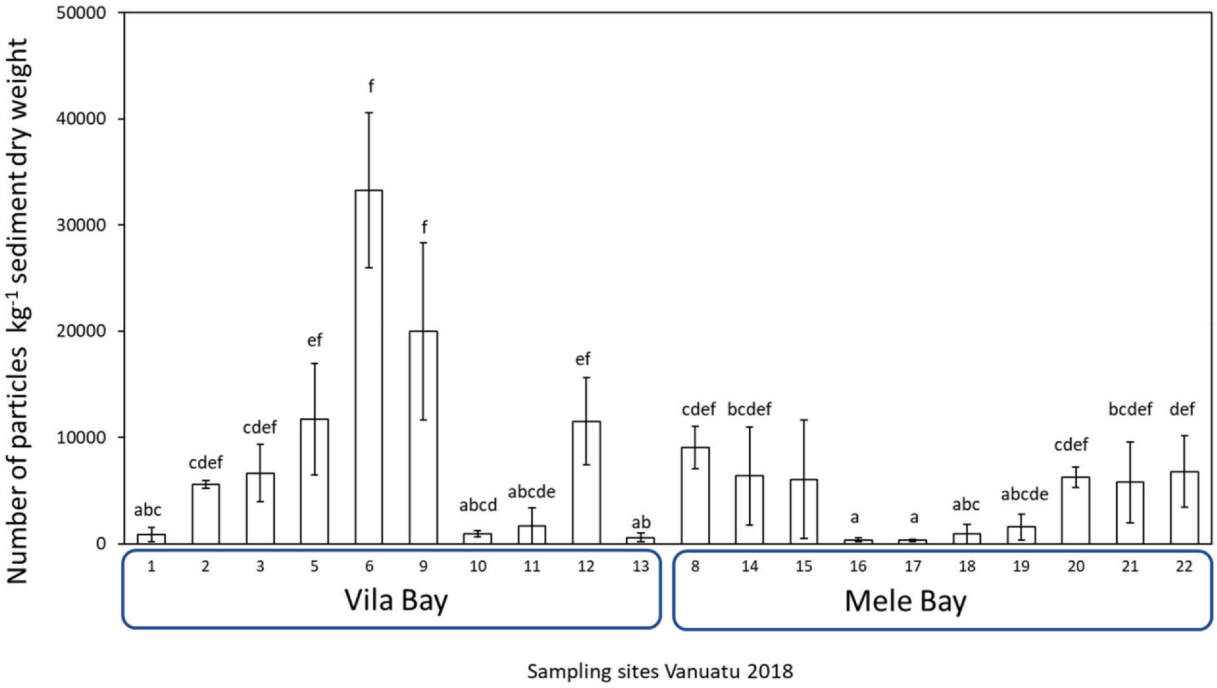

Fig. 4. Number of particles kg^{-1} dry weight sediment (n = 3, average ± SD) for sediment samples collected in 2018. Letters refer to grouping following a one-way ANOVA using a Tukey (HSD) post hoc test. Means that do not share a letter are significantly different ($p = 0.01$).

10/11 and 13/9 for 2017/2018 respectively (Fig. 5). Concentration of microplastics was significantly similar for the two consecutive years for sites 7/13 ($p = 0.0650$) and 9/12 ($p = 0.0787$). However, a significant increase ($p = 0.0404$) was recorded for sites 13/9 with an increase from 5800 ± 400 to 20,000 ± 8335 particles kg^{-1} dry weight sediment between 2017 and 2018.

Particle size analysis (PSA) was also carried out for sediment samples collected in Vanuatu for 2018 (Fig. S11). A scatterplot of both the number of particles kg^{-1} dry weight sediment and the standard deviation values against % gravel, % sand and % silt/clay were plotted (Figs. S12 and S13). Results indicated an increase in the concentration of microplastics with a higher percentage of silt/clay as compared to higher percentages of gravel and sand (Fig. S12). However, a higher variability between the replicates was also observed with an increase in standard deviations (SD) for samples with a higher percentage of silt/clay (Fig. S13).

3.2.2. Solomon Islands

The concentration of microplastics for the Solomon Islands ranged from 450 ± 180 to 15,167 ± 8661 particles kg^{-1} dry weight sediment for sites 1 and 2 respectively corresponding to Boneghi and Poha Rivers respectively (Figs. 1 and 6). The high concentration of microplastics recorded for Poha river (site 2 in Fig. 1) was not significantly different ($p > 0.01$) to sites 11 and 12, corresponding to sediment

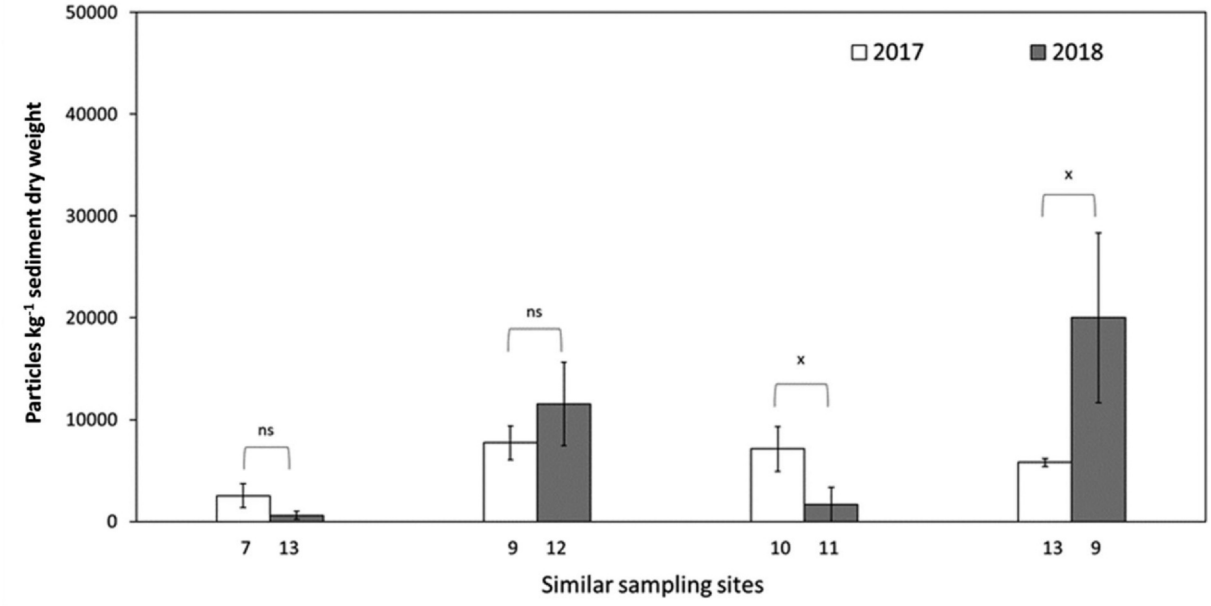

Fig. 5. Comparison between the number of particles kg^{-1} dry weight sediment for geographically similar sampling sites from 2017 and 2018 (n = 3, average ± SD). x significant ($p = 0.01$) and ns non-significant.

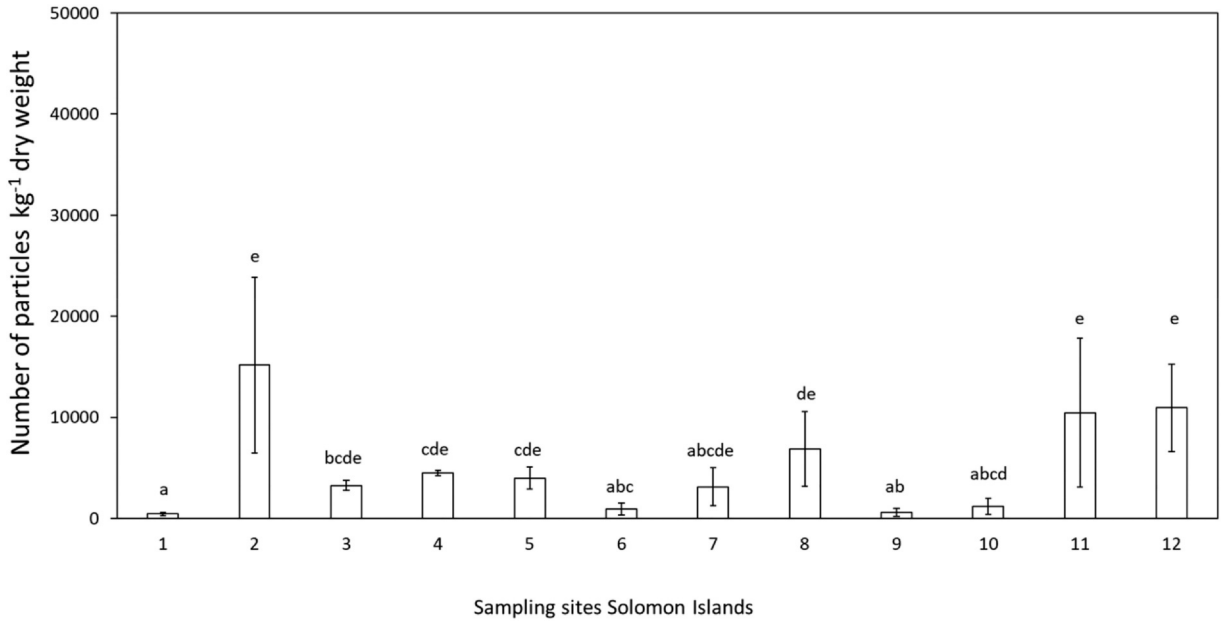

Fig. 6. Number of particles kg^{-1} dry weight sediment (n = 3, average ± SD) for riverine and costal/offshore samples collected in 2017. Letters refer to grouping following a one-way ANOVA using a Tukey (HSD) post hoc test. Means that do not share a letter are significantly different ($p = 0.01$).

samples from two different locations along the Lunga river with concentrations of 10,458 ± 7374 and 10,967 ± 4325 particles kg^{-1} dry weight sediment respectively (Figs. 1 and 6).

3.3. Microplastics in biota

Biota under investigation included the analysis of crabs ($n = 14$), fish ($n = 20$) and Yellow fin tuna ($n = 6$) from Vanuatu. No mesoplastics were found for all the biota samples under investigation. Of the crabs investigated, 57% contained at least one microplastic. From the 14 individuals, 24 items were found, with an average of 1.71 ± 2.27 items $individual^{-1}$. This corresponded to 0.021 ± 0.028 items g^{-1} wet weight tissues. Individual data are presented in supporting Table S9. Of the reef fish investigated, 35% contained microplastics in their guts. A total of 58 items (n = 20) were extracted from the tissues with an average of 2.9 ± 4.6 items $individual^{-1}$ corresponding to an average of 0.11 ± 0.19 items g^{-1} wet weight. Individual data are presented in supporting Table S10. From the Yellow Fin Tuna stomachs, 83% of those investigated contained at least one microplastic. A total of 26 items were found for the 6 individuals, with an average of 4.3 ± 5.13 items $individual^{-1}$. This corresponded to 0.02 ± 0.02 items g^{-1} wet weight tissues. Individual data are presented in supporting Table S11.

4. Discussion

4.1. Validity of method and limitations

The Nile red screening method for microplastics was applied for a fast and cost-effective assessment of the occurrence of microplastics in sediment, surface waters and biota for Vanuatu and Solomon Islands. This method has recently been used as a fast and cost-effective tool for the large-scale mapping of microplastics (Wang et al., 2018). Due to the geographic remoteness of the sites under investigation, only small, portable items were used during this study. This included the use of a portable ATR-FTIR for plastic particle validation and polymer identification with a validation step restricted to particles down to about 250 μm in size. No micro-FTIR or Raman spectroscopy was accessible on site and are commonly used for the identification of smaller size particles (Kniggendorf et al., 2019). Additional steps are therefore required for plastic confirmation, including visual observation using digital imaging and microscopy. The presence of false positives has been previously identified as a source of error when applying the Nile red screening method (Maes et al., 2017a; Kukkola et al., 2020). Visual observation of the fluorescent particles as well as the use of the automated based ArcGIS counting tool limited the occurrence of false positives with the screening of the lower fluorescence items. However, as a result, concentrations generated from this study were considered as indicative and comparative, rather than absolute (Kukkola et al., 2020).

4.2. Microplastics in surface waters

Data on the occurrence and abundance of floating microplastics in the South Pacific are limited and there is urgent need to fill this knowledge gap with more monitoring baseline data. The concentration of microplastics in surface waters from Vanuatu ranged from 9779 to 101,700 particles km^{-2} with a mean concentration of 51,144 items km^{-2}. This corresponded to a range of 0.09 to 0.57 items m^{-3} with a mean concentration of 0.28 items m^{-3}. Results suggested that the concentration of floating microplastics was significantly higher ($p = 0.027$) for Vila Bay as compared to Mele Bay (Fig. 2).

Graham et al. (2020) investigated how the coastal circulation around Port Vila can affect dispersal of pollutants released into the ocean. This modelling study focused on dispersal of buoyant plumes, representing sewage outflow sources through rivers and storm drains. Most plastic, including microplastic that enters the sea, originates from land-based sources and is transported by sewage and storm water into the marine environment (e.g., Jambeck et al., 2015; Lebreton et al., 2017). Other sources include ocean-based plastic inputs from maritime activities, including discarded and lost fishing items (Li, 2018). Pathways of coastal circulation can thus impact the distribution of marine litter. Graham et al. (2020) showed that pathways of dispersal and accumulation are primarily dependent on local bathymetry and coastline. From this modelling study, the highest concentrations in the Port Vila vicinity were typically found close to sources along the coast. However, enclosed regions around Port Vila (e.g. Vila Bay and Fatumaru Bay) were also shown to be likely “accumulation zones”, due to build-up from effluent and runoff outflows along the coast as well as transport pathways within the bay.

The highest concentration of floating microplastics observed here

was attributed to trawl transect 3 in Vila Bay alongside the area covering the boatyard, the cruise ship port and the recently completed Lapetasi International Wharf, where anthropogenic inputs are high and currents within the sheltered bay likely result in an accumulation zone for floating microplastics (Fig. 1c). The lowest concentration was found for transect 9 which was a transect out in Mele Bay. This region has fewer anthropogenic activities, and the exposed location likely leads to greater dispersion (by wind and currents) rather than accumulation.

It is worth noting that the model presented in Graham et al. (2020) only considers known outfall pipes and rivers as potential sources of pollutants, whereas marine litter sources likely come from a wider distribution of both known and unknown sources around the coastline. For example, the model did not consider outflows around the southern end of Vila Bay and the cruise ship port. Despite this, the model presented does show high concentrations in southern Vila Bay, suggesting that circulation patterns will contribute to high concentrations here, in addition to any local sources. Distribution of microplastics may be influenced by both large-scale as well as local circulation pathways. Large-scale ocean currents could also have the potential to transport plastics into the area. For offshore regions (such as Mele Bay), it is worth noting that the model neglects ocean sources, such as the loss of fishing gear, that may be more relevant here.

PS, PE and PP were the most abundant polymer types for both *meso* and microplastics present in water samples. Their proportion however varied according to their size fractions (meso vs microplastics) PE represented about 41% of analysed mesoplastics compared to only 14% for microplastics (Figs. S8 and S9). PS represented the main polymer type for microplastics with 72%. While polymer types were similar, their percentages differed from results in other locations around the Pacific Ocean. Reisser et al. (2013) found floating microplastics in Australian waters to consist of PE (68%), PP (31%) and polystyrene (1%). Ferreira et al. (2020) reported a prevalence of PE (39%) and PP (39%) for fibres in surface waters from Fiji while PS, latex and nitrile represented the main polymer types (17%) for floating fragments. Additionally, Pan et al. (2019) found microplastic pollution in the Northwestern Pacific Ocean to be roughly split between polyethylene (60%) and polypropylene (~40%). The differences in polymer composition might well reflect the different usage in those subregions of the Pacific.

The maximum reported abundance for this study (101,700 items km^{-2}) was substantially below the mean abundance of 334,271 km^{-2} reported for the North Pacific central gyre known as a convergence and accumulation zone for floating marine litter (Moore et al., 2001; Van Sebille et al., 2012). The range reported in this study was in the same order of magnitude as the one reported by Eriksen et al. (2013) for the South Pacific Ocean with a concentration ranging from 0 to 400,000 items km^{-2}. The abundances from this study were substantially higher than reported values for other locations including Australian waters (mean of 4258 items km^{-2}), the Gulf of Maine (1500 ± 200 items km^{-2}), Caribbean Sea (1400 ± 110 items km^{-2}) and Cape Province South-Africa (3640 items km^{-2}) indicating a much higher occurrence of floating plastic items for surface waters in Vanuatu (Morét-Ferguson et al., 2010; Reisser et al., 2015; Ryan, 1988). The average number of 0.28 items m^{-3} for this study was comparable with concentrations reported for different locations in Fiji with concentrations of 0.09 ± 0.02, 0.10 ± 0.02 and 0.24 ± 0.07 items m^{-3} for Lucala Bay, Suva Harbour and *vanua* Navakavu, respectively, showing consistency between studies for the same area (Ferreira et al., 2020). The average number of 0.28 items m^{-3} for this study was however substantially higher than the 0.12 items m^{-3} reported for the North Pacific by Goldstein et al. (2012) (Table 2).

4.3. Microplastics in sediment

4.3.1. Vanuatu

Seafloor sediments have been suggested as a likely final sink for microplastics in the marine environment (Van Cauwenberghe et al., 2013; Woodall et al., 2014; Näkki et al., 2019). Microplastic particles were detected in all sediment samples collected around Efate. The concentration of microplastics for Vanuatu ranged from 833 ± 333 to 19,167 ± 5085 particles kg^{-1} dry weight sediment in 2017 and from 333 ± 115 to 33,300 ± 7300 particles kg^{-1} dry weight sediment in 2018. Spatial variations between sediment samples were investigated and significantly higher number of particles were reported for Vila Bay as compared to Mele Bay ($p = 0.0496$) (Table S6). This suggested a reduction in the abundance of microplastics from coastal to more offshore sampling sites. The same trend was reported by Graca et al. (2017) when investigating the abundance of microplastics in sediments of the of the Southern Baltic Sea with a decrease in plastic abundance as distance from shore increases (Graca et al., 2017).

In 2017, the lowest and highest abundance was found for sites 8 (Vila Bay) and 14 (La Colle River) with concentrations of 833 ± 333 and 19,167 particles kg^{-1} dry weight sediment, respectively (Figs. 1 and 3). Higher concentrations of microplastics were also reported for Fatumaru Bay as compared to Vila Bay. This could be explained by the shallower bathymetry surrounding this area, restricting flow into and out of the region (Cefas, 2016; Graham et al., 2020). Sewage inputs into this system, combined with restricted water exchange, is also likely to lead to rapid biofouling of microplastic particles, loss of buoyancy and relatively rapid sedimentation (Kaiser et al., 2017; Michels et al., 2018; Wu et al., 2019). Graham et al. (2020) revealed this to be an accumulation region for sources further upstream in Vila Bay, transported along the coast. Similarly, Graham et al. (2020) also show higher accumulation in southern Vila Bay, that may help to explain the highest concentrations, observed around sites 6/9 in 2018 (Figs. 1 and 4).

The occurrence of microplastics varied according to the year of sampling. Comparison of the concentration of microplastics can be made for several sites for the two consecutive sampling years (2017 and 2018). This indicated a significant increase ($p > 0.01$) for sites 13/9 (2017/2018), corresponding to the site between Iririki Island and Port Vila Lapatasi International Wharf (Fig. 1). In 2017, the channel of the southern end of the Inner harbour was dredged to 5.5 m depth to facilitate entry to the Inner harbour of Port Vila for sailing boats and small sized yachts. Dredged materials and water were pumped into a reclamation bund (constructed lagoon) and dredged materials were left to settle. Excess water was then released back to the sea through a controlled weir gate causing fine sand and other particles, such as microplastics, to settle and accumulate in that area.

Other likely sources of microplastics in Vila Bay are city dust and urban runoffs (Kole et al., 2017). Graham et al. (2020) show that there is temporal variability in runoff in the region as a result of seasonality in local rainfall. Longer term trends in rainfall and runoff could affect the source of microplastics to the ocean environment. However, as microplastics may derive from either direct sources or from degradation of macroplastics over many years, the observed concentration in sediments may also vary due to either a long-term trend or a more recent change in source or circulation. The number of particles observed elsewhere varies greatly between studies and geographical areas reported in literature (Table 3). As previously stated, the absence of a globally accepted protocol is making comparison between datasets difficult. The highest concentration found from this study (33,000 particles kg^{-1} dry weight sediment) was substantially higher than the majority of concentrations reported for Europe and America, with the exception of the Central North Sea, with a maximum reported concentration of 31,000 particles kg^{-1} dry weight (Norwegian Environment Agency, 2018). However, the maximum concentration in microplastic reported from this study was in agreement with the maximum concentration reported for Jakarta Bay, with a concentration of 38,790 particles kg^{-1} dry wet (Manalu et al., 2017).

Particle size analysis (PSA) was also in agreement with previous studies that have shown that microplastic density was directly proportional to the content of silt/clay (Maes et al., 2017b; Kazmiruk et al., 2018; Wahyuningsih et al., 2018).

Table 2
Number of items in surface waters per cubic metre (m^3) and squared kilometre (km^2) reported in the literature for several locations.

Region	Mesh size/collection method	Number of items m^{-3}	Number of items km^{-2}	References
South Pacific Ocean, Vanuatu	330 μm	0.05–0.57 (mean: 0.28)	9779–101,700 (mean: 51,144)	This study
South Pacific Ocean, Fiji, Laucala Bay	125 μm	0.09 ± 0.02		(Ferreira et al., 2020)
South Pacific Ocean, Fiji, Vanua Navakavu	125 μm	0.24 ± 0.007		(Ferreira et al., 2020)
South Pacific Ocean, Fiji, Suva Harbour	125 μm	0.10 ± 0.02		(Ferreira et al., 2020)
North Western Pacific	330 μm		640–42,000 (mean: 10,000)	(Pan et al., 2019)
North Western Pacific- Kuroshio Current area	330 μm		170,000 - 470,000	(Yamashita and Tanimura, 2007)
Seta Inland Sea	Neuston net	0.39		(Isobe et al., 2015)
South Pacific Ocean - South Pacific subtropical gyre	331 μm		0–400,000	(Eriksen et al., 2013)
North Pacific central gyre	330 μm		31,982–969,777 (mean: 334,271)	(Moore et al., 2001)
Australian waters	330 μm		4256.4	(Reisser et al., 2013)
North Pacific		0.12		(Goldstein et al., 2012)
North Eastern Pacific Ocean	Saltwater intake system of the vessel		8000 - 9200	(Desforges et al., 2014)
East Asian seas around Japan	350 μm		1,700,000	(Isobe et al., 2017)
Western North Atlantic Ocean	330 μm		0–580,000	(Morét-Ferguson et al., 2010)
North Atlantic (accumulation rate)	150 μm	1.7		(Reisser et al., 2015)
Cape Basin, South Atlantic	Neuston sledge		1874.3	(Morris, 1980)
Cape Province, South Africa	Neuston net		3640	(Ryan, 1988)
Fernando de Noronha, Abrolhos and Trindade, Brazil	Manta net	0.03		(Do Sul et al., 2014)
Giona estuary, Brazil	Conical plankton net	0.26		(Lima et al., 2014)
Caribbean Sea	330 μm		1400 ± 110	(Morét-Ferguson et al., 2010)
Gulf of Maine	330 μm		1500 ± 200	(Morét-Ferguson et al., 2010)
Mediterranean	330 μm		0–900,000	(Collignon et al., 2012)
NW Mediterranean Sea - Gulf of Lion	780 and 330 μm		6000 - 1,000,000 (mean: 112,000)	(Schmidt et al., 2018)
Arabian Bay	300 μm		44,000 - 1500,000	(Abayomi et al., 2017)
Arctic Ocean	330 μm		28,000	(Lusher et al., 2015c)
East China Sea	333 μm	0.167 ± 0.138		(Zhao et al., 2014)

4.3.2. Solomon Islands

The occurrence and abundance of microplastics in sediments were investigated for rivers and coastal sites along the coast of Guadalcanal, to establish the relative importance of rivers as pathways for the transport of microplastics from land to sea. Microplastics were also detected in all sediment samples, indicating their widespread distribution for the area. The abundance of microplastics was significantly higher for sediments collected from Poha and Lunga Rivers with

Table 3
Number of particles kg^{-1} dry weight (dw) sediment reported in the literature for several locations.

Region	Sampling area	Characteristics	Numbers of particles kg^{-1} dw sediment	References
Oceania	South Pacific Ocean	Port Vila, Vanuatu	333–33,300	This study
	South Pacific Ocean	Honiara, Solomon Islands	450–15,167	This study
Oceania	South Pacific Ocean	Laucala Bay, Suva Harbour and *vanua* Navalaku, Fiji	19.8 ± 4.2	(Ferreira et al., 2020)
America	Canada	Baynes Sound and Lambert Channel, British Columbia	up to 25,000	(Kazmiruk et al., 2018)
	Canada	Intertidal, Halifax Harbour, Nova Scotia	2000 - 8000	(Mathalon and Hill, 2014)
	Canada	Shoreline	83–161.8	(Crichton et al., 2017)
	Canada	Canadian Lake Ontario nearshore, tributary and beach sediments	20–27,830	(Ballent et al., 2016)
Africa	Northern Tunisia	Lagoon-Channel of Bizerte	3000–18,000	(Abidli et al., 2017)
Arctic Ocean		Deep sea	42–6595	(Bergmann et al., 2017)
Asia	Tokyo	Tokyo Bay	1900	(Matsuguma et al., 2017)
	China	Beibu Gulf/Coastline of China Sea	5020–8720	(Qiu et al., 2015)
	Jakarta	Jakarta Bay	18,405 - 38,790	(Manalu et al., 2017)
	Jakarta	Mangrove area Pantai Indah Kapuk (PIK)	216.8–2218.4	(Manalu et al., 2017)
	Eastern Asia	Gulf of Thailand	100–1900	(Matsuguma et al., 2017)
Europe	Belgium	Continental Shelf	97.2–166.7	(Claessens et al., 2011)
	Barents Sea	Norwegian Continental Shelf	830–3900	(Norwegian Environment Agency, 2018)
	Central North Sea	Norwegian Continental Shelf	180–31,000	(Norwegian Environment Agency, 2018)
	Norway	Reference areas in the Norwegian coastal shelf	1–400	(Mareano, 2017)
	The Netherlands	Subtidal	100–3600	(Leslie et al., 2017)
	Italy	Venice Lagoon	672–2175	(Vianello et al., 2013)
	Italy	Lido di Dante (Beach)	1512	(Lots et al., 2017)
	Sweden	Subtidal	16–2590	(KIMO Sweden, 2007)
	Slovenia	Beach	170.4–177.8	(Laglbauer et al., 2014)
	UK	North Sea and English Channel	0–3146	(Maes et al., 2017b)
	Kachelotplate Island	Beach transects	0–62,100	(Liebezeit and Dubaish, 2012)
	Romania	Beach	100–5500	(Popa et al., 2014)
	Baltic Sea	Isle of Rügen	55.01–114.72	(Hengstmann et al., 2018)

Table 4
Number of items g^{-1} wet weight and individual^{-1} reported in the literature for several locations. Results from this study are included in grey for fish, crabs and Tuna samples.

Organism		Location	Number of items g^{-1} wet weight	Number of items individual^{-1}	Reference
Crabs					
Red claw crab	***Cardisoma carnifex***	**Efate, Vanuatu**	**0.021 ± 0.028**	**1.71 ± 2.27**	**This study**
Blue crab	*Callinectes sapidus*	Corpus Christi Bay, TX		**0.87**	(Waddell et al., n.d.)
European spider crab	*Maja squinado*	Celtic Sea		**1.39 ± 0.79**	(Welden et. al, 2018)
Snow crab	*Chionoecetes opilio*	Chukchi and Bering seas continental shelf		**0.04 – 1.67**	(Fang et al., 2018)
Fish					
Marine fish	***A. Pyroferus, A. triostegus, A. lineatus, A. auranticavus, S. oviceps, Chlorusus spp., B. viridescens, Carangidae spp.***	**Efate, Vanuatu**	**0.11 ± 0.19**	**2.9 ± 4.6**	**This study**
	Mugil spp., Siganus spp., Lutjanus spp., Lethrinus spp. and Chanos chanos	**Laucala Bay, *vanua* Navakavu and Suava Harbour Fiji**		**5.5 ± 9.4**	(Ferreira et al., 2020)
Whiting	*S. Sihama*	Western Pacific Oceans	0.25	1.5	(Abbasi et al., 2018)
Greater lizardfish	S. tumbil	Western Pacific Oceans	0.37	2.8	(Abbasi et al., 2018)
Pelagic and demersal fish		English channer		1.9 ± 0.10	(Lusher et al., 2013)
Adriatic fish mullet		Adriatic Sea, Italy		1 – 1.78	(Avio et al., 2015)
Large pelagic fish	*Xiphias gladius, Thunnus; thynnus and Thunnus alalunga*	Mediterranean Sea		4 - 16	(Romeo et al., 2015)
Demersal fish		Spanish Atlantic and Mediterranean coasts		1.56 ± 0.5	(Bellas et al., 2016)
Pelagic and demersal fish		North and Baltic Sea		0.03 ± 0.18	(Rummel et al., 2016)
Sunfish bluegill and Longear	*Lepomis Macrochirus & Lepomis megalotis*	Brazos River Basin, Central Texas, USA		10.1 – 13.9	(Peters and Bratton, 2016)
Demersal & pelagic fish		Northeast Atlantic around Scotland		1.8 ± 1.7	(Murphy et al., 2017)
Flying fish	*C. rapanouiensis*	South Pacific coastal waters around Easter Island		1.0 ± 0.0	(Chagnon et al., 2018)
Commercial fish		Mondego estuary in Portugal		1.67 ± 0.27	(Bessa et al., 2018)
Sardines and Anchovy	*Sardina pilchardus and Engraulis encrasicolus*	Spanish Mediterranean coast		0 - 3	(Compa et al., 2018)

(*continued on next page*)

Table 4 (*continued*)

Tuna					
Yellow Fin Tuna	*T. albacares*	**South Pacific Ocean Vanuatu**	**0.02 ± 0.02**	**4.3 ± 5.1**	**This study**
Yellow Fin Tuna	*T. albacares*	South Pacific Ocean Coastal waters of the Eastern Island		5.0	(Chagnon et al., 2018)
Marine worms					
	A. marina	French_Belgian_Dutch coastline	1.2 ± 2.8		(Van Cauwenberghe et al., 2015)
Sea snails					
	C. abbreviatus	Western Pacific Oceans	0.16	2.9	(Abbasi et al., 2018)
Prawns					
	P. indicus	Western Pacific Oceans	0.59	2.3	(Abbasi et al., 2018)
Mussels					
	M. edulis	French_Belgian_Dutch coastline	0.2 ± 0.3		(Van Cauwenberghe et al., 2015)
	M. edulis	French Atlantic coast		0.61 ± 0.56	(Phuong et al., 2018)
	M. edulis	China	0.9 – 4.6		(Li et al., 2016)
	M. edulis	UK	0.7 – 2.9	1.1 – 6.4	(Li et al., 2018)
	M. modiolus	UK	0.086 ± 0.031	3.5 ± 1.29	(Catarino et al., 2017)
	M. edulis	Norway	0 – 24.45	0 – 14.67	(Lusher et al., 2017)
	M. edulis	Germany	0.36 ± 0.07		(Van Cauwenberghe and Janssen, 2014)
	M. edulis	Canada		34 - 178	(Mathalon and Hill, 2014)
	Mytilus edulis, Perna viridis	Coastal waters of China	1.52 – 5.36		(Qu et al., 2018)
	Bivalves	China		4.3 – 57.2	(Li et al., 2015)
Oysters					
	Saccostrea cucullata	China	1.5 – 7.2	1.4 – 7.0	(Li et al., 2018)
	C. gigas	Brittany, France	0.47 ± 0.16		(Van Cauwenberghe and Janssen, 2014)
	C. gigas	French Atlantic coast		2.10 ± 1.71	(Phuong et al., 2018)
Whales					
True's beaked whale	*Mesoplodon mirus*	Northern Ireland		2.95	(Lusher et al., 2015b)
Sea Birds					
Northern Fulmars	*Fulmarus glacialis*	Pacific and Grays Harbor counties, Washington		13.3	(Terepocki et al., 2017)
Sooty Shearwaters	*Ardenna grisea*	Pacific and Grays Harbor counties, Washington		19.5	(Terepocki et al., 2017)

Note: Lusher et al. (2015b).

concentrations of 15,167 ± 8661 and 10,967 ± 4325 particles kg^{-1} dried weight respectively ($p < 0.05$ and $p = 0.015$) (Fig. 6). In comparison, marine coastal sampling sites ranged from 600 ± 400 to 6867 ± 3700 particles kg^{-1} dry weight for sites 9 and 8 respectively (Fig. 1). The results from this study were in agreement with other studies, finding rivers to be a major pathway for ocean plastic waste (Lebreton et al., 2017). Regarding the relative importance of local rivers to transport microplastics from land sources to the marine environment the main sources followed the order: Poha River > Lunga River > Mataniko River > White River > Tanavasa River > Boneghe

River.

Investigation into the abundance of microplastics in sediments identified specific accumulation zones for microplastic contamination. Specific data on sources of plastic pollution for the Solomons is limited. The Poha River site was observed to have a high concentration of microplastics in sediments. However, unlike the Lunga and Mataniko rivers, Poha river is not located near large urban areas and likely sources did not include major commercial activities related to tourism or from large populated areas. The Poha river is heavily impacted by logging activities including cutting, skidding, on-site processing and loading of logs onto trucks. These industrial activities, in addition to the absence of beach or river cleaning activities in the area could contribute to the higher abundance of microplastics.

Lunga River enters the sea on the northern coast of Guadalcanal, near Lunga Point (Fig. 1), and showed high concentrations of microplastics in sediments at both its mouth and upstream. Lunga River is the longest river in the Solomons with a catchment area of 377 km^2 (UNESCO, 2012). Being close to the centre of population, it is also expected for this river to be heavily impacted by a series of anthropogenic activities such as littering, industrial and commercial based activities including fishing and shipping activities from the nearby Leroy Wharf Port.

The Mataniko River site was identified as a likely source for microplastic contamination for the area, found in central Honiara. However, the abundance of plastic particles was less than half the abundance recorded for Poha and Lunga Rivers with concentrations of 4000 ± 1114, 15,167 ± 8661 and 10,967 ± 4325 particles kg^{-1} dry weight sediment, respectively. The potential sources of plastic pollution identified for the area were varied and included commercial activities (e.g. restaurants and catering facilities) as well as communities located close to the river using it as a waste disposal system for household litter.

Relatively high concentrations of microplastics were also recorded for coastal sediment samples, with a maximum concentration of 6867 ± 3700 particles kg^{-1} dry wet sediment observed at point 8 in central Honiara. Likely sources of contamination included densely populated areas, commercial activities (e.g. restaurants and catering facilities) and surface runoffs. Urban areas have been identified as important sources of microplastics with potential for transport to the marine environment (GESAMP, 2016).

Another likely source of microplastics is surface runoff from roads transporting plastic and microplastic items from land to the sea. City dust and urban runoffs include paint polymers and synthetic rubber particles from car tyres. (Kole et al., 2017). Significantly higher abundances of microplastics detected for sites 7 and 8 coincide with the proximity to a main road. Interestingly, there was no buildings or constructions in this particular site that could act as physical barriers for the trapping of plastic particles and the runoff generated from road activities. Graham et al. (2020) showed this portion of coastline was likely to have higher concentrations of pollutant due to an increased number of sewage outfall pipes in the area.

In contrast to Vanuatu, the Honiara coastline is more exposed, with depth increasing rapidly with a short distance offshore. The exposed coastline may partly account for the lower maximum values observed here, when compared with those in Vila Bay. Graham et al. (2020) showed that the highest concentrations of pollutants around Honiara were found close to source, with no obvious regions of accumulation as shown for the enclosed bays on Efate. Concentrations reported for Vanuatu and the Solomon Islands were much higher than the reported average concentration of 19.8 ± 4.2 particles kg^{-1} dry weight sediment for Fiji (Ferreira et al., 2020). It is worth noting that Ferreira et al. (2020) collected particles over a 300 μm sieve compared to a 0.2 μm porosity filter in the present study. As a result, the smaller size fraction (< 300 μm) was under reported in their study. Previous studies have showed a prevalence of particles below 300 μm in size in sediment samples (Strand et al., 2019).

4.4. Microplastics in biota

This study confirmed the occurrence of microplastics in GITs of terrestrial crabs, coral reef fish and pelagic fish from the South Pacific. Microplastics have been reported for a wide range of marine organisms ranging from marine worms to seabirds with varying concentrations according to the species and locations (Table 4). Microplastics were present in 57% of the land crabs under investigation with an average concentration of 0.021 ± 0.028 microplastics g^{-1} wet weight corresponding to an average of 1.71 ± 2.27 microplastics $individual^{-1}$. Data on the occurrence of microplastics in crabs is relatively scarce to allow for global comparisons. While our study included the terrestrial crab *Cardisoma carnifex*, reported data in terrestrial crab were compared to marine species as an indication of potential for human uptake from crabs as food sources. Data from this study were comparable with the studies of Welden et al. (2018) who reported an average concentration of 1.39 ± 0.79 items $individual^{-1}$ for the European spider crab (*Maja squinado*) collected from the Celtic Sea and the study from Fang et al. (2018) for the Snow crab (*Chionoecetes opilio*) from the Chukchi and Bering seas continental shelf with a concentration of microplastics ranging from 0.04 to 1.67 items $individual^{-1}$ (Fang et al., 2018; Welden et al., 2018).

Our study also indicated that microplastics were present in 38% of the gastrointestinal tract of reef fish collected in Vanuatu with an average concentration of 0.11 ± 0.19 items g^{-1} wet weight corresponding to an average of 2.9 ± 4.6 items $individual^{-1}$. Microplastics have been reported for fish species for several locations (Table 4). Average reported concentrations for microplastics in fish from Vanuatu (2.9 ± 4.6 items $individual^{-1}$) was within the range of reported concentrations of microplastics for the South Pacific with reported values of between 1.0 ± 0.0 particle $individual^{-1}$ (Chagnon et al., 2018) to 5.5 ± 9.4 items $individuals^{-1}$ (Ferreira et al., 2020). Ferreira et al. (2020) however reported a much higher incidence of microplastics in fish collected in Fiji with a 68% occurrence compared to 38% for Vanuatu. Previous studies reported occurrences of microplastics in 37% of the pelagic and demersal fish investigated from the English Channel and 11% of the *meso*pelagic fish from the North Atlantic sea (Lusher et al., 2013; Lusher et al., 2015a). Reported concentrations from this study were however comparable to concentrations of microplastics in fish from the Western Pacific Ocean with an average abundance of 1.5 items $individual^{-1}$ for the whiting *S. sihama* and 2.8 items $individual^{-1}$ for the Greater lizardfish (*S. tumbil*) (Abbasi et al., 2018).

The abundance of microplastics in Yellow Fin Tuna from this study (4.3 ± 5.13 items $individual^{-1}$) was also in agreement with the abundance of microplastics reported by Chagnon et al. (2018) for the Yellow Fin Tuna (*T. albacares*) from coastal waters off Easter Island with an average of 5 items $individual^{-1}$ (Table 4).

Data indicated that microplastics were present for a range of food items from terrestrial to marine sources at different concentrations for crabs, fish and Yellow Fin Tuna with average number of 1.7 ± 2.27, 2.9 ± 4.6 and 4.3 ± 5.13 items $individual^{-1}$. The occurrence of microplastics in biota has caused several concerns ranging from its effects on biodiversity and populations to potential risks to food safety and human health. Microplastics can cause a physical effect to biota following exposure with the disruption of normal physiological processes (Besseling et al., 2013; Cole et al., 2015, 2019) with unclear impacts at a population and ecosystem level. Microplastics have also been shown to sorb hydrophobic organic compounds (HOCs) and metals from the surrounding water with potential for transfer to organism following ingestion (Bakir et al., 2014; Brennecke et al., 2016). However recent studies have shown that transfer of sorbed HOCs from plastics to biota following ingestion would be negligible compared to other routes of uptake (i.e. direct uptake from water and via ingestion of contaminated preys) (Bakir et al., 2016). It is however still unclear whether plastic additives, often added at high concentrations, could have a chemical impact on biota following ingestion (Koelmans et al.,

2016). Regarding human health implications, it has been suggested by the FAO that the transfer of sorbed co-contaminants and additives from the ingestion of plastic particles would be negligible due to the low dietary exposure to such contaminants (Lusher et al., 2017).

5. Conclusion

This study described for the first time the occurrence and abundance of microplastics for Vanuatu and the Solomons Islands. A protocol based on the fluorescence tagging of polymers using Nile red coupled with digital imaging and an automated data processing tool was applied for the fast screening of microplastics in surface waters and biota for Vanuatu and in sediments for both countries. Results have indicated the widespread occurrence of microplastics which were detected in all surface water and sediment samples for both Vanuatu and the Solomon Islands. This method was in alignment with previous recommendations that the Nile red method is a promising approach for the large-scale mapping of microplastics in a monitoring context. Recommendations include further monitoring for macro, meso and microplastics in Vanuatu and the Solomon Island for the characterisation of additional accumulation zones and associated activities having substantial impacts in the marine environment. A better understanding of the types, quantities and sources of marine litter will help to define better regulation and legislation. As most plastics are entering the marine environment from land-based sources, it would be strongly recommended to include more monitoring for rivers including water catchment areas to identify sources. Identification of the main sources would allow the prioritisation of remediation procedures. More monitoring is needed on the occurrence and abundance of microplastics in biota with the sampling of a larger number of individuals. The investigation of a wide range of species from specific areas would allow the characterisation of an appropriate bioindicator for microplastic contamination for the South Pacific area. Addressing the knowledge gaps in monitoring data would allow the creation of “exposure risk maps” for the area with the mapping of accumulation zones for microplastics and more sensitive areas (e.g. marine protected areas, fishing zones). Exposure risk maps would be the first step in developing robust and reliable risk assessment frameworks based on environmental relevant conditions such as concentrations, size and type of plastics items in the natural environment.

CRediT authorship contribution statement

A.B. co-analysed the samples, co-analysed the data, conducted the statistical analysis, and wrote the manuscript. M.D. co-analysed the samples and co-analysed the data, T.W. developed the method, N.V.H. carried out sample collection and produced the figures. R.A. contributed in sampling design and arranging in-country laboratory facilities, S.A. contributed in sampling design and sample collection. J.G. co-wrote the paper and T.M. designed the experiments, supervised the research and commented on earlier versions of the manuscript.

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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