Ecological Indicators 154 (2023) 110836

Contents lists available at ScienceDirect

Ecological Indicators

journal homepage: www.elsevier.com/locate/ecolind

Progress in selecting marine bioindicators for nanoplastics ecological risk assessment

Ilaria Corsi [a,c,*], Arianna Bellingeri [a,c], Elisa Bergami [b,c]

[a] Department of Physical, Earth and Environmental Sciences, University of Siena, Italy
[b] Department of Life Sciences, University of Modena and Reggio Emilia, Italy
[c] NBFC, National Biodiversity Future Center, Palermo 90133, Italy

ARTICLE INFO

Keywords:
Plastic pollution
Nanoplastics
Marine environment
Bioindicators
Ecological risk assessment
Ecosystem services

ABSTRACT

Nanoplastic (<1 μm) pollution in the marine environment is a cause of growing concern due to the current difficulties in measuring their occurrence in abiotic and biotic matrices, with consequent uncertainties on their ecological risk for natural communities and associated ecosystem services. Most investigations dealing with marine nano-ecotoxicity have been conducted on a bench-scale by examining the effects on single model species under short-term exposure conditions and at high concentrations (>50 mgL^{-1}). Both negligible impacts and detrimental effects, although poorly descriptive of the real environmental exposure scenarios, have been documented on different trophic levels and ecological functionalities. Polystyrene nanospheres (<100 nm) are by far the most tested as a proxy for nanoplastics, even though the occurrence of nanoplastics composed by other polymers and shapes (*i.e.,* irregular and fibers) has been reported in seawater column and sediments. Limited information on bioaccumulation in marine species hamper the selection of key bioindicator species following various criteria (*i.e.,* target, highly sensitive, endangered, etc) for pollution monitoring and ecological risk assessment (ERA) purposes. A holistic approach is thus required starting from setting concentrations as environmentally relevant coupled with chronic exposure, and selecting bioindicators including those having a key role in marine ecosystem processes, functions and services, also relevant for human consumption (shellfish and seafood). The present mini-review aims to provide a framework for the selection of the best bioindicators for nanoplastic in the marine environment along with current knowledge on sources, circulation and behavior in temperate and polar environments and potential compartments/species more at risk of exposure, to support nanoplastic ERA. Less investigated ecological niches and habitats, which should deserve more attention in future studies, are also identified.

1. Introduction

In the last decade, the research on plastic pollution at sea has been massive and both occurrence and biological effects on marine species have been widely documented. Initially, upon the discovery of oceanic plastic gyres, major efforts have been made on the understanding of their origin and magnitude (Lebreton and Andrady, 2019). Primarily originating by sewage treatment plant effluents released in marine coastal waters as well as from rivers and land run-off, nanoplastics (<100 nm) can be considered contaminants of emerging concerns (CECs) along with pharmaceutical, personal care products and other commercial and industrial chemicals (Hernandez et al., 2017; Bundschuh et al., 2018; Sun et al., 2019). Likewise, a secondary source of nanoplastics has been demonstrated from the breakdown of large plastic items into the sub-micron fraction <1 μm, although quantitatively unknown in terms of global mass balance reaching the marine environment and associated physical chemical properties (Gewert et al., 2015; Lambert and Wagner, 2016a; Lambert and Wagner, 2016b; Ter Halle et al., 2016; Ekvall et al., 2019; Sander et al., 2019; Singh et al., 2019; Enfrin et al., 2020).

Local circulation, as well as global long-range transport of the smallest plastics by either air masses or oceanic current have been recently described, with evidence of micro- and nanoplastics occurrence even in the most remote and pristine environments such as the Arctic and Antarctica (Lusher et al., 2017; Horton and Barnes, 2020; Bergmann et al., 2022; Caruso et al., 2022; Rota et al., 2022).

* Corresponding author at: Department of Physical, Earth and Environmental Sciences, University of Siena, Italy.
E-mail address: ilaria.corsi@unisi.it (I. Corsi).

https://doi.org/10.1016/j.ecolind.2023.110836
Received 30 March 2023; Received in revised form 29 July 2023; Accepted 17 August 2023
Available online 20 August 2023

Due to the initial paucity of field data on the environmental occurrence of nanoplastics in seawater and sediments caused by analytical challenges due to the nanosize range (Gigault et al., 2016; Ter Halle et al., 2017; Nguyen et al., 2019; Caputo et al., 2021), the majority of ecotoxicity studies have been conducted at very high concentrations (>50 mgL^{-1}) poorly descriptive of real exposure scenarios, being far higher than those recently reported in marine coastal waters and sand water extracts (*e.g.*, Dutch Wadden Sea, 1–3 m water depth range 1–15 μgL^{-1}) (Siti et al., 2015; Al-Sid-Cheikh et al., 2018; Davranche et al., 2020; Materić et al., 2022a). In addition, standardized short-term acute ecotoxicity tests have prevailed over chronic ones without taking into consideration transformations occurring with time upon nanoplastic release into the sea, which indeed affect their behavior and fate in seawater and sediments (Alimi et al., 2018; Corsi et al., 2020; Frehland et al., 2020).

In terms of hazards, nanoplastics exposure, intended as those in the nanoscale range between 1 and –100 nm, raise major concerns due to their ability to pass through biological barriers and enter inside cells. Their colloidal properties and high surface area to volume ratio make them not only far easily bioavailable to marine species but able to interact with other compounds including existing toxic chemicals in seawater and sediments (Singh et al., 2019; Bhagat et al., 2021; Gigault et al., 2021; Venel et al., 2021).

Polystyrene nanospheres (PS NPs, <50 nm) have been the most widely tested by far as a proxy for nanoplastics due to the paucity of commercially available polymeric nanoscale materials for ecotoxicological studies, though being only partially environmentally relevant. Most recent field monitoring surveys have shown the occurrence of a range of synthetic polymers (*e.g.*, polypropylene-PP, polyethylene-PE, polyvinyl chloride-PVC, poly(methyl methacrylate)-PMMA)of various size and shape including microfibers as the most prevalent in either abiotic (sea spray, water column and sediment) and biotic matrices (marine species) (Schirinzi et al., 2019; Avio et al., 2020; Suaria et al., 2020; Santini et al., 2022).

While microplastic presence has been massively documented in marine biota, including potential biomagnification along food chains (Miller et al., 2020), nanoplastic occurrence (including the sub-micron fraction below 1 μm, Hartmann et al., 2019) has yet to be proven, mainly due to challenges associated with their extraction and quantitation from complex biological matrices (Valsesia et al., 2021). Synthetised metal-doped or radiolabelled nanoplastics have been used so far in laboratory-controlled studies, with the aim to understand fate and behavior in complex environmental matrices including marine organisms (Al-Sid-Cheikh et al., 2018; Mitrano et al., 2019). Such knowledge gaps still limit a proper ecological risk assessment (ERA) of nanoplastics in the marine environment and, more importantly, the recognition of best bioindicators to be selected for monitoring and risk assessment purposes.

The present mini-review aims to provide a framework for the selection of the best bioindicators for nanoplastics in the marine environment along with current knowledge on sources, circulation and behavior of nanoscale particles (<100 nm) in temperate and polar environments and potential compartments/species more at risk with the final aim to support nanoplastic ERA. We also highlight less investigated ecological niches (*e.g.*, benthos) and habitats that should deserve more attention in future studies based on sources and circulation of nanoplastics in the marine environment at different spatial scales.

2. Nanoplastic fate in seawater: How to set exposure conditions

Up until a few years ago, the occurrence of nano-sized plastic fragments in the marine environment was considered plausible based on the million tonnes of plastic entering the oceans each year and based on laboratory evidence of the fragmentation, down to the nanoscale, of different types of plastic exposed to weathering (Gigault et al., 2016; Lambert and Wagner, 2016a; Lambert and Wagner, 2016b). In 2017, first evidence of sub-micron plastic fragments (1–999 nm) in water column from within the North Atlantic subtropical gyre was given by Ter Halle et al. (2017), later followed by others (Schirinzi et al., 2019; Llorca et al., 2021; Materić et al., 2022a).

What emerges is that nanoplastic behavior in water can be associated to that of colloids, as the polymers (*e.g.*, PVC and polyethylene terephthalate-PET) composing the nano-sized fraction were negatively buoyant compared to water and yet were sampled in the water column (Ter Halle et al., 2017). While the behavior and agglomeration/aggregation of microplastics is considered to be mainly dominated by buoyancy and shear flows, for nanoplastic objects, Brownian motion and water chemistry prevail in determining their behaviour and fate (Fig. 1) (Gigault et al., 2018; Sun et al., 2021). Hence, the reduction of size under a certain threshold corresponds to a change in the factors determining particle dispersion in water. For instance, not being affected by density may allow nanoplastics to stay longer in the euphotic zone, rich in biomass and biodiversity, resulting in higher possibility of encounters with marine biota (Gigault et al., 2016).

Key factors ruling agglomeration/aggregation and fate of nanoplastics in natural waters are numerous: among those, surface properties and shape, as well as the concentration ratio between nanoplastics, electrolytes, organic matter and inorganic colloids (Oriekhova and Stoll, 2018; Veclin et al., 2022).

Since environmental conditions are subject to temporal and spatial variations, it becomes difficult to draw a general guideline on nanoplastic fate in the marine environment. For instance, Venel et al. (2021) observed that the variable conditions in the interface between land and sea, as in a mangrove swamp, would probably cause the aggregation and sedimentation of most of the nanoplastics flowing through, leaving just a few particles in suspension. Gondikas et al. (2020) wondered if it is correct to look for a final sink for colloids in the natural environment (*e.g.*, sediment) or if it is better to consider it as a cyclical transformation process, involving resuspension and sinking phases. The cyclical nature is attributed to the fluctuation in the amount of extracellular polymeric substances (EPS) in seawater, due to the seasonality of algal blooms. For example, when EPS are more abundant, engineered nanoparticles are more stable, reducing the rapid aggregation induced by the high ionic strength of seawater, which would presumably result in the aggregate sinking and deposition to the seafloor (Gondikas et al., 2020). This was demonstrated under laboratory-controlled conditions for PS NPs incubated with diatom-secreted EPS, whose aggregation resulted less pronounced compared to the absence of EPS (Grassi et al., 2020). While EPS is known to modify nanoplastic behavior, the opposite is also true. PS NPs were observed to induce and accelerate the assembly of microgels from free EPS (Shiu et al., 2020a), while phytoplankton exposed to PS NPs produced EPS with a higher protein-to-carbohydrate ratio (P/C), which increased the matrix stickiness (Shiu et al., 2020b). This might result in the incorporation of nanoplastics inside biogenic aggregates (*e.g.*, marine snow) with implications for the vertical transport in the water column of both nanoplastics and marine snow, and repercussions on carbon fluxes to the seafloor as well as on the bioavailability of nanoplastic for benthic filter feeders and dwellers (Ward and Kach, 2009; Long et al., 2015; Porter et al., 2018). Even if a general guideline on nanoplastic fate in the marine environment is far from being accomplished, it appears clear that nanoplastics and microplastics behave differently in seawater and, as such, will have a different biological impact, especially when considering possible target organisms.

3. Ecotoxicity testing of nanoplastics

Concerning the ecotoxicological evaluation of nanoplastics, a major gap exists between experimental conditions and what could be considered a realistic exposure scenario. First, tested nanoplastics are generally perfectly round spheres which hardly resemble nanometric irregular fragments resulting from the degradation of plastic waste (Gigault et al., 2018). Moreover, the suspensions of these *ad hoc* synthetized proxies

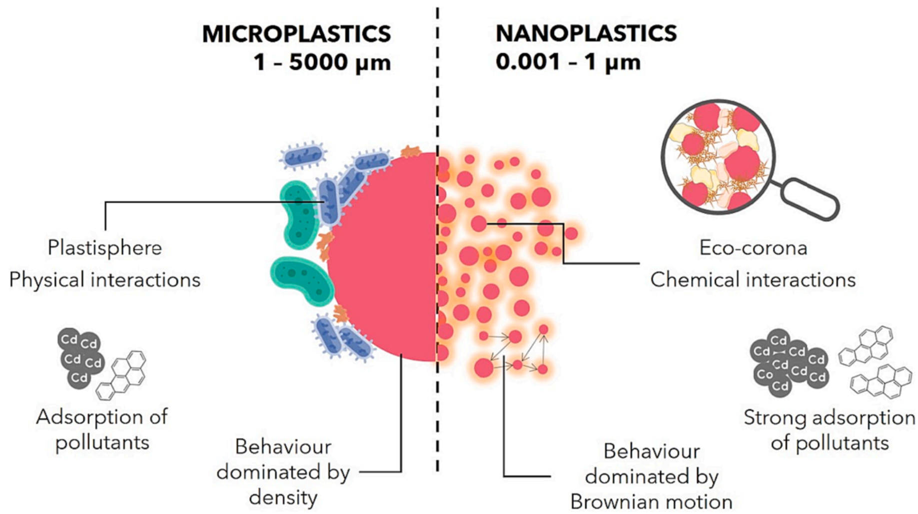

Fig. 1. Difference in behavior and interactions of microplastics (left) and nanoplastics (right) in the marine environment: microplastic circulation in the water column is mainly dominated by density, plastisphere composition and characterized by physical interactions, while nanoplastic behavior is dominated by Brownian motion and chemical interactions, including strong adsorption of pollutants and eco-corona formation.

usually contain surfactants and/or tracers, such as fluorophores, which may play a role in the observed behavior and ecotoxicological effects (Pikuda et al., 2018; Catarino et al., 2019).

Another relevant gap is the one existing between tested polymers and polymers detected in the colloidal fraction of environmental sampling. As it comes up from the relative percentages of sub-micron polymers sampled in the North Atlantic subtropical gyre (Ter Halle et al., 2017), the prevalent one is PVC (70%), followed by PET (17%), PS (9%) and PE (4%) (Fig. 2a). Data reported by other studies about the composition of sub-micron plastic fragments sampled in the marine environment (Llorca et al., 2021; Materić et al., 2022a) do not allow a comparison of the relative percentages of the different polymers; however, the overall composition varied with sampling spots, with the main represented polymers being PP, polyisoprene (PI) and PS by llorca et al. (2021), while a prevalence of PS and PET was reported by Materic et al. (2022a). Such data hardly find correspondence with ecotoxicological studies conducted so far, in which PS is nearly the only tested polymer (Fig. 2b). By searching the words "nanoplastic" "toxicity/effects" and "marine organisms" on the online Scopus database, excluding reviews and non-pertinent results, a total of 125 research papers are found, the 87% percent of which is carried out on PS NPs (Fig. 2b, Table S1). Among those, only 2% of analyzed studies (Env NP) are performed on nanoplastics obtained from the fragmentation of micro- and macro-plastics sampled in the marine environment, mainly composed of PE and PP (Baudrimont et al., 2020; Arini et al., 2022; Roman et al., 2023). Species-sensitivity distribution (SSD) analysis has shown that the polymer core of model nanoplastics is crucial in driving their ecotoxicity towards marine species (Venâncio et al., 2019). Although scarce, available data do not show such a marked predominance of PS in the colloidal fraction of marine environmental samplings, hence ecotoxicological data should better reflect the possible composition of environmental nanoplastics.

Together with the polymer core, surface charge also plays a crucial role in nanoplastic ecotoxicity. Model PS nanospheres with different surface charges (*i.e.,* amino-modified, $-NH_2$, or carboxyl-modified, –COOH) have been tested, with the aim to better understand the bio-nano-interactions in seawater media (Wheeler et al., 2021). Surface functionalization was recognized as an important driver of behavior and ecotoxicity in seawater, affecting uptake and ecotoxicological outcomes in marine model species (Bergami et al., 2016; Marques-Santos et al., 2018; Grassi et al., 2019; Grassi et al., 2020; Murano et al., 2021). However, specific functionalization, such as $-NH_2$ surface groups, are unlikely to be found in the marine environment unless occurring during ageing and/or transformations through interactions with other existing molecules present in seawater (*i.e.*, dissolved organic matter, proteins) (Lehner et al., 2019). Instead, the use of surface functionalization/charges resembling those of environmentally weathered plastics (*e.g.,* carbonyl and carboxyl groups and negative surface charge) should be encouraged.

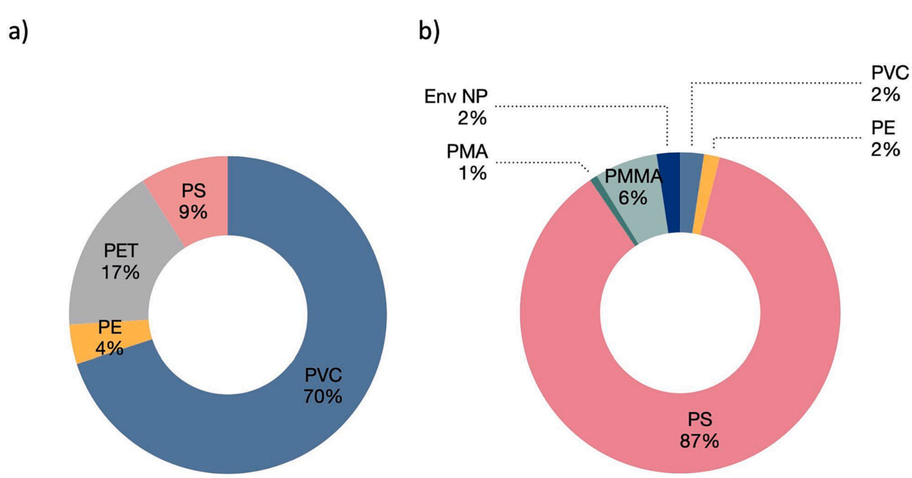

Fig. 2. Relative percentages of polymer composition of sub-micron fragments sampled in the North Atlantic subtropical gyre (a) and relative percentages of polymers employed for marine ecotoxicological studies (b). Data obtained from Ter Halle et al. (2017) (a) and data obtained from Scopus (b), search queries: TITLE-ABS-KEY (nanoplastic OR plastic AND nanoparticles AND effects OR toxicity AND marine AND organisms OR environment) AND (LIMIT-TO (DOC-TYPE, "ar")). "Env NP" refers to plastics sampled in the marine environment as micro- and macro-plastics and fragmented to the nanosize for exposure.

As it should be encouraged the use of concentrations resembling those predicted for marine waters. Although relevant in terms of understanding toxicological outcomes and, in some cases, mode of action of nanoplastics towards marine organisms, concentration tested in most studies are at least two orders of magnitude higher than those predicted in surface waters (*i.e.,* >1 mgL^{-1} vs 0.001–20 μgL^{-1}, (Lenz et al., 2016)). This questions their relevance towards real exposure scenarios, though such high concentrations may still represent hotspot regions for plastic pollution such as coastal areas of the Mediterranean Sea (Everaert et al., 2020), in which nanoplastic abundance may approach toxicity thresholds.

Besides the use of model nanoplastics, most marine environmental risk assessment studies have been performed using model organisms rather than bioindicators. This resulted in ecotoxicological outcomes which might be far from being realistic in terms of uptake and disposition of nanoplastics, as well as their toxicodynamic and mode of action. Phyto- and zoo-plankton species have been the most investigated (52%) for nanoplastic ecotoxicity, while species belonging to the benthic flora and endobenthic fauna have been poorly studied (Fig. 3a) (Chae and An, 2017). This could be explained by several factors, such as the reliability of standardized ecotoxicity assays based on ISO, OECD and/or ASTM protocols (*e.g.,* ISO, 2006; OECD 201, 2011; OECD 317, 2021), which are easy to handle at bench-scale, quick and costly, and the assumption that the colloidal nature of nanoplastics makes them more bioavailable to marine planktonic species in surface waters. However, as recent models underline, nanoplastics fate in seawater is not straightforward and nanoplastics might be able to reach the sea floor due to agglomeration/aggregation phenomena, being thus bioavailable for endobenthic organisms, by far less investigated (Pradel et al., 2021a; Alimi et al., 2022). Another relevant drawback in terms of representativeness of ERA scenarios, is the single-species toxicity test which not only relies on a very limited number of model organisms but, more importantly, does not take into consideration that marine phytoplankton is composed of highly diverse taxa (Caron et al., 2017) with different functional traits that define the fitness of planktonic communities and functioning of pelagic ecosystems (Otero et al., 2020).

Therefore, is the limited number of species available for ecotoxicity testing enough to represent natural phytoplanktonic communities and assess ecological risk? And, which one could thus represent the best match in terms of reliability and feasibility at bench-scale test level and representativeness of impact of nanoplastics on natural communities and associated ecosystem functions and services?

3.1. Selected models and their ecological relevance/representativeness

Microalgae including diatoms have been vastly investigated (Fig. 3b) and alterations on photosynthetic efficiency, production of reactive oxygen species up to DNA and mitochondria damages with consequences on cell division and growth have been described (Sjollema et al., 2016; Bergami et al., 2017; Sendra et al., 2019; Gomes et al., 2020; González-Fernández et al., 2020). We recently reported a reduction of chain length in marine diatom *Skeletonema marinoi* upon exposure to PS NPs suggesting that nanoplastics could limit their sinking on the sea floor with ecological implications for the marine biological carbon pump (Bellingeri et al., 2020).

Zooplanktonic suspension feeders such as rotifers, copepods and brine shrimps (Bergami et al., 2017) as well as embryos/larvae stages of ascidians, microcrustaceans, bivalves and sea urchins have been widely represented in nanoplastic toxicity testing (Fig. 3b). Indeed, larval stages are the most sensitive since any effect has direct implications for predicting recruitment and survival upon exposure to environmental stressors, including nanoplastics. Alteration in feeding behavior, delay in growth and gut retention of PS NPs have been reported in the copepod *Paracyclopina nana* (PS NPs of 50 nm, up to 10 μgmL^{-1}) (Jeong et al., 2017), in *Brachionus koreanus* and *Brachionus plicatilis* regardless of surface charges (0.5–5 μgmL^{-1}), with PS-NH_2 significantly reducing lifespan (LC_{50} = 2.75±0.67 μgmL^{-1}) (Jeong et al., 2016; Manfra et al., 2017).

Nanoplastics bearing positive surface charges (50 nm PS-NH_{22} up to 20–25 μgmL^{-1}) have been shown to disrupt embryo development and biomineralization in mussels (*Mytilus galloprovincialis*) and oysters (*Crassostrea gigas*) (Balbi et al., 2017; Tallec et al., 2018) and cause several degrees of malformations in sea urchins and ascidian larvae (*Ciona robusta*) (Della Torre et al., 2014; Pinsino et al., 2017; Eliso et al., 2020; Eliso et al., 2023). Long-term exposure studies (14 days) have revealed more severe outcomes in brine shrimp larvae (*Artemia franciscana*), with delay in larvae development and mortality (50 nm PS-NH_2, LC_{50} = 0.83 μgmL^{-1}) and significant gut retention of PS-COOH agglomerates (Bergami et al., 2017). Such a scenario could anticipate disruption in organism' growth and potential food chain transfer of nanoplastics up to direct predators such as marine fish.

As far as benthic invertebrates, nanoplastics have been shown to affect cell-mediated innate immune responses on sea urchin *Paracentrotus lividus* and bivalves (*Mytilus spp.*) with positive surface charged PS NPs (PS-NH_2, 50 nm) inducing apoptotic-like nuclear alterations, impairing cell viability and phagocytosis (Marques-Santos et al., 2018; Murano et al., 2020; Murano et al., 2021). PS NPs can enter inside immune cells by endocytic pathways as shown in mussel hemocytes and sea urchin coelomocytes (Gaspar et al., 2018; Sendra et al., 2020). By far less investigated, nanoplastics (PS NPs, 0.0005–50 mg/L) affect keystone species of intertidal and marine coastal environment such as the polychaeta *Hediste diversicolor* by altering regeneration capacity,

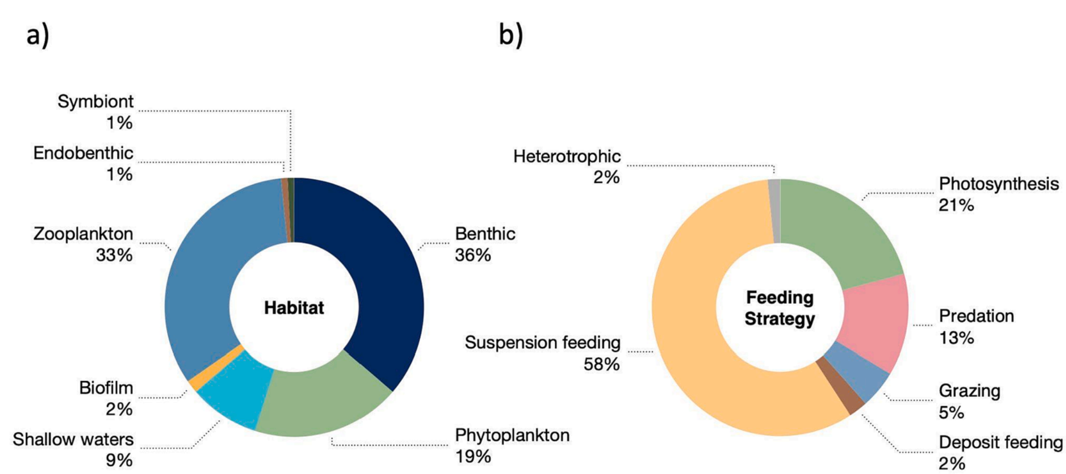

Fig. 3. Relative percentages of habitat (a) and feeding strategies (b) of marine organisms investigated in nanoplastic ecotoxicity studies.

neurotransmission and behaviour which may have consequences on sediment nutrient cycling and (endo-)benthic fauna (Silva et al., 2020a; Silva et al., 2020b).

Regarding vertebrate species, fish and marine mammals have been so far overlooked in terms of nanoplastic effect assessment. Following the current EU regulation on animal testing (Directive 2010/63 EU), nanoplastics either tested on fish cell lines (PS NPs of 100 nm, 0.001–10 mgL^{-1}) from seabass and seabream and adult of gilthead sea bream (PMMA NPs of 45 nm, 0–10 μgmL^{-1}) cause disruption in their immune system, lipid metabolism and cell viability (Almeida et al., 2019; Brandts et al., 2021) (Table S1).

3.2. Overlooked taxa with high ecological values

In comparison with other marine organisms, marine plants have been overlooked in terms of impact of nanoplastics, even though the seafloor is now considered the final sink for plastics coming from water column and land (Corsi et al., 2021).

Seagrass meadows, for instance, have been recently shown to easily interact with plastic debris reaching the seafloor, playing a role in their removal and transfer to shorelines and along the marine food chain (*e.i.,* herbivorous fishes). Plastics of various size and polymeric nature present on the shallow seafloor can thus be transported back into the shorelines with consequences on species living in them. Considering their important ecological role and associated ecosystem services as CO_2 absorption and nutrient circulation, energy and carbon production, climate change mitigation and nursery and refuge areas for many species including supporting fishery production, any study assessing nanoplastics interference on their health status is becoming mandatory (Duarte et al., 2013).

Root degeneration and impairment of photosynthetic machinery have been reported in seagrass *Cymodocea nodosa* (Ucria) upon acute exposure (12 days) to PS NPs (30 nm, 68 μgL^{-1}) with serious consequences on ecosystem services they provide on abiotic (water column oxygenation, water flow attenuation) and biotic parameters (maintaining biodiversity) (Menicagli et al., 2022).

Recent findings revealed accumulation of microplastics in seagrass sediments and adhesion to seagrass blades and associated epiphyte assemblages whose density strongly affect the retention ability of the blade itself (Jones et al., 2020; Gerstenbacher et al., 2022). Water column and sediment microplastics are either intercepted by blades, with fibers and smaller fragments the most abundant among other shapes. Indeed, the ability of seagrass meadow to retain and act as sink for the sub-micron plastics cannot be ruled out with still unknown ecotoxicological outcomes so far.

Evidence of large plastic fibers and fragments (average size of 1 cm) entrapped in the aegagropilae of the Mediterranean phanerogam *Posidonia oceanica* raised concerns on possible consequences on their ecological role supporting detritivorous trophic webs and providing a significant amount of carbon and nitrogen for beach biological assemblages (Pietrelli et al., 2017; Sanchez-Vidal et al., 2021). *P. oceanica* high abundance, primary production and biomass make it the most productive ecosystem in the Mediterranean. However, this major habitat-engineering species is currently under threat (13–50% reduction in areal extent, cover and shoot density) and was recently included in the list of endangered or threatened species of the Barcelona Convention (Annex II) and in the Annex I (Strictly Protected Flora Species) of the Convention on the Conservation of European Wildlife and Natural Habitats (Bern Convention) (de Los Santos et al., 2019).

Vascular plants are known to significantly retain nanoplastics through surface foliar and root adsorption and even internalize and translocate them through the vascular system; phytotoxicity and effects on growth and photosynthesis have been reported with size and surface charges of the NP having a leading role (Sun et al., 2020; Yin et al., 2021).

Nanoplastic toxicity on aquatic macrophytes has been investigated only on freshwater species. For example, Lemnaceae and Poaceae have been shown to facilitate nanoplastic entrance in the food web, with consequences on a wide range of species, ecosystem functions and services (Mateos-Cárdenas et al., 2021; Gerstenbacher et al., 2022).

Very limited studies have been conducted on brackish and marine species and only on microplastics. Recent findings reported the ability of mangrove leaves to capture microplastics from tidal waters and air and mangrove wetland acting as a sink (Li et al., 2022).

Evidence of nanoplastic negative impact on terrestrial plants underlines the importance of focusing future studies on seagrass and marine vascular plants as they represent important bioindicators of the marine environment given their precious role in numerous ecosystem services and marine goods (Cullen-Unsworth and Unsworth, 2013; Reynolds et al., 2016; Nordlund et al., 2018; Kalčíková, 2020).

As far as other marine habitat-engineers, coral reefs have been barely investigated for nanoplastic impact despite their huge ecological role as hotspots of biodiversity and associated ecosystem goods and services and facing serious threats from other anthropogenic stressors (Hughes et al., 2018; Woodhead et al., 2019). Microplastic ingestion and toxicological outcomes on corals have been vastly documented, including the negative impact of plastisphere-associated microbes on coral diseases (Lamb et al., 2018; Reichert et al., 2018; Rezania et al., 2018). Only a couple of recent studies on symbiotic dinoflagellates and the coral host *Stylophora pistillata* reported how the impairment of photosynthetic capacity in symbionts upon exposure to nanoplastics (PS NPs of 20–42 nm, 0.5–10 mgL^{-1}) can have a significant role in coral bleaching (Ripken et al., 2020; Marangoni et al., 2022).

3.3. New frontiers: Nanoplastics impact in the Southern Ocean

Although in the ERA of nanoplastics the choice of best bioindicators should not rely on peculiar properties of the marine ecosystem, high latitude oceans deserve a separate mention. The Arctic and Antarctica are characterized by extreme environmental conditions, which, through millions of years, have determined unique eco-physiological adaptations in the endemic species inhabiting these regions. Cold-adapted marine organisms are typically stenothermal, they display slow metabolism, development and growth rates (Peck, 2018) and they are characterized by low genetic variability (Rogers, 2007). As a result, polar marine ecosystems are considered less resilient to changes compared to tropical and temperate regions and the study of CECs such as nanoplastics allows us to understand how human-driven perturbations are affecting the most pristine environments.

Polar regions are known to act as ‘cold trap’ for major chemical pollutants, which can reach remote regions through the movement of air masses and ocean currents at a global scale. Long-range transport pathways can be hypothesised also for nanoplastics, which have recently been found in remote settings such as Siberian Arctic surface waters far from potential emission sources (Obbard, 2018; Materić et al., 2022c). Nevertheless, while several studies have addressed the occurrence and distribution of microplastics in polar marine ecosystems (reviewed in Caruso et al. (2022) and Rota et al. (2022)), so far evidence of nanoplastic contamination in polar marine environments is confined to a single observation in the sea ice from McMurdo Sound (>77° S) in the Ross Sea, one of the most remote regions of Antarctica (Materić et al., 2022b). Within sea ice, Materic and co-authors reported a prevalence of low-dense nano-sized polymers (*i.e.,* PE and PP, with PE contributing to 50% by mass) with average concentrations of 0.052 mgL^{-1}. The absence of other polymers, such as PS, which were previously found in the microplastic composition of the Eastern Antarctic Sea ice (Kelly et al., 2020), was attributed to the complex sea ice dynamics regulating the incorporation/release processes of these anthropogenic particles. By mimicking sea ice growth-thawing on a bench-scale set up, Pradel et al. (2021b) have indeed demonstrated that, differently from microplastics that accumulate within sea ice, nanoplastics can be expelled from brine channels during sea ice growth. In polar waters, low temperatures approaching the freezing point of seawater can be associated to lower reactivity of nanoplastics, which are likely to form large agglomerates (Bergami et al., 2019) but can be stabilised by NOM at the sea ice-water interface (Pradel et al., 2021b). However, passive entrainment of nanoplastics in sea ice might still occur as a result of changes in environmental conditions (*e.g.,* increase in salinity) and interaction with EPS released by sea ice algae (Hoffmann et al., 2020).

Sea ice is a highly productive habitat which supports diverse phytoplankton (both within or underneath the sea ice) as well as cryopelagic communities, providing nutrients, food and shelter to polar marine wildlife (Peck, 2018). The accumulation of nanoplastics at the interface with sea ice indicates a high probability to encounter polar marine organisms at the base of the trophic web, such as phyto- and zooplankton. Hoffman et al. (2020) reported that large PS NPs (500 nm, at 90,000 nmL^{-1}) incubated with cells of the alga *Fragillariopsis cylindrus*, which dominates Arctic and Antarctic Sea ice communities, reduced the number of algal cells within the sea ice, suggesting potential alteration on their annual recolonization of sea ice and changes in algae community structure.

Nanoplastics have been shown to impact pigment and lipid composition as well as assemblage structure in marine microalgae including diatoms (Bellingeri et al., 2020; González-Fernández et al., 2020), leading to potential disruption of biomass productivity implications at the ecosystem-scale (Casabianca et al., 2021). In polar marine ecosystems, sea ice algae should be considered as target organisms to study nano-biological interactions to predict cascading effects on higher trophic level species as well as effects on primary productivity.

Ecotoxicological studies conducted so far on polar marine organisms have the same two main limitations as those on temperate ones: (i) they were conducted with commercially available PS nanospheres, thus not representing the different polymers and shapes (mainly irregular or fibrous) of the nanoplastics found in marine environments; (ii) nano-PS were tested at rather high concentrations (in the range of 1 – 2.5 mgL^{-1}), although predicted environmental concentrations for polar regions are still lacking.

In the Southern Ocean, the Antarctic krill *Euphausia superba*, the keystone species of pelagic ecosystems, has recently been shown to internalise microplastics (mainly microfibers of polyamide-PA, PE and PET) (Wilkie Johnston et al., 2023; Zhu et al. 2023), with the ability to fragment them into nanoplastics, as demonstrated through a bench-scale experiment (Dawson et al., 2018). Nanoplastics generated by the Antarctic krill become readily available to enter Antarctic marine food webs. By understanding the impacts of nanoplastics on Antarctic krill, it is possible to shed light into the consequences at higher trophic levels. Antarctic krill has a circumpolar distribution, it reaches high biomass and it exerts a bottom-up control on the food webs, being the key trophic link between primary producers and higher predators (Atkinson et al., 2012). PS NPs have been found to alter the physiology of Antarctic krill juveniles following short-term exposure, with PS-NH_2 leading to decreased swimming and altered exuviae release (Bergami et al., 2020). Surface charged PS NPs were further found to be excreted and incorporated in krill faecal pellets, altering their settling behavior in the water column. Considering the pivotal role of Antarctic krill in ensuring deep-sea carbon export (Manno et al., 2020) these results can be considered an early warning of the ecosystem-scale impact of nanoplastics on particulate organic carbon flux through the mesopelagic zone in the Southern Ocean.

Other key sentinel zooplankton species of Arctic and Antarctic pelagic ecosystems to be considered in future studies should include other zooplanktonic taxa, such as copepods and amphipods, which play key roles in polar open ocean and sea ice food webs. In addition, salps (*Salpa thompsoni*) could be taken into account as target organisms in climate change scenarios in which the krill-based ecosystem model is expected to decline (Rowlands et al., 2021). Compared to pelagic ecosystems, our knowledge regarding the presence and impacts of nanoplastics on polar benthic communities remains unexplored. Microplastics accumulate in Arctic and Antarctic deep-sea sediments (Munari et al., 2017; Adams et al., 2021; Bergmann et al., 2022) and are found in polar benthic organisms (Sfriso et al., 2020; Deng et al., 2021; Bergami et al., 2023). However, no data are available on the occurrence of small microplastics (<10 µm, lower size limit reported in Fang et al., (2018)) and nanoplastics, limiting the impact assessment on benthos. Our study on the sea urchin *Sterechinus neumayeri* immune cells (Bergami et al., 2019) was the first attempt to evaluate nanoplastic toxicity on Antarctic benthos. We showed that both PS-COOH and PS-NH_2 (50–60 nm, at 1 and 5 mgL^{-1}) were able to impair phagocytic capacity and generated an inflammatory response in sea urchin cells following *in vitro* exposure.

The Antarctic sea urchin *S. neumayeri* was chosen as a relevant target organism for nanoplastics due to its abundance in Antarctic shallow waters and its major role in trophic webs such as in the McMurdo Sound (Brey 1995). Furthermore, in phylogenetics, *S. neumayeri* is a sister species of the Mediterranean sea urchin *P. lividus*, in which the biological responses to model nanoplastics (PS) have been widely characterized (Pinsino et al., 2017; Marques-Santos et al., 2018; Murano et al., 2021), thus allowing for a close comparison upon the effects of nanoplastics at the molecular level. Other key taxa that need to be considered in future studies should include arthropods, bivalves, whelks and other echinoids to fully represent polar benthic community dynamics. Our understanding of nanoplastics pathways and impacts towards polar regions is still at an early stage. Unknowns currently range from lack of field monitoring observations to potential effects on the ecophysiology of polar endemic species. As emerging contaminants in polar environments, nanoplastics need to be addressed together with other anthropogenic and environmental stressors impacting polar marine ecosystems.

Up to now few studies have investigated nanoplastic effects under future climate change conditions, *e.g.*, increase in sea temperature and ocean acidification. Rowlands et al. (2021) showed that PS-NH_2 (size of 160 nm, at 2.5 mgL^{-1}) and reduced pH (7.7) significantly impacted the early development of Antarctic krill, while the single stressors (PS-NH_2, low pH) did not cause any effect compared to the control group. Through an intergenerational study, Rowlands et al. (2023) further underlined the resilience of krill embryos to PS-NH_2 as single stressor under short-term exposure conditions.

Similarly, PS-NH_2 (50 nm, at 1 mgL^{-1}) and reduced pH (7.8) significantly increased the mortality rate in the sub-Antarctic pteropod *Limacina retroversa* through an additive effect (Manno et al., 2022). These studies show the ability of nanoplastics to lower the biological thresholds of Antarctic marine ecosystems and underline the importance of multi-stressor experiments resembling natural scenarios or near future conditions.

4. Models vs bioindicators and marine ecosystem services

Along with the increase of field monitoring campaigns, several marine species have been recognized as potential targets of plastic pollution, from top predators such as marine mammals to less complex organisms such as microalgae (Casabianca et al., 2021; Fossi et al., 2018), with consequences not only on survival of natural populations but also on their important ecological role. Marine model species have been selected by using the same criteria applied for chemical risk assessment in order to meet regulatory requirements and standard parameters (*i.e.,* easy to handle, knowledge on eco-physiology and responsive to external chemical stressors). On the other hand, by studying model organisms in laboratory settings and under standardised replicable conditions, we can scarcely infer conclusions about the real natural environment and ecosystem services and this is particularly true for nanoplastics. The demonstrated significant changes to which nanoplastics undergo when released in seawater (*i.e.,* aggregation/agglomeration, eco-corona formation) together with the paucity of data on the actual nanoplastic amount at sea, strongly limit ERA for the marine environment (Corsi et al., 2024; Corsi et al., 2021).

Plastic pollution has been recognized as a threat for ocean carbon sequestration, for the carbon pump and pool by affecting phytoplankton and zooplankton (Shen et al., 2020b; Galgani and Loiselle, 2021; Galgani et al., 2022). How nanoplastics could potentially act in this regard is still in infancy although their colloidal behavior and documented bio-nano-interactions with both phytoplanktonic and zooplanktonic species raise concerns (Sander et al., 2019; Bellingeri et al., 2020; Bergami et al., 2020).

First attempts to quantify the negative impact of plastic pollution estimated from 1 to 5% loss or reduction in marine ecosystem services, resulting in a calculated economic impact of US$ 2.5 trillion (Kumar 2021). Tourism, fisheries with supply of seafood and cultural benefits are only some among those economic sectors which could face serious constraints due to plastic pollution, with loss of jobs and income being connected also to human welfare (Beaumont et al., 2019). Connections among plastic production, marine litter and global warming have been made both in terms of industrial emission (Shen et al., 2020a) and plastic-induced variations in the solar radiation along the water column; as such they can alter physical processes at the ocean surface and near-surface layers as well as trigger climate feedback mechanisms (VishnuRadhan et al., 2019; Cornejo-D'Ottone et al., 2020).

Nanoplastics could thus amplify the impact of multi-stressors (*i.e.,* either physical and chemical with climate-induced stressors), which could compromise population and ecosystem health and resilience.

The selection of bioindicators as representative of marine ecosystem services is utmost important considering the increasing threats faced by marine coastal areas, whose habitats and their taxa account for 20% of ecosystem services, linking structures, processes and functions with the derived economic and social values and benefits (Scherber et al., 2016; Ferreira et al., 2017; Culhane et al., 2018). Coastal zones display unique marine ecosystems heavily exploited and polluted with various degree of degradation and loss in estuarine and coastal wetlands, as marshes and mangroves (35–50%), sand beaches and dunes, seagrass beds (29%), and coral reefs (30%) (Barbier, 2017; Carrasco De La Cruz, 2021). According to the network model proposed by Culhane et al. (2018) >50% of ecosystem services are provided by biotic groups, with macroalgae and epifauna in the highest amount followed by macrophytes and infauna while bacteria, whales and microphytobenthos in the lowest amount. A constant decrease in the number of services from coastal zones to the deep sea is recognized, underlining the importance of a higher degree of protection towards any source of anthropogenic impact including pollution (Culhane et al., 2018).

Since the environmental fate of nanoplastics still relies on models and no field-data on most affected marine compartments are available, the ERA of nanoplastics should be conducted based on different requirements. Model species in short-term acute exposure scenarios are easy-to-handle, reliable and reproducible, but often far from relevant in terms of ecological exposure. Selecting representative bioindicators based on their ecological role and importance in ecosystem functioning and services could provide more useful data for the evaluation of the real threat posed by nanoplastics in specific ecological compartments. Furthermore, an ecosystem-oriented approach should be embraced based on multi-trophic micro- and mesocosm studies resembling environmentally relevant physical-chemical exposure conditions (*i.e.*, light, temperature, pH, etc.) in long-term exposure scenarios. More ecological cascade effects should be investigated by widening marine taxa considered more at risk as well as marine biocenosis. A more grounded knowledge of mechanisms of ecological interactions will provide a more holistic view predictable of impact on ecosystem function and services (Fig. 4).

With the aim of identifying a set of best bioindicators for nanoplastic contamination in the marine environment, a framework that takes into account multiple factors, from trophic position and ecological role with associated ecosystem services, to the protection of endangered species and the risk to humans following dietary transfer, is here proposed. There are already several examples of studies whose aim is to identify the best bioindicators for monitoring marine microplastics such as bioaccumulators (*i.e.,* fish), representative of different components of marine ecosystems and descriptive of a wide range of plastics (Savoca et al., 2022), including iconic and/or endangered species (Fossi et al., 2018).

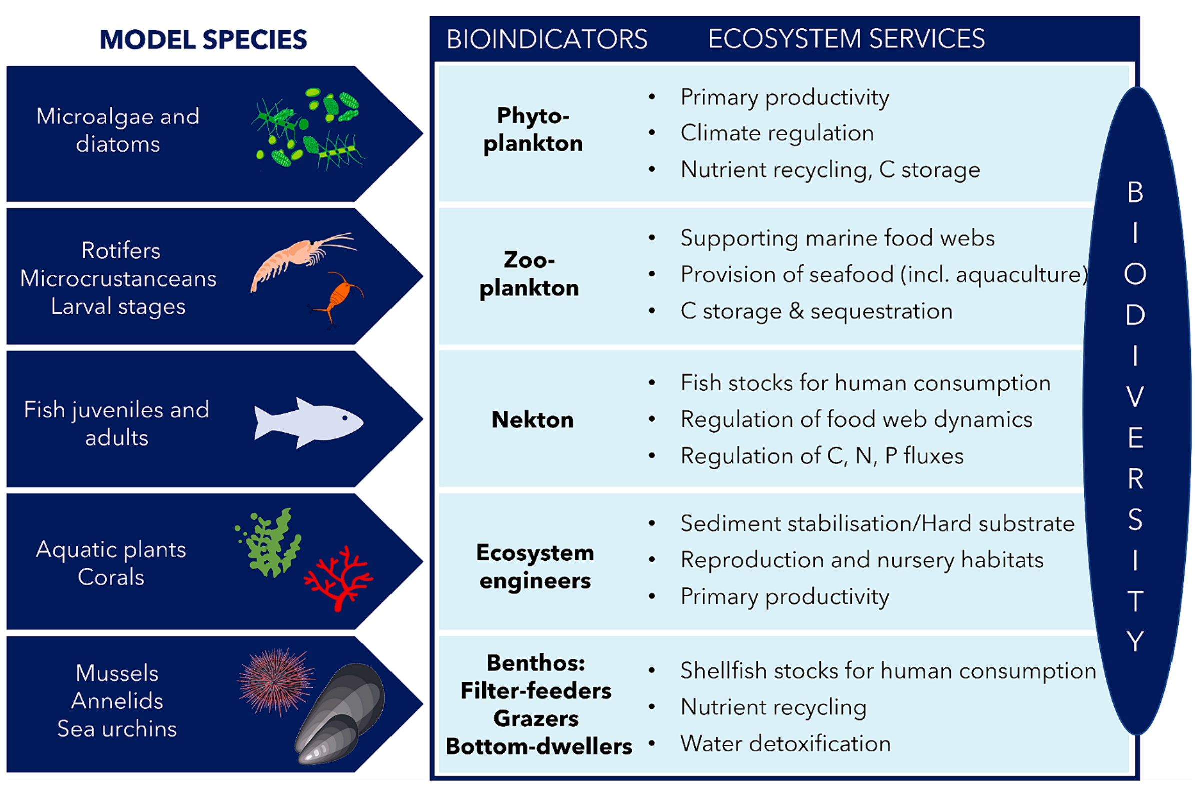

Fig. 4. Models vs bioindicators: the importance of linking model organisms to bioindicators of pelagic and benthic realms and their related ecosystem services.

However, as such, this approach, even though can be applied also for the smaller fractions as nanoplastics, requires a more deep investigation to fulfill some major knowledge gaps.

The risk associated with nanoplastic exposure, also and above all in function of the peculiar properties both in terms of behavior in the saline (*i.e.,* transformations) and biological medium (*i.e.,* easily uptaken), requires particular attention in relation to the risks for humans and the environment (Multisanti et al., 2022).

A 4-tiered holistic approach is proposed here which takes into account the specific needs relating to the choice of bioindicators, not only to the level of protection towards a particular natural resource or ecosystem and associated services, but also to the health of the entire marine ecosystem and consequently of human health (One Health).

The four levels can be identified as such: target species, species most at risk, key species and species of interest for commercial purposes (*i.e.,* seafood) (Fig. 5):

1- target species as those more likely to be exposed to nanoplastics based on their patterns of dispersion and behavior in the marine environment and the position of the species in the food web, including feeding habits. In this case, filter-feeding molluscs could be considered among the best candidates together with other benthic species following processes such as aggregation/agglomeration of nanoplastics, which could lead to significant levels of accumulation within marine sediments. Top predators will be also on the list being potentially receiving high amounts through the biomagnification of nanoplastics along food webs, information which is still scarce or absent and merit more attention. The movement of marine currents and waves could also make the phenomenon of aggregation and precipitation less stable, with a fraction of the nanoplastics still floating and capable of interacting with phyto- and zooplankton, thus identifying another possible target. The lack of knowledge about how nanoplastics can be redistributed in the water column in conditions of realistic environmental concentrations and in the presence of natural factors such as organic matter, waves, light, etc. underlines the need to fill these knowledge gaps in order to identify which taxa could represent potential targets for these CECs. Biomagnification along with bioaccumulation are also relevant for adding species at the top of the food chain in the list.

2- The definition of the endangered species is very close to that of the target species but with a more negative meaning in terms of potential harmful effects for individuals survival and recruitment. The species at risk are to be considered undoubtedly those already affected by other environmental and anthropogenic stressors and/or included in the IUCN list. Whether exposure to nanoplastics occurs, it could affect an already precarious state of health and lead to far more serious consequences on survival than in species in good health. Undoubtedly more studies need to be devoted to understanding the potential exposure of endangered species, both in terms of effect concentrations and ecotoxicological consequences that could lead to major health damages up to, and including, extinction of the species also taking into account synergisms with other stressors.

3- Although key species have a pivotal role for natural ecosystems and some of them are already under stress also in the marine environment (*i.e.,* corals, seagrass meadows), they are often little studied from an ecotoxicological point of view and not at all in terms of nanoplastics. Numerous stress factors, also linked to global climate changes, are putting the survival of natural ecosystems at risk and key species are excellent candidates for plastic pollution monitoring and effects. The first studies conducted on microplastics reveal significant risks for corals and seagrass meadows, also in the Mediterranean area considered a hotspot for plastic pollution, and therefore research on these species is urgently needed for management and mitigation purposes of nanoplastics.

4- With the increase in consumption and production of fishery products globally, microplastic contamination and the risk to human health due to consumption of contaminated food are one of the hot topics of scientific research. Although nanoplastics can have even more serious effects on human health than those of higher dimensions, the lack of accurate methods of size analysis in seafood still represent a knowledge gap on the choice of the best taxa to monitor. Nanoplastics in fishery products and the risk of transfer to humans should therefore receive greater attention in the future, also for regulatory purposes that ensure their safety for the groups most at risk (children, pregnant women, etc.), as is already the case for other marine contaminants (*e.g.*, mercury).

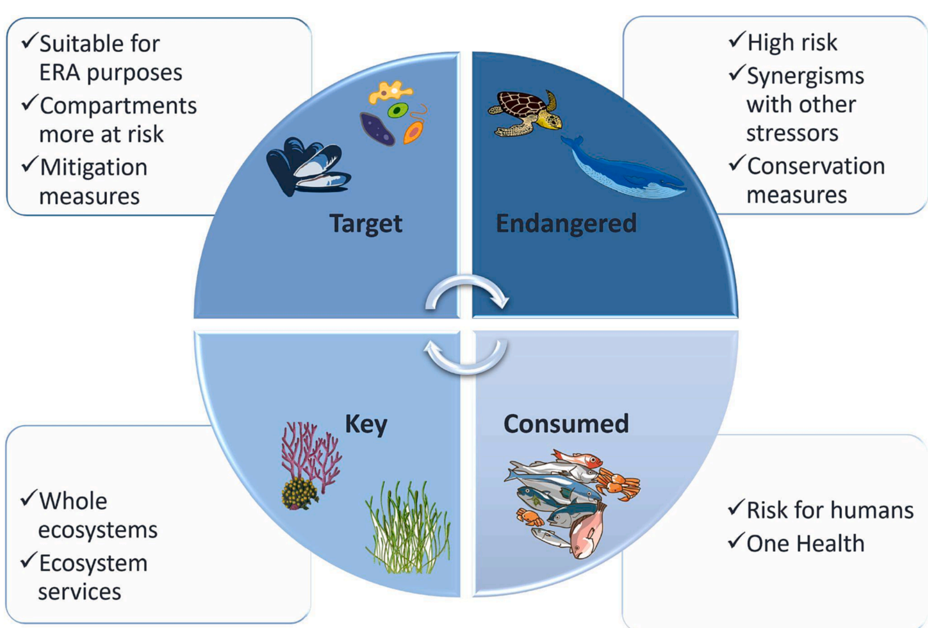

Fig. 5. Proposed framework for selecting bioindicators of nanoplastics in the marine environments: their relevance and interconnections.

5. Conclusions

By coupling the benefits of using model organisms and biological indicators, we can compare marine environmental quality and nanoplastic ERA. To this aim, in future nanoplastic ecotoxicity studies it is crucial to select bioindicators based on specific goals, from those most (likely) encountering them (target) useful for monitoring purposes, to the most threatened and already at risk (endangered) for conservation and protection of marine biodiversity, to those less investigated but having a key ecological role (key species) allowing to predict higher risk of the ecological hierarchy up to ecosystem services (*i.e.,* populations, communities and ecosystems), and to those more linked to human health due to their consumption (consumed seafood). Accordingly, the choice of the appropriate endpoint is of utmost importance, as this should consider the ecological functions played by the selected bioindicator and the possible repercussions on the biotic compartment relying on it. As complex as the matter is, efforts are needed to further examine nanoplastics interactions and fate in the marine environment, in order to understand which are the most affected compartments and define the best practices for the evaluation of nanoplastics ecological impact. To this regard, comparative nano-ecotoxicity studies between temperate and polar species occupying the same ecological niche are encouraged to disclose the vulnerability of different marine ecosystems and the combined effects of multiple stressors scenarios including nanoplastics.

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.

Data availability

Data will be made available on request.

Acknowledgements

The authors would like to thank Mrs Tatiana Rusconi, *PhD candidate*, for the artwork done in the graphical abstract of the ms (https://tatianarusconiart.wixsite.com/tatianarusconiart).

Project funded under the National Recovery and Resilience Plan (NRRP), Mission 4 Component 2 Investment 1.4 – Call for tender No. 3138 of 16 December 2021, rectified by Decree n.3175 of 18 December 2021 of Italian Ministry of University and Research funded by the European Union – NextGenerationEU; Award Number: Project code CN_00000033, Concession Decree No. 1034 of 17 June 2022 adopted by the Italian Ministry of University and Research, CUP B63C22000650007, Project Title "National Biodiversity Future Center – NBFC" .

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecolind.2023.110836.

References

Adams, J.K., Dean, B.Y., Athey, S.N., Jantunen, L.M., Bernstein, S., Stern, G., Diamond, M.L., Finkelstein, S.A., 2021. Anthropogenic particles (including microfibers and microplastics) in marine sediments of the Canadian Arctic. Science of The Total Environment 784, 147155.

Alimi, O.S., Farner Budarz, J., Hernandez, L.M., Tufenkji, N., 2018. Microplastics and nanoplastics in aquatic environments: aggregation, deposition, and enhanced contaminant transport. Environmental science & technology 52, 1704–1724.

Alimi, O.S., Farner, J.M., Rowenczyk, L., Petosa, A.R., Claveau-Mallet, D., Hernandez, L. M., Wilkinson, K.J., Tufenkji, N., 2022. Mechanistic understanding of the aggregation kinetics of nanoplastics in marine environments: Comparing synthetic and natural water matrices. Journal of Hazardous Materials Advances 7, 100115.

Almeida, M., Martins, M.A., Soares, A.M., Cuesta, A., Oliveira, M., 2019. Polystyrene nanoplastics alter the cytotoxicity of human pharmaceuticals on marine fish cell lines. Environmental Toxicology and Pharmacology 69, 57–65.

Al-Sid-Cheikh, M., Rowland, S.J., Stevenson, K., Rouleau, C., Henry, T.B., Thompson, R. C., 2018. Uptake, whole-body distribution, and depuration of nanoplastics by the scallop *Pecten maximus* at environmentally realistic concentrations. Environmental science & technology 52, 14480–14486.

Arini, A., Gigault, J., Venel, Z., Bertucci, A., Baudrimont, M., 2022. The underestimated toxic effects of nanoplastics coming from marine sources: A demonstration on oysters (*Isognomon alatus*). Chemosphere 295, 133824.

Atkinson, A., Ward, P., Hunt, B., Pakhomov, E., Hosie, G., 2012. An overview of Southern Ocean zooplankton data: abundance, biomass, feeding and functional relationships. Ccamlr Science 19, 171–218.

Avio, C.G., Pittura, L., d'Errico, G., Abel, S., Amorello, S., Marino, G., Gorbi, S., Regoli, F., 2020. Distribution and characterization of microplastic particles and textile microfibers in Adriatic food webs: General insights for biomonitoring strategies. Environmental Pollution 258, 113766.

Balbi, T., Camisassi, G., Montagna, M., Fabbri, R., Franzellitti, S., Carbone, C., Dawson, K., Canesi, L., 2017. Impact of cationic polystyrene nanoparticles (PS-NH_2) on early embryo development of *Mytilus galloprovincialis*: Effects on shell formation. Chemosphere 186, 1–9.

Barbier, E.B., 2017. Marine ecosystem services. Current Biology 27, R507. R510.

Baudrimont, M., Arini, A., Guégan, C., Venel, Z., Gigault, J., Pedrono, B., Prunier, J., Maurice, L., Ter Halle, A., Feurtet-Mazel, A., 2020. Ecotoxicity of polyethylene nanoplastics from the North Atlantic oceanic gyre on freshwater and marine organisms (microalgae and filter-feeding bivalves). Environmental Science and Pollution Research 27, 3746–3755.

Beaumont, N.J., Aanesen, M., Austen, M.C., Börger, T., Clark, J.R., Cole, M., Hooper, T., Lindeque, P.K., Pascoe, C., Wyles, K.J., 2019. Global ecological, social and economic impacts of marine plastic. Marine Pollution Bulletin 142, 189–195.

Bellingeri, A., Casabianca, S., Capellacci, S., Faleri, C., Paccagnini, E., Lupetti, P., Koelmans, A.A., Penna, A., Corsi, I., 2020. Impact of polystyrene nanoparticles on marine diatom *Skeletonema marinoi* chain assemblages and consequences on their ecological role in marine ecosystems. Environmental Pollution 114268.

Bergami, E., Bocci, E., Vannuccini, M.L., Monopoli, M., Salvati, A., Dawson, K.A., Corsi, I., 2016. Nano-sized polystyrene affects feeding, behavior and physiology of brine shrimp *Artemia franciscana* larvae. Ecotoxicology and environmental safety 123, 18–25.

Bergami, E., Pugnalini, S., Vannuccini, M., Manfra, L., Faleri, C., Savorelli, F., Dawson, K., Corsi, I., 2017. Long-term toxicity of surface-charged polystyrene nanoplastics to marine planktonic species *Dunaliella tertiolecta* and *Artemia franciscana*. Aquatic Toxicology 189, 159–169.

Bergami, E., Emerenciano, A.K., González-Aravena, M., Cárdenas, C., Hernández, P., Silva, J., Corsi, I., 2019. Polystyrene nanoparticles affect the innate immune system of the Antarctic sea urchin *Sterechinus neumayeri*. Polar Biology 42, 743–757.

Bergami, E., Manno, C., Cappello, S., Vannuccini, M., Corsi, I., 2020. Nanoplastics affect moulting and faecal pellet sinking in Antarctic krill (*Euphausia superba*) juveniles. Environment International 143, 105999.

Bergami, E., Ferrari, E., Löder, M.G., Birarda, G., Laforsch, C., Vaccari, L., Corsi, I., 2023. Textile microfibers in wild Antarctic whelk *Neobuccinum eatoni* (Smith, 1875) from Terra Nova Bay (Ross Sea, Antarctica). Environmental Research 216, 114487.

Bergmann, M., Collard, F., Fabres, J., Gabrielsen, G.W., Provencher, J.F., Rochman, C.M., van Sebille, E., Tekman, M.B., 2022. Plastic pollution in the Arctic. Nature Reviews Earth & Environment 3, 323–337.

Bhagat, J., Nishimura, N., Shimada, Y., 2021. Toxicological interactions of microplastics/nanoplastics and environmental contaminants: Current knowledge and future perspectives. Journal of Hazardous Materials 405, 123913.

Brandts, I., Barría, C., Martins, M., Franco-Martínez, L., Barreto, A., Tvarijonaviciute, A., Tort, L., Oliveira, M., Teles, M., 2021. Waterborne exposure of gilthead seabream (*Sparus aurata*) to polymethylmethacrylate nanoplastics causes effects at cellular and molecular levels. Journal of Hazardous Materials 403, 123590.

Bundschuh, M., Filser, J., Lüderwald, S., McKee, M.S., Metreveli, G., Schaumann, G.E., Schulz, R., Wagner, S., 2018. Nanoparticles in the environment: where do we come from, where do we go to? Environmental Sciences Europe 30, 1–17.

Caputo, F., Vogel, R., Savage, J., Vella, G., Law, A., Della Camera, G., Hannon, G., Peacock, B., Mehn, D., Ponti, J., 2021. Measuring particle size distribution and mass concentration of nanoplastics and microplastics: addressing some analytical challenges in the sub-micron size range. Journal of Colloid and Interface Science 588, 401–417.

Caron, D.A., Alexander, H., Allen, A.E., Archibald, J.M., Armbrust, E.V., Bachy, C., Bell, C.J., Bharti, A., Dyhrman, S.T., Guida, S.M., 2017. Probing the evolution, ecology and physiology of marine protists using transcriptomics. Nature Reviews Microbiology 15, 6–20.

Carrasco De La Cruz, P.M., 2021. The knowledge status of coastal and marine ecosystem services-challenges, limitations and lessons learned from the application of the ecosystem services approach in management. Frontiers in Marine Science 8, 684770.

Caruso, G., Bergami, E., Singh, N., Corsi, I., 2022. Plastic occurrence, sources, and impacts in Antarctic environment and biota. Water Biology and Security 1, 100034.

Casabianca, S., Bellingeri, A., Capellacci, S., Sbrana, A., Russo, T., Corsi, I., Penna, A., 2021. Ecological implications beyond the ecotoxicity of plastic debris on marine phytoplankton assemblage structure and functioning. Environmental Pollution 290, 118101.

Catarino, A.I., Frutos, A., Henry, T.B., 2019. Use of fluorescent-labelled nanoplastics (NPs) to demonstrate NP absorption is inconclusive without adequate controls. Science of the Total Environment 670, 915–920.
Chae, Y., An, Y.-J., 2017. Effects of micro-and nanoplastics on aquatic ecosystems: Current research trends and perspectives. Marine Pollution Bulletin 124, 624–632.
Cornejo-D'Ottone, M., Molina, V., Pavez, J., Silva, N., 2020. Greenhouse gas cycling by the plastisphere: The sleeper issue of plastic pollution. Chemosphere 246, 125709.
Corsi, I., Bergami, E., Grassi, G., 2020. Behavior and bio-interactions of anthropogenic particles in marine environment for a more realistic ecological risk assessment. Frontiers in Environmental Science 8, 60.
Corsi, I., Bellingeri, A., Eliso, M.C., Grassi, G., Liberatori, G., Murano, C., Sturba, L., Vannuccini, M.L., Bergami, E., 2021. Eco-interactions of engineered nanomaterials in the marine environment: Towards an eco-design framework. Nanomaterials 11, 1903.
Culhane, F.E., Frid, C.L., Royo Gelabert, E., White, L., Robinson, L.A., 2018. Linking marine ecosystems with the services they supply: what are the relevant service providing units? Ecological Applications 28, 1740–1751.
Cullen-Unsworth, L., Unsworth, R., 2013. Seagrass meadows, ecosystem services, and sustainability. Environment: Science and policy for sustainable development 55, 14–28.
Davranche, M., Lory, C., Le Juge, C., Blancho, F., Dia, A., Grassl, B., El Hadri, H., Pascal, P., Gigault, J., 2020. NanoImpact 20, 100262.
Dawson, A.L., Kawaguchi, S., King, C.K., Townsend, K.A., King, R., Huston, W.M., Nash, S.M.B., 2018. Turning microplastics into nanoplastics through digestive fragmentation by Antarctic krill. Nature communications 9, 1001.
de Los Santos, C.B., Krause-Jensen, D., Alcoverro, T., Marbà, N., Duarte, C.M., Van Katwijk, M.M., Pérez, M., Romero, J., Sánchez-Lizaso, J.L., Roca, G., 2019. Recent trend reversal for declining European seagrass meadows. Nature communications 10, 3356.
Della Torre, C., Bergami, E., Salvati, A., Faleri, C., Cirino, P., Dawson, K., Corsi, I., 2014. Accumulation and embryotoxicity of polystyrene nanoparticles at early stage of development of sea urchin embryos *Paracentrotus lividus*. Environmental Science & Technology 48, 12302–12311.
Deng, H., He, J., Feng, D., Zhao, Y., Sun, W., Yu, H., Ge, C., 2021. Microplastics pollution in mangrove ecosystems: a critical review of current knowledge and future directions. Science of the Total Environment 753, 142041.
Duarte, C.M., Losada, I.J., Hendriks, I.E., Mazarrasa, I., Marbà, N., 2013. The role of coastal plant communities for climate change mitigation and adaptation. Nature climate change 3, 961–968.
Ekvall, M.T., Lundqvist, M., Kelpsiene, E., Šileikis, E., Gunnarsson, S.B., Cedervall, T., 2019. Nanoplastics formed during the mechanical breakdown of daily-use polystyrene products. Nanoscale Advances 1, 1055–1061.
Eliso, M.C., Bergami, E., Manfra, L., Spagnuolo, A., Corsi, I., 2020. Toxicity of nanoplastics during the embryogenesis of the ascidian *Ciona robusta* (Phylum Chordata). Nanotoxicology 14, 1415–1431.
Eliso, M.C., Bergami, E., Bonciani, L., Riccio, R., Belli, G., Belli, M., Corsi, I., Spagnuolo, A., 2023. Application of transcriptome profiling to inquire into the mechanism of nanoplastics toxicity during *Ciona robusta* embryogenesis. Environmental Pollution 318, 120892.
Enfrin, M., Lee, J., Gibert, Y., Basheer, F., Kong, L., Dumée, L.F., 2020. Release of hazardous nanoplastic contaminants due to microplastics fragmentation under shear stress forces. Journal of Hazardous Materials 384, 121393.
Everaert, G., De Rijcke, M., Lonneville, B., Janssen, C., Backhaus, T., Mees, J., van Sebille, E., Koelmans, A., Catarino, A.I., Vandegehuchte, M.B., 2020. Risks of floating microplastic in the global ocean. Environmental Pollution 267, 115499.
Fang, C., Zheng, R., Zhang, Y., Hong, F., Mu, J., Chen, M., Song, P., Lin, L., Lin, H., Le, F., 2018. Microplastic contamination in benthic organisms from the Arctic and sub-Arctic regions. Chemosphere 209, 298–306.
Ferreira, A.M., Marques, J.C., Seixas, S., 2017. Integrating marine ecosystem conservation and ecosystems services economic valuation: Implications for coastal zones governance. Ecological Indicators 77, 114–122.
Fossi, M.C., Pedà, C., Compa, M., Tsangaris, C., Alomar, C., Claro, F., Ioakeimidis, C., Galgani, F., Hema, T., Deudero, S., Romeo, T., Battaglia, P., Andaloro, F., Caliani, I., Casini, S., Panti, C., Baini, M., 2018. Bioindicators for monitoring marine litter ingestion and its impacts on Mediterranean biodiversity. Environmental Pollution 237, 1023–1040.
Frehland, S., Kaegi, R., Hufenus, R., Mitrano, D.M., 2020. Long-term assessment of nanoplastic particle and microplastic fiber flux through a pilot wastewater treatment plant using metal-doped plastics. Water Research 182, 115860.
Galgani, L., Loiselle, S.A., 2021. Plastic pollution impacts on marine carbon biogeochemistry. Environmental Pollution 268, 115598.
Galgani, L., Goßmann, I., Scholz-Böttcher, B., Jiang, X., Liu, Z., Scheidemann, L., Schlundt, C., Engel, A., 2022. Hitchhiking into the Deep: How Microplastic Particles are Exported through the Biological Carbon Pump in the North Atlantic Ocean. Environmental Science & Technology 56, 15638–15649.
Gaspar, T.R., Chi, R.J., Parrow, M.W., Ringwood, A.H., 2018. Cellular bioreactivity of micro-and nano-plastic particles in oysters. Frontiers in Marine Science 5, 345.
Gerstenbacher, C.M., Finzi, A.C., Rotjan, R.D., Novak, A.B., 2022. A review of microplastic impacts on seagrasses, epiphytes, and associated sediment communities. Environmental Pollution 119108.
Gewert, B., Plassmann, M.M., MacLeod, M., 2015. Pathways for degradation of plastic polymers floating in the marine environment. Environmental Science: Processes & Impacts 17, 1513–1521.
Gigault, J., Pedrono, B., Maxit, B., Ter Halle, A., 2016. Marine plastic litter: the unanalyzed nano-fraction. Environmental Science: Nano 3, 346–350.
Gigault, J., Ter Halle, A., Baudrimont, M., Pascal, P.-Y., Gauffre, F., Phi, T.-L., El Hadri, H., Grassl, B., Reynaud, S., 2018. Current opinion: what is a nanoplastic? Environmental Pollution 235, 1030–1034.
Gigault, J., El Hadri, H., Nguyen, B., Grassl, B., Rowenczyk, L., Tufenkji, N., Feng, S., Wiesner, M., 2021. Nanoplastics are neither microplastics nor engineered nanoparticles. Nature nanotechnology 16, 501–507.
Gomes, T., Almeida, A.C., Georgantzopoulou, A., 2020. Characterization of cell responses in Rhodomonas baltica exposed to PMMA nanoplastics. Science of the Total Environment 726, 138547.
Gondikas, A., Gallego-Urrea, J., Halbach, M., Derrien, N., Hassellöv, M., 2020. Nanomaterial fate in seawater: A rapid sink or intermittent stabilization? Frontiers in Environmental Science 8, 151.
González-Fernández, C., Le Grand, F., Bideau, A., Huvet, A., Paul-Pont, I., Soudant, P., 2020. Nanoplastics exposure modulate lipid and pigment compositions in diatoms. Environmental Pollution 262, 114274.
Grassi, G., Landi, C., Della Torre, C., Bergami, E., Bini, L., Corsi, I., 2019. Proteomic profile of the hard corona of charged polystyrene nanoparticles exposed to sea urchin *Paracentrotus lividus* coelomic fluid highlights potential drivers of toxicity. Environmental Science: Nano 6, 2937–2947.
Grassi, G., Gabellieri, E., Cioni, P., Paccagnini, E., Faleri, C., Lupetti, P., Corsi, I., Morelli, E., 2020. Interplay between extracellular polymeric substances (EPS) from a marine diatom and model nanoplastic through eco-corona formation. Science of The Total Environment 725, 138457.
Hartmann, N.B., Hüffer, T., Thompson, R.C., Hassell€ov, M., Verschoor, A., Daugaard, A. E., Rist, S., Karlsson, T., Brennholt, N., Cole, M., Herrling, M.P., Hess, M.C., Ivleva, N. P., Lusher, A.L., Wagner, M., 2019. Are we speaking the same language? Recommendations for a definition and categorization framework for plastic debris. Environmental Science & Technology. 53. 1039–1047.
Hernandez, L.M., Yousefi, N., Tufenkji, N., 2017. Are there nanoplastics in your personal care products? Environmental Science & Technology Letters 4, 280–285.
Hoffmann, L., Eggers, S.L., Allhusen, E., Katlein, C., Peeken, I., 2020. Interactions between the ice algae *Fragillariopsis cylindrus* and microplastics in sea ice. Environment International 139, 105697.
Horton, A.A., Barnes, D.K., 2020. Microplastic pollution in a rapidly changing world: Implications for remote and vulnerable marine ecosystems. Science of The Total Environment 738, 140349.
Hughes, T.P., Anderson, K.D., Connolly, S.R., Heron, S.F., Kerry, J.T., Lough, J.M., Baird, A.H., Baum, J.K., Berumen, M.L., Bridge, T.C., 2018. Spatial and temporal patterns of mass bleaching of corals in the Anthropocene. Science 359, 80–83.
Jeong, C.-B., Won, E.-J., Kang, H.-M., Lee, M.-C., Hwang, D.-S., Hwang, U.-K., Zhou, B., Souissi, S., Lee, S.-J., Lee, J.-S., 2016. Microplastic size-dependent toxicity, oxidative stress induction, and p-JNK and p-p38 activation in the monogonont rotifer (*Brachionus koreanus*). Environmental Science & Technology 50, 8849–8857.
Jeong, C.-B., Kang, H.-M., Lee, M.-C., Kim, D.-H., Han, J., Hwang, D.-S., Souissi, S., Lee, S.-J., Shin, K.-H., Park, H.G., 2017. Adverse effects of microplastics and oxidative stress-induced MAPK/Nrf2 pathway-mediated defense mechanisms in the marine copepod *Paracyclopina nana*. Scientific Reports 7, 41323.
Jones, K.L., Hartl, M.G., Bell, M.C., Capper, A., 2020. Microplastic accumulation in a Zostera marina L. bed at Deerness Sound, Orkney, Scotland. Marine Pollution Bulletin 152, 110883.
Kalčíková, G., 2020. Aquatic vascular plants–A forgotten piece of nature in microplastic research. Environmental Pollution 262, 114354.
Kelly, A., Lannuzel, D., Rodemann, T., Meiners, K., Auman, H., 2020. Microplastic contamination in east Antarctic sea ice. Marine Pollution Bulletin 154, 111130.
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, 460–462.
Lambert, S., Wagner, M., 2016a. Characterisation of nanoplastics during the degradation of polystyrene. Chemosphere 145, 265–268.
Lambert, S., Wagner, M., 2016b. Formation of microscopic particles during the degradation of different polymers. Chemosphere 161, 510–517.
Lebreton, L., Andrady, A., 2019. Future scenarios of global plastic waste generation and disposal. Palgrave Communications 5, 1–11.
Lehner, R., Weder, C., Petri-Fink, A., Rothen-Rutishauser, B., 2019. Emergence of nanoplastic in the environment and possible impact on human health. Environmental Science & Technology 53, 1748–1765.
Lenz R., Enders K., Nielsen T.G., 2016. Microplastic exposure studies should be environmentally realistic. Proceedings of the National Academy of Sciences. 201606615.
Li, R., Wei, C., Jiao, M., Wang, Y., Sun, H., 2022. Mangrove leaves: an undeniably important sink of MPs from tidal water and air. Journal of Hazardous Materials 426, 128138.
Llorca, M., Vega-Herrera, A., Schirinzi, G., Savva, K., Abad, E., Farré, M., 2021. Screening of suspected micro (nano) plastics in the Ebro Delta (Mediterranean Sea). Journal of Hazardous Materials 404, 124022.
Long, M., Moriceau, B., Gallinari, M., Lambert, C., Huvet, A., Raffray, J., Soudant, P., 2015. Interactions between microplastics and phytoplankton aggregates: Impact on their respective fates. Marine Chemistry 175, 39–46.
Lusher, A., Welden, N., Sobral, P., Cole, M., 2017. Sampling, isolating and identifying microplastics ingested by fish and invertebrates. Analytical Methods 9, 1346–1360.
Manfra, L., Rotini, A., Bergami, E., Grassi, G., Faleri, C., Corsi, I., 2017. Comparative ecotoxicity of polystyrene nanoparticles in natural seawater and reconstituted seawater using the rotifer *Brachionus plicatilis*. Ecotoxicology and Environmental Safety 145, 557–563.

Manno, C., Fielding, S., Stowasser, G., Murphy, E., Thorpe, S., Tarling, G., 2020. Continuous moulting by Antarctic krill drives major pulses of carbon export in the north Scotia Sea. Southern Ocean. Nature Communications 11, 6051.
Manno, C., Peck, V.L., Corsi, I., Bergami, E., 2022. Under pressure: Nanoplastics as a further stressor for sub-Antarctic pteropods already tackling ocean acidification. Marine Pollution Bulletin 174, 1131176.
Marangoni, L.F., Beraud, E., Ferrier-Pagès, C., 2022. Polystyrene nanoplastics impair the photosynthetic capacities of Symbiodiniaceae and promote coral bleaching. Science of the Total Environment 815, 152136.
Marques-Santos, L., Grassi, G., Bergami, E., Faleri, C., Balbi, T., Salis, A., Damonte, G., Canesi, L., Corsi, I., 2018. Cationic polystyrene nanoparticle and the sea urchin immune system: biocorona formation, cell toxicity, and multixenobiotic resistance phenotype. Nanotoxicology 12, 847–867.
Mateos-Cárdenas, A., van Pelt, F.N., O'Halloran, J., Jansen, M.A., 2021. Adsorption, uptake and toxicity of micro-and nanoplastics: Effects on terrestrial plants and aquatic macrophytes. Environmental Pollution 284, 117183.
Materić, D., Holzinger, R., Niemann, H., 2022a. Nanoplastics and ultrafine microplastic in the Dutch Wadden Sea-The hidden plastics debris? Science of the Total Environment 846, 157371.
Materić, D., Kjær, H.A., Vallelonga, P., Tison, J.-L., Röckmann, T., Holzinger, R., 2022b. Nanoplastics measurements in Northern and Southern polar ice. Environmental Research 208, 112741.
Materić, D., Peacock, M., Dean, J., Futter, M., Maximov, T., Moldan, F., Röckmann, T., Holzinger, R., 2022c. Presence of nanoplastics in rural and remote surface waters. Environmental Research Letters 17, 054036.
Menicagli, V., Castiglione, M.R., Balestri, E., Giorgetti, L., Bottega, S., Sorce, C., Spanò, C., Lardicci, C., 2022. Early evidence of the impacts of microplastic and nanoplastic pollution on the growth and physiology of the seagrass *Cymodocea nodosa*. Science of the Total Environment 838, 156514.
Miller, M.E., Hamann, M., Kroon, F.J., 2020. Bioaccumulation and biomagnification of microplastics in marine organisms: A review and meta-analysis of current data. PLoS One1 15, e0240792.
Mitrano, D.M., Beltzung, A., Frehland, S., Schmiedgruber, M., Cingolani, A., Schmidt, F., 2019. Synthesis of metal-doped nanoplastics and their utility to investigate fate and behaviour in complex environmental systems. Nature Nanotechnology 14, 362–368.
Multisanti, C.R., Merola, C., Perugini, M., Aliko, V., Faggio, C., 2022. Sentinel species selection for monitoring microplastic pollution: A review on one health approach. Ecological Indicators 145, 109587.
Munari, C., Infantini, V., Scoponi, M., Rastelli, E., Corinaldesi, C., Mistri, M., 2017. Microplastics in the sediments of terra nova bay (ross sea, Antarctica). Marine Pollution Bulletin 122, 161–165.
Murano, C., Agnisola, C., Caramiello, D., Castellano, I., Casotti, R., Corsi, I., Palumbo, A., 2020. How sea urchins face microplastics: uptake, tissue distribution and immune system response. Environmental Pollution 264, 114685.
Murano, C., Bergami, E., Liberatori, G., Palumbo, A., Corsi, I., 2021. Interplay between nanoplastics and the immune system of the mediterranean sea urchin *Paracentrotus lividus*. Frontiers in Marine Science 8, 647394.
Nguyen, B., Claveau-Mallet, D., Hernandez, L.M., Xu, E.G., Farner, J.M., Tufenkji, N., 2019. Separation and analysis of microplastics and nanoplastics in complex environmental samples. Accounts of Chemical Research 52, 858–866.
Nordlund, L.M., Jackson, E.L., Nakaoka, M., Samper-Villarreal, J., Beca-Carretero, P., Creed, J.C., 2018. Seagrass ecosystem services–What's next? Marine Pollution Bulletin 134, 145–151.
Obbard, R.W., 2018. Microplastics in polar regions: the role of long range transport. Current Opinion in Environmental Science & Health 1, 24–29.
OECD 317, 2021. Guidance Document on Aquatic and Sediment Toxicological Testing of Nanomaterials. Series on Testing and Assessment No 317. ENV/JM/MONO(2020)8.
Oriekhova, O., Stoll, S., 2018. Heteroaggregation of nanoplastic particles in the presence of inorganic colloids and natural organic matter. Environmental Science: Nano 5, 792–799.
Otero, J., Álvarez-Salgado, X.A., Bode, A., 2020. Phytoplankton diversity effect on ecosystem functioning in a coastal upwelling system. Frontiers in Marine Science 7, 592255.
Peck, L.S., 2018. Antarctic marine biodiversity: adaptations, environments and responses to change. Oceanography and Marine Biology. Taylor & Francis.
Pietrelli, L., Di Gennaro, A., Menegoni, P., Lecce, F., Poeta, G., Acosta, A.T., Battisti, C., Iannilli, V., 2017. Pervasive plastisphere: first record of plastics in egagropiles (*Posidonia spheroids*). Environmental Pollution 229, 1032–1036.
Pikuda, O., Xu, E.G., Berk, D., Tufenkji, N., 2018. Toxicity assessments of micro-and nanoplastics can be confounded by preservatives in commercial formulations. Environmental Science & Technology Letters 6, 21–25.
Pinsino, A., Bergami, E., Della Torre, C., Vannuccini, M.L., Addis, P., Secci, M., Dawson, K.A., Matranga, V., Corsi, I., 2017. Amino-modified polystyrene nanoparticles affect signalling pathways of the sea urchin (*Paracentrotus lividus*) embryos. Nanotoxicology 11, 201–209.
Porter, A., Lyons, B.P., Galloway, T.S., Lewis, C., 2018. Role of marine snows in microplastic fate and bioavailability. Environmental Science & Technology 52, 7111–7119.
Pradel, A., Ferreres, S., Veclin, C., El Hadri, H., Gautier, M., Grassl, B., Gigault, J., 2021a. Stabilization of fragmental polystyrene nanoplastic by natural organic matter: insight into mechanisms. ACS ES&T Water 1, 1198–1208.
Pradel, A., Gautier, M., Bavay, D., Gigault, J., 2021b. Micro-and nanoplastic transfer in freezing saltwater: implications for their fate in polar waters. Environmental Science: Processes & Impacts 23, 1759–1770.
Reichert, J., Schellenberg, J., Schubert, P., Wilke, T., 2018. Responses of reef building corals to microplastic exposure. Environmental Pollution 237, 955–960.
Reynolds, L.K., Waycott, M., McGlathery, K.J., Orth, R.J., 2016. Ecosystem services returned through seagrass restoration. Restoration Ecology 24, 583–588.
Rezania, S., Park, J., Din, M.F.M., Taib, S.M., Talaiekhozani, A., Yadav, K.K., Kamyab, H., 2018. Microplastics pollution in different aquatic environments and biota: A review of recent studies. Marine Pollution Bulletin 133, 191–208.
Ripken, C., Khalturin, K., Shoguchi, E., 2020. Response of coral reef dinoflagellates to nanoplastics under experimental conditions suggests downregulation of cellular metabolism. Microorganisms 8, 1759.
Rogers, A.D., 2007. Evolution and biodiversity of Antarctic organisms: a molecular perspective. Philosophical transactions of the royal society B: Biological Sciences 362, 2191–2214.
Roman, C., Mahé, P., Latchere, O., Catrouillet, C., Gigault, J., Métais, I., Châtel, A., 2023. Effect of size continuum from nanoplastics to microplastics on marine mussel *Mytilus edulis*: Comparison in vitro/in vivo exposure scenarios. Comparative Biochemistry and Physiology Part C: Toxicology & Pharmacology 264, 109512.
Rota, E., Bergami, E., Corsi, I., Bargagli, R., 2022. Macro-and microplastics in the Antarctic environment: Ongoing assessment and perspectives. Environments 9, 93.
Rowlands, E., Galloway, T., Manno, C., 2021. A Polar outlook: Potential interactions of micro-and nano-plastic with other anthropogenic stressors. Science of the Total Environment 754, 142379.
Rowlands, E., Galloway, T., Cole, M., Lewis, C., Hacker, C., Peck, V.L., Thorpe, S., Blackbird, S., Wolff, G.A., Manno, C., 2023. Scoping intergenerational effects of nanoplastic on the lipid reserves of Antarctic krill embryos. Aquatic Toxicology 261, 106591.
Sanchez-Vidal, A., Canals, M., De Haan, W.P., Romero, J., Veny, M., 2021. Seagrasses provide a novel ecosystem service by trapping marine plastics. Scientific Reports 11, 1–7.
Sander, M., Kohler, H.-P.-E., McNeill, K., 2019. Assessing the environmental transformation of nanoplastic through 13C-labelled polymers. Nature Nanotechnology 14, 301–303.
Santini, S., De Beni, E., Martellini, T., Sarti, C., Randazzo, D., Ciraolo, R., Scopetani, C., Cincinelli, A., 2022. Occurrence of Natural and Synthetic Micro-Fibers in the Mediterranean Sea: A Review. Toxics 10, 391.
Savoca, M.S., Kühn, S., Sun, C., Avery-Gomm, S., Choy, C.A., Dudas, S., Hong, S.H., Hyrenbach, K.D., Li, T., Ka-yan, C.N., Provencher, J.F., Lynch, J.M., 2022. Towards a North Pacific Ocean long-term monitoring program for plastic pollution: A review and recommendations for plastic ingestion bioindicators. Environmental Pollution 310, 119861.
Scherber, C., Scheu, S., Sutherland, W.J., Tamaddoni-Nezhad, A., ter Braak, C., Termansen, M., Thompson, M.S., Tscharntke, T., Vacher, C., van der Geest, H., 2016. Networking Our Way to Better Ecosystem Service Provision. Trends in Ecology & Evolution 31, 105–115.
Schirinzi, G.F., Llorca, M., Seró, R., Moyano, E., Barceló, D., Abad, E., Farré, M., 2019. Trace analysis of polystyrene microplastics in natural waters. Chemosphere 236, 124321.
Sendra, M., Staffieri, E., Yeste, M.P., Moreno-Garrido, I., Gatica, J.M., Corsi, I., Blasco, J., 2019. Are the primary characteristics of polystyrene nanoplastics responsible for toxicity and ad/absorption in the marine diatom *Phaeodactylum tricornutum*? Environmental Pollution 249, 610–619.
Sendra, M., Saco, A., Yeste, M.P., Romero, A., Novoa, B., Figueras, A., 2020. Nanoplastics: From tissue accumulation to cell translocation into *Mytilus galloprovincialis* hemocytes. resilience of immune cells exposed to nanoplastics and nanoplastics plus Vibrio splendidus combination. Journal of Hazardous Materials 388, 121788.
Sfriso, A.A., Tomio, Y., Rosso, B., Gambaro, A., Sfriso, A., Corami, F., Rastelli, E., Corinaldesi, C., Mistri, M., Munari, C., 2020. Microplastic accumulation in benthic invertebrates in Terra Nova Bay (Ross Sea, Antarctica). Environment International 137, 105587.
Shen, M., Huang, W., Chen, M., Song, B., Zeng, G., Zhang, Y., 2020a. (Micro) plastic crisis: un-ignorable contribution to global greenhouse gas emissions and climate change. Journal of Cleaner Production 254, 120138.
Shen, M., Ye, S., Zeng, G., Zhang, Y., Xing, L., Tang, W., Wen, X., Liu, S., 2020b. Can microplastics pose a threat to ocean carbon sequestration? Marine Pollution Bulletin 150, 110712.
Shiu, R.-F., Vazquez, C.I., Tsai, Y.-Y., Torres, G.V., Chen, C.-S., Santschi, P.H., Quigg, A., Chin, W.-C., 2020a. Nano-plastics induce aquatic particulate organic matter (microgels) formation. Science of The Total Environment 706, 135681.
Shiu, R.-F., Vazquez, C.I., Chiang, C.-Y., Chiu, M.-H., Chen, C.-S., Ni, C.-W., Gong, G.-C., Quigg, A., Santschi, P.H., Chin, W.-C., 2020b. Nano-and microplastics trigger secretion of protein-rich extracellular polymeric substances from phytoplankton. Science of the Total Environment 748, 141469.
Silva, M., Oliveira, M., Lopéz, D., Martins, M., Figueira, E., Pires, A., 2020a. Do nanoplastics impact the ability of the polychaeta *Hediste diversicolor* to regenerate? Ecological Indicators 110, 105921.
Silva, M.S., Oliveira, M., Valente, P., Figueira, E., Martins, M., Pires, A., 2020b. Behavior and biochemical responses of the polychaeta *Hediste diversicolor* to polystyrene nanoplastics. Science of the Total Environment 707, 134434.
Singh, N., Tiwari, E., Khandelwal, N., Darbha, G.K., 2019. Understanding the stability of nanoplastics in aqueous environments: effect of ionic strength, temperature, dissolved organic matter, clay, and heavy metals. Environmental Science: Nano 6, 2968–2976.
Siti, H.N., Kamisah, Y., Kamsiah, J., 2015. The role of oxidative stress, antioxidants and vascular inflammation in cardiovascular disease (a review). Vascular Pharmacology 71, 40–56.

Sjollema, S.B., Redondo-Hasselerharm, P., Leslie, H.A., Kraak, M.H., Vethaak, A.D., 2016. Do plastic particles affect microalgal photosynthesis and growth? Aquatic Toxicology 170, 259–261.

Suaria, G., Achtypi, A., Perold, V., Lee, J.R., Pierucci, A., Bornman, T.G., Aliani, S., Ryan, P.G., 2020. Microfibers in oceanic surface waters: A global characterization. Science. Advances 6, eaay8493.

Sun, J., Dai, X., Wang, Q., Van Loosdrecht, M.C., Ni, B.-J., 2019. Microplastics in wastewater treatment plants: Detection, occurrence and removal. Water Research 152, 21–37.

Sun, H., Jiao, R., Wang, D., 2021. The difference of aggregation mechanism between microplastics and nanoplastics: Role of Brownian motion and structural layer force. Environmental Pollution 268, 115942.

Sun, X.-D., Yuan, X.-Z., Jia, Y., Feng, L.-J., Zhu, F.-P., Dong, S.-S., Liu, J., Kong, X., Tian, H., Duan, J.-L., 2020. Differentially charged nanoplastics demonstrate distinct accumulation in Arabidopsis thaliana. Nature Nanotechnology 15, 755–760.

Tallec, K., Huvet, A., Di Poi, C., González-Fernández, C., Lambert, C., Petton, B., Le Goïc, N., Berchel, M., Soudant, P., Paul-Pont, I., 2018. Nanoplastics impaired oyster free living stages, gametes and embryos. Environmental Pollution 242, 1226–1235.

Ter Halle, A., Ladirat, L., Gendre, X., Goudouneche, D., Pusineri, C., Routaboul, C., Tenailleau, C., Duployer, B., Perez, E., 2016. Understanding the fragmentation pattern of marine plastic debris. Environmental Science & Technology 50, 5668–5675.

Ter Halle, A., Jeanneau, L., Martignac, M., Jardé, E., Pedrono, B., Brach, L., Gigault, J., 2017. Nanoplastic in the North Atlantic Subtropical Gyre. Environmental Science & Technology 51, 13689–13697.

Valsesia, A., Parot, J., Ponti, J., Mehn, D., Marino, R., Melillo, D., Muramoto, S., Verkouteren, M., Hackley, V.A., Colpo, P., 2021. Detection, counting and characterization of nanoplastics in marine bioindicators: A proof of principle study. Microplastics and Nanoplastics 1, 1–13.

Veclin, C., Desmet, C., Pradel, A., Valsesia, A., Ponti, J., El Hadri, H., Maupas, T., Pellerin, V., Gigault, J., Grassl, B., 2022. Effect of the surface hydrophobicity–morphology–functionality of nanoplastics on their homoaggregation in seawater. ACS ES&T Water 2, 88–95.

Venâncio, C., Ferreira, I., Martins, M.A., Soares, A.M., Lopes, I., Oliveira, M., 2019. The effects of nanoplastics on marine plankton: A case study with polymethylmethacrylate. Ecotoxicology and Environmental Safety 184, 109632.

Venel, Z., Tabuteau, H., Pradel, A., Pascal, P.-Y., Grassl, B., El Hadri, H., Baudrimont, M., Gigault, J., 2021. Environmental fate modeling of nanoplastics in a salinity gradient using a lab-on-a-chip: where does the nanoscale fraction of plastic debris accumulate? Environmental Science & Technology 55, 3001–3008.

VishnuRadhan, R., Eldho, T., David, T.D., 2019. Can plastics affect near surface layer ocean processes and climate? Marine Pollution Bulletin 140, 274–280.

Ward, J.E., Kach, D.J., 2009. Marine aggregates facilitate ingestion of nanoparticles by suspension-feeding bivalves. Marine Environmental Research 68, 137–142.

Wheeler, K.E., Chetwynd, A.J., Fahy, K.M., Hong, B.S., Tochihuitl, J.A., Foster, L.A., Lynch, I., 2021. Environmental dimensions of the protein corona. Nature Nanotechnology 16, 617–629.

Wilkie Johnston, L., Bergami, E., Rowlands, E., Manno, C., 2023. Organic or junk food? Microplastic contamination in Antarctic krill and salps. Royal Society Open Science 221421, 221421.

Woodhead, A.J., Hicks, C.C., Norström, A.V., Williams, G.J., Graham, N.A., 2019. Coral reef ecosystem services in the Anthropocene. Functional Ecology 33, 1023–1034.

Yin, L., Wen, X., Huang, D., Du, C., Deng, R., Zhou, Z., Tao, J., Li, R., Zhou, W., Wang, Z., 2021. Interactions between microplastics/nanoplastics and vascular plants. Environmental Pollution 290, 117999.

Zhu, W., Liu, W., Chen, Y., Liao, K., Yu, W., Jin, H., 2023. Microplastics in Antarctic krill (Euphausia superba) from Antarctic region. Science of The Total Environment 870, 161880.