Assessing plastic size distribution and quantity on a remote island in the South Pacific

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ABSTRACT

Plastics are an environmental threat; however, their fate once in the pelagic environment is poorly known. We compare results from assessments of floating plastics in the South Pacific Ocean with accumulated beach plastics from Henderson Island. We also compare accumulated plastic mass on Henderson during 2015 and 2019 and investigate the presence of nanoplastics. There were differences between the size classes of beach and pelagic plastics, and an increase in microplastics (0.33–5 mm) on the beach between 2015-2019. Micro- and nanoplastics were found at all sites (mean ± SE: 1960 ± 356 pieces/kg dw). Across the whole beach this translates to >4 billion plastic particles in the upper 5 cm. This is concerning, particularly given Henderson is uninhabited and distant from urban centres (~2350 km from Pape'ete, French Polynesia). The vast number of small particles on Henderson may make nearshore filter feeders susceptible to ingestion and subsequent detrimental impacts.

Keywords: Anthropogenic debris; Henderson Island; marine debris; microplastic; nanoparticle; plastic pollution

1. Introduction

Life without plastic is unimaginable in today's society, yet the mass production of this fossil-fuel derived material began just 70 years ago (PlasticsEurope, 2013, 2019). In 2018, 359 million tonnes of plastic were produced globally and current rates of production double approximately every 10 years (PlasticsEurope, 2019). Plastics have myriad uses due to product longevity and malleability, leading to numerous societal improvements (e.g., medical treatments and sanitation; Carney Almroth and Eggert, 2019). However, the high demand for plastics and inefficient waste management has resulted in plastic permeating throughout the natural environment, from the deepest ocean trench and highest mountain top, to the Earth's atmosphere (Chiba et al., 2018; Napper et al., 2020; Zhang et al., 2020). The lightweight and durable properties of most plastics allow the items to persist for decades and be transported long distances by wind and currents (Barnes et al., 2009; Eriksen et al., 2014; Lebreton et al., 2018) with 19-23 million metric tonnes of plastic waste entering the marine environment in 2016 alone (Borrelle et al., 2020).

Plastics pose a significant risk to marine ecosystems and biota through entanglement, transportation of invasive species or contaminants(Barnes et al., 2009). The ingestion of plastic can damage the digestive tract, reduce growth rates, and cause starvation which can sometimes lead to death of wildlife (Lavers et al., 2014; Wilcox et al., 2018). Plastics can also act as a vector for chemicals that are absorbed from surrounding waters, or incorporated during the manufacturing process (Lithner et al., 2011; Turner, 2017). Wildlife and humans are therefore at risk of increased exposure to chemicals through direct consumption of plastic, or indirect consumption of contaminated organisms (Lehner et al., 2019; Oliveira and Almeida, 2019).

The abundance of plastics in the environment and diversity of shapes, sizes, and properties has highlighted a need for common definitions and structured collection and reporting methods that will enable comparisons among studies (Cowger et al., 2020; Hartmann et al., 2019; Provencher et al., 2017; Serra-Goncalves et al., 2019). Standardising plastic definitions will also enhance global policy objectives in attempt to reduce further environmental damage from these products (Provencher et al., 2020; Serra-Goncalves et al., 2019). Plastic size categories are commonly reported as macroplastic (>20 mm), mesoplastic (5–20 mm), and microplastic (<5 mm; Arthur et al., 2009; Barnes et al., 2009). More recently, microplastics have been further categorised into large microplastics (1–5 mm), small microplastics (<1 mm), and nanoplastics (1–1000 nm; Galgani et al., 2013; Koelmans et al., 2015). However, there remains substantial variation in how these categories are delimited, and consensus is yet to be reached (Hartmann et al., 2019; Provencher et al., 2017). For example, although the upper limit of microplastics is often 5 mm, the lower limit, and sizing terminology, varies considerably, and nanoplastics variously refer to items <100 or <1000 nm (Tables 1 and 2; Gigault et al., 2018; Mendoza et al., 2018; Wyer et al., 2020).

Regardless of the size of items, the fate of individual plastics entering the marine environment is uncertain (Cressey, 2016). Ocean gyres, formed by strong winds and circular ocean currents, can accumulate significant quantities of floating plastic debris (Eriksen et al., 2014; Law et al., 2010). Though plastic inputs into the ocean are increasing annually, floating surface plastic may account for 1% of all plastic found at sea (Cózar et al., 2014; Jambeck et al., 2015; Law et al., 2010; Van Sebille et al., 2015). The absence of such high quantities of plastics from the ocean's surface could result from a range of factors, including fragmentation, where plastics break up (i.e., through wave or UV exposure) into smaller particles and thus lose buoyancy over time (Andrady, 2011; Egger et al., 2020). Particles may also be ingested by organisms, which through deposition (e.g., contaminated fecal matter) may remove plastics from the marine environment (Bourdages et al., 2020; Van Franeker and Law, 2015). Plastics may also attract and accumulate organisms (biological fouling) which could lead to the sinking of particles and resurfacing of low density plastic particles through de-fouling (Andrady, 2011). Finally, plastics of all sizes can wash ashore in large quantities, often becoming buried in beach sediments over time (Cózar et al., 2014; Lavers and Bond, 2017). While there is growing interest in the fate of this "missing" debris, few studies provide robust data or descriptions that help elucidate debris dynamics (Cózar et al., 2014; Egger et al., 2020; Pohl et al., 2020).

Cózar et al. (2014) suggested plastic particles <1 mm to be the least abundant size class in surface waters, and that this may result from size selective removal processes. A more recent study found that >50% of plastics found deeper within the water column were <1.5 mm, and the vast majority of plastics found below the ocean’s surface were <5 mm (Egger et al., 2020). Research into size removal processes (i.e., aquatic sediments, marine organism ingestion/disturbances, and beached debris) could lead to a greater understanding of plastic behaviour and fate once in the marine environment (Cózar et al., 2014; Egger et al., 2020). This is particularly true for nanoparticles which are poorly documented in both terrestrial and aquatic environments (Koelmans et al., 2015), including the South Pacific region where data are especially limited (Bakir et al., 2020; Cózar et al., 2014).

Some of the highest quantities of beached plastics were found washed ashore on Henderson Island in the Pitcairn Islands in 2015 (Lavers and Bond, 2017). This remote and uninhabited island is in close proximity to the western boundary of the South Pacific gyre, which is thought to influence plastic accumulation on Henderson’s beaches (Lavers and Bond, 2017). A lack of direct human influence has meant beach-washed debris items on Henderson Island can act as an indicator of marine pollution in the broader region (Lavers and Bond, 2017). Here we quantify marine debris present on Henderson Island to (1) compare the size distribution of plastics recorded on land (Henderson Island) and in the pelagic surface waters of the South Pacific (Cózar et al., 2014), (2) determine whether the density of macro- and microplastics on Henderson have increased since 2015 and 2019, and (3) investigate the abundance of micro- and nanoplastics are present in beach sediments.

2. Materials and methods

2.1. Study location

Henderson Island (24.36°S, 128.30°W) is a 43 km^2 raised coral island and UNESCO World Heritage Site in the South Pacific Ocean and is one of four islands in the Pitcairn Island group (Fig.1). The closest human population (Pitcairn Island; ~45 permanent residents) is 155 km west. Samples were collected on East Beach, a 2.25 km sand beach inside the fringing reef which faces the predominant currents in the region (Irving and Dawson, 2012), from 13 June–24 July 2015 (Lavers and Bond, 2017) and 7–21 June 2019.

2.2 Field data collection

We defined size classes based on common mesh sieve sizes: macroplastics >5 mm, and microplastics 0.33-5 mm (Andrady, 2011; Eriksen et al., 2014; Masura et al., 2015). Microplastics were further divided into two categories to allow comparison across existing studies: large microplastics (1–5 mm) and small microplastics (0.33–1 mm). Plastics from beach core samples (see below) ranged from 50 µm to 5 mm, and are collectively referred to as microplastics and nanoplastics, as no size distribution was available.

2.2.1. Beach surface macro- and microplastics

For macro-and microplastics, collection methods during 2019 followed Lavers and Bond (2017), with a few modifications (fewer transects/quadrats, and buried debris not recorded) due to time and logistical constraints. Macro- and microplastics, including plastic, glass, wood, and metal items, were sampled in five 1 × 1 m quadrats placed randomly along the strandline of East Beach (Fig. 2A). Sediment and debris from the top 1 cm of each quadrat were collected as a bulk sample using a flat trowel. Due to permit requirements, initial sorting was completed *in situ* using our smallest sieve (0.33 mm) in order to separate the bulk sample into debris items (retained for further analysis) from large organic materials (e.g., leaves, coral, algae). Organic materials and bulk sediment were discarded. Pre-sorted debris samples were then transferred to individually labelled, sterile plastic sealed bags for transport to the lab. In addition to the quadrats, macroplastics were also recorded in a 50 m transect covering the entire beach width (7 m) placed randomly. All debris was counted and sorted by size and type with small items (micro- and nanoplastics) weighed using an electronic balance (Mettler Toledo PB303-S) and macroplastics using a spring balance (± 1 g) following Lavers and Bond (2017).

2.2.2 Nanoplastic beach core samples

Following Maes et al. (2017b), a 50 g bulk sediment sample was collected from each of the five quadrats at a depth of 5 cm using a 50 mm diameter stainless steel corer. The samples were placed in individually labelled glass jars. Prior to use, each jar was washed in hot water to remove any residue, then thoroughly rinsed with reverse osmosis water before being sealed with aluminium foil (also rinsed with reverse osmosis water) to prevent contamination.

2.3. Lab data collection

2.3.1. Beach surface macro- and microplastic

Care was taken to prevent contamination of samples with plastic particles from outside sources: we used Milli-Q water (Vermaire et al., 2017), work benches and tools were wiped clean with sterile paper and 70% ethanol between samples. We sourced lab consumables made of glass and researchers wore lab coats and clothing made of non-synthetic materials whenever possible (Torre et al., 2016).

For each of the five quadrats, the pre-sorted samples were placed in stacked Tyler mesh sieves (0.33, 1.00, and 4.75 mm) to sort the samples into three size categories. Microplastics that could not be confirmed by the naked eye (typically <1 mm) were identified using a dissecting microscope (40× power; Lusher et al., 2020). Plastic particles were sorted by size based on particle width, allowing for long thin particles to be counted as a size class smaller than the total length (Lusher et al., 2020).

As plastics <1 mm require further identification methods, we used density separation via floatation in sodium chloride (NaCl) for samples 0.33–1 mm in order to identify and remove any residual non-plastic items (e.g., coral; Galgani et al., 2013; Hidalgo-Ruz et al., 2012; Masura et al., 2015). A saturated NaCl solution (1.2 g cm^{-3}) was prepared using Milli-Q water and stirred with a stainless steel stirrer. Each plastic sample was then introduced to their allocated glass beakers and stirred manually for 2 minutes and left to settle for one hour. Complete samples were then extracted from the solution, poured through a 0.33 mm metal sieve, and rinsed with Milli-Q water to remove any residual salt solution. Samples were then transferred to glass jars, covered in aluminium foil, and left to air dry for 72 hours. Settled particles (e.g., rock, coral, sand, and plastics) from each beaker were examined under dissection microscope to remove any identifiable plastics which were of a higher density than the salt solution. One beaker containing only NaCl solution was placed alongside each sample to act as a blank. Blanks were covered with aluminium foil, with a total of two blanks used throughout the extraction process; both blanks had no plastics detected within our size categories.

Once dry, a fine natural fibre, non-shedding brush was used to spread particles from each sample evenly across a sheet of velvet fabric. Samples were then photographed using a Canon 70D camera and 100 mm lens. Plastics were then counted using ImageJ 1.53a software (Schneider et al., 2012) (Fig. 2B).

2.3.3. Micro- and nanoplastic beach core samples

Bulk sediment samples from the stainless corer were sent to the Centre for Environment, Fisheries and Aquaculture Science (Cefas) Laboratories in Lowestoft, United Kingdom for analysis and identification of micro- and nanoplastics using density separation and spectroscopy. All chemical solutions used were previously filtered using a 0.2 µm regenerated cellulose membrane. Sediment samples were transferred to 100 mL glass beakers and the lids were replaced with 15 cm Whatman 509 filter papers, held into place using small metal wires. Each sample was dried in a drying cabinet below 50°C for 72 h. Five grams of the sediment were weighed into three 50 mL polypropylene centrifuge tubes in a biological safety cabinet with ventilation.

Density separation was carried out by using a 1.37 g/mL^{-1} solution of zinc chloride. Approximately 35 mL of zinc chloride solution was added to each of the centrifuge tube and 37 mL was added to an empty tube as a control. Each tube was centrifuged at 3900 × G for 5 min. Each supernatant 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 same filter. Each filter was then carefully transferred to previously cleaned 100 mL glass beakers with a glass lid for the alkaline digest process. Samples were incubated using a 30% potassium hydroxide:sodium hypochlorite solution to remove organic matter at 40°C on a rotary shaker at 120 rpm prior to filtration using a 0.2 µm regenerated cellulose membrane. Following filtration, the filters were stained using Nile red (Maes et al., 2017a). Automatic counting was also validated by visual inspection and visual validation. The lower detection limit for micro- and nanoplastics was 50 µm.

2.4. Statistical analyses

All statistical analyses were conducted in R 4.0.2 (R Core Team, 2020). Values are stated as mean ± standard deviation (SD), unless otherwise specified, and results were considered significant when $p < 0.05$.

2.4.1. Pelagic and beach surface comparison

To test whether there was a difference between the size distribution in pelagic debris items as recorded by Cózar et al. (2014) in the South Pacific Ocean and macro- and microplastic debris from Each Beach, Henderson Island in 2019, we pooled the 28 size classes by Cózar et al. (2014) so they were consistent with the categories used in this study, with the lowest size corresponding to the smallest mesh sieve, and the upper size being the width of the quadrat (0.33–1 mm, 1–5 mm, 5 to

1000 mm). Although a 4.75 mm sieve was used on Henderson Island, we pooled sizes to 5 mm for ease of comparability. We calculated the mean number of each of these size classes for comparison. A log-linear model was used to assess the size distributions of plastics across both pelagic and beach sources.

2.4.2. Henderson Island macro- and microplastic comparison

To compare our findings with the 2015 survey of Henderson Island, we used the surface debris count data from three of the transects completed in 2015 (total area surveyed 280 m^2) (see Fig. 2 in Lavers and Bond, 2017). Differences between the accumulated mass of debris in 2015 and 2019 were examined using a general linear model (GLM). Changes in the mass of microplastics were not examined due to the small sampling area in 2019.

2.4.3. Micro- and nanoplastic beach core samples

The mean number and SD of micro- and nanoplastic items was determined per kg of dry sediment of each individual quadrat. The total means of all five quadrates were calculated along with the standard error (SE).

We extrapolated the total number of plastics in the top 5 cm of East Beach by using 10,000 bootstraps drawn from the five quadrat values, and multiplying this by the standard density of beach sand (2.65 g/cm^3; Manger, 1963) and dimensions of East Beach (2.25 km × 7 m; Lavers & Bond 2017), and present the mean value with 95% confidence intervals.

3. Results

3.1. Comparing pelagic and beach plastics in the South Pacific

A total of 17,501 plastic items were collected from five 1 × 1 m quadrats on Henderson Island during 2019, with 13.1% macroplastics (n = 2291), 51.2% large microplastics (n = 8974), and 35.6% small microplastics (n = 6236; Fig. 3). The distribution was similar to that modelled by Cózar et al. (2014)

with 13.2% of pelagic debris also being macroplastic, however the proportion of large microplastics (67.8%) and small microplastics (18.8%) differed between pelagic and beach environments. Overall, there was a significant difference between the size distributions of pelagic and beach-based plastic items ($\chi^2 = 380.23$, df = 2, $p < 0.001$).

3.2. Beach plastic abundance in 2015 and 2019

There was no significant difference between the macroplastic density between 2015 (458.326 ± 438.671 g/m^2) and 2019 (408.326 ± 591.864 g/m^2; $F_{1,17} = 0.016$, $p = 0.902$). The density of microplastics, however, was an order of magnitude greater in 2019 (23.391 ± 11.7 g/m^2) than 2015 (2.128 ± 3.285 g/m^2; $F_{1,17} = 39.351$, $p < 0.001$; Table 3).

3.3. Micro- and nanoplastics in beach core samples

Micro- and nanoplastic particles were identified from all three replicates of the beach core sediment samples taken from all five quadrats on East Beach in 2019. Total micro- and nanoplastics extracted ranged between 1467 ± 643 and 3333 ± 1976 pieces/kg dw (mean ± SE: 1960 ± 356 pieces/kg dw; Table 3).

Across the whole of East Beach, we estimated 4.089 billion micro- and nanoplastic particles (95% confidence interval: 3.088-5.565 billion particles) in the top 5 cm of sand.

4. Discussion

Our findings demonstrate the value of remote islands as marine pollution indicators. We found differences in plastic size distributions across beach and open water environments, and a significant increase in microplastic density between 2015-2019. While our study documented micro- and nanoplastics at a depth of 5 cm in beach sediments, previous work has shown that these plastic sizes are likely ubiquitous throughout ecosystems (La Daana et al., 2020; Napper et al., 2020). Thus, many current estimates of plastic abundance are likely underrepresented as nanoplastics are typically unaccounted for (Lindeque et al., 2020; Peng et al., 2020).

4.1. Plastic size distribution

The size distribution of macroplastics was similar across both Henderson Island and particles modelled for the open surface waters of the South Pacific Ocean by Cózar et al. (2014). However, the size distribution of large (pelagic: 67.8%; beach: 51.2%) and small microplastics (pelagic: 18.8%; beach: 35.6%) were significantly different across the two systems. Ocean surface waters transport buoyant plastics to beaches, and in our case, items deposited on Henderson Island may initially reflect the size of debris in surface waters, but over time, the size distribution may change due to greater elemental exposure in a beached environment (e.g., mechanical abrasion; Andrady, 2011; Koelmans et al., 2015; Song et al., 2017). The differences identified between both large and small microplastics from beach and surface waters could also result from different removal processes or exposures that may be occurring between the two locations (Cózar et al., 2014). For example, small microplastics from surface waters can lose buoyancy and sink either as a result of biofouling or polymer type (Egger et al., 2020), which could remove these particles from the ocean's surface where most debris sampling has taken place to date. In contrast, beached plastics can fragment into small microplastics, where they may remain relatively *in situ* (Andrady, 2011), allowing them to accumulate and be more readily recorded in sampling efforts than at sea. Few studies have described the size distribution of beached plastics, however, in southern China, microplastics between 0.33–5 mm accounted for 98% of plastic items located on the beach surface, compared to 2% macroplastics from 5–10 mm (Fok et al., 2017). Overall, our understanding of the mechanisms influencing the fate of plastic in the ocean and in beach sediments are relatively limited (Cózar et al., 2014; Egger et al., 2020; Fok et al., 2017).

4.2. Density of plastic on East Beach, Henderson Island

We found no significant difference in the density (g/m^2) of macroplastics on East Beach between 2015 and 2019 (Lavers and Bond, 2017). In contrast, the quantity of microplastics differed significantly and has increased on East Beach by an order of magnitude. The similarity in macroplastic density between the two years is surprising considering the increased volume of plastic waste entering the marine environment every year (Borrelle et al., 2020; Jambeck et al., 2015). There may be multiple explanations for these findings. Firstly, the beaches of Henderson Island are surrounded by a fringing reef. As plastic items pass over the reef, wave exposure may cause mechanical abrasion which could induce rapid fragmentation of macroplastics into smaller particles (Andrady, 2011; Lavers and Bond, 2017). Large storm events may bury beached macroplastics or allow greater numbers of items to wash ashore, thereby influencing the quantity of macroplastic present on the beach in 2019 (Song et al., 2017). To our knowledge, comparable microplastic mass data are not yet available for remote islands, although future modelling predicts that marine microplastic density will double in the next 10 years (Isobe et al., 2019). Our findings are concerning given that Henderson Island is uninhabited and distant from urban centres. Considering these factors, and the variability around plastic density estimates, continued and consistent sampling effort will be required to establish trends in debris accumulation on Henderson Island (Serra-Goncalves et al., 2019).

4.3. Micro- and nanoplastics in beach sediments

We estimate each 1 × 1 m quadrat contained 3333 ± 1973 micro- and nanoplastic particles/kg dw at 5 cm depth, and more than 4 billion micro- and nanoplastics across the whole of East Beach. Few studies have quantified nanoplastics within the marine environment because of the challenges of identifying tiny particles, which requires specialised, and often expensive equipment (Hidalgo-Ruz et al., 2012; Provencher et al., 2020). Although there is a lack of nanoplastic data to compare our findings with, one study from Norderney, an island off Germany's mainland, found >213 nano fibres/kg dw of beach sediment (Dekiff et al., 2014). A comparative study from Solomon Island coastal sediment found microplastics (defined in the study as <5 mm) between 600-6,867 items per kg/dw, however, these finding were in close proximity to densely populated coastal zones and other anthropogenic influences (Bakir et al., 2020).

Buried beach plastics can have less environmental exposures (e.g., wind) than surface plastics, and can therefore act as reliable plastic pollution indicators (Tavares et al., 2020). In our study, micro- and nanoplastics at a depth of 5 cm were the only buried items sampled during 2019. However, Lavers and Bond (2017) found 68% of all macro- and microplastics on Henderson Island during the 2015 survey were buried within the sediment at <10 cm. The number, type or size of plastic particles in sediments >10 cm depth is unknown for Henderson Island, but items are likely distributed throughout the beach substrate as continuous movement of sand via waves and bioturbation may cause gradual movement of plastics (Song et al., 2017; Tavares et al., 2020). For example, plastics are found up to 2 m deep in beach sediment off São Paulo, Brazil, although with a likely reduction in plastic abundance at increased depth (Turra et al., 2014). Overall, the high number of micro- and nanoplastics detected in beach sediments on Henderson Island, thousands of miles from the nearest urban centre, highlights the difficulty, and likely underrepresentation, of current debris estimates for beaches, worldwide.

4.4. Plastic: risks to Henderson Island's wildlife

Our understanding of the effects of plastic on Henderson's wildlife is limited. Seabirds on Henderson ingest small quantities of plastic (0–16% of 93 samples in three species; Imber et al., 1995), but follow-up data do not exist. Henderson's strawberry hermit crabs (*Coenobita perlatus*) experience high rates of entrapment and mortality in beached plastics, however impacts on the population are not known (Lavers et al., 2020). Beached plastics can pose a hazard to nesting turtles, such as increased entanglement for laying females and hatchlings (Aguilera et al., 2018; Gündoğdu et al., 2019). Similar data are not available for Henderson, but nesting green turtles *Chelonia mydas*

(Brooke, 1995) likely experience some of these effects due to the accumulated debris, with reports of adult turtles entangled in ghost nets (Lavers and Bond, 2017). Debris accumulated on Henderson's beaches can also alter the sediment temperature profile (Lavers et al., in review). Increasing sediment temperature, often associated with climate change, has been attributed to female biased sex ratio in sea turtles (Jensen et al., 2018; Tomillo et al., 2014), therefore debris accumulation on Henderson may significantly impact the nesting turtle population.

While the effect of nanoplastics on Henderson's ecology are unknown, the exposure of marine and freshwater organisms to nanoplastics (mostly within controlled laboratory settings) is associated with a range of harmful effects (Haegerbaeumer et al., 2019). They can impact growth and reproduction of algae and small benthic invertebrates (Besseling et al., 2014; Sendra et al., 2019). Marine benthic filter feeders (e.g., sea cucumbers) could ingest or resuspend nanoplastics embedded within sandy sediments, making them available to other marine organisms (Renzi et al., 2018). Thus, as Henderson is home to myriad marine species, including filter feeding organisms (Irving and Dawson, 2012), and heavily contaminated with nanoplastics there is a need to better understand the implications of nanoplastic exposure on this unique and vulnerable ecosystem.

4.5. Study limitations

Spatial and temporal trends are an important aspect of ecological research, including quantifying and predicting plastic accumulation patterns over time. However, gathering the necessary, robust datasets can be extremely challenging due to the variability in sampling and reporting methods, difficulty associated with working in remote areas (i.e., dangerous weather conditions; Browne et al., 2015), and seasonality in debris accumulation (Maharani et al., 2020; Ríos et al., 2018). For example, seasonal variability was identified in the remote island beaches of the Azores archipelago, likely a result of temporal wind exposure, and human influences (i.e., tourists more prevalent in summer leading to more beach clean-ups in this season)(Ríos et al., 2018). However, we attempted to minimise the effects of seasonality (i.e., south-easterly trade wind patterns) by completing our surveys in June 2015 and June 2019. A handful of recent review papers made recommendations in an attempt to standardise reporting, however inconsistencies remain and most recommendations are yet to be fully adopted. Based on a subset of recent review papers (Table 1), nine different definitions for micro and nano-plastics were recommended (Frias and Nash, 2019; Gigault et al., 2018; Hanvey et al., 2017; Hartmann et al., 2019; Mendoza et al., 2018; Provencher et al., 2017). Due in part to this uncertainty, ten different size classes were used for microplastics in subsequent scientific articles (Table 2). Overall, despite an abundance of review papers that have synthesised information and made recommendations for reporting, there is little evidence this is having a positive effect on standardizing how subsequent data are collected. Thus greater efforts are needed to improve communication within the plastic research community to reach a consensus on methods and categorisations, as this will ultimately impact the comparability of future research.

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6. References

Abidli, S., Toumi, H., Lahbib, Y., El Menif, N.T., 2017. The first evaluation of microplastics in sediments from the complex lagoon-channel of Bizerte (Northern Tunisia). Water, Air, & Soil Pollution 228, 262-272.

Aguilera, M., Medina-Suárez, M., Pinós, J., Liria-Loza, A., Benejam, L., 2018. Marine debris as a barrier: Assessing the impacts on sea turtle hatchlings on their way to the ocean. Marine Pollution Bulletin 137, 481-487.
Andrady, A.L., 2011. Microplastics in the marine environment. Marine Pollution Bulletin 62, 1596-1605.
Arthur, C., Baker, J., Bamford, H., 2009. International research workshop on the occurrence, effects, and fate of microplastic marine debris, NOAA Technical Memorandum NOS-OR&R-30 (2008).
Bakir, A., Desender, M., Wilkinson, T., Van Hoytema, N., Amos, R., Airahui, S., Graham, J., Maes, T., 2020. Occurrence and abundance of meso and microplastics in sediment, surface waters, and marine biota from the South Pacific region. Marine Pollution Bulletin 160, 111572.
Barnes, D.K., Galgani, F., Thompson, R.C., Barlaz, M., 2009. Accumulation and fragmentation of plastic debris in global environments. Philosophical Transactions of the Royal Society B: Biological Sciences 364, 1985-1998.
Besseling, E., Wang, B., Lürling, M., Koelmans, A.A., 2014. Nanoplastic affects growth of *S. obliquus* and reproduction of *D. magna*. Environ. Sci. Tec. 48, 12336-12343.
Borrelle, S.B., Ringma, J., Law, K.L., Monnahan, C.C., Lebreton, L., McGivern, A., Murphy, E., Jambeck, J., Leonard, G.H., Hilleary, M.A., 2020. Predicted growth in plastic waste exceeds efforts to mitigate plastic pollution. Science 369, 1515-1518.
Bourdages, M.P., Provencher, J.F., Baak, J.E., Mallory, M.L., Vermaire, J.C., 2020. Breeding seabirds as vectors of microplastics from sea to land: Evidence from colonies in Arctic Canada. Science of The Total Environment, 142808.
Brooke, M.d.L., 1995. Seasonality and numbers of green turtles Chelonia mydas nesting on the Pitcairn Islands. Biological Journal of the Linnean Society 56, 325-327.
Browne, M.A., Chapman, M.G., Thompson, R.C., Amaral Zettler, L.A., Jambeck, J., Mallos, N.J., 2015. Spatial and temporal patterns of stranded intertidal marine debris: Is there a picture of global change? Environ. Sci. Tec. 49, 7082-7094.
Carney Almroth, B., Eggert, H., 2019. Marine plastic pollution: Sources, impacts, and policy issues. Review of Environmental Economics and Policy 13, 317-326.
Chiba, S., Saito, H., Fletcher, R., Yogi, T., Kayo, M., Miyagi, S., Ogido, M., Fujikura, K., 2018. Human footprint in the abyss: 30 year records of deep-sea plastic debris. Mar. Pol. 96, 204-212.
Chubarenko, I., Esiukova, E., Bagaev, A., Bagaeva, M., Grave, A., 2018. Three-dimensional distribution of anthropogenic microparticles in the body of sandy beaches. Science of the Total Environment 628, 1340-1351.
Chubarenko, I., Esiukova, E., Khatmullina, L., Lobchuk, O., Grave, A., Kileso, A., Haseler, M., 2020. From macro to micro, from patchy to uniform: Analyzing plastic contamination along and across a sandy tide-less coast. Marine Pollution Bulletin 156, 111198.
Cowger, W., Booth, A., Hamilton, B., Primpke, S., Munno, K., Lusher, A., Dehaut, A., Vaz, V.P., Liboiron, M., Devriese, L.I., 2020. EXPRESS: Reporting guidelines to increase the reproducibility and comparability of research on microplastics. Applied Spectroscopy, 1066-1077.
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., 2014. Plastic debris in the open ocean. Proceedings of the National Academy of Sciences 111, 10239-10244.
Cressey, D., 2016. The plastic ocean. Nature 536, 263-265.
De-la-Torre, G.E., Dioses-Salinas, D.C., Castro, J.M., Antay, R., Fernández, N.Y., Espinoza-Morriberón, D., Saldaña-Serrano, M., 2020. Abundance and distribution of microplastics on sandy beaches of Lima, Peru. Marine Pollution Bulletin 151, 110877.
De Ruijter, V.N., Milou, A., Costa, V., 2019. Assessment of microplastics distribution and stratification in the shallow marine sediments of Samos Island, Eastern Mediterranean Sea, Greece. Mediterranean Marine Science 20, 736-744.
Dekiff, J.H., Remy, D., Klasmeier, J., Fries, E., 2014. Occurrence and spatial distribution of microplastics in sediments from Norderney. Environmental Pollution 186, 248-256.

Delvalle de Borrero, D., Fábrega Duque, J., Olmos, J., Garcés-Ordóñez, O., Amaral, S.S.G., Vezzone, M., de Sá Felizardo, J.P., Meigikos dos Anjos, R., 2020. Distribution of plastic debris in the Pacific and Caribbean beaches of Panama. Air, Soil and Water Research 13.
Di, M., Wang, J., 2018. Microplastics in surface waters and sediments of the Three Gorges Reservoir, China. Science of the Total Environment 616, 1620-1627.
Dobaradaran, S., Schmidt, T.C., Nabipour, I., Khajeahmadi, N., Tajbakhsh, S., Saeedi, R., Mohammadi, M.J., Keshtkar, M., Khorsand, M., Ghasemi, F.F., 2018. Characterization of plastic debris and association of metals with microplastics in coastline sediment along the Persian Gulf. Waste Management 78, 649-658.
Dong, Z., Zhu, L., Zhang, W., Huang, R., Lv, X., Jing, X., Yang, Z., Wang, J., Qiu, Y., 2019. Role of surface functionalities of nanoplastics on their transport in seawater-saturated sea sand. Environmental Pollution 255, 113177.
Egessa, R., Nankabirwa, A., Basooma, R., Nabwire, R., 2020. Occurrence, distribution and size relationships of plastic debris along shores and sediment of Northern Lake Victoria. Environmental Pollution 257, 113442.
Egger, M., Sulu-Gambari, F., Lebreton, L., 2020. First evidence of plastic fallout from the North Pacific Garbage Patch. Scientific Reports 10, 1-10.
Eriksen, M., Lebreton, L.C., Carson, H.S., Thiel, M., Moore, C.J., Borerro, J.C., Galgani, F., Ryan, P.G., Reisser, J., 2014. Plastic pollution in the world's oceans: More than 5 trillion plastic pieces weighing over 250,000 tons afloat at sea. PloS one 9, e111913.
Fok, L., Cheung, P.K., Tang, G., Li, W.C., 2017. Size distribution of stranded small plastic debris on the coast of Guangdong, South China. Environmental Pollution 220, 407-412.
Frias, J., Nash, R., 2019. Microplastics: Finding a consensus on the definition. Marine Pollution Bulletin 138, 145-147.
Galgani, F., Hanke, G., Werner, S., Oosterbaan, L., Nilsson, P., Fleet, D., Kinsey, S., Thompson, R., Van Franeker, J., Vlachogianni, T., 2013. Guidance on monitoring of marine litter in European Seas. MSFD Technical Subgroup on Marine Litter (TSG-ML). JRC Technical Report. European Commission, Joint Research Centre. EUR83985.
Ghaffari, S., Bakhtiari, A.R., Ghasempouri, S.M., Nasrolahi, A., 2019. The influence of human activity and morphological characteristics of beaches on plastic debris distribution along the Caspian Sea as a closed water body. Environ. Sci. Poll. 26, 25712-25724.
Gigault, J., Ter Halle, A., Baudrimont, M., Pascal, P.-Y., Gauffre, F., Phi, T.-L., El Hadri, H., Grassl, B., Reynaud, S., 2018. Current opinion: What is a nanoplastic? Environmental pollution 235, 1030-1034.
Gündoğdu, S., Yeşilyurt, İ.N., Erbaş, C., 2019. Potential interaction between plastic litter and green turtle *Chelonia mydas* during nesting in an extremely polluted beach. Marine Pollution Bulletin 140, 138-145.
Haegerbaeumer, A., Mueller, M.-T., Fueser, H., Traunspurger, W., 2019. Impacts of micro-and nano-sized plastic particles on benthic invertebrates: A literature review and gap analysis. Frontiers in Environmental Science 7, 17.
Hanvey, J.S., Lewis, P.J., Lavers, J.L., Crosbie, N.D., Pozo, K., Clarke, B.O., 2017. A review of analytical techniques for quantifying microplastics in sediments. Analytical Methods 9, 1369-1383.
Hartmann, N.B., Huffer, T., Thompson, R.C., Hassellov, M., Verschoor, A., Daugaard, A.E., Rist, S., Karlsson, T., Brennholt, N., Cole, M., Herrling, M.P., Hess, M.C., Ivleva, N.P., Lusher, A.L., Wagner, M., 2019. Are we speaking the same language? Recommendations for a definition and categorization framework for plastic debris. Environ. Sci. Tec. 53, 1039-1047.
Hidalgo-Ruz, V., Gutow, L., Thompson, R.C., Thiel, M., 2012. Microplastics in the marine environment: a review of the methods used for identification and quantification. Environ. Sci. Tec. 46, 3060-3075.
Imber, M., Jolly, J., Brooke, M.d.L., 1995. Food of three sympatric gadfly petrels (*Pterodroma spp.*) breeding on the Pitcairn Islands. Biological Journal of the Linnean Society 56, 233-240.

Imhof, H.K., Sigl, R., Brauer, E., Feyl, S., Giesemann, P., Klink, S., Leupolz, K., Löder, M.G., Löschel, L.A., Missun, J., 2017. Spatial and temporal variation of macro-, meso-and microplastic abundance on a remote coral island of the Maldives, Indian Ocean. Marine Pollution Bulletin 116, 340-347.
Irving, R., Dawson, T., 2012. The marine environment of the Pitcairn Islands. Dundee University Press, United Kingdom.
Isobe, A., Iwasaki, S., Uchida, K., Tokai, T., 2019. Abundance of non-conservative microplastics in the upper ocean from 1957 to 2066. Nature Communications 10, 1-13.
Jambeck, J.R., Geyer, R., Wilcox, C., Siegler, T.R., Perryman, M., Andrady, A., Narayan, R., Law, K.L., 2015. Plastic waste inputs from land into the ocean. Science 347, 768-771.
Jensen, M.P., Allen, C.D., Eguchi, T., Bell, I.P., LaCasella, E.L., Hilton, W.A., Hof, C.A., Dutton, P.H., 2018. Environmental warming and feminization of one of the largest sea turtle populations in the world. Current Biology 28, 154-159.
Karthik, R., Robin, R., Purvaja, R., Ganguly, D., Anandavelu, I., Raghuraman, R., Hariharan, G., Ramakrishna, A., Ramesh, R., 2018. Microplastics along the beaches of southeast coast of India. Science of The Total Environment 645, 1388-1399.
Koelmans, A.A., Besseling, E., Shim, W.J., 2015. Nanoplastics in the aquatic environment. Critical review, Marine anthropogenic litter. Springer, Cham, pp. 325-340.
La Daana, K.K., Gardfeldt, K., Krumpen, T., Thompson, R.C., O'Connor, I., 2020. Microplastics in sea ice and seawater beneath ice floes from the Arctic ocean. Scientific Reports 10, 1-11.
Lavers, J.L., Auty, J., Bond, A.L., in review. Plastic debris increases circadian temperature extremes in beach sediments. Proceedings of the National Academy of Sciences of the United States of America.
Lavers, J.L., Bond, A.L., 2017. Exceptional and rapid accumulation of anthropogenic debris on one of the world's most remote and pristine islands. Proceedings of the National Academy of Sciences of the United States of America 114, 6052-6055.
Lavers, J.L., Bond, A.L., Hutton, I., 2014. Plastic ingestion by Flesh-footed Shearwaters (*Puffinus carneipes*): Implications for fledgling body condition and the accumulation of plastic-derived chemicals. Environmental Pollution 187, 124-129.
Lavers, J.L., Sharp, P.B., Stuckenbrock, S., Bond, A.L., 2020. Entrapment in plastic debris endangers hermit crabs. Journal of Hazardous Materials 387, 121703.
Law, K.L., Morét-Ferguson, S., Maximenko, N.A., Proskurowski, G., Peacock, E.E., Hafner, J., Reddy, C.M., 2010. Plastic accumulation in the North Atlantic Subtropical Gyre. Science 329, 1185-1188.
Lebreton, L., Slat, B., Ferrari, F., Sainte-Rose, B., Aitken, J., Marthouse, R., Hajbane, S., Cunsolo, S., Schwarz, A., Levivier, A., 2018. Evidence that the Great Pacific Garbage Patch is rapidly accumulating plastic. Scientific Reports 8, 1-15.
Lee, J., Lee, J., Hong, S., Hong, S.H., Shim, W.J., Eo, S., 2017. Characteristics of meso-sized plastic marine debris on 20 beaches in Korea. Marine Pollution Bulletin 123, 92-96.
Lehner, R., Weder, C., Petri-Fink, A., Rothen-Rutishauser, B., 2019. Emergence of nanoplastic in the environment and possible impact on human health. Environ. Sci. Tec. 53, 1748-1765.
Lindeque, P.K., Cole, M., Coppock, R.L., Lewis, C.N., Miller, R.Z., Watts, A.J., Wilson-McNeal, A., Wright, S.L., Galloway, T.S., 2020. Are we underestimating microplastic abundance in the marine environment? A comparison of microplastic capture with nets of different mesh-size. Environmental Pollution, 114721.
Lithner, D., Larsson, Å., Dave, G., 2011. Environmental and health hazard ranking and assessment of plastic polymers based on chemical composition. Science of the total environment 409, 3309-3324.
Lots, F.A., Behrens, P., Vijver, M.G., Horton, A.A., Bosker, T., 2017. A large-scale investigation of microplastic contamination: Abundance and characteristics of microplastics in European beach sediment. Marine Pollution Bulletin 123, 219-226.
Lusher, A., Bråte, I.L.N., Munno, K., Hurley, R., Welden, N.A., 2020. Is it or isn't it: The importance of visual classification in microplastic characterization. Applied Spectroscopy 74, 1139-1153.

Maes, T., Jessop, R., Wellner, N., Haupt, K., Mayes, A.G., 2017a. A rapid-screening approach to detect and quantify microplastics based on fluorescent tagging with Nile Red. Scientific Reports 7, 44501.
Maes, T., Van der Meulen, M.D., Devriese, L.I., Leslie, H.A., Huvet, A., Frère, L., Robbens, J., Vethaak, A.D., 2017b. Microplastics baseline surveys at the water surface and in sediments of the North-East Atlantic. Frontiers in Marine Science 4, 135.
Maharani, A., Yuliadi, L., Syamsuddin, M., Ismail, M., 2020. Seasonal effect on the spatial distribution of macro debris in Tunda Island, Banten, IOP Conference Series: Earth and Environmental Science. IOP Publishing, p. 012006.
Manger, G.E., 1963. Porosity and bulk density of sedimentary rocks. Geological Survey Bulletin 1144, 1-55.
Masura, J., Baker, J.E., Foster, G.D., Arthur, C., Herring, C., 2015. Laboratory methods for the analysis of microplastics in the marine environment: Recommendations for quantifying synthetic particles in waters and sediments. NOAA Technical Memorandum.
Mendoza, L.M.R., Karapanagioti, H., Álvarez, N.R., 2018. Micro (nanoplastics) in the marine environment: Current knowledge and gaps. Current Opinion in Environmental Science and Health 1, 47-51.
Naji, A., Nuri, M., Amiri, P., Niyogi, S., 2019. Small microplastic particles (S-MPPs) in sediments of mangrove ecosystem on the northern coast of the Persian Gulf. Marine Pollution Bulletin 146, 305-311.
Napper, I.E., Davies, B.F., Clifford, H., Elvin, S., Koldewey, H.J., Mayewski, P.A., Miner, K.R., Potocki, M., Elmore, A.C., Gajurel, A.P., 2020. Reaching new heights in plastic pollution—preliminary findings of microplastics on Mount Everest. One Earth 3, 621-630.
Oliveira, M., Almeida, M., 2019. The why and how of micro (nano) plastic research. TrAC Trends in Analytical Chemistry 114, 196-201.
Peng, L., Fu, D., Qi, H., Lan, C.Q., Yu, H., Ge, C., 2020. Micro-and nano-plastics in marine environment: Source, distribution and threats—A review. Science of the Total Environment 698, 134254.
PlasticsEurope, 2013. Plastics – the facts 2013: An analysis of European plastics production, demand and waste data. Plastics Europe Market Reserach Group, Brussels, Belgium.
PlasticsEurope, 2019. Plastics – the facts 2019: An analysis of European plastics production, demand and waste data. Plastics Europe Market Reserach Group, Brussels, Belgium.
Pohl, F., Eggenhuisen, J.T., Kane, I.A., Clare, M.A., 2020. Transport and burial of microplastics in deep-marine sediments by turbidity currents. Environ. Sci. Tec.
Provencher, J.F., Bond, A.L., Avery-Gomm, S., Borrelle, S.B., Rebolledo, E.L.B., Hammer, S., Kühn, S., Lavers, J.L., Mallory, M.L., Trevail, A., 2017. Quantifying ingested debris in marine megafauna: A review and recommendations for standardization. Analytical Methods 9, 1454-1469.
Provencher, J.F., Covernton, G.A., Moore, R.C., Horn, D.A., Conkle, J.L., Lusher, A.L., 2020. Proceed with caution: The need to raise the publication bar for microplastics research. Science of the Total Environment 748, 141426.
Quinn, B., Murphy, F., Ewins, C., 2017. Validation of density separation for the rapid recovery of microplastics from sediment. Analytical Methods 9, 1491-1498.
R Core Team, 2020. R: a language and environment for statistical computing, 4.0.2 ed. R Foundation for Statistical Computing, Vienna, Austria.
Renzi, M., Blašković, A., Bernardi, G., Russo, G.F., 2018. Plastic litter transfer from sediments towards marine trophic webs: A case study on holothurians. Marine Pollution Bulletin 135, 376-385.
Ríos, N., Frias, J.P., Rodríguez, Y., Carriço, R., Garcia, S.M., Juliano, M., Pham, C.K., 2018. Spatio-temporal variability of beached macro-litter on remote islands of the North Atlantic. Marine Pollution Bulletin 133, 304-311.
Schneider, C.A., Rasband, W.S., Eliceiri, K.W., 2012. NIH Image to ImageJ: 25 years of image analysis. Nature Methods 9, 671-675.

Sendra, M., Staffieri, E., Yeste, M.P., Moreno-Garrido, I., Gatica, J.M., Corsi, I., Blasco, J., 2019. Are the primary characteristics of polystyrene nanoplastics responsible for toxicity and ad/absorption in the marine diatom *Phaeodactylum tricornutum*? Environmental Pollution 249, 610-619.
Serra-Goncalves, C., Lavers, J.L., Bond, A.L., 2019. Global review of beach debris monitoring and future recommendations. Environ. Sci. Tec. 53, 12158-12167.
Song, Y.K., Hong, S.H., Jang, M., Han, G.M., Jung, S.W., Shim, W.J., 2017. Combined effects of UV exposure duration and mechanical abrasion on microplastic fragmentation by polymer type. Environ. Sci. Tec. 51, 4368-4376.
Strungaru, S.-A., Jijie, R., Nicoara, M., Plavan, G., Faggio, C., 2019. Micro-(nano) plastics in freshwater ecosystems: Abundance, toxicological impact and quantification methodology. TrAC Trends in Analytical Chemistry 110, 116-128.
Tavares, D.C., Moura, J.F., Ceesay, A., Merico, A., 2020. Density and composition of surface and buried plastic debris in beaches of Senegal. Science of The Total Environment 737, 139633.
Tomillo, P.S., Oro, D., Paladino, F.V., Piedra, R., Sieg, A.E., Spotila, J.R., 2014. High beach temperatures increased female-biased primary sex ratios but reduced output of female hatchlings in the leatherback turtle. Biological Conservation 176, 71-79.
Torre, M., Digka, N., Anastasopoulou, A., Tsangaris, C., Mytilineou, C., 2016. Anthropogenic microfibres pollution in marine biota. A new and simple methodology to minimize airborne contamination. Marine Pollution Bulletin 113, 55-61.
Turner, A., 2017. Trace elements in fragments of fishing net and other filamentous plastic litter from two beaches in SW England. Environmental Pollution 224, 722-728.
Turra, A., Manzano, A.B., Dias, R.J.S., Mahiques, M.M., Barbosa, L., Balthazar-Silva, D., Moreira, F.T., 2014. Three-dimensional distribution of plastic pellets in sandy beaches: Shifting paradigms. Scientific Reports 4, 4435.
Van Franeker, J.A., Law, K.L., 2015. Seabirds, gyres and global trends in plastic pollution. Environmental Pollution 203, 89-96.
Van Sebille, E., Wilcox, C., Lebreton, L., Maximenko, N., Hardesty, B.D., Van Franeker, J.A., Eriksen, M., Siegel, D., Galgani, F., Law, K.L., 2015. A global inventory of small floating plastic debris. Environmental Research Letters 10, 124006.
Vermaire, J.C., Pomeroy, C., Herczegh, S.M., Haggart, O., Murphy, M., 2017. Microplastic abundance and distribution in the open water and sediment of the Ottawa River, Canada, and its tributaries. Facets 2, 301-314.
Wilcox, C., Puckridge, M., Schuyler, Q.A., Townsend, K., Hardesty, B.D., 2018. A quantitative analysis linking sea turtle mortality and plastic debris ingestion. Scientific Reports 8, 12536.
Wyer, H., Polhemus, D., Moore, S., Weisberg, S., Coffin, S., Rochman, C., 2020. EXPRESS: Steps scientists can take to inform aquatic microplastics management: A perspective informed by the California experience. Applied Spectroscopy 74, 971–975.
Zhang, Y., Kang, S., Allen, S., Allen, D., Gao, T., Sillanpää, M., 2020. Atmospheric microplastics: A review on current status and perspectives. Earth-Science Reviews 203, 103118.

Table 1. Plastic size categorisations recommended by a random sample of plastic review papers, from 2017–2020. Variation is demonstrated within terminology, size ranges and unit measurements (e.g., nm, µm, mm, and cm).

Terminology	Size	Source
Megaplastic	>100 mm	Provencher et al. (2017)
Macroplastic	>1 cm	Hartmann et al. (2019)
	2-10 cm	Provencher et al. (2017)
	>20 cm	Hanvey et al. (2017)
Mesoplastic	5–20 cm	Hanvey et al. (2017)
	5–20 mm	Provencher et al. (2017)
	1–10 mm	Hartmann et al. (2019)
Microplastic	1–5 mm	Provencher et al. (2017)
	1–1000 µm	Hartmann et al. (2019)
	1 µm – 5 mm	Frias and Nash (2019)
Large microplastic	1–5 mm	Hanvey et al. (2017)
Small microplastic	1–1000 µm	Hanvey et al. (2017)
	25 µm – 1 mm	Gigault et al. (2018)
Nanoplastic	1–1000 nm	Gigault et al. (2018); Hartmann et al. (2019)
	<1000 nm	Hanvey et al. (2017)
	1–100 nm	Mendoza et al. (2018)

Table 2. Plastic size categorisations used in a sample of research papers from 2017–2020.

Terminology	Size	Source
Macroplastic	>25 mm	Egessa et al. (2020); Ghaffari et al. (2019); Imhof et al. (2017)
Mesoplastic	5–25 mm	Egessa et al. (2020); Ghaffari et al. (2019); Imhof et al. (2017); Lee et al. (2017)
	>4.75 mm	Karthik et al. (2018)
Microplastic	<5 mm	De Ruijter et al. (2019); Dobaradaran et al. (2018); Egessa et al. (2020)
	1–5 mm	De-la-Torre et al. (2020); Delvalle de Borrero et al. (2020)
	0.5–5 mm	Chubarenko et al. (2018); Chubarenko et al. (2020); Di and Wang (2018)
	0.33–5 mm	Abidli et al. (2017); Lots et al. (2017)
	0.33–4.75 mm	Karthik et al. (2018)
Large microplastic	1–5 mm	Ghaffari et al. (2019); Imhof et al. (2017); Quinn et al. (2017)
	2–5 mm	Chubarenko et al. (2020)
Small microplastic	0.5–2 mm	Chubarenko et al. (2020)
	<1 mm	Quinn et al. (2017)
	0.1–1000 µm	Naji et al. (2019)
Nanoplastic	<1 µm	Dong et al. (2019)
	<100 nm	Strungaru et al. (2019)

Table 3. Henderson Island macro- and microplastic density (g/m^2) records from 2015 (data adapted from Lavers and Bond (2017)) and 2019. Items per m^2 and buried micro- and nanoplastics Pieces/kg dw) were included for 2019 only. Total refers to mean ± SD. Numbers marked with* are reported with standard error (SE). Nanoplastics are mean counts of three replicates per quadrat.

		Density (g/m^2)		**Density (Items/m^2)**			**Density (pieces/kg dw)**
Sampling	**Dimensions**	**Macro**	**Large-micro**	**Macro**	**Large-micro**	**Small-micro**	**Nano**
2015							
Transect	30 × 7 m	43.179	0.26				
Transect	30 × 7 m	414.561	0.202				
Transect	10 × 7 m	917.24	5.922				
Total		458.326 ± 438.671	2.128 ± 3.285				
2019							
Quadrat	0.5 × 0.5 m	44.398	12.891	271	1130	381	1467 ± 643
Quadrat	0.5 × 0.5 m	117.362	12.711	309	1086	705	1667 ± 503
Quadrat	0.5 × 0.5 m	207.063	32.81	494	2066	1713	1993 ± 115
Quadrat	0.5 × 0.5 m	355.768	20.178	606	1887	2457	1400 ± 346
Quadrat	0.5 × 0.5 m	129.292	38.365	611	2805	980	3333 ± 1973
Transect	50 × 7 m	1,597.14					
Total		408.326 ± 591.864	23.391 ± 11.700	458.326 ± 161.073	1794.800 ± 715.351	1247.200 ± 836.101	1960 ± 356*

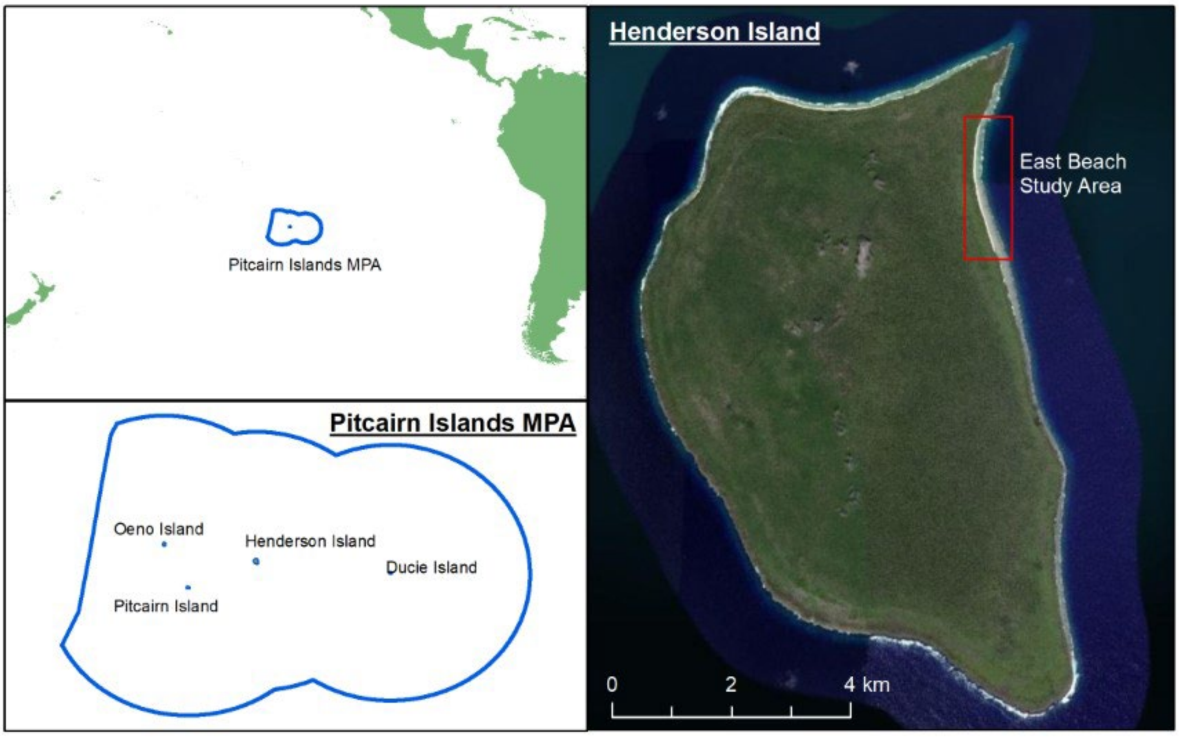

Fig. 1. East Beach study site on Henderson Island, South Pacific (24.36°S, 128.30°W). Data from ESRI (2020). "World Imagery" (basemap). Scale: 1:100,000. November 20, 2020. (https://services.arcgisonline.com/ArcGIS/rest/services/World_Imagery/MapServer; accessed 04 December 2020).

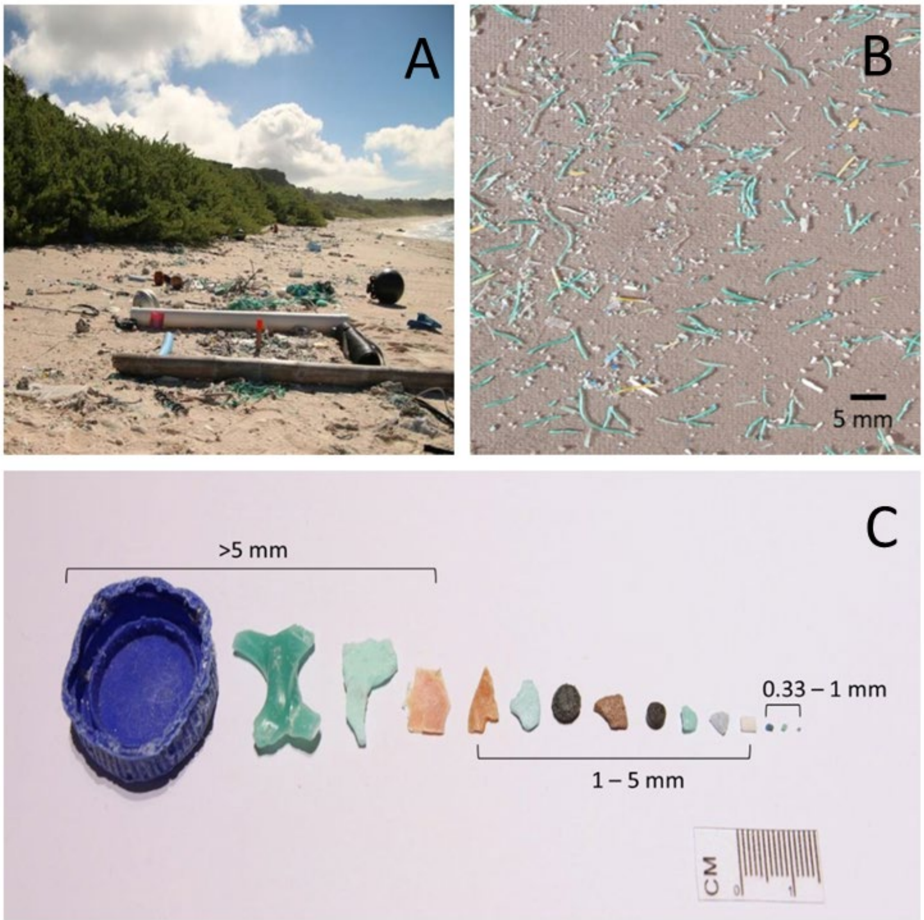

Fig. 2. A 1 m^2 quadrat on East Beach, Henderson Island. (B) Sample of small microplastics from one quadrat within the 0.33–1 mm range (small microplastics categorised by width, not length). (C) Visual representation of the three plastic size categories used to compare with Cozar et al. (2014): macroplastic (>5 mm), large microplastic (1–5 mm) and small microplastic (0.33–1 mm).

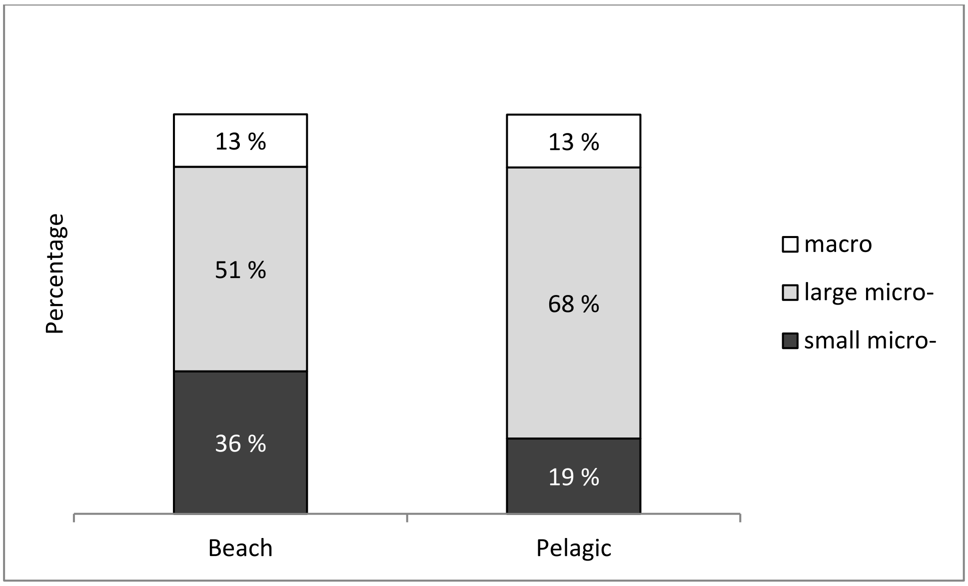

Fig. 3. Percentage of plastic items per size class, collected on beaches of Henderson Island and pelagic surface waters of the South Pacific, as modelled by Cózar et al. (2014). Percentages rounded to the nearest whole figure.