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**Ecological Impacts of Marine Plastic
Pollution, Microplastics' Foodweb
Bioaccumulation Modelling and Global
Ocean Footprint: Insights into the
Problems, the Management Implications
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The Nippon Foundation-Ocean Litter Project (2019-2023)**

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Year: 2023

Email: j.alava@oceans.ubc.ca

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Ecological Impacts of Marine Plastic Pollution, Microplastics' Foodweb Bioaccumulation Modelling and Global Ocean Footprint: Insights into the Problems, the Management Implications and Coastal Communities Inequities

Juan José Alava^{1}, Marcia Moreno- Báez², Karly McMullen¹, Mine Banu Tekman³, Abigail. P.W. Barrows^{4,5}, Melanie Bergmann³, Dana Price¹, Wilf Swartz⁶, Yoshitaka Ota⁷*

Affiliations:

¹Ocean Pollution Research Unit & Nippon Foundation-Ocean Litter Project, Institute for the Oceans and Fisheries, University of British Columbia, Vancouver, BC V6T 1Z4, Canada

²Center for International Environment and Resource Policy, The Fletcher School, Tufts University, Medford, Massachusetts, USA.

³Alfred Wegener Institute Helmholtz Centre for Polar and Marine Research, Am Handelshafen 12, 27570 Bremerhaven, Germany

⁴Adventure Scientists, PO Box 1834, Bozeman, MT 59771, USA

⁵College of the Atlantic, Department of Biology, 105 Eden Street, Bar Harbor, ME 04609, USA

⁶Marine Affairs Program, Faculty of Science, Life Sciences Centre, Dalhousie University, Room 808, 1459 Oxford Street, PO Box 15000 Halifax NS B3H 4R2, Canada

⁷Nippon Foundation Ocean Nexus Center, School of Marine and Environmental Affairs EarthLab, University of Washington, Box 355674, Seattle, WA 98195-5674, United States

*Contact information for lead author: j.alava@oceans.ubc.ca

Abstract:

Macroplastics, microplastics, and nanoplastics are increasingly becoming pollutants of great concern in the world's oceans. Many studies have revealed adverse health impacts in marine ecosystems and organisms resulting from microplastic and nanoplastics exposure, ingestion and contamination. Marine biodiversity is readily affected by plastic pollution and coastal communities strongly relying on traditional seafoods and commercial fishing for subsistence are particularly susceptible to the global footprint of ocean plastics. Understanding potential bioaccumulation and biomagnification processes of microplastics in marine foodwebs is critical to advance microplastic science. Concerted bioaccumulation studies and foodweb bioaccumulation modelling of microplastics, addressing trophic transfer, ingestion, bioaccumulation potential and elimination/egestion rates in marine biota and in foodwebs are urgently needed as part of ecotoxicological and human risk assessments. To address these research gaps, this report presents primary research fronts focused on: 1) key contributions from the development of a comprehensive foodweb-bioaccumulation and biomagnification modeling approach for microplastics, using the well-known Chinook salmon-southern resident killer whale foodweb of the Northeastern Pacific, as a practical tool to understand the bioaccumulation and biomagnification behaviour of microplastics; 2) a synthesis of the application of trophic dynamic-ecosystem modeling applying Ecopath and Ecosim (EwE) models with the Ecotracer module; 3) the projection of the global ocean distribution and concentration levels of microplastics, using the databases Litterbase and the Global Microplastic Initiative to track the bioaccumulation and biomagnification potential in tandem with the development of the global microplastic footprint exposure index in selected marine ecosystems and Indigenous coastal communities of the world's ocean; and, 4) a critical narrative of the implications for plastic pollution mitigation and socially equitable interventions and solutions for addressing marine plastic pollution. In conclusion, continued biomonitoring efforts and application of sound bioaccumulation modelling tools in tandem

with the prioritization of knowledge mobilization and community participation via equitable interventions is of paramount importance to ensure effective solutions and mitigation policies that are socially and equitably fair to reduce inequalities and halt marine plastic pollution, following preventive measures and the precautionary approach.

Keywords:

Marine plastic pollution, ocean plastics, macroplastics, microplastics, nanoplastics, Plasticene, exposure, persistence, bioaccumulation, biomagnification potential, trophic magnification factor, toxicity, foodweb bioaccumulation modelling, ecosystem modelling, Ecopath with Ecosim (EwE model), Ecotracer, zooplankton, fish, sea turtles, seabird, marine mammals, seafoods, health effects, coastal communities, microplastic footprint, microplastic potential exposure index, microplastic concentration index, Indigenous coastal communities, equity, wellbeing

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Introduction: Scope of the problem

In the Plasticene (“The Age of Plastics” Haram et al., 2020), ocean pollution by plastics of all sizes (i.e., macroplastics, microplastics, and nanoplastics) is one the most pervasive ecological footprints generated by anthropogenic activities. It presents an urgent pollution and health risks due to the accelerated rate of plastic production and emissions with associated contamination of the global ocean environment and coastal zones (Alava, 2019; Andrady, 2022; Barrows et al. 2018; Bergmann et al. 2015; Bergmann et al. 2017; Ericksen et al., 2014; Jambeck et al. 2015; Kane et al., 2020; Lebreton et al., 2017; Lebreton et al., 2018; MacLeod et al., 2021). The benefits of plastics for society and the economy have indeed come with high environmental costs, mainly negatively affecting marine biodiversity and jeopardizing the public health, wellbeing, and equity dimensions of coastal communities and marginalized minority groups of people (Bennett et al., 2022; Liboiron, 2021; Simon et al., 2021; UNEP, 2021a; UNEP, 2021b; Vandenberg & Ota, 2022).

The first reported statistics of the volume of plastic production was in 1950 when ~2 million tonnes of plastics were produced, globally (Crawford and Quinn, 2017; Geyer et al., 2017; Gibb, 2019). In 2020, a total volume of 367 million tonnes of plastics was produced at the global level (Plastics Europe, 2021), equivalent to 2.45 million blue whales (i.e., considering a blue whale mass of 150 metric tonnes). According to the global assessment of marine litter and plastic pollution by the United Nations Environmental Program (2021), the chronic emissions of plastic waste into aquatic ecosystems are projected to nearly triple by 2040, if meaningful actions are not implemented. Plastic pollution per se is now exceeding and overwhelming the planetary boundaries at unprecedented scales (Arp et al., 2021; MacLeod et al., 2021; Persson et al. 2022). The environmental economic cost resulting from the damage to marine ecosystems by ocean plastic debris has been estimated at more than US\$13 billion annually (Avio et al., 2017). Despite the evident ecological impacts and health effects on marine biodiversity, ecosystems and coastal communities, the current mitigation actions to combat marine plastic pollution are insufficient and outpaced by the increasing plastic production rate and associated emissions into the global ocean.

As part of the Nippon Foundation-Ocean Litter project developed from 2019 to 2023, research fronts on marine plastic pollution and microplastic bioaccumulation were conducted to assess the ecological impacts of ocean plastics and the bioaccumulative behaviour of plastic particles in marine foodwebs. Doing so, in this working paper, we contribute to the ecological, organismal and coastal community impacts of marine plastic pollution and new knowledge on microplastic foodweb bioaccumulation modelling to support ecotoxicological and human risk assessment and inform equitable interventions and management actions for the exposed coastal communities. Thus, the novel research fronts were focused on the following themes:

1. The major findings resulting from the development of a pragmatic kinetic foodweb-bioaccumulation model for microplastics (Alava’s model; Alava 2020), using a well-known marine food web of the Northeastern Pacific (i.e., Chinook salmon-southern resident killer whale), to simulate and predict the accumulation potential of plastic particles as a practical exercise and product aimed to be applied to understand the bioaccumulation and biomagnification behaviour of microplastics with implications for marine species conservation and ecotoxicological and human risk assessments;
2. A synthesis of the application of trophic dynamic-ecosystem modeling applying Ecopath and Ecosim (EwE) models with the Ecotracer module in combination with the global distribution and concentration levels of microplastics in the oceans, using the global database Litterbase (Bergmann et al., 2017), to simulate and track the bioaccumulation and biomagnification potential of microplastics.

Our goal in doing so is to illustrate a global overview of microplastics bioaccumulation in selected large marine ecosystems of the world's ocean;

3. Geospatial modelling analysis and development of the microplastic concentrations index and microplastic potential exposure index in selected marine ecosystems of the global ocean level with exposure risk implications for Indigenous coastal communities heavily reliant on seafood consumption; and,
4. Critical insights on the management of plastic waste, life-cycle assessment and circular economy of plastic with equity implications to propose interventions with mitigation strategies that are socially equitable, environmentally sustainable and economically viable to championing solutions addressing marine plastic pollution.

Macroplastics and Microplastics impacts on marine ecosystems

While large plastics (i.e., macroplastics with a length > 5mm) have been known to roam the oceans for a long time, microplastics are now considered a class of global pollutants of great concern exacerbating the threat of other ocean pollutants due to their persistent, hazardous and ubiquitous nature (Browne et al., 2007; Andrady, 2011; Rochman et al., 2013; Bergmann et al., 2017; Rochman et al., 2019). Microplastics are defined as particles with a size or length from 1 μm to below 5 mm and categorized as: (1) primary microplastics, which are deliberately manufactured, including microbeads in cosmetics, industrial cleaners, virgin resin pellets for manufacturing and nurdles; and, (2) secondary microplastics, which are break-down products of large plastics (macroplastics) that exceed 5 mm such as clothing, ropes, bags, and bottles and many single-use plastics (Andrady, 2011; Browne et al., 2007; Duis and Coors, 2016; Galgani et al., 2013; GESAMP, 2010; GESAMP, 2016; Hartmann et al., 2019; Moore, 2008). On the other hand, nanoplastics are particles defined with a size ranging from <1 to 1000nm (=1 μm), according to Gigault et al., (2018).

The contamination by both macroplastics and microplastics in our oceans is not just an aesthetic visual impact issue along shorelines, tourist beaches, open ocean environments and coastal regions, but tangible evidence of marine pollution in the Plasticene (Table 1). Ocean plastic emissions from the world's coastal cities (based on 192 coastal countries, equivalent to 93% of the world's population; see Jambeck et al. 2015) range from ~5 to ~13 million metric tonnes per year (average ~8.75 million tonnes). According to the United Nations Environment Programme, an estimated volume ranging from 75 to 199 million tonnes of plastic has accumulated in the ocean (UNEP, 2021a). While previous studies estimated that there are 5.25 to 50 trillion plastic particles floating in the global ocean, equivalent to 236,000 to 268,940 tonnes (Ericksen et al., 2014; van Sebille et al., 2015), new research has just revealed that the global abundance of small plastics floating in the ocean in 2019 was estimated to be around 82 to 358 trillion plastic particles, weighing 1.1–4.9 million tonnes, with a mean of ~170 trillion plastic particles, weighing 2.3 million tonnes (Ericksen et al., 2023). Similarly, a study assessing the distribution of seafloor microplastics found microplastics hotspots of up to 1.9 million pieces/m² (Kane et al., 2020). More recently, the total mass of microplastics in global seafloor sediments was estimated to reach 3.05 million tonnes (Harris et al., 2023). In terms of the class of plastic particles, microfibers are the most common microplastic type, accounting for >90% of observed plastic particles, distributed on regional seas and the global ocean (Barrows et al., 2018).

The origin, transport and final fate of most plastics is derived from human-made activities and land-based sources, including urban areas, household, and industrial water, as well as fisheries, aquaculture, maritime shipping, and tourism (Andrady, 2022; GESAMP, 2016; Jambeck et al., 2015; UNEP 2021a). The emissions of large plastic waste and microplastics from global rivers into the ocean are also major sources and pathways contributing to marine plastic pollution (Lambert et al., 2018; Lebreton et al. 2017). The long-range atmospheric transport of plastic particles is also a major mechanism of transport via air deposition in the oceans and coastal environment (Allen et al., 2022; Dris et al., 2016).

Recently, neopelagic communities have been recognized as ocean-floating plastic debris serving as habitats to support new sea surface (epipelagic) communities composed of coastal and oceanic species at sea which have significant ecological implications and cascading consequences of plastic pollution at the open ocean, causing changes in the marine environment (Haram et al. 2021)

Moreover, the impact of ocean plastics on the biological carbon flux pump through their influence on microbial cycling of carbon and nutrients in the ocean is of great concern (Galgani et al., 2022; Tetu et al., 2019). Plastics in fecal pellets may influence buoyancy, alter sinking rates and change the functioning of the biological pump, impacting the vertical carbon transfer from surface water to euphotic regions (Cole et al., 2022; Macali & Bergami, 2020). Galgani et al., (2022) found that the carbon contained in plastic may well represent up to 3.8% of the total downward flux of particulate organic carbon in tandem with key pathways regulating the transport of microplastics and potential interactions with the marine carbon cycling system, prompting microplastic removal via the “biological plastic pump”. Similarly, the most abundant photosynthetic bacteria (*Prochlorococcus*) in the ocean which is critical for oxygen production and ocean productivity can be impaired by leachate from high-density polyethylene bags and polyvinyl chloride matting, underscoring potential negative impacts to global primary production and carbon cycling (Tetu et al., 2019).

At present, the large-scale use of personal protection equipment (PPE) during the COVID-19 pandemic since early 2020 lead to arbitrary PPE littering including disposable facemasks, discharged gloves and empty sanitizer plastic bottles (see Box 1; Figure 1), which have become an inter- and intra-pandemic and post-pandemic plastic pollution issue of emerging concern (Canning-Clode et al., 2020; Klemeš et al., 2020; Silva et al., 2020; Silva et al., 2021; Zhang et al., 2021). The mismanagement of PPE during the COVID-19 pandemic, embracing a monthly estimated use of 129 billion face masks and 65 billion gloves at the global level, caused widespread environmental contamination (Prata et al., 2020). A recent study estimated an average of 8.4 million tonnes of COVID-19 pandemic-associated plastic generated from 193 countries, from which close to 26,000 tonnes were released into the global ocean (Peng et al., 2021). The emissions of discarded or misused PPE represented a looming public health risk as PPE waste is a potential physical vector for SARS-CoV-2 virus, surviving up to three days on plastic surfaces (Alava et al., 2022; Prata et al., 2020; Zhang et al., 2021).

As a result of unabated point and non-point sources of pollution along with continued production and emissions of plastics, the pervasive plastic footprint, including microplastic pollution, is drastically impacting the oceans’ biogeochemical cycling, and marine biodiversity with negative consequences for coastal human communities, heavily reliant on traditional seafoods.

Box 1: The COVID-19 pandemic and PPE plastic pollution

The COVID-19 pandemic prompted and caused the massive use and misuse of personal protection equipment (PPE), following current public health recommendations and requirements to prevent the spread of the virus since early 2020. While the severity of COVID-19 necessitated the use of protective equipment, poor PPE disposal and critical hazardous waste management issues due to the ongoing viral pandemic have been raised (Canning-Clode et al., 2020; Klemeš et al., 2020; Silva et al., 2021; Zhang et al., 2021). The emergence and increase in PPE usage and subsequent discharging and littering of discarded PPE supplies (personal, household and medical waste) such as facemasks, gloves, and empty sanitizer bottles not only exacerbated plastic pollution and disrupted plastics' life cycles (Canning-Clode et al., 2020; Klemeš et al., 2020), but the looming threat and public health risks of PPE-plastic debris as a source and physical vectors of contamination that can be potentially contaminated with SARS-CoV-2 (Alava et al., 2022; Canning-Clode et al., 2020; Prata et al., 2020; Zhang et al., 2021). While the plastics industry took advantage of the COVID-19 crisis and began spreading the inappropriate narrative that single-use plastics were safer than reusable plastics, there were insufficient reusable products to keep up with such high demand of PPE products (Prata et al., 2020; Silva et al., 2021). Unfortunately, many nations and cities around the world listened and rolled back plastic bans temporarily and progress on single-use plastic reduction policies was stalled or reversed in the face of pandemic uncertainty and industry seized this advantage (Silva et al., 2020). The mismanagement of pandemic-related wastes was responsible for causing “widespread environmental contamination” (Prata et al., 2020; Zhang et al., 2021). Thus, a surge and a plastic wave in the amount of PPE that has been littered in the environment are now emerging on and polluting the oceans and coastal areas. As an intra-pandemic plastic pollution problem at the global level, external contamination with these plastic types potentially carrying SARS-CoV-2 from infected people is possible and cannot be ruled out (Alava et al., 2022; Prata et al., 2020; Zhang et al., 2021). Thus, concerted ecotoxicological research fronts and biomedical monitoring to investigate the health risks by COVID-19 induced-plastic pollution are of paramount importance.



Figure 1. Discarded personal protection equipment (PPE) stemming from the massive demand, production, overuse and misuse of PPE-plastic waste and pollution during and after the COVID-19 pandemic: a) blue, medical facemask black (see white arrow) entangled in a white mangrove tree branches (*Laguncularia racemosa*); b) plastic (non-medical) facemask observed (see white arrow) on red mangrove leaves (*Rhizophora* sp.) in coastal waters of the Puerto Ayora, Santa Cruz Island, Galápagos (Ecuador); c) blue medical mask discharged on storm drain/sewer (BC, Canada), and; d) a pair of surgical blue gloves dumped close to a sidewalk (BC, Canada). Photo credits: (a) K. McMullen (Galápagos); (b) J.J. Alava (Galápagos) and (c, d) N. Alava (BC, Canada). Images and caption adapted from Alava et al. (2022).

Macroplastic and microplastic impacts on marine organisms

Many studies have attempted to isolate physiological, chemical, biological and behavioural effects related to the presence of plastics in marine biota, but the diverse nature of plastics and microplastic particles makes it challenging to generate all-encompassing implications, based on the weight of evidence (i.e., mode or mechanisms of toxic action) and best available data reported in the existing literature (Table 1). The impacts of marine plastic pollution became even more evident with the increasing reports of sea turtles, seabirds, cetaceans, sea lions and seals becoming entangled in marine debris (i.e., derelict and discarded fishing gear, ghost nets; see Macfadyen et al., 2009; Gilman, 2015) and interacting with or ingesting marine litter, sometimes causing deleterious injurious and/or death (Kühn and van Franeker 2020, Kruse et al., 2023; López-Martínez et al., 2021; Ryan, 2015).

In fact, seabirds, sea turtles and mammals received the most attention in the early days of marine litter literature (Kühn et al., 2015; Kühn & van Franeker, 2020; Lynch, 2018; Nelms et al., 2016; Roman et al., 2019; Ryan, 2015; Schuyler et al., 2014; Wilcox et al. 2015), but there is a growing subset of literature devoted to microplastic implications for invertebrates, including fish, crustaceans, molluscs, jellyfish, and more (Table 1).

That said, many studies have revealed adverse health effects resulting from macroplastics and microplastic ingestion and contamination. As shown in Table 1, internal lesions, digestive tract blockages, drowning, diminished predator avoidance, impaired feeding ability or falsified satiation, blockage of enzyme production, reduced growth rates, lowered steroid hormone levels, delayed ovulation and reproductive failure, and also absorption of toxic chemicals are a few of the related effects of microplastic intake (Athey et al., 2022; Cole et al., 2011; Everaert et al., 2018; Everseaert et al., 2020; Galgani et al., 2010; Wright et al., 2013; Ryan, 2015).

Despite compelling research identifying the adverse implications of microplastics in the marine ecosystem, some researchers remain skeptical regarding the magnitude of the problem. The skepticism is not surprising given that some studies have used a disproportionately high abundance of microplastic quantities compared to those found in nature (Rochman et al., 2019); likewise, some critics hesitate to quantify the level of threat microplastics pose given that many studies reveal non-fatal implications at an individual level. Certainly, biological implications of microplastic exposure assessed at an individual level are difficult to extrapolate to the entire population, however, ingested microplastic implications on behaviour, can provide clues to the potential impacts at a higher, ecosystem level (Galloway et al., 2017). Recent research found associations between microplastic ingestion and individual changes in predation patterns and weight gain, and related studies have revealed that social vertebrates may accept a greater risk to consume food if they are increasingly hungry (Galloway et al., 2017).

Additionally, zooplankton, which form an important trophic link for pelagic food webs, have exhibited changes in energy uptake when exposed to microplastics, which could lead to changes in energy uptake across the food web, not to mention, changes in the density of fecal pellets and thus changes in carbon transport to deep waters (Cole et al., 2022; Galloway et al., 2017; Macali & Bergami, 2020). When extrapolated to a species or ecosystem level, the aforementioned sublethal implications become increasingly more daunting (Galloway et al., 2017). The preliminary weight of evidence on neurotoxic effects by microplastics and nanoplastics in different species and in vitro suggests that exposure to plastic particles can induce oxidative stress, likely resulting in cellular damage and increased susceptibility to develop neuronal disorders (Prüst et al., 2020).

Microplastics are highly dynamic in nature (Rochman et al., 2019; Galloway et al., 2017); the ecotoxicology of microplastics depends on a variety of factors such as particle size, shape, crystallinity, surface chemistry, polymer and additive composition (Lambert et al., 2017). Therefore, it is necessary to differentiate microplastics in order to identify the potential hazards they pose (Lambert et al., 2017; Moore et al., 2020). For example, there is evidence that polypropylene fibers are more toxic than polypropylene beads commonly used in laboratory studies (Lambert et al., 2017). Additives designed to enhance durability also have the potential to leach from microplastics as they degrade in the ocean (Cole et al., 2011). Additionally, microplastic properties, shape and size for example, are constantly changing as they navigate through the water column, which in turn, constantly changes their potential impact.

The potential of microplastics fragmenting into nanoplastics (<1-1000nm) introduces an additional dynamic component in which the nanoplastics, with high surface area and higher ability to aggregate with other particles, can enter organism tissue and cells (Cole et al., 2011). For example, fin whales that ingested microplastics also had concentrations of phthalates in their blubber, which could be associated with the ingested microplastics (Fossi et al., 2014; Ivar do Sul & Costa, 2014).

Microplastics and nanoplastics may also collect layers of organic matter as they pass through organisms, through ingestion and subsequent egestion, building multiple layers of material. After recent years in which a pandemic spread at an unprecedented speed, the concern of inorganic material laced with biomolecules should be critically considered. Additionally, selectively binding chemicals have the potential to interfere with biological interaction, including mating and predator-prey behaviour (Galloway et al. 2017); more specifically, there is evidence to suggest plastic particles can pick up their own pheromone signals. The accumulation of substances on plastic particles as they navigate, likely changes the dispersion, density, sinking rate and bioavailability of microplastics (Galgani et al., 2022; Harris et al., 2023; Kooi et al., 2017).

Table 1. Summary of macroplastic and microplastic impacts and health effects in marine organisms and ecosystems

Ecological and Marine Biota Impacts	Source
Ocean plastics can function as physical vectors and floating platforms of invasive species or non-indigenous species arriving, impacting coastal environments, and affecting, competing with and eliminating endemic and native species in marine-coastal communities.	Galloway et al. (2017); Rochman et al. (2019); Haram et al., (2021)
Ocean plastics and microplastics can serve as physical vectors of biological pollution and pathogens, invasive species, as well as organic material (i.e., epiplastic microbial communities or Plastisphere as part of the Eco-corona), including pathogenic bacteria likely or potentially able to cause diseases or emerging infectious disease in humans and free-ranging marine animals/wildlife with implications for public and environmental health.	Alava et al., (2022); Debeljak et al. (2017); De Tender et al. (2015); Galloway et al. (2017); Kirstein et al. (2016); Harrison et al. (2014); McCormick et al. (2014); Oberbeckmann et al. (2015); Zettler et al. (2013)
Ocean plastics and microplastics are chemical vectors and carriers of adsorbed persistent organic pollutants (POPs) and other organic contaminants (e.g., Bisphenol A, plasticizers, hydrocarbons, poly- and perfluoralkyl substances or perfluorinated compounds [PFAS], emerging flame retardants), as well as metals (e.g., mercury) that can cause harm and health toxic effects in humans and marine organisms.	Athey et al. (2022); Bakir et al. (2016); Diepens & Koelmans, (2018); Galloway et al. (2017); Hartmann et al. (2017); Koelmans et al. (2013a,b); Koelmans et al. (2014a,b); Koelmans (2015); Rochman (2015)
Field and experimental studies demonstrate that marine biota is directly and indirectly affected by ingesting microplastics; and also by ingesting macroplastics (e.g., plastic bags, cups, bottles) with associated GI tract obstruction, affecting growth, feeding rates, nutrition and absorption of food, as well as deleterious and lethal entanglements, mutilations and asphyxiations with macroplastics, nylon, ropes and nets (discarded fishing gear and ghost nets), mainly impacting charismatic and high trophic level marine megafauna such as sea turtles, seabirds, and marine mammals. 'Plasticosis' as a new plastic-induced fibrotic disease has been discovered in stomach tissues of seabirds in 2023.	For example: Marine invertebrates (Botterrell et al., 2019; Bour et al., 2020; Everaert et al., 2018; Fulfer & Menden-Deuer, 2021; Everaert et al., 2020; Prüst et al., 2020; Tetu et al., 2019) Fish (Azevedo-Santos et al., 2019; Fossi et al., 2014; Jovanović, 2017; Prüst et al., 2020; Savoca et al., 2021) Sea turtles (Duncan et al., 2019; ; Kühn et al., 2015; Kühn & van Franenker, 2020; López-Martínez et al., 2021; Lynch, 2018; Matiddi et al., 2017; Nelms et al., 2016; Ryan, 2015; Schuyler et al., 2014; Yaghmour et al., 2021) Seabirds (Amelineau et al., 2016; Kühn et al., 2015; Kühn & van Franenker, 2020; Roman et al., 2019; Wilcox et al., 2015) Marine mammals (Besseling et al., 2015; Battaglia et al., 2020; Fossi et al., 2016; Fossi et al., 2014; Fossi et al., 2020; Kühn et al., 2015; Kühn & van Franenker, 2020; Kruse et al., 2023; López-Martínez et al., 2021; Lusher et al., 2015; Lusher et al. 2018; Moore et al., 2020; Nelms et al., 2018; Nelms et al., 2019; Ryan, 2015; Unger et al., 2016; Zantis et al., 2021)
Microplastics and nanoplastics can readily be ingested through direct ingestion of plastic particles per se in marine organisms, and wildlife. Bioconcentration, species-specific bioaccumulation or lack of bioaccumulation; and potential predicted biomagnification or no biomagnification can take place as a function of the specific passage time, retention time, and elimination rate of microplastics in species and individuals.	Alava et al. (2020); Covernton et al. (2022); Ma & You (2021); Miller et al. (2020); Miller et al. (2023); Hamilton et al. (2021); Provencher et al. (2019)
Depending on the size and shape of microplastics, the plastics particles can plausibly cause injuries and lesions on the GI tract and other tissues and organs, triggering immune response and inflammation in marine organisms and humans. 'Plasticosis' as a new plastic-induced fibrotic disease has been discovered in stomach tissues of seabirds in 2023.	Cole et al. (2011); Everaert et al. (2018); Everaert et al. (2020); Ryan, (2015); Wright et al., (2013); and 'Plasticosis' in seabirds (Charlton-Howard et al., 2023)

There is still a growing need to determine the ecotoxicological effects of marine species chronically exposed to microplastics in environmental concentrations (Everart et al., 2018; Nelms et al., 2019). Carpenter et al. (1972) suggested that plastics could be a source of some polychlorinated biphenyls (PCBs) that had been reported in marine organisms. The large surface area of microplastics suggest potential to

adsorb waterborne organic pollutants (Teuten et al., 2007). Persistent organic pollutants (POPs) reportedly adsorbed onto plastic have been detected in streaked shearwater (*Calonectris leucomelas*) and short-tailed shearwater (*Puffinus tenuirostris*) (Ivar do Sul & Costa, 2014). There is evidence that high concentrations of polychlorinated biphenyls (PCBs) on microplastics can be transferred to seabird tissues (Ivar do Sul & Costa, 2014). While some evidence suggests microplastics can act as vectors for toxic substances such as POPs, emerging chemicals of concern, and trace metals (Hartmann et al., 2017; Moore et al., 2020), there is evidence that this transfer only represents a small portion of toxic substances compared to those ingested via dietary exposure and food web trophic transfer (Bakir et al., 2016; Diepens & Koelmans, 2018; GESAMP, 2010; Koelmans et al., 2014; Koelmans, 2015; Table 1). Additionally, depending on the chemical concentration in the organism's tissues and ingested plastic, the plastic can either clean or contaminate the species by way of achieving equilibrium within the biota (Koelmans, 2015).

Litterbase, an online-interactive portal amalgamating marine litter literature (Bergmann et al., 2017; Tekman et al., 2023a; Tekman et al., 2023b), provides a reliable database summarizing and mapping (at present) >2,480 species interactions with ocean plastics/microplastics from >1,375 scientific studies on marine litter interactions with biota and 1,065 studies on marine litter distribution (Tekman et al., 2023a). Of literature on ingested plastic (macro- and microplastics) by marine organisms, the majority of studies has undoubtedly been focused in the Global North. Specifically, literature based in the South Pacific or South Atlantic accounts for only 17% of all assessed papers on ingested plastic, whereas literature based in the North Pacific or North Atlantic accounts for 50% of all assessed papers, while research in the Mediterranean contributed to 9% of the total (Tekman et al., 2023a).

A recent ecotoxicological risk assessment by Everaert et al. (2018) and Everaert et al. (2020) recently proposed a threshold for microplastic risk assessment. These studies suggest there will be direct effects from free-floating microplastic concentrations higher than the threshold of 6,650 or 7,990 particles/m³ of seawater (Everaert et al., 2018, 2020). Initially, the authors predicted concentrations of 9.6 to 48.8 particles/m³ by 2100 generally in the global ocean (Everaert et al., 2018), however, some oceanic zones already exceed these minimum safe limits (Everaert et al., 2018). In a more concerted follow-up study, Everaert et al., (2020) projected that 0.52% and 1.62% of the global ocean surface layer (0 to 5 m) will exceed the unacceptable threshold concentration risk of 1.21×10^5 microplastics per m³ by 2050 and 2100, respectively, under the worst-case scenario predicting future plastic discharge into the ocean. This study also predicted threshold levels between 32 and 144 particles/kg dry sediment in benthic areas, but suspect the range to exceed the safe limit of 540 particles/kg preceding 2050 (Everaert et al., 2018). The authors call for urgent studies on chronic microplastic exposure equivalent to environmental concentrations (Everaert et al., 2018; Everaert et al., 2020).

Ocean plastics impact in coastal communities and marginalized ethnic groups

As microplastic ecotoxicological research in marine ecosystems continues to grow, the threat to public health and human beings, especially those dependent on seafood consumption, still lingers (Macali & Bergami, 2020; Smith et al., 2018; Lusher et al., 2017; Revel et al., 2018; Rochman et al., 2015; Santillo et al., 2017). Of particular concern are the direct and indirect implications of microplastics on oceanic and coastal food webs with humans as the top predators and how microplastics are disproportionately distributed and affecting coastal communities around the global ocean.

Recent research has reported data estimates for the microplastic daily intake by humans or microplastic consumption rate per capita from seafood. A study including only 15% of a typical human diet projected that human beings may consume 74,000-121,000 microplastics annually (Cox et al., 2019). In Europe, for instance, humans are readily exposed to plastic particles via dietary ingestion of cultured bivalves, including blue mussels (*Mytilus edulis*) and Pacific/Japanese oysters (*Crassostrea gigas*), which contain in average around 0.3 and 0.5 microplastics per gram (wet weight) at the time of the consumption, respectively (Van Cauwenberghe & Janssen, 2014). More recently, Everaert et al. (2018) suggested that human beings may intake in average 864 microplastics per 2.4kg of blue mussels consumed in an annual basis, while for an average European, as avid consumers of mussels, they may ingest from 1,550 to 9,474 microplastics per 2.4kg of mussels consumed per year. Conversely, Domenech & Marcos (2021) estimated that the global per capita consumption of microplastics and nanoplastics via seafood is in the order of 22.04×10^3 plastic particles per year. These estimates underscore food safety and security implications for people with a diet rich in seafoods. As it stands, the direct consequences of marine litter are certainly not experienced in equal magnitudes across different ecosystems and different geographic locations.

The accumulation of nano- and microplastics in commercial fish, shellfish and crustaceans with the potential trophic transfer into the human food chain may compromise human health given the health risks by consuming microplastic-contaminated diet, drinks and seafood with associated toxic chemicals, especially impacting small-scale (artisanal) fishers, aquaculture industries, coastal communities, and Indigenous people, who strongly rely on traditional seafoods (Domenech & Marcos, 2021; O'Neill & Lawler, 2021; Kumar, 2018; Kumar et al., 2022; Lusher et al., 2017; Revel et al., 2018; Santillo et al., 2017; Smith et al., 2018; UNEP, 2021a; Walkinshaw et al., 2020). The potential toxic effects to human health by plastic particles in the short and long terms (acute and chronic toxicity, respectively) are of particular concern as microplastics and nanoplastics have already been detected in human blood (Leslie et al., 2022), placenta (Ragusa et al., 2021), breastmilk (Ragusa et al., 2022), colon (Ibrahim et al., 2021); liver (Horvatits et al., 2022), and lungs (Amato-Lourenço et al., 2021; Jenner et al., 2022).

Moreover, while the direct toxicity risks of plastic particle ingestion to human beings requires further study, the fate of nanoplastics and microplastics, their translocation and the mechanism of toxic action in human tissues and organs are now becoming more evident (Kumar et al., 2022; Ramsperger et al., 2023). The potential health risks for human carcinogenesis induction have also been suggested (Kumar et al., 2022; Revel et al., 2018). The weight of evidence also indicates that plastics' consumption may be markedly harmful to the reproductive health of women via immunotoxicity and endocrine disruption by associated toxic chemicals; there are also effects on the physical and mental health of exposed children, exhibiting immature defense mechanisms and making them particularly vulnerable (Kumar et al., 2022; O'Neill & Lawler, 2021; Landrigan et al., 2023; Rollin et al., 2022; Sripada et al., 2022). Thus, the pollution and exposure risks by microplastics are of utmost importance given the strong reliance of many communities on seafood for consumption, as aforementioned. According to the recent Report of the

Minderoo-Monaco Commission on Plastics and Human Health (Landrigan et al., 2023): “*Plastics cause disease, impairment, and premature mortality at every stage of their life cycle, with the health repercussions disproportionately affecting vulnerable, low-income, minority communities, particularly children.*”

Table 2. Summary of macroplastic, microplastic and nanoplastics impacts on public or human health of coastal communities

Indirect and Direct Impacts: public/human health risk, wellbeing implications and inequities	Source
Microplastics and nanoplastics are potentially accumulated through dietary consumption of contaminated food items including seafoods from fisheries and aquaculture, magnified by trophic transfer in marine fauna and coastal marine food webs, particularly impacting human communities that are heavily dependent on seafood. The net bioaccumulation of microplastics depends on the specific retention time and elimination rate of microplastics in species and individuals.	Alava (2020); Cox et al. (2019); Domenech & Marcos (2021); Galloway, (2015); Lusher et al. (2017); O'Neill & Lawler, (2021); Revel et al. (2018); Rochman et al. (2015); Santillo et al. (2017); Smith et al. (2018); van Cauwenberghe & Janssen (2014);
While questions still remain on the health effects and toxicity risks of plastic particles in humans, micro- and/or nanoplastics have been already detected in human placenta, blood, breast milk, colon, and lungs. These findings indicate the capacity of plastic particles to translocate to and accumulate in human tissues or organs and suggest that coastal communities heavily reliant on seafoods are at a larger risk of exposure than other groups.	Blood (Leslie et al., 2022); placenta (Ragusa et al., 2021); breastmilk (Ragusa et al., 2022); colon (Ibrahim et al., 2021); liver (Horvatits et al., 2022), and lungs (Amato-Lourenço et al., 2021; Jenner et al., 2022). Fate and translocation to human tissues/organs (Ramsperger et al., 2023); and potential carcinogenesis (Kumar et al., 2022); diseases and mortality (Landrigan et al., 2023)
Transferring of POPs from ingested microplastics is generally small relative to POPs readily bioaccumulated from dietary uptake and GI tract magnification and absorption, but still may be of concern due to varying exposure risks and sensitivity in the most exposed human populations.	Ivar do Sul & Costa, (2014); Koelmans et al. (2014); Koelmans (2015); Moore et al. (2020); Walkinshaw et al. (2020); Landrigan et al., (2023)
Indigenous/First Nation peoples from small island nations and continental coastal communities strongly relying on seafood can inequitably be most exposed to the impacts by ocean plastics and microplastic by being in direct contact with macroplastics (including associated pathogen transmission) and by accidentally or indirectly intaking microplastics through dietary ingestion of seafood.	Alava et al. (2022); Bennett et al. (2022); (Landrigan et al., 2023); Macali & Bergami, (2020); Rochman et al. (2015); Smith et al. (2018); UNEP (2021b)
Women and children are particularly exposed to potential negative health effects by microplastics and nanoplastics, affecting reproduction, physical and mental health. Children with a developing and immature immune system are especially susceptible to the exposure to plastic particles and associated toxic chemicals or additives, as well as being at particularly high risk of plastic-related health effects, diseases impairment, and premature mortality at every stage of their early life and young age. The health repercussions disproportionately affect vulnerable, low-income, minority communities, and children.	Kumar et al. (2022); Landrigan et al. (2023); O'Neill & Lawler, (2021); Rollin et al. (2022); Sripada et al. (2022)
The socio-economy, equity and wellbeing dimensions and human health of coastal communities, including Indigenous peoples, as well as minorities from underrepresented and marginalized groups that relying significantly on artisanal fisheries, traditional seafoods, commercial fishing and ecotourism (e.g., plastic – contaminated beaches and charismatic marine life are affected by ocean plastics.) are indirectly impacted by the pervasive presence and unequal distribution burdens of plastic pollution along coastal areas and oceanic islands.	Chami et al., (2020); Bennett et al. (2022); Liboiron, (2021); Vandenberg & Ota (2022); UNEP (2021b)

Ultimately, it is important to highlight how the threat of macroplastics and microplastics is unequally felt at the human level and in complex socio-ecological systems. The people living in oceanic, remote and

continental coast areas, mainly ancestral, Indigenous and marginalized native communities, from developed and developing countries, have common and unique health issues in the face of pervasive ocean pollution by marine plastic and microplastics (Simon et al., 2021; Vandenberg & Ota, 2022; UNEP 2021b). The marine plastic pollution problem is exacerbated by lack of equity, wellbeing and environmental justice due to inequality gaps dismissing equitable interventions and fair access to basic resources (e.g., effective solid waste management, appropriate sanitation and hygiene levels, primary education and public health programs) in the most affected and exposed communities, as well as overburden minority groups (Bennett et al., 2022; McMullen, 2022; Vandenberg & Ota, 2022). Prompted by global capitalism expansion, the accelerate rate of plastic production with consequential marine plastic pollution have evolved to become a form of colonialism, affecting marginalized groups and Indigenous people (Liboiron, 2021; UNEP 2021b).

Towards foodweb bioaccumulation modeling for microplastics

Microplastic bioaccumulation modeling in cetacean foodwebs of the Northeastern Pacific: A regional exercise case

Bioaccumulation and biomagnification of nanoplastics and microplastics are key processes that has yet to be assessed and comprehended within the context of bioaccumulation science for these emerging pollutants of concern. While bioaccumulation frameworks for screening and assessing the bioaccumulation and biomagnification potential of bioaccumulative substances (e.g., Persistent Organic Pollutants-POPs) is fairly well understood under the application of bioaccumulation criteria and metrics (Gobas et al., 2009); concerted bioconcentration, and bioaccumulation assessments of microplastics, addressing exposure, trophic transfer, ingestion and elimination/egestion rates in marine biota and thus plastic biomagnification potential in foodwebs is strongly needed as part of the ecotoxicological and human risk assessments (Alava, 2020; Burns and Boxall 2018; Covernton et al., 2022; Gouin et al., 2019; Hamilton et al., 2021; Miller et al., 2020; Miller et al., 2023; Provencher et al. 2019). Within this rationale, the appropriate definition in tandem with the application of bioaccumulation modelling, indexes or metrics to advance the microplastic bioaccumulation science is critical.

Microplastics can readily be ingested by marine organisms by direct ingestion; and, indirectly, through trophic transfer, which is likely to be the main pathway for microplastics to bioaccumulate in top predators (Alava, 2020; Miller et al., 2020; Nelms et al., 2018; Nelms et al., 2019; Provencher et al., 2019). Specifically, considering the foraging behavior of marine mammal species and exposure to plastic particles (see Figure 2), microplastics may be ingested directly through incidental consumption, including feeding by pelagic and deep diving cetaceans and indiscriminate filter-feeding of small particles from large volumes of water by baleen (mysticete) whales (Germanov et al., 2018; Kahane-Rapport et al., 2022; Besseling et al., 2015; Lusher et al., 2015; Lusher et al., 2018; Nelms et al., 2019; Simmonds, 2012). Microplastics with a size of 1mm, for example, were found in the gastrointestinal tract of a stranded humpback whale (*Megaptera novaeangliae*), indicating unselective ingestion by this baleen whale species (Besseling et al., 2015). Microplastics can also be indirectly ingested through trophic transfer, in which predators consume microplastics-contaminated prey items (Battaglia et al., 2020; Hernandez-Gonzalez et al., 2018; Lusher, 2015; Nelms et al, 2019). For example, during raptorial foraging behavior performed by most pinnipeds and dolphins (Hocking et al., 2017), or via bottom feeding and suction feeding on benthic organisms contaminated with microplastics from sediments (e.g., walruses, belugas and gray whales). Incidental consumption of contaminated sediment may be considered direct exposure, as well.

A recent study predicted that krill-feeding whales like the blue whale (*Balaenoptera musculus*) are more exposed to microplastics ingestion compared to fish-feeding whales, like the humpback whales (*M. novaeangliae*) (Kahane-Rapport et al. 2022). In a daily basis, for instance, a blue whale foraging on krill may well ingest 10 million microplastics, while a fish-eating humpback whale likely ingests 200,000 microplastics (Kahane-Rapport et al., 2022). Additionally, the ingestion of plastic debris and possibly microplastics as a result of stranding event processes in cetaceans cannot be ruled out (Simmonds, 2012; Unger et al., 2016). Inhalation of air-borne particles, fibers or aerosols at the water-air interface in the oceanic atmosphere is also a potential route of exposure for air-breathing organisms such as marine mammals (Lusher, 2015).

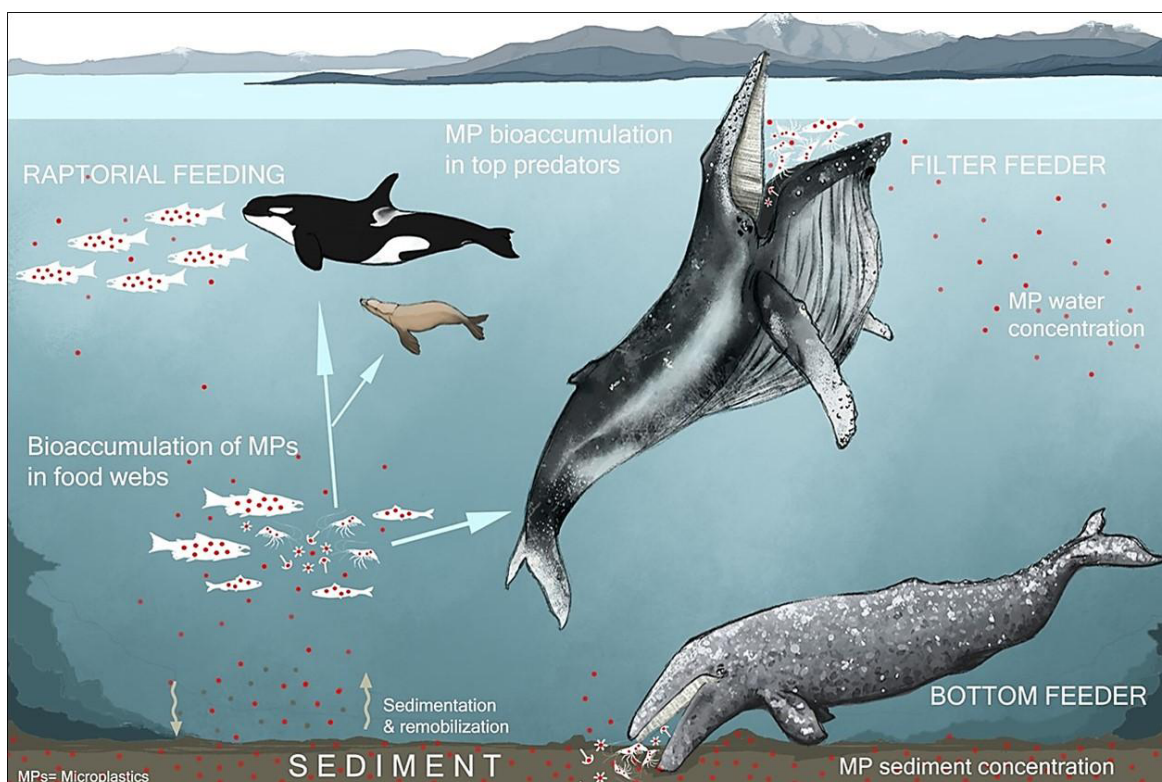


Figure 2. Illustration of pathways for bioaccumulation of microplastics (MPs) in marine mammalian foodwebs, indicating the feeding preferences and foraging strategies in marine mammals (e.g., fish-eating killer whales, pinnipeds, filter-feeding humpback whales and bottom-feeding grey whales) and potential microplastic exposure via prey (zooplankton/krill, benthic crustaceans and fish), water and sediments (Conceptual Designing: Dr. Juan José Alava; Artwork by Nastenka Alava). Learn more about Microplastic bioaccumulation/biomagnification at the following microplastic bioaccumulation video Link: <https://vimeo.com/452051237>

Because bioaccumulation and biomagnification of microplastics in marine mammalian foodwebs is scarcely known, a bioaccumulation model for microplastics was developed for the fish-eating resident killer whale (i.e. Chinook salmon-southern resident killer whale foodweb) and the filter-feeding humpback whale (i.e., zooplankton-Pacific herring-humpback whale food web) of the Northeastern Pacific (see Alava, 2020) with the aim to preliminarily explore whether microplastics bioaccumulate in food webs and marine mammals (Figure 3), with implications for the food safety and public health of coastal communities heavily reliant on fish or traditional seafoods for commercial use and subsistence. Here, we synthesize the fundamental principles of the model application and main findings resulting from the foodweb bioaccumulation modelling work based on Alava's model (Alava, 2020).

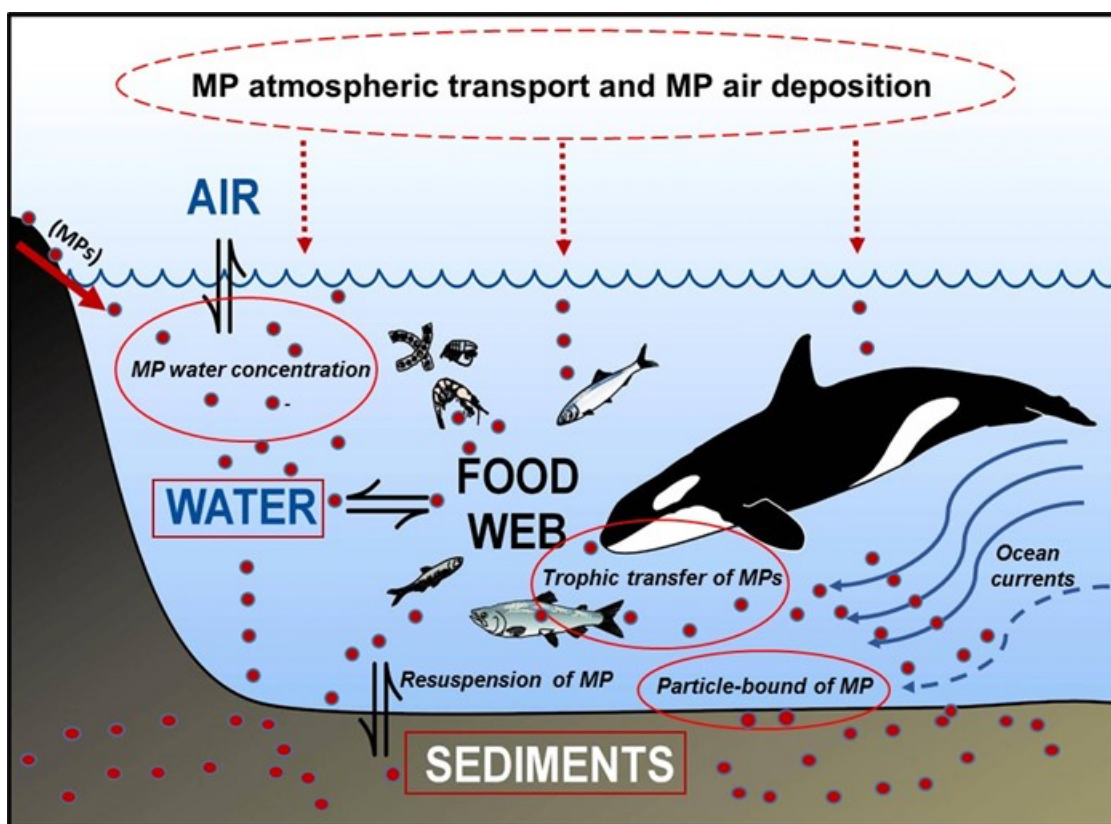


Figure 3. Environmental fate and transport of microplastics in the coastal-marine and ocean environments including exposure pathways that facilitate plausible bioaccumulation of microplastics in marine organisms and potential biomagnification in foodwebs. Microplastic partitioning and distribution among abiotic compartments (i.e., air, water, and sediments) in the marine environment provide routes of exposure as sources (water) and sinks (sediments), as well as long-range atmospheric transport and deposition of microplastics for marine biota; hypothetically, plastic particles can undergo bioaccumulation and biomagnification in biota and foodwebs, depending on the capacity to ingest or eliminate these contaminants from the organism as a function of the dietary intake/ingestion rate, passage time, retention or residence time and elimination rate from the gastrointestinal (GI) tract.

Model rationale

This primary research front introduces and presents the development of Alava's pragmatic kinetic bioaccumulation model for microplastics, using a known marine food web of the Northeastern Pacific (i.e., Chinook salmon-southern resident killer whale). It simulates and predicts the accumulation potential of microplastics as a practical exercise and product with the aim to understand the bioaccumulation and biomagnification behaviour of microplastics (Alava, 2020). The developed model serves as a prospective tool to be adapted and applied for other food webs, involving fish, seabirds, marine mammals, and humans as apex predators to support ecotoxicological and human risk assessment and decision-making.

The rationale and principles of the food web bioaccumulation model consist in a modelling framework integrating fundamental concepts of bioaccumulation science, as well as the best data available and reliable information from the existing literature on this subject by considering the main kinetic mechanisms for bioaccumulation with associated variables and parameters, summarized as follows:

- Microplastic concentration data in environmental compartments from the peer-review literature published for the study area (marine region of British Columbia, including offshore and nearshore waters of the west and east coast of Vancouver Island, respectively; and coastal waters of the Strait of Georgia) were used as input data for abiotic matrices: seawater (Desforges et al., 2014; Collicutt et al.,

2019) and sediments (Kazmiruk et al., 2018; Collicutt et al., 2019) to set up low, moderate and high exposure concentration scenarios (see Alava, 2020; Appendix I).

- Dietary preferences (% diet matrix) and trophic levels of food web organisms or species were based on previous studies using information for the southern resident killer whale's foodweb and diet compositions (Alava et al., 2012; Alava and Gobas, 2016; Alava et al., 2018)
- Dietary uptake/intake rate constant (k_D) of species (trophic level) was retrieved from the existing literature (see Alava, 2020)
- Calculated egestion/elimination rate constant (k_E) from documented data for the retention time (t_r) of microplastics in the digestive system or GI tract in marine animal species ($t_r = 1/k_E$; then, $k_E = 1/t_r$) were used as described in Alava (2020).

The model (Alava, 2020) was developed as a practical tool in Microsoft Excel that can be adapted and applied for any marine and coastal food webs without the need of conducting the field sampling of all the species in the food web. The complete description of the model's theory and kinetic bioaccumulation model with the food web bioaccumulation modeling scenarios are freely available (open access) in Alava (2020). A flowchart diagram illustrating the conceptual model framework of the modeling components and mechanism is portrayed in Figure 4.

Based on this approach, the model equations are described in Alava (2020) under the principles of kinetic processes that can be combined in a basic toxicokinetics model for the bioaccumulation of microplastic in aquatic or marine biota, using a mass balance of uptake and loss rates, expressed by the following differential equation for water respiring organisms:

$$dC_B/dt = k_W \cdot C_W + k_D \cdot C_D - (k_G + k_E) \cdot C_B$$

For air-respiring organisms, the differential equation is:

$$dC_B/dt = k_A \cdot C_A + k_D \cdot C_D - (k_G + k_E) \cdot C_B$$

where C_B is the microplastic concentration in the organism (in particles/organisms or in g/kg); k_W is the rate constant for microplastic uptake from water (particles/organism/d or g/kg/d); C_W is the microplastic concentration (particles/L or g/L) in the water; k_A is the rate constant for microplastic uptake from air (particles/organism/d or g/kg/d); C_A is the microplastic concentration in the air (particles/L or g/L); C_D is microplastic concentration in the diet of the organism (particle/organism or g/kg), which can also include sediment concentrations (C_S) of microplastic (in particle/kg dry weight [dw] or g/kg dw) for detritophages; k_D is the dietary uptake rate (kg/kg/d); k_G is the growth dilution rate constant, and k_E is the elimination rate constant (in units of $d^{-1} = 1/d$) of microplastic in organisms.

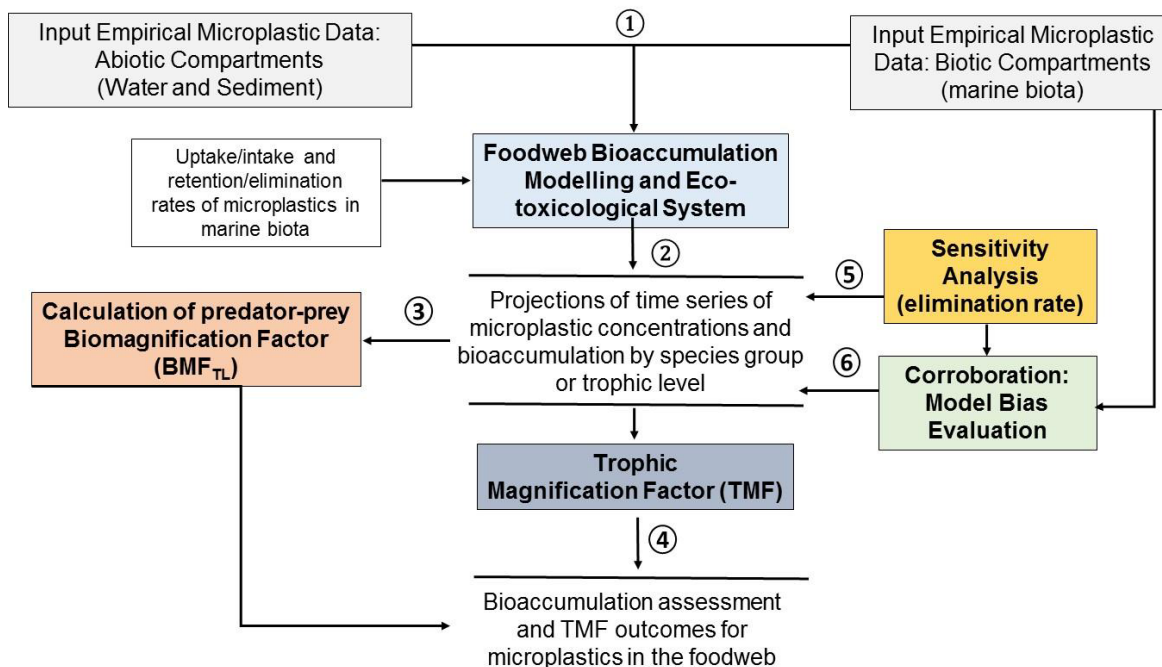


Figure 4. Conceptual model framework illustrating the basic relationships of modelling components for the prediction and assessment of the microplastic levels and bioaccumulation potential in the marine food web. (1) The empirical data for microplastics (i.e., documented empirical concentrations measured in water and sediments from the studied region under low, moderate and high concentration scenarios; Appendix I) is a key input for the modelling work to predict concentrations in a particular species or functional groups of species in the marine food webs over time to simulate the bioaccumulation potential in the food web; (2) Then, following the inclusion of observed microplastic data in water and sediment, the food web model is run to predict and simulate the concentrations in each species or functional group to produce the projections of time series in the food web; (3) The predicted microplastic data in marine biota is used as input data to estimate the predator-prey magnification factor (BMF_{TL}) aimed to analyze preliminary biomagnification potential in predators relative to the predicted microplastic data in preys; (4) Following the application of BMF_{TL} , the contaminant data generated from it also used to compute trophic magnification factors (TMF) for microplastics to further explore the potential magnification of microplastics at each trophic level in the marine food web; (5) A sensitivity analysis is performed on the elimination rate (k_E) of microplastics in a given organisms and a fundamental trophic level or a functional group (e.g., zooplankton) to assess how sensitive is this parameter and the model to changes in the parameter values; and, (6) To corroborate the projections resulting from the simulations, a model bias (MB) evaluation approach is required to assess the performance of the food web bioaccumulation model (i.e. whether the model is reproducing fairly well concentration values for microplastics similar to those observed from the empirical contaminant data available). Conceptual framework flowchart diagram based on Alava (2020).

Bioaccumulation metrics

Description of the rationale and calculation of bioaccumulation/biomagnification metrics used in bioaccumulation science and applied to this modelling work, including the predator-prey biomagnification factor (BMF_{TL}) and trophic magnification factor (TMF), as indexes to assess biomagnification, are described as follows:

Predator-prey biomagnification factor (BMF_{TL})

To investigate biomagnification in predators (e.g., marine mammals) relative to prey items and to assess the effect of the magnitude of trophic level differences on this biomagnification index, the predator-prey biomagnification factor (BMF_{TL}) was applied as a practical tool to assess preliminary biomagnification potential of microplastics in predators (Alava 2020). The criterion applied to indicate the capacity of microplastics to biomagnify was a $BMF_{TL} > 1$, while a $BMF_{TL} < 1$ is an indication of lack of biomagnification capacity (see Alava, 2020). If the BMF is statistically greater than 1, then it indicates that a contaminant is a probable bioaccumulative contaminant (Gobas et al., 2009). Following this approach, the microplastic concentrations projected in selected predators were divided by predicted concentration in the prey. Thus, the model-based predator-prey biomagnification factor normalized to trophic position (i.e., $BMF_{TROPIC\ LEVEL} : BMF_{TL}$) is calculated using the following equation (Borga et al., 2004):

$$BMF_{TL} = \frac{(C_{PREDATOR} / C_{PREY})}{TL_{PREDATOR} - TL_{PREY}}$$

Where $C_{PREDATOR}$ and C_{PREY} are the MP concentrations in the predator and prey, expressed in units of mass (g per kg of predator) and the concentration in prey (g per kg of prey); and, $TL_{PREDATOR}$ and TL_{PREY} are the trophic levels of the predator and prey. The BMF_{TL} values were used to measure biomagnification in the cetaceans' food chain between two adjacent trophic levels (i.e., the difference in TL between predator and prey is small), assuming steady state in MP concentrations between predator and prey, as reported in Alava (2020).

The trophic magnification factor (TMF)

The trophic magnification factor (TMF) is a food web biomagnification metric that is often used to investigate the biomagnification of pollutants at each trophic level in an entire food web (Borga et al., 2012; Conder et al., 2012; Walters et al., 2016). This approach was applied to further assess the microplastic biomagnification potential in the entire marine foodweb (Alava 2020). The TMF is calculated as the antilog of the regression slope of the linear regression between the logarithmic-transformed concentrations of microplastics (Log MPs) predicted in the GI tract of organisms of the food web and their trophic level, TL (Alava, 2020), i.e., $\text{Log [MP]} = a + bTL$, which in the equivalent exponential mathematical terms is expressed as $TMF = 10^b$, where b is the slope.

The TMF (slope, b) is statistically evaluated using a significance level (α) of 0.05. A $TMF > 1$ ($b > 0$) indicates that the contaminant biomagnifies in the foodweb. A $TMF < 1$ ($b < 0$) indicates trophic dilution of the contaminants, while a $TMF=1$ ($b = 0$) indicates no change in contaminant concentrations among organisms of a food web (Borga et al., 2012).

Sensitivity analysis, and model bias

The formulation of scenarios, definition of sensitivity analysis and model bias for the foodweb bioaccumulation model are briefly described in the Appendix I (see also Alava, 2020; and Alava, 2021).

Limitations and uncertainty

The main limitation and uncertainty are related to the availability of empirical data to test the model bias and the availability of published values for elimination rates of microplastics in the food web species or functional groups included. However, the model contains default or universal rates for many species of the food web and can pragmatically be applied to other species, based on the best available data and the existing published literature (see Alava, 2020). Thus, there is no need to concurrently conduct additional experiments in the lab for elimination rates, if the datasets are not readily available. The application of this modelling tool is critical when there are limitations on time, logistics, finances, and ethics to collect species or biota samples (e.g., small population of endemic species, threatened or endangered species).

Model outcomes

The key findings from the bioaccumulation and biomagnification modelling are described as follows (for more information on results see Alava, 2020):

- Bioaccumulation and biomagnification of microplastics can be a species-specific or foodweb-specific process, which is dictated by the microplastic retention time (t_r) and elimination rate (k_E) of marine organisms, as shown in Figure 5.
- Compared to the high bioaccumulation behaviour of persistent organic pollutants (POPs) in marine food webs, microplastics appears to show low bioaccumulation potential in cetaceans with specific foraging behaviour: fish-eating resident killer whale (raptorial feeding) versus Chinook salmon; and filter-feeding humpback whales versus Pacific herring (Figure 5).
- Scarce biomagnification capacity of microplastic was predicted in the cetacean food webs, depending on the magnitude of abiotic concentrations (low or high observed water and sediment concentration data for microplastics) used in the modelling work (Figure 6).
- The moderate to high microplastic bioaccumulation predicted in some lower trophic level marine organisms highlights the health risks of toxic exposure to marine fauna and coastal communities that strongly rely on seafood.
- The developed model provides a tool to assess the bioaccumulation potential and impact of microplastics in the marine environment to support risk assessment and inform plastic waste management. Doing so, the application of the model can be used to recommend and set up water and sediment quality guidelines for microplastics.

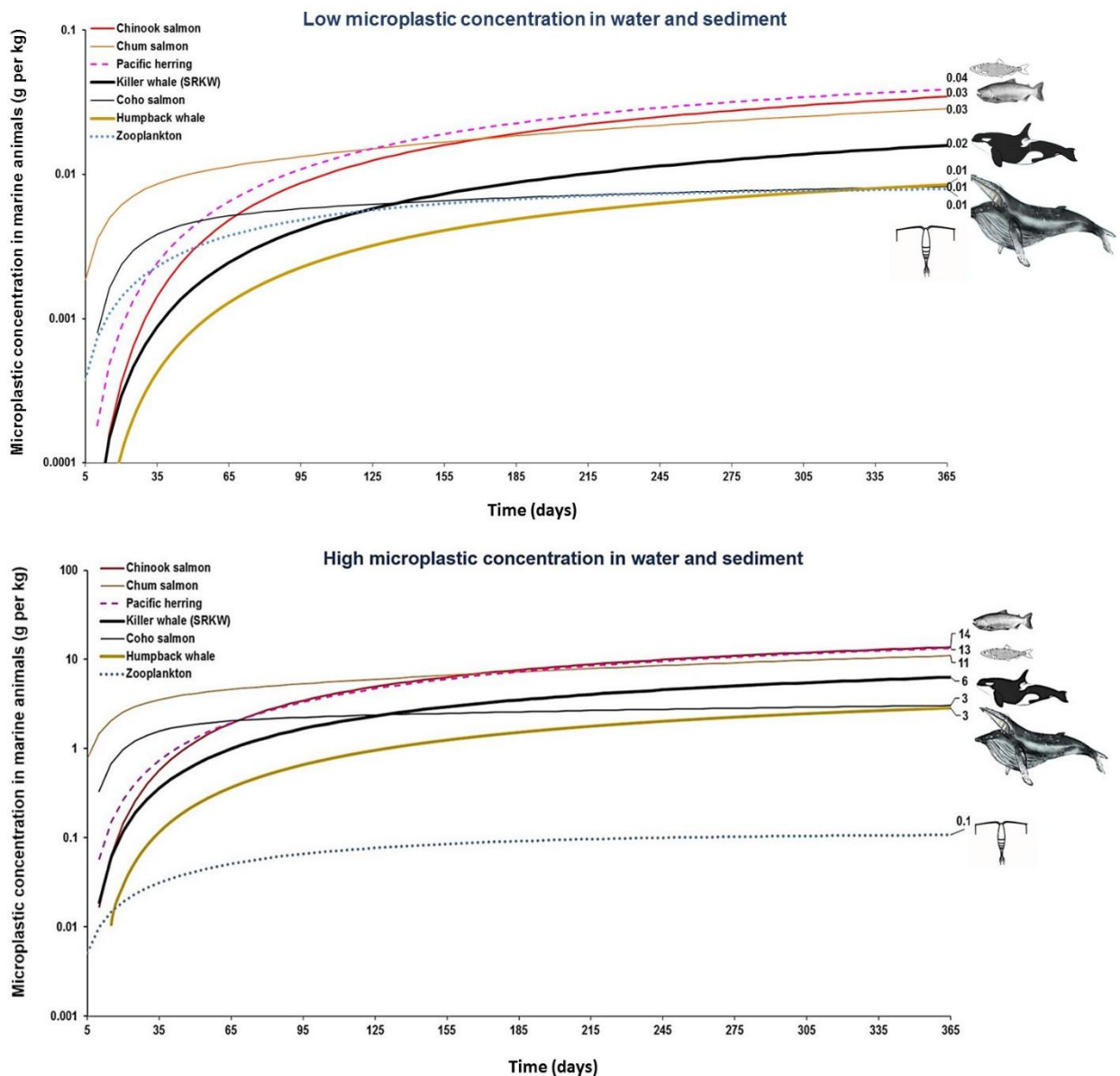


Figure 5. Simulations of the bioaccumulation model, showing the projections of microplastics (MPs) bioaccumulation in the cetaceans' food webs (zooplankton-Pacific herring-filter feeding humpback whale; and, Pacific herring-Chinook salmon-fish eating killer whale) under low concentration (above): seawater = [0.003 g/L]; and, sediment = [0.266 g/kg dw]; and a high concentration (bottom): seawater = [0.04 g/L]; and, sediment = [111 g/kg dw]. The simulations for bioaccumulation include the elimination rates and growth dilution for most organisms based on the literature reported in Alava (2020). For zooplankton, the key trophic level for the initial uptake of microplastics, a $k_E = 0.143/\text{d}$ (i.e., retention time = 7 days) was used as a least conservative scenario, based on the study by Cole et al. (2013). Figures adapted from Alava (2020).

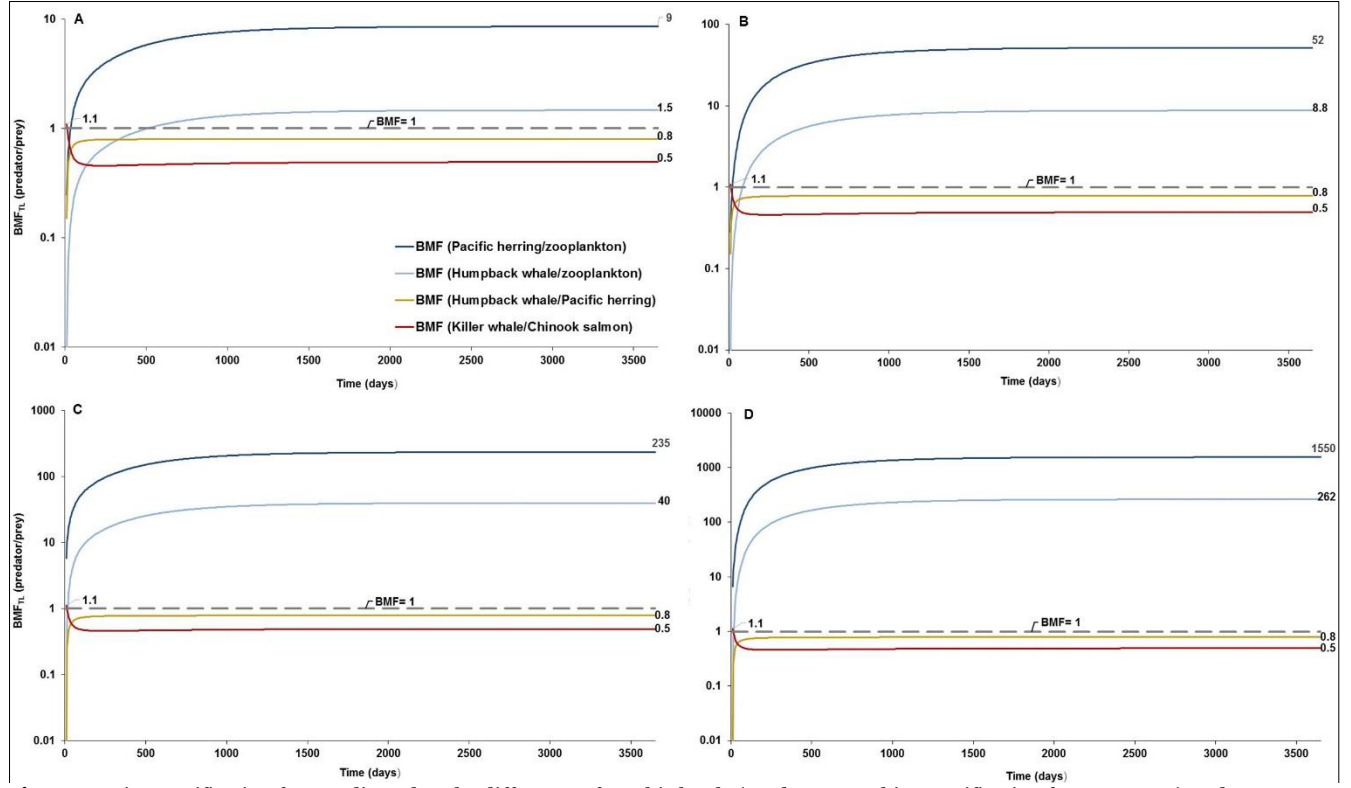


Figure 6. Biomagnification factor adjusted to the difference of trophic levels (predator-prey biomagnification factor: BMF_{TL}) under the testing of two elimination rates, based on data from Cole et al. (2013): (A) BMF_{TL} simulation with a low elimination rate of $k_E = 0.143/\text{d}$ at a low concentration scenario in seawater and sediment (0.003 g/L ; and 0.266 g/kg dw , respectively) as a conservative scenario; (B) BMF_{TL} simulation with a high elimination rate of $k_E = 1/\text{d}$ at low concentration scenario as a least conservative scenario; (C) BMF_{TL} simulation with a $k_E = 0.143/\text{d}$ at a high concentration scenario in water and sediment (0.04 g/L ; and 111 g/kg dw , respectively) as a conservative scenario; and (D) BMF_{TL} simulation with a $k_E = 1/\text{d}$ at a high concentration scenario, as a least conservative scenario. Dashed line represents equal distribution of MP concentrations ($BMF_{TL} = 1$) between predator and prey. Figures adapted from Alava (2020).

Ecosystem modelling (EwE models) to explore and track microplastic bioaccumulation in foodwebs in the Global Ocean

Foodweb ecosystem and bioaccumulation models: A global overview of marine bioaccumulation of microplastics

To further support the development of the kinetic food web bioaccumulation model for microplastics (Alava, 2020), this component embraced the application of food web-trophic dynamic ecosystem modeling using Ecopath with Ecosim (EwE) models along with the Ecotracer routine (Christensen & Walters, 2004) as a practical exercise to further understand microplastic bioaccumulation. The application of EwE ecosystem modelling to track the food web bioaccumulation of microplastics is critical as very few models have been used to test the performance and reliability of the Ecotracer routine to simulate the bioaccumulation potential of microplastics in aquatic food webs. For example, EwE ecosystem models with Ecotracer to predict microplastic bioaccumulation have been developed for the Baiyangdian Lake ecosystem in China; Ma & Yu, 2021) and for the Galapagos Island ecosystems, i.e., the Galapagos penguin food web model (McMullen, 2023). The theory and rationale defining the principles of the EwE model and Ecotracer module can be found in Appendix II.

In this context, this ecosystem modelling application was aimed to track and simulate the bioaccumulation potential in 23 geographical marine areas and ecosystems of the global ocean. The modeling exercise was conducted using the field observed or estimated ocean surface water concentrations of microplastics/nanoplastics measured in selected marine-coastal regions of the global ocean available in Litterbase (<http://Litterbase.org>; Bergmann et al., 2017; Tekman et al., 2023b). The distribution of the microplastic water concentration systematically reported in the Litterbase data (<https://Litterbase.awi.de/litter>) and the Global Microplastics Initiative dataset were then entered as input data (i.e., environmental concentration) in the Ecotracer module of existing EwE models (Appendix III, Table A1).

The suit of EwE models was retrieved from the Ecobase database (Colléter et al., 2013; Colléter et al., 2015; <http://ecobase.ecopath.org/> or <http://sirs.agrocampus-ouest.fr/EcoBase/>), which represent selected geographical marine regions that match and/or overlap with the distribution of microplastic water concentration pinpointed in the global ocean maps of Litterbase and the Global Microplastics Initiative datasets, as illustrated in Figure 7. The Ecotracer module of these models was run using microplastic concentration data (entered as the environmental compartment concentration in Ecotracer) and documented microplastics' elimination rates in marine biota adopted from the foodweb bioaccumulation model developed by Alava (2020).

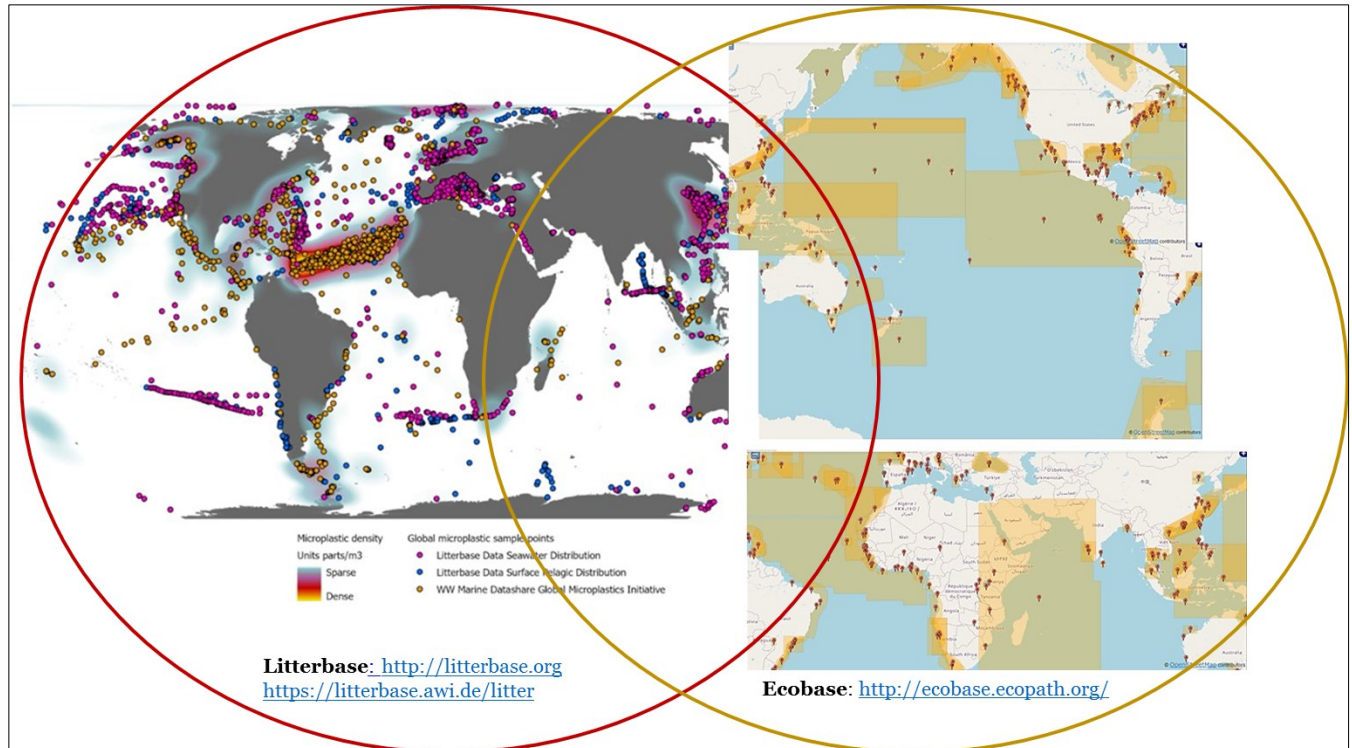


Figure 7. Conceptual illustration of the combined modelling approach to explore and track microplastic bioaccumulation in foodwebs in the global ocean using the field observed or estimated ocean surface water concentration data of microplastics/nanoplastics measured in selected marine-coastal regions of the global ocean available in Litterbase (<http://Litterbase.org>; Bergmann et al., 2017; Tekman et al., 2023b) used as input data for selected EwE ecosystem models retrieved from Ecobase database (<http://ecobase.ecopath.org/>; Colléter et al., 2013; Colléter et al., 2015) for marine ecosystems matching or overlapping marine regions exposed to microplastic pollution in the global ocean.

Outputs of the EwE ecosystem modelling

A total of twenty-one EwE models representing marine and coastal regions of the global oceans were retrieved from the Ecobase database (Appendix II). The data analysis of the EwE models' simulations and outputs (see Figure 8) along with the calculation of trophic magnification factors (*TMFs*) used as an index of biomagnification, for the marine ecosystems and regions are shown Table 3 and Figures 9 and 10.

The overall simulation outcomes resulting from the EwE modelling exercise predicted the plausible bioaccumulation and biomagnification of microplastics in the biomass of species and/or species' functional groups composing the assessed marine foodwebs (Table 3; Figures 9 and 10). The relationship between the concentration of microplastics (as logarithmic transformed data) and the species' trophic levels shows that the *TMF* was significantly greater than 1 (i.e., $TMF > 1$; when the slope is statistically different or greater than zero in a positive, significant linear regression) in twenty food webs of the marine regions, highlighting the potential biomagnification of microplastics in these specific food webs.

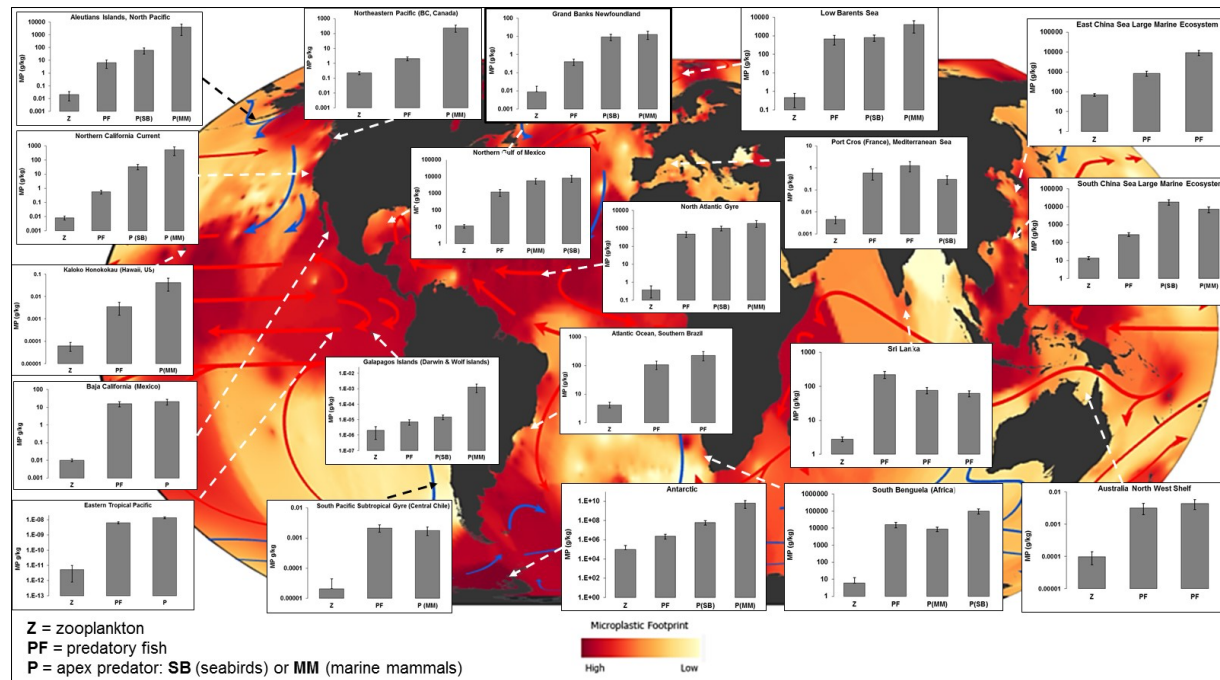


Figure 8. Examples of the projections of microplastic accumulation, based in water concentration data from Litterbase, in zooplankton (Z), predatory fish (PF), and top predators, including seabirds (SB) and marine mammals (MM) predicted by the Ecotracer module of EwE ecosystem models developed in different marine regions of the world's oceans. The background map represents the geospatial analyses and interpolation (IDW layer) of the distribution of microplastic concentration (in g/m³) in the global ocean to portray a preliminary global ocean exposure-risk map (please, see next section on 'Global Ocean exposure-risk index for microplastics pollution footprint'). It illustrates the microplastic footprint (see the scale in colors) based on the database Litterbase (<https://Litterbase.awi.de/litte>; Bergmann et al., 2017; Tekman et al., 2023b) and the Global Microplastics Initiative dataset (<https://www.adventurescientists.org/microplastics.html>; Christiansen, 2018); the blue and red arrows represent major ocean currents (NOAA National Weather Service and the US Army).

The highest *TMF* value was predicted in the Antarctica Peninsula ecosystem (*TMF* = 55; Figure 9a), followed by those projected in marine ecosystems and regions of the Pacific Ocean (i.e., *TMF* values ranging from 10 to 30; Table 2, Figure 9b-9g) which were also among the highest. These Pacific regions include the Northern California Current, Northeastern Pacific, Baja California (Mexico), Floreana Island's rocky reef ecosystem in the Galapagos Islands, Australian Northwest Shelf and South Pacific subtropical Gyre (waters off the Chilean central coast). Moderate *TMF* values were projected for the East China Sea (ECS) and South China Sea (SCS) Large Marine ecosystems, Hawaii (Kaloiko-Honokōhau), a marine-coastal ecosystem of Sri Lanka (Indian Ocean) and Eastern Tropical Pacific, as shown in Figures 10a-10e.

Similar to the Pacific Ocean, other marine regions with higher *TMF* values were projected in the Atlantic Ocean such as the Benguela Current Large Marine Ecosystem (along Namibia and South Africa coasts), the northern Gulf of Mexico, and the North Atlantic Gyre, as well as further north in the Arctic circumpolar circle (Low Barents Sea) (Table 1; Figures 9h-9l). Conversely, relative lower *TMF* values (*TMF* > 5.1-9.6; Table 3) were predicted in the Grand Banks, Newfoundland (Figure 10f), Pros Crous Marine National Park in the northwestern Mediterranean (Figure 10g), and southern coast of Brazil (Figure 10h), as well as Darwin and Wolf Islands in the northern region of the Galápagos Islands, where the largest biomass of sharks exists (Figure 10i).

The highest concentration levels of microplastics were generally predicted in top predators representing the highest trophic levels, though there were several cases in which other upper trophic levels even below the apex predator position exhibited the highest concentrations, as projected in the foodwebs of Baja California (Figure 9g), northern Gulf of Mexico (Figure 9h), Benguela Current Marine Ecosystem (Figure

9j), SCS Large Marine Ecosystem (Figure 10a), Sri Lanka (Figure 10d) and Eastern Tropical Pacific (Figure 10e) and Darwin and Wolf Islands in the Galápagos Islands (Figure 10i). No trophic dilution of microplastics (a decline in contaminant concentrations as the trophic level increases, $TMF < 1$) was predicted in any of the marine foodwebs assessed.

The lowest TMF values predicted in some regions result from the very low MP concentration reported such as those found in the Eastern tropical Pacific close to the Galápagos Islands, and also in Darwin and Wolf Islands (Galapagos Islands), based on old data reported by Spears et al. (1995). They are also influenced by the composition of trophic level and the structure of the food web (Alava, 2020). Newly published empirical data can be used to update the model contaminant input data for the Ecotracer routine for some of these remote marine ecosystems. In the Galápagos Islands, for instance, the newest published data from Jones et al., (2021) and McMullen (2023), can be used.

Table 3. Apparent trophic magnification factors (*TMF*) and regression statistics for the linear regression models of the log of microplastic (MP) concentrations versus trophic level (TL).

Marine regions/ecosystems	Slope (<i>b</i>)	<i>TMF</i>*	Biomagnification metric outcome
Antarctica Peninsula	1.7398	54.9	Potential Biomagnification
Northern California Current	1.4795	30.2	Potential Biomagnification
Aleutians Islands (Alaska, North Pacific)	1.4642	29.12	Potential Biomagnification
South Pacific Subtropical Gyre (Central Chile)	1.4636	29.08	Potential Biomagnification
Galapagos Islands (Floreana Island rocky reef)	1.3769	23.8	Potential Biomagnification
North Eastern Pacific (British Columbia, Canada)	1.3480	22.3	Potential Biomagnification
Baja California (Mexico)	1.3109	20.5	Potential Biomagnification
South Benguela Current Large Marine Ecosystem (southeastern African coastlines: Namibia and South Africa)	1.3103	20.4	Potential Biomagnification
Galapagos Islands (Bolívar Channel Ecosystem)	1.2231	16.7	Potential Biomagnification
Australia Northwest Shelf	1.2031	16.0	Potential Biomagnification
Northern Gulf of Mexico	1.1867	15.4	Potential Biomagnification
North Atlantic Gyre	1.1594	14.4	Potential Biomagnification
Low Barents Sea	1.1382	13.7	Potential Biomagnification
South China Sea Large Marine Ecosystem (SCS LME)	1.1320	13.6	Potential Biomagnification
Kaloko Honokōkai (Hawaii, US)	1.0942	12.4	Potential Biomagnification
East China Sea Large Marine Ecosystem (ECS LME)	1.0663	11.6	Potential Biomagnification
Sri Lanka (Indian Ocean)	1.0533	11.3	Potential Biomagnification
Eastern Tropical Pacific	1.0163	10.4	Potential Biomagnification
Grand Banks Newfoundland-southeastern Labrador	0.9815	9.58	Potential Biomagnification
Northwestern Mediterranean Sea (Port Cros)	0.9472	8.86	Potential Biomagnification
Southern Brazil (Atlantic Ocean)	0.7094	5.12	Potential Biomagnification
Galapagos Islands (Darwin & Wolf islands)	0.6701	4.68	Potential Biomagnification

**TMF*, trophic magnification factor calculated as the anti-log of the slope: $TMF = 10^b$, where *b* is the slope. All *TMF* values were statistically and significantly greater than one ($p < 0.001$ or $p < 0.0001$; $TMF > 1$).

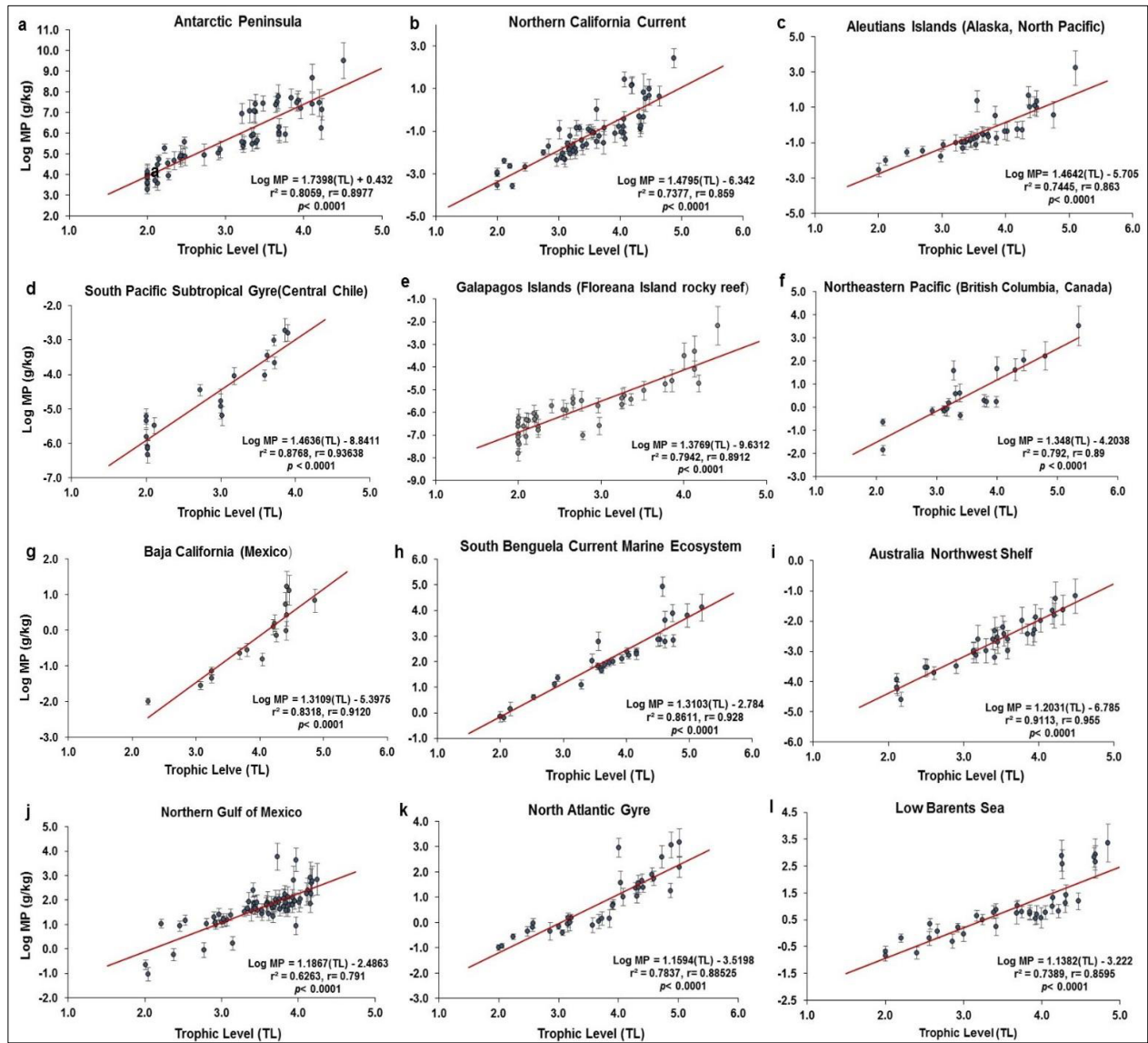


Figure 9. Linear regressions showing the statistically significant relationships between predicted concentrations of microplastics (log-transformed data) and trophic levels (TL) in marine foodwebs from ecosystem and regions of the global ocean assessed in this work. The trophic magnification factor (*TMF*) is calculated as the anti-log of the slope: $TMF = 10^b$, where b is the slope. All *TMF* values were statistically and significantly greater than one ($p < 0.001$ or $p < 0.0001$; $TMF > 1$).

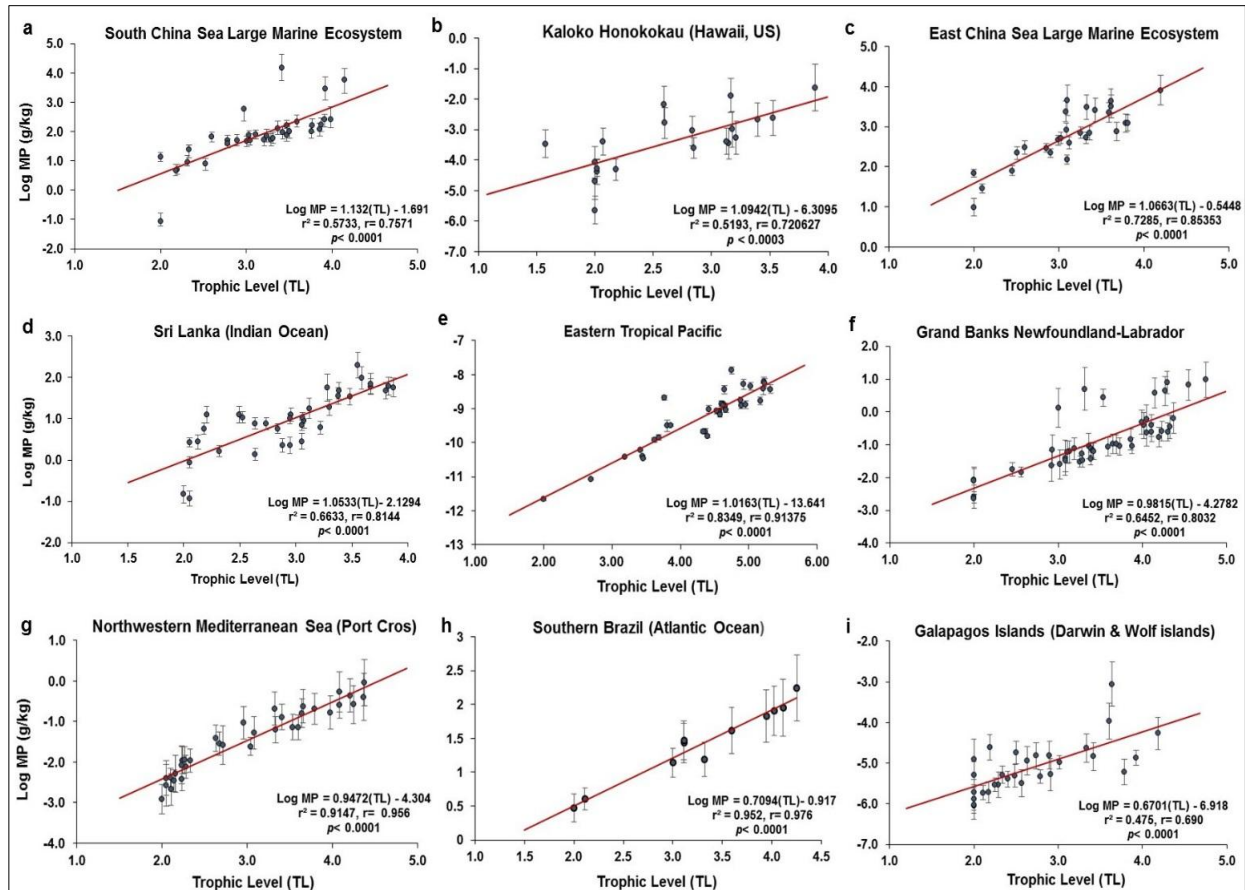


Figure 10. Linear regressions showing the statistically significant relationships between predicted concentrations of microplastics (log-transformed data) and trophic levels (TL) in marine foodwebs from ecosystem and regions of the global ocean. The trophic magnification factor (*TMF*) is calculated as the anti-log of the slope: $TMF = 10^b$, where b is the slope. All *TMF* values were statistically and significantly greater than one ($p < 0.001$ or $p < 0.0001$; $TMF > 1$).

Key finding and concluding remarks from EwE modelling application

- The projections of microplastic bioaccumulation produced by the application of trophic dynamic ecosystem modelling (i.e., EwE models) reflect the intrinsic mechanistic outputs of the transferring of energy and biomass flowing throughout the species and/or species' functional groups of the food web from preys to predators (i.e., the mortality of prey is survival for predator) by mass balancing the consumption and biomass production flow in the ecosystem.
- The EwE and Ecotracer modeling applications show that top predators are exposed to high levels of microplastics, underscoring pollution health risks for marine fauna and coastal wildlife.
- The microplastic accumulation, nonetheless, can be counteracted by the elimination or egestion rate constant as function of the retention time of the plastic particle in the GI tract as these two parameters are inversely related (i.e., the lower the retention time, the higher the elimination rate) and they have specific and different values for each species or functional group (e.g., high elimination rate in zooplankton versus low elimination rate in marine mammals).
- While there is a potential bioaccumulation impact for marine species and food webs, questions linger concerning the public health implications and food safety in coastal communities, heavily reliant on seafoods, such as those found in the Northeastern Pacific coast, Baja California (Mexico), southeastern African coastlines along Namibia and South Africa (South Benguela Current Large

Marine Ecosystem), ECS and SCS Large Marine ecosystems, Hawaii (Kaloko-Honokōhau), Galapagos Islands (rocky reef of Floreana Island) and coastal Sri Lanka.

- Following the precautionary principle, the predictions of food web biomagnification of microplastics documented in Table 1 in concert with the significant positive relationships shown in Figures 5 and 6 should be considered and interpreted with substantial caution until we have a better understanding on the role of retention times and elimination rates of microplastics.
- The egestion or elimination rate is a key parameter dictating the bioaccumulation behaviour of these emerging micropollutants in marine biota and food webs, and once more empirical data from field studies and lab exposure experiments become available, the models can be adjusted and improved.

The preliminary simulations of the ecosystem modelling revealed apparent bioaccumulation and biomagnification potential in all foodwebs and top predators in the suit of marine ecosystems assessed. These outcomes are comparable to limited existing microplastic bioaccumulation and biomagnification simulations using EwE (Ma & You, 2021; McMullen, 2023). Ma and You (2021) predicted that microplastics bioaccumulate quickly in fish food webs of Baiyangdian Lake (China) via EwE ecosystem modelling. The projections of microplastic concentrations in biota revealed that top predators are likely exposed to higher levels of microplastics accumulated through their diet items. Likewise, microplastic bioaccumulation modelling in a cetacean food web (Alava, 2020) with comparable EwE results indicates that species-specific bioaccumulation of microplastics is likely, while biomagnification is highly dependent on species-specific elimination rates. Miller et al. (2020) documented evidence supporting the bioaccumulation of microplastics in marine species; conversely, biomagnification of these pollutants in the food web has yet to be confirmed by field or empirical data.

In general, future modelling research should incorporate scenarios of microplastic accumulation across trophic levels under scenarios testing changes in the elimination or egestion rates as a function of GI tract retention times, and physical-chemical characteristics (shape, size, polymer composition) of retained microplastics (see McMullen, 2023). Via mediation processes in the EwE models, different iterations with microplastics set at trophic level 1 as the primary consumer level can be tested. Microplastics, for instance, may well be incorporated as a specific functional group in EwE and mediation applied to increase plastic particle consumption when phytoplankton abundance is low and vice versa (D. Price; pers. comm., 2022; following recommendation by V. Christensen & C. Walters, Institute for Ocean and Fisheries, University of British Columbia; see also Chapter 3 in McMullen, 2023).

Global ocean exposure-risk index for microplastics pollution footprint

As an emerging global environmental problem, marine plastic debris of all sizes represents a potential threat to our ecosystems where all organisms, including humans, are exposed to it through deleterious entanglements, physical trauma and ingestion (Galloway, 2015; Galloway et al., 2017; Kühn et al., 2015; Kühn & van Franeker, 2020; López-Martínez et al., 2021; Nelms et al., 2018; Rochman et al., 2015; Schuyler et al., 2014; Wilcox et al., 2015). In a recent study, Savoca et al., (2021) revealed that the abundance of plastic in surface waters was positively correlated to plastic ingestion by fish. As such, it is necessary to continually develop our understanding of the exposure that marine life has to plastics around the world and where these plastics may be coming from.

Additionally, concerns about bioaccumulation of microplastic and plastic-derived contaminants in humans due to seafood consumption have been raised by different authors, even though there is not yet a comprehensive understanding of the direct effects (Barboza et al., 2018; Carbery et al., 2018; Domenech & Marcos, 2021; Lusher et al., 2017; O'Neill, & Lawler, 2021; Revel et al., 2018; Santillo et al., 2017; Smith et al., 2018). While the direct contamination and health risks of nanoplastics and microplastics due to indirect ingestion or direct consumption via dietary exposure in human beings requires more research, some concerns about the public health impacts on humans who consume seafood that may have ingested microplastics or any associated toxic pollutants have been identified (Cox et al., 2019; Santillo et al., 2017; Smith et al., 2018; van Cauwenberghe and Janssen, 2014).

The weight of evidence in tandem with the epidemiology concerning microplastic prevalence, associated toxicity and carcinogenesis, and known translocation to many human organs and tissues, including blood, placenta, breastmilk, colon, lungs and liver, sufficiently draws concern regarding the implications of long-term exposure (Amato-Lourenço et al., 2021; Horvatits et al., 2022; Ibrahim et al., 2021; Jenner et al., 2022; Kumar et al., 2022; Leslie et al., 2022; Ragusa et al., 2021; Ragusa et al., 2022; Ramsperger et al., 2023).

Plastics in the marine environment are increasing and as a result, microplastics are ingested by many species of marine wildlife including fish and shellfish (Azevedo-Santos et al., 2019; Everaert et al., 2018; Jovanović, 2017; Lusher et al., 2017; Savoca et al., 2021), which is becoming a great concern for marine life and potentially human health. Addressing these research gaps is a critical priority due to the nutritional importance of seafood consumption and associated long-term food security implications for coastal communities and the most exposed populations.

Using the global ocean distribution data for microplastic water concentration from Litterbase (<https://Litterbase.awi.de/litter>; Bergmann et al., 2017; Tekman et al., 2023b) plus the Global Microplastics Initiative dataset (<https://www.adventurescientists.org/microplastics.html>; Barrows et al., 2018; Christiansen, 2018) in combination with the application Arc-GIS geospatial analysis and statistical tools, this research component contributes to the development of a preliminary global ocean exposure-risk map to portray the microplastic footprint (i.e., the interpolated global ocean heat map shown in Figure 12) and pin point the plausible microplastic exposure of coastal communities, which are heavily dependent on seafood. This is done around the global ocean using *per capita* seafood consumption rates.

Methodology

Spatial and Temporal Distribution of Microplastics in the Ocean

To understand the spatial and temporal distribution of microplastics, two datasets were compiled (i.e., Litterbase and the Global Microplastics Initiative dataset) and standardized to determine abundances and masses of both microplastics and nanoplastics. The initial datasets included 5,097 sample points characterized by habitat distribution (i.e., pelagic, coastal, open ocean and surface) and comprised data collected from 1960 – 2019 with the highest number of samples (1,159) collected during 2014 (Figure 11).

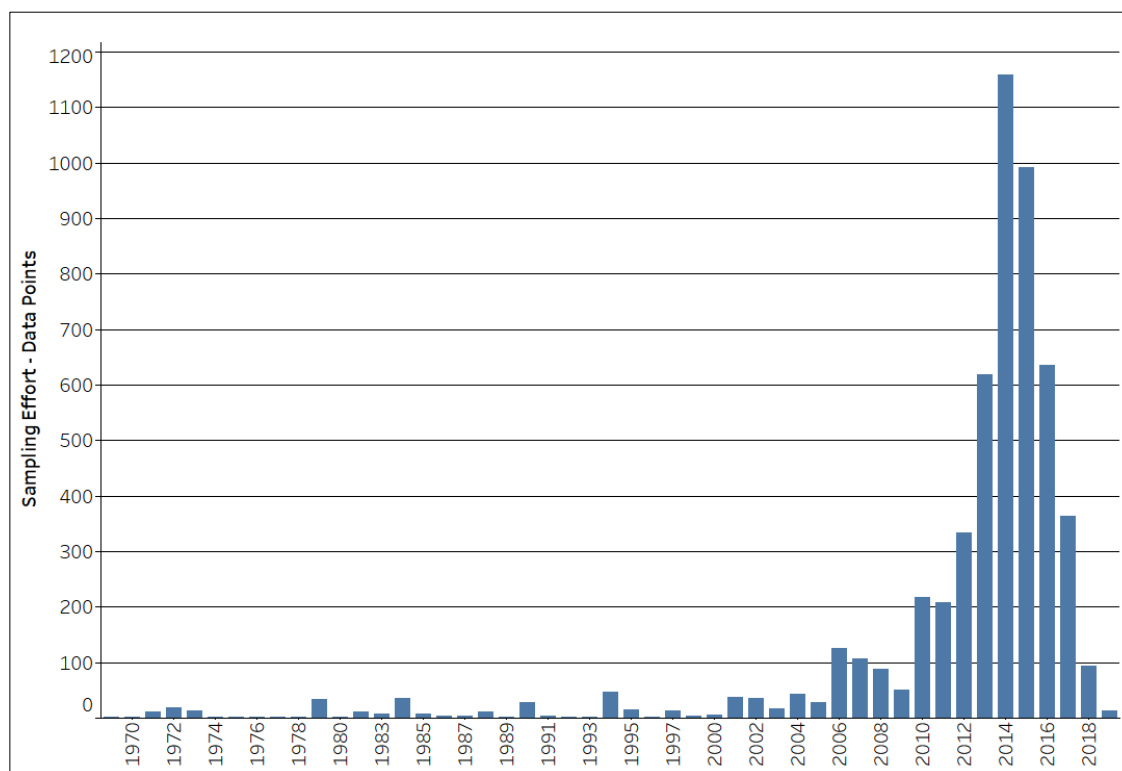


Figure 11. Distribution of the sampling effort for ocean plastic samples (e.g., plastic particles) in a time scale (1970-2019). Sources of information are Litterbase (<https://Litterbase.awi.de/litte>; Bergmann et al., 2017; Tekman et al., 2023b) and Global Microplastics Initiative dataset (<https://www.adventurescientists.org/microplastics.html>; Barrows et al., 2018; Christiansen, 2018).

The Litterbase data contains litter quantities taken from publications (Tekman et al., 2023b) while the Global Microplastics Initiative dataset included samples of water collected by volunteers (Haab, & Haab 2016; Barrows, et al 2017; Waller et al., 2017; Barrows et al., 2018). Outliers were identified and eliminated, along with other properties excluded for the purpose of this work. These excluded records were those containing habitat ice, a sampling method corer, the aquatic system freshwater, data prior to the year 2010 and values higher than 300,000 g/m³. A total of 3,351/5,097 records were considered for analysis (Figure 12). Given that the compiled data came from different sources using various sampling designs, devices, and reported units, a standardized concentration unit (particles/m³) for microplastic was established.

A preliminary global microplastic sample effort map portraying the geographic distribution can be seen in Figure 12. This dataset was used to generate a visual representation of densities and identify the hotspots in terms of sample effort and microplastic pollution (Figure 12). Based on this density analysis, the regions with the most samples of microplastics were the North Atlantic, the Arctic and the Mediterranean.

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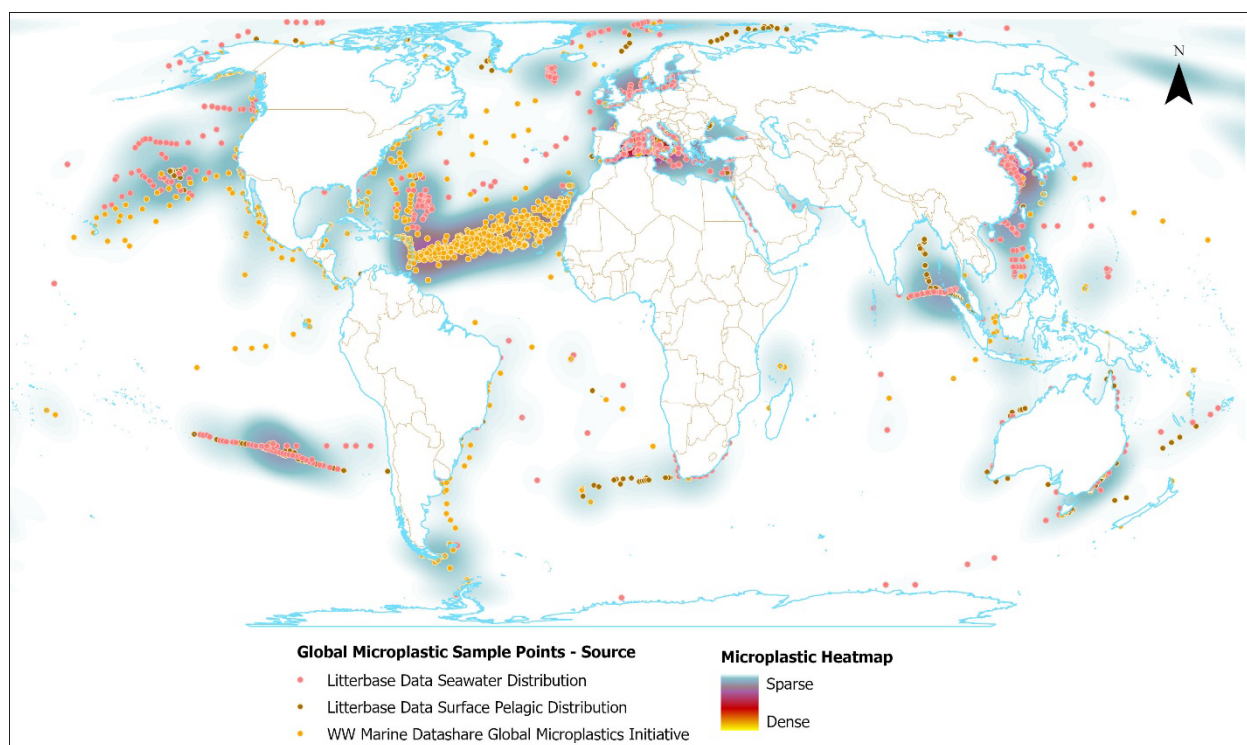


Figure 12. Global microplastic sample points integrated in a geographic information system portraying the spatial distribution of sample effort and microplastic density (particles/m³). Sources of information are Litterbase (<https://Litterbase.awi.de/litte>; Bergmann et al., 2017; Tekman et al., 2023b); and Global Microplastics Initiative dataset (<https://www.adventurescientists.org/microplastics.html>; Barrows et al., 2018; Christiansen, 2018).

Microplastic Pollution Index

For the purpose of this report, a comprehensive analysis of microplastic distribution using ocean surface or deep-water currents was not included. An Ordinary Kriging (OK) method was used to produce prediction maps of the spatial distribution of microplastics using a spatial unit of 2.5 km² pixel size; the semi-variogram model used was linear with a lag size of 2500 map units (Oliver & Webster, 1990; Snyder, 1997). These sample concentrations ranged from 71,782,400 to 5.02 x10⁻⁷ p/m³. Microplastic interpolated values were standardized using a linear function, changing to 0 to 10 scale, using 6,650 and 73,150 p/m³ as thresholds (calculation based on Everaert et al. 2020) to calculate a microplastic concentration index (MCI). This standardization set a minimum microplastic toxic threshold for marine organisms of 7,990 p/m³ based on Everaert (2020) and a maximum value of ten times the minimum threshold value, as shown in Figure 13. Of particular interest is the magnitude of microplastic concentration distributed towards the Central Pacific and North Atlantic, close to the oceanic subtropical gyres existing in these regions (Figure 13). Similarly, a high-density concentration footprint is projected at the North and South Poles, the eastern coast (Atlantic coast) of both Canada and US, North and Central Atlantic Gyre, the North and South Pacific Gyres, Caribbean Sea, South China Seas and the East China Sea Large Marine Ecosystem. This finding may well be biased due to higher sampling efforts and field research work campaigns deployed in Northern Hemisphere relative to less studied oceanic and coastal regions in lower latitudes (i.e., southeastern Tropical Pacific, and South Atlantic, and the Indian Ocean).

This analysis supports the identification of the low global sample rates along with potential countries and regions with low MCI.

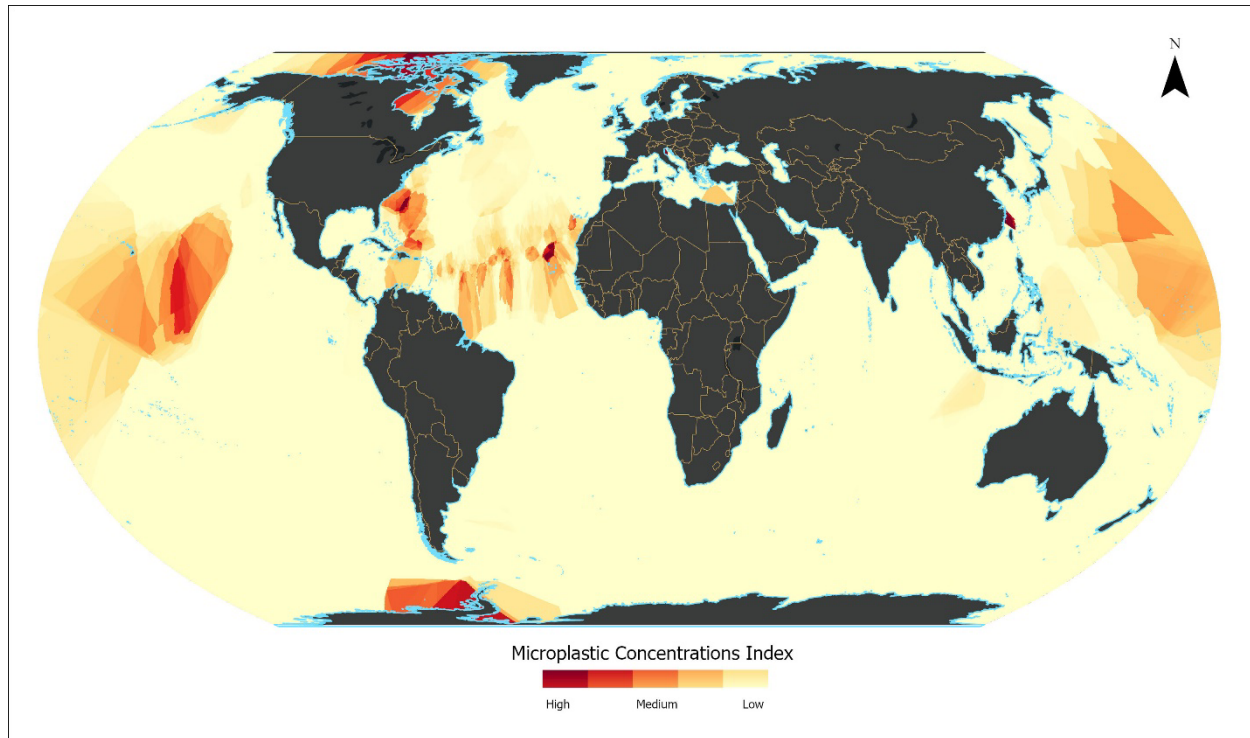


Figure 13. Global Ocean map portraying the microplastic concentration index (MCI) (see scale in colors) based on the database Litterbase (<https://Litterbase.awi.de/litte>; Bergmann et al., 2017; Tekman et al., 2023b); and Global Microplastics Initiative dataset (<https://www.adventurescientists.org/microplastics.html>; Barrows et al., 2018; Christiansen, 2018). Original conceptualization of idea and study design: Dr. J.J. Alava. Designing, geospatial analysis and map development by Dr. M. Moreno-Baez.

Global Ocean Potential Exposure Index

The Microplastic Potential Exposure Index (MPEI) assumes that the abiotic compartment (i.e., concentration in ocean water: $C_{MP-water}$ in units of g/m³), containing the potential concentration of microplastics ingested by marine organisms ($C_{MP-biota}$ in units of g/kg), is estimated considering the microplastic-bioconcentration factor (BCF_{MP} in units of m³/kg), defined as the distribution ratio between the microplastics concentration in $C_{MP-biota}$ to $C_{MP-water}$, i.e., $BCF_{MP} = [C_{MP-biota}]/[C_{MP-water}]$ (Alava, 2020; Alava, 2021). Thus, it calculates a degree of exposure based on the global distribution of microplastic concentration in the ocean water as the abiotic compartment ($C_{MP-water}$ in units of particles/m³) and includes the potential concentration of microplastics in marine organisms as the biotic compartment ($C_{MP-biota}$ in units of g/kg). The $C_{MP-biota}$ is estimated considering the microplastic-bioconcentration factor (BCF_{MP} in units of m³/kg), which is defined as the distribution ratio between the microplastics concentration in the biotic compartment ($C_{MP-biota}$) and the concentration in ocean water ($C_{MP-water}$), as follows (Alava, 2020; Alava, 2021):

$$BCF_{MP} = \frac{[C_{MP-biota}]}{[C_{MP-water}]}$$

This bioaccumulation metric approach assumes a steady state (i.e., chemical concentrations no longer change over time), and equilibrium, in which concentrations are equally distributed between marine biota and water, i.e., $BCF_{MP} = 1$. Thus, solving for:

$$[C_{MP-biota}] = [BCF_{MP}] * [C_{MP-water}]$$

The mean *per capita* seafood ingestion (kg/per capita/year) based on Cisneros-Montemayor et al., (2016) was used with the assumption that seafood consumption comes mainly from local fishing zones. This dataset includes ~1,900 Indigenous communities around the world and provides information about seafood consumption for communities that heavily depend on ocean-based protein for their livelihoods. The fishing zones were determined by calculating a buffer of 120 km around each Indigenous community and the land area was then extracted to obtain an exclusively marine area that was considered as the influence zone per community. Next, the mean *per capita* seafood ingestion (kg/per capita/year) was standardized to a scale of 1 – 10 using a linear function with original min and max values as thresholds (7.5-241.9) to generate a seafood consumption index (SCI). Using ESRI ArcGIS Pro 2.9.5 zonal statistics the MCI value in the biotic compartment ($C_{MP-biota}$) was determined by influence zone (i.e., x/km²) per community.

$$MPEI = \frac{\overline{MCI} * \overline{SCI}}{10}$$

Where MPEI is the Microplastic Potential Exposure Index; MCI is the microplastic concentration index; and SCI is the standard seafood consumption for that community. Descriptive statistics for MPEI values (average and maximum) and average seafood consumption are shown in Table 4 summarized for Indigenous communities by regions (continents), subregion, countries and subregion countries.

A potential microplastic exposure index was developed based on the assumption that concentrations of microplastics in the water can potentially be equivalent to the concentration of microplastics in fish or marine invertebrates in a given marine region or area. Doing so, a MPEI was developed with two variables: seafood consumption and the exposure to concentration of microplastics.

Table 4. Top 35 MPEI highest values of coastal Indigenous communities summarized by region/country.

Region	Subregion	Country	SubRegion Country	# Localities	Avg. MPEI	Max. MPEI	Average Estimated Consumption per capita
Africa	Eastern Africa	Comoros	Eastern Africa Comoros	2.00	0.70	0.70	164.0
		Kenya	Eastern Africa Kenya	5.00	0.70	0.70	164.0
		Mauritius	Eastern Africa Mauritius	1.00	0.70	0.70	164.0
		Tanzania	Eastern Africa Tanzania	8.00	0.70	0.70	164.0
		Transboundary	Eastern Africa Transboundary	3.00	0.70	0.70	164.0
	Middle Africa	Angola	Middle Africa Angola	3.00	0.70	0.70	164.0
		Cameroon	Middle Africa Cameroon	8.00	0.70	0.70	164.0
		Eq Guinea	Middle Africa Eq Guinea	1.00	0.70	0.70	164.0
		Sao Tome Prn	Middle Africa Sao Tome Prn	2.00	0.73	0.73	164.0
		Transboundary	Middle Africa Transboundary	4.00	0.70	0.70	164.0
	Northern Africa	Transboundary	Northern Africa Transboundary	1.00	0.70	0.70	164.0
	Southern Africa	South Africa	Southern Africa South Africa	1.00	0.63	0.63	145.7
	Western Africa	Cape Verde	Western Africa Cape Verde	1.00	1.32	1.32	90.3
		Cote D'Ivoire	Western Africa Cote D'Ivoire	10.00	0.59	0.59	133.9
		Ghana	Western Africa Ghana	1.00	0.70	0.70	164.0
		Senegal	Western Africa Senegal	2.00	0.92	0.94	164.0
		Sierra Leone	Western Africa Sierra Leone	1.00	0.74	0.74	164.0
		Transboundary	Western Africa Transboundary	14.00	0.64	0.73	119.4
Americas	Northern America	Canada	Northern America Canada	181.00	0.46	1.54	59.7
		Greenland	Northern America Greenland	3.00	1.05	1.75	162.0
		USA	Northern America USA	190.00	0.45	1.71	75.7
	South America	Fr Guiana	South America Fr Guiana	1.00	0.61	0.61	46.3
		Transboundary	South America Transboundary	9.00	0.35	0.71	46.3
		Venezuela	South America Venezuela	1.00	0.66	0.66	43.0
Asia	Eastern Asia	Japan	Eastern Asia Japan	1.00	0.79	0.79	158.7
	Southeastern Asia	Indonesia	Southeastern Asia Indonesia	21.00	0.49	0.69	99.9
Oceania	Australia and New Zealand	Australia	Australia and New Zealand Australia	318.00	0.25	0.70	47.4
	Micronesia	FS Micronesia	Micronesia FS Micronesia	19.00	1.17	1.17	93.0
		Kiribati	Micronesia Kiribati	1.00	1.91	1.91	139.5
		Marshall Is	Micronesia Marshall Is	1.00	0.76	0.76	49.0
		Palau	Micronesia Palau	3.00	0.71	0.71	109.5
	Polynesia	Niue	Polynesia Niue	1.00	0.63	0.63	84.0
		Tokelau	Polynesia Tokelau	1.00	0.63	0.63	74.6
		Tuvalu	Polynesia Tuvalu	1.00	0.80	0.80	115.5
		Wallis and Futuna	Polynesia Wallis and Futuna	3.00	0.62	0.62	74.0

Key Findings

The Microplastic Potential Exposure Index aimed to calculate a degree of exposure based on the global distribution of microplastic concentration in the water (C_{Water}), based on the two datasets (i.e., Litterbase and the Global Microplastics Initiative dataset), as well as the potential concentration of microplastics in marine organisms (C_{Biota}), estimated from the application of the BCF, as aforementioned. Data and results were classified by region and subregion to display the MPEI results, including the top 35 MPEI highest values of coastal Indigenous communities (Table 4). The results help visualize the geography of exposure at different scales (Figure 14).

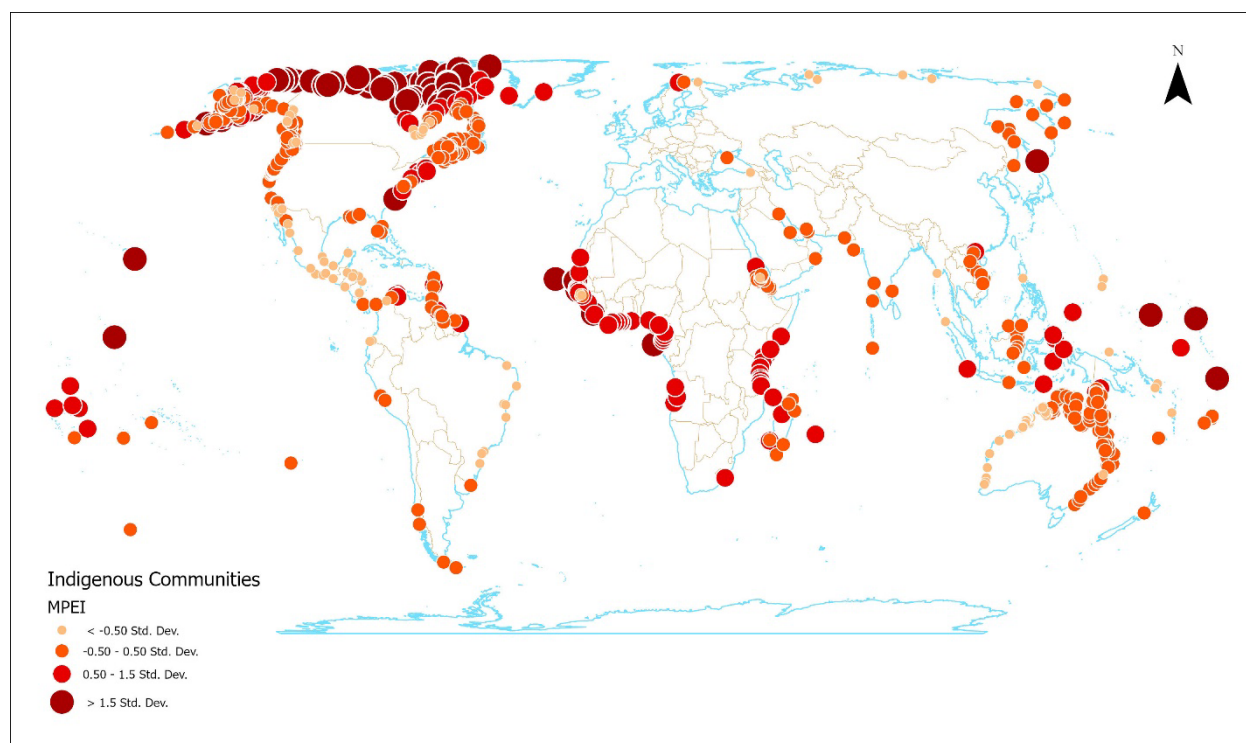


Figure 14. Microplastic Potential Exposure (MPE) index (see scale in colors) for coastal communities based on the global estimate of seafood consumption by coastal indigenous peoples (Cisneros-Montemayor et al., 2016) and the global estimate of microplastic concentrations. Original conceptualization of idea and study design: Dr. J. J. Alava and Dr. M. Moreno-Baez; geospatial analysis and map development by Dr. M. Moreno-Baez.

It is important to note that these results show a degree of exposure; however, it is not yet possible to quantify the health risk that this exposure might have in human populations until more human ecotoxicological risk assessments and epidemiological data on the mechanisms of toxic action and direct toxicity or health effects become available (see review by Ramsperger et al., 2023). While the global per capita consumption of plastic particles via seafood has been estimated at $\sim 22 \times 10^3$ plastic particles per year (Domenech & Marcos, 2021), questions linger on the human and public health risk and impacts on marine-resource dependent communities inhabiting coastlines of the global ocean.

As it stands, there is still a need for more extensive work that includes measures of plastic contamination with a systematic sampling process in seafood. In addition, it is of paramount importance to understand fish and seafood consumption at smaller scales, which would imply understanding in detail where consumed fish and seafood originate. Our preliminary results are made available in an attempt to assist future research and policy makers with selecting locations for future monitoring and modelling research efforts to support ecotoxicological and human risk assessments.

Future work implications for human health and nutrition

- Standardize data collection and analysis methods for microplastic occurrence in the marine environment.
- Standardize data collection assessing major seafood production types, consumption and seafood producing countries.
- Collect data on presence, identity and quantity of degraded plastic in seafood, and data on the how microplastics persist through the aquatic food web and human food system.
- Assess microplastics' impact on food safety and improve understanding of potential exposure risks, particularly in communities that heavily depend on ocean-based protein.
- Identify, if possible, specific areas of high seafood consumption, and understand the interactions of microplastics with high-consumed species, in order to promote adjustments rather than consumer avoidance of seafood.

Addressing the inequity gap for solution-oriented research and equitable interventions framework

The increase in public awareness of the problem of ocean plastics presents an opportunity to capitalise on this attention and implement bold solutions that will tackle the root cause of not only plastics, but also issues in environmental justice and equity (Bennett et al., 2022; Vandenberg & Ota, 2022). The people living in oceanic, remote and continental coast areas, of which many are Indigenous communities have common and unique food security, food safety and public health issues in the face of pervasive ocean pollution by marine plastics (see Table 2).

While the circular economy of plastics per se is an ideal concept, we cannot ignore the fact that the plastic life cycle is not in fact a closed circle. There is value in promoting a life cycle assessment strongly relying on the circularity of plastic materials for plastic waste management by considering a new plastics economy constantly flowing back and forth from plastic production to consumption via a closed loop system through recycling, reduction and reuse (Ellen MacArthur Foundation, 2016; O’Neil, 2019; Jones et al., 2023), though it may still perpetuate inequality gaps and neglect potential environmental justice consequences because of the lack of equitable interventions and solutions for the most exposed coastal communities, many of which are minority Indigenous peoples (Bennett et al., 2022; O’Neil, 2019; Vandenberg & Ota, 2022). One such example of the limited effectiveness of the circular plastic framework is a study of 2016 industries that identified only ~9% of Canadian plastic waste was recycled, indicating linearity more so than circularity (Deloitte & Cheminfo Services Inc., 2019). In fact, only about 8 to 9% of 3.3 million tonnes of plastic waste produced per year in Canada is recycled, while 85% (2.8 million tonnes per year) ends up in landfills and 1%, (~29,000 tonnes) enters coastal environments, annually (ECCC, 2020; ECCC, 2021; Naddaf, 2021). Developing countries may well be concerned with the circular economy approach because these nations have less capacity both legally and technically to implement an infrastructure and a system to foster and support the circular economy approach without escaped plastics entering the marine environment and impacting other regions.

The social implications and inequitable consequences of a circular plastics economy approach in developing and underdeveloped nations are critical. In this context, inequality exists in who causes plastic pollution, who experiences its impacts and consequences, who can provide solutions, and who has the political will to make the decisions to divorce from a plastic dependence, fostering proactive end of the plastic life cycle assessments (Simon et al., 2021). The people exhibiting high consumption rates of plastics steaming from a pervasive plastic demand and those who supply it disproportionately affect the success or failure of plastic pollution mitigation.

A circular economy of plastic, for example, may not work as intended for small island developing nations (SIDS), communities inhabiting remote, oceanic islands and some developing or undeveloped countries (Jones et al., 2023). Thus, questions linger as to whether the circular economy of plastics is able to address, mitigate and minimize health risks and ecological impacts of plastic-associated chemicals and additives and potential microbial pathogens from plastics that would cause health risks and ecological impacts. We argue that there should not be plastics leaking into these coastal communities and SIDS from other regions under an ideal circular economy model. Thus, within this premise, we argue that: (1) the circularity of plastic economy may not be fully implemented and possible in remote oceanic-coastal communities and SIDS; and, (2) plastics affect coastal and developing nations more than other developed nations.

With the aim to address the plastic pollution problem surrounding and exacerbating the inequity gaps in the most affected and exposed communities and the corresponding burden on minority groups, it is of paramount to question and identify the sources of inequities, as follows:

What institutions, policies, and management systems perpetuates these inequities?

- Single-use plastic industries are the main sources of producing and marketing these kinds of ubiquitous plastics, polluting the marine and coastal environments and the world's oceans. (In December 2022, for examples, The Government of Canada banned six categories of harmful single-use plastics, including checkout bags, cutlery, foodservice ware made from or containing problematic plastics that are hard to recycle, ring carriers, stir sticks and straws, prohibiting their manufacture and import for sale (ECCC, 2022). On the other hand, Canada's ban does not include plastics bottles.
- Clothing, textile and apparel industries are responsible for producing and emitting microfibers from synthetic textiles such as nylon and polyester into the marine coastal areas and oceans (Athey et al., 2022).
- The industrial and small-scale fisheries sectors accidentally or/and intentionally discharge, abandon or release plastic objects/items, derelict fishing gear (e.g., ghost nets) and devices (e.g., fishing aggregating devices, FADs) at sea, severely impacting marine life, mainly epipelagic organisms and marine megafauna (Gilman, 2015; Kühn and van Franeker 2020, Kruse et al., 2023; López-Martínez et al., 2021; Macfadyen et al., 2009; Ryan, 2015).
- Aquaculture and/or mariculture facilities owned by the private sector can act as point pollution sources of plastic and microplastic emissions in coastal areas, wetlands/marshes and the open ocean (Lusher et al., 2017).
- The automotive sector generates automotive plastic waste during primary productions and in the form of microplastics that result from tires wearing on road surfaces (Wagner et. al., 2018).
- The electronic industries generate plastic waste from electrical and electronic components and equipment, i.e., e-waste (Babu et al., 2007; Jia et al., 2022).
- Local and/or regional governments, authorities and industries from developed and developing nations lack of a proactive responsibility for solid waste management system and policy to control, mitigate, prevent and eliminate plastic waste with equitable interventions to attack the root cause of the problem rather than prioritizing end-of-life approaches that are focused on symptoms-targeted solutions (Vandenberg & Ota, 2022). As it stands, the industry creates responsibility "scapegoats" serving as distractions from the continuing efforts of big business to avoid responsibilities, deflect blame or derail regulatory actions (Vandenberg & Ota, 2022).

In this context, new collaborative research frameworks and solutions-oriented research focused on the root cause of the plastics crisis are vital to ensure that the health and environmental protection needs of people living in coastal, rural and remote communities can be assisted with equitable interventions, appropriate care, mitigation strategies and environmental and health education programs to ensure equal access to hygiene, public health and pollution prevention measures for a healthy ocean environment and oceans free of plastics (Alava et al., 2022; Bennett et al., 2022; Onyena et al., 2022; Silva et al., 2020; Vandenberg & Ota, 2022).

What are socially equitable solutions for addressing marine litter and plastic pollution?

The following solutions are predominantly based on Alava (2019):

- Development of a comprehensive solutions-oriented framework and engagement with the community and key shareholders to equitably embrace the people's needs and contribute with ideas and solutions

to inform the designing of equitable interventions for the end of plastic products life cycle with appropriate plastic waste disposal and management.

- Devise product designs and innovations considering a bottom-up approach with policy and consultation processes to divorce from plastics generated from fossil fuels.
- Implement a proactive end of the life-cycle approach, from the design of plastic products to their production, use, collection, recycling, and recovery to foster a “zero plastic waste” or “single plastic use-free products” at the corporate or industrial-level.
- Target private sector and industries with policies to minimize plastic production and packaging and favour the designing of biodegradable plastic-free products by means of incentives, policy, and regulations
- Offer and implement good subsidies and fisheries incentives to small-scale or artisanal fisheries to collect or “fish” ocean plastics instead of fishing (i.e., fishers receive an economic incentive or salary from a plastic bank or depository by returning plastic collected/harvested from the sea and beaches).
- Promote knowledge and social mobilization and offer incentives for eco-friendly products and taxes for single-use plastics paired with market-based instruments (e.g., plastic bottle deposit/cash back for containers) to encourage and call on consumers to alter consumption, littering, and throw-away habits.
- Design a school curriculum to democratize a network of knowledge on plastic pollution and education, integrating best individual/community practices and habits through environmental awareness programs and outreach to eliminate plastic pollution framed within a “Plastic Pollution Solution Curriculum” in elementary schools, and secondary/high schools, and in colleges and universities.

The pursuit of a risk assessment criteria framework to categorize macro-, micro-, and nanoplastics and their associated chemicals in the global ocean and terrestrial environments as a new class of persistent (P), bioaccumulative (B), and toxic (T) pollutants (i.e., PBT pollutants), subject to the long-range atmospheric and oceanic transport, has also been proposed (Alava et al., 2023). This categorization framework is being proposed as powerful tool to support policy efforts directed at combating plastic pollution and highlights the implementation of bottom-up policy decisions along with precautionary actions aimed to champion global plastic governance by including equitable interventions and equal access to pollution prevention and mitigation strategies (Alava et al., 2023).

Using key findings and data from our applied research (i.e., projections of foodweb bioaccumulation of microplastics in marine foodwebs, global ocean pollution footprint of microplastics, and assessment of potential microplastic exposure and risks in coastal communities) in tandem with solution-oriented research from other research components, we discuss and contribute with actions and multi-target strategies to address plastic pollution and waste. We encourage communities, not-for-profit/non-profit organizations, businesses, industries and governments to work together to create innovative policies; support research and innovation; invest in best management practices for wastewater and solid waste infrastructure within a feasible, fair and sustainable circular economy of plastics in tandem with industrial transitions and innovative design of plastic-free products or substitutes (biodegradable or smart bioplastic materials); and shift mindsets and proactive changes in our behavioral practices.

In summary, concerted, bottom-up policy decisions along with precautionary actions and regulatory enforcement to cap and reduce plastic production, along with a reform for plastics’ end-of-life solutions, are urgently needed to combat the roots of global plastic pollution and implement just-transitions. These

policy efforts should also champion global plastic governance by including equitable interventions and equal access to pollution prevention and mitigation strategies. This is of paramount importance to address the inequality gap framework, with the aim to foster ocean equity and environmental justice in plastic pollution management for the most exposed people and impacted remote, oceanic-coastal communities of the global oceans.

Concluding Remarks

In this work, we have contributed to the assessment of the overall ecological impacts by marine plastic pollution with microplastic-food web bioaccumulation and ecosystem models that predict bioaccumulation potential in several marine foodwebs, as well as geospatial modelling analysis projecting the global geographic distribution of ocean plastics. This analysis reveals moderate to high microplastic concentration exposure to the majority of coastal marine-resource dependent communities of the world's oceans, represented largely by Indigenous peoples. In view of this accelerated rate of global pollution by marine plastics of all sizes (large plastics, microplastics and nanoplastics), which have been further exacerbated by the plastic pollution associated with the COVID-19 pandemic, concerted equitable interventions are needed. Such efforts include improved solid waste management and proactive public health strategies to hamper and eliminate plastic pollution, following the precautionary principle. The food web bioaccumulation models and global ocean map projections for microplastics' exposure developed and applied here will serve as practical tools and approaches to support ecotoxicological and human risk assessment, plastic waste management, and policy decisions to address the inequality gap framework for the most exposed people in coastal communities, and conserve marine biodiversity.

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Appendix I

Microplastic bioaccumulation modeling in cetacean food webs of the Northeastern Pacific: A regional exercise case

Model Scenarios

To explore the bioaccumulation capacity of organisms and biomagnification in the foodweb, the model was run with the three scenarios using different microplastic concentrations measured in abiotic compartments (seawater and sediments). Thus, based on the documented data for the study region, three scenarios were modeled (see Alava, 2020): (i) scenario 1: a low concentration scenario (i.e., water concentration = 0.66 particles/L; and sediment concentration = 60 particles/kg dw, which in units of mass are equivalent to 0.003 g/L and 0.266 g/kg dw, respectively); (ii) scenario 2: a moderate concentration scenario (i.e., water concentration = 2.08 particles/L; and sediment concentration = 200 particles/kg dw, equivalent to 0.010 g/L and 0.886 g/kg dw); and (iii) scenario 3: a high concentration scenario (i.e., water concentration = 9.18 particles/L; and sediment concentration = 25000 particles/kg dw, equivalent to 0.040 g/L and 111 g/kg dw). The simulation time for the foodweb bioaccumulation model was run for different times series (e.g., 1–100 year) starting at 0 day with 5-day intervals ($dt = 5$).

Sensitivity Analysis

The sensitivity of the model was mainly assessed by testing changes in the microplastic elimination rate in zooplankton, as the fundamental entry point for uptake and bioaccumulation of microplastics at the bottom of the food web. This was conducted by comparing the outcomes of the model at a high elimination rate in zooplankton ($k_E = 1/\text{d}$) versus a low elimination rate ($k_E = 0.143/\text{d}$) and running the model with empirical abiotic concentrations of microplastics observed in water and sediment (i.e., scenario 1: low concentration as conservative scenario; and, scenario 2: moderate concentration as the least conservative scenario). Under scenario 1 or scenario 2, the average \pm SD percentage of microplastic concentrations in zooplankton increased by $75\% \pm 17\%$ over a simulation time of 1 year (Figure A1A). The equivalent average increase ($85 \pm 2.0\%$) in microplastic concentrations in zooplankton subject to changes in the elimination rates is also predicted under either scenario 1 or scenario 2 at 100 year (Figure A1B).

While the changes in concentrations in other low and mid-trophic levels are generally lower (e.g., 0.5% to >2%), chum and coho salmon, and squid exhibited increases of ~4%, >6%, and >7%, respectively (at scenarios 1 and 2 at 1 year; Figure A1A). Relatively lower concentration changes are also projected in these organisms at 100 year of simulation. The incremental changes in the humpback whale were of 16 to 17% at 1 year and ~9–10% by 100 year, indicating that herring feeding on zooplankton with low or high elimination rate is a biological driver in the microplastic concentrations in humpback whales. As for Chinook salmon and resident killer whales, the changes in microplastic concentrations were minor at ~3% and >2–3.6% for all scenarios, respectively (Figures A1A and A1B).

Overall, the sensitivity analysis illustrates that the changes in the elimination rate (i.e., retention time from 1 to 7 days) in zooplankton are mainly propagated from filter and suspension feeding invertebrates through zooplanktivorous fish up to humpback whales, directly ingesting herring and zooplankton, but with little impact in high trophic level organisms such as Chinook salmon and fish-eating killer whales.

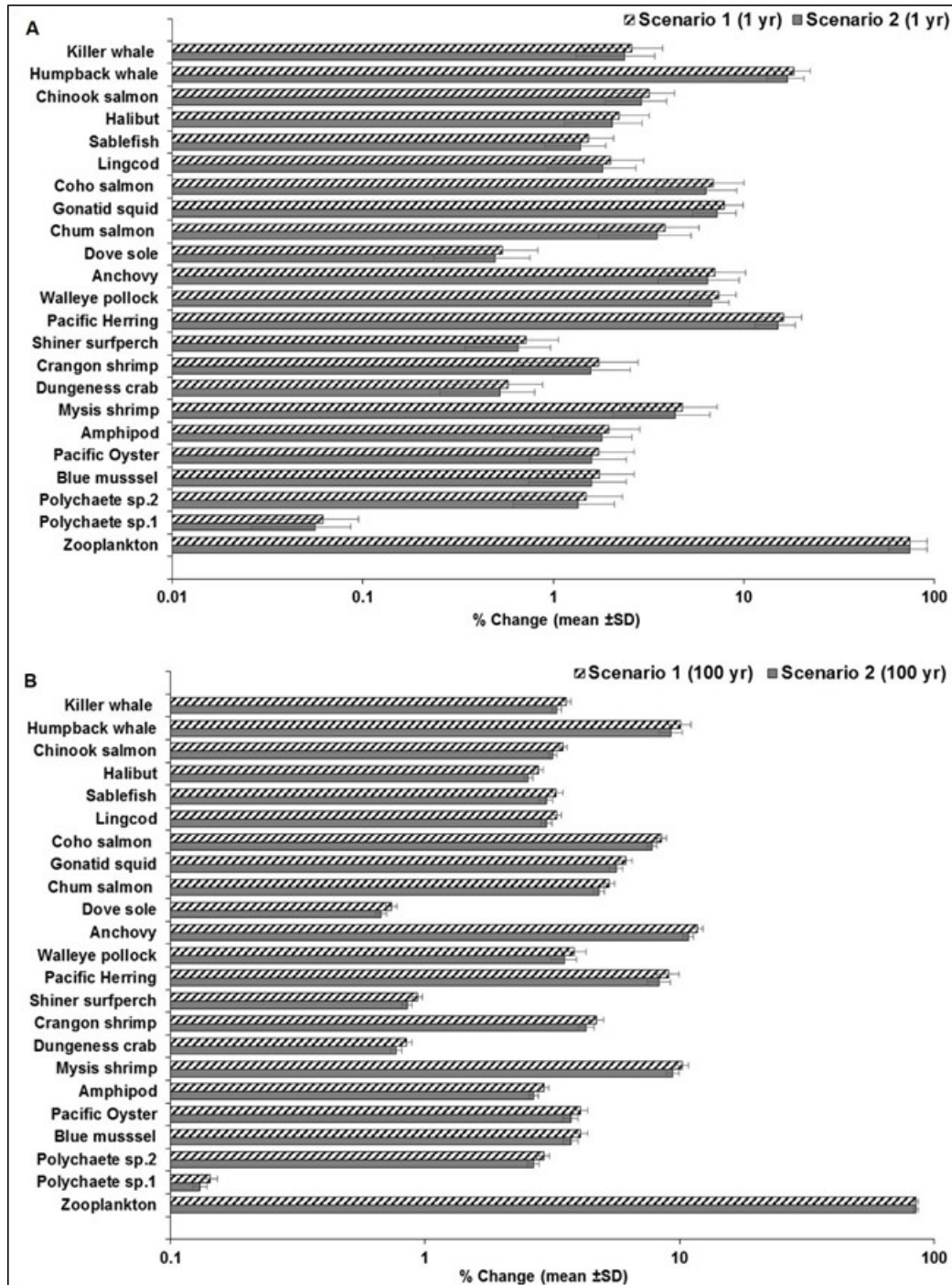


Figure A1. Outcomes of the sensitivity analysis showing the response of the food web to changes in the elimination rate of zooplankton at 1 and 100 years. (A) scenario 1: from $k_E = 1/d$ to $k_E = 0.143/d$ with low concentrations in water and sediment (0.003 g/L and 0.266 g/kg dw, respectively) at 1 year, as a conservative scenario; scenario 2: from $k_E = 1/d$ to $k_E = 0.143/d$ with moderate concentrations in water and sediment (0.010 g/L and 0.886 g/kg dw) at 1 year, as a least conservative scenario (scenario 1); and (B) scenario 1: from $k_E = 1/d$ to $k_E = 0.143/d$ at low concentrations in water and sediment (same as above) at 100 years, as a conservative scenario; scenario 2: from $k_E = 1/d$ to $k_E = 0.143/d$ with moderate concentrations in water and sediment (same as above) at 100 years, as a least conservative scenario. Figures adapted from Alava (2020).

Model Bias (MB)

A model bias (*MB*) approach was applied to assess the performance of the foodweb model and corroborate the projections of microplastics under the three abiotic concentrations' scenarios. Despite the limited empirical data of microplastic concentrations in most of the organisms composing the food web, the performance of the model was analyzed in terms of the model bias ratio: $MB = C_{BP,iMP}/C_{BO,iMP}$; where $C_{BP,iMP}$ and $C_{BO,iMP}$ are the model calculated and observed microplastic concentrations in species *I*, respectively. This analysis was done by comparing the projected microplastic concentrations in wild juvenile Chinook salmon to the observed microplastic data reported for this species in coastal BC.

Thus, the microplastic mean concentration (i.e., mean \pm SD: 1.2 ± 1.4 microplastics/individual) for wild Chinook salmon from the east coast of Vancouver Island reported by Collicutt et al. (2019) was used as empirical data. This concentration reported in microplastics/organism is equivalent to ~ 0.890 g/kg (~ 1.0 g/kg), using a microplastic mean mass of 0.00443 g (based on Alava, 2020), and a mean wet weight for juvenile Chinook salmon of 6.01 g (~ 0.006 kg) documented in Collicutt et al. (2019). The *MB* ratio can indicate the model's systematic over-prediction ($MB > 1$) or under-prediction ($MB < 1$) of the concentrations of a chemical contaminant in biota (Alava et al., 2012, Alava et al. 2018).

Comparing the projected concentration of MPs in Chinook salmon to observed data in wild juvenile Chinook salmon (i.e., 0.89 ± 1.0 g/kg, derived from 1.2 ± 1.4 particles/individual; Collicutt et al., 2019), the outcomes of the *MB* ratio analysis reveal systematic under prediction ($MB < 1$), with a *MB* ranging 0.0–0.1 and 0.0–0.3 at low (scenario 1) and moderate (scenario 2) abiotic concentrations throughout the simulation, respectively (Figure A2). In scenario 3 (high concentrations), the *MB* ranged from 0.0 to 33.3 over the simulation time, with $MB = 1.0$ at 35–50 days of simulation, indicating that observed concentrations in Chinook salmon are reproduced fairly well by model's predicted data in this predatory fish at those time steps (Figure A2). Systematic overprediction with an order of magnitude higher ($MB > 10$) is generated beyond 200 days because of the continued exposure of prey to high abiotic concentrations of microplastics.

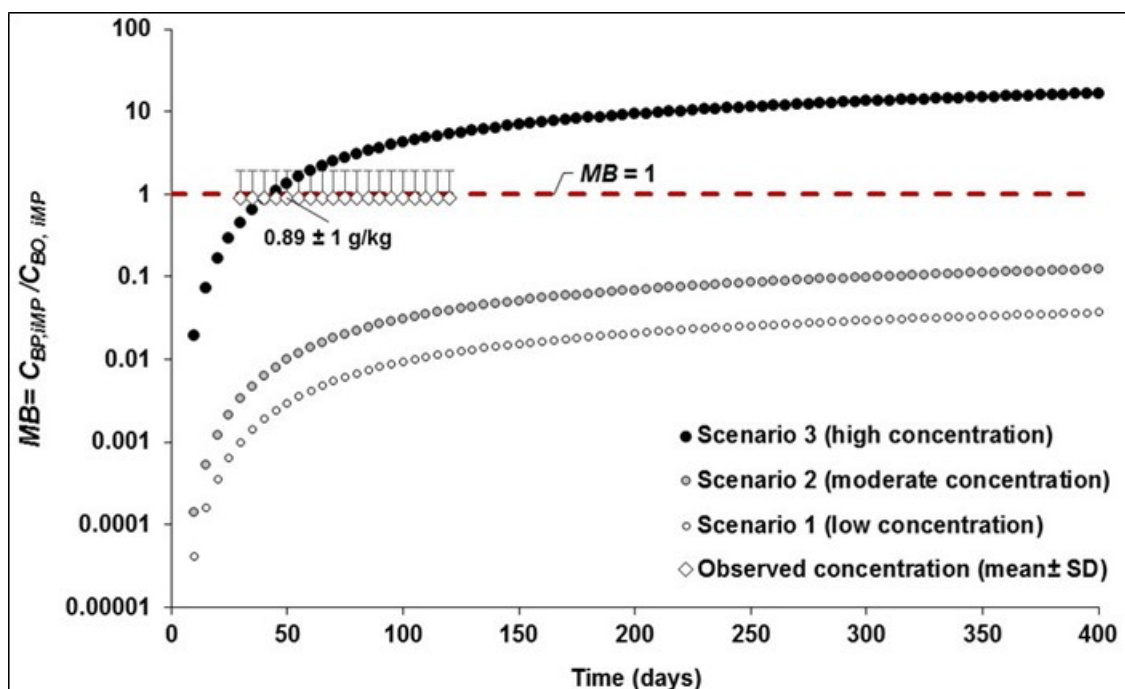


Figure A2. Assessment of the model bias ($MB = C_{BP,IMP}/C_{BO,IMP}$; where $C_{BP,IMP}$ and $C_{BO,IMP}$ are the model predicted and observed MP concentrations in Chinook salmon, respectively) and performance of the model by comparing the simulations of MP concentrations projected in Chinook salmon from 0 to >365 days under three abiotic concentrations scenarios (i.e., scenario 1: low concentration; scenario 2: moderate concentration; and, scenario 3: high concentration; see Scenarios above) to the empirical field data (mean \pm SD = 1.2 ± 1.4 particles/individual or 0.89 ± 1.0 g/kg) of MP concentration observed in wild juvenile Chinook salmon (Collicutt et al., 2019). The red dashed line represents equal concentration of MPs ($MB = 1$) between predicted and observed concentration data. Figures adapted from Alava (2020).

Appendix I Supporting References:

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- Alava, J. J., Ross, P. S., Lachmuth, C. L., Ford, J. K. B., Hickie, B., & Gobas, F. A. P. C. (2012). Habitat based PCB environmental quality criteria for the protection of endangered killer whales (*Orcinus orca*). *Environmental Science & Technology* 46, 12655–12663. doi: 10.1021/es303062q
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Appendix II

Ecopath with Ecosim (EwE) and Ecotracer model:

EwE is a trophodynamic simulation model that integrates biotic and abiotic components of an ecosystem. It uses assumptions of mass balance and a system of linear equations describing and tracking the average flows of mass and energy between functional groups (i.e., biomass pools: species or groups of species aggregated according to life-history and niche characteristics) according to a diet composition matrix while accounting for energy lost in respiration, emigration, and decomposition through time (Christensen & Pauly, 1992; Christensen & Walters, 2004; Christensen et al., 2005). Ecotracer is a module within EwE that tracks and assesses the bioaccumulation of pollutants in marine food webs over time (Christensen & Walters, 2004; Combs, 2004; Booth et al., 2016). Details on the core principles and equations of EwE can be found in the EwE user guide available online (Christensen et al., 2008).

Ecotracer uses Ecosim to predict movement and accumulation of contaminants (Combs, 2004; Alava et al., 2018; Booth et al., 2016). Specifically, changes in concentrations of chemicals are predicted using flow rates from Ecosim along with decay or elimination rates and physical exchange rates (Christensen & Walters, 2004). The linear dynamical equation for time changes in contaminant concentration in a given functional group (pool) or species i is expressed as:

$$CiBi/dt = (Cj \cdot GCi \cdot Qji / Bj) + (ui \cdot Bi \cdot Co) + (ci \cdot Ii) - [(Ci \cdot Qij / Bi) + Ci \cdot MOi + ((1 - GCi) \cdot \sum j Cj \cdot Qji / Bj + ei \cdot Ci + di \cdot Ci)]$$

Thus, the time dynamic changes in contaminant concentration in the biomass of a given functional group or species i ($CiBi$) can explicitly be described by the following components, based on Christensen & Walters (2004):

1. Uptake from food: $Cj \cdot GCi \cdot Qji / Bj$ where Cj = conc in food j , GCi = proportion of food assimilated by type i organisms; Qji = biomass flow rate from j to i (estimated in Ecopath as $Bi \cdot (Q/B)I \cdot DCij$) i , Bj =food j biomass;
2. Direct uptake from environment: $ui \cdot Bi \cdot Co$, where ui =parameter representing uptake per biomass per time, per unit environmental concentration, Bi =biomass, Co =environmental concentration;
3. Concentration in immigrating organisms: $ci \cdot Ii$, where ci = parameter (tracer per unit biomass in immigrating biomass), Ii = biomass of pool i immigrants per time;
4. Predation: $Ci \cdot Qij / Bi$, where Ci =concentration in pool i , Qij = consumption rate of type i organisms by predator type j , Bi = biomass in pool i ;
5. Detritus: $Ci \cdot MOi + (1-GCi) \cdot \sum j Cj \cdot Qji / Bj$, where MOi = non-predation death rate of type i (per year), GCi = fraction of food intake assimilated, Qji = intake rate if type j biomass by type i ;
6. Emigration: $ei \cdot Ci$, where ei = emigration rate (per year);
7. Metabolism: $di \cdot Ci$, where di = metabolism + decay rate for the material while in pool i .

In some EwE models of the assessed marine ecosystems and regions, the contaminant concentration in immigrating organisms ($ci \cdot Ii$) and emigration ($ei \cdot Ci$) were considered to be negligible (i.e., set to zero) for the purpose of the modeling work. In doing so, the equation is simplified as:

$$CiBi/dt = (Cj \cdot GCi \cdot Qji / Bj) + (ui \cdot Bi \cdot Co) - [(Ci \cdot Qij / Bi) + Ci \cdot MOi + ((1 - GCi) \cdot \sum j Cj \cdot Qji / Bj + di \cdot Ci)]$$

Appendix II Supporting References:

Christensen, V., & Walters, C.J. (2004). Ecopath with Ecosim: methods, capabilities and limitations. *Ecological Modelling*, 172(2-4): 109-139. <https://doi.org/10.1016/j.ecolmodel.2003.09.003>

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Christensen, V., Walters, C. J. & Pauly, D. (2005). Ecopath with Ecosim: a User's Guide. Fisheries Centre, University of British Columbia, Canada. 154 pp.

Christensen, V., Walters, C., Pauly, D., & Forrest, R. (2008). Ecopath with Ecosim Version6: User Guide. Lenfest Ocean Futures Project.

Coombs, A.P. (2004). Marine mammals and human health in the eastern Bering Sea: Using an ecosystem-based food web model to track PCBs. MSc Thesis, University of British Columbia, Vancouver, BC. 91p.

Booth, S. *et al.* (2016). Pollutants in the seas around us, in *Global Atlas of Marine Fisheries. A critical appraisal of catches and ecosystem impacts* (eds. Pauly, D. & Zeller, D.). Island Press, Washington, Covelo, London. pp. 152-170.

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Appendix III

Table A1. Data on Ecopath with Ecosim models (EwE) downloaded from Ecobase and microplastic concentration distribution retrieved from Litterbase.

Model #	EwE model geographical location: source	EwE model period	Published microplastic data available in Litterbase for marine coastal/oceanic regions matching EwE model geographical location	Authors	Publication year	Publication Title	Publication Link	Habitat studied	Microplastic quantity at location	Sampling year
1	Central and western Aleutians and of southeast Alaska (USA): Guénette S., Heymans S.J.J., Christensen V., Trites A.W. (2006). Ecosystem models show combined effects of fishing, predation, competition, and ocean productivity on Steller sea lions (<i>Eumetopias jubatus</i>) in Alaska. Canadian Journal of Fisheries and Aquatic Sciences 63(11): 2495-2517	1963-1977	Aleutian Islands, Alaska, Northeastern Pacific Ocean	Doyle, M. J., Watson, W., Bowlin, N. M., Sheavly, S. B.	2011	Plastic particles in coastal pelagic ecosystems of the Northeast Pacific Ocean	http://dx.doi.org/10.1016/j.marenvres.2010.10.001	surface	342337.1 items/km ²	2006
			North Pacific	Day, R.H., Shaw, D. G., Ignell, S. E.	1989	The quantitative Distribution and Characteristics of Neuston Plastic in the North Pacific Ocean, 1985-1988	https://books.google.ca/books?hl=en&lr=&id=pl5H8jtl7PoC&oi=fnd&pg=PA247&ots=2umyaMrpY_&sig=eIFyoTdPwpNTEGOLUP41p3bH7mo#v=onepage&q&f=false	surface	1990 items/km ²	1988

2	<p>Northeastern Pacific, British Columbia, Canada: Alava, J. J., Cisneros-Montemayor, A. M., Sumaila, U. R., Cheung, W. W. (2018). Projected amplification of food web bioaccumulation of MeHg and PCBs under climate change in the Northeastern Pacific. Scientific Reports 8(1): 13460. Model based on Field, J. C. (2004) Application of ecosystem-based fishery management approaches in the northern California Current. PhD thesis, University of Washington, Seattle, WA, USA. 408 pp.</p>	2005-2100	Northeastern Pacific Ocean	Desforges, J. W., Galbraith, M., Dangerfield, N., Ross, P. S.	2014	Widespread distribution of microplastics in subsurface seawater in the NE Pacific Ocean	http://dx.doi.org/10.1016/j.marpolbul.2013.12.035	pelagic		2012
3	<p>Northern Californian Current (USA): Field, J. C. (2004) Application of ecosystem-based fishery management approaches in the northern California Current. PhD thesis, University of Washington, Seattle, WA, USA. 408 pp.</p>	1990-2000	Northeastern Pacific Ocean	Lebreton, L. Slat, B. Ferrari, F. Sainte-Rose, B. Aitken, J. Marthouret, R. Hajbane, S. Cunsolo, S. Schwarz, A. Levivier, A. Noble, K. Debeljak, P. Maral,	2018	Evidence that the Great Pacific Garbage Patch is rapidly accumulating plastic	www.doi.org/10.1038/s41598-018-22939-w	surface	7.80e+04 items/km ²	2015

				H. Schoeneich-Argent, R. Brambini, R. Reisser, J.						
4	North Pacific (Kaloko Honokohau, Hawaii, USA): Wabnitz C.C.C., Balazs G., Beavers S., Bjorndal K.A., Bolten A.B., Christensen V., Hargrove S., Pauly D. (2010). Ecosystem structure and processes at Kaloko Honokohau, focusing on the role of herbivores, including the green sea turtle <i>Chelonia mydas</i> , in reef resilience Marine Ecology Progress Series 420: 27-44	2005	North Pacific Ocean	Uchida, K., Hagita, R., Hayashi, T., Tokai, T.	2016	Distribution of small plastic fragments floating in the western Pacific Ocean from 2000 to 2001	http://doi.org/10.1007/s12562-016-1028-2	surface	1.80e+03 items/km ²	2000-2001
5	Baja California Sur, Mexico: Cisneros-Montemayor A.M., Christensen V., Arreguín-Sánchez F., Sumaila R.U. (2012). Ecosystem models for management advice: An analysis of recreational and commercial fisheries policies in Baja California Sur, Mexico. Ecological Modelling 228: 8-16	1970	Northeastern Pacific Ocean, Southern California	Doyle, M. J., Watson, W., Bowlin, N. M., Sheavly, S. B.	2011	Plastic particles in coastal pelagic ecosystems of the Northeast Pacific Ocean	http://dx.doi.org/10.1016/j.marenvres.2010.10.001	surface		2006-2007

6	Northern Gulf of Mexico (USA): Sagarese, S. R., Laurretta, M. V., & Walter III, J. F. (2017). Progress towards a next-generation fisheries ecosystem model for the northern Gulf of Mexico. Ecological modelling, 345, 75-98.	2005-2009	Gulf of Mexico, northern Gulf of Mexico	Di Mauro, R., Kupchik, M. J., Benfield, M. C.	2017	Abundant plankton-sized microplastic particles in shelf waters of the northern Gulf of Mexico	https://doi.org/10.1016/j.envpol.2017.07.030	surface		2015
			Gulf of Mexico, northern Gulf of Mexico	Hoss, D. E., & Settle, L. R. (1990).	1990	Ingestion of plastics by teleost fishes.	In Proceedings of the Second International Conference on Marine Debris. NOAA Technical Memorandum . NOAA-TM-NMFS-SWFSC-154. Miami, FL (pp. 693-709). https://pdfs.semanticscholar.org/1d66/ofd9e28af0d581e0939af853f3769d7f6dd0.pdf	surface		1981
7	East China Sea Shelf: Li, Y., & Zhang, Y. (2012). Fisheries impact on the East China Sea Shelf ecosystem for 1969-2000. Helgoland Marine Research 66(3): 371-383. https://doi.org/10.1007/s10152-011-0278-8 East China Sea Shelf: Li, Y., Chen, Y., Olson, D., Yu, N., & Chen, L. (2009). Evaluating ecosystem	1969-2000	East China Sea Large Marine Ecosystem	Zhao, S., Zhu, L., Wang, T., Li, D.	2014	Suspended microplastics in the surface water of the Yangtze Estuary System, China: First observations on occurrence, distribution	http://dx.doi.org/10.1016/j.marpolbul.2014.06.032	surface		2013
				Zhao, S., Zhu, L., Li, D.	2015	Microplastic in three urban estuaries, China	http://dx.doi.org/10.1016/j.envpol.2015.08.027	surface		2013

	structure and functioning of the East China sea shelf ecosystem, China. Hydrobiologia, 636(1), 331–351. https://doi.org/10.1007/s10750-009-9964-9			Xu, P., Peng, G., Su, L., Gao, Y., Gao, L., Li, D.	2018	Microplastic risk assessment in surface waters: A case study in the Changjiang Estuary, China	https://doi.org/10.1016/j.marpolbul.2018.06.020	surface		2017
				Luo, W., Su, L., Craig, N. J., Du, F., Wu, C., Shi, H.	2019	Comparison of microplastic pollution in different water bodies from urban creeks to coastal waters	https://doi.org/10.1016/j.envpol.2018.11.081	surface		2017
				Qu, X., Su, L., Li, H., Liang, M., Shi, H.	2018	Assessing the relationship between the abundance and properties of microplastics in water and in mussels	https://doi.org/10.1016/j.scitotenv.2017.11.284	pelagic		2017
				Shim, W. J., Song, Y. K., Hong, S. H., Jang, M.	2016	Identification and quantification of microplastics using Nile Red staining	http://dx.doi.org/10.1016/j.marpolbul.2016.10.049	surface	1.15e+00 items/m ³	2016
				Song, Y. K. Hong, S. H. Eo, S. Jang, M. Han, G. M. Isobe, A. Shim, W. J.	2018	Horizontal and vertical distribution of microplastics in Korean coastal waters	www.doi.org/10.1021/acs.est.8b04032	surface	3.52e+02 items/m ³ 1.15e+03 items/m ³ 6.70e+02 items/m ³ 6.70e+02 items/m ³	2017

8	China Sea, North South China, Viet Nam: Cheung W.L. (2007). Vulnerability of marine fishes to fishing: from global overview to the Northern South China Sea. PhD Thesis Dissertation, University of British Columbia.	2000	South China Sea Large Marine Ecosystem	Tsang, Y. Y., Mak, C. W., Liebich, C., Lam, S. W., Sze, E. T. P., Chan, K. M., 2017	Microplastic pollution in the marine waters and sediments of Hong Kong	http://dx.doi.org/10.1016/j.marpolbul.2016.11.003	surface		2016
				Cai, M., He, H., Liu, M., Li, S., Tang, G., Wang, W., Huang, P., Wei, G., Lin, Y., Chen, B., Hu, J., Cen, Z., 2018	Lost but can't be neglected: Huge quantities of small microplastics hide in the South China Sea	https://doi.org/10.1016/j.scitotenv.2018.03.197	pelagic		2017
				Qu, X., Su, L., Li, H., Liang, M., Shi, H., 2018	Assessing the relationship between the abundance and properties of microplastics in water and in mussels	https://doi.org/10.1016/j.scitotenv.2017.11.284	pelagic		2017
				Tang, G., Liu, M., Zhou, Q., He, H., Chen, K., Zhang, H., Hu, J., Huang, Q., Luo, Y., Ke, H., Chen, B., Xu, X., Cai, M., 2018	Microplastics and polycyclic aromatic hydrocarbons (PAHs) in Xiamen coastal areas: Implications for anthropogenic impacts	https://doi.org/10.1016/j.scitotenv.2018.03.336	surface		2017

				Pan, Z., Liu, Q., Sun, Y., Sun, X., Lin, H.	2019	Environmental implications of microplastic pollution in the Northwestern Pacific Ocean	https://doi.org/10.1016/j.marpolbul.2019.06.031	surface		2018
9	Eastern Tropical Pacific: Olson R.J., Watters G.M.(2003). A model of the pelagic ecosystem in the eastern tropical Pacific Ocean. Inter-American Tropical Tuna Commission Bulletin 22(3): 135-248	1993-1997	Eastern Tropical Pacific (between 110°W and 150°W longitude)	Spear, L. B., Ainley, D. G., Ribic, C. A.	1995	Incidence of plastic in seabirds from the tropical pacific, 1984–1991: Relation with distribution of species, sex, age, season, year and body weight	http://dx.doi.org/10.1016/0141-1136(94)00140-K	pelagic	1.37e+02 items/km ² 2.40e+01 items/km ²	1988-1991
			Western Pacific Ocean	Uchida, K., Hagita, R., Hayashi, T., Tokai, T.	2016	Distribution of small plastic fragments floating in the western Pacific Ocean from 2000 to 2001	http://doi.org/10.1007/s12562-016-1028-2	surface	1.00e+03 items/km ²	2001
10	Darwin and Wolf Islands (Galápagos Islands): Ruiz DJ, Banks S, Wolff M. (2016) Elucidating fishing effects in a large-predator dominated system: The case of Darwin and Wolf Islands (Galápagos). Journal of Sea Research 107 (2): 1–11.	2006-2007	Galápagos Islands, Ecuador	Spear, L. B., Ainley, D. G., Ribic, C. A.	1995	Incidence of plastic in seabirds from the tropical pacific, 1984–1991: Relation with distribution of species, sex, age, season, year and body weight	http://dx.doi.org/10.1016/0141-1136(94)00140-K	pelagic	1.37e+02 items/km ² 2.40e+01 items/km ²	1988-1991

	Floreana Island's rocky reef (Galápagos Islands): Okey T.A.,Banks S.,Born A.F.,Bustamante R.H.,Calvopiña M.,Edgar G.J.,Espinoza E.,Fariña J.M.,Garske L.E.,Reck G.K.,Salazar S.,Shepherd S.,Toral-Granda V.,Wallem P.(2004). A trophic model of a Galápagos subtidal rocky reef for evaluating fisheries and conservation strategies. Ecological Modelling 172 (2-4): 383-401.	2000-2001		Spear, L. B., Ainley, D. G., Ribic, C. A.	1995	Incidence of plastic in seabirds from the tropical pacific, 1984–1991: Relation with distribution of species, sex, age, season, year and body weight	http://dx.doi.org/10.1016/o141-1136(94)00140-K	pelagic	1.37e+02 items/km ² 2.40e+01 items/km ²	1988-1991
	Bolivar Channel Ecosystem (Galápagos Islands): Ruiz, D., & Wolff, M. (2011). The Bolivar Channel Ecosystem of the Galapagos Marine Reserve: Energy flow structure and role of keystone groups. <i>Journal of Sea Research</i> , 66(2), 123-13	2004-2008		McMullen, K.	2023	McMullen, K. (2023). The Galapagos penguin as the “Canary in the Coal Mine” for Microplastic Research in the Galapagos Marine Reserve & Plastic Pollution Perceptions in Ecuadorian mangrove communities. MSc Thesis, University of British Columbia, Vancouver, BC. 205p.	N/A	pelagic	420 items/m ³ (0.00411 t/km ²)	2021

11	Low Barents Sea (Russian Federation): Blanchard, J.L., Pinnegar, J.K. and Mackinson, S. (2002). Exploring marine mammal-fishery interactions using 'Ecopath with Ecosim': modelling the Barents Sea ecosystem. Sci. Ser. Tech Rep., CEFAS Lowestoft, 117: 52pp.	1995-1996	Barents Sea, Arctic Central Basin	Lusher, A. L., Tirelli, V., O'Connor, I., Officer, R.	2015	Microplastics in Arctic polar waters: the first reported values of particles in surface and sub-surface samples	http://dx.doi.org/10.1038/srep14947	pelagic	2.68e+00 items/m ³ 3.40e-01 items/m ³	2014
				Kanhai, L. D. K., Gårdfeldt, K., Lyashevsk a, O., Hassellöv, M., Thompson, R. C., O'Connor, I.	2018	Microplastics in sub-surface waters of the Arctic Central Basin	https://doi.org/10.1016/j.marpolbul.2018.03.011	surface	Closest data point to Barents Sea: 2.50e+00 items/m ³ ; Average = 26.30 items/m ³	2016
12	Grand Banks of Newfoundland (North Atlantic, Canada): Heymans J.J. (2003). Ecosystem models of Newfoundland and Southeastern Labrador: Additional information and analyses for 'Back to the Future' Fisheries Centre Research Reports	1990-1997	The Gully' on the continental slope, Nova Scotia, Canada	Dufault, S., Whitehead, H.	1994	Floating marine pollution in 'the Gully' on the continental slope, Nova Scotia, Canada	http://dx.doi.org/10.1016/0025-326X(94)90522-3	surface	8.48e+04 items/km ² 4.95e+04 items/km ²	1990

13	North Atlantic: Vasconcellos, M. and R. Watson 2004 Mass balance models of oceanic systems in the Atlantic. pp. 171- 214 In Palomares, M.L.D. Pauly, D. (eds.) West African marine ecosystems: models and fisheries impacts. Fisheries Centre Research Reports 12(7). Fisheries Centre, UBC, Vancouver.	1997- 1998	North Atlantic Subtropical Gyre (NASG)	Kooi, M., Reisser, J., Slat, B., Ferrari, F. F., Schmid, M. S., Cunsolo, S., Brambini, R., Noble, K., Sirks, L. A., Linders, T. E. W., Schoeneic h-Argent, R. I., Koelmans, A. A.	2016	The effect of particle properties on the depth profile of buoyant plastics in the ocean	http://doi.org/10.1038/srep33882	surface	2.50e+00 items/m ³	2015
			North Atlantic Subtropical Gyre (NASG)	Enders, K., Lenz, R., Stedmon, C. A., Nielsen, T. G.	2015	Abundance, size and polymer composition of marine microplastics ≥ 10 µm in the Atlantic Ocean and their modelled vertical distribution	http://dx.doi.org/10.1016/j.marpolbul.2015.09.027	pelagic		2014
			North Atlantic Subtropical Gyre (NASG): North-Western Atlantic Ocean	Branch, L., Deixonne, P., Bernard, M. F., Durand, E., Desjean, M. C., Perez, E., van Sebille, E., ter Halle, A.	2018	Anticyclonic eddies increase accumulation of microplastic in the North Atlantic subtropical gyre	https://doi.org/10.1016/j.marpolbul.2017.10.077	surface		2015

14			the North Atlantic Subtropical Gyre	Ter Halle, A., Jeanneau, L., Martignac, M., Jardé, E., Pedrono, B., Brach, L., Gigault, J.	2017	Nanoplastic in the North Atlantic Subtropical Gyre	http://dx.doi.org/10.1021/aacs.est.7b03667	surface	10000-7000000 items/km ²	2015
			North Atlantic	Debroas, D., Mone, A., Ter Halle, A.	2017	Plastics in the North Atlantic garbage patch: A boat-microbe for hitchhikers and plastic degraders	https://doi.org/10.1016/j.scitotenv.2017.05.059	surface		
	Central Atlantic: Vasconcellos M., Watson R.(2004). Mass-balance models of oceanic systems in the Atlantic West African marine ecosystems: models and fisheries impacts. pp 171-214. In Palomares M.L.D., Pauly D.	1990-1991	Atlantic Ocean	Enders, K., Lenz, R., Stedmon, C. A., Nielsen, T. G.	2015	Abundance, size and polymer composition of marine microplastics $\geq 10 \mu\text{m}$ in the Atlantic Ocean and their modelled vertical distribution	http://dx.doi.org/10.1016/j.marpolbul.2015.09.027	pelagic		2014
			Northeastern Brazil	Lima, A. R. A., Barletta, M., Costa, M. F., Ramos, J. A. A., Dantas, D. V., Melo, P. A. M. C., Justino, A. K. S., Ferreira, G. V. B.	2015	Changes in the composition of ichthyoplankton assemblage and plastic debris in mangrove creeks relative to moon phases	https://online.library.wiley.com/doi/epdf/10.1111/jfb.12838	surface		2008

			Fernando de Noronha, Abrolhos and Trindade (western tropical Atlantic Ocean), Brazil	Ivar do Sul, J. A., Costa, M. F., Fillmann, G.	2014	Microplastics in the pelagic environment around oceanic islands of the Western Tropical Atlantic Ocean	http://dx.doi.org/10.1007/s11270-014-2004-z	surface		2013
15	North Western Mediterranean Sea: Valls A., Gascuel D., Guenette S., Francour P. (2012). Modeling trophic interactions to assess the effects of a marine protected area: case study in the NW Mediterranean Sea Marine Ecology Progress Series 456: 201–214.	1998-2008	Mediterranean Sea, North Western Mediterranean Sea	Collignon, A., Hecq, J. H., Galgani, F., Voisin, P., Collard, F., Goffart, A.	2012	Neustonic microplastics and zooplakton in the North Western Mediterranean Sea	http://dx.doi.org/10.1016/j.marpolbul.2012.01.011	surface		2010
16a	East Brazil Large Marine Ecosystem, Brazil: de Meirelles Felizola Freire K. (2005). Fishing impacts on marine ecosystems off Brazil, with emphasis on the northeastern region. PhD Thesis Dissertation, University of British Columbia. Model is not available (N/A) in Ecobase	1970-1970	Northeastern Brazil	Lima, A. R. A., Barletta, M., Costa, M. F., Ramos, J. A. A., Dantas, D. V., Melo, P. A. M. C., Justino, A. K. S., Ferreira, G. V. B.	2015	Changes in the composition of ichthyoplankton on assemblage and plastic debris in mangrove creeks relatively to moon phases	https://online.library.wiley.com/doi/epdf/10.1111/jfb.12838	surface		2008
			Fernando de Noronha, Abrolhos and Trindade (western tropical Atlantic Ocean), Brazil	Ivar do Sul, J. A., Costa, M. F., Fillmann, G.	2014	Microplastics in the pelagic environment around oceanic islands of the Western Tropical	http://dx.doi.org/10.1007/s11270-014-2004-z	surface		2013

						Atlantic Ocean				
16b	Southern Brazil, Brazil: Vasconcellos M., Gasalla M.A. (2001). Fisheries catches and the carrying capacity of marine ecosystems in southern Brazil. Fisheries Research. pp 279-295	1980-1981	Guanabara Bay, southeastern Brazil,	Figueiredo, G. M., Vianna, T. M. P.	2018	Suspended microplastics in a highly polluted bay: Abundance, size, and availability for mesozooplankton	https://doi.org/10.1016/j.marpolbul.2018.07.020	pelagic	5.10e+00 items/m ³ 25500000 items/km ²	2016
			Guanabara Bay, Rio de Janeiro, Brazil (Brazilian southern coast)	Olivatto, G. P., Martins, M. C. T. Montagne, C. C., Henry, T. B., Carreira, R. S.	2019	Microplastic contamination in surface waters in Guanabara Bay, Rio de Janeiro, Brazil	https://doi.org/10.1016/j.marpolbul.2018.12.042	surface	4.10e+00 items/m ³ 5.55e+00 items/m ³ 1.16e+01 items/m ³ Geometric mean = 32073642 items/km ²	2016
17	Southern Bengala (Namibia and South Africa, Africa): Shannon L.J., Moloney C.L., Jarre A., Field J.G. (2003). Trophic flows in the southern Benguela during the 1980s and 1990s Journal of Marine Systems 39 (1-2): 83-116.	1978-1979	Southeastern African coastline (South Africa, Africa)	Nel, H. A., Froneman, P. W.	2015	A quantitative analysis of microplastic pollution along the south-eastern coastline of South Africa	http://dx.doi.org/10.1016/j.marpolbul.2015.09.043	pelagic		2014
			Southern African coastline (Africa)	Nel, H.A., Hean, J.W., Noundou, X.S., Froneman, P.W.	2017	Do microplastic loads reflect the population demographics along the southern African coastline?	10.1016/j.marpolbul.2016.11.056	surface		2016

18	<p>Sri Lanka: Haputhantri S.S.K., Villanueva M.C.S., Moreau J. (2008). Trophic interactions in the coastal ecosystem of Sri Lanka: An ECOPATH preliminary approach Estuarine, Coastal and Shelf Science. pp 304-318</p>	2000-2001	Indian Ocean, Sri Lanka southern coast	<p>Bimali Koongolla, J., Andrad, A. L., Terney Pradeep Kumara, P. B., Gangabadi, C. S.</p>	2018	Evidence of microplastics pollution in coastal beaches and waters in southern Sri Lanka	<p>https://doi.org/10.1016/j.marpolbul.2018.10.031</p>	surface		2018
19a	<p>Gulf of Carpentaria (Australia): Okey T.A., Griffiths S., Pascoe S., Kenyon R., Miller M., Dell Q., Pillans R., Buckworth R.C., Gribble N., Engstrom N., Bishop J., Milton D., Salini J., Stevens J. (2007). The effect of illegal foreign fishing on the ecosystem in the Gulf of Carpentaria: management options and downstream effects on other fisheries Final Report to Australian Fisheries Management Authority Project 2006/825</p>	1990-1991	Gulf of Carpentaria, Australia	<p>Reisser, J., Shaw, J., Wilcox, C., Hardesty, B. D., Proietti, M., Thums, M., Pattiaratchi, C.</p>	2013	Marine Plastic Pollution in Waters around Australia: Characteristics, Concentrations, and Pathways	<p>http://dx.doi.org/10.1371/journal.pone.0080466</p>	surface		2012

19b	Australia North West Shelf: Bulman C. (2006). Trophic webs and modelling of Australia's North West Shelf. CSIRO. Marine and Atmospheric Research. North West Shelf Joint Environmental Management Study (NWSJEMS) Technical Report 9	1986-1991	Australia North West Shelf (Australia)	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	http://dx.doi.org/10.1371/journal.pone.0080466	surface		2012
20	Central Chile, Chile: Neira S., Arancibia H., Cubillos L. (2004). Comparative analysis of trophic structure of commercial fishery species off Central Chile in 1992 and 1998 Ecological Modelling. pp 233-248	1998-1999	South Pacific subtropical gyre	Eriksen, M., Maximenko, N., Thiel, M., Cummins, A., Lattin, G., Wilson, S., Hafner, J., Zellers, A., Rifman, S.	2013	Plastic pollution in the South Pacific subtropical gyre	http://dx.doi.org/10.1016/j.marpolbul.2012.12.021	surface	1486.58 items/km ² 817.62 items/km ² 605.65 items/km ² ; Geometric mean = 903 items/km ²	2011
21	Antarctic Peninsula, Antarctic: Hoover C. (2009). Ecosystem effects of climate change in the Antarctic Peninsula Ecopath 25 years conference proceedings: Extended abstracts. pp 96-97. In Palomares M.L., Morissette L., Cisneros-Montemayor A., Varkey D., Coll M., Piroddi C.	1970-1971	Antarctic Peninsula (Rothera Research Station, Antarctica)	Reed, S., Clark, M., Thompson, R., Hughes, K. A.	2018	Microplastics in marine sediments near Rothera Research Station, Antarctica	https://doi.org/10.1016/j.marpolbul.2018.05.068	pelagic		2016

Glossary

Microplastics: plastic particles less than 5mm in size, a class of contaminants, including numerous chemical compositions.

Nanoplastics: plastic particles ranging 1nm to 1000nm (1 μ m) in size, a class of contaminants, including numerous chemical compositions

Microplastic bioaccumulation / microplastic accumulation: the gradual increase of total microplastic particles in biota throughout the life span of an organism.

Microplastics trophic magnification / biomagnification: the significant increase of microplastic particle intake, trophic transfer and amplification as trophic level increases in the foodweb.

Predator-prey biomagnification factor (BMF_{TL}): To investigate biomagnification in predators (e.g., marine mammals) relative to prey items and to assess the effect of the magnitude of trophic level differences on this biomagnification index, the predator-prey biomagnification factor (BMF_{TL}) is applied as a practical tool to assess preliminary biomagnification potential of microplastics in predators (Alava 2020). The criterion applied to indicate the capacity of microplastics to biomagnify was a $BMF_{TL} > 1$, while a $BMF_{TL} < 1$ is an indication of lack of biomagnification capacity (see Alava 2020). If the BMF is statistically greater than 1, then it indicates that a contaminant is a probable bioaccumulative contaminant (Gobas et al. 2009). Following this approach, the concentration of microplastics projected in selected predators was divided by predicted concentration in the prey. Thus, the model-based predator-prey biomagnification factor normalized to trophic position (i.e., $BMF_{TROPIC LEVEL}$: BMF_{TL}) is calculated using the following equation (Borga et al., 2004):

$$BMF_{TL} = \frac{(C_{PREDATOR}/C_{PREY})}{TL_{PREDATOR} - TL_{PREY}}$$

Where $C_{PREDATOR}$ and C_{PREY} are the microplastic concentrations in the predator and prey, expressed in units of mass (g per kg of predator) and the concentration in prey (g per kg of prey); and, $TL_{PREDATOR}$ and TL_{PREY} are the trophic levels of the predator and prey. The BMF_{TL} values were used to measure biomagnification in cetaceans' food chains between two adjacent trophic levels (i.e., the difference in TL between predator and prey is small), assuming steady state in microplastic concentrations between predator and prey, as reported in Alava (2020).

The trophic magnification factor (TMF): is a food web biomagnification metric that is often used to investigate the biomagnification of pollutants at each trophic level in an entire food web (Borga et al., 2012; Conder et al., 2012; Walters et al., 2016). This approach was applied to further assess the microplastic biomagnification potential in the entire marine food web (Alava 2020). The TMF is calculated as the antilog of the regression slope of the linear regression between the logarithmic-transformed concentrations of microplastics (Log MPs) predicted in the GI tract of organisms of the food web and their respective trophic levels, TL (Alava 2020), i.e. $\text{Log [MP]} = a + bTL$, which in the equivalent exponential mathematical terms is expressed as $TMF = 10^b$, where b is the slope.

The TMF (slope, b) is statistically evaluated using a significance level (α) of 0.05. A $TMF > 1$ ($b > 0$) indicates that the contaminant biomagnifies in the foodweb. A $TMF < 1$ ($b < 0$) indicates trophic dilution of the contaminants, while a $TMF=1$ ($b=0$) indicates no change in contaminant concentrations among organisms of a food web (Borga et al., 2012).

Kinetic Food web model: considers the main kinetic mechanisms for bioaccumulation with associated variables and parameters, summarized as follows:

Basic kinetic bioaccumulation model for microplastics (MP):

[Intake rate] times [food MP-concentration] minus [elimination rate] times [organism MP-concentration]
Or

[Intake rate] times [prey MP-concentration] minus [elimination rate] times [predator MP-concentration]

- Microplastic concentration data in environmental compartments (abiotic matrices: seawater and sediment) under low, moderate and high concentration scenarios
- Dietary preferences (% diet matrix) and trophic levels of food web organisms or species
- Dietary uptake/intake rate constant (k_D) of species (trophic level) from existing literature
- Calculated egestion/elimination rate constant (k_E) from documented data for the retention time (t_r) of microplastics in the digestive system or GI tract in marine animal species ($t_r = 1/k_E$; then, $k_E = 1/t_r$).

Trophic dynamic food web model: reflects the intrinsic mechanistic outputs of the transferring of energy and biomass flowing throughout the species and/or species' functional groups of the food web, from prey to predators (i.e., the mortality of prey is survival for predator), by mass balancing the consumption and biomass production flow in the ecosystem.

Microplastics exposure / exposure-risk / degree of exposure: the number of possible interactions with microplastics throughout an organism's daily activities. It does not include the implications of said exposure levels.

Human Health / public health: According to The World Health Organization, health is a state of complete physical, mental and social well-being and not merely the absence of disease or infirmity (WHO, 1946). Public health is the science of protecting and improving the health of people and their communities. This work is achieved by promoting healthy lifestyles, researching disease and injury prevention, and detecting, preventing and responding to infectious diseases. Overall, public health is concerned with protecting the health of entire populations. These populations can be as small as a local neighborhood, or as big as an entire country or region of the world. [As defined by the Center for Disease Control Foundation. (2023) <https://www.cdcfoundation.org/what-public-health>]. The Public Health Agency of Canada defines public health as activities focused on preventing disease and injuries, responding to public health threats, promoting good physical and mental health, and providing information to support informed decision making. [As defined on the Government of Canada. (2023). <https://www.canada.ca/en/public-health.html>].

Toxicity: a dose or quantity/quantities that are harmful or toxic to an organism's physiological, chemical, biological, or behavioural health.

Local-fishing zone / community-fishing zone: incorporates a 120 km buffer surrounding a coastal community or around each Indigenous coastal community.

Indigenous peoples / Aboriginal and native communities: "Indigenous peoples" is a collective name for the original peoples of North America and their descendants. Often, "Aboriginal peoples" is also used. The Canadian Constitution recognizes 3 groups of Aboriginal peoples: Indians (more commonly referred to as First Nations), Inuit and Métis. These are 3 distinct peoples with unique histories, languages, cultural practices and spiritual beliefs. [As defined by the Government of Canada. (2002); <https://www.rcaanc-cirnac.gc.ca/eng/1100100013785/1529102490303>]

Inequity gap framework: Within the context of marine plastic pollution for this document, it is defined as the lack the information necessary to implement equitable solutions that do not further disadvantage the marginalized, the disempowered, and those otherwise unable to equally access or benefit from oceans. For instance, inequitable impacts of uneven burdens plastic waste and marine plastic pollution extended across social, political, and economic contexts and effects that disproportionately affect people of color and low-income communities, where burdens of responsibility are placed on other stakeholders. The burden of these impacts is often disproportionately experienced by communities who are marginalized and most vulnerable to the impacts of plastic pollution. [Adapted from and as defined by Vandenberg & Ota. (2022). <https://oceanx nexus.uw.edu/equity-marine-plastic-pollution-report/>].

Environmental justice and equity dimensions: Environmental justice refers broadly to the distribution of environmental benefits and burdens, and the fair treatment and meaningful involvement of all people in environmental decision making and legal frameworks. The field of environmental justice initially developed out of a concern for the disproportionate distribution and impacts of environmental pollution and hazardous waste disposal on groups that have been historically and structurally marginalized, including Indigenous, and People of Color populations and socio-economically disadvantaged communities. More recent environmental justice scholarship has expanded geographically and focused on a broader set of environmental hazards and harms, such as climate change impacts, environmental pollution, biodiversity and habitat loss, and ecosystem service declines. [As defined by Bennett et al., 2022. <https://doi.org/10.1016/j.marpol.2022.105383>].

There is no universally agreed upon definition of equity (Campbell and Hanich, 2015), but according to the report on "*Towards an Equitable Approach to Marine Plastic Pollution*" by Vandenberg & Ota (2022), the term equity refers to fair or just treatment among individuals or groups (Law et al., 2018). The environmental management literature increasingly recognizes equity as a multidimensional concept that includes distributional, procedural, recognition, and contextual dimensions (Friedman et al., 2018; Law et al., 2018; Pascual et al., 2014; McDermott et al., 2013). [As defined by Vandenberg & Ota (2022) <https://oceanx nexus.uw.edu/equity-marine-plastic-pollution-report/>].

Global ocean pollution footprint of microplastics: marine and coastal areas and fishing zones of the global ocean impacted by anthropogenic plastics of all sizes, including macroplastics, microplastics, and nanoplastics.

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