# Microplastic distribution in urban vs pristine mangroves: using marine sponges as bioindicators of environmental pollution

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## Highlights

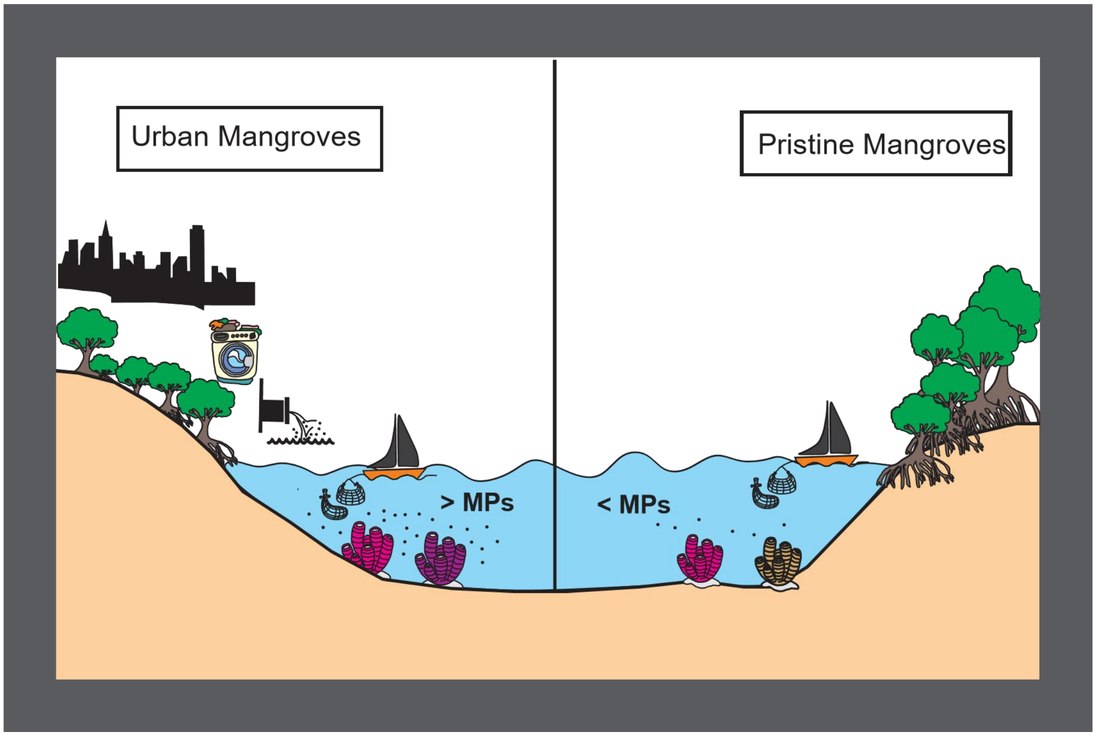
- Fibres were the only microplastic type in marine sponges from Isla del Carmen.
- Sponges exhibited MP concentrations from 556 to 5000 items kg$^{-1}$.
- Sponges bioaccumulate 17 times more MPs than seawater concentrations.
- Marine sponges act as a quantitative bioindicator of MPs to aquatic environments.
- Fishing, sewage and laundry activities were the main anthropogenic sources of MPs.

## Abstract

Sessile benthic organisms are considered good bioindicators for monitoring environmental quality of coastal ecosystems. However, these environments are impacted by new pollutants such as microplastics (MPs), where there is limited information about organisms that can be used as reliable bioindicators of these emerging contaminants. We evaluated MP concentrations in three compartments: surface sediment, water and in three marine sponge species (*Haliclona implexiformis*, *Halichondria melanadocia* and *Amorphinopsis atlantica*), to determine whether these organisms accumulate MPs and reflect their possible sources. Results showed MPs in all three compartments. Average concentrations ranged from 1861 to 3456 items kg$^{-1}$ of dry weight in marine sponges, 130 to 287 items L$^{-1}$ in water and 6 to 11 items kg$^{-1}$ in sediment. The maximum MP concentration was in the sponge *A. atlantica*, which registered 5000 items kg$^{-1}$ of dry weight, in water was 670 items L$^{-1}$ and in sediment was 28 items kg$^{-1}$, these values were found in the disturbed study area. The three sponge species exhibited MP bioaccumulation and showed significant differences between disturbed and pristine sites ($F = 11.2$, $p < 0.05$), suggesting their use as bioindicators of MP.

**Capsule abstract:** Bioaccumulation of marine sponges show how these organisms could be considered as possible bioindicators to evaluate MP pollution in coastal ecosystems.

**Keywords:** Microplastics, Marine Sponges, Mangroves, Isla del Carmen, Gulf of Mexico. **Graphical Abstract**




**Introduction**

Microplastics (MPs) are recognized, together with a range of new pollutants, as emergent contaminants (Anderson et al., 2018; Yin et al., 2020). They can be defined as any synthetic material with a specific density, polymeric matrix, colour, size less than 5 mm and undefined shape (fibres, foams and microbeads) (NOAA, 2018; Stead et al., 2020 and Fred et al., 2020). This emergent contaminant has recently been noted as a global concern, because of their potential environmental impacts and wide distribution in marine and estuarine ecosystems (McEachern et al., 2019; Velez et al., 2019 and Zhang et al., 2020). Urban wastewater, sewage discharge and agricultural runoff are identified as point sources of MPs (Yeats et al., 2010; Li et al., 2020). Human activities such as fishing, shipping, tourism, cosmetic and textile industries are also linked to plastic and MP contamination and can be considered as sources of MP inputs to aquatic environments, either directly or indirectly (Nuelle et al., 2014; Aliabad et al., 2019).

In tropical and subtropical coastal systems worldwide, mangroves are considered as natural pollution filters (Maghsodian et al., 2021; Rezaei et al., 2021), because they are usually located close to urban or industrial environments that dump their waste with little or no treatment. In general, these ecosystems are recognized worldwide as natural environments that are exposed to a range of pollutants including MPs, in part due to their role as a conduit between the terrestrial and marine realms (Ward et al., 2016; Celis et al., 2020). Indeed, due to their role as nursery habitats for a range of marine organisms, mangroves have been suggested as the first step in the pathway of MP transference to higher levels of the trophic chain and begin a bioaccumulation and biomagnification process along all the food webs.

Bioindicators are organisms that provide information about environmental quality where they occur (Holt and Miller, 2011). They can reflect spatial and temporal variations in environmental conditions due to changes in their diversity, abundances or for their capacity to accumulate pollutants (Zukal et al., 2015; Bonanno and Vymazal, 2017). In general, these organisms should have some basic properties such as natural abundance, wide geographical distribution, ease of identification, ease of sampling and they must show a moderate tolerance to disturbance and stress (Carignan and Villard, 2002; Caro et al., 2010; Urban et al., 2012) that help to integrate environmental information. Marine organisms that have beennproposed as bioindicators of plastic debris are: seabirds (e.g., *Fulmarus glacialis*, Linnaeus, 1761), loggerhead sea turtles (e.g., *Caretta caretta* Linnaeus, 1758), mussels (e.g., *Mytilus edulis*, Linnaeus, 1758), and other taxonomic groups such as fish, mammals, polychaetes, bryozoans, holothurians and also bacterial communities (Bonanno and Orlando, 2018). In the case of marine sponges (phylum Porifera), they have been proposed as bioindicators for heavy metals, polycyclic aromatic hydrocarbons, and microbial pollution; due to their sessile condition, type of feeding (by filtration), their high sensitivity to environmental changes and their relative abundance in benthic ecosystems (Mahaut et al., 2013; Batista et al., 2014). However, the use of marine sponges as bioindicators of MPs has hardly been investigated and there is limited information about marine sponges that enable us to understand if these organisms are capable of reflecting the environmental quality of aquatic ecosystems and if they could be used as bioindicators for MPs (Baird, 2016; Karlson et al., 2017; Girard et al., 2020). Our hypothesis is that if they accumulate MPs in their body, MP concentration is likely to be related to that recorded in the surrounding environment (e.g., water column and surface sediments) where they occur.

Here we examine the accumulation of MPs in marine sponges, sediment and water from two mangrove areas with different levels of human disturbance located within Laguna de Terminos, a Natural Protected Area located in the southern Gulf of Mexico. The study evaluated whether the concentration of MPs in these compartments varied spatially (between sites), and in the case of sponges, between species. An assessment was also undertaken to assess whether sponges have the potential to be used as bioindicators of MP pollution in mangrove areas.

## Study area

Mangrove ecosystems from Isla del Carmen belong to a natural protected area known as Flora and Fauna Protection Area Laguna de Terminos, which includes the second largest coastal lagoon environment in Mexico (about 7050.16 $km^2$) (INEGI, 2018) (Figure 1). Isla del Carmen is a barrier island which 77% of its area is covered by mangrove forests, while the remaining 23% is covered by the urban area of Ciudad del Carmen, which is the second most populated city (248,303 inhabitants) in Campeche State (INEGI, 2018). The inner part of Isla del Carmen is characterized by shallow seagrass meadows (dominated by a mix of *Thalassia testudinum* Banks ex Köning, 1805 and *Halodule wrightii* Ascherson 1868) bordered by mangrove areas (*Rhizophora mangle*, *Avicennia germinans*, *Laguncularia racemosa* and *Conocarpus erectus*). There, *Amorphinopsis atlantica* Carvalho, Hadju, Mothes & van Soest, 2004, *Halichondria melanadocia* Laubenfels, 1936 and *Haliclona implexiformis* (Hechtel, 1965) are among the most common sponge species in the area (Castellanos-Pérez et al., 2020) (Figure 2). Although this is a natural protected area, artisanal fishing activities are permitted throughout the year.

## Materials and methods

**Field sampling**

In order to determine whether marine sponges reflect the environmental quality of mangrove ecosystems, two sampling sites with a clear and contrasting anthropogenic influence were selected. The disturbed area was identified as site "A" (18° 38´24.07" N, 91° 47´49.86" W), characterized by mangroves impacted by untreated urban water discharges presence from Ciudad del Carmen as well as wastewater from an adjacent slaughterhouse. The distance from the sampling point to the shoreline was approximately 50 m and the depth was 1.1 m at high tide.

The undisturbed area is located 30 km away from area "A" and was identified as site "B" (18° 44´31.62" N, 91° 32´13.66"). This site is characterized by pristine mangroves, where the only human activities are artisanal fishing and the distance from this sampling point to the shoreline was approximately 50 m and the depth 1.45 m at high tide. At every site 3 sediment samples, 3 water samples and 10 sponge individuals were taken. Surface sediment samples were taken using a handheld aluminium corer. Water samples were obtained taking surface water with a gallon container. Samples of two different sponge species were also collected (by hand while snorkelling) at each site. At site A, five individuals of *H. implexiformis* and five of *A. atlantica* were collected, and at site B, five *H. implexiformis* and five *H. melanadocia* individuals were collected (Figure 2), representing the dominant species in both areas (Castellanos-Pérez et al., 2020). The collection methods were stratified random point for each compartment and were undertaken in August 2019.

**Laboratory work**

Organic and inorganic samples were processed as follow: Sponge samples were cleaned for epibionts and washed with distilled water to remove sediment and detritus on the surface. They were then dried in an oven at 40°C for 7 days to obtain the dry weight. In each sponge individual, three subsamples from 0.2 to 1.2 g were extracted with a 1.2 cm diameter aluminium corer to examine MPs inside the sponge body. These subsamples were crushed in a pestle and mortar to obtain a finely ground material. Before crushing these subsamples were observed under a Carl Zeiss Stemi 305 stereomicroscope at 4x magnification to observe and remove any material (rocks, shells or macroplastics) to facilitate the process. Each sponge subsample was introduced into a solution of distilled water, hydrogen peroxide and ammonium hydroxide in equal parts for three days at 50°C to digest the organic material. After this treatment the material was distributed in glass petri dishes and put to dry in an oven for 3 days. Once the material was dried, it was observed under the stereomicroscope. MPs were carefully removed using dissecting forceps, quantified by shape and colour, and placed in flasks with distilled water for subsequent analysis. Sediment samples were dried at 50°C for 72 h. MP separation from the sediment was undertaken by flotation using a saturated solution of NaCl, where 100 **g** of dried sediment was shaken by a magnetic stirrer in 1 L of saturated solution for 5 min and 2 min rest lo let the sediment settle in the flask bottom. Later, using a vacuum system the supernatant was filtered in 0.45 µm nitro cellulose filters (Millipore) and rinsed with 250 mL of distillate water to avoid salt precipitation during the filter drying process. After this, each filter was dried at 50°C for 24 h and it kept in glass petri dishes until MP assessment using the stereomicroscope was undertaken. Finally, employing a vacuum system 100 mL of seawater was filtered on 0.45 µm nitro cellulose Millipore and each filter was treated at the same way those used for sediment sampling. Care was taken to avoid any potential environmental contamination throughout the analysis and sample process. Simple precautions such as exhaustive rinsing with distillate water were undertaken for all the materials and equipment. Glassware material was used instead of plastic, wherever it was possible, and each sample was covered before and after analysis to avoid atmospheric contamination.

**Microplastics identification**

In this study, all MPs were verified using the hot needle test. Although this test cannot be used to identify MP polymer types such as polyethylene, propylene, and PET, it is acceptable as an economical way to verify particles are MPs based on their response to a hot needle (Lusher et al., 2017; Campbell et al., 2017; Silva et al., 2018; Kapp and Yeatman, 2018). Additionally, we used photographs published in scientific journals as base data.

**Data analysis**

The normality and homoscedasticity of the data (MP concentrations in sponges, sediments, and water) were examined using the Shapiro-Wilk and Levene's test, respectively. One-way Analysis of Variance (ANOVA) were performed to determine whether there were significant inter-site variations in MPs concentration and colour in the three marine compartments (sponges, sediment and water column). One-way analysis of variance (ANOVA) followed by Tukey's post hoc test was used to assess variations in MPs concentration between sponge species. These analyses were performed using Statistica 6.0 software. To evaluate the marine sponges' ability to accumulate MPs with respect to their availability in the environment, we used the bio-accumulation factor (BAF), which is defined as the uptake of a contaminant by an organism from the abiotic environment (non-living chemical and physical components of the environment that affect living organisms, for instance sunlight, sediment, water and pollution) (Rand et al. 1995). Usually, the BAF for sponges is calculated as the ratio between the chemical concentration (in this case MPs) recorded in sponges divided by MP concentration in seawater or sediment (Negri et al., 2006; Venkateswara et al., 2009; Padovan et al., 2012; Batista et al., 2014; Orani et al., 2018). For this work we used MP concentration in seawater to obtain the BAF values. Values equal or higher than 1 are considered indicative of bioaccumulation.

**Results**

**Microplastic abundance in water, sediment and marine sponges in the pristine and disturbed environments**

Results showed MP presence in the three compartments in both, the disturbed (site A) and pristine (site B) environments (Figure 3). In seawater, the average abundance of MPs was 287 items $L^{-1}$ in site A (ranged from 20 to 670 items $L^{-1}$) and 130 items $L^{-1}$ in site B (from 20 to 670 items $L^{-1}$). In sediments, the average abundance was 11 items $kg^{-1}$ of DW in site A (6 to 28 items $kg^{-1}$) and 6 items $kg^{-1}$ of DW in site B (from 4 to 10 items $kg^{-1}$). While in marine sponges the average abundances were 3456 items $kg^{-1}$ of DW in site A (from 2000 to 5000 $kg^{-1}$) and 1861 items $kg^{-1}$ in site B (from 556 to 4444 items $kg^{-1}$). In general, every compartment had the highest MP concentration in site A, but MP average concentration in the seawater and sediments did not vary significantly between the sites ($F=0.58$, $p > 0.05$, $F=0.32$, $p > 0.05$, respectively). Although, the differences between water and sediment were not significant, there was almost a 2:1 ratio between the disturbed and pristine environments that shows a clear spatial distinction in MP abundance.

## Microplastic kind, size and colour found in water, sediment and marine sponges

In this work, only MP fibres were recorded in every compartment and studied site. Their average sizes ranged from 2.3 to 6.2 mm and their colours were blue, red, black and white (Figure 4). In both sites, we found blue and red fibres in water, sediment and sponges. White fibres were found only in water; while black fibres were only identified in sediment and sponge samples. Blue fibres were more abundant (between 40% and 74%) than red (15% and 30%) and black (11% and 30%) in the sediment and sponges, respectively; while white fibres were more abundant (41%) than blue (39%) and red (20%) fibres in water.

## Inter-site and inter-species variability of MPs concentration

The results show the presence and bioaccumulation of MPs in the three studied sponge species and inter-site differences between average concentrations. The average concentration of MPs in the species *A. atlantica* and *H. implexiformis* were 3455 and 3457 MPs $kg^{-1}$ DW of sponge and their ranges per species were from 2000 to 5000 and from 2500 to 4444, respectively in site A; while in the species *H. melanadocia* and *H. implexiformis* concentrations were 2033 and 1688 MPs $kg^{-1}$ DW and they varied from 556 to 4444 and from 909 to 2143, respectively in site B (Figure 5). MP concentration in sponges did not vary between species (site A: $F= 0.0001$, $P > 0.05$; site B: $F= 0.17$, $P> 0.05$) but did vary significantly between sampling sites ($F= 11.2$, $p < 0.05$). These results suggest that the sponges *H. melanadocia* and *H. implexiformis* from site A exhibited significantly lower MPs concentration than the species *A. atlantica* and *H. implexiformis* from site B. The average bioaccumulation factor (BAF) of MPs in the species *A. atlantica* and *H. Implexiformis* was 17 for both species within site A. At site B the BAF was 10 and 8 in the species *H. melanadocia* and *H. implexiformis* respectively. These results suggest that marine sponges can bioaccumulate MPs by a factor of at least 8 compared to MP concentrations in the surrounding seawater.

# Discussion

## Microplastic origin, sources and possible chemical composition

The chemical composition of MPs is a good tool to identify the origin of these pollutants, because it is possible to identify the type of plastic that is deposited in aquatic environments and infer what was made with it. However, it is both costly and complicated, due to the nature of microplastics (small), and chemical analysis is not the only way to undertake an evaluation of the source. Using physical characteristics rather than chemical composition is a good and cheap approximation. Plastics from different sources obtain different shapes when degraded by weathering conditions and it is feasible to use MP shape to identify the origin and source. Borges et al. (2020) identified that most fibres found in aquatic environments are related to fishing and laundry activities. Wang et al. (2019) suggested that films are likely to be derived from plastic bags and agricultural films. Foams have been suggested to be related to the breakdown of foam material such as expanding foam, floating fishing gear and floating containers that are used for packaging and fisheries industries (Zhou et al., 2020) and pellets and microbeads are related to cosmetic and washing products (Andrady et al., 2017). Therefore, if MPs of different shapes are found in a specific environment, this suggests that they come from different origins. Several studies worldwide have noted that urban sewages and waste water treatment plants can be considered as an important source of MPs and also have noted that fibres are the main shape for MPs from this source (Fendal et al., 2009; Browne et al., 2011; Deng et al., 2020). This is because sewage systems usually receive most of the wastewater generated by laundry activities in urban centres.

Our results suggest that MPs obtained in this work come from the same origins, because, we find fibres as the only MP shape in both study sites. Both fishing activities and sewage outfall are well known as environmental MP fibre sources. Therefore, it is suggested that these pollutants are likely to come from the degradation of fishing nets, ropes, and laundry activities undertaken in Ciudad del Carmen. Also, the sewage presence in the degraded site could explain why the average MP abundances (in the three compartments) were almost double in the disturbed site compared to the pristine site. Borges et al. (2020) identified Polyamide 6 + Polyamide 6.6, Polystyrene and nylon in fibres collected in six fish species of the Campeche Bay (the same area as this current study). These polymers are typically used in packages and packaging, twine, ropes, water sport items such as life jackets, clothes, and fishing lines respectively. Chang et al. (2019) observed that fibres collected in the Hong Kong coast were made of nylon, polystyrene, and polyethylene. Historically, the main polymers in aquatic environments were found to be polyamides and polyester, which are common materials in fishing activities (Pruter, 1987) and these polymers have been recorded in high abundance in every aquatic environment (Wang et al., 2018). Therefore, it is highly likely that plastics related to fisheries activities and clothing such as polyamides, nylon, polystyrene, and polyester represent the chemical composition of the fibres found in this work.

**Microplastic bioavailability**

Microplastic bioavailability in organisms is related to physical characteristics such as size, shape and colour. Usually, their bioavailability is due to their small size that allows them to be easily ingested by aquatic organisms (Jovanovic et al., 2017). Their shape and colour are identified as factors that produce a misunderstanding in aquatic predators about what they are eating (Wright et al., 2013; Masia et al., 2019) as MPs can be readily mistaken for prey, particularly where biofouling has occurred. Borges et al., (2020) noted MP presence in the gastrointestinal track of six commercial marine fish species caught for human consumption along the coast of Campeche, Mexico. They found fibres, fragments, and pellets were the main shapes of MPs. Borges et al. (2020), recorded black, white, red, blue, orange, green and brown fibres and pellets. Their results highlighted differences in the shape and colour of MPs found between demersal fish and pelagic species. These differences showed evidence that the density of the MP material plays a key role in determining their fate in marine fish habitats. Sun et al. (2017) evaluated the characteristics of MPs ingested by different groups of zooplankton in natural conditions, including copepods, chaetognaths, jellyfish, shrimps and fish larvae and the encounter rates between them. Their results highlighted that the size of MPs ingested by zooplankton ranged from 125 μm to 167 μm. They identified fibres, particles, and other types of MPs between the zooplankton groups. They also found that the proportion of shapes of the MPs varied between zooplankton groups. For example, fibre proportions decreased from copepods to fish larvae and the encounter rates of MPs/ zooplankton increased from copepods to chaetognaths, jellyfish, shrimps and fish larvae. Although MP toxicity is related to the additives, and the organic and inorganic pollutants adsorbed that can be transferred to aquatic organisms, some works have noted that MP colour could provide clues about the capacity of these particles to adsorb heavy metals and organic pollutants. Huang et al. (2020) have noted that MPs with dark colours (black and blue) can better adsorb heavy metals and Frias et al. (2010) suggested that white MPs contained less adsorbed persistent organic pollutants than MPs with other colours.

Our data recorded MP fibres in water, sediment and benthic organisms. These results showed the presence of MPs in all three compartments, within both urban/degraded and pristine mangrove ecosystems of Isla del Carmen and highlighted MP bioaccumulation in organisms. Although we expected to find the same shapes and colours in marine sponges, we noted that white fibres were not present in sediment or sponge samples (Figure 6). This suggests that white fibres could remain in the water column because they are made of a less dense material or they are consumed by fish that prevent settling of this material to the sediment and being filtered by marine sponges. Borges et al. (2020) noted that demersal fish collected close to the coast of Campeche Bay ingested more fibres than pelagic species and the main MPs collected in all fish species analysed were white. They attributed these differences to the fact that demersal fish inhabits in sandy bottoms covered by *Thalassia testudinum* where fibre MPs are apparently more abundant than fragments. The Borges et al. (2020) study was conducted in the same region as our study sites and this removal from the water column, may in part explain why we did not find white MPs in marine sponges. Moreover, in this work we did not find MPs smaller than 2 mm which would suggest bioaccumulation in zooplankton as these organisms bioaccumulate MPs < 200 µm. However, sponge predators such as polychaetes, crabs, fish and turtles (Pawlik, 1983; Padilla et al., 2010) are likely to bioaccumulate and biomagnify MPs through the trophic chain.

**Feasibility of using sponges as bio-indicators of MPs pollution in coastal ecosystems**

Marine sponges could be considered good bioindicators of MPs because they are efficient filter feeders (Reiswig, 1971; Riisgård et al., 1993). Their physiology lets them filter between 100 to 1200 mL per hour and per gram of dry weight through their pores and channels (Vogel, 1977). This fact makes them suitable organisms to detect MPs suspended in the water column. Also, they are geographically well distributed, spread along many coastal ecosystems such as coral and algal reefs, seagrasses and mangroves (Batista et al., 2014). Although, there is a lack of information about using marine sponges as MP bioindicators, some works have noted that these organisms can capture microparticles in their bodies. Recently, Girard et al. (2021) proposed sponges as bioindicators for microparticulate pollutants, where MPs were included. They identified 34 different particle types such as polystyrene, particulate cotton, titanium dioxide and blue pigmented particles. Their results showed that marine sponges can incorporate a variety of microparticles inside their skeletal fibres, the ectosome or both depending on the particle size and sponge species. These results suggested that the fluctuation in material ratios found inside the sponges' tissues was a response to the spatial variation of surrounding microparticles. Modica et al. (2020) used a sponge collection from the Cantabrian Sea Maritime museum to detect synthetic textile microfibers in sponges collected 20 years ago in the North-east Atlantic Ocean. They analyzed 170 samples that belonged to 34 sponge families. Their results showed that specimens of all the sponge families examined (n=34 families) were able to retain MPs. They also found that the presence of MPs on the sponge samples were not related to sampling depth (as these particles were present along the whole sampling depth range, from 1 to 23.5 m) nor the ecotope where they were collected (e.g., *Laminaria* spp., rocky wall, rocky shade, cave and sand).

Regarding the bio-accumulation factor (BAF), as we expected, the three sponge species showed values higher than 1 and confirm the ability of these sponge species to bioaccumulate MPs from the two contrasting mangrove environments examined. The higher BAF values recorded in the sponges from the disturbed site could be due to the fact that the organisms had higher MP concentrations than sponges in the pristine site. Regardless of the fact that MP concentration in seawater was not recorded as statically significant between the two sites. Also, the inter-site variation found in MPs concentration in sponges suggests that marine sponges can reflect spatial variations of MPs from places with different pollution level as has been documented in the mussel *Mytilus edulis* from the North Sea (Karlsson et al., 2017). There were no significant differences in the average concentration of MPs among the sponge species studied, thus, these results suggest that these three species exhibit a similar capacity to accumulate MPs in their tissues. A search for information available on MPs in sponges was conducted to compare the concentrations reported in this study with other species collected from the natural environment. Data were only found for the sponge *Hymeniacidon perlevis* (Montagu, 1814) from Lake Veere, Netherlands (Karlsson et al., 2017) and the concentrations reported for this species (25000-57000 particles $kg^{-1}$ DW of sponge) were much higher than those recorded in the sponge species of our study. This lack of information about MP concentrations on marine sponges highlights the importance of our study.

## Conclusion

The results of this study provide evidence that the three sponge species have the capacity to incorporate MPs, and concentrations of this contaminant are reflected by spatial variations in the degree of exposure to potential sources. Furthermore, because they are common, relatively abundant and inhabit accessible areas for monitoring, we consider that their use as bioindicators in integrated monitoring programs for these coastal environments along their distribution range would be feasible and low cost. In addition, given the scarce knowledge about the accumulation of MPs in sponges, it is recommended to continue examining this capacity (as potential bioindicators of MPs) in a wider range of sponge species and from different types of environments.

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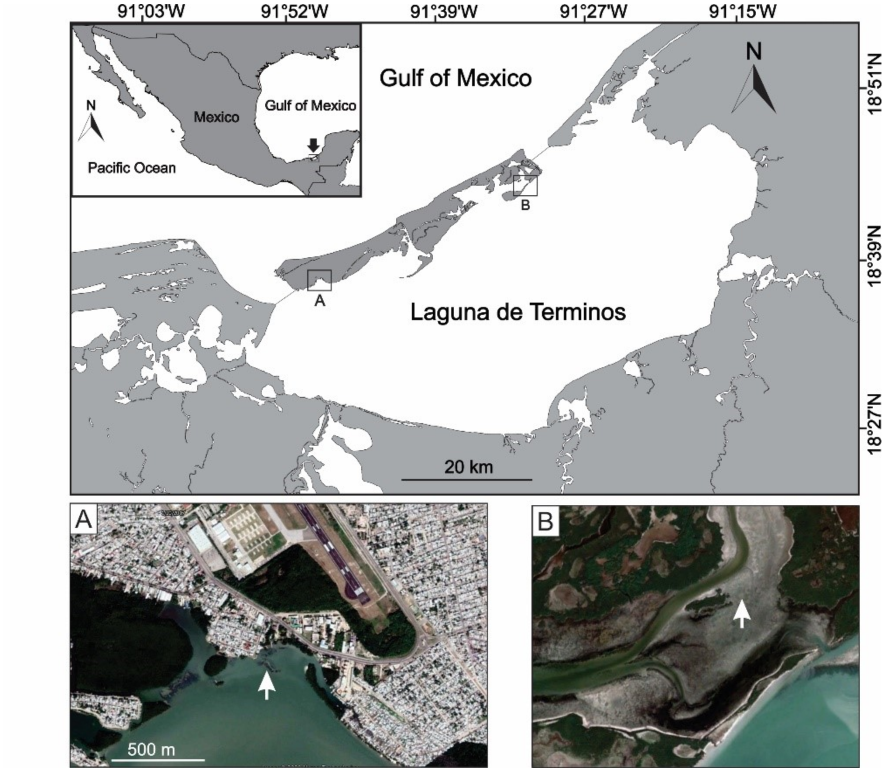
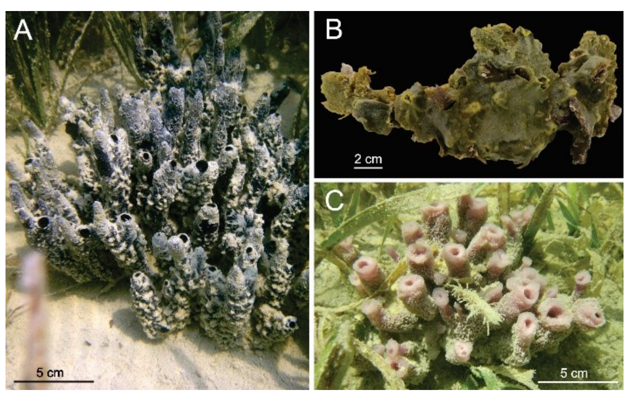
Figure 1 Study area and sampling location



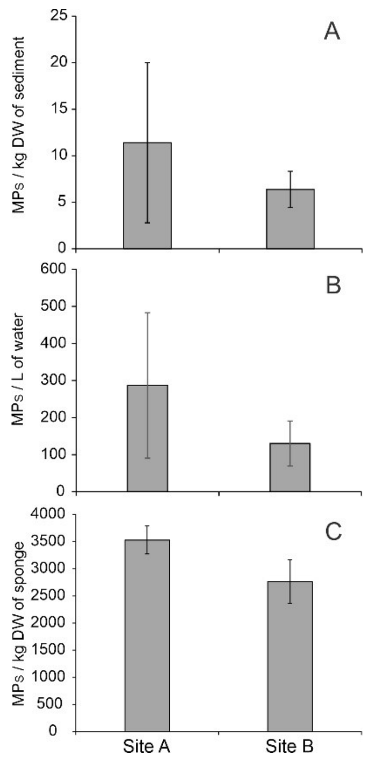

Note: Sampling places identified as follow: (A): Disturbed area (El rastro), (B): Undisturbed area (La deseada).

Figure 2 Sponge species



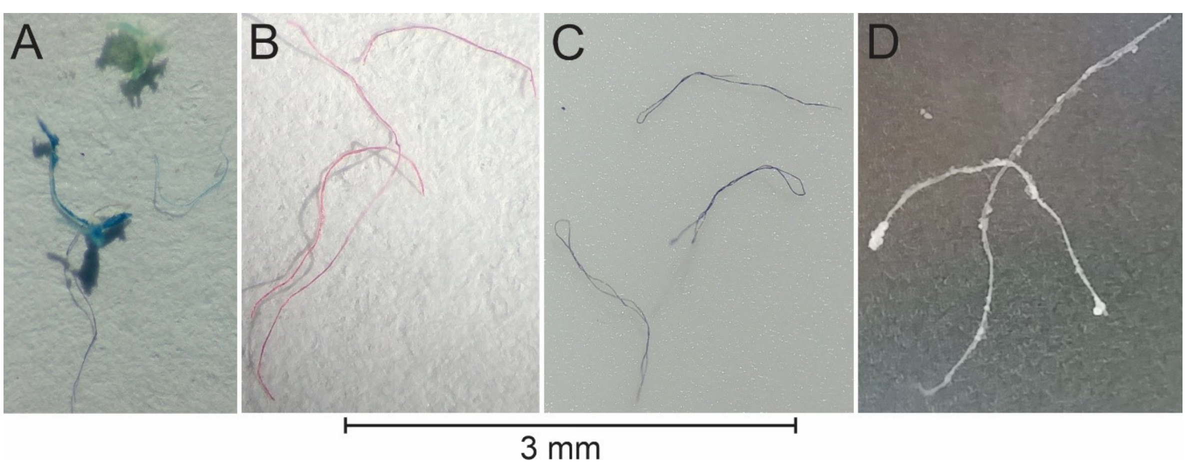

Note: Sponge species used in this work and identified as follow: **A**: *H. melanadocia*, **B**: *A. atlantica* and **C**: *H. implexiformis*.

Figure 3 Average microplastic concentration per compartment



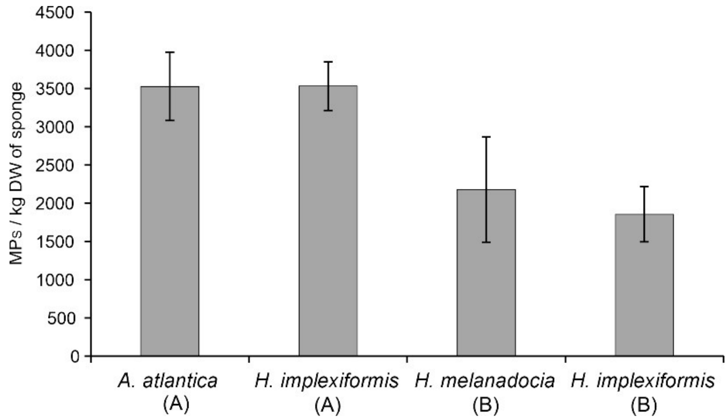

Note: Sampling places identified as follow: (A): Disturbed area, (B): Undisturbed area and DW means dry weight.

Figure 4 Microplastics fibers found in water, sediment and sponges




Note: Type of MPs and colours identified in the three compartments. **A**: Blue fibres, **B**: Red fibres, **C**: Black fibres and **D**: White fibres.

Figure 5 Microplastic concentration between species




Note: Sampling places identified as follow: (A): Disturbed area (El rastro), (B): Undisturbed area (La deseada)