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# Microplastics on the Portuguese coast

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ABSTRACT

Marine anthropogenic litter was analysed in eleven beaches along the Portuguese coast, over a two-year period (2011 − 2013). Of all collected items, 99% were plastic and 68% were microplastics (MP; 1–5 mm in diameter). Higher MP concentrations were found in winter/autumn, near industrial areas and/or port facilities and in beaches exposed to dominant winds. Resin pellets (79%) were the dominant category close to industrial areas and high concentrations of fragments and polymeric foams were found near fishing ports. The most frequent pellet size classes were 4 and 5 mm (respectively 47% and 42%). Results suggest that MP have predominately a land-based origin and are deliberately discarded or accidentally lost in watercourses and/or coastal areas. A combination of measures within stakeholders, namely industry and fishing sectors and share of good practices are needed to prevent marine anthropogenic litter.

## 1. Introduction

Microplastics (MP) - plastic particles smaller than five millimeters (≤5 mm in diameter) - have been recognized as a ubiquitous marine environmental problem, accentuated over the last decades (Eriksen et al., 2013, Lusher, 2015; UNEP, 2016). The persistence of plastic materials and their slow degradation in the environment enable them to enter and accumulate in aquatic ecosystems (Zarlf et al., 2011),

Several studies have documented MP accumulation in different environments – open ocean water bodies (Wang et al., 2017), beaches and coastlines (Hengstmann et al., 2018; Zhang et al., 2018), subtropical oceanic gyres (Brach et al., 2018), polar areas (Lusher et al., 2015; Obbard, 2018), deep ocean sediments (Van Cauwenberghe et al., 2015), and freshwater systems (Vaughan et al., 2017; Su et al., 2018; Wagner and Lambert, 2018).

MP can either enter the aquatic ecosystem from direct leakages during transfer and transport to/from industries, and known as primary MP (e.g. pellets and microbeads) (Hohenblum et al., 2015) or resulting from breakdown of larger plastics, known as secondary MP (e.g. fragments) (Arthur et al., 2009; Andrady, 2011; Ryan, 2015).

Resin pellets (usually ≤10 mm in diameter), are the raw material used in moulding and packaging in order to produce all sorts of plastic items used in modern society (Fendall and Sewell, 2009; Cole et al., 2011). MP beads used on personal care products (Lei et al., 2017), such as exfoliating facial scrubs, toothpastes, shower gels are able to enter the marine environment through wastewater discharges (Ziajahromi et al., 2017). Other vectors of MP transport to oceans are river flows, extreme events (e.g. storms), winds or animals (Galgani et al., 2013). Maritime activities (Ryan et al., 2009) and agricultural activities (Briassoulis et al., 2010) are also potential sources of MP input into aquatic ecosystems.

Regarding polymer type, there are a wide range of polymers annually produced, namely: polyethylene (PE), polypropylene (PP) and polystyrene (PS), polyvinylchloride (PVC), polyamide (PA), polyethylene terephthalate (PET) or polyvinyl alcohol (PVA), (PlasticsEurope, 2016). These polymers are usually mixed with additives (Avio et al., 2016). PE, PP and PS float on seawater and consequently are able to travel long distances being found far away for their sources (Zhang et al., 2018). PVC, PA, PET or PVA have the ability to sink due to their intrinsic high density, potentially accumulating in benthic sediments.

Lower density plastics might also have the tendency to sink and accumulate in the ocean floor when colonized by organisms, i.e. biofouling (Lobelle and Cunliffe, 2011).

Polymeric molecular structures degrade over time when exposed to UV radiation (resulting in plastic photo-oxidation) (Cai et al., 2018), during abrasion processes caused by impacts of sediment, currents, winds (increase bigger plastics fragmentation) (Mathalon and Hill, 2014), interaction with marine organisms and/or microbial action (Caruso, 2015).

The impacts of MP to marine wildlife have also been investigated by several authors and are essentially related either to ingestion or

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entanglement (Gall and Thompson, 2015). Regarding ingestion, MP particles are mistaken by food due to their size, shape and colour (Schuyler et al., 2014). Studies have traced impacts of MP in small animals such as zooplankton and larval fish (Cole et al., 2013; Sun et al., 2017; Steer et al., 2017), sessile invertebrates (Wright et al., 2013; Thushari et al., 2017) and in larger animals, such as sea turtles (Camedda et al., 2014; Pham et al., 2017), marine birds (Van Franeker et al., 2011) and fish species (Neves et al., 2015; Ory et al., 2017) either directly or through consumption of prey, and therefore understanding beach hotspots is relevant to minimize potential accumulation of MP in the food web.

MP also act as dispersal vectors of chemical additives (Rochman, 2015), once organic and metal pollutants are absorbed onto their surface, due to their hydrophobic nature (Mato et al., 2001; Teuten et al., 2007; Teuten et al., 2009; Frias et al., 2010; Antunes et al., 2013). Adsorbed pollutants can potentially be transferred to organisms tissues following ingestion (Rochman et al., 2013), depending on their size (Yamashita et al., 2011; Bakir et al., 2012; Tanaka et al., 2013; Cannon et al., 2016 and Nadal et al., 2016). The highest concentrations of these contaminants are often detected in estuaries and coastal areas (Bergmann et al., 2015; Wagner and Lambert, 2018).

Therefore, this study aimed to quantify MP accumulation in beaches along the coast of Portugal and to identify a proximity relationship between MP abundance and potential hotspot areas such as proximity to port facilities, urban areas, plastic moulding and packaging industries and receiving river streams. Results of this study constitute the first identification of MP accumulation in beaches of the coast of mainland Portugal.

## 2. Materials and methods

### 2.1. Sampling sites

A total of eleven sampling sites were chosen according to the following criteria; 1) proximity to potential sources and/or inputs of MP, i.e. major receiving stream, industrial and/or port facilities (cargo and fisheries) and densely populated urban areas and 2) exposure to dominant northerly winds.

The eleven sampled beaches were divided into 3 regions: North (n = 3) - Matosinhos (MT), Espinho (ES), Mira (MI); Centre (n = 5) - Vieira de Leiria (VL), Paredes de Vitória (PV), Peniche (PEN), Cresmina (CR), Fonte da Telha (FT); and South of mainland Portugal (n = 3) - Sines (SI), Bordeira (BOR) and Ancão (ANC) (Fig. 1).

Sampling campaigns took place during spring (April 2011 and March 2013, n = 63) and in winter/autumn (January 2012 and September 2012, n = 99). For the later, sampling occurred immediately after extreme events dominated by strong winds and high hydrodynamics to guarantee marine debris accumulation on the beaches.

All sampled beaches are exposed to prevailing northerly winds (MT, ES, MI, VL, PV, PEN, CR and FT are exposed to North winds and SI, BOR and ANC are exposed to NNW winds). With the exception of five beaches (MI, VL, PV, PEN and BOR), the remaining ones are protected within bays (CR and FT) or by infrastructures (MT, ES and SI) that minimize the effects of strong winds and currents. PEN beach is the location exposed to high speed (20 kts) winds for longer periods of time (IPMA, 2015).

Precipitation is an important fact to disperse plastic pollution, namely MP. During spring, there are more precipitation events in the Centre, occurring between 67 mm to 95 mm per month from PV to SI. In winter/autumn, events of precipitation are more common in the North (MT and ES) and in two special zones (PEN and SI) (SNIRH, 2018).

A total of 162 sediment samples (top 2–3 cm) were scooped with a metal spoon at the high tide line (accumulation zone) using 50 × 50 cm squares (3 to 5 replicates), following methodologies previously used by Frias et al., 2010 and Antunes et al., 2013.

#### 2.1.1. Laboratory procedures

Once at the laboratory, sediment samples were dried at room temperature for at least 24 h. Large plastic items (large microplastics and mesoplastics) were removed with a metal tweezers prior to the density separation procedure.

Dried sediments (with an approximate volume of 0.5 × 0.5 × 0.02 m per sample) were introduced into a 25 L glass aquarium to which was added a high-density sodium chloride (NaCl; 1.2 g $cm^{-3}$) solution (Galgani et al., 2010). Sediment was stirred vigorously and left to settle for approximately 1 h. Floating particles (including natural items and visible MP particles) were collected from the surface of the saturated solution using metal tweezers, saved in glass Petri dishes and dried in a muffle at 40 °C. This procedure was repeated until no visible particles were floating. The supernatant solution was filtered with a Millipore vacuum pump, onto Whatman® GF/C filters (~1.2 μm pore; ø = 47 mm), in order to recover MP with smaller dimensions. Filters were observed under a Leica® MZ125 stereomicroscope. All steps were conducted using 100% cotton labcoats and precautions to avoid cross-contamination were taken (e.g. airborne fibers). All collected items were counted and measured into 5 mm size classes, (1–5 mm), and separated into 7 categories (pellets, fragments, styrofoam, foam sponges, filaments, swabs, films), according to Ogi and Fukumoto, 2000.

### 2.2. Statistical analyses

After invalidation of homogeneity of variances as determined by Levene's test, nonparametric statistical analyses were performed on collected data.

The Kruskal-Wallis test was applied for comparisons between regions (North, Centre and South), to MP collected (items·$m^{-2}$), in winter/autumn and spring.

Correlations were determined between sampled sites and MP exposure criteria (population density, receiving streams average annual flow, ports and industries and prevailing winds) using the Spearman test.

A principal component analysis (PCA) was conducted to estimate the variables that contributed the most to explaining seasonal variations.

The significance level for all analysis was set at 95% ($\alpha = 0.05$). All calculations were performed with software Statistica 7 (Statsoft Inc., Tulsa, OK, USA).

## 3. Results

Ninety-nine per cent of collected items were plastic (666 ± 2642 items·$m^{-2}$, n = 162, Table 2), of which 68% were MP (≤5 mm) (454 ± 1908 items·$m^{-2}$). MP were dominated by resin pellets (79%–358 ± 1679 items·$m^{-2}$), followed by fragments (14%–63 ± 169 items·$m^{-2}$), and polymeric foams: Styrofoam® (6%–30 ± 152 items·$m^{-2}$) and foam sponge (1%–3 ± 20 items·$m^{-2}$) (Fig. 2). All filaments, swabs and films analysed had dimensions higher than > 5 mm in diameter.

MP were predominant in all regions (54% in the North (236 ± 369 items·$m^{-2}$, n = 48); 74% in the Centre (707 ± 2790 items·$m^{-2}$, n = 69, Table 2) and 60% in the South (251 ± 346 items·$m^{-2}$, n = 35) (Table 2) (Kruskal-Wallis test, $p < 0.05$).

Size distribution (mm) is presented in Fig. 3, where items with 4 and 5 mm in diameter are more abundant, followed by 3 mm pellets (47% 211 ± 733 items·$m^{-2}$, 42% 208 ± 1124 items·$m^{-2}$ and 10% 48 ± 133 items·$m^{-2}$ of the total MP, respectively). These categories were dominant due to the high quantities of resin pellets.

Significant differences between size classes 4 and 5 mm were registered comparatively to the remaining sizes (Kruskal-Wallis test, $p < 0.05$).