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A systematic review of microplastics in coral reef ecosystems: Abundance, distribution, toxicity, and future research directions

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ABSTRACT

Coral reefs, among the most biodiverse ecosystems providing critical ecological services, are increasingly exposed to microplastics (MPs), an emerging contaminant of concern found throughout freshwater and marine environments. Despite the growing number of studies, the lack of harmonized methodologies and reporting units for MP abundance in coral ecosystems hinder meaningful comparison across studies. This inconsistency poses challenges for ecotoxicology, where coral toxicity studies rely on accurate and comparable environmental data. Here, we adopted a systematic quantitative literature review approach to analyze 125 publications including 69 studies on the abundance of MPs in coral reefs environment and 56 studies on the toxicity of MPs on corals. Various MP forms, polymer types, colors and sizes have been reported. Reported MP concentrations in coral reef environments varied among sampled locations, spanning over six orders of magnitude in water (0.046 to 389,280 MPs/m^3), up to five orders of magnitude in sediment (3.35 to 431.67 MPs/kg), and over two orders in coral tissue (0.03 to 14.18 MPs/g). To date, knowledge of the impact of environmentally relevant MPs on coral health remains limited. However, studies suggest these effects can be both species-specific and depend on polymer type and MP shape. Reported impacts include reduced growth and calcification, altered photosynthetic efficiency, increased mucus production, and in some cases coral bleaching and tissue necrosis. Furthermore, this review highlights the lack of appropriate quality assurance and quality control (QA/QC) protocols and discusses recommendations for future studies to enhance the data quality and relevance.

1. Introduction

Due to its low cost, durability, and broad application, global plastic production has rapidly increased over the past 50 years, with a total estimate of approximately 9200 million metric tons produced (Lyu et al., 2021; Walker and Fequet, 2023). In 2021 alone, more than 400 million tons of plastics were produced (Hadiyanto et al., 2022) and each year up to 23 million tons are estimated to enter the aquatic environment (Horton, 2023). Once in the environment, large plastic debris undergoes photolytic, mechanical and biological degradation, ultimately forming microplastics (MPs), plastic particles smaller than 5 mm in size (Sales-Ribeiro et al., 2020; Ziajahromi and Leusch, 2023). MPs can be classified either as primary, which are intentionally manufactured as micro-sized particles for domestic and industrial such as microbeads in personal care products, pellets for plastic manufacture or industrial scrubbers (Adam et al., 2019; Nava and Leoni, 2021) or secondary, which are formed by degradation of larger plastic items (Burns and Boxall, 2018). MPs can be released directly into the environment via land-based sources such as wastewater treatment plants (Dris et al., 2015; Ziajahromi et al., 2016) or via aquatic-based source such as physical and microbial decomposition of discarded plastics (Bhan et al., 2025). In addition, MPs can become airborne and transported globally in the atmosphere, another crucial pathway to marine ecosystems (Zhang et al., 2020).

Research on the abundance, distribution and impact of MPs in marine environments has been rapidly increasing, with MPs detected in various parts of the ocean including polar regions and deep seas (Khalid et al., 2021). Coral reefs are one of the most productive marine ecosystems and play a crucial role in maintaining high biodiversity (Huang et al., 2021), providing ecological services such as spawning habitat, nursery and feeding areas (Cordova et al., 2018), as well as ecosystem services to humans (Cheang et al., 2018). Coral reefs are being increasingly exposed to MP pollution, which has been recognized as a

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significant threat to the reef ecosystem. A review by Huang et al. (2021) showed that MP abundance in the surface water within those regions reaches tens of thousands of particles/m^3. Nevertheless, factors such as population density, commercial fishing and sewage discharge influence the variations in the abundance (Rahman et al., 2023). MPs can interact with corals in various ways including entanglement, trapping, surface coverage and ingestion (Ashrafy et al., 2023). Due to those interactions, coral reefs have been considered to act as a sink for MPs, with over 10 % of MPs in shallow water being removed yearly (Soares et al., 2023).

Despite growing research in MP pollution, current studies on MPs in coral reef ecosystems are often inconsistent in scope and methodology. Currently, there is a lack of comprehensive synthesis that integrates data on MP abundance, methodological approaches, ecological impacts and study quality specifically within coral reef context. This lack of in-depth understanding limited our ability to effectively compare findings, identify key knowledge gaps and develop relevant and reliable toxicity assessment approaches.

Here we review the current state of the science on (i) abundance and distribution of MPs in coral reef regions; (ii) methods used for sampling and sample processing in coral reef regions; (iii) the impact of MPs on coral species; (iv) experimental design of toxicity studies of MPs on coral species; (v) and quality assurance and quality control (QA/QC) measures. This review will provide critical insight into current knowledge on MPs in coral reef ecosystems and their impacts and identify future research priorities.

2. Methods

A systematic quantitative literature review was undertaken, following Preferred Reporting Items for Systematic Reviews and Meta-Analyses (PRISMA) guidelines. A PRISMA diagram of the selection process is provided in Supplementary Information (Fig. S1).

2.1. Search criteria

Relevant scientific papers from January 2013 to December 2024 were retrieved using the Web of Science and Scopus databases. The primary search keywords included (*microplastic** AND *coral**), combined using the OR operator with the following keywords: *reef*, *plastic*, *plastic debris* and *plastic pollution.* This literature search retrieved 543 publications from Scopus, and 483 publications from Web of Science. Duplicate records were removed, and each title and abstract were screened for relevance. Review articles and peer-reviewed research articles were additionally screened for publications missed during the search. A total of 125 studies were reviewed including 69 studies on the abundance of MPs, and 56 studies on the toxicity of MPs. Detailed inclusion and exclusion criteria are provided in Supplementary Information Section S1.

2.2. Data processing

Data from all publications were recorded in Microsoft Excel. For each abundance study information was recorded about: location (author's affiliation, area of study), sample type (water, inter-tidal, sub-tidal, biota-coral), sampling technique including replication, sample processing (digestion, density separation), identification of MPs (instrument/method used), abundance of MPs, characterization of MPs (polymer type, form, color, size classes). To better visualize sampling locations, sampling sites were zoned based on Large Marine Ecosystems (LME) — noting that six sites were outside. To further visualize MP abundance across the sampled sites, a map was acquired using GIS, based on the preexisting LME layout from Van Graafeiland (2017). For each toxicity study information was recorded about: location (author's affiliation), coral's origin, coral species used, experimental design (co-exposure, duration of exposure, replication, coral feeding during exposure, toxicity endpoints), characterization of MPs used (polymer type, form, concentration), and QA/QC protocols were examined for both types of study.

Due to non-standardized reporting units, concentrations of MPs reported in abundance studies were converted to the most commonly used units, particles/m^3 in water samples and particles/kg in sediment samples, respectively. Nevertheless, for easier comparison between abundance and toxicity studies, the concentration in water and laboratory exposures was also converted to particles/L. When reported as g/L, concentrations were converted to particles/L using previously published equations (Leusch and Ziajahromi, 2021). From each study, only the mean concentration was recorded and if no mean concentration was provided, raw data were extracted to calculate the mean if possible.

In terms of toxicity studies, where it was needed, we normalized results from the reviewed studies to only include plastic particles. Additionally, percentage values were often extracted from supplementary information and/or from figures included in publications (Rohatgi, 2024).

3. Results and discussion

3.1. Abundance of microplastics in coral reef

Among the 69 studies on abundances of MPs in coral reefs, 43, 41 and 20 studies investigated MPs in sediment, water and biota, respectively, and 24 studies reported MPs in more than one matrix (Fig. 1A). MP abundances in coral reef ecosystems were first reported in 2013 (Reisser et al., 2013), but most publications on this topic (60 out of 69) were published after 2019 (Fig. S2).

Studies investigating the abundance of MPs in coral reefs generally sample one of three matrices: water, sediment or coral specimens/tissues. For the purposes of this review, sediment was divided into sub-tidal (n = 30) and inter-tidal zones (n = 13). Of the 69 studies, 45 (65.2 %) sampled only one matrix, 13 (18.8 %) sampled two matrices, often water and either sub-tidal or inter-tidal sediments, and 11 studies (16.0 %) sampled three matrices (Fig. S3). No study sampled all four matrices (water, intertidal sediment, subtidal sediment and coral tissues). Airborne MPs are rarely studied and only a handful of studies have

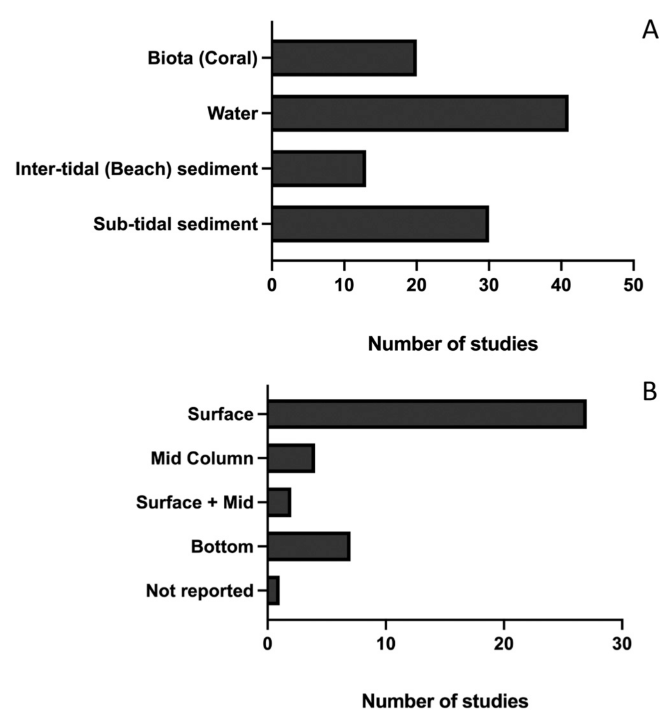

Fig. 1. (A) Number of studies investigating each sample type. (B) From which water column samples were taken.

investigated atmospheric MPs and their role in the distribution of MPs in the marine environment. No studies had yet assessed airborne MPs in coral reef areas, and this is therefore recommended in future studies.

Water samples were mainly collected from the surface (65.8 %, n = 27) or close to the benthos (17.1 %, n = 7) (Fig. 1B). While higher density MPs such as polyvinyl chloride (PVC) (density 1.4 g/cm^3) and polyethylene terephthalate (PET) (density 1.38 g/cm^3) are generally expected to disperse to deeper water and low density MPs such as polystyrene (PS) (density 1.06 g/cm^3) and polypropylene (PP) (density 0.9 g/cm^3) tend to float, environmental process such as biofouling (the accumulation of microorganisms on plastic surface) can alter their buoyancy (Al Nabhani et al., 2022). Therefore, it is important to collect water samples from various depths to accurately estimate the distribution of various MP in water (Al Nabhani et al., 2022; Cai et al., 2022). Therefore, future studies should collect samples from various depths ideally at the same location.

When quantifying MP abundance within corals, most studies collected several coral species. The most common coral genera sampled were *Acropora* (n = 12), *Pocillopora* (n = 12), *Galaxea* (n = 11), *Porites* (n = 9), and *Montipora* (n = 8) (Fig. S4).

A large percentage of studies were performed in Asia (54.9 % n = 39), while only one study was published in Africa, highlighting a knowledge gap in particular for East Africa's coral ecosystems (Fig. S5). Most studies have been done in the South China Sea, Bay of Bengal and Indonesian Sea, where a high MP abundance was also reported (Fig. S6).

The concentration of MPs varied among LME's. The differences in reported values were largely dependent on the sampling approaches and the specific location from which the samples are taken. For example, locations with high tourism and/or large population are expected to have high MP concentrations. Nevertheless, studies use a variety of reporting units and comparison of results is difficult. Only studies using the most used unit were included when comparing the abundance (see Fig. 2 and Table S1).

In water, the highest abundance of MPs was reported in the South China Sea (389,280 particles/m^3; MPs < 350 µm (Yusof et al., 2023). In comparison, sites within Insular Pacific–Hawaiian LME had the lowest abundances of MPs (0.046 particles/m^3; MPs < 330 µm) (Axworthy et al., 2024). Additionally, abundance varies across different sampling depths, with generally higher concentrations found in surface water (Fig. S7). The concentration of MPs in sediment ranged between 3.35 particles/kg in Pacific Central-American Coastal LME (Pérez et al., 2022) to 682 particles/kg in South China Sea (Lin et al., 2024). In coral specimens, the lowest and highest abundance were both reported in the Indonesian Sea, 0.0122 (Sabdono et al., 2022) and 14.18 (Wijayanti et al., 2024) particles/g, respectively. The ingestion of MPs in corals is highly influenced by polyp sizes and feeding strategies (Pantos, 2022; Reichert et al., 2019).

3.1.1. *Sampling technique*

A wide range of sampling techniques have been used for detecting MPs in the marine environment. While these methods have been comprehensively discussed in previous studies, the most common approaches include bulk and volume-reduced sampling for water, and various coring or grab methods for sediments, depending on the depth and zone of collection (Hidalgo-Ruz et al., 2012; Wootton et al., 2024). 60 % of studies that sampled MPs in water used towed nets to collect samples, with only 14.6 % of studies conduct bulk sampling (Fig. 3A). A significant relationship has been identified between MP size and abundance, with an increasing number of particles with decreasing particle size (Leusch et al., 2023). Thus, the size of the net will have a significant effect on the concentration of MPs reported, with smaller net sizes resulting in higher particle counts. There is no standard mesh size for tow nets with most studies using a mesh size between 300 and 400 µm (Fig. 3B). Sediment samples on the other hand are commonly collected directly into sampling container using a shovel or a spoon (Fig. 3C, D).

Coral sampling approaches varied widely across studies investigating MP contamination in corals. Of the 20 studies investigating MPs in corals, half examined both internal accumulation and external

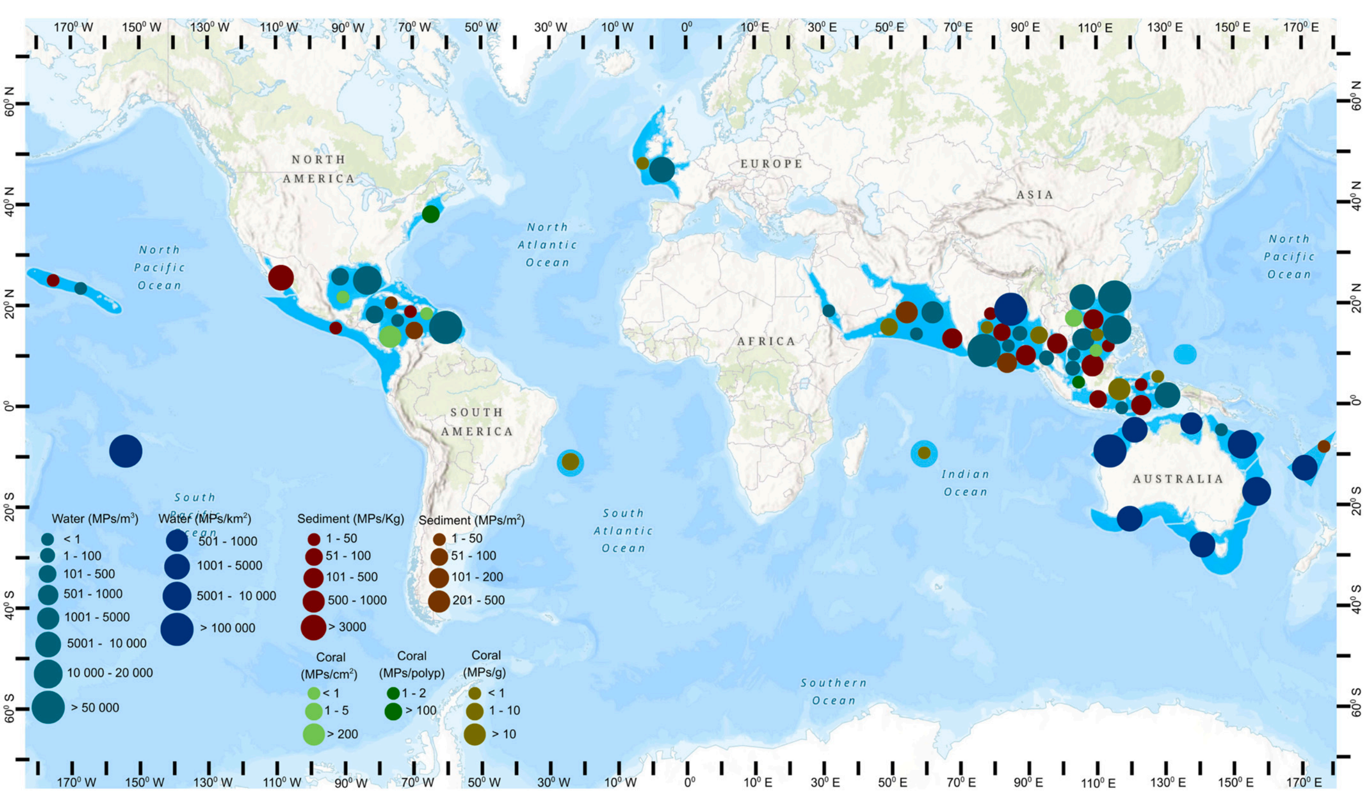

Fig. 2. Global abundance of MPs in coral reef ecosystems. The light blue polygon represents the sampled locations, within and outside LME's. Map acquired using GIS. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

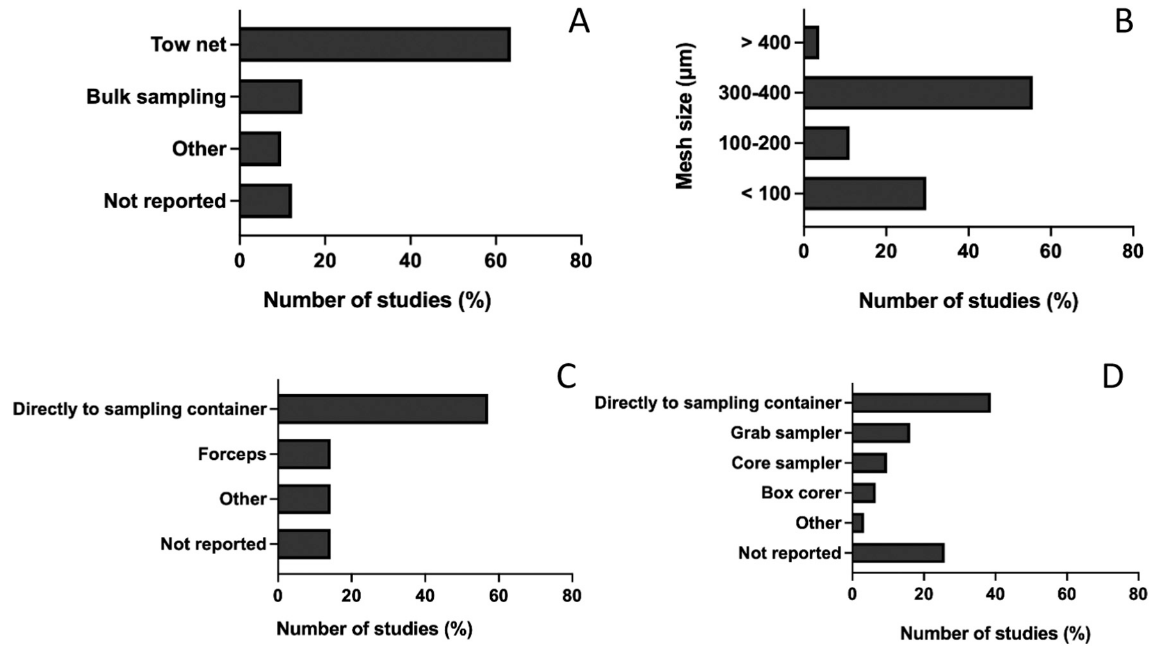

Fig. 3. (A) Sampling method to collect water, other indicates the use of hydrophore, pump or ROV. (B) Towed net mesh size used to collect MPs from water. (C) Sampling method for intertidal sediments, other indicates sieving method. (D) Sampling method for subtidal sediments, other indicates the use of cut-off syringe.

absorption of MPs, while the remaining half investigated internal accumulation only. Most corals are non-selective suspension feeders, and ingest MPs from their surrounding environment (Savinelli et al., 2020). MPs were found retained in mesenterial tissues and gastrovascular tissues (Grillo et al., 2021; Hall et al., 2015; Okubo et al., 2020; Rotjan et al., 2019). Several different coral genera were collected among the studies (Fig. S4) which resulted in a varied number of coral specimen sample collected across species, ranging from fewer than five to 131 (Fig. S8).

3.1.2. *Sample processing*

Depending on the sample type the process of extracting MPs generally consists of digestion, density separation and filtration. Digestion is especially important to remove organic material attached to MPs that could otherwise impact the accuracy of MPs identification (Wootton et al., 2024). Detailed information on sample processing is presented in the Supporting Information Section S2.

Processing of coral samples depends on the research objectives, and whenever internal accumulation or external adsorption of MPs is examined. For internal accumulation, corals (tissue and skeleton) are often digested using various agents with majority of studies ($n = 7$) used HCl or a combination of several digestion agents such as KOH + HCl or HCl + sodium hydroxide (NaOH) (Fig. S9A). Despite HCl being the most commonly used, its effectiveness to digest corals requires further specific evaluation. Use of KOH has been popular among studies investigating MPs in biota due to its high digestion efficiency without damaging MPs making it a suitable digesting agent for soft animal tissues (Lu et al., 2024; Prata et al., 2019). However, its effectiveness for coral samples may be limited, as KOH does not digest the calcium carbonate skeleton, and may be more appropriate when only soft coral tissue is targeted. KOH has been identified as the most economical and least time-consuming when used to extract MPs from bivalve tissue (Thiele et al., 2019). However, KOH has only been used in 15 % ($n = 3$) of reviewed studies (Fig. S9A). A combination of H_2O_2 and KOH, which was used by one study, was found to be the most effective to extract MPs from fish and mussel samples (Tsangaris et al., 2021). To date, there is insufficient knowledge on the method suitability for MPs extraction specifically for corals. Recently, HNO_3 was found to be an effective digestion agent with efficiency of 99 % (Santana et al., 2022). The study showed 70 % HNO_3 have 100% recovery rate for commonly found polymer types (Santana et al., 2022). Despite the apparent benefits of HNO_3, none of the reviewed studies used it as a digestion agent.

To investigate the adsorption of MPs on the external part of corals, corals were often sonicated in a density separation agent. Five studies used NaCl in combination with sonication, while two studies added digestion using H_2O_2 as well (Fig. S10B). Similarly to the water or sediment sample processing, filtration was often the last step with 50 % ($n = 10$) of studies using filter/mesh with pore size <1 µm (Fig. S10C).

To differentiate between plastic and non-plastic particles, both instrumental and non-instrumental techniques are available. Most studies (72.5 %, $n = 50$) used Fourier Transform Infrared (FTIR) spectroscopy while Raman or both instruments were used only occasionally (Fig. 4). Those instrumental techniques produce a spectrum for individual particles that are compared with a reference spectrum, to identify the polymer (Cutroneo et al., 2020). As this process can be time consuming, studies often use only a fraction of the extracted particles for identification. The remaining studies either did not characterize MPs further or used a non-instrumental technique, hot needle test, by placing a hot needle on or close to a particle to assess whether it melts and, therefore, is plastic (Patti et al., 2020). This test does not provide information on chemical composition of the particle and should not be considered an identification technique.

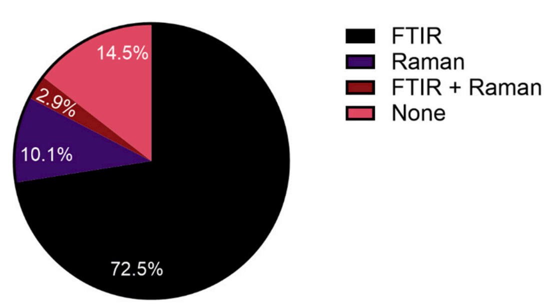

Fig. 4. Identification instrument for MPs used as a percentage of reviewed studies. “None” also includes 3 studies that used the hot needle test.

3.1.3. Characteristics of microplastics

Information on the chemical and physical properties of MPs including MP form, polymer type, color and size is critical for identifying their origin. Such data also provide essential baseline parameters for toxicity assessments. However, the reporting is often inconsistent between the studies. Few studies provide percentage values, and particles made of natural material such as cellulose are often included in the results.

Chemical composition is crucial to understanding the source of a particle, and yet 26.3 % (n = 10), 29.3 % (n = 12) and 15 % (n = 3) of studies on sediment, water and biota, respectively, did not report polymer type in their results. Polyethylene (PE) was the most abundant polymer type in sediment and water samples, but not in corals, accounting for 27.8 %, 28.6 % and 16.3 % respectively (Fig. 5). PE is a widely produced polymer, often used in single-use disposable plastics (Kalčíková et al., 2020). Polyethylene terephthalate (PET) was the most abundant in coral tissues, accounting for 34.6 % (Fig. 5). PET was also highly abundant in sediment (23.2 %) and water (25.3 %) samples (Fig. 5). PET is commonly used in synthetic fabric, and is primarily released into the environment through wastewater discharge (Dris et al., 2015; Ziajahromi et al., 2017). PET has a high density of 1.38 g/cm^3, and as mentioned in Section 3.1.2 the processing method could affect the recovery and thus reported abundance of PET. Polypropylene (PP) is another polymer that is widely used in today's society. This is reflected in the findings of the reviewed studies with PP abundance of 26.3 % in sediment and 22.4 % in water (Fig. 5). In comparison, only a small proportion (9.4 %) of PP was reported in coral specimens (Fig. 5).

Several MP forms are present in the environment. Granules, beads and pellets are spherical particles, and many studies do not differentiate between them. Similarly, fibers and filaments are often reported together. For this review and easier comparability, granules were classified as "beads", and as filaments were only reported in five studies they are included here as "fibers". More than half of the reviewed studies (n = 12) on corals, and a small portion of studies on sediment (n = 5) and water (n = 8) did not report the form of MPs. Where form was reported, fibers were the most abundant across all sample matrices, accounting for 56 % (sediment), 57.9 % (water) and 75.1 % (coral) (Fig. 5). Fragments were also highly abundant, having 27 % (sediment), 25.2 % (water) and 16.8 % (coral) (Fig. 5). Other reported forms included film, foam, pellets and beads. Beads in particular were the least abundant (<1 %) for the three sampling types. Beads are primarily used in domestic products such as personal care or cosmetics (Li et al., 2018; Liu et al., 2019). However, countries have been implementing bans around the use of beads, which could explain the reported low abundance.

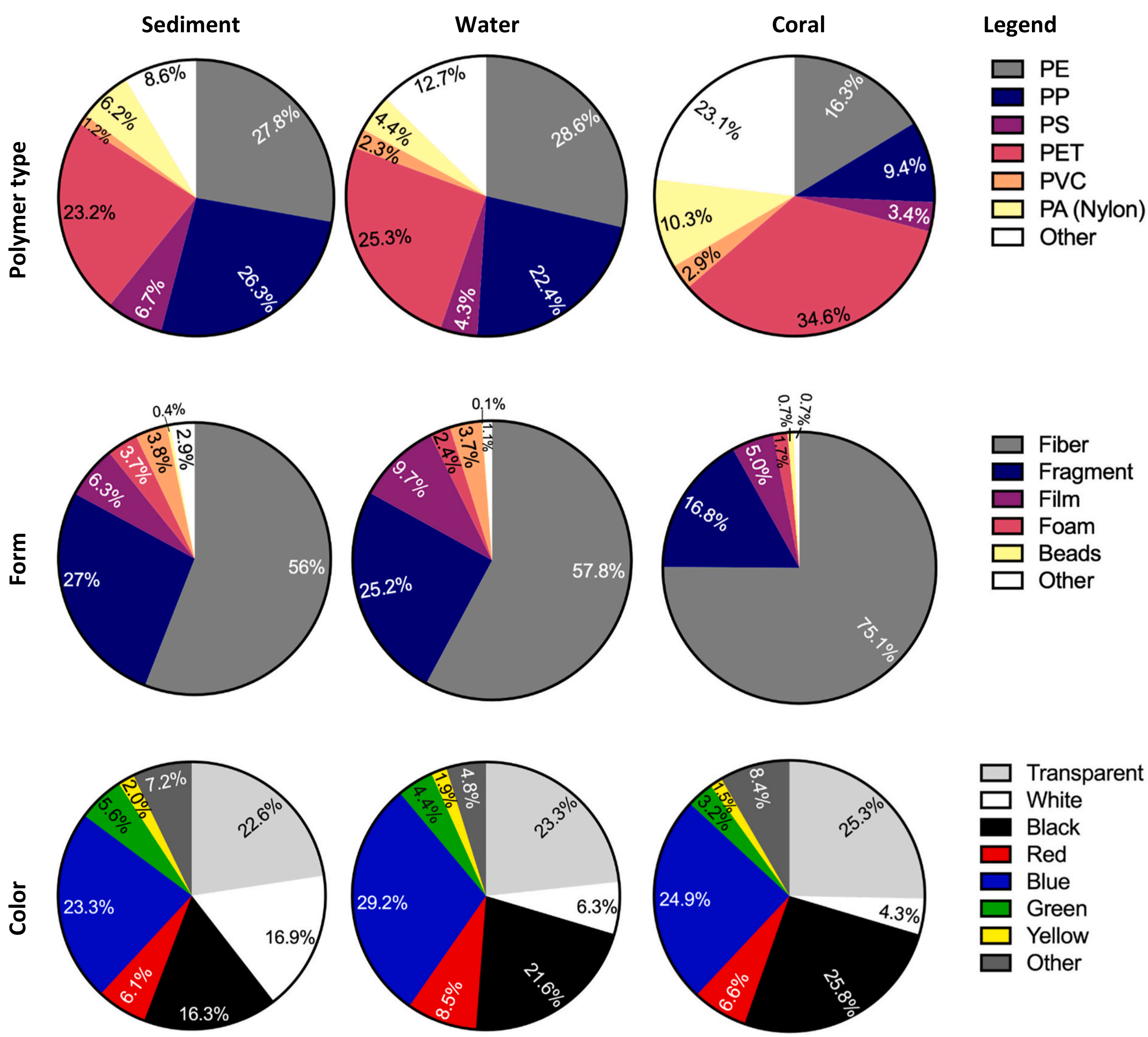

Fig. 5. Characterization (polymer type, form, color) of MPs found in sediment (left), water (middle) and coral specimen (right). The polymer type was not reported in 10 (sediment), 12 (water) and 3 (coral) studies. The form was not reported in 5 (sediment), 8 (water) and 12 (coral) studies. The color was not reported in 13 (sediment), 12 (water) and 4 (coral) studies. One study reported color of MPs particles together for sediment, water and biota.

Color of MPs is the least reported among the studies and was not reported by 32 % (n = 22) of reviewed studies. Blue fragments were the most common, compromising of 23.3 % (sediment), 29.2 % (water), and corals (24.9 %) of the total MPs reported (Fig. 5). Followed by transparent with 22.6 % (sediment), 23.3 % (water) and 25.3 % (corals), white (16.9 %) and red (16.3 %) (Fig. 5). Some studies categorized colored particles simply as "colored" without providing the specific color or having a range variety of colors under the category "other".

MPs are most commonly defined as plastic particles between 1 and 5000 μm (Coady et al., 2020). The size of reported MPs is heavily influenced by the filter pore size used for sample collection and processing (Ziajahromi and Leusch, 2022). Smaller MPs (<300 μm), which are more abundant in the environment, are often not sampled due to the use of larger filter/net pore size, especially in water sampling. Despite this, MPs below 1 mm are reported as more abundant than MPs in 1–5 mm size range (Fig. 6). Overall, out of 69 reviewed studies, 33 studies reported particle size and only 8 studies reported mean size detected. MPs below 1 mm were most abundant in all three matrices (Fig. 6). Another important factor impacting the reported size of MPs is the instrumental technique used for MP identification. For instance, FTIR-based methods do not detect MPs between 1 and 20 μm. Therefore, it is almost certain that smaller MPs (1–20 μm) are significantly underreported due to these methodological limitations.

3.2. Toxicity studies on corals

The number of studies investigating the impact of MPs on corals has been increasing over the past few years, and in particular since 2020 with 41 studies (out of 56) published in the last 4 years (Fig. S2), with majority (n = 24) conducted in Europe (Fig. S5). Ingestion of MPs by corals (in *Dipsastrea pallida*) was first reported in 2015 (Hall et al., 2015). The study showed corals are capable of ingesting MPs (100 μm–2 mm) at a rate up to ~50 μg particles $cm^{-2}\,h^{-1}$, and able to retain particles for up to 24 h. MPs have since been reported to physically damage tissue (Reichert et al., 2018), reduce growth and alter photosynthesis (Reichert et al., 2019).

Feeding activity has been studied the most (60.7 %, n = 34), followed by photosymbiont state (35.7 %, n = 20) and impact on coral's health (28.6 %, n = 16), whereas trophic transfer has only been studied twice, with both studies examining the transfer of zooplankton contaminated with MPs to corals (Fig. S11).

The environmental relevance of MP exposure studies on corals, is limited by the use of pristine MPs that do not reflect the complexity of those found in natural environment, which may limit the ecological relevance of their findings. Key factors such as particle size, shape, polymer type and surface properties – including weathering and aging – play a crucial role in determining MP behavior and toxicity. Aged particles for instance have altered surface chemistry, increased sorption capacity for pollutants and different biological interactions compared to pristine (Leistenschneider et al., 2023). Additionally, studies utilize many different units to report MP concentration, which makes it hard to compare between studies. Over half of reviewed studies (53.8 %, n = 28) used concentrations well above environmentally relevant levels (Fig. 7). The remaining studies, but one, used concentrations at the higher end of environmental relevance found only in the most polluted coral reefs, for instance 30.8 % (n = 16) used high concentrations (50–400 particles/L), and 13.5 % (n = 7) used medium concentrations (1–50 particles/L; Fig. 7). Therefore, incorporating environmentally realistic exposure conditions - including representative MP characteristics and concentrations - is essential for improving the ecological relevance of future studies and ensuring accurate risk assessments.

Corals are exposed not only to MPs but also to other environmental stressors such as heat, acidity, and other anthropogenic pollutants. Therefore, it is crucial to investigate the impact of MPs in co-exposure with other environmental stressors, particularly considering the existential threat of global warming. Yet most studies (82.1 %, n = 46) only investigated the effect of MPs alone with remaining studies co-exposed corals to MPs and heat or investigated the effect of MPs in combination with other chemical pollutants such as heavy metals, and pH (Fig. S12).

3.2.1. Experimental design

The effect of MPs was studied on various coral species, and 18 out of the 56 studies investigated the impact of MPs on more than one coral species. The most common species used in toxicity experiments were *Acropora* (35.7 %, n = 20) and *Pocillopora* (28.6 %, n = 16) (Fig. S13). Coral specimens were generally collected from the field (61.1 %, n = 36) (Fig. S14).

Corals were acclimated to laboratory conditions for between 7 h and 6 months. Over half the studies (58.6 %, n = 34) exposed corals to a short-term MP exposure (≤7 d) with majority (61 %, n = 21) had an exposure duration of less than 48 h (Fig. 8B). Conversely, 24 studies (41.4 %) investigated longer-term effect on corals (>7 d), most (41.7 %, n = 10) with an experiment duration over 8 weeks (Fig. 8C). Jiang et al. (2020) exposed *Protopalythoa* sp. to short-term (four days) and long-term (28 days) MP using PE and PVC beads, and polymethyl methacrylate (PMMA) fragments. After short-term exposure only chlorophyll-content was significantly increased in PE treatment, but long-term exposure to PE and PMMA significantly decreased chlorophyll, and a decreased Symbiodiniaceae density in all three MP treatments in long-term exposure (Jiang et al., 2020). This illustrates the importance of more realistic long-term exposure conditions.

Replication is essential in toxicity studies, but eleven studies (19.6 %) did not report replicate numbers, and five studies (8.9 %) reported no replication (Fig. 9). The most studies reported using three replicates.

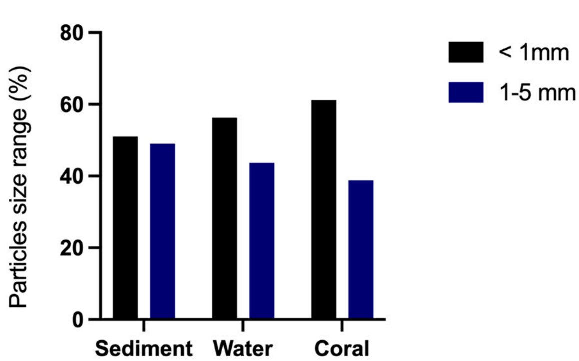

Fig. 6. Percentage of MPs particle size ranges for sediment, water and coral specimen samples.

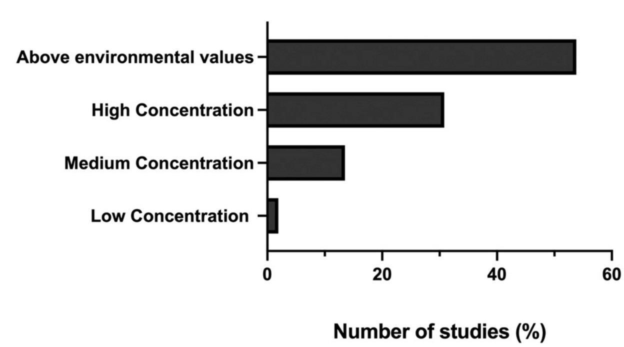

Fig. 7. Concentration used in toxicity studies reported as percentage, with low (<1 MPs/L), medium (1–50 MPs/L) and high concentrations (50–400 MPs/L) referring environmental concentrations. Ten studies used concentrations within two separate categories. One study had concentrations wihtin three separate categories. Seven studies provided concentation in mg/L, one study provided in mL and one in mg, and those studies were unable to convert the units to particles/L. Two studies directly fed one particle to coral.

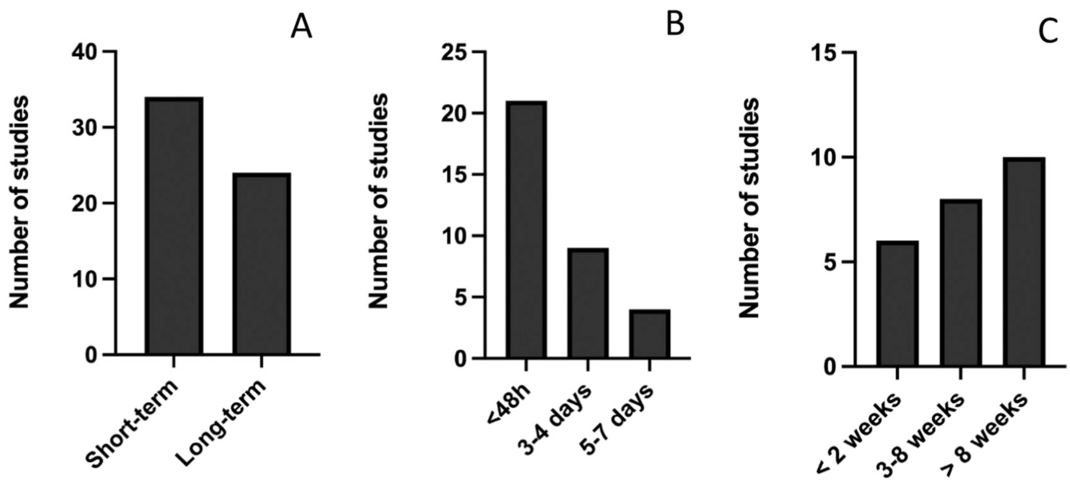

Fig. 8. Length of exposure. (A) Number of studies conducted short-term (≤7 days) and long-term (>7 days) exposure. (B) Length of short-term exposure per number of studies. (C) Length of long-term exposure per number of studies.

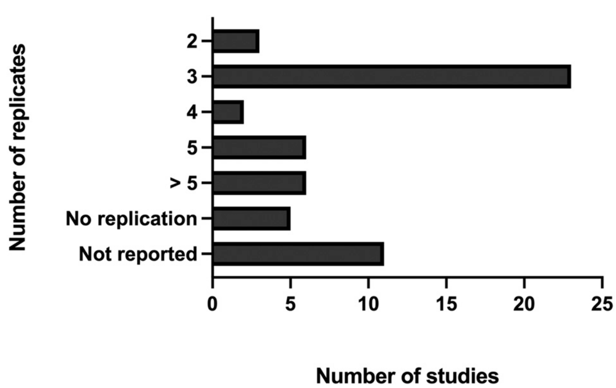

Fig. 9. Number of replicates per number of toxicity studies. Samples were only considered a replicate if the samples were under separate conditions, sudo-replicates were considered as no replication.

Feeding during exposure can affect the outcome of toxicity studies, allows for longer experiments and can arguably be more representative of natural conditions. During treatments coral specimens were either not provided with food, fed or were receiving food via connected systems (indirect feeding). Most studies (39.7 %, n = 23) fed coral specimen during exposure, while eight studies (13.8 %) did not (Fig. S15). In five studies, corals were fed indirectly via connected mesocosm systems. But many studies (37.9 %) did not report whether corals were fed or not (Fig. S15). Corals are more likely to ingest more MPs if not fed through the exposure. Savinelli et al. (2020) observed *Astroides calycularis* ingested significantly more natural food than aged PE fragments. Supported by Reichert et al. (2024a) where several coral species ingested natural food more frequently than pristine or biofouled PE particles. In contrast, Rades et al. (2022) reported no preferential feeding in four coral species *Acropora muricata*, *Pocillopora verrucosa*, *Porites lutea* and *Heliopora coerulea*. Nevertheless, studies should reflect natural conditions and therefore (reasonable) feeding during exposure is recommended, as the presence and absence of food can influence ingestion rate and experimental outcome.

3.2.2. Experimental design — characteristics of microplastics used

Most experimental designs do not represent environmentally relevant MPs. MPs do not occur in the environment on their own or in a pristine state, but rather undergo several natural processes that alter their properties, behavior and fate (Kalčíková et al., 2020). Kedzierski et al. (2018) investigated how plastic aging changes three different polymer types (PVC, PET and polybutylene adipate terephthalate). The study found that some plastics can release estrogenic compounds and absorb heavy metals with aging. Most studies (60.6 %, n = 40) used pristine particles (Fig. 10A). The remaining studies used aged particles, using different approaches for aging such as submerged in seawater prior the exposure, MPs that were aged naturally, often directly collected from beach sediments, or artificially aged (Fig. 10A).

MPs can also act as a carrier for toxic chemicals such as persistent organic pollutants (Hazeem et al., 2020), heavy metals (Shen et al., 2019), organic pollutant (Wang et al., 2020), pathogens (Fallon and Freeman, 2021), and other contaminants that may be associated with MPs such as plasticizers (Montano et al., 2020). For instance, study investigated five phthalate esters (PAEs) and observed >95 % of collected scleractinian corals were contaminated, with an average concentration of 30 ng/g per coral (maximum 173.4 ng/g) (Montano et al., 2020). In addition, MPs can attach microorganisms and form biofilm which can increase the particle density (Rehse et al., 2016). For example, Kaiser et al. (2017) incubated PS and PE MPs in coastal seawater to identify how biofouling changes their sinking behavior, and observed an 81 % increase in sinking velocity of biofouled PS. The presence of biofilm can also affect palatability of the MPs, which can enhance feeding response in *Viminella flagellum* with biofouled MPs compared to clear MPs (Pereira et al., 2024). Yet biofilm is often ignored in MP toxicity studies, with most (77 %, n = 47) using particles without biofilm (Fig. 10B). Of the handful of studies that did use biofouled MPs (23 %, n = 14), six conducted exposures using both particles with and without biofilm. It is important to note that during long-term exposures biofilm may naturally form, however most long-term studies did not report on the formation of biofilm.

The most common polymer type used in studies investigated the impact of MPs was PE, corresponding to 47.0 % (n = 39) (Fig. 10C). The use of PE is important as it is the most abundant polymer type detected in coral reef ecosystems (Fig. 5). However, studies should avoid using less relevant particles made of PS and PVC as those polymer types are not commonly detected in natural environment with PS and PVC accounting for 3.4–6.7 % and 1.2–2.9 % detected in water, sediment and corals (Fig. 5). If PS and PVC particles are use, the size of MPs needs to be considered. Both PS and PVC have been detected in marine environment as nano-sized particles (<1 μm), ranging from 1 % to 9 % (PS) and from 1 % to 70 % (PVC) (Materić et al., 2022; Ter Halle et al., 2017). However, the knowledge on abundance of nanoplastics is limited. For instance, PE significantly decrease the chlorophyll-a content, compare to PVC at the same concentration and size where no significant effect was observed (Jiang et al., 2020). In addition, plastic particles occur as mixtures of different polymers, but only a few studies have so far

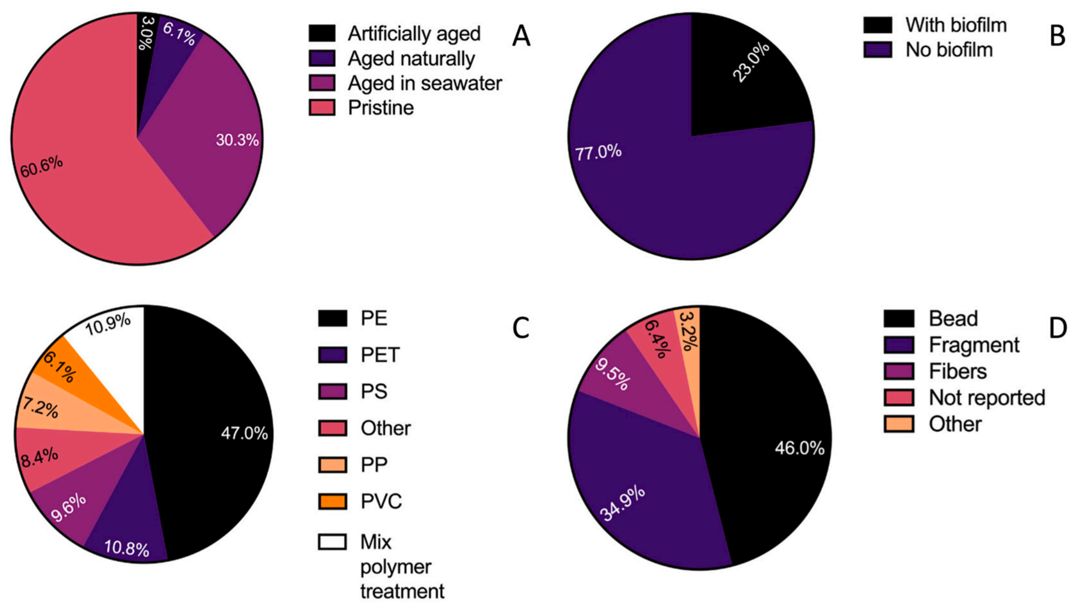

Fig. 10. Characterization of MPs particles used in toxicity studies. (A) Percentage of studies that used pristine or aged MPs. Aged naturally refers to particles that were collected in the environment. Aged in seawater refer to particles that were aged in during a long-term exposure or through prior soaking in seawater. Artificially aged refers to a particles that were aged via UV light or in artificial environmental conditions. (B) Percentage of studies that included formed biofilm on MPs. The study had to confirm whenever a biofilm was formed. In some studies, biofilm was formed during long-term exposure. Six studies used MPs that were with and without biofilm. (C) Polymer type of MPs used. Six studies used two, fours studies used three, three studies used four and one study used six different polymers. (D) MP form used. Other includes powder and pellets. Eight studies used two different forms.

investigated the impact of mixtures of MP polymers (10.9 %, n = 9) (Fig. 10C).

The use of appropriate forms of MPs is also critical to correctly assess the toxicity of MPs. Beads were the most common form of MPs used in the reviewed studies, accounting for 46 % (n = 29). Since beads are rarely detected (<1 %) in the marine environment, particularly in coral reefs, the beads should not be used in toxicology studies. Fibers are the most common form of MPs found in the coral reefs (Fig. 5), however only a few studies (9.5 %, n = 6) used fibers in their experiments (Fig. 10D). Fragments, the second most common MPs form (Fig. 5), were used in 34.9 % (n = 22) studies (Fig. 10D). MP form is an especially important factor when considering experimental design. Corals have been shown to be impacted differently when exposed to different MP forms. For example, fibers reduced photosynthetic efficiency in *Acropora* sp., whereas beads at comparable concentrations increased photosynthetic efficiency in *Seriatopora hystrix* (Mendrik et al., 2021). This can also vary between species: fibers were observed to have more significant impact on *Pocillopora verrucosa* compared to *Stylophora pistillata* in a recent study (Reichert et al., 2024b). These studies not only prove that effects can be form-specific but also species-specific.

3.2.3. Impact of microplastics on corals

Despite a recent growth in the number of studies, the true impact that MPs may have on corals is still largely understudied. The impact of MPs on corals is complex, and the results of toxicity experiments are influenced by experimental design. In particular, exposure concentration and duration have a dramatic impact on possible effects, and many studies use extremely high, environmentally unrealistic, exposure concentrations. Overall, MPs have been shown to physically damage tissues (Isa et al., 2023), reduce growth (Rades et al., 2022), alter photosynthesis (Rocha et al., 2020), cause oxidative stress (Tang et al., 2024) and coral bleaching (Reichert et al., 2018). The toxic impacts of MPs on corals is very species-specific and may be influenced by factors such as polyp and prey size, and feeding strategy, which leads to some species being impacted more than others (de Oliveira Soares et al., 2020; Pantos, 2022; Reichert et al., 2019). For example, a study exposed *Acropora cervicornis* and *Pseudodiploria clivosa* to PE fragments of various sizes (at a very high concentration of 30 mg/L). Buoyant weight was significantly reduced in *A. cervicornis*, while no such effect was observed in *P. clivosa* (Hankins et al., 2021). Similarly, when *Lophelia pertusa* and *Madrepora oculate* were exposed to biofouled 500 μm PE beads (at a high concentration of 350 particles/L), MPs altered growth rate of *L. pertusa* but not *M. oculate* (Mouchi et al., 2019).

Passive adhesion of MPs is the primary mechanisms by which corals sequester MP from the water column (Table S2) (Corona et al., 2020; Martin et al., 2019). While adhesion is the dominant mechanism, MPs can also interact with corals actively through ingestion, and partial or complete egestion. Rades et al. (2022) exposed four coral species (*Acropora muricata*, *Pocillopora verrucosa*, *Porites lutea*, and *Heliopora coerulea*) to biofouled PE fragments (184 μm) at 200 and 1850 particles/L for 15 months exposure with 24 h pulse feeding. The study confirmed no preferential feeding, and showed corals feed on both their natural food and MPs (Rades et al., 2022). But other studies have contradicted these findings and reported corals feeding more on natural food, and no impact of biofilm after exposing various coral species to pristine and biofouled PE fragments (100–355 μm) (Table 1) (Reichert et al., 2024a). Similarly, Bejarano et al. (2022) observed *Stylophora pistillata* preferentially feeding on natural prey compared to PET fragments. Irrespective of food preference, certain MPs are more likely to be ingested than others, as influenced by individual coral species morphology and feeding behavior. Reichert et al. (2024b) exposed *P. verrucosa* and *S. pistillata* to various MPs (Table S2) and reported *P. verrucosa* preferably ingested mainly PE fragments with ~65 % of particles egested. On the other hand, *S. pistillata* ingested mainly MP fibers (mixture of acrylic, PA, PES) with 100 % egestion (Reichert et al., 2024b). A similar observation was reported by Corinaldesi et al. (2021) where *Corallium rubrum* exposed to a mixture of different polymer types (Table S2) showed preferential feeding of PP fragments (20 μm–1 mm). Those experimental feeding exposures indicate some coral species can distinguish between MPs and natural prey.

Corals can egest MPs and only a fraction of ingested MPs are ultimately retained by coral, however egestion of MPs is energetically costly for corals (Utami et al., 2021). Table S2 provides a summary of studies investigating the feeding activity in corals.

Table 1
Summary of impact of MPs on coral specimens after short-term exposures (≤7 d).

Species	Microplastics	Concentration	Duration of exposure	Results	Reference
Montastraea cavernosa, *Orbicella faveolata*	Biofouled PE beads (90–106 μm, 425–500 μm, 850–1000 μm)	30 mg/L (~60–83,611 MPs/L*)	48 h	No effect on calcification observed	Hankins et al. (2018)
Favites chinensis	PS beads (3 μm)	Not reported	72 h	MPs suppress the uptake of *Symbiodinium* into the host	Okubo et al. (2018)
Pocillopora damicornis	PS beads (1 μm)	50 mg/L (9×10^{10} MPs/L)	24 h	No significant effect on symbiotic zooxanthellae density MPs caused significant increase in chlorophyll content at 12 h of exposure MPs significantly increased antioxidant enzymes activities	Tang et al. (2018)
Acropora tenuis	PP fragment (0.5, 1, 2 μm), PE beads (1.6 μm)	PP: 5 MPs/L, 15 MPs/L, 50 MPs/L PE: 25 MPs/L, 100 MPs/L, 200 MPs/L	2.5 h 48 h	Lower fertilization when exposed to the largest weathered MPs, and effect was not concentration dependent No significant effect on embryo development and larval settlement	Berry et al. (2019)
Zoanthus sociatus	Aged PE fragments (63–125 μm), PVC fragments (63–125 μm)	1 mg/L (PE: ~1040–8125 MPs/L; PVC: ~708–5534 MPs/L*) 10 mg/L (PE: ~10,402–81,255 MPs/L, PVC: ~7085–55,347 MPs/L*)	96 h	MPs-specific impact observed PVC (10 mg/L) increased photosynthetic efficiency, lipid peroxidation and antioxidant defenses	Rocha et al. (2020)
Porites porites	PS pellets (<20 μm)	1 mg/L (~1.5×10^7 MPs/L*) 10 mg/L (~1.5×10^8 MPs/L*) 100 mg/L (~1.5×10^9 MPs/L*) 1000 mg/L (~1.5×10^{10} MPs/L*)	96 h	No significant difference in catalase enzymatic activity or melanin content for all concentrations	Grillo et al. (2021)
Protopalythoa sp.	PE beads (1–40 μm), PVC beads (1–40 μm), PMMA (1–20 μm)	50 mg/L (PE: ~1.3×10^7 MPs/L; PVC: ~8.6×10^6 MPs/L; PMMA: ~1×10^7 MPs/L*)	4 d	MPs-specific impact Significant increase in chlorophyll-a content only in PE treatment	Jiang et al. (2020)
Tubastrea aurea	PE beads (1–10 μm), PET beads (1–10 μm), PVC beads (1–10 μm), PA66 beads (1–10 μm)	300 mg/L (PE: ~4.9×10^9 MPs/L; PET, PVC: ~3.3×10^9 MPs/L; PA66: 4×10^9 MPs/L*)	24 h	MP exposure decreased energy metabolism, antioxidant status, immune system, and calcification process of *T. aurea* MPs cause significant decreases in activities of SOD, CAT, AKP, PK, Ca-ATP, Mg-ATP, Na, K-ATP, Ca, Mg-ATP, and concentrations of TAC and GSH. PET MPs had the most significant impact	Liao et al. (2021)
Acropora sp.	PE fragments, PET fragments, PA 66 fragments	50 mg/L	96 h	MPs caused decrease in endosymbiont density and chlorophyll content	Xiao et al. (2021)
Goniopora columna	PE powder (40–48 μm)	5 mg/L (~1.25×10^5 MPs/L*) 10 mg/L (~2.5×10^5 MPs/L*) 50 mg/L (~1.3×10^6 MPs/L*) 100 mg/L (~2.5×10^6 MPs/L*) 300 mg/L (~7.5×10^6 MPs/L*)	7 d	Symbiont density significantly increased at different MPs concentrations compare to control At different MPs concentrations catalase, glutathione peroxidase, glutathione transferase, glutathione, malondialdehyde activity was increased	Chen et al. (2022)
Goniopora columna	PE fragments (40–48 μm)	5 mg/L (~1.25×10^5 MPs/L*) 10 mg/L (~2.5×10^5 MPs/L*) 50 mg/L (~1.3×10^6 MPs/L*) 100 mg/L (~2.5×10^6 MPs/L*) 300 mg/L (~7.5×10^6 MPs/L*)	7 d	MPs at the highest concentration significantly changed the dynamics of a bacterial community structure with increased abundance of Bacillus (*Firmicutes phylum*) compared to control	Hung et al. (2022b)
Coelogorgia palmosa	PE beads (180–212 μm)	10 mg/L (~2131–3483 MPs/L*) 50 mg/L (~1.3×10^4 MPs/L*) 70 mg/L (~1.8×10^4 MPs/L*)	48 h	Significant increase in biomarkers apart from Hsp60 Concentration above 50 mg/L caused significantly different cellular response to corals exposed to 10 mg/L	Montalbetti et al. (2022)
Tubastrea aurea	PVC beads (5–10 μm)	1 mg/L (~1.1×10^7–1.4×10^6*)	72 h	Co-exposure of MPs and Cu(II) caused significant decrease in lipid peroxide activity	Xiao et al. (2022)

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Table 1 (*continued*)

Species	Microplastics	Concentration	Duration of exposure	Results	Reference
		30 mg/L (~4.2 × 10^7–3.3 × 10^8*)		MPs decreased the toxicity of Cu(II)	
Pinnigorgia flava	Artificially aged PP fibers	0.001 mg/L 0.1 mg/L 1 mg/L 10 mg/L	72 h	MPs did not cause mortality Concentration above 1 mg/L caused the significant increase in mucus production, polyp retraction, coral tissue swelling and bleaching	Isa et al. (2023)
Acropora pruinosa	PE fragments	2 mg/L 5 mg/L 10 mg/L 20 mg/L	7 d	All MPs concentrations caused significant decrease in endosymbiont density Co-exposure of PE and Cr(III) had significantly negative effect on energy metabolism of endosymbionts	Xiao et al. (2023)
Pocillopora damicornis	PE beads (180–212 μm)	1 mg/L (~212–348 MPs/L*) 10 mg/L (~2132–3483 MPs/L*)	72 h	Bleaching recorded with an increase in temperature and MPs concentration MPs and increased temperature significantly impacted corals cellular protein homeostasis Overall, impact of MPs was less significant to increase temperature	Isa et al. (2024)
Briareum violacea	PE fragments (40–48 μm)	5 mg/L (~1.25 × 10^5 MPs/L*) 10 mg/L (~2.5 × 10^5 MPs/L*) 50 mg/L (~1.3 × 10^6 MPs/L*) 100 mg/L (~2.5 × 10^6 MPs/L*) 300 mg/L (~7.5 × 10^6 MPs/L*)	7 d	MPs caused skeleton degradation — 10 mg/L MPs caused transformation of coral skeletons from aragonite calcium carbonate crystals to amorphous calcium carbonate	Lin et al. (2025)
3D model of *Acropora* genus	Biofouled melamine fragments (1–5 mm)	80 g	1 h	79–80 % of MPs retained in coral canopies with denser canopies retaining 15 % more MPs compare to sparse canopy	Mendrik et al. (2024)
Montipora capitata	Aged and pristine PP beads (40–76 μm), PE fragments (40–76 μm), PA fragments (40–76 μm)	50 MPs/L 100 MPs/L 200 MPs/L	Not reported	MPs did not impact gamete fertilization PE MPs leachate treatment significantly reduced fertilization rate	Wilkins et al. (2024)
Pocillopora acuta	PE fragments (400–500 μm, 900–1000 μm)	5 mg/L (~10–158 MPs/L*) 10 mg/L (~20–317 MPs/L*)	Not reported	Higher number of MPs trapped in more dense colonies than sparse colonies 5 mg/L caused decrease in calcite crystals 10 mg/L caused transformation to amorphous calcium carbonate	Yen et al. (2024)

* MPs concentration range was converted using MPs size range provided by study.

Overall, toxicity studies suggest various negative effects of MPs depending on the length of exposure. Even after short-term exposure, MPs can suppress the uptake of *Symbiodinium*, affect production of chlorophyll, impact cellular response, alter photosynthetic efficiency, impact symbiont density, increase mucus production, alter development processes, and in some cases cause bleaching (Table 1) — although it is important to note that many of these studies used concentrations far higher than those found in the environment. Exposure concentration is a crucial factor in toxicity studies, for instance a study found that concentrations above 1 mg/L caused significant increase in mucus production, polyp retraction, and swelling of coral tissues, but those effects were not detected at lower concentrations (Isa et al., 2023) (Table 1).

Longer exposures were often associated with a more significant impact, and a lot of those impacts were species-specific (Table 2). When *Acropora muricata*, *Pocillopora verrucosa*, *Porites lutea* and *Heliopora coerulea* were exposed for 6 months to PE fragments (65–410 μm) at 203 ± 69 particles/L, *A. muricata* and *H. coerulea* specimens showed reduced growth, but not *P. lutea* or *P. verrucosa* (Reichert et al., 2019) (Table 2). Photosynthetic efficiency was likewise evaluated, and alterations in photosynthetic performance were only observed in *A. muricata* and *P. verrucosa* (Reichert et al., 2019) (Table 2). Similarly, Rades et al. (2024) exposed the same four species to the same PE fragments (175.5 μm, 200 particles/L) for 11 months and showed significantly reduced growth in *H. coerulea*, but not in any of the other three species — although MPs altered photosynthetic efficiency in *A. muricata* and *P. lutea* (Rades et al., 2024) (Table 2). Differences in experimental set up and the length of exposure could have influenced the slight differences in results between those two studies.

Co-exposure of MPs with other pollutants and stressors was examined in handful of studies. For example, Xiao et al. (2022) exposed *Tubastrea aurea* to PVC beads (5–10 μm) at two concentrations (1 and 30 mg/L) in combination with copper (Cu(II)). The study showed that co-exposure of MPs and Cu(II) cause significant decrease in lipid peroxide activity, however it also showed that MPs can decrease the toxicity of Cu(II) (Xiao et al., 2022). MPs were also tested in combination with chromium (Cr(III)), and it caused adverse effects on energy metabolism of endosymbionts in *Acropora pruinose* (Xiao et al., 2023). However, our understanding of the impact of MPs in combination with other pollutants on corals remains limited.

Lastly, MPs may act as vectors for pathogens, resulting in the spread

Table 2
Summary of impact of MPs on coral specimens after long-term exposure (>7 d).

Species	Microplastics	Concentration	Duration of exposure	Results	Reference
Lophelia pertusa	Biofouled LDPE beads (500 μm)	350 MPs/L	2 mo	Corals growth was significantly reduced MPs did not affect polyp activity, prey capture rate, but calcification was decreased	Chapron et al. (2018)
Acropora humilis, Acropora millepora, Pocillopora verrucosa, Pocillopora damicornis, Porites lutea, Porites cylindrica	PE fragments (37–163 μm) (biofilm formed during exposure)	3799 ± 546 MPs/L	28 d	Species-specific impact with *P. verrucosa* being affected the most MPs cause negative effects on health such as bleaching and tissue necrosis	Reichert et al. (2018)
Montipora capitata, Pocillopora damicornis	PE beads (150–180 μm)	2000 MPs/L	10 d with 1 h feeding exposure	Species-specific effect observed No significant difference was observed in *M. capitata*	Axworthy and Padilla-Gamiño (2019)
Lophelia pertusa, Madrepora oculata	Biofouled PE beads (500 μm)	350 MPs/L	5 mo	Species-specific impacts observed MPs impair prey capture and growth rates of *L. pertusa* *M. oculate's* growth and feeding was not impacted by MPs	Mouchi et al. (2019)
Acropora muricata, Pocillopora verrucosa, Porites lutea, Heliopora coerulea	PE fragments (65–410 μm)	203 ± 69 MPs/L	6 mo	Species-specifi impact on coral growth, photosynthetic efficiency and overall health (bleaching, tissue necrosis). *A. muricata* & *H. coerulea* showed reduced growth *A. muricata* & *P. verrucosa* showed alteration in photosynthetic efficiency	Reichert et al. (2019)
Acropora formosa	PE fragments (<100 μm, 100–200 μm and 200–500 μm)	50 mg/L (~812–1 × 10^5 MPs/L*) 100 mg/L (~1625–2 × 10^5 MPs/L*) 150 mg/L (~2438–3 × 10^5 MPs/L*)	14 days	MPs cause dose and size specific effect on corals health. Fragments <100 μm at 150 mg/L caused the most severe impact with 93.6 % corals experiencing bleaching and 5.9 % tissue necrosis	Syakti et al. (2019)
Stylophora pistillata	PE beads (106–125 μm)	5000 MPs/L 50,000 MPs/L	28 d	Photosynthetic activity was impacted at higher concentration Higher concentration caused significant change to metabolite profiles	Lanctôt et al. (2020)
Pseudodiploria clivosa, Acropora cervicornis	PE fragment (212–250 μm, 425–500 μm, 850–1000 μm, 1700–2000 μm, 2400–2800 μm)	30 mg/L (~2.8–6397 MPs/L*)	3 mo	Species-specific effect observed MPs significantly decreased buoyant weight in *A. cervicornis* In both species, MPs significantly decreased tissue surface area and calcification	Hankins et al. (2021)
Acropora valida, Montipora capricornis, Pocillopora damicornis, Seriatopora hystrix	PET fragments (5–5000 μm)	500 mg/L (~5.5–5.5 × 10^9 MPs/L*)	~5 mo	Species-specific effect observed *S. hystrix* showed higher mucus production in the first hours of exposure compared to the other species	Hierl et al. (2021)
Protopalythoa sp.	PE beads (1–40 μm), PVC beads (1–40 μm), PMMA (1–20 μm)	50 mg/L (PE: ~1.3 × 10^7 MPs/L; PVC: ~8.6 × 10^6 MPs/L; PMMA: ~1 × 10^7 MPs/L*)	28 d	MPs-specific impact Chlorophyll-a content was significantly decreased in PE and PMMA treatment, but not in PVC. Symbiodiniaceae density significantly decreased in all MPs treatments. Immunocompetence and oxidation resistance were also impacted by MPs	Jiang et al. (2020)
Acropora sp., *Seriatopora hystrix*	PS beads (500–1000 μm), fibers of unknown polymer (500 μm–1 cm)	0.1 mg	12 d	MPs-specific and species-specific effect observed. Fibers significantly reduced photosynthetic activity of *Acropora* sp. Beads significantly increased photosynthetic activity of *S. hystrix*	Mendrik et al. (2021)

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Table 2 (*continued*)

Species	Microplastics	Concentration	Duration of exposure	Results	Reference
Acropora muricata, *Pocillopora verrucosa*, *Porites lutea*, *Heliopora coerulea*	PE fragments (65–200 μm) (Biofilm formed over the course of exposure)	2.5 mg/L (~634–18,495 MPs/L*)	75 wk	Low number of MPs accumulated in coral tissues (up to 2 particles/cm^2), up to 84 particles/cm^2 found in coral skeleton	Reichert et al. (2022)
Acropora muricata, *Montipora digitata*, *Pocillopora verrucosa*, *Stylophora pistillata*, *Porites cylindrica*	PE fragments (100–355 μm), MPs fragment collected from environment (PE, PP, PA, PU, PS, PVC, PET and other) (346 μm), MPs fibers (acrylic, PA, PES) (1006 μm), tire wear (100–255 μm)	2.5 MPs/L 25 MPs/L 230 MPs/L 250 MPs/L 2500 MPs/L 10 mg/L for environmental MPs, fibers and tire wear	11 wk	No significant effect on overall corals health (bleaching or necrosis)	Reichert et al. (2021)
Anomastraea irregularis, *Pocillopora verrucosa*	Mixture of PP fragment (10–100 μm), PES fibers (10–100 μm)	0.8 mg/L (PP: ~1697–1.6 × 10^6 MPs/L; PES: ~1107–1.1 × 10^6 MPs/L*)	28 d	No significant in respiration, photosynthetic, growth rate, symbiodiniaceae, chlorophyll and lipid contents when corals were mixture of zooplankton and MPs compare those only zooplankton fed	Boodraj and Glassom (2022)
Goniopora columna	PE fragments (40–48 μm)	5 mg/L (~1.25 × 10^5 MPs/L*) 10 mg/L (~2.5 × 10^5 MPs/L*) 50 mg/L (~1.3 × 10^6 MPs/L*) 100 mg/L (~2.5 × 10^6 MPs/L*) 300 mg/L (~7.5 × 10^6 MPs/L*)	14 d	MPs at all concentrations significantly changed composition and dynamics of microbial communities 300 mg/L caused increase in the abundance of *Donghicola*	Hung et al. (2022a)
Acropora cervicornis	Biofouled PE beads (212–355 μm)	11.8 MPs/L	17 d	No significant effect (bleaching, tissue necrosis and zooxanthellae density) of MPs on *A. cervicornis*	Plafcan and Stallings (2022)
Acropora muricata, *Pocillopora verrucosa*, *Porites lutea*, *Heliopora coerulea*	Biofouled PE fragments (184 μm)	200 MPs/L 1850 MPs/L	15 mo with 24 h pulse feeding	MPs did not cause changes in defense mechanisms such as mucus production and extrusion of mesenterial filaments	Rades et al. (2022)
Acropora cervicornis	Biofouled PE beads (180–212 μm 355–425 μm)	125 mg/L (1.6 × 10^4 MPs/L*)	22 d	Co-exposure (pH and temperature) treatment exhibited the strongest gene expression response	Bove et al. (2023)
Merulina ampliata	PET fragments (200–300 μm)	55 MPs/L 280 MPs/L	28 d	MPs did not cause a significant effect on growth, symbiodiniaceae density, chlorophyll-a concentrations	Ng and Todd (2023)
Acropora muricata, *Pocillopora verrucosa*, *Porites lutea*, *Heliopora coerulea*	PE fragments (175.5 μm)	200 MPs/L	11 mo	Species-specific effect observed *H. coerulea* showed significantly reduced growth *P. lutea* and *A. muricata* showed changes in photosynthetic efficiency	Rades et al. (2024)
Pocillopora verrucosa, *Stylophora pistillata*	PE fragments (100–355 μm), MPs collected from environment (PE, PP, PA, PU, PS, PVC, PET and other) (100–1000 μm), MPs fibers (acrylic, PA, PES) (1006 μm), tire wear (100–355 μm)	10 mg/L (PE: ~453–20,317 MPs/L*)	8 wk	MPs-specific and species-specific impact observed Fibers and tire wear impacted *P. verrucosa* more significantly than *S. pistitata* MPs caused decreased calcification in *P. verrucosa* Photosynthetic efficiency increased in both species	Reichert et al. (2024b)
Turbinaria mesenterina	PS beads (9.5–11.5 μm)	0.01 mg/L (~11,959–21,214 MPs/L*) 0.1 mg/L (~1.2 × 10^5–2.1 × 10^5 MPs/L*)	10 d	MPs cause no significant effect on symbiotic algae density, photosynthetic efficiency or chlorophyll content Oxidative stress significantly increase in both MPs concentrations	Tang et al. (2024)
Pocillopora damicornis, *Dipsastraea pallida*	Pristine and aged PS fragments (<45 μm, 90–200 μm)	0.1 mg/L (~22–249 MPs/L*)	14 d	Species-specific impact observed Co-exposure with MPs and bisphenol A leachate caused decrease in chlorophyll and total protein content in *P. damicomis* No effect recorded in *D. pallida*	Washington et al. (2024)

[*] MPs concentration range was converted using MPs size range provided by study.

of diseases on corals (Lamb et al., 2018). A 2018 study showed that contact with plastic increased the likelihood of disease spreading to corals from 4 % to 89 % (Lamb et al., 2018). Pathogens like *Halofolliculina* spp. and *Rhodobacterales* can spread through floating plastic debris, and have been linked to outbreaks of coral diseases (Nama et al., 2023). A study exposed corals to 10–25 PE microbeads (size 170.5–230 μm) with *Escherichia coli*, and observed mortality in all polyps after four weeks post-ingestion (Rotjan et al., 2019). We have just starting scratching the surface of this topic, and more studies are needed to understand the full impact of MPs on the spread of pathogens.

3.3. Quality assurance and quality control (QA/QC)

QA/QC procedures are crucial for both studies on abundance and toxicity of MPs throughout the entire process, to prevent over or under reporting of either MPs concentrations or their effects. Implementation of QA/QC improves the accuracy and reliability of the study, and poor implementation of QA/QC measures has been recognized as a significant limitation in MP studies overall (Brander et al., 2020). In the sections below, the value in bracket indicates the number of studies across all reviewed publications that adopted a particular QA/QC aspect.

QA/QC of studies on abundance was categorized into four groups: 1) Use of field blank during sample collection (n = 11); 2) Use of procedural blank during sample processing (n = 39); 3) Method validation to test analysis efficiency (n = 8) and 4) Measures to control contamination. The fourth group was further sub-categorized into: 1) Avoiding plastic tools, such as petri dish, sampling container etc. (n = 16); 2) Cleaning equipment prior to sample collection and processing (n = 32); 3) Processing performed in a laminar flow cabinet (n = 16); 4) Wearing non-plastic clothing such as cotton, hemp etc. (n = 32); 5) Covering samples and equipment when not in use (n = 22); and 6) Pre-filtering all solutions used during sample processing (n = 17). Twenty-one studies (30.4 %) did not report any QA/QC considerations.

Replication is another crucial QA/QC component, for improving the accuracy and reliability of the study. Yet a high percentage of studies did not include replicate samples (16.2 % for sediment and 22 % for water samples) nor report whether replicates were collected (27 % for sediment and 17 % for water samples) (Fig. 11). For example, tow net sampling approach is often performed without any replication. For studies that did collect replicate samples, 3 replicates were the most common (27 % sediment, 19.5 % water) (Fig. 11). However, sites such as intertidal sediment with high disturbance of sediments may require higher number of replicates to accurately report the concentration of MPs, with some groups suggesting a minimum of five replicate samples for sediments (Institute for Environment and Sustainability (Joint Research Centre) and MSFD Technical Subgroup on Marine Litter Group, 2013). This is supported by Besley et al. (2017) who reported a rapid decrease in confidence interval above 5 replicate samples.

QA/QC categories for toxicity studies were based on de Ruijter et al. (2020) with slight modifications. Measures were categorized into four groups, with each group further divided into sub-categories: 1) Laboratory preparation: a) Inclusion of control to verify the exposure (n = 47); b) Inclusion of procedural black to limit background contamination (n = 4); c) Wearing non-plastic clothing such as cotton, hemp etc. (n = 5); d) Cleaning equipment prior to exposure (n = 3); 2) Particle characterization: a) Polymer type was validated using an instrument (n = 17); b) Particle size was validated (n = 6); c) Shape of the particles was validated (n = 3); d) Other which includes measuring of zeta potential and validating aging and/or biofilm formation (n = 4); 3) Experimental design: a) Validation of exposure concentration either during or at the end of exposure (n = 15); b) The use of concentration was justified (n = 21); c) Three or more replicates were included (n = 37); and 4) Ecological relevance: a) Natural food was added to avoid force feeding (n = 28); b) Concentrations used were environmentally relevant (n = 24); c) MPs particles were aged and/or underwent process to form biofilm (n = 24); d) Diversity of MPs including variety of polymer types and shapes (n = 14); e) Exposure time was 28 d or longer (n = 16).

Figs. 12 and 13 provide an overview of the number of QA/QC measures applied among the reviewed studies. In abundance publications, only a small percentage of studies used field blanks (n = 15). Field blanks are critical to understanding and mitigating contamination from not only airborne MPs, but also from equipment or personnel. Together with field blanks, the use of procedural blanks can minimize the risk of misinterpretation of the results. As for toxicity studies, 85.5 % reported the use of control exposure, where MPs were not added to the exposure. The use of control is essential for those studies to accurately determine whenever the effect reported is indeed due to MPs.

4. Recommendations for future studies

Based on a systematic analysis of current studies of occurrence and toxicity of MPs on corals, future studies on abundance of MPs in coral reef ecosystems should focus on the following:

4.1. Methodological improvements

Accurate assessment of MP abundance and toxicity in coral reef ecosystems relies heavily on standardized and rigorous methodological approaches. Future studies should:

I) Use consistent reporting units such as particles/m^3 and particles/L for water, particles/kg for sediment samples., to enable comparison across studies. Raw data on the concentrations should be provided in supplementary information to enable unit conversion and meta-analysis.

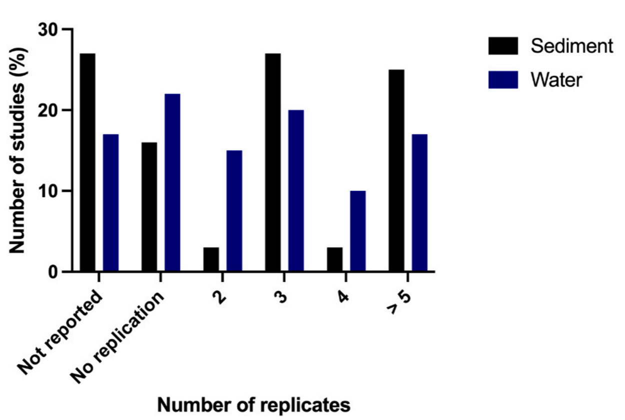

Fig. 11. Number of replications in sediment and water sampling per percentage number of studies. One study (sediment) not included as MPs (1–5 mm) were manually picked. If needed number of replicates was calculated using the number of sites and number of samples collected in total.

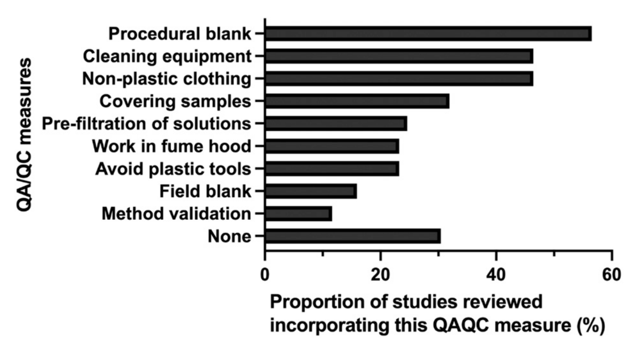

Fig. 12. QA/QC measures implemented by reviewed studies on abundance.

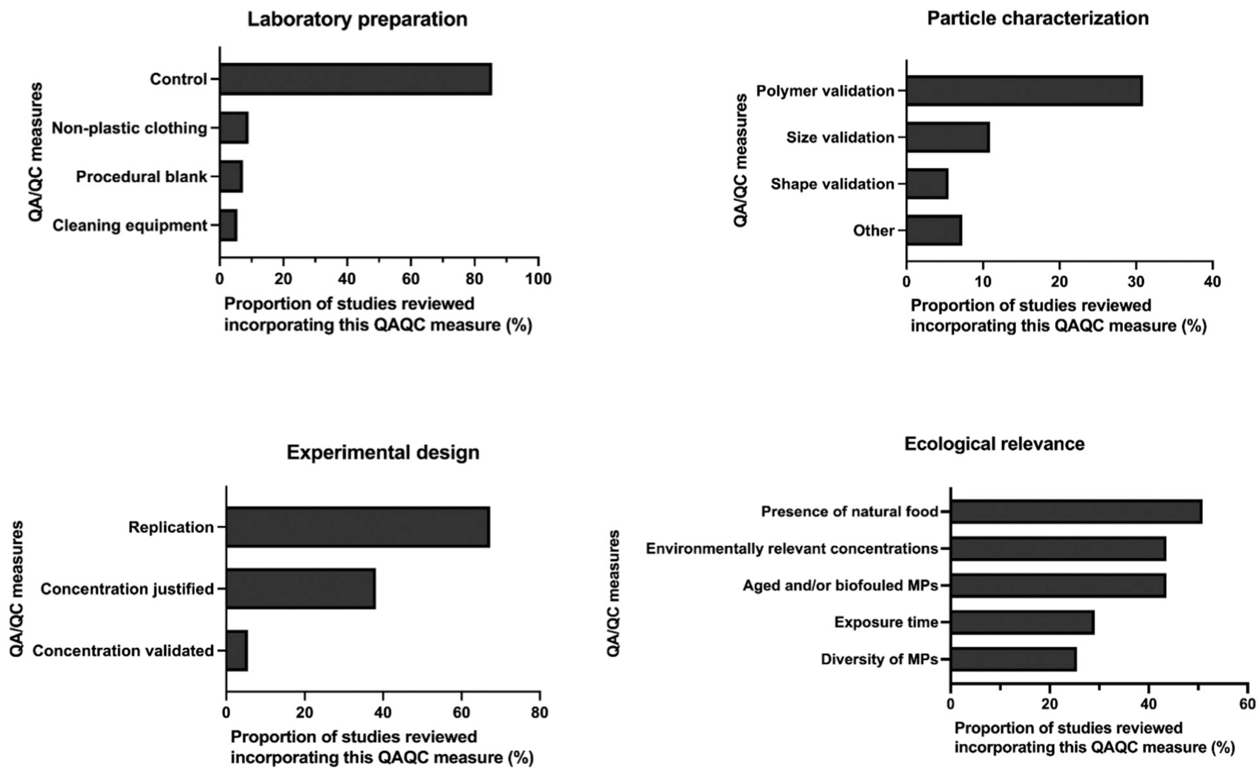

Fig. 13. QA/QC measured implemented by reviewed studies on the toxic impact of MPs.

II) Use appropriate sampling technique with small mesh sizes, depending on the analytical technique, such as FTIR detection limit of 20 μm, to ensure that smaller sized, more prevalent particles, are effectively captured.
III) Provide detailed information on the characteristics of MPs including particle size, polymer composition, shape and color. Additionally, minimum MPs size detected should be reported in a clearly defined and comparable range to facilitate inter-study comparisons.
IV) Expand sampling design to capture spatial variability including measurement of MP concentrations at various depths in the water column. Moreover, airborne MPs around coral reef areas should be measured through air sampling to develop an understanding of the contribution of airborne MPs to the total MP load in coral reef ecosystems.

4.2. Data quality and standardization

Ensuring transparency and reproducibility in MP research requires high-quality data and clear methodological documentation. Future work should:

V) Provide comprehensive methodological details including sampling methods, number of replicates, sample processing steps, and characteristics of MPs used in toxicity studies (e.g., size, polymer type, shape, any pre-treatment such as aging or biofouling). Any potential limitations in the methodology should be clearly reported.
VI) Implement a rigorous QA/QC protocol to instill confidence in data accuracy and reliability in both abundance and toxicity study results.

4.3. Research gaps and future directions

Several areas remain understudied and require attention to fully understand the occurrence and ecological impacts of MPs on coral reef ecosystems.

VII) Address geographical research gaps by monitoring MPs in coral reefs in under-studied regions, such as the East Coast of Africa.
VIII) It is crucial that toxicity studies investigating the impact of MPs use environmentally relevant MPs including concentration, shape, polymer type as well as the use of aged and biofouled MPs and pollutants associated with MPs.
IX) To accurately determine the effect of MPs exposure, it is necessary to simulate environmental conditions such as feeding coral specimens- and investigate long-term impact of MPs using various endpoints and coral genera.
X) More studies are needed to understand the role that plastics and MPs play in the spread of pathogens to coral ecosystems.
XI) Coral reefs are not exposed to single contaminants, and toxicity experiments need to include mixtures of MPs (of different polymers, sizes and forms) and other stressors, such as pH, temperature and other co-existing chemical contaminants.
XII) The occurrence and impact of nano-sized particles have so far been largely overlooked due to technical and instrumental challenges but should be focused on.

5. Conclusion

Coral reefs have unique ecological importance and despite a recent rapid rise in the number of studies over the last few years, our knowledge on MP abundance within those regions and the impact of MPs on coral is still limited by methodological shortcomings, unrealistic exposure conditions, and limited uptake of essential QA/QC measures. Addressing these limitations is crucial for generating accurate risk assessment and informing effective conservation strategies. This review provided a critical summary of the current state of the science, and highlighted limitations of current methodologies that contribute the most to difficulties in comparing across studies. The abundance of MPs varies across locations and was affected by a variety of factors. MPs cause various effects on coral species, and those were highly influenced by the experimental design and species. Although some studies applied several QA/QC, many studies still lack the use of contamination and exposure controls. Future studies need to focus on implementing adequate methodologies that will improve the knowledge on MPs and comparability between studies. This review provides 13 key research areas to improve our understanding of the occurrence and risk of MPs in coral ecosystems.

CRediT authorship contribution statement

Nikol Slynkova: Writing – original draft, Visualization, Formal analysis, Data curation, Conceptualization. **Frederic D.L. Leusch:** Writing – review & editing, Supervision, Conceptualization. **Kylie A. Pitt:** Writing – review & editing, Supervision, Conceptualization. **Mia O. Hoogenboom:** Writing – review & editing, Supervision, Conceptualization. **Shima Ziajahromi:** Writing – review & editing, Supervision, Conceptualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.marpolbul.2025.119010.

Data availability

Data will be made available on request.

References

Adam, V., Yang, T., Nowack, B., 2019. Toward an ecotoxicological risk assessment of microplastics: comparison of available hazard and exposure data in freshwaters. Environ. Toxicol. Chem. 38 (2), 436–447.

Al Nabhani, K., Salzman, S., Shimeta, J., Dansie, A., Allinson, G., 2022. A temporal assessment of microplastics distribution on the beaches of three remote islands of the Yasawa archipelago, Fiji. Mar. Pollut. Bull. 185, 114202.

Ashrafy, A., Liza, A.A., Islam, M.N., Billah, M.M., Arafat, S.T., Rahman, M.M., Rahman, S. M., 2023. Microplastics pollution: a brief review of its source and abundance in different aquatic ecosystems. J. Hazard. Mater. Adv. 9, 100215.

Axworthy, J.B., Padilla-Gamiño, J.L., 2019. Microplastics ingestion and heterotrophy in thermally stressed corals. Sci. Rep. 9 (1), 18193.

Axworthy, J.B., Lasdin, K.S., Padilla-Gamiño, J.L., 2024. Low incidence of microplastics in coral reefs of Kāne'ohe Bay, Hawai'i, USA. Mar. Pollut. Bull. 208, 116996.

Bejarano, S., Diemel, V., Feuring, A., Ghilardi, M., Harder, T., 2022. No short-term effect of sinking microplastics on heterotrophy or sediment clearing in the tropical coral *Stylophora pistillata*. Sci. Rep. 12 (1), 1468.

Berry, K.L., Epstein, H.E., Lewis, P.J., Hall, N.M., Negri, A.P., 2019. Microplastic contamination has limited effects on coral fertilisation and larvae. Diversity 11 (12), 228.

Besley, A., Vijver, M.G., Behrens, P., Bosker, T., 2017. A standardized method for sampling and extraction methods for quantifying microplastics in beach sand. Mar. Pollut. Bull. 114 (1), 77–83.

Bhan, C., Kumar, N., Elangovan, V., 2025. Microplastics pollution in the rivers, its source, and impact on aquatic life: a review. Int. J. Environ. Sci. Technol. 22 (3), 1937–1952.

Boodraj, P., Glassom, D., 2022. Experimental exposure to microplastics does not affect the physiology of healthy or moderately bleached *Anomastraea irregularis* and *Pocillopora verrucosa* corals. Mar. Biol. 169 (4), 48.

Bove, C.B., Greene, K., Sugierski, S., Kriefall, N.G., Huzar, A.K., Hughes, A.M., Sharp, K., Fogarty, N.D., Davies, S.W., 2023. Exposure to global change and microplastics elicits an immune response in an endangered coral. Front. Mar. Sci. 9, 1037130.

Brander, S.M., Renick, V.C., Foley, M.M., Steele, C., Woo, M., Lusher, A., Carr, S., Helm, P., Box, C., Cherniak, S., Andrews, R.C., Rochman, C.M., 2020. Sampling and quality assurance and quality control: a guide for scientists investigating the occurrence of microplastics across matrices. Appl. Spectrosc. 74 (9), 1099–1125.

Burns, E.E., Boxall, A.B., 2018. Microplastics in the aquatic environment: evidence for or against adverse impacts and major knowledge gaps. Environ. Toxicol. Chem. 37 (11), 2776–2796.

Cai, M., Liu, M., Qi, H., Cui, Y., Zhang, M., Huang, P., Wang, L., Xie, M., Li, Y., Wang, W., 2022. Transport of microplastics in the South China Sea: a review. Gondwana Res. 108, 49–59.

Chapron, L., Peru, E., Engler, A., Ghiglione, J.-F., Meistertzheim, A.-L., Pruski, A.M., Purser, A., Vétion, G., Galand, P.E., Lartaud, F., 2018. Macro- and microplastics affect cold-water corals growth, feeding and behaviour. Sci. Rep. 8 (1), 15299.

Cheang, C.C., Ma, Y., Fok, L., 2018. Occurrence and composition of microplastics in the seabed sediments of the coral communities in proximity of a metropolitan area. Int. J. Environ. Res. Public Health 15 (10), 2270.

Chen, Y.-T., Ding, D.-S., Lim, Y.C., Singhania, R.R., Hsieh, S., Chen, C.-W., Hsieh, S.-L., Dong, C.-D., 2022. Impact of polyethylene microplastics on coral *Goniopora columna* causing oxidative stress and histopathology damages. Sci. Total Environ. 828, 154234.

Coady, K.K., Burgoon, L., Doskey, C., Davis, J.W., 2020. Assessment of transcriptomic and apical responses of *Daphnia magna* exposed to a polyethylene microplastic in a 21-d chronic study. Environ. Toxicol. Chem. 39 (8), 1578–1589.

Cordova, M.R., Hadi, T.A., Prayudha, B., 2018. Occurrence and Abundance of Microplastics in Coral Reef Sediment: A Case Study in Sekotong (Lombok-Indonesia).

Corinaldesi, C., Canensi, S., Dell'Anno, A., Tangherlini, M., Di Capua, I., Varrella, S., Willis, T.J., Cerrano, C., Danovaro, R., 2021. Multiple impacts of microplastics can threaten marine habitat-forming species. Commun. Biol. 4 (1), 431.

Corona, E., Martin, C., Marasco, R., Duarte, C.M., 2020. Passive and active removal of marine microplastics by a mushroom coral (*Danafungia scruposa*). Front. Mar. Sci. 7, 128.

Cutroneo, L., Reboa, A., Besio, G., Borgogno, F., Canesi, L., Canuto, S., Dara, M., Enrile, F., Forioso, I., Greco, G., 2020. Microplastics in seawater: sampling strategies, laboratory methodologies, and identification techniques applied to port environment. Environ. Sci. Pollut. Res. 27, 8938–8952.

Dris, R., Imhof, H., Sanchez, W., Gasperi, J., Galgani, F., Tassin, B., Laforsch, C., 2015. Beyond the ocean: contamination of freshwater ecosystems with (micro-) plastic particles. Environ. Chem. 12 (5), 539–550.

Fallon, B.R., Freeman, C.J., 2021. Plastics in Porifera: the occurrence of potential microplastics in marine sponges and seawater from Bocas del Toro, Panamá. PeerJ 9, e11638.

Grillo, J.F., Sabino, M.A., Ramos, R., 2021. Short-term ingestion and tissue incorporation of polystyrene microplastic in the scleractinian coral *Porites porites*. Reg. Stud. Mar. Sci. 43, 101697.

Hadiyanto, H., Muslihuddin, M., Khoironi, A., Pratiwi, W.Z., Fadlilah, M.a.N., Muhammad, F., Afiati, N., Dianratri, I., 2022. The effect of salinity on the interaction between microplastic polyethylene terephthalate (PET) and microalgae *Spirulina* sp. Environ. Sci. Pollut. Res. 29, 7877–7887.

Hall, N., Berry, K., Rintoul, L., Hoogenboom, M., 2015. Microplastic ingestion by scleractinian corals. Mar. Biol. 162, 725–732.

Hankins, C., Duffy, A., Drisco, K., 2018. Scleractinian coral microplastic ingestion: potential calcification effects, size limits, and retention. Mar. Pollut. Bull. 135, 587–593.

Hankins, C., Moso, E., Lasseigne, D., 2021. Microplastics impair growth in two Atlantic scleractinian coral species, *Pseudodiploria clivosa* and *Acropora cervicornis*. Environ. Pollut. 275, 116649.

Hazeem, L.J., Yesilay, G., Bououdina, M., Perna, S., Cetin, D., Suludere, Z., Barras, A., Boukherroub, R., 2020. Investigation of the toxic effects of different polystyrene micro-and nanoplastics on microalgae *Chlorella vulgaris* by analysis of cell viability, pigment content, oxidative stress and ultrastructural changes. Mar. Pollut. Bull. 156, 111278.

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. Technol. 46 (6), 3060–3075.

Hierl, F., Wu, H.C., Westphal, H., 2021. Scleractinian corals incorporate microplastic particles: identification from a laboratory study. Environ. Sci. Pollut. Res. 28 (28), 37882–37893.

Horton, A.A., 2023. Plastic Pollution in the Global Ocean, pp. 1–20.

Huang, W., Chen, M., Song, B., Deng, J., Shen, M., Chen, Q., Zeng, G., Liang, J., 2021. Microplastics in the coral reefs and their potential impacts on corals: a mini-review. Sci. Total Environ. 762, 143112.

Hung, C.-M., Chen, C.-W., Huang, C.-P., Hsieh, S.-L., Dong, C.-D., 2022a. Ecological responses of coral reef to polyethylene microplastics in community structure and extracellular polymeric substances. Environ. Pollut. 307, 119522.

Hung, C.-M., Huang, C.-P., Hsieh, S.-L., Chen, Y.-T., Ding, D.-S., Hsieh, S., Chen, C.-W., Dong, C.-D., 2022b. Exposure of *Goniopora columna* to polyethylene microplastics (PE-MPs): effects of PE-MP concentration on extracellular polymeric substances and microbial community. Chemosphere 297, 134113.

Institute for Environment and Sustainability (Joint Research Centre), MSFD Technical Subgroup on Marine Litter Group, 2013. Guidance on Monitoring of Marine Litter in European Seas. Publications Office.

Isa, V., Becchi, A., Napper, I.E., Ubaldi, P.G., Saliu, F., Lavorano, S., Galli, P., 2023. Effects of polypropylene nanofibers on soft corals. Chemosphere 327, 138509.

Isa, V., Seveso, D., Diamante, L., Montalbetti, E., Montano, S., Gobbato, J., Lavorano, S., Galli, P., Louis, Y.D., 2024. Physical and cellular impact of environmentally relevant microplastic exposure on thermally challenged *Pocillopora damicornis* (Cnidaria, Scleractinia). Sci. Total Environ. 918, 170651.

Jiang, S., Zhang, Y., Feng, L., He, L., Zhou, C., Hong, P., Sun, S., Zhao, H., Liang, Y.-Q., Ren, L., 2020. Comparison of short-and long-term toxicity of microplastics with different chemical constituents on button polyps (*Protopalythoa* sp.). ACS Earth Space Chem. 5 (1), 12–22.

Kaiser, D., Kowalski, N., Waniek, J.J., 2017. Effects of biofouling on the sinking behavior of microplastics. Environ. Res. Lett. 12 (12), 124003.

Kalčíková, G., Skalar, T., Marolt, G., Kokalj, A.J., 2020. An environmental concentration of aged microplastics with adsorbed silver significantly affects aquatic organisms. Water Res. 175, 115644.

Kedzierski, M., d'Almeida, M., Magueresse, A., Le Grand, A., Duval, H., César, G., Sire, O., Bruzaud, S., Le Tilly, V., 2018. Threat of plastic ageing in marine environment. Adsorption/desorption of micropollutants. Mar. Pollut. Bull. 127, 684–694.

Khalid, N., Aqeel, M., Noman, A., Hashem, M., Mostafa, Y.S., Alhaithloul, H.A.S., Alghanem, S.M., 2021. Linking effects of microplastics to ecological impacts in marine environments. Chemosphere 264, 128541.

Lamb, J.B., Willis, B.L., Fiorenza, E.A., Couch, C.S., Howard, R., Rader, D.N., True, J.D., Kelly, L.A., Ahmad, A., Jompa, J., 2018. Plastic waste associated with disease on coral reefs. Science 359 (6374), 460–462.

Lanctôt, C.M., Bednarz, V.N., Melvin, S., Jacob, H., Oberhaensli, F., Swarzenski, P.W., Ferrier-Pagès, C., Carroll, A.R., Metian, M., 2020. Physiological stress response of the scleractinian coral *Stylophora pistillata* exposed to polyethylene microplastics. Environ. Pollut. 263, 114559.

Leistenschneider, D., Wolinski, A., Cheng, J., ter Halle, A., Duflos, G., Huvet, A., Paul-Pont, I., Lartaud, F., Galgani, F., Lavergne, É., Meistertzheim, A.-L., Ghiglione, J.-F., 2023. A critical review on the evaluation of toxicity and ecological risk assessment of plastics in the marine environment. Sci. Total Environ. 896, 164955.

Leusch, F.D., Lu, H.-C., Perera, K., Neale, P.A., Ziajahromi, S., 2023. Analysis of the literature shows a remarkably consistent relationship between size and abundance of microplastics across different environmental matrices. Environ. Pollut. 319, 120984.

Leusch, F.D., Ziajahromi, S., 2021. Converting mg/L to particles/L: reconciling the occurrence and toxicity literature on microplastics. Environ. Sci. Tech. 55 (17), 11470–11472.

Li, J., Liu, H., Chen, J.P., 2018. Microplastics in freshwater systems: a review on occurrence, environmental effects, and methods for microplastics detection. Water Res. 137, 362–374.

Liao, B., Wang, J., Xiao, B., Yang, X., Xie, Z., Li, D., Li, C., 2021. Effects of acute microplastic exposure on physiological parameters in *Tubastrea aurea* corals. Mar. Pollut. Bull. 165, 112173.

Lin, J., Zhao, Y.-M., Zhan, Z.-G., Zheng, J.-Y., Zhou, Q.-Z., Peng, J., Li, Y., Xiao, X., Wang, J.-H., 2024. Microplastics in remote coral reef environments of the Xisha Islands in the South China Sea: source, accumulation and potential risk. J. Hazard. Mater. 469, 133872.

Lin, P.-Y., Hsieh, S.-L., Ding, D.-S., Chen, C.-T.A., Beck, D.E., Hsieh, S., 2025. Microplastics alter crystal growth in coral skeleton structures. Environ. Chem. Lett. 23 (1), 7–11.

Liu, G., Zhu, Z., Yang, Y., Sun, Y., Yu, F., Ma, J., 2019. Sorption behavior and mechanism of hydrophilic organic chemicals to virgin and aged microplastics in freshwater and seawater. Environ. Pollut. 246, 26–33.

Lu, H.-C., Smith, J.L., Ziajahromi, S., Leusch, F.D.L., 2024. Microplastics and other anthropogenic fibres in large apex shark species: abundance, characteristics, and recommendations for future research. Chemosphere 349, 140957.

Lyu, K., Cao, C., Li, D., Akbar, S., Yang, Z., 2021. The thermal regime modifies the response of aquatic keystone species *Daphnia* to microplastics: evidence from population fitness, accumulation, histopathological analysis and candidate gene expression. Sci. Total Environ. 783, 147154.

Martin, C., Corona, E., Mahadik, G.A., Duarte, C.M., 2019. Adhesion to coral surface as a potential sink for marine microplastics. Environ. Pollut. 255, 113281.

Materić, D., Kjær, H.A., Vallelonga, P., Tison, J.-L., Röckmann, T., Holzinger, R., 2022. Nanoplastics measurements in northern and southern polar ice. Environ. Res. 208, 112741.

Mendrik, F., Henry, T., Burdett, H., Hackney, C., Waller, C., Parsons, D., Hennige, S., 2021. Species-specific impact of microplastics on coral physiology. Environ. Pollut. 269, 116238.

Mendrik, F., Houseago, R.C., Hackney, C.R., Parsons, D.R., 2024. Microplastic trapping efficiency and hydrodynamics in model coral reefs: a physical experimental investigation. Environ. Pollut. 342, 123094.

Montalbetti, E., Isa, V., Vencato, S., Louis, Y., Montano, S., Lavorano, S., Maggioni, D., Galli, P., Seveso, D., 2022. Short-term microplastic exposure triggers cellular damage through oxidative stress in the soft coral *Coelogorgia palmosa*. Mar. Biol. Res. 18 (7–8), 495–508.

Montano, S., Seveso, D., Maggioni, D., Galli, P., Corsarini, S., Saliu, F., 2020. Spatial variability of phthalates contamination in the reef-building corals *Porites lutea*, *Pocillopora verrucosa* and *Pavona varians*. Mar. Pollut. Bull. 155, 111117.

Mouchi, V., Chapron, L., Peru, E., Pruski, A.M., Meistertzheim, A.-L., Vétion, G., Galand, P.E., Lartaud, F., 2019. Long-term aquaria study suggests species-specific responses of two cold-water corals to macro- and microplastics exposure. Environ. Pollut. 253, 322–329.

Nama, S., Shanmughan, A., Nayak, B.B., Bhushan, S., Ramteke, K., 2023. Impacts of marine debris on coral reef ecosystem: a review for conservation and ecological monitoring of the coral reef ecosystem. Mar. Pollut. Bull. 189, 114755.

Nava, V., Leoni, B., 2021. A critical review of interactions between microplastics, microalgae and aquatic ecosystem function. Water Res. 188, 116476.

Ng, M.S., Todd, P.A., 2023. The comparative effects of chronic microplastic and sediment deposition on the scleractinian coral *Merulina ampliata*. Mar. Environ. Res. 191, 106135.

Okubo, N., Takahashi, S., Nakano, Y., 2018. Microplastics disturb the anthozoan-algae symbiotic relationship. Mar. Pollut. Bull. 135, 83–89.

Okubo, N., Tamura-Nakano, M., Watanabe, T., 2020. Experimental observation of microplastics invading the endoderm of anthozoan polyps. Mar. Environ. Res. 162, 105125.

de Oliveira Soares, M., Matos, E., Lucas, C., Rizzo, L., Allcock, L., Rossi, S., 2020. Microplastics in corals: an emergent threat. Mar. Pollut. Bull. 161, 111810.

Pantos, O., 2022. Microplastics: impacts on corals and other reef organisms. Emerg. Top. Life Sci. 6 (1), 81–93.

Patti, T.B., Fobert, E.K., Reeves, S.E., da Silva, K.B., 2020. Spatial distribution of microplastics around an inhabited coral island in the Maldives, Indian Ocean. Sci. Total Environ. 748, 141263.

Pereira, J.M., Carreiro-Silva, M., de Vilar, A.S., Godinho, A., Lewis, C., Pham, C.K., 2024. Cold-water octocoral interactions with microplastics under laboratory conditions. Deep-Sea Res. I Oceanogr. Res. Pap. 213, 104400.

Pérez, A.A., Cordero, A.M., Duarte, G.G., Rojas, A.G., Bolaños, H.V., Zumbado, F.A., Adolio, D.S., Naranjo, K.U., 2022. Microplastics found in the World Heritage Site Cocos Island National Park, Costa Rica. Mar. Fish. Sci. 35 (3), 403–420.

Plafcan, M.M., Stallings, C.D., 2022. Microplastics do not affect bleaching of *Acropora cervicornis* at ambient or elevated temperatures. PeerJ 10, e13578.

Prata, J.C., da Costa, J.P., Girão, A.V., Lopes, I., Duarte, A.C., Rocha-Santos, T., 2019. Identifying a quick and efficient method of removing organic matter without damaging microplastic samples. Sci. Total Environ. 686, 131–139.

Rades, M., Schubert, P., Wilke, T., Reichert, J., 2022. Reef-building corals do not develop adaptive mechanisms to better cope with microplastics. Front. Mar. Sci. 9, 863187.

Rades, M., Poschet, G., Gegner, H., Wilke, T., Reichert, J., 2024. Chronic effects of exposure to polyethylene microplastics may be mitigated at the expense of growth and photosynthesis in reef-building corals. Mar. Pollut. Bull. 205, 116631.

Rahman, N., Shozib, S.H., Akter, Y., Islam, A.R.T., Islam, S., Sohel, S., Kamaraj, C., Rakib, R.J., Idris, A.M., Sarker, A., 2023. Microplastic as an invisible threat to the coral reefs: sources, toxicity mechanisms, policy intervention, and the way forward. J. Hazard. Mater. 454, 131522.

Rehse, S., Kloas, W., Zarfl, C., 2016. Short-term exposure with high concentrations of pristine microplastic particles leads to immobilisation of *Daphnia magna*. Chemosphere 153, 91–99.

Reichert, J., Schellenberg, J., Schubert, P., Wilke, T., 2018. Responses of reef building corals to microplastic exposure. Environ. Pollut. 237, 955–960.

Reichert, J., Arnold, A.L., Hoogenboom, M.O., Schubert, P., Wilke, T., 2019. Impacts of microplastics on growth and health of hermatypic corals are species-specific. Environ. Pollut. 254, 113074.

Reichert, J., Tirpitz, V., Anand, R., Bach, K., Knopp, J., Schubert, P., Wilke, T., Ziegler, M., 2021. Interactive effects of microplastic pollution and heat stress on reef-building corals. Environ. Pollut. 290, 118010.

Reichert, J., Arnold, A.L., Hammer, N., Miller, I.B., Rades, M., Schubert, P., Ziegler, M., Wilke, T., 2022. Reef-building corals act as long-term sink for microplastic. Glob. Chang. Biol. 28 (1), 33–45.

Reichert, J., Tirpitz, V., Oponczewski, M., Lin, C., Franke, N., Ziegler, M., Wilke, T., 2024a. Feeding responses of reef-building corals provide species-and concentration-dependent risk assessment of microplastic. Sci. Total Environ. 913, 169485.

Reichert, J., Tirpitz, V., Plaza, K., Wörner, E., Bösser, L., Kühn, S., Primpke, S., Schubert, P., Ziegler, M., Wilke, T., 2024b. Common types of microdebris affect the physiology of reef-building corals. Sci. Total Environ. 912, 169276.

Reisser, J., Shaw, J., Wilcox, C., Hardesty, B.D., Proietti, M., Thums, M., Pattiaratchi, C., 2013. Marine plastic pollution in waters around Australia: characteristics, concentrations, and pathways. PLoS One 8 (11), e80466.

Rocha, R., Rodrigues, A., Campos, D., Cícero, L., Costa, A., Silva, D., Oliveira, M., Soares, A., Silva, A.P., 2020. Do microplastics affect the zoanthid *Zoanthus sociatus*? Sci. Total Environ. 713, 136659.

Rohatgi, A., 2024. WebPlotDigitizer, Version 5.2, p. 860.

Rotjan, R.D., Sharp, K.H., Gauthier, A.E., Yelton, R., Lopez, E.M.B., Carilli, J., Kagan, J. C., Urban-Rich, J., 2019. Patterns, dynamics and consequences of microplastic ingestion by the temperate coral, *Astrangia poculata*. Proc. R. Soc. B 286 (1905), 20190726.

de Ruijter, V.N., Redondo-Hasselerharm, P.E., Gouin, T., Koelmans, A.A., 2020. Quality criteria for microplastic effect studies in the context of risk assessment: a critical review. Environ. Sci. Technol. 54 (19), 11692–11705.

Sabdono, A., Ayuningrum, D., Sabdaningsih, A., 2022. First evidence of microplastics presence in corals of Jepara coastal waters, Java Sea: a comparison among habitats receiving different degrees of sedimentations. Pol. J. Environ. Stud. 31, 825–832.

Sales-Ribeiro, D., Brito-Casillas, Y., Fernandez, A., Caballero, M.J., 2020. An end to the controversy over the microscopic detection and effects of pristine microplastics in fish organs. Sci. Rep. 10.

Santana, M.F., Kroon, F.J., van Herwerden, L., Vamvounis, G., Motti, C.A., 2022. An assessment workflow to recover microplastics from complex biological matrices. Mar. Pollut. Bull. 179, 113676.

Savinelli, B., Fernández, T.V., Galasso, N.M., D'Anna, G., Pipitone, C., Prada, F., Zenone, A., Badalamenti, F., Musco, L., 2020. Microplastics impair the feeding performance of a Mediterranean habitat-forming coral. Mar. Environ. Res. 155, 104887.

Shen, M., Zhang, Y., Zhu, Y., Song, B., Zeng, G., Hu, D., Wen, X., Ren, X., 2019. Recent advances in toxicological research of nanoplastics in the environment: a review. Environ. Pollut. 252, 511–521.

Soares, M.O., Rizzo, L., Ximenes Neto, A.R., Barros, Y., Martinelli Filho, J.E., Giarrizzo, T., Rabelo, E.F., 2023. Do coral reefs act as sinks for microplastics? Environ. Pollut. 337, 122509.

Syakti, A.D., Jaya, J.V., Rahman, A., Hidayati, N.V., Raza'i, T.S., Idris, F., Trenggono, M., Doumenq, P., Chou, L.M., 2019. Bleaching and necrosis of staghorn coral (*Acropora formosa*) in laboratory assays: immediate impact of LDPE microplastics. Chemosphere 228, 528–535.

Tang, C.-H., Lin, C.-Y., Li, H.-H., Kuo, F.-W., 2024. Microplastics elicit an immune-agitative state in coral. Sci. Total Environ. 908, 168406.

Tang, J., Ni, X., Zhou, Z., Wang, L., Lin, S., 2018. Acute microplastic exposure raises stress response and suppresses detoxification and immune capacities in the scleractinian coral *Pocillopora damicornis*. Environ. Pollut. 243, 66–74.

Ter Halle, A., Jeanneau, L., Martignac, M., Jardé, E., Pedrono, B., Brach, L., Gigault, J., 2017. Nanoplastic in the North Atlantic subtropical gyre. Environ. Sci. Technol. 51 (23), 13689–13697.

Thiele, C.J., Hudson, M.D., Russell, A.E., 2019. Evaluation of existing methods to extract microplastics from bivalve tissue: adapted KOH digestion protocol improves filtration at single-digit pore size. Mar. Pollut. Bull. 142, 384–393.

Tsangaris, C., Panti, C., Compa, M., Pedà, C., Digka, N., Baini, M., D'Alessandro, M., Alomar, C., Patsiou, D., Giani, D., 2021. Interlaboratory comparison of microplastic extraction methods from marine biota tissues: a harmonization exercise of the Plastic Busters MPAs project. Mar. Pollut. Bull. 164, 111992.

Utami, D.A., Reuning, L., Konechnaya, O., Schwarzbauer, J., 2021. Microplastics as a sedimentary component in reef systems: a case study from the Java Sea. Sedimentology 68 (6), 2270–2292.

Van Graafeiland, K., 2017. Large Marine Ecosystems of the World. ArcGIS Online.

Walker, T.R., Fequet, L., 2023. Current trends of unsustainable plastic production and micro (nano) plastic pollution. TrAC Trends Anal. Chem. 160, 116984.

Wang, Q., Wangjin, X., Zhang, Y., Wang, N., Wang, Y., Meng, G., Chen, Y., 2020. The toxicity of virgin and UV-aged PVC microplastics on the growth of freshwater algae Chlamydomonas reinhardtii. Sci. Total Environ. 749, 141603.

Washington, E.A., Paley, A.S., Gulizia, A.M., Motti, C.A., Vamvounis, G., Hoogenboom, M.O., 2024. Bisphenol A leachate from polystyrene microplastics has species-specific impacts on scleractinian corals. Sci. Total Environ. 957, 177604.

Wijayanti, D.P., Indrayanti, E., Haryanti, D., Sibero, M.T., Widyarizia, M.E., Zahra, A.Q. A., Bachtiar, M., 2024. Characteristic and histological evidence of microplastic in scleractinian corals of Java Sea. Egypt. J. Aquat. Res. 50 (1), 95–102.

Wilkins, K.W., Yew, J.Y., Seeley, M., Richmond, R.H., 2024. Microplastic leachate negatively affects fertilization in the coral *Montipora capitata*. Integr. Comp. Biol. 64 (4), 1131–1140.

Wootton, N., Reis-Santos, P., Adyel, T., Blewitt, M., Clarke, B., Crutchett, T., Hamann, M., Hardesty, D., Lavers, J., Leterme, S., Leusch, F., Lynch, S., Motti, C., O'Brien, A., Okoffo, E., Perera, K., Puskic, P., Razzell Holls, J., Roman, L., Santana F.M, M., Snigirova, A., Tuuri M, E., Wilson, S., Ziajahromi, S., Gillanders M, B., 2024. In: Przeslawski R, F.S. (Ed.), Field Manuals to Monitor Australian Waters, Version 3. National Environmental Science Program.

Xiao, B., Li, D., Liao, B., Zheng, H., Yang, X., Xie, Y., Xie, Z., Li, C., 2021. Effects of microplastics exposure on the *Acropora* sp. antioxidant, immunization and energy metabolism enzyme activities. Front. Microbiol. 12, 666100.

Xiao, B., Wang, J., Liao, B., Zheng, H., Yang, X., Xie, Z., Li, D., Li, C., 2022. Combined effects of copper and microplastics on physiological parameters of *Tubastrea aurea* corals. Environ. Sci. Pollut. Res. 1–7.

Xiao, B., Li, D., Liao, B., Zheng, H., Yang, X., Xie, Y., Xie, Z., Li, C., 2023. Effects of microplastic combined with Cr (III) on apoptosis and energy pathway of coral endosymbiont. Environ. Sci. Pollut. Res. 30 (14), 39750–39763.

Yen, L.P., Yong, C.L.X., Todd, P.A., 2024. The effect of coral colony morphology, coral surface condition, particle size, and seeding point on the trapping and deposition of microplastics. Sci. Total Environ. 921, 171077.

Yusof, K.M.K.K., Anuar, S.T., Mohamad, Y., Jaafar, M., Mohamad, N., Bachok, Z., Mohamad, N., Ibrahim, Y.S., 2023. First evidence of microplastic pollution in the surface water of Malaysian Marine Park islands, South China Sea during COVID-19. Mar. Pollut. Bull. 194, 115268.

Zhang, Y., Kang, S., Allen, S., Allen, D., Gao, T., Sillanpää, M., 2020. Atmospheric microplastics: a review on current status and perspectives. Earth Sci. Rev. 203, 103118.

Ziajahromi, S., Leusch, F.D., 2023. Plastic Pollution in the Global Ocean, pp. 205–232.

Ziajahromi, S., Leusch, F.D.L., 2022. Systematic assessment of data quality and quality assurance/quality control (QA/QC) of current research on microplastics in biosolids and agricultural soils. Environ. Pollut. 294, 118629.

Ziajahromi, S., Neale, P.A., Leusch, F.D., 2016. Wastewater treatment plant effluent as a source of microplastics: review of the fate, chemical interactions and potential risks to aquatic organisms. Water Sci. Technol. 74 (10), 2253–2269.

Ziajahromi, S., Neale, P.A., Rintoul, L., Leusch, F.D., 2017. Wastewater treatment plants as a pathway for microplastics: development of a new approach to sample wastewater-based microplastics. Water Res. 112, 93–99.