The Effects of Plastic Pollution on Arctic Biota

AMAP

Arctic Monitoring and Assessment Programme (AMAP)

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Contents

Preface

The Arctic Monitoring and Assessment Programme (AMAP) is the Working Group of the Arctic Council that monitors and assesses the status of the Arctic region with respect to pollution and climate change and their effects on ecosystems and human health. The Arctic Council Ministers have requested AMAP to:

- produce integrated assessment reports on the status and trends of Arctic ecosystems;
- identify possible causes for the changing conditions;
- detect emerging problems, their possible causes, and the potential risk to Arctic ecosystems including Indigenous Peoples and other Arctic residents; and
- recommend actions required to reduce risks to Arctic ecosystems.

This analysis of the effects of plastic pollution on Arctic biota was conducted between 2021 and 2025 by an international group of experts. The expert group members and lead authors were appointed following an open nomination process coordinated by AMAP. A similar process was used to select international experts who independently reviewed this report. Information contained in this report is fully referenced and based first and foremost on results of research and monitoring. Care was taken to ensure that any critical probability statements made in this assessment were based only on peer-reviewed materials.

This report provides the accessible scientific basis and validation for statements and recommendations made in related derivative products, including the *Effects of Plastic Pollution on Arctic Animals. Summary for policy-makers* report that was delivered to the 14th Meeting of the Arctic Council in May 2025. The present report includes background data and references to scientific literature, and details the sources for analyses and graphics reproduced in derivative products. Whereas the related summary for policy-makers contains recommendations that focus on policy-relevant actions, the conclusions and recommendations presented in this report also cover issues of a more scientific nature, such as proposals for filling gaps in knowledge, and recommendations relevant to future research and monitoring work.

AMAP would like to express its appreciation to all experts who have contributed their time, efforts and data, in particular the lead authors who coordinated the production of this report. Thanks are also due to Carolyn Mallory for her work as scientific editor to the report and the reviewers who contributed to the peer-review process and provided valuable comments that helped to ensure the quality of the report. A list of contributors is included in the acknowledgments at the start of this report and lead authors are identified at the start of each chapter. The acknowledgments list is not comprehensive. Specifically, it does not include the many national institutes, laboratories and organizations, and their staff that have been involved in plastics-related research and monitoring. Apologies, and no lesser thanks are given to any individuals unintentionally omitted from the list.

The support from the Arctic nations and Permanent Participants and from non-Arctic nations and organizations that implement research and monitoring in the Arctic is vital to the success of AMAP. AMAP work relies heavily on ongoing activities of these nations and organizations, including on the lands or territories of Indigenous Peoples. The nations, Permanent Participants, and other countries and organizations also provide the necessary support for most of the experts involved in the preparation of the AMAP reports. In particular, AMAP would like to acknowledge Canada and Norway for their lead role for this report, and thank Canada, Norway and the Nordic Council of Ministers for their financial support of this work.

All AMAP scientific and assessment reports are freely available from the AMAP Secretariat and on the AMAP website: www.amap.no, and their use for educational purposes is encouraged.

The AMAP Working Group is pleased to present this report to the Arctic Council and the international science community.

Jennifer Provencher (Expert Group co-lead, Canada)

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Sarah Kalhok Bourque (AMAP Chair)

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Tromsø, February 2026

Introduction

Authors: E. Farmen, F. Collard, B. Hamilton, D. Herzke, A. Lusher, T. Kögel, J. R. Larsen, M. L. Mallory, I. L. Pollet, J. F. Provencher, and K. Vorkamp

Background

Plastic production has rapidly increased since the 1950s (Geyer et al. 2017), leading to high plastic emissions, which are projected to continue to increase (Borrelle et al. 2020). Plastic pollution of both plastic debris and microplastics (< 5 mm) is a global concern, including the widespread occurrence of plastics in the Arctic environment (Zhang et al. 2020, Collard and Ask 2021, Bergmann et al. 2022, Citterich et al. 2023, Martin et al. 2023).

Multiple studies have documented the presence of plastics both in the abiotic environment of the Arctic and in Arctic fauna (Cau et al. 2019, Collard and Ask 2021, Martin et al. 2021, Bergmann et al. 2022). Plastic pollution is a complex problem because plastic varies in size, morphology, and chemical constituents (Rochman et al. 2019). Plastic can elicit various effects: it can deleteriously affect an animal through entanglement (Butterworth 2016, Ryan 2018, Lusher et al. 2022) or through ingestion or inhalation (e.g., Cózar et al. 2014, Unger et al. 2016, Derraik 2002). For example, marine species, including pinnipeds (Jepsen and Nico de Bruyn 2019) and seabirds (Ryan 2018), have been found entangled in plastic. Ingested plastic can affect organismal health by causing starvation (Pierce et al. 2004), internal damage, and plastic-related fibrosis, or "plasticosis," in tissues (Charlton-Howard et al. 2023). Moreover, smaller plastics, such as microplastics, can cause an array of effects, including oxidative stress in some animals (Solomando et al. 2020). Plastics contain chemical additives such as plasticisers, flame retardants, or UV-filters that have been shown to have a range of toxicological effects, such as endocrine disruption, and they can also cause oxidative stress and cytotoxicity (Zimmermann et al. 2019). Plastics contain numerous other intentionally and non-intentionally added chemicals, potentially harmful to the environment and to human health.

Indicators for plastic pollution have been called for at the global level to improve the understanding of the environmental fate of plastic particles by reliably tracking quantities of plastic pollution in the environment over time and space, as well as quantifiying if plastic reduction measures (i.e. actions/policy changes/laws) are effective in reducing plastic pollution. The Arctic Monitoring and Assessment Programme (AMAP) proposed a set of first and second priority indicators for circum-Arctic monitoring, identified by AMAP's Litter and Microplastic Expert Group (LMEG; AMAP 2021). These indicators included beach litter, microplastics in water and sediment, and ingested plastics in seabirds as first priority indicators (LMEG; AMAP 2021b). These monitoring activities have the primary goal of providing data on plastic occurrence in the Arctic that are comparable over space and time, allowing the assessment of the state of pollution and the effectiveness of counteractions.

Work on indicators is also ongoing in other organizations and fora, such as the Northern Contaminants Program (Canada), the Oslo/Paris Convention for the Protection of the Marine Environment of the North East Atlantic (OSPAR), the International Council for the Exploration of the Sea (ICES), the European Union Marine Strategy Framework Directive (MSFD), etc. Therefore, coordination and information sharing are priorities to further advance our understanding of the topic.

Negative biological effects from plastic pollution in the Arctic have been considered in some scientific studies, as well as in a few reviews such as the Desktop Study on Marine Litter by the Arctic Council Working Group on the Protection of the Arctic Marine Environment (PAME; PAME 2019). However, no common indicators for effects have been proposed or implemented to date in the Arctic region, or elsewhere, to specifically measure biological effects of plastic pollution. The main bottleneck for proposing effect indicators is related to a lack of data on occurrence, susceptibility, and most relevant toxicological endpoints. The Northern Fulmar (*Fulmarus glacialis*) is an important monitoring species in the OSPAR region and has also been proposed for plastic monitoring under AMAP. However, the occurrence of plastic in fulmar stomachs is an environmental indicator of the level of plastic pollution, not of its effects.

In 2022, the United Nations (UN) endorsed a resolution to develop a legally binding agreement to end global plastic pollution. An Intergovernmental Negotiating Committee (INC) was established to work toward a global treaty. Effects on marine and coastal species, including their entanglement and ingestion of plastics, are mentioned as one of the major threats associated with plastic pollution. Besides climate change and pollution, nature loss contributes to the triple planetary crisis. Therefore, the effects on animal health are likely to be included in the final text of the global agreement.

Subsequent to the development of indicators and guidelines for plastic monitoring, the second phase of AMAP LMEG's work aimed to review the biological effects of plastic pollution on Arctic animals with a view to potential future monitoring and assessments of negative ecosystem effects. This report summarizes the state of knowledge on negative biological effects of plastic litter and microplastic particles on Arctic animals and analyzes the current gaps in our knowledge in this field. Based on this gap analysis, LMEG proposes future work on indicators for plastic effects, including recommendations for future research to support the further development of biological indicators of these effects.

The report addresses the following:

1. Effects on biota from entanglement in large plastic items (macrolitter);
2. Physical effects on biota caused by the ingestion of plastics;

3. Physiological effects on biota caused by the uptake of micro- and nanoplastics; and
4. Physiological effects on biota caused by exposure to chemical additives in plastics.

Arctic animals can be affected by pollution in several ways. Effects can occur at the individual or at the population level, and both must be considered to understand the impact plastic pollution may have on the environment. Second, these effects can be both direct (e.g., entanglement in plastics that leads to an animal drowning), or indirect (e.g., an entangled bird cannot dive properly to forage, which leads to starvation). Additionally, toxicity, inflammation, and mild entanglement can have a cost in the dynamic energy budget of an organism. Therefore, sublethal impacts can increase the susceptibility of lifeforms for other adversities, such as climate change resilience, infections, or predator performance.

In Chapter 1, the authors focus on the entanglement of animals in plastic waste, typically macroplastics. Unlike some of the other impacts, entanglement is a threat that is widely documented to directly affect individuals, i.e., seabird chicks at their nest or fish and marine mammals in derelict fishing gear at sea. Although individual occurrences are well documented and reported, it is difficult to measure the effects at a population level or on a broader geographical scale. To assess the potential population-level impacts, both the number of animals that die or are impaired each year from plastic entanglement and the total population size is needed.

In Chapter 1, we summarized the effects of litter entanglement on marine mammals, including pinnipeds, seabirds, fish, and benthic communities. We explored the different available databases that could help us estimate the effects of entanglement in plastic at a population level. No one dataset perfectly fits this need, and at this point, quantitative analyses are not possible due to a lack of comparable methods in sampling and reporting. The low human population in the Arctic is a barrier to the systematic recording of entanglement. We advocate for a cross-promotional approach to improve the collection of information on litter entanglement in the Arctic.

One of the better-known threats of plastic pollution globally is ingestion by biota, in which ingested plastic debris can cause physical damage to the digestive tract (e.g., de Stephanis et al. 2013, Kühn and van Franeker 2020) or have other adverse effects such as absorption and translocation of plastic particles and associated chemicals. Chapter 2 focuses on the evidence of physical damage from ingested plastic debris. Plastic debris is found in many types of Arctic animals, from small invertebrates to whales (reviewed in Collard and Ask 2021), although sampling for plastic ingestion remains limited for Arctic species and across geographic regions. As of 2023, there was only one report of physical damage from plastic ingestion in Arctic wildlife (Tulatz et al. 2023), and one separate study that suggests plastic debris has had deleterious effects on the microbiome of Arctic seabirds (Fackelmann et al. 2023). The chapter summarizes the large gaps in knowledge faced in assessing effects of plastic ingestion in the Arctic, which make it difficult to draw conclusions about the extent and severity of these impacts. The chapter also notes the challenges with testing for physical effects, i.e., this principally requires dead organisms, and lethal sampling is a logistic and ethical challenge for many areas.

In chapter 3, the effects from ingested plastics are expanded to include physiological effects caused by microplastics. However, most studies on Arctic animals focused on the presence rather than the effects of microplastics. No effect studies on Arctic mammals or birds were available. For birds, studies from outside the Arctic showed damage to stomach lining and inflammation of spleen, liver, and stomach glands, in correlation with microplastic occurrence. Effect studies on fish indicated that long exposure times may affect lipid storage, hormone pathways, egg production, sexual maturation, and malformation. Translocation of microplastic particles from the gastrointestinal tract to muscle tissue was also shown in Arctic fish. Microplastics also occur in an array of Arctic invertebrates, and physiological effects were reported for laboratory studies with invertebrates from plankton to lobster. However, some of these studies used microplastic concentrations that exceeded those found in environmental studies. For blue mussels, concentrations that cause effects were listed, e.g., for malformations, neurotoxicity, growth, and immune deficiency. The chapter identifies data gaps currently preventing a risk assessment of microplastics, such as those on plastic particle concentrations in tissues and on the presence of smaller particle sizes including in subsistence foods such as reindeer and birds. The chapter highlights that different methods were applied, which can lead to biased results and recommends mitigation steps.

Plastic chemical additives are intentionally added to polymers during the manufacturing process. These plastic additives are not chemically bound to the polymer matrix and therefore are able to migrate or leach into the surrounding environment. In Chapter 4, we summarized the current state of knowledge pertaining to plastic chemical additive effects in Arctic biota across multiple levels of biological organization. Currently, knowledge on the effects of plastic additives in Arctic biota is lacking. The majority of published data, which is minimal, focuses on the occurrence of plastic additives. Chapter 4 highlights these knowledge gaps and provides a framework for future monitoring and research priorities. As we begin to understand the chemical effects of plastic pollution, it is important to assess effects from an Arctic lens to better understand impacts at an organismal, population, and community level across the circumpolar North.

The following chapters reflect the state of the science and literature as of the end of March 2024. As the field of knowledge continues to grow, LMEG will aim to update this work on a regular basis as part of its workplan under AMAP.

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1. Biological Effects of Litter Entanglement

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1.0 Introduction

Plastic is present everywhere, and interaction with wildlife takes multiple forms: colonization, ingestion, and entanglement, each of which have been reviewed in detail several times (Jepsen and de Bruyn 2019, Collard and Ask 2021, Bergmann et al. 2022). Plastic litter serves as a vector for chemical and biotic transport (Werner et al. 2016). Entanglement events occur via different mechanisms. Biota can encounter floating litter, but also abandoned, lost, and derelict fishing gear (ALDFG) in rivers or at sea, and litter that has washed ashore. Entanglement can happen at any level of the water column from the surface to the sea floor (Tekman et al. 2017, Alfonso and Fidelis 2023). Birds can build their nests using plastic materials in which either breeding adults, or more often juveniles, may get entangled (Votier et al. 2011, Bond et al. 2012).

Numerous taxa are vulnerable to entanglement (Laist 1997, Gregory 2009, Gall and Thompson 2015, Kühn et al. 2015, Rodríguez et al. 2022). In the Arctic, entanglement records include at least 12 marine mammal species, 3 terrestrial mammal species, and 24 seabird species (Appendix A).

For many marine mammal species, individuals can get entangled in active fishing gear and swim away with some gear still attached. It is then difficult to differentiate between bycatch and entanglement in ALDFG (e.g., Moore et al. 2009). Importantly, these two types of entanglement are often considered separately due to the policy implications, however, entanglement in ALDFG is not considered bycatch, but marine litter (Smolowitz 1978, Davies et al. 2009).

Systematic monitoring of entanglement could be carried out in locations with high animal density, such as seabird colonies and pinniped haul-out sites and colonies (Phillips et al. 2010, Allen et al. 2012). For example, due to their large nests, Northern Gannets (*Morus bassanus*) are commonly used to assess marine debris and entanglement at their colonies (Votier et al. 2011, O'Hanlon et al. 2019), and the amount of fishing gear in their nests is related to the fishing effort surrounding their colonies (Bond et al. 2012). However, Northern Gannets can also be observed at sea for entanglement (Barrett 1988, Rodríguez et al. 2013) or when dead birds are washed ashore (Camphuysen 2001). In the Arctic, much monitoring has been done regarding plastic ingestion in seabirds, specifically in Northern Fulmar (*Fulmarus glacialis*), but entanglement is mostly limited to opportunistic observations of marine mammals (PAME 2019, Baak et al. 2020).

1.1 Effects of litter entanglement

From an individual standpoint, entanglement has different effects depending on the concerned species. Along with bycatch, entanglement is the main reason whales get injured and die due to human activities (Basran and Rasmussen 2020). In pinnipeds and marine mammals, entanglement in marine debris could lead to lacerations (Delean et al. 2020). Subsequently, those wounds could result in infection and amputation (Barreiros and Raykov 2014). In many species, entanglement results in a lack of mobility, increased energy requirements, and reduced foraging capacity (Feldkamp et al. 1989, Cassoff et al. 2011). Entanglements can also increase the probability of being preyed upon and drowning (Allen et al. 2012, Knowlton et al. 2012, Stelfox et al. 2016).

Seabird entanglement leads to drowning or starvation of individuals, either at sea or on land (Derraik 2002, Phillips et al. 2010). Entanglement can also occur at the colonies, and chicks have been reported entangled in materials used to build the nests (Votier et al. 2011). Finally, marine litter on beaches could be a problem for both coastal bird species and migrating birds that use this habitat to forage and hide from predators (Day et al. 1985, Laist 1997, Cadée 2002).

The effects of litter on benthic communities are poorly known, however, certain types of large litter items, such as lost fishing gear, cause negative physical impacts, especially in fragile habitats such as coral reefs (Mortensen et al. 2005). In the Arctic, fishery-related litter is most dominant (Buhl-Mortensen and Buhl-Mortensen 2017, Haarr et al. 2023). Local activities are a more common source of litter than long transport or drifting, and this agrees with findings from other areas with high fishing activities, such as on oceanic ridges and seamounts (Pham et al. 2014, Woodall et al. 2015). Benthic organisms caught in traps and nets will suffer from a reduction in food intake and starvation (Erzini et al. 2008, Goodman et al. 2021).

Lost fishing gear causes entanglement in corals or other complex habitats, ghost fishing, and physical damage to living organisms in general (Bergmann and Klages 2012, Ragnarsson et al. 2017). Trawls and longlines entangled in the corals may cause tissue damage and erosion to the skeleton (Mortensen et al. 2005). Corals with tissue damage are more susceptible to colonization by epibionts. Bavestrello et al. (1997) found that damage to colonies by fishing line, with subsequent colonization of epibionts, is the major cause of mortality for the gorgonian *Paramuriecea clavata* in the Ligurian Sea. From retrieval actions of ALDFG, there are several records of entanglement of benthic invertebrates like crustaceans and various fish species in Arctic waters, e.g., from Norway and Greenland (Greenland Institute for Natural Resources 2022). Smothering, i.e., the covering and abrasive action of seafloor litter on benthic organisms, can also have adverse effects (Canals et al. 2021).

Assessing population-level effects of entanglement is challenging (Adimey et al. 2014). First, entanglements are generally sporadic events, distributed over vast areas. It is also difficult to assess the number of animals that drowned, are decomposed, or depredated before their mortality could be observed and documented (Kirkwood et al. 1997, Laist 1997). Data on northern fur seal (*Callorhinus ursinus*), when the species was still commercially harvested, show that up to 0.7% of the juvenile seals caught were entangled (Fowler et al. 1990) and that 15% of those juveniles died as a result of entanglement (Fowler 1987). A recent compilation of published studies on population-level effects of plastic entanglement on marine megafauna (Senko et al. 2020) found only two studies on seabirds (Votier et al. 2011, Lavers et al. 2013), which both failed to detect a population effect. To date, coordinated sampling efforts and monitoring schemes with a corresponding database to understand the population impact of entanglement have been limited (UNEP/MAP 2018).

Marine mammals and seabirds are long-lived species with delayed age of maturity. As such, survival is important for the stability of the population (Pfister 1998, Sæther and Bakke 2000). The higher rate of juvenile entanglement observed in some species could potentially decrease recruitment rates (Votier et al. 2011, Knowlton et al. 2012). To best address the entanglement issue, scientists need to know where litter is most likely to accumulate (Richardson et al. 2019), which litter items are more likely to cause entanglement, where entanglement events are more likely to happen, and what species are more prone to entanglement.

An inventory and review of the existing databases whose primary focus is to report entanglement, or databases that could provide some level of information relating to entanglement was undertaken. For each database, the geographic scale (local, regional, or global), and if it concerned biota, plastic, or both

Table 1.1 Inventory of databases that include entanglement data.

Organization	Description and Database	Pros	Cons
Environmental Research Institute (North Highland College UHI and University of the Highlands and Islands, UK).	Birds and Debris collects and catalogues images and reports of birds' interactions with plastic debris from all over the world. This includes entanglements or incorporation of plastic debris in nests. (BirdsandDebris.com)	Global database A user guide has been developed in the Greenlandic language (https://wildlifeand debris.com/gl/)	Difficult to access pictures when several pictures are in close or same location No filter to specify ingestion, nest incorporation, or entanglement Data not downloadable, difficult to analyze (see Ammendolia et al. 2022)
NOAA's National Marine Mammal Entanglement Response Networks	Network operations are coordinated by NOAA Fisheries, and the network includes volunteers, non-profit organizations, academia, industry, and government organizations. Data on large whale entanglements for the entire USA are presented in yearly reports. There are specific annual reports for the Alaska Marine Mammal Stranding Network, which include pinnipeds, small cetaceans, and large whales. (https://www.fisheries.noaa.gov/nationa l/marine-life-distress/large-whale- entanglement-response#national-entanglement-reports)	Source of entanglement is provided (for the large whale entanglement reports)	USA only
North Atlantic Marine Mammal Commission (NAMMCO)	NAMMCO is an international body for cooperation on conservation, management, and study of cetaceans and pinnipeds in the North Atlantic. Members include the Faroe Islands, Greenland, Iceland, and Norway. BYCELS is a working group within NAMMCO that deals with animal welfare, including entanglement and live strandings. Their reports mention the number of entangled mammals in each member country (NAMMCO 2021; https://nammco.no/bycels-reports/)	Covers a large region of the Arctic	Very little to no details on the material marine mammals are entangled in
Litterbase	Litterbase offers a global dataset of scientific, peer-reviewed publications reporting interactions (colonization, entanglement, ingestion, and other) between litter and any aquatic life (Bergmann et al. 2017). Different filters allow searching for species interactions, publication dates, aquatic system (marine, freshwater, or estuary), and habitat (beach, benthic, pelagic, or surface). At a selected location, a pop-up window indicates the source of the information with all the details related to the paper. (Litterbase.awi.de)	Global database Data can be filtered	Data are not downloadable
Canadian Wildlife Health Cooperative (CWHC)	A database maintained since 1992 for wildlife health surveillance (Wildlife Health Intelligence Platform or WHIP). The platform offers georeferenced wildlife health events, the animals collected or sampled, and the diagnostic/necropsies conducted (King et al. 2023). In the case of entanglement, it would be included in the report. (http://www.cwhc-rcsf.ca/index.php)	Long running database	Data only accessible to the CWHC network
U.S. Fish and Wildlife Service	An "Injury and Mortality Reporting" portal to report injured or dead birds and bats. You need to be U.S. Fish and Wildlife Service staff, or employees of federal or state agencies, Tribes, or NGO to enter data, and you need an Environmental Conservation Online System account to be able to contribute to the portal. (https://ecos.fws.gov/imr/welcome)	Very specific data pertaining to birds and bats	Data are only available to a subset of federal employees People uploading data can only see data related to their own project

was recorded. The accessibility of the databases, in terms of the number of languages proposed on the website or app of respective databases, and the ease to either contribute data or download data were also recorded. Finally, the order of magnitude of entries to give an idea of the importance of each database, the time frame it covers, and if images are available (Appendix A) were appraised.

1.2 Inventory of databases

Databases were grouped into three categories: those that include entanglement data (Table 1.1), bird/wildlife databases that could be used to monitor entanglement (Table 1.2), and litter databases that could be used to monitor entanglement (Table 1.3).

Table 1.2 Inventory of bird/wildlife databases that could be used to monitor entanglement.

Organization	Description and Database	Pros	Cons
Cornell Lab of Ornithology	eBird is a checklist of birds seen or heard submitted by volunteer birdwatchers. Survey effort is captured by the duration and distance traveled. eBird is a global dataset with millions of checklists created and billions of observations made worldwide (Sullivan et al. 2009). Data collected have been used in different ways, from research to monitoring and conservation planning (Sullivan et al. 2017). (ebird.org)	Truly global Data are downloadable	No specific category for entangled individuals User must look through a long photo catalog manually
California Academy of Science	iNaturalist is a social network for sharing biodiversity information. Within iNaturalist, members can create projects in which observations from other members that fit the project mandate will be collected. For example, several projects fall under the "Nature and Plastic" portal. It provides valuable open data through the creation of different projects (i.e., Beach finds and Washashore) and can be used by scientists to answer conservation issues (Winton et al. 2018). The "Tangled & Trapped" project includes many non-target species caught in illegal (or legal) traps. These projects have a diverse geographic scale, but for the most part, are limited in the number of observations (100s). A promising project within iNaturalist is "Beached Birds." Any entries that specify their sighting was a dead bird, and if it is within a certain distance of the coast, will automatically be added to the project. It has 10,000s of observations, but there is no filter for entanglement. (https://www.iNaturalist.org)	Global Specific "Beached Birds" project	No specific filter for entanglement User must look through a long photo catalog manually
Observation International	An organization started in 2015 and based in the Netherlands. It provides a platform to store nature information on a global scale. Observations can be filtered by general group (birds, mammals, fish, butterflies), by country, and by rarity. Some entries have associated images. (www.observation.org)	Global	Downloadability is uncertain, but data are open and free to use No filter for entanglement User must look through a long photo catalog manually
U.S. Geological Survey Bird Banding Laboratory	A website to report found bands. Band reporting can be made from either live or dead birds. An option to specify if the bird was found entangled. Banding and recapture records are available from 1960 with a yearly updated release of the dataset. Current dataset includes 5 million encounters of 800 species. (https://www.pwrc.usgs.gov/BBL/bblretrv/index.cfm)	Data are downloadable Easy to query for entanglement	
Norwegian Species Observation Service	Has been published since 2008 by the Norwegian Biodiversity Information Centre, on behalf of the Ministry of Climate and Environment. Observers can upload their sightings including photos, which go through some internal quality assurance process. (https://www.artsobservasjoner.no/) (www.biodiversity.no)	Covers all taxa Data are downloadable A filter option allows to select for, e.g., "death," "death by fishing net," "harm by fishing gear," and/or a text string	Only available in Norwegian
SLU Swedish Species Information Centre	Artportalen is a national Swedish portal, run by the SLU Art Data Bank, for reporting and searching for species sightings in Sweden. Observers can upload their sightings including photos, which go through some internal quality assurance process. The search results are downloadable. Description of entanglement events may be included as a comment. (https://www.artportalen.se/)	Covers all taxa Data are downloadable A filter option allows to select for, e.g., "death" and "drowned in fishing net," but not for a text string	Only available in Swedish
Danish Ornithological Association	DOFbasen is a database started in 2002. Observers can upload their sightings including photos, which go through some internal quality assurance process. Users can also search the database by location or dates. The behavior dropdown search button includes "dead," which is a good start to look for entangled birds. Since January 2024, two new subcategories for "death caused by fishing gear" and "death caused by plastic litter" have been included, but not implemented retroactively. Therefore, description of entanglement events may also be noticed as a comment in the previous records. (https://dofbasen.dk/)	Data are downloadable A filter option allows to search for text string	Does not include records from the Arctic Database is only in Danish Covers only birds
SmugMug	A website for people to post their photos. People can share their photos with anyone or have them privately stored. You can search the open access photos with text strings. Some photos have their location in the comment, while others are geo-localized. (www.flickr.com)	Site is available in nine languages	Some photos might be missed if they are not tagged User must manually go through thousands of photos

1.3 Case study: the Northern Gannet

As an example of data acquired through the different datasets listed above, a wide search for records specifically addressing the entanglement of Northern Gannet in Nordic countries (Denmark, Norway, Sweden, Iceland, and Faroe Islands) was performed. National databases in Norway, Denmark, and Sweden contained 75% of the records, and international databases contained only 7% (Fig. 1). Additional records (17%) on entangled gannets from the Nordic countries were also found by doing a Google search using native words for "gannet" in combination with "dead," "entanglement," "rope," "fishing net," "fishing line," "hook," and/or "string." Only three records on entanglement of gannets from Nordic countries were found in scientific papers using a Google Scholar search. Altogether, 329 records on entangled gannets, dead or alive, have been found (until now) through this first search for such types of records.

~10% of the Nordic records are from Arctic waters, i.e., Northern Norway, Faroe Islands, or Iceland (Strand et al., *unpublished data*). What holds true for Nordic countries might not apply to Canada, in which international citizen science databases like eBird, iNaturalist, and Observation.com may be the major/only contributors to such entanglement records.

Table 1.3 Inventory of litter databases that could be used to monitor entanglement.

Organization	Description and Database	Pros	Cons
National Oceanic and Atmospheric Administration (NOAA)	Marine Debris Tracker has worldwide data, but most surveys are in the northern hemisphere (Jambeck and Johnsen 2015). Marine Debris Tracker was first established in 2006 by the U.S. Congress. It has regional coordination, and its main focuses are the removal, prevention, response, and research related to plastic pollution. (www.debristracker.org)	Global dataset, but mostly in North America	Data are difficult to access
Oslo/Paris Convention for the Protection of the Marine Environment of the Northeast Atlantic (OSPAR)	OSPAR is a legislative instrument that regulates international cooperation for the environmental protection of that area. OSPAR has guidelines and protocols to conduct shoreline litter monitoring (OSPAR 2020). The OSPAR beach litter database may include a note section in the survey data in which entanglement events could be recorded. (https://beachlitter.ospar.org/)	Data are downloadable	No place on the downloadable data for entangled animals
Ocean Wise Shoreline Clean- up	The Ocean Wise Shoreline Clean-up aka the Great Canadian Shoreline Cleanup, was started in British Columbia in 1994 by the employees and volunteers of the Vancouver Aquarium. It has become a national conservation organization and promotes clean-up events in Canada and the United States (Chen et al. 2019). (https://ocean.org/pollution-plastics/shoreline- cleanup/)	Over one million volunteers over the years	Data are not downloadable
International Coastal Cleanup	Created by Ocean Conservancy in 1986 with its international partners. It operates in over 90 countries. In their yearly reports, the number of entangled animals is not always provided, and when the number of entangled animals is provided, there is no indication about the location or the species (Ocean Conservancy 2005). The numbers of entangled animals reported (46 birds in 2004) appear to be underestimated relative to the number of volunteers (298,347) and the quantity of debris collected (3.5 million kg of debris). (https://oceanconservancy.org/trash-free- seas/international-coastal-cleanup/annual-data- release/)	There are some entanglement data	Data are not downloadable
Ocean Conservancy Trash Information and Data for Education and Solutions (TIDES)	Data of trash collected are entered through an app called "Clean Swell." Data are available from 2015 to present. Users must log in to view reports, but the app does offer a specific field for entangled animals. Entanglement data reveal location and animal type (fish, bird, turtle, crab, rat, sea lion), if the animal was dead or injured, and sometimes, the debris the animal was in. (https://www.coastalcleanupdata.org/)	Global Reports (by country, summary of items, top 10 items, detailed summary with locations and items found) can be downloaded Available in English, Spanish, French, and Portuguese	
European Environment Agency Marine Litter Watch	Marine Litter Watch was created in 2014 with initiatives to collect litter from beaches. Most of the locations are in Europe, with a few locations outside the continent. There is no report of entanglement in the data, and such events would only be recorded as a note. (https://www.eea.europa.eu/publications/marine- litter-watch/briefing)	Data are readily downloadable	No information related to entanglement Limited regional relevance outside of EU
Norwegian Directorate of Fisheries	A portal listing the locations where lost fishing gear was recovered and the associated biota found entangled in it. The description of the biota is often vague (i.e., lots of crab, small fish). So far, the clean-ups removed 2669 pot traps, 20,000 m of gill nets, and 172,450 m of rope for a total of 11,000 kg of fish and 15,000 crabs. (https://www.fiskeridir.no/English/Coastal- management/Marine-litter)	Data are downloadable	
European Marine Observation and Data Network	The European Marine Litter Database (MLDB) holds data on litter abundance for various environmental matrices and is linked to other databases (OSPAR, ICES DATRAS). The data report entangled animals, but the numbers are really low (30 animal for more than 12,300 surveys). (https://emodnet.ec.europa.eu/en/marine-litter)	Enables access to data from different environmental matrices (i.e., coastlines, floating, seafloor)	Mostly European based, with some Arctic data

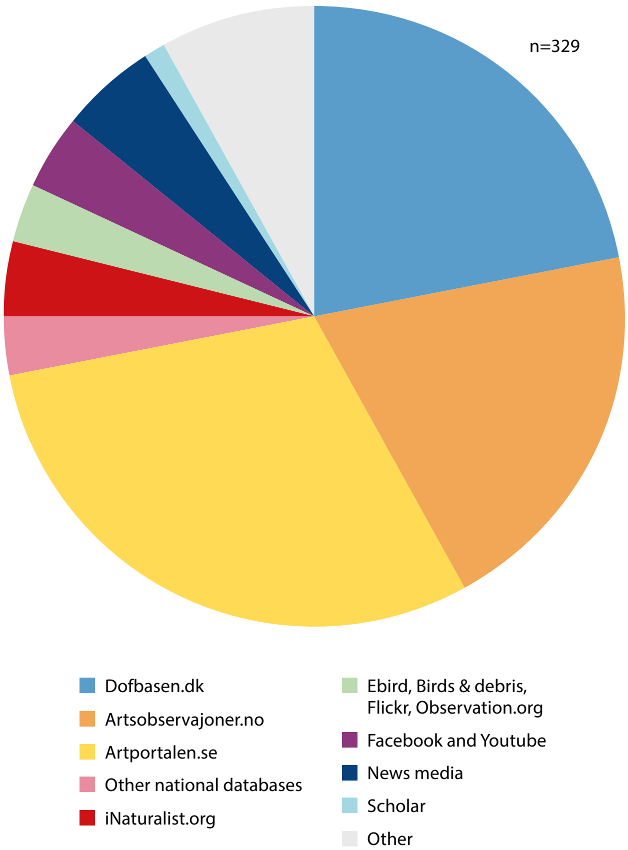

Figure 1.1 Distribution of database sources with records (n = 329) on entangled Northern Gannets (*Morus bassanus*) in Nordic marine waters (Denmark, Norway, Sweden, Iceland, and Faroe Islands).

1.4 Conclusions

The identification of a database that could act as a repository for data generated on biota entanglement in the AMAP region to evaluate which species were most affected and the locations of entanglement hotspots was a primary goal. In general, there is a paucity of data in the Arctic region (Stelfox et al. 2016), or the data are hidden within scientific committee reports (e.g., International Whaling Commission Report of the Scientific Committee). Datasets were found that provide (or potentially provide data) on entanglement records from a wide array of backgrounds. Each dataset was designed with specific objectives in mind, and at this point, it is expected that no single dataset satisfies our requirements. In general, the global datasets tend to offer little information regarding entanglement, and it is often sporadic rather than systematic. Specific entanglement datasets are generally run by organizations working at a local or regional scale, with their trained volunteers following a protocol, and with surveys done on a regular basis. In either local or global datasets, there is a lot to ponder regarding the type of quality control and quality assurance applied to the data. How to discriminate between records of animals entangled in litter in the environment versus active fishing gear is possibly the most pressing question to be addressed. Additionally, what steps are taken to make sure the same animal is not counted multiple times within the same dataset? How often is an entangled animal recorded in multiple datasets? For example, entanglement events recorded in COASST database are not entered in other databases by COASST staff, but it is not certain that COASST volunteers do not enter the entangled animal found during their beach survey in an external database. Other issues include the time and effort required to go through pictures to locate entanglement events from several datasets. The lack of harmonized circumpolar entanglement data and the difficulty accessing these data (Appendix A) are the main gaps to moving forward with an analysis of entanglement data in the Arctic.

Difficulties encountered to get meaningful entanglement data are even more pronounced in the Arctic where human population is sparse. As a result, reviews on entanglement studies do not offer much more than a list of impacted species because quantitative analyses are not possible due to a lack of standardized methods in sampling and reporting (Kühn and van Franeker 2020). Despite the presence of a strong seabird researcher community, the Arctic region has no specific database available to collect information regarding plastic and seabird interactions (either ingestion or incorporation into nests), as explored by Linnebjerg et al. (2021). At this point, it is very difficult to pinpoint the locations and species most affected by entanglement in the Arctic, although global maps and species sensitivity distributions are emerging (Høiberg et al. 2022). It should be a priority for every observed entanglement event to be systematically recorded. It is suggested that any entanglement record should at least include: date observed, species involved, location, and litter item categories causing entanglement (as specifically described as possible, e.g., fishing-related litter items, consumerism plastic, etc.), and other key information as outlined in Appendix B. In addition, photos of the entanglement, including the animal and material, would be beneficial.

One possible recommendation is to create a cross-promotional approach, in which organizations responsible for clean-up events would have a clear statement in their protocol regarding the actions to take in case of entanglement encounters by their volunteers or staff (Appendix C). The very low human population in the Arctic is a barrier to the development of many of the databases inventoried here because many of them rely on countless hours of volunteer engagement, citizen scientists, or individual observations (Nevins et al. 2011, Parrish et al. 2019). However, scientists working on seabird or seal colonies could record entanglements to monitor the change in debris according to existing protocols (Galgani et al. 2023). Trends in entanglement data might provide a benchmark to evaluate the impact of clean-up actions and mitigation measures against marine litter. Similarly, entanglements in sessile organisms could be systematically recorded during video transects, and documentation of litter-wildlife interactions should be an integral part of any seafloor macro-litter monitoring (Tekman et al. 2017). The use of drones might help in the quantification of some species (Claro et al. 2019), but at this point, the use of existing knowledge from non-Arctic species might be the most sensible approach, despite the different specificities inherent to the various regions.

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Appendix A: Published records on entanglement of marine mammals in Arctic countries.

	Russia	USA	Canada	Greenland	Faroe Islands	Iceland	Norway
Marine mammals							
Right whale (*Eubalaena* spp.)		NOAA Report 2017	Benjamins et al. 2012				
Bowhead whale (*Balaena mysticetus*)		Philo et al. 1992, Rolland et al. 2019	Benjamins et al. 2012, Treble and Stewart 2012	Kapel 1985, Treble and Stewart 2010, Levermann 2016			
Blue whale (*Balaenoptera musculus*)		NOAA Report 2017					
Gray whale (*Eschrichtius robustus*)		NOAA Report 2017					
Humpback whale (*Megaptera novaeangliae*)		Neilson et al. 2009, NOAA Report 2017	Benjamins et al. 2012	Levermann 2016		Basran et al. 2019	
Common minke whale (*Balaenoptera acutorostrata*)		NOAA Report 2017	Benjamins et al. 2012	Levermann 2016			
Fin whale (*Balaenoptera physalus*)		NOAA Report 2017	Benjamins et al. 2012	Levermann 2016		Sadove and Morreale1989	
Sei whale (*Balaenoptera borealis*)		NOAA Report 2017					
Bearded seal (*Erignathus barbatus*)							Bergmann et al. 2017, Hallanger and Gabrielsen 2018
Harbor seal (*Phoca vitulina*)							Bergmann et al. 2017, Hallanger and Gabrielsen 2018
Steller sea lion (*Eumetopias jubatus*)	CAFF Report 2019	Manville 1990					
Northern fur seal (*Callorhinus ursinus*)		Fowler 1987, Derraik 2002					
Terrestrial mammals							
Polar bear (*Ursus maritimus*)							Bergmann et al. 2017
Reindeer (*Rangifer tarandus*)		Beach et al. 1976					Bergmann et al. 2017
Arctic fox (*Vulpes lagopus*)	Gavrilo 2019						
Seabirds							
Laysan albatross (*Phoebastria* immutabilis)	Gavrilo 2019						
Northern Fulmar (*Fulmarus glacialis*)	Gavrilo 2019						Camphuysen 2000
Sooty Shearwater (*Ardenna grisea*)	Gavrilo 2019	Manville 1990					
Short-tailed Shearwater (*Ardenna tenuirostris*)	Gavrilo 2019						
Leach's Storm-petrel (*Hydrobates leucorhous*)		Manville 1990					
Northern Gannet (*Morus bassanus*)	Gavrilo 2019			O'Hanlon et al. 2019			Barrett 1988
Great Cormorant (*Phalacrocorax carbo*)	Gavrilo 2019						
Japanese Cormorant (*Phalacrocorax capillatus*)	Gavrilo 2019						

	Russia	USA	Canada	Greenland	Faroe Islands	Iceland	Norway
American Herring Gull (*Larus smithsonianus*)	Gavrilo 2019						
Black-headed Gull (*Chroicocephalus ridibundus*)	Gavrilo 2019						
Black-legged Kittiwake (*Rissa tridactyla*)	Gavrilo 2019						
Arctic Tern (*Sterna paradisaea*)							Bergmann et al. 2017, Hallanger and Gabrielsen 2018
Thick-billed Murre or Brunnich's Guillemot (*Uria lomvia*)	Gavrilo 2019						Camphuysen 2000
Dovekie or Little Auk (*Alle alle*)							Camphuysen 2000
Ancient Murrelet (*Synthliboramphus antiquus*)	Gavrilo 2019						
Tufted Puffin (*Fratercula cirrhata*)	Gavrilo 2019						
Atlantic Puffin (*Fratercula arctica*)					Anker-Nilssen et al. 2018		Camphuysen 2000
Red-throated Loon (*Gavia stellata*)	Gavrilo 2019						
Black-throated Loon (*Gavia arctica*)	Gavrilo 2019						
Yellow-billed Loon (*Gavia adamsii*)		Naves and Zeller 2017					
Common Eider (*Somateria mollissima*)	Gavrilo 2019						
Common Merganser (*Mergus merganser*)	Gavrilo 2019						
Red-breasted Merganser (*Mergus serrator*)	Gavrilo 2019						
Tufted Duck (*Aythya fuligula*)	Gavrilo 2019						

Appendix B: Inventories of databases that potentially could provide entanglement data. Cell colors indicate data availability: green = high availability; orange = limited or partial availability; red = low availability or missing information

Name	Web address	Entries	Scale	Year database covers	Biota data	Plastic data	Images	Upload data	Downloadable	Available languages
Entanglement databases										
Birds and Debris	www.birdsanddebris.com	1,000	Global		Birds			Website No login required	No	English Greenlandic
NOAA National Marine Mammal Entanglement Response Networks	https://www.fisheries.noaa.gov/national/marine-life-distress/large- whale-entanglement-response#national-entanglement-reports	100s	National (USA)	2007-	Marine mammals			Phone or App to report stranding	Yearly reports	English
BYCELS from NAMMCO	https://nammco.no/bycels-reports/		Faroe Islands, Greenland, Iceland, Norway	2018-	Marine mammals	no details about entanglement material			Reports	English
Litterbase	www.litterbase.awi.de	1,000	Global	Publications from at least 1972				No input by public	No	English, German, Turkish
CWHC - WHIP	http://www.cwhc-rcsf.ca/	?	National (Canada)	1992-			?		Upon request?	English, French?
IMR	https://ecos.fws.gov/imr/welcome	?	National (USA)	?			?	Need an Environmental Conservation Online System account	Upon Request?	English
Biodiversity databases										
eBird	www.eBird.org	1,000,000	Global	2002 – (but you can add older data)	Birds			ebird app	Yes, upon request	17 languages including English Spanish French
iNaturalist	www.inaturalist.org	10,000	Global	2008-				inaturalist app	Yes	English
Observation International	https://observation.org/	1,000,000	Global	2015-			Sometimes	ObsIdentify iObs and ObsMapp apps	No, but free to use data	English, French, Spanish, German, Dutch
Bird Banding Lab	https://www.pwrc.usgs.gov/BBL/bblretrv/index.cfm	?	National (USA and Canada)		Birds	You can report entanglement	Images of bird or band can be uploaded	Website	No	English, French, Spanish
Debris databases										
Marine Debris tracker	www.debristracker.org	10,000	Global	2010-				Marine Debris Tracker app	Difficult	English
Ocean Wise Shoreline Cleanup	https://ocean.org/pollution-plastics/shoreline-cleanup/	10,000	Regional (Canada and USA)	1994-				Submit online data card	No	English French

International Coastal Cleanup	https://oceanconservancy.org/trash-free-seas/international-coastal-cleanup/	10,000	Global	1986-	Probably underestimated			Clean Swell app	No	Over 100 languages
Nurdle Patrol	https://nurdlepatrol.org/	1,000	Global (but North America focused)	2018-		Nurdles	Optional	Website, no login required		English, Spanish
TIDES	https://www.coastalcleanupdata.org/	100,000	Global	2015-	Entangled animal			Clean Swell app	Yes	Spanish, French, English, Portuguese
European Marine Litter Database	https://emodnet.ec.europa.eu/en/marine-litter	10,000 +	Europe	2001-	Report of entangled animals					English
Norwegian Directorate of Fisheries	https://portal.fiskeridir.no/portal/apps/webappviewer/index.html?id=9e35f133ef924d68bfa0455965230f5a		National (Norway)	1985?	Not specific	Ghost gear	No		Yes	Norwegian

Appendix C: Recommendations of key information identified for Arctic-wide assessments of animal entanglement in litter/debris in the environments.

Record ID No.	
Country	
Entanglement category	• Entanglement (general) • Entanglement in colonies/breeding sites • Plastic used as nest material • Entanglement in retrieved ghost nets/long lines/traps • Ingestion of plastic • Other
Taxonomic group	Bird / Mammal / Fish / Invertebrate / Other
Species name (lowest taxonomic level possible)	
Scientific name	
Status of animal	Dead animal / Live animal / Rescued alive / Nest material
Number of individuals	
Date for observation	dd-mm-yyyy
Location name	
County/municipality	
Sea (sub)region	
Longitude, GPS position	GPS DD (decimal degrees), Alternatively, approx. position may be later extracted from location name
Latitude, GPS position	GPS DD (decimal degrees), Alternatively, approx. position may be later extracted from location name
Data Source / Reference	Name of database or reference to report
Weblink to record	Example: https://macaulaylibrary.org/asset/169999231
Photo	Yes / No
Category of litter item	Preferably described using same detailed terminology as for beach litter categories, alternatively just fishery related litter / consumerism plastic
Condition of dead animals	Fresh corpse / Decayed corpse / Not reported /Not relevant
Age of individual	Adult / immature / Not reported
Comments made	Description of observation, if available

2. Physical Effects of Plastic Ingestion

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2.0 Introduction

In the 21st Century, habitats around the globe are threatened from a variety of stressors, principally of anthropogenic origin (Mazor et al. 2018); some of the more obvious ones include climate change (Post et al. 2009, AMAP 2021), resource and land development (Theobald et al. 1997), overharvest of various resources (Burgess et al. 2013), human overpopulation (Ganivet 2020), and chemical contamination (Noyes and Lema 2015, Borgå et al. 2022). However, no environmental issue has so rapidly galvanized public attention like that of plastic pollution, in part because it is truly global (MacLeod et al. 2021) and obvious to any person. Indeed, plastic garbage is found everywhere from urban streets to remote wilderness, and people feel they can have positive impacts on this issue by picking it up from the environment and disposing of plastics properly. However, as learned recently, the plastic pollution threat is far larger than some "garbage in the street," with microplastics (plastics < 5 mm in size) now found effectively in every habitat on the planet (e.g., Ivar do Sul and Costa 2014, Peeken et al. 2018, Ajith et al. 2020). Plastic pollution—a human-derived threat with its genesis in the 20th century— has increased dramatically with the growth of the plastics industry and the myriad types and uses of plastics (e.g., Rochman et al. 2013, Borrelle et al. 2020). At the same time, poor management of waste plastics is a global issue, with a collective, decades-long lack of action to reduce plastic pollution, to the point where it has become a ubiquitous threat to wildlife and habitats (MacLeod et al. 2021).

There have been countless extensive reviews of the occurrence of plastic debris in the environment and of the types of organisms affected by plastic pollution (e.g., Bucci et al. 2020, Kühn and van Franeker 2020, Bergmann et al. 2022). From an organismal perspective, there are two main ways that plastic debris can have a deleterious, physical effect on an animal: through entanglement (Butterworth 2016, Ryan 2018, Lusher et al. 2022, AMAP 2025, Chapter 1) or ingestion/uptake (e.g., Derraik 2002, Cózar et al. 2014, Unger et al. 2016). Once plastic debris enters an organism, it can then have three further means of affecting organismal health: transferring deleterious chemicals to the animal, which can affect organismal health (e.g., Kühn et al. 2020, AMAP 2025, Chapters 3 and 4), breaking down and entering other organs, cells, or tissues (e.g., Charlton-Howard et al. 2023), or blocking and/or damaging the digestive tract (e.g., de Stephanis et al. 2013, Kühn and van Franeker 2020). In this chapter, the focus is on the occurrence of physical damage from ingested plastic debris to the digestive systems of wildlife in the Arctic.

It is well-established that ingestion of plastic debris can have deleterious effects on wildlife (Machovsky-Capuska et al. 2019, Puskic et al. 2020, MacLeod et al. 2021), but what is less well understood is how much plastic may be required to do this, how impacts vary by types of plastics (e.g., fragments vs. threads), what the sublethal effects of plastic ingestion may be, and how variable this is across species. In practice, when organisms ingest plastics, it can pass through their body and be eliminated with solid fecal waste; it can physically reduce or block food passage to their digestive system; it can physically irritate, penetrate, or cut surrounding tissues, causing lesions and exposing wildlife to the risk of infection and costs of tissue repair; or it can have other related, deleterious effects on tissues (e.g., plasticosis; Charlton-Howard et al. 2023). Certainly, these types of deleterious effects have been documented across many species (Wilcox et al. 2015, Kühn and van Franeker 2020, Tulatz et al. 2023).

Reduced body condition due to starvation, or at least a reduction in nutritional food intake due to blockage or a feeling of partial satiation, has been found in some cases (e.g., turtles, birds, whales; Puig-Lozano et al. 2018, Puskic et al. 2020, Santos et al. 2020, Tulatz et al. 2023), and sometimes this was thought to be the cause of mortality (Pierce et al. 2004, Puig-Lozano et al. 2018). Elsewhere, some wildlife species have been found with lesions in their digestive systems (e.g., turtles, birds, marine mammals; Lusher et al. 2018, Roman et al. 2019, Lima et al. 2022), but even in locations where there are high levels of plastic ingestion with often large, ingested loads, internal damage to tissues is not necessarily common (e.g., seabirds; Sileo et al. 1990, Seif et al. 2018). Some species, like gulls, can ingest plastics (e.g., Seif et al. 2018, Benjaminsen et al. 2022), but may regurgitate these hard, potentially deleterious items (Hammer et al. 2016) before they cause internal problems. Consequently, understanding the digestive system and behavior of a species is critical to interpreting if or how it may be affected by ingested plastics.

2.1 Plastic ingestion in Arctic wildlife

Almost all the examples of plastic ingestion and the related harm noted in Section 2.0 come from studies outside of the Arctic. Indeed, most of the recent reviews on threats from plastic ingestion are based on terrestrial or marine wildlife in tropical and temperate regions (e.g., Egbeocha et al. 2018, Haegerbaeumer et al. 2019, Puskic et al. 2020, Santos et al. 2021, Savoca et al. 2021). However, Arctic wildlife is not immune to the threat of plastic ingestion.

The occurrence of plastics in Arctic environments and wildlife, and the ability to monitor this threat has been the focus of recent reviews (e.g., Lusher et al. 2015, 2022, Baak et al. 2020, Kühn and van Franeker 2020, Collard and Ask 2021). Indeed, despite the remoteness of Arctic environments and the relatively low human population density across this region, plastic debris has been found in most Arctic ecosystems and in some wildlife (Bergmann et al. 2022). Arctic marine waters (Lusher et al. 2015, Cózar et al. 2017), coastlines (Pollet et al. 2023), sea ice (Peeken et al. 2018), snow (Rosso et al. 2024), glaciers

(Stefánsson et al. 2021, Hamilton et al. 2022), and freshwaters (Citterich et al. 2023) are all polluted at some level with plastics. The majority of these plastics move to the Arctic from temperate and tropical areas, through atmospheric or ocean transport mechanisms, but some debris is clearly of local origin (Mallory et al. 2021, Strand et al. 2021).

Given its pervasive and ubiquitous occurrence across habitats, it comes as no surprise that plastic debris is found in many types of Arctic wildlife (reviewed in Collard and Ask 2021; see Box 2.1). Few of the invertebrate species at the bottom of Arctic marine food webs have been sampled for plastic uptake (~ingestion; Grøsvik et al. 2022), but microplastics have been detected in lower trophic level species from many organism groups including polychaetes, echinoderms (e.g., *Asterias rubens*), crustaceans (e.g., *Pandalus borealis*), and mollusks (gastropods and bivalves; e.g., Fang et al. 2018, Knutsen et al. 2020, Grøsvik et al. 2022). At higher trophic levels in Arctic marine food webs, fish are key predators that also ingest plastics in the Arctic (Collard and Ask 2021, Kögel et al. 2022), perhaps most notably Arctic cod (*Boreogadus saida*), a keystone, sea-ice linked species (Moore et al. 2020, Pedro et al. 2020). Seabirds are a widespread, common feature of Arctic pelagic and coastal zones, and are perhaps the best-known group of organisms in which macroplastic ingestion in the Arctic is common and likely best known globally (Wilcox et al. 2015). The more common Arctic-breeding seabird species in which plastics has been found include Thick-billed Murres (*Uria lomvia*), Black-legged Kittiwakes (*Rissa tridactyla*), Northern Fulmars (*Fulmarus glacialis*), Little Auk or Dovekies (*Alle alle*), Glaucous Gulls (*Larus hyperboreus*; Benjaminsen et al. 2022), and Arctic Skuas (Parasitic Jaeger; *Stercorarius parasitica*; reviewed in Baak et al. 2020). Plastics are also commonly found in some waterfowl, gulls, and shorebirds (Holland et al. 2016, Flemming et al. 2022), although it is unclear for these groups whether debris were ingested while the bird was present in the Arctic. Arctic whales, such as beluga (*Delphinapterus leucas*) ingest plastic (Moore et al. 2020) and larger whales in Arctic waters have been harvested with macroplastic in their digestive tracts (reviewed in Collard and Ask 2021), although it is impossible to determine where ingestion of the plastics occurred. Pinnipeds have also been sampled for plastics (Collard and Ask 2021), with surprisingly only two species showing evidence of plastic ingestion, including northern fur seals (*Callorhinus ursinus*) and hooded seals (*Cystophora cristata*; Pinzone et al. 2021). In the terrestrial environment, anthropogenic materials including macroplastics have been found consumed by Arctic foxes (*Vulpes lagopus*; Anthony et al. 2000, Hallanger et al. 2022, Technau et al. 2022) and Arctic wolves (*Canis lupus arctos*; Marquard-Petersen 1998). The top Arctic predator, the polar bear (*Ursus maritimus*) is known to ingest plastics (Lusher et al. 2022), probably both from direct consumption of anthropogenic household wastes (e.g., from landfills) as well as trophic transfer.

Box 2.1. List of knowledge gaps (in no particular order) to fill about the physical effects of plastic ingestion on Arctic biota.

1. **Terrestrial biota:** No invertebrates and few vertebrates have been sampled and none monitored.
2. **Freshwater fish:** Few sampled and tested, none monitored.
3. **Marine invertebrates:** Few sampled and tested, none monitored.
4. **Marine fish**: Few sampled and tested, none monitored.

Collectively, then, ingested plastics are found in many types of Arctic wildlife (Collard and Ask 2021). Despite this fact, as of 2023, most Arctic wildlife species have not been examined for plastic ingestion. In the terrestrial environment, we have no reports for invertebrates and very few reports for terrestrial birds and mammals, or freshwater fish. The marine environment has received far more attention, but there are myriad species there, too, yet to be tested. Nonetheless, the more we look for plastic ingestion, the more we find.

Table 2.1. Generalized summary of plastic ingestion and effects in Arctic wildlife. Categories of effects were defined as follows: *Unknown* – there is no published study from the Arctic on this effect; *Yes* – at least one study from the Arctic has reported the effect; *Insufficient data* – although the effect is known, too few studies have reported to data to generalize; *Probably* – more than one study has reported the effect from different regions, and given information from elsewhere our expert opinion is that this is likely, but given the diversity of species or limited sampling it is unclear; *No* – studies have occurred, opportunities have been undertaken in which the effect could be examined, and none have been reported.

Wildlife group	Plastic ingestion			Physical damage from ingestion?
	Occurs?	Across many taxa?	Widespread geographically?	
Marine invertebrates	Yes	Insufficient data	Insufficient data	Unknown
Marine fish	Yes	Probably	Probably	Insufficient data
Marine birds	Yes	Yes	Yes	1 species, 2 individuals
Marine mammals	Yes	Yes	Probably	No
Freshwater fish	Yes	Unknown	Unknown	Unknown
Terrestrial invertebrates	Unknown	Unknown	Unknown	Unknown
Terrestrial birds	Yes	Insufficient data	Insufficient data	Insufficient data
Terrestrial mammals	Yes	Insufficient data	Insufficient data	Insufficient data

2.2 Physical damage from plastic ingestion in Arctic wildlife

Studies of plastic ingestion in the Arctic terrestrial environment are very limited in number, for either microplastic (< 5 mm) or macroplastic (> 5 mm), and to date there are no reports of physical damage from ingestion, either in studies looking for damage, but largely because there have been few studies undertaken (Table 2.1). Similarly, in the marine environment, not many studies on plastic ingestion have been undertaken for wild marine fish and invertebrates (Collard and Ask 2021), but within those, microplastics have been found in many of the species examined but none have reported physical damage to organisms (Kögel et al. 2022). More studies have been undertaken on Arctic marine mammals, most of which have reported very few microplastics in gastrointestinal tracts, and none have reported damage from plastics to these tissues. Finally, Arctic marine birds have received the most attention; hundreds of Arctic marine birds have undergone dissections and many of those have contained macroplastic debris in low levels. Some other articles reported high numbers of ingested plastics in some individuals (e.g., Avery-Gomm et al. 2018, Collard et al. 2022a, van Franeker et al. 2022, Tulatz et al. 2023,), and many studies reported frequencies of occurrence close or equal to 100% of the birds sampled (e.g., Amélineau et al. 2016, Collard et al. 2022b, van Franeker et al. 2022, Tulatz et al. 2023). None of these have reported mortality and only one reported physical damage from ingested plastics; Tulatz et al. (2023) reported two cases of possible physical stomach damage from two different individuals of Northern Fulmars from Svalbard. The first case was the occurrence of a hole in the proventriculus of a fledgling, most likely due to the presence of a large, sharp plastic fragment found stuck horizontally during dissection. That bird was observed regurgitating prior to capture and did not eject that big piece of plastic. The second case was a green rigid thread perforating the intestine wall. While cutting the bird open, a tip of this thread was observed outside the intestine, the other part inside (Tulatz et al. 2023). This bird was shot as an adult and therefore, it cannot be excluded that the hole could have been made by the bullet rather than the rigid thread, but the hole was much smaller than the diameter of the bullet used in the shotgun. Despite these examples of tissue damage, both birds were in good health when they were collected.

Most studies do not perform histological examinations of tissues to check for sublethal tissue responses to plastic ingestion (e.g., Charlton-Howard et al. 2023). Most effects of plastic ingestion are likely sublethal and may be difficult to detect; molecular markers may offer some solutions (e.g., Pannizolo et al. 2023), but clearly much more study is required. Consequently, there remains considerable uncertainty on the chronic or sublethal effects of ingested plastics in most wildlife. With this in mind, and that there is effectively only one report, to our knowledge, of internal physical damage from plastic ingestion in Arctic wildlife (as of 2023), is physical damage not occurring in these species? That is really a matter of perspective. The findings indicate a few Arctic wildlife individuals (mostly seabirds) with high plastics loads, with few large fragments that may damage internal tissues, which suggests that the exposure to this threat, or the frequency with which they may actually occur in the Arctic, is very low—that is encouraging. An alternative explanation is that the Arctic has exceedingly low human population density and much of the wildlife remains remote from human habitation. Thus, if an organism did have considerable physical damages that led to mortality, there would be little chance that anyone would see it, despite scientists visiting many of the large bird colonies, pinniped haul-outs, or cetacean aggregation sites. However, we acknowledge that Indigenous hunters are active in many parts of the Arctic and rely on these species for sustenance (e.g., Kinloch et al. 1992), and they do not report mortality of young like we occasionally see at sites where plastic ingestion has led to mortality (e.g., Sileo et al. 1990, Charlton-Howard et al. 2023).

2.3 A new type of physical effect?

Although we have focused above on the main physical effects of digestive blockage or damage to digestive tissues, new research is finding evidence of more subtle, potentially deleterious effects of plastic ingestion. For example, Hewins and Gibson (2022) found that tissues and fecundity were altered in eastern mud snails (*Ilyanassa obsoleta*) that had ingested microplastics, and this also resulted in a shift in the microbiome to reduce alpha and beta diversity in blue mussels (*Mytilus edulis*; Ferguson et al. 2022). In recent years, scientists have discovered the role that microbiomes play in organismal health (e.g., Zhu et al. 2021, Bodawatta et al. 2022), and this has become a very active field of research. Santos et al. (2022) reviewed studies and found considerable evidence of a variety of deleterious effects at the microbiome level from ingested microplastics. Fackelmann et al. (2023) recently found that higher amounts of microplastic in two marine bird species (one of which, the Northern Fulmar, came from Arctic samples) was significantly correlated with higher amounts of pathogenic or antibiotic-resistant bacteria and fewer commensal ("good") bacteria. It is unclear whether this was attributable simply to the numbers of microplastics (i.e., a physical effect) or the chemicals coming from those microplastics, but the net effect was a physical adjustment of the key composition of the gut microbiome, in a direction suggesting deleterious but sublethal effects. Although just at the early stages of research, the potential effects of plastic ingestion on organismal health and fitness through adjustments in the gut microbiome would seem a fertile avenue for future research.

A second, "novel" physical effect is "plasticosis," whereby seabirds that consume considerable amounts of plastics may develop extensive scar tissue in their digestive tract (Charlton-Howard et al. 2023). To date this has only been reported in laboratory studies and one species of wild seabird in the Southern Ocean, which experience exceptionally high occurrence and mass of plastics in their digestive systems, but in principle this could affect Arctic seabirds as well.

2.4 Key issues and gaps

There are several challenges, and some huge gaps, to refine our understanding of the physical impacts of plastic ingestion on Arctic wildlife. Perhaps the main challenge is the need for dead animals to acquire the data. A carcass that washes up (e.g., Haelters et al. 2018) can be necropsied, or digestive tracts can be acquired by collaborating with Indigenous hunters (e.g., Baak et al. 2020, Moore et al. 2022), but fundamentally wildlife needs to be killed to check for internal damage. Beached birds are not common and/or easily accessible in most parts of the Arctic, and plastic ingestion may differ among different methods of collection (beached versus bycaught or shot animals). This presents a problem for sampling in some locations with limited Indigenous harvest and declining or threatened numbers of various species (e.g., seabirds in Norway; Frederiksen et al. 2016). In other cases, this may present an ethical problem—should animals be killed simply to check if they are experiencing problems with plastics when there is almost no evidence that physical impacts of ingestion are occurring? At the same time, if samples are needed, enough individuals need to be sampled to be statistically confident in the interpretations (e.g., Provencher et al. 2015). For example, measuring plastic debris in a minimum of 40 seabirds is a regularly cited sample size needed for some confidence in determining frequency of plastics occurrence (e.g., Provencher et al. 2017). However, that value has been established for species with a high frequency of occurrence of plastics (fulmars) and limited variability in occurrence regionally. It actually may require hundreds of sampled birds in Arctic Canada (likely everywhere in the Arctic) to detect change, compared to only 40 required near The Netherlands (e.g., Provencher et al. 2015). This may be possible to acquire by working with hunters for regularly harvested species (e.g., seals, murres, eiders, caribou). However, this is much more challenging for species not regularly shot nor consumed, yet perhaps more likely to ingest plastics and potentially have physical impacts (e.g., gulls, ravens). Species that are not consumed may be expensive to sample because scientific expeditions would have to be undertaken to collect samples, such as sessile filter feeders, offshore pelagic invertebrates, or deep-sea fish. Finally, some species may be susceptible to plastic ingestion through their foraging habits or upper positions in food chains, and yet can never be sampled unless a carcass is found. Examples of these species include Ivory Gulls (*Pagophila eburnea*, endangered in several Arctic countries and known to feed in landfills; Mallory et al. 2020), falcons, and several whale species (Jacobsen et al. 2010, Im et al. 2020, Lefort et al. 2022, Merrill et al. 2023).

Irrespective of the challenges involved, the recent scientific reviews (Baak et al. 2020, Collard and Ask 2021, Lusher et al. 2022) clearly note considerable challenges in interpreting the threat of plastic debris in Arctic wildlife due to major spatial and taxonomic gaps in our knowledge of plastic ingestion. The huge expanse of the Russian Arctic remains largely undocumented for the extent or intensity of plastic ingestion or impacts on wildlife (Collard and Ask 2021), even for the common groups (seabirds, marine mammals) documented elsewhere, although plastic ingestion by wildlife does occur there (Lusher et al. 2022). Furthermore, other geographic regions have been sampled patchily and opportunistically; additional sampling will help fill in our regional knowledge. Across all trophic levels, there is little information on most components of terrestrial ecosystems. This should be possible to address since sport or Indigenous hunters harvest ptarmigan, breeding geese, and caribou in many parts of the Arctic, while foxes are still trapped across many regions. Enhanced collaborations with hunters may help fill in this gap both on ingestion and on possible damage from that plastic ingestion. In the marine environment, most invertebrates have not been tested for plastic ingestion (Grøsvik et al. 2022), nor have many fish (Kögel et al., 2022). For the latter, there are ample commercial or subsistence harvests (for example Arctic char *Salvelinus alpinus*, Greenland halibut *Reinhardtius hippoglossoides*, pollock *Pollachius virens*, or bycatch of Greenland shark *Somniosus microcephalus*; Yan et al. 2022), in which collaborations with fishers (commercial or subsistence) should be investigated for samples (see Box 2.2).

An important first step to developing a framework that monitors (at some level) damage from plastic ingestion is to undertake a SWOT (strengths, weaknesses, opportunities, threats) analysis to examine what data are currently available in the circumpolar Arctic that may be adapted to fill this need, where there are existing missed opportunities, and where there are glaring holes in data availability (e.g., Marrazi et al. 2020). For example, the development of a pan-Arctic marine mammal stranding network, similar to (or indeed linked to) what exists in other countries (e.g., Lusher et al. 2018), along with standardized sampling protocols for potential damage from plastic ingestion, would be a welcome development to improve knowledge on plastic ingestion and possible damage from that behavior. Concurrently, standardized, or harmonized methods of plastic pollution metrics must be recorded and aligned with existing global programs (e.g., Marine Strategy Framework Directive; Galgani et al. 2023), and standards must be developed or adopted for reporting plastic ingestion damage (presumably from wildlife veterinary frameworks).

Box 2.2. List of recommendations for future research on the physical effects of plastic ingestion n Arctic biota.

1. Establish or enhance sampling programs with Indigenous or sport harvesters in various Arctic countries to acquire digestive tracts (this should be possible in many locations). Where possible, encourage citizen or community science protocols (at least for macroplastic identification).
2. Work with commercial industries (e.g., fisheries) to acquire samples from at least a subset of harvested wildlife (e.g., char, halibut, caribou). Consider a community-led analysis of macro- or microplastic (> 1 mm) identification, perhaps in collaboration with local colleges.
3. Given the negligible examples of damage to date in any Arctic biota, issues regarding physical damage from plastic ingestion should be lower priority than attention (or resources) going toward entanglement or effects of ingested chemicals, both likely of much higher concern.

2.5 Conclusions

Considering the potential physical impacts of plastic ingestion on Arctic wildlife, four key points need to be considered from existing efforts:

1. Relatively few samples of most species have been collected or examined to date. Given the low incidence of seeing deleterious effects in wildlife, even those susceptible to high loads of plastics (e.g., Daoust et al. 2021), more samples need to be examined in Arctic wildlife to ascertain how pervasive negative effects may be.
2. Of those samples that have been examined, there is only one that reports internal physical damage in wildlife with plastic ingestion (Tulatz et al. 2023). Although frequency of occurrence may be relatively high in some species (Baak et al. 2020), mass of ingested plastics is generally low (e.g., van Franeker et al. 2021), at least for migratory wildlife when sampled in the part of their annual cycle that occurs in the Arctic.
3. With only one piece of evidence of deleterious physical impacts from plastic ingestion in any Arctic wildlife at the level of an individual organism, there is currently little concern for subsequent effects from impacts of ingestion at a population level (Senko et al. 2020). Nonetheless, we need to remain vigilant as new evidence accrues, such as plastics in the stomachs of sperm whales in Norway (*Physeter macrocephalus*; Similä et al. 2022).
4. Given 1) and 2), the evidence to date does not suggest there should be conservation or wildlife health concerns for Arctic wildlife from physical damages due to ingesting plastics (unlike some southern seabirds; Roman et al. 2019, 2021, Charlton-Howard et al. 2023). This of course may change with increased shipping or development in Arctic regions (Pizzolato et al. 2016), but at current levels, physical damage from plastics appears to be very infrequent. Instead, we suggest that entanglement or transfer of chemicals from plastics, both of which we know occur in the Arctic, may be more of an issue for Arctic wildlife than damage from plastic ingestion, although the degree to which those are a concern varies by species (see other chapters).

In many Arctic locations, wildlife carcasses are accessible for scientists and can be checked for both plastics and damage from plastics when carcasses are available. However, as a plan moving forward for the collective investigation and monitoring of threats from plastic debris in the Arctic, checking for damage from plastic ingestion in wildlife should be undertaken as a component of standard monitoring (e.g., for frequency of occurrence of plastics, notably monitored in Northern Fulmars; Kühn et al. 2020, van Franeker et al. 2021), but not as a monitoring program in its own.

Box 2.3. List of policy implications on the physical effects of plastic ingestion on Arctic biota.

1. Within the OSPAR region, there is a target, at least for the seabird Northern Fulmar, to have < 10% of birds having > 0.1 g of plastic in their digestive tracts. It is also adopted as a regional threshold value for Good Environmental Status (GES) by EU MSFD (2023).

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3. Physiological Effects from Ingested Micro- and Nanoplastics in Arctic Animals – a Critical Review and Gap Analysis

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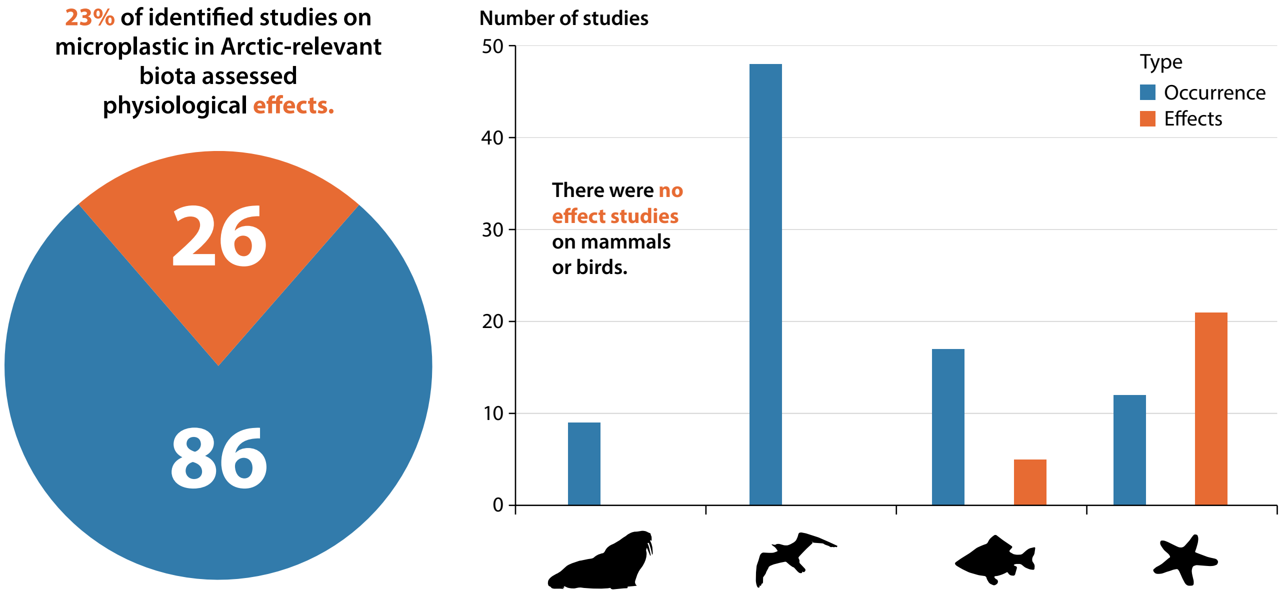


Observed effects in Arctic species were on:

- Reproduction
- Metabolism
- Tissue changes
- Movement
- Molting

3.0 Introduction

The occurrence of plastics in the Arctic abiotic environment and fauna has been shown in multiple studies (Cau et al. 2019, Collard and Ask 2021, Martin et al. 2021, Bergmann et al. 2022). In addition, specific questions have been addressed regarding plastic pollution in the Arctic, such as the source identification of plastic items (Strand et al. 2021), transport pathways (Obbard et al. 2014, Huserbråten et al. 2022), and interactions with sea ice (Peeken et al. 2018). Arctic animals can ingest both microplastic (MP; defined as plastic, rubber, and silicone particles < 5 mm; Lusher et al. 2022, Grøsvik et al. 2023a, Kögel et al. 2023) and larger items, including mesoplastic (5–25 mm), with potential deleterious effects, such as false satiation, internal injuries, and physiological impacts (Morrison et al. 2022, Mallory et al. 2024). Occurrence of nanoplastics (NP) in Arctic animals remains to be investigated (no studies in Annex Tables A3.1–A3.6).

The uptake of MP into biota occurs via direct or indirect ingestion, i.e., directly from the environment or through the diet. Fish and invertebrates can also take up MP through respiration and via their gills (Baechler et al. 2020). Once taken up, MP can transfer across trophic levels (Baechler et al. 2020). Furthermore, translocation (i.e., movement) of ingested particles from the gut into other tissues and organs (e.g., liver, hepatopancreas, eggs) has been documented in fish and crabs *in vitro* (Baechler et al. 2020). Translocation has also been observed in wild-caught fish, including fish from Arctic locations (McIlwraith et al. 2021, Hamilton 2023). Ingestion of MP by feeding animals is likely to also occur via microalgae, because MP, predominantly < 10 µm, were found to be associated with sea ice algae (*Melosira arctica*) with concentrations one order of magnitude higher than in the surrounding water (Bergmann et al. 2023). Grazers feed on these algae below the ice, but the algae also sink to the sediment where they are eaten by benthic feeders (Bergmann et al. 2023). However, although MP accumulate in gastrointestinal tracts (GIT), this accumulation might occur only in the lumen. Evidence of micro- and nanoplastics (MNP) bioaccumulation and biomagnification occurring in tissues in wild living animals is lacking and warrants further investigation (Hamilton et al. 2021, Covernton et al. 2022, Ding et al. 2023, Hermabessiere et al. 2023).

The size of plastics that can be ingested or taken up by animals, and the consequences of this uptake, may be related to the size of the animals. Estimates suggest a size ratio of roughly 20:1 between body length and the largest plastic the animal may ingest (Jâms et al. 2020), and a recent study on particle characteristics suggests there "is sufficient evidence that particle size is a critical determinant of toxicological outcomes" (Thornton-Hampton et al. 2022a). However, for the smaller size range of MNPs, there is evidence that smaller MP (< 10 µm) and NP may cause greater negative effects compared to larger MP, and observed effects in fish include impacts on survival, growth, metabolism, oxidative stress, neurofunction, intestinal microbiome, and behavior (Kögel et al. 2020). Furthermore, the occurrence of MNP in fish and potential adverse effects on the fish may also have an impact on food security and safety (Kögel et al. 2022, 2023).

Adverse effects of MNP on invertebrates, including zooplankton, larger crustaceans, and molluscs, include decreased feeding/filtering and movement, survival, growth, reproduction, metabolic rates, energy storage and slowed development, increased oxidative stress, inflammation, cytotoxicity, and rates of malformations (Kögel et al. 2020, Jeong et al. 2024). The extent of some of these effects are species-specific (Kögel et al. 2020). Additionally, for crustaceans such as zooplankton, shrimp, crab and lobster, and beach hoppers, MNP were associated with effects on molting (Kögel et al. 2020). For copepods, transgenerational effects of NP and small MP were shown (Lee et al. 2013). Studies on corals reported suppressed symbiosis, bleaching, and tissue necrosis, and studies on ascidians showed inhibition of metamorphosis (Kögel et al. 2020).

Human exposure to MP can occur, in addition to inhalation, through ingestion, e.g., of seafood, with endocytotic and paracellular transfer across epithelial tissues proposed as mechanisms for uptake into human blood and tissues (Baechler et al. 2020). In recent years, MP presence has been demonstrated in human feces (Schwabl et al. 2019), endometria (Sun et al. 2024), lungs (Amato-Lourenço et al. 2021, Jenner et al. 2022), blood (Leslie et al. 2022), breastmilk (Liu et al. 2023), and MNP in placenta (Ragusa et al. 2021, 2022, Amereh et al. 2022, Liu et al. 2023). A correlation between the presence of plastic in arterial plaque with death and stroke was published recently (Marfella et al. 2024). Although concerns have been raised about the methods (Kuhlman 2022), these findings warrant thorough investigation.

Because MP particle counts in foods are still strongly influenced by the analytical methods, published estimates are likely to be highly uncertain (Baechler et al. 2020). Even though 90% of ingested MP are thought to be removed from the human body through the excretory system (EFSA Panel on Contaminants in the Food Chain 2016), particles either retained or excreted may affect human health (Baechler et al. 2020). Expected adverse effects include oxidative stress, altered metabolism, transfer of sorbed pollutants, disturbance of the gut microbiome, inflammatory responses, reproductive toxicity, carcinogenicity, neurological effects, and immune response (Baechler et al. 2020, Rahman et al. 2021, Khan and Jia 2023, Sofield et al. 2024). In addition, plastic additives and other residues may cause toxicity (Fauser et al. 2022, Wagner et al. 2024, AMAP 2025, Chapter 4).

Due to this complexity, effect studies are often conducted under controlled laboratory conditions and focus on single species and stressors, such as a single type of polymer of a clearly defined size and shape (Thornton-Hampton et al. 2022b). However, in the environment, organisms are exposed to diverse MNP in combination with other stressors. Thus, the effects of MNP can be influenced by the presence of other toxic contaminants (Lebordais et al. 2021, Li et al. 2021, Na et al. 2021, Varshney et al. 2023) and pathogens that can travel on plastics and carry with them antibiotic resistance genes (Radisic et al. 2020, Radisic and Marathe 2021). The rapidly changing climate in the Arctic might also lead to changes of MNP occurrence (Welden and Lusher 2017) and effects due to changes in temperature (Lins et al. 2022, Sulukan et al. 2022, D'Avignon et al. 2023, Hasan et al. 2023, Na et al. 2023). Interactions between climate-driven changes in ecosystems and the toxicity of organic pollutants are not fully understood, and MNP add a further level of complexity (Borgå et al. 2022).

To estimate if a given effect of MNP observed in laboratory studies will occur in the environment and affect biota, the exposure concentrations and MP characteristics (polymer identity, size, shape; Kögel et al. 2020) leading to an effect need to be compared to occurrence levels in the environment, including a critical analysis of the methods applied (Savoca et al. 2021). Likewise, exposure studies need to select adequate particles for toxicity testing and study designs, allowing for the derivation of dose-response curves (Thornton-Hampton et al. 2022b).

Indicators for plastic pollution have been developed under several regulatory bodies such as the Oslo/Paris Convention for the Protection of the Marine Environment of the Northeast Atlantic (OSPAR) and the European Union Marine Strategy Framework Directive (MSFD 2008). The OSPAR System of Ecological Quality Objectives includes one objective according to which fewer than 10% of beached Northern Fulmars (*Fulmarus glacialis*) should have more than 0.1 g of plastics in their stomachs (OSPAR Commission 2008). Similarly, the MSFD addresses "Trends in the amount and composition of litter ingested by animals, e.g., stomach analysis" (MSFD descriptor 10 https://environment.ec.europa.eu/topics/marine-environment/descriptors-under-marine-strategy-framework-directive_en; UNEP 2021 Table S1). The MSFD states that the Northern Fulmar was chosen as an indicator for the northern European waters, while sea turtles, particularly the loggerhead species *Caretta caretta*, were chosen as an indicator for the Mediterranean Basin. For those two species, threshold values have been suggested, whereas indicators for other taxa have not been developed to the same extent (Galgani et al. 2023). The original OSPAR Ecological Quality Objective has been incorporated as a Fulmar-threshold value (TV) into the MSFD. Furthermore, MSFD recommended identifying and quantifying MP in several groups of indicator animals: birds, fish, invertebrates (bivalves), and mammals regarding trophic level, time, and space, and suggested sample preparation and analysis protocols. However, these indicators are considered to have a low level of maturity (Galgani 2023). The MSFD describes concerns about plastic ingestion that include (1) obstructions impairing feeding and digestion, (2) particulates-type toxicity, (3) transfer of toxic substances to biota upon ingestion, and (4) physical damage to organs and tissues (Section 4.4 of the MSFD descriptor 10: *Amount, distribution and composition of microparticles*). The European Union Scientific Advisory Mechanism has published a report on the potential effects of microplastics on human health and the environment, stressing that it is not yet possible to assess risks to human health (European Commission 2019). On a global scale, the UN Sustainable Development Goals (2015) include the assessment of plastics in the marine environment, however ingestion of plastics by biota is only listed as a supplementary national indicator.

The monitoring of plastics in fulmar stomachs under OSPAR has allowed the analysis of trends over time (van Franeker et al. 2011, 2021, Baak et al. 2024). This work shows that about 10% of Canadian Arctic fulmars exceed the threshold value of 0.1 g of ingested plastic. Of North Sea fulmars, 51% exceeded that threshold, but linear regression over 2009–2018 showed a significant decline, suggesting that compliance may be reached by 2054. However, the indicator focuses on particles > 1 mm in size and primarily acts as an indicator of the abundance of floating litter in surface waters. It is not intended as an indicator of the effects of plastics.

Previous work of the Arctic Monitoring and Assessment Programme (AMAP) focused on the development and implementation of monitoring initiatives for plastics in the Arctic and identified beach litter, MP in water and sediment, and plastics in seabirds as priority indicators (AMAP 2021, Provencher et al. 2022). The primary goal of these monitoring activities is to provide data on plastic occurrence in the Arctic that are comparable over space and time, allowing an assessment of the state of pollution and the effectiveness of counter measures. The AMAP guidelines were aligned with OSPAR plastic monitoring, identifying Northern Fulmars as a Priority 1 indicator for immediate trend monitoring, although the inclusion of other species should be considered, given the breadth of the Arctic and species' ranges (Lusher et al. 2022, Provencher et al. 2022). Both invertebrates and fish were included as Priority 2 indicators to facilitate baseline mapping and future trend monitoring on occurrence. It is pivotal that indicator animals are now defined on an animal group level, but further work is needed to identify the most suitable indicator animals on a species level, based on ecology, physiology, and impacts, as well as to ensure maturity of the monitoring methods.

The objective of this chapter was to review the state of knowledge on physiological effects of MNP on Arctic biota. With the view to a risk evaluation for Arctic fauna and human consumers, we explored whether information on MNP type, size, and concentration range reported in occurrence studies from environmental samples matched with that from effect studies in which animals were exposed to MNP in laboratory experiments. We have not considered the effects of larger plastic items > 5 mm associated with entanglement and ingestion, nor the isolated effects of chemical additives because these aspects are covered in other chapters (Chapters 1, 2, 4). Based on this information, major knowledge gaps were identified and recommendations put forward on how to close them.

3.1 Methods

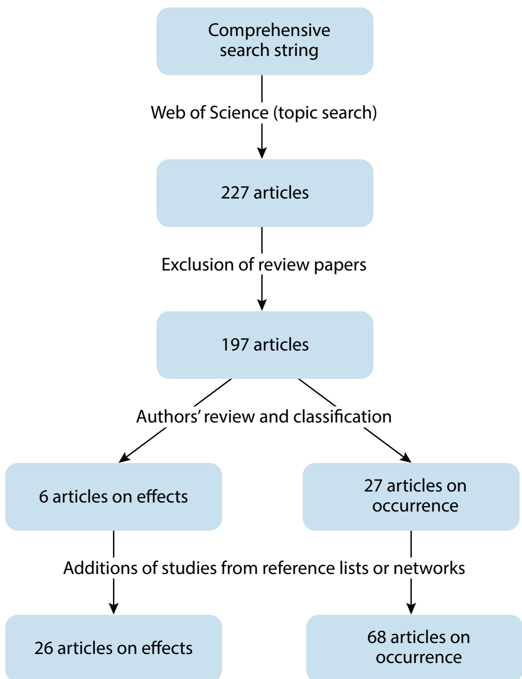
A literature search was carried out in a Web of Science topic search in April 2023, using a comprehensive combination of search string terms related to plastics, species, effects, and the Arctic region. Details are provided in the Annex.

This search string yielded 227 articles. Review articles were not included in the next step. Each of the remaining 197 articles was screened by 2 of the authors or contributors and sorted into 2 categories, i.e., studies on (1) effects and (2) occurrence (Figure 3.1). Although the focus of this review was on the effects of MNP on Arctic biota, data on the presence of MNP in Arctic animals were included as well to study whether first steps toward risk evaluations would be possible. For two groups of species, mammals and birds, no effects data were available, therefore only occurrence data are listed in the tables and in the Annex for these groups.

Thus, the literature review included laboratory effect studies on species occurring in the Arctic and occurrence studies in biota from Arctic locations. The Arctic area was defined according to AMAP (2021). Consequently, this approach excluded research on matrices other than biota, occurrence studies on animals outside the AMAP area, and studies on macroplastics or chemical additives. Gray literature, such as scientific reports, were included. In total, 6 publications were included on effects and 30 publications were included on occurrence. However, three of these did not contain original data and were therefore removed from further data analysis. Despite the comprehensive search string, an additional 41 publications of original occurrence data and 12 on effects known to the authors or listed in the references of the articles of the systematic literature search were found and added, and 8 studies on the effects of bivalves were included, resulting in 68 publications on occurrence of MP in Arctic animals and 26 publications on effects (Figure 3.1).

Selected metadata and main findings from the publications were extracted into separate tables for mammals (9 and 0 papers on occurrence and effects, respectively), birds (30 occurrence, 0 effects), fish (17 occurrence, 5 effects), and invertebrates (12 occurrence, 21 effects), refer to Table 3.1 following the main fauna groups used in the AMAP monitoring guidelines (AMAP 2021). The literature screening also identified two publications on occurrence of bacteria, algae, fungi, and cnidarians (plastisphere) on plastic particles and their effects (Rüthi et al. 2020, Xin et al. 2022). These were not included for further analysis.

Figure 3.1 Summary of the literature search.



3.2 Results and discussion

Figure 3.2 shows a geographical representation of the data on MP occurrence according to animal groups. Figure 3.3 shows which sample types of the animal groups have been analyzed. Figure 3.4 shows which effects were tested for, and if an effect was found. However, the data were of varying and often questionable quality. Studies on effects of MNP on Arctic species were not available for mammals and birds. In addition, effect data were generally of limited comparability, both among effect studies and regarding available occurrence data. For this reason, we did not prepare graphics on available MP effect studies in the Arctic, or a tentative risk evaluation. Micro- and nanoplastics are diverse in chemical composition, particle size, and shape, all of which are critical factors for MP toxicity (Kögel et al. 2020, Zimmermann et al. 2020, Gomes 2022). In general, information on MNP type, size, and concentration ranges reported in occurrence studies rarely matched the corresponding information in effect studies.

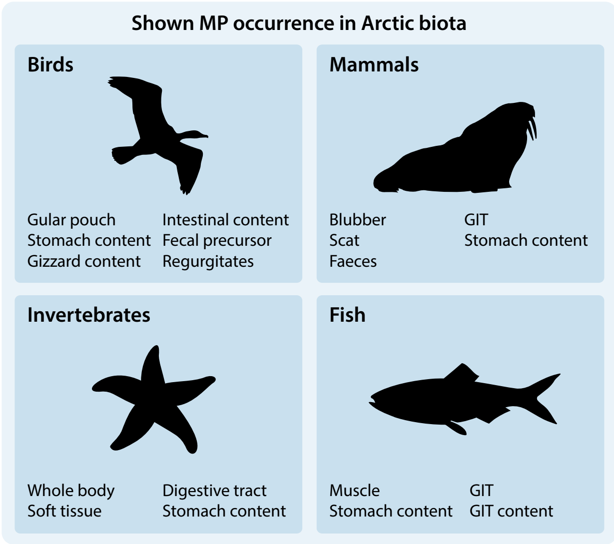


Figure 3.3 Organs, tissues, and other sample types that contained MPs in different Arctic animals. GIT: Gastrointestinal tract.

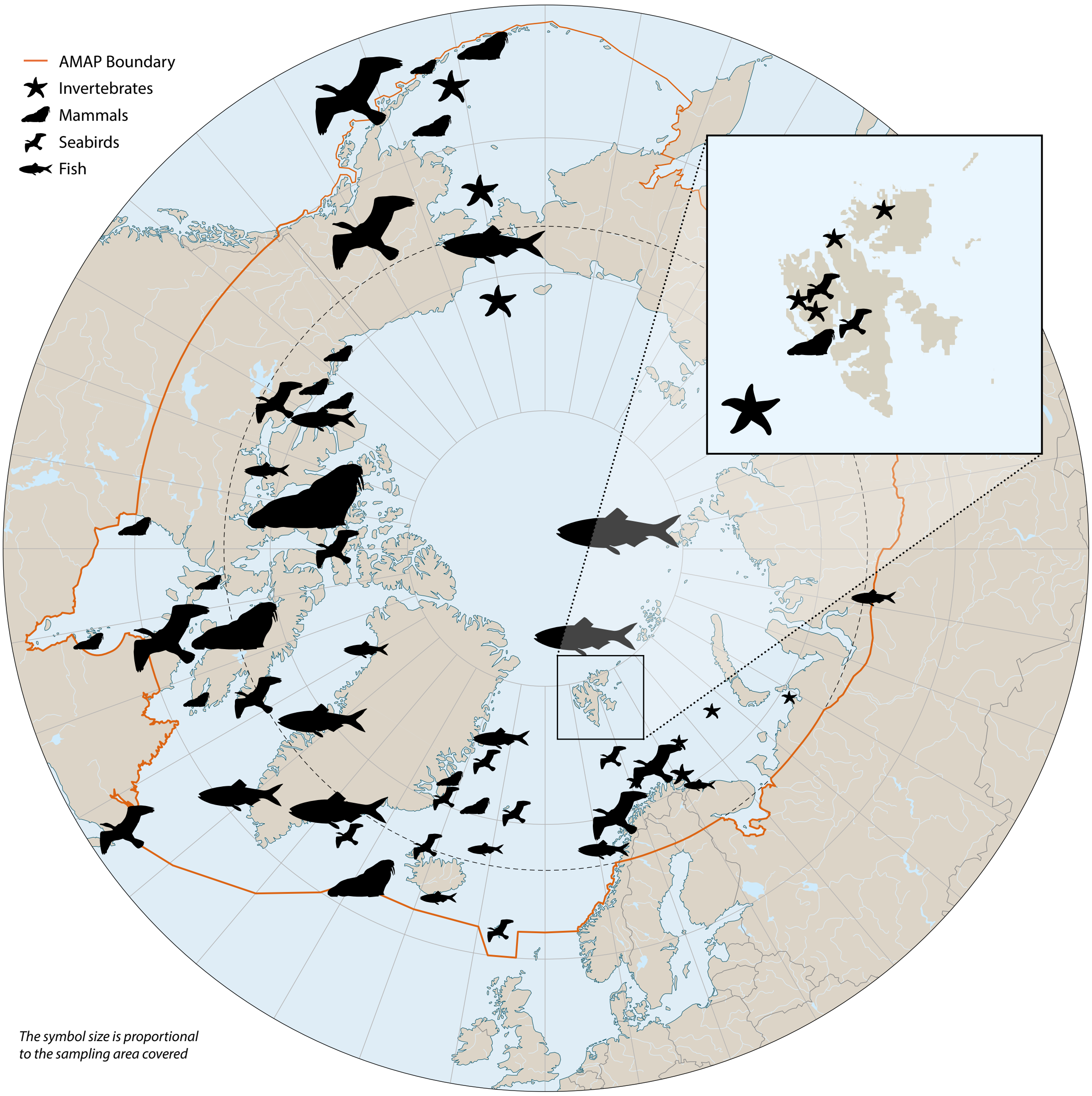


Figure 3.2 Map of the sampling locations of the studies investigating the occurrence of microplastics in biota within the Arctic region as defined by AMAP; modified from (Culp et al. 2012) and (Collard and Ask 2021). Symbols represent the four biota groups analyzed for MP occurrence in this chapter.

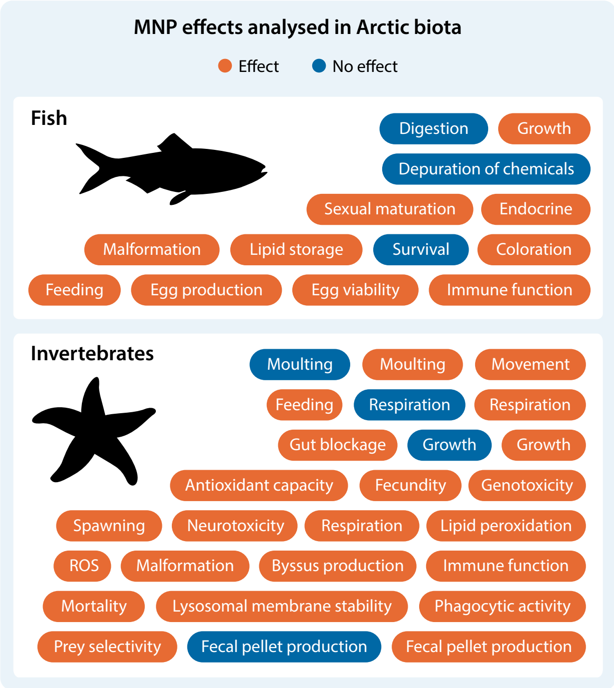


Figure 3.4 Effects analyzed for MNPs in different Arctic animals. Orange color for found effects, blue color for absence of effects.

3.2.1 Arctic mammals

3.2.1a Occurrence of microplastics in Arctic marine mammals

The occurrence of MP in Arctic marine mammals was recently reviewed by Lusher et al. 2022. Four additional publications were identified, bringing the total number of publications to nine. None of the studies investigated NP. All available studies were performed on marine mammals (Table 1; more details including methods in Annex, Table A1). Arctic marine species investigated belonged mostly to pinnipeds and cetaceans, in addition to one recent study on polar bears. Most studies were on seals, i.e., ringed seals (*Phoca hispida*) and fur seals (*Callorhinus ursinus*), followed by walrus (*Odobenus rosmarus*), and then whales, mostly fin whales (*Balaenoptera physalus*). A complete count of individuals per species is included in the Annex.

Occurrence in GIT, feces/scat, and blubber was studied. Studies on the stomach dominated the available data (N = 5); one study investigated the whole GIT and three studies analyzed feces/scat. One study analyzed the blubber of bearded seals—important subsistence resources for coastal Arctic communities (Scherdin et al. 2022) and therefore important for food safety. Most of the studies were from Alaska and the Canadian Arctic, while two studies were from Svalbard and Iceland (Annex, Table A1).

Table 3.1 Number of studies reviewed per group of Arctic biota.

	Mammals	Birds	Fish	Invertebrates
Occurrence	9	30	17	12
Effects	0	0	5	21

3.2.1b Methods applied for analyzing occurrence of microplastics in Arctic animals

The methods for sample preparation and MP detection varied (Table A1, and summary in the Annex), with implications for the lowest MP size limit and general comparability between studies. Furthermore, quality assurance/control (QA/QC) measures varied between studies and did not always address both contamination (via blanks) and losses (via recovery tests), i.e., risks of false positives and false negatives. Studies in which blanks were included found that fibers were present across samples, highlighting the challenges of accounting for and preventing procedural contamination while working with MP, especially in the field, as reported previously from temperate and tropical regions (Lusher and Hernandez-Milian 2018, Panti et al. 2019). In three of the studies in which blanks were analyzed, the amount of MP in samples did not exceed the amounts in procedural laboratory blanks, or field blanks. Only two studies performed positive controls/recovery tests. These are recommended for future studies because they can provide an indication of the probability of underestimating the MP content with the chosen methodology.

3.2.1c Pinnipeds – consideration of scat analysis

Among the five studies on pinnipeds, the three studies on stomachs of seals and walrus did not find any MP or no MP content that was different from blanks. Note that the lowest detectable size limit was roughly 0.5 mm or 1 mm, i.e., smaller particles might have been overlooked. Power analyses performed in two studies on walrus and seal identified that the sample sizes (N = 36 and 10) were too low to determine a relationship between particle presence (> 0.5 or 1 mm) in the stomachs compared to the procedural blanks (Jardine et al. 2023a, b). Therefore, higher sample numbers will be necessary if using stomach analysis in pinnipeds in less contaminated areas with size limits > 0.5 mm. Scat analysis presented a different picture: for fur seals from Alaska, 0–86 MP (polyethylene) were detected per scat, with a frequency of occurrence (FO) of 55% for the lowest size limit of 250 µm (Donohue et al. 2019). The numbers of fragments and fibers were higher in the size range < 1 mm (N = 325 and 39, respectively) compared to those > 1 mm (N = 73 and 32). Although the difference was most pronounced for fragments, the data again emphasize the importance of analyzing smaller MP. Regional differences were demonstrated but not statistically evaluated. For walrus, MP were found in feces at 34 MP/kg (Carlsson et al. 2021), but not in stomachs (Jardine et al. 2023a). Polar bear scat was found to contain 0–1.4 MP/g dry weight for MP > 100 µm. Comparability is also limited here due to different size ranges and units. However, the mean for polar bear was < 1 MP/g, which is consistent with the data for walrus, which contained 0.034 MP/g, but does not provide a sufficient resolution for detailed comparison.

Differences in the lowest detectable size limits prevailed and are bound to have an impact on the detected total and grouped numbers of MP. Several studies present data for MP according to size classes, but do not use the same size classes for grouping, rendering it impossible to compare the data. Therefore, we recommend making the metadata and study details (e.g., actual size of all MP detected) available in linked databases and/or supplementary material.

3.2.1d Whales – investigations found microplastics in digestive tracts and blubber

The three studies on whales also differed in species, target organs/tissues, and methodology for MP analysis, including reporting unit (Table 3.2). One study on fin whales found 57 MP/kg in krill retrieved from the stomach of the whales (Garcia-Garin et al. 2021). The MIKRONOR study of fin whales found two MP (ca. 0.1 MP/g) in a blubber sample (Merrill et al. 2023). Beluga whales contained 18–147 MP/GIT with an FO of 100%, or 1 MP in a blubber sample. One gray whale was shown to contain 9.5 µg PVC/g of blubber and another 3 MP in a blubber sample, whereas the blubber of one seal contained a lower amount of PVC, i.e., 0.59 µg/g in the same study (Table 3.2). Further investigated whale species contained less plastic. However, one needs to keep in mind that only one individual sample per species was investigated (Merrill et al. 2023). Even though the numbers of MP are not high, MP occurrence seems to be ubiquitous in whales and seems to translocate to blubber to a larger degree than in pinnipeds. This could have implications for human exposure in regions where whale blubber is consumed. More details on the methods of the occurrence studies can be found in Annex, Table A1.

3.2.1e No data on the effects of microplastics in Arctic mammals

No data were available on the effects of MP in Arctic marine or larger mammals. Laboratory studies on MNP effects on mice have assessed pathological changes in the gut and liver including reduced mucus secretion, gut barrier dysfunction, microbiota changes, inflammation, oxidative stress, lipid changes, gastrointestinal and hepato-toxicity, reproduction disorders, neurotoxic effects as well as metabolic disorder, and showed particle accumulation in several tissues (Banerjee and Shelver 2021, da Silva Brito et al. 2022). Also, although uptake into organs should be most frequent for particles below 10 µm (da Silva Brito et al. 2022), there are hardly any data on MNP occurrence including that size range (Table 3.2). Even though there are many different plastic polymer types in the environment, most exposure studies have only investigated polystyrene or polyethylene (Banerjee et al. 2021). So far, data are insufficient to rank endpoints according to their sensitivity and to determine predicted no effect concentrations (PNECs) for risk assessments (Liu et al. 2023).

Table 3.2 Microplastic occurrence in Arctic mammals.

Species	N	Lowest MP size limit	Amount FO%	Matrix	Reference
Pinnipeds					
Fur seal (*Callorhinus ursinus*)	35	250 µm	0–86 MP per scat; FO 55%	Scat	Donohue et al. 2019
Ringed seal (*Pusa hispida*) Bearded seal (*Erignathus barbatus*) Harbor seal (*Phoca vitulina*)	135 6 1	425 µm	None	Stomach content	Bourdages et al. 2020
Walrus (*Odobenus rosmarus*)	8	500 µm	34 MP/kg FO 87.5%	Feces	Carlsson et al. 2021
Atlantic walrus (*Odobenus rosmarus rosmarus*)	36	80 µm, visible MP	no difference to control	Stomach	Jardine et al. 2023a
Ringed seal	10	80 µm, visible MP	no difference to control	Stomach	Jardine et al. 2023b
Bearded seal	1	py-GC/MS 1 µm	0.59 µg/g	Blubber	Merrill et al. 2023
Whales					
Beluga whale (*Delphinapterus leucas*)	7	20 µm	18–147 MP/GIT; FO 100%	GIT	Moore et al. 2020
Fin whale (*Balaenoptera physalus*)	25	1.2 mm	57 MP/kg krill	Stomach content	Garcia-Garin et al. 2021
Gray whale (*Eschrichtius robustus*) Raman: Beluga whale, Humpback whale (*Megaptera novaeangliae*), Gray whale, Fin whale, Minke whale (*Balaenoptera acutorostrata*)	1 4	py-GC/MS 1 µm Raman: handled with forceps	Py-GC/MS: gray whale: 9.5 µg/g Raman: 0.05–0.19 MP/g	Blubber	Merrill et al. 2023
Bears					
Polar bear	15 scats, 15 colon feces	100 µm	0–1.4 MP/g d.w. mean < 1 MP/g	Feces	Iyare et al. 2024

GIT = gastrointestinal tract (oesophagus, stomach, intestines), MP = microplastic, FO = frequency of occurrence. FO partially calculated by authors of this report. py-GC/MS = Pyrolysis gas chromatography – mass spectrometry, d.w. = dry weight. For more details, see Annex, Table A1.

3.2.1f Potential human exposure

Because muscle, liver, or blubber tissues of marine mammals can be part of traditional food in the Arctic, MP occurrence in these tissues could lead to human exposure. Thus, more knowledge about smaller particles that reach these tissues would be relevant. Currently, studies on MP in Arctic pinniped stomachs mainly target particles > 500 µm. The MP of this size are unlikely to enter other tissues via translocation (EFSA Panel on Contaminants in the Food Chain 2016). They are likely retained in the GIT or egested with other waste material, which is supported by the studies on feces of Arctic animals. Arctic terrestrial mammals, such as reindeer/caribou (*Rangifer tarandus*), which are important for human consumption need to be investigated for the presence of MP. If one regards the rodent studies as model studies for human impacts, these studies also imply that health effects might occur if humans are exposed to MP.

3.2.1g Summary: microplastics in Arctic mammals

In summary, limited evidence is available on MP presence from gut, tissue, and feces of Arctic marine mammals, and no direct data exist on effects. The data in feces suggest that at least the larger MP are egested with feces, while smaller particles still need to be better investigated for potential transfer to other tissues. However, it is noteworthy that MP are detected in the blubber of whales and seals, with higher levels in whales. The sparsity of the data and heterogeneity of methods and reporting prevent comparisons across the studies. Research should focus on consolidating a harmonized investigative approach and ensuring the publication of details and metadata for better comparability.

3.2.2 Arctic birds

Since the 2021 AMAP monitoring guidelines, 34 more publications on MP occurrence in Arctic birds have been found, thus this report has a total of 48 publications (Table 3.3 and Annex, Table A2).

3.2.2a Northern Fulmar as a bioindicator

Seabirds constitute the most studied Arctic vertebrate groups with regard to plastic occurrence, with most studies on the Northern Fulmar (Table 3.3; more details including methods in Annex, Table A2). As described in the introduction, the Northern Fulmar is a bioindicator for plastic pollution under OSPAR (OSPAR Commission 2008) and also suggested for the MSFD. It has also been recommended as a Priority 1 indicator in the Arctic (AMAP 2021). The fulmar is one of the bird species investigated from the Arctic with the highest plastic contamination. Its feeding strategy and morphology explain its susceptibility to plastic accumulation. However, one study found no plastic in 233 fulmar stomachs from northern Devon Island, in Canada, west of Greenland in 2010 (Byers et al. 2010). The circumpolar distribution of the Northern Fulmar simplifies geographical comparisons of plastic ingestion.

3.2.2b Feeding strategies play a pivotal role for plastic occurrence in birds

The fulmar is an opportunistic pelagic sea surface-feeder with two stomachs separated by a narrow constriction, preventing it from regurgitating the content of the second stomach, the gizzard (van Franeker et al. 2011). The main role of the gizzard is to grind down hard items such as squid beaks, until they are small enough to pass through the pylorus and reach the intestine. Hard items, such as plastic pieces, are also stuck in the gizzard until ground down. Plastic ingestion by Arctic seabirds was reviewed recently (Baak et al. 2020a, Collard and Ask 2021), with more studies published in recent years (Table 3.3). Studies investigating Arctic birds other than fulmars are mostly restricted to the Northeast Canadian Arctic. Most of those species showed lower plastic occurrence than the fulmar and often had an absence of plastic in the stomach or digestive tract.

Other studies reporting high to intermediate FO% (> 25) of plastic were on Short-tailed Shearwater (*Puffinus tenuirostris*), Sooty Shearwater (*Puffinus griseus*), Fork-tailed Storm Petrel (*Oceanodroma furcata*), Leach's Storm Petrel (*Hydrobates leucorhous*), Parakeet Auklet (*Aethia psittacula*), Cassin's Auklet (*Ptychoramphus aleuticus*), Crested Auklet (*Aethia cristatella*), Northern Pharalope (*Phalaropus lobatu*), Horned Puffin (*Fratercula corniculate*), Tufted Puffin (*Fratercula cirrhata*), Little Auk/Dovekie (*Alle alle*), Thick-billed Murre (*Uria lomvia*), Glaucous-winged Gull (Larus glaucescens), Black-legged Kittiwake (*Rissa tridactyla*), Pelagic Cormorant (*Phalacrocorax pelagicus*), and Red-faced Cormorant (*Phalacrocorax urile*). In this list, Short-tailed Shearwater, Fork-tailed Storm Petrel, and Parakeet Auklet are interesting because all available reports (7, 2, and 4, respectively) show high FO% over 75% and no lower FO% were reported for these species. For comparison, with Northern Fulmar, there were 11 reports with FO% > 75, but also 12 between 25% and 74%, and 3 below 25% (Table 3.3 and Annex, Table A2).

Inter-species differences in plastic contamination are likely due to different feeding strategies because the pursuit-diving seabirds (e.g., Thick-billed Murre, N = 2380, 9 studies) and the Common Murre (*U. aalge*; N = 436, 6 studies) ingested much less plastics than surface feeders or dipping feeders such as the fulmar or the Short-tailed Shearwater (N = 651, 8 studies), gulls, or the Black-legged Kittiwake (N = 520, 8 studies; Table 3.3; Provencher et al. 2014, Poon et al. 2017, Baak et al. 2020a). However, the auklets with high FO% and cormorants and puffins with intermediate FO levels, which are all pursuit divers, do not follow this pattern, but the amount of data is too small to be certain. For some seabird species, plastic debris can resemble their prey; these seabirds tend to ingest more plastic. For example, the Little Auk/Dovekie (N = 128, 4 studies) feeds underwater while swimming and captures its prey by suction-feeding (Enstipp et al. 2018). Little Auks/Dovekies have not been studied as extensively as the Northern Fulmar in the context of plastic pollution, but the most recent study showed 100% FO of MP in gular pouch samples (Amélineau et al. 2016). The authors argued that the size, color, and even the shape of the MP ingested could have led to confusion with the birds' prey, *Calanus* copepods.

Table 3.3 Microplastic occurrence in Arctic birds.

Species	N	Lowest MP size limit	Amount, FO%	Matrix	Reference
37 species	1968	1.2 mm	0–6.17 MP/ind. 0–0.24 g/ind. FO > 75% in Short-tailed Shearwater, Fork-tailed Storm Petrel, and Parakeet Auklet, FO 50–74% in Northern Fulmar and Northern Phalarope; FO 25–49% in Leach's Storm Petrel, Sooty Shearwater, Cassin's Auklet, and Horned Puffin	Stomach content	Day 1980
Parakeet Auklet	unknown	Visible	"Almost all birds had plastic." 0–40 MP/ind	Stomach content	Mikhtaryantz 1981
9 species	62	Visible	FO 36% in Northern Fulmar, FO 0% in other species	Stomach content	Mehlum and Giertz 1984
Northern Fulmar	51	Visible	FO 4.5–4.7 %	Stomach content	van Franeker 1985
Little Auk/Dovekie White-winged Scoter (*Melanitta deglandi*), Barrow's Goldeneye (*Bucephala islandica*), Greater Scaup (*Aythya marila*), Harlequin Duck (*Histrionicus histrionicus*), Long-tailed Duck (*Clangula hyemalis*), Surf Scoter (*Melanitta perspicillata*)	408	Visible	"present" in Little Auk/Dovekie, FO 0% in other species	Stomach content	Day et al. 1985
Northern Fulmar Black-legged Kittiwake	26	Visible	FO 50% Not mentioned	Stomach content	Gjertz et al. 1985
Little Auk/Dovekie Thick-billed Murre Northern Fulmar Black-legged Kittiwake Common Eider (*Somateria mollissima*), Atlantic Puffin (*Fratercula arctica*), Black Guillemot (*Cepphus grylle*), Glaucous Gull (*Larus hyperboreus*)	144	Visible	FO 45% FO 24% FO 15% FO 5% FO 0%	Stomach content	Lydersen et al. 1989 Lydersen et al. 1989
Thick-billed Murre	202	Visible	23 MP < 10 mm	Stomach content	Falk and Durinck 1993
Northern Fulmar Common Eider, Black Guillemot, Arctic Tern (*Sterna paradisaea*), Glaucous Gull	25	Visible	FO 20%, 0.02 g/ind. FO 0%	Stomach content	Weslawski et al. 1994
24 species	1851	500 µm	FO 0–93.8%; FO > 75% in Short-tailed Shearwater, Fork-tailed Storm Petrel, and Parakeet Auklet, 50–74% in Northern Fulmar; 25–49% in Sooty Shearwater, Leach's Storm Petrel, Cassin's Auklet, and Horned Puffin	Stomach content	Robards et al. 1995
Northern Fulmar	22	Visible	FO 82% (1–21 items)	Stomach contents	Camphuysen and van Franeker 1997
Little Auk/Dovekie	104	Visible	FO 8.65%	Stomach contents	Pedersen and Falk 2001
Short-tailed Shearwater	330	Visible	FO 84%	Stomach content	Vlietstra and Parga 2002
Herring Gull (*Larus argentatus*)	187	Visible	FO 19.8%	Stomach content	Turovskaya and Nichkevich 2005
Northern Fulmar	42	Visible	1.3 MP/ind. 0.02–0.31 g/ind. FO 36%	Stomach content	Mallory 2006
Northern Fulmar	102	Visible	7.4 MP/ind. 0.31 g/ind. FO 31%	Stomach content	Mallory 2008
Northern Fulmar	25	Visible	5.6 MP/ind. 0.094 g/ind. FO 85%	GIT content	Provencher et al. 2009

Species	N	Lowest MP size limit	Amount, FO%	Matrix	Reference
Northern Fulmar Black Guillemot	278	Visible	None	Stomach content	Buyers et al. 2010
Great Skuas (*Stercorarius skua*)		Visible	MP in 12 pellets from 5 nests	Regurgitated pellets	Knutsen 2010
Thick-billed Murre	186	Visible	0.2 MP/ind. 0.0016 g/ind. FO 11%	GIT content	Provencher et al. 2010
Short-tailed Shearwater	99	Visible	Mean: 15.1 (± 2.9) MP, 0.23 g/ind. Fragment > resin pellet/sheet > fiber > foam	Stomach content	Yamashita et al. 2011
Herring Gull	306	Visible	MP "present"	Stomach content	Tolmacheva 2012
Northern Fulmar	647	1 mm	15 MP/ind. 0.21 g/ind. FO 44%	Stomach content	van Franeker 2012
Northern Fulmar	58	1 mm	6.0 MP/ind. 0.13 g/ind.	Stomach content	Kühn and van Franeker 2012
Common Murre Thick-billed Murre	1662	Not disclosed, visible	Specified per period	GIT content	Bond et al. 2013
Short-tailed Shearwater	12	Visible	FO 100% 0.04–0.59 g per bird	Stomach content	Tanaka et al. 2013
Northern Fulmar Atlantic Puffin Thick-billed Murre Northern Common Eider 13 more species	1580	1 mm	FO 51% FO 13% FO 2% FO < 1% FO 0%	GIT content	Provencher et al. 2014
Short-tailed Shearwater, Parakeet Auklet Northern Fulmar, Thick-billed Murre Common Murre/Common Guillemot, Tufted Puffin, Whiskered Auklet (*Aethia pygmaea*), Red-legged Kittiwake (*Rissa brevirostris*)	25	visible	FO 100% FO 25–49% FO 0%		Artukhin 2014
Northern Fulmar Faroe Islands and Iceland	67	1 mm	FO 33.3%, 0.23 ± 0.35 g/ind., 13.9 ± 29 MP/ind., 0–152 FO 90%, 0.12 ± 0.02 g/ind.	Stomach content	Trevail et al. 2014
Northern Fulmar	40	1 mm	FO 87.5% 0.08 g/ind.	Stomach content	Trevail et al. 2015
Little Auk/Dovekie	44	0.1 mm	8.99 and 9.99 MP/ind. FO 100%	Gular pouch content	Amélineau et al. 2016
Great Skua	1034	Not disclosed, visible	FO 6%	Regurgitated pellets	Hammer et al. 2016
Northern Fulmar	75	1 mm	FO 81%	Stomach content	Herzke et al. 2016
Northern Fulmar Black-legged Kittiwake Thick-billed Murre Black Guillemot	53	1 mm	3.4 MP/ind. 0.18 MP/ind. 0 MP/ind. 0 MP/ind.	GIT content	Poon et al. 2017
Northern Fulmar	30	1 mm	1.9 MP/ind. FO 47%	Last 10 cm intestine, fecal precursor	Provencher et al. 2018
Northern Fulmar	70	1 mm	11.6 MP/ind. FO 79%	Stomach content	Avery-Gomm et al. 2018
Short-tailed Shearwater	1	Visible	FO 100%, 6 MP/ind.	Stomach content	Golovnyuk et al. 2019
Short-tailed Shearwater	2	Visible	FO 100%	Stomach content	Solovyeva et al. 2020

Species	N	Lowest MP size limit	Amount, FO%	Matrix	Reference
Northern Fulmar	86	1 mm	FO 87.3%, 0.18 g/ind., 11.4 MP/ind. *Faroese* FO 90.3%, 0.06 g/ind., 6.16 MP/ind. *Greenland*	Stomach content	Ask et al. 2020
Northern Fulmar Black-legged Kittiwake Thick-billed Murre Black Guillemot	164	1 mm	FO 72%; 1.7MP/ind. FO 15%; 0.2 MP/ind. FO 0% FO 0%	GIT content	Baak et al. 2020b
Crested Auklet Tufted Puffin Northern Fulmar Glaucous-winged Gull Black-legged Kittiwake Pelagic Cormorant Horned Puffin Red-faced Cormorant Common Murre Pigeon Guillemot (*Cepphus columba*)	75	Not disclosed, visible	FO 70% FO 63% FO 57% FO 50% FO 45% FO 33% FO 30% FO 30% FO 16.7% FO 0%	Stomach content	Padula et al. 2020
Northern Fulmar Thick-billed Murre	57	800 µm	1.8 MP/ind. 0 MP/ind.	GIT content	Bourdages et al. 2021
Glaucous Gull	21	1 mm	14.3% FO 0.14 MP/ind. 0.008 g/ind.	GIT content	Benjaminsen et al. 2022
Northern Fulmar	145	1 mm	13.5 MP/ind. 0.14 g/ind., FO 86%	Stomach content	van Franeker et al. 2022
Northern Fulmar	43	1 mm	10.3 MP/ind. 0.07 g/ind.	Stomach content	Collard et al. 2022a
Northern Fulmar	20	1 mm	12.4 MP/ind. 0.15 g/ind., FO 95%	Stomach content	Collard et al. 2022b
Northern Fulmar	39	1 mm	36.1 MP/ind. 0.21 g/ind. 0 (1 piece in total)	Stomach content; intestine content	Tulatz et al. 2023
Black-legged Kittiwake (2008), Prince Leopold Island Black-legged Kittiwake (2021), Qikiqtarjuaq Black-legged Kittiwake (2021), Pond Inlet Black Guillemot	67		FO 20%, 0.004 g/ind., 2 MP/ind. FO 17%, 0.003 g/ind., 2.7 MP/ind. FO 0%	GIT content	Baak et al. 2024

For more details, see Annex, Table A2. MP = microplastic, FO = frequency of occurrence, ind. = individual. FO% > 50 and > 25 are highlighted in red and orange, respectively.

3.2.2c Influence of analysis methodology and investigated tissue, organ, or body fluid

According to Provencher et al. (2014), the Little Auk/Dovekie had not ingested plastic at all when sampled in the 1980s. However, Amélineau et al. (2016) analyzed smaller-sized plastic pieces, making the studies difficult to compare. The reason for differences in plastic counts may be heavily influenced by the methodology used, for example, if a sieve or microscope is used (Avery-Gomm et al. 2016). Plastic ingestion is most often studied in the stomach or the whole GIT content of the birds. Limited data exist on the occurrence and presence of plastics in the intestine, but portions of the small intestine and the cloaca, gular pouch, and regurgitated pellets were represented in the reviewed studies (Table 3.3). Provencher et al. (2018) showed a positive correlation between the number of MP in the stomach and in the fecal precursors of Northern Fulmars, with differences in the shape, proportion of user plastic vs. industrial plastic, and FO. This shows the importance of investigating the intestinal content, which provides complementary information to that acquired through the analysis of the stomach content. More knowledge of MP in the intestinal content will also help in assessing the retention time in the stomach(s) and also provide information on the ability of the birds, and especially the fulmar, to grind down the hard plastic pieces in the gizzard allowing for their evacuation. Furthermore, some birds are part of a traditional Arctic diet, making it relevant to study MP occurrence in tissues used for consumption, including whole birds as in the case of Little Auks/Dovekies (Johansen et al. 2004, Ebel 2019).

3.2.2d Harmonization benefits and caveats

The size of plastics extracted and quantified in Arctic seabirds has been harmonized, especially in recent years. Based on the bioindicator introduced by OSPAR (OSPAR Commission 2008), researchers usually focus on plastics larger than 1 mm at their maximum dimension to improve comparability between studies. However, this could be a drawback because smaller MP have been disregarded in scientific research, resulting in a scarcity of data on plastic pieces below 1 mm in Arctic birds. None of the studies investigated NP. Species that do not ingest plastic greater than 1 mm in size may still contain MP and NP (Bourdages et al. 2021). Depending on the species and on its prey, including smaller MP would be relevant, particularly regarding the potential effects, tissue distribution, and indirect ingestion through contaminated prey. Different size classes can be reported separately to still allow for comparison across studies.

3.2.2e Effect studies outside the Arctic indicate microplastic-induced tissue damage and inflammation as well as changes in body mass

Although an increasing amount of data is being generated on plastic occurrence in seabirds, the impacts are far from understood. Considering the great number of plastic pieces in the gizzard of some fulmars (Collard et al. 2022a, van Franeker et al. 2022, Tulatz et al. 2023), such contamination potentially impairs and slows down the normal process of food digestion from the first stomach, the proventriculus, until the intestine. However, this assumption still needs to be confirmed.

Effects on seabirds have been observed in regions other than the Arctic: studies performed in Australia on Flesh-footed Shearwater fledglings (*Puffinus carneipes*) evidenced direct impacts due to the occurrence of plastic in the stomach (Charlton-Howard et al. 2023, Rivers-Auty et al. 2023). Charlton-Howard et al. (2023) found a significant and positive correlation between the number of ingested plastics and the extent of fibrosis in the stomach lining, in contrast to effects from the occurrence of natural hard items such as pumice. Rivers-Auty et al. (2023) examined the spleen, the kidney, and the proventriculus. All three organs had embedded MP, their amounts correlating with macroplastic exposure, and had suffered tissue damage. Flesh-footed Shearwaters that had ingested plastics displayed significant inflammation in the proventricular (especially in the inferior) region (Rivers-Auty et al. 2023). Fibrosis in the proventriculus was also observed in birds with a high plastic burden. Such plastic-induced fibrosis has now been called “plasticosis” (Charlton-Howard et al. 2023). The relationship between plastic numbers and tissue damage in the spleen or kidney was less obvious and might have originated from the stress induced by the proventriculus pathology (Rivers-Auty et al. 2023). In an Arctic context, the fulmar would be the candidate of choice for such impact studies, given the numbers of plastic they ingest.

Experimental studies on the terrestrial Japanese Quail (*Coturnix japonica*) all showed effects on body mass or growth rates (Roman et al. 2019, de Souza et al. 2022, Monclús et al. 2022). They reported subtle effects, for example in terms of biochemical alterations (e.g., reflecting oxidative stress and redox imbalances; de Souza et al. 2022), and induction of enzyme activities (Monclús et al. 2022), the latter with interesting differences for smaller and larger MP particles. Roman et al. (2019) concluded that although endocrine effects were observed in a multi-generational study, there were no lasting effects such as mortality or changes in reproductive outputs.

3.2.2f Summary: microplastics in Arctic birds

In summary, the monitoring of plastic pollution via Northern Fulmar as a bioindicator is established in the OSPAR region and in other parts of the Arctic, and it has generated data that allow trend analyses (van Franeker et al. 2011, Baak et al. 2024). Data on other bird species provide complementary information that suggests a link in the plastic occurrence to the birds’ feeding habits, migration, and general ecology. However, no such information is available regarding effects on Arctic birds. Laboratory studies and field studies outside the Arctic have indicated changes in various biomarkers and physiological effects on organs, but it is unknown whether these observations can be transferred to other species that likely experience a different exposure situation.

3.2.3 Arctic fish

3.2.3a Occurrence in Arctic fish

The occurrence of MP in Arctic fish has been previously reviewed (Kögel et al. 2023). In this updated report, we have included three additional publications on the occurrence of MP (Table 3.4; more details including methods in Annex, Table A3; Frank et al. 2020, 2023, Wardlaw et al. 2022). None of the studies investigated NP. The studies cover 12 marine and 4 freshwater species, the latter from Russian and Canadian rivers and lakes. Atlantic cod is the best studied species, including 1627 individuals. For a more detailed summary, see the Annex.

3.2.3b Microplastic occurrence in fish is mainly studied in the gastrointestinal tract – first results in muscle tissue

The vast majority of the included studies addressed MP in the GIT of the fish. One study (Hamilton 2023) investigated MP in the muscle tissue of Arctic char (*Salvelinus alpinus*) and lake trout (*Salvelinus namaycush*) in the central Canadian Arctic, reporting an average of 73.8 µg/g for MP in Arctic char and 6.01 µg/g in lake trout. These studies suggest translocation of MP into the fillets of Arctic fish. Interestingly, Hamilton (2023) did not observe bioaccumulation of MP within the fish fillet, in terms of increasing amounts over time, which could be the result of growth dilution or recirculation of particles within an organism. Ding et al. (2023) reported more MP in older fish when investigating GIT content in Alaska pollock (*Gadus chalcogrammus*), but MP residence in the fish gut is transitory and might therefore not reflect bioaccumulation but feeding amounts. Although translocation and plastic burden in fish tissue may be species dependent, it does highlight the need for further data on translocation and bioaccumulation of MP in wild, northern fish (Hamilton et al. 2022, Kögel et al. 2023), which are often consumed in northern communities.

Table 3.4 Microplastic occurrence in Arctic fish.

Species	N	Lowest size limit MP	Amount, FO%	Matrix	**Reference**
Greenland shark (*Somniosus microcephalus*)	30	1 mm	0–1 MP/fish, FO 3.33%	Stomach content	Nielsen et al. 2014
Atlantic cod (*Gadus morhua*)	302	3.2 mm	0–8 MP/fish, FO 18.8%	Stomach content	Bråte et al. 2016
Atlantic cod	71	1 mm	0–2 MP/ fish, FO 2.4%	GIT content	Liboiron et al. 2016
Juvenile polar cod (*Boreogadus saida*)	72	650 400 μm 590* 170 μm	2 MP in total, FO 2.8%	GIT content	Kühn et al. 2018
Bigeye sculpin (*Triglops nybelini*) Polar cod	71 85	90 μm	1–2 MP/fish sculpin FO 34%; polar cod FO 18%	GIT	Morgana et al. 2018
Atlantic cod Atlantic salmon (*Salmo salar*) Capelin (*Mallotus villosus*)	1010 69 350	1 mm	0–2 MP/fish, salmon FO 0%, capelin FO 0% , cod FO 1.68%	GIT content	Liboiron et al. 2019
Atlantic cod Saithe/pollock (*Pollachius virens*)	39 46	80 μm	0.23 MP/cod 0.28 MP/saithe cod: FO 20.5%; saithe: FO 17.4%	GIT	de Vries et al. 2020
Greenland cod (*Gadus ogac*)	9	20 μm	12 ± 6 MP/fish, 100% FO	GIT	Granberg et al. 2020
Atlantic cod	205	1 mm	0–1 MP/fish, FO 1.4%	GIT content	Saturno et al. 2020
Common dace (*Leuciscus leuciscus L.*)	13	Not provided. Filtration: 0.45 μm	204 ± 28.7 MP/fish, FO 100%	GIT	Frank et al. 2020
Atlantic mackerel (*Scomber scombrus*)	50	100 μm	1.3 MP/fish, FO 23%	GIT	Malinen 2021
Arctic/polar cod Saffron cod (*Eleginus gracilis*) Arctic cisco (*Coregonus autumnalis*) Four-horn sculpin (*Myoxocephalus quadricornis*) Capelin	20 26 26 7 28	15 μm	0.18–0.7 MP/fish FO 19–43%	GIT	Moore et al. 2022
White sucker (*Catostomus commersonii*) Common carp (*Cyprinus carpio*)	173 58	34.5 μm	White sucker: 0–14 (0.59) MP/fish, FO 44%; Common carp: 0–128 (49) MP/fish, FO 31%	GIT	Wardlaw et al. 2022
Siberian dace (*Leuciscus baicalensis*)	40	150 μm	1.55 MP/fish, 43.1 MP/kg, FO 60%	GIT	Frank et al. 2023
Alaska pollock	80	26 μm	2.7 ± 2.8 (0–14) MP/ fish, FO 85%	GIT	Ding et al. 2023
Arctic char Lake trout	20 7	1 μm	73.8 μg/g (char) 6.01 μg/g (lake trout) FO 100%	Muscle	Hamilton 2023
Arctic char	20	45 μm	26 MP/fish, FO 95%	GIT	Hamilton et al. 2024

GIT = gastrointestinal tract, MP = microplastic, FO = frequency of occurrence, ind. = individual.

3.2.3c Limited comparability – results influenced by methods – and lack of recovery studies, information on small MPs and NPs, and occurrence in tissue

As discussed for mammals and birds, comparability across studies is also limited for fish because important parameters (e.g., lowest detectable size) vary among the studies (Table 3.4). This, along with other method parameters, influences the FO, which ranged between 0 and 100% in the fish studies (Table 3.4). Studies with lowest detectable size > 600 μm reported FO% of 0–18%, while those including size classes < 20 μm reported FO% of 19–100%. None of the studies with FO% above 85% had a cut-off size > 45 μm. Hence, FO%, in the datasets published so far, may reflect the method rather than the species or location investigated.

Only the most recent studies on fish have carried out recovery experiments. The sparseness of data on small MP and on their occurrence in tissues other than the GIT is a critical knowledge gap because studies from regions other than the Arctic have shown that small MP particles can be found in fillets and livers of different fish (Collard et al. 2017, Abbasi et al. 2018, Akhbarizadeh et al. 2018, Gomiero et al. 2020). The Arctic study by Hamilton (2023) suggested that this was also the case for

Arctic fish. The particles found in fish tissue were mostly below 50 µm (Gomiero et al. 2020). The highest reported average MP concentration identified so far outside the Arctic in fish fillets was 18.5 MP/g, for the fish species *Pangasius indicus* from the Persian Gulf (Akhbarizadeh et al. 2018). However, the study did not report crucial QA/QC information, such as the levels in procedural blanks, so the results might be overestimated (Primpke et al. 2023). More information on the methods applied in the occurrence studies can be found in Annex, Table A3.

3.2.3d Effects shown in studies with long exposure time, high concentrations, or nanoplastics

We retrieved five research papers on effects of MNP in fish species endemic to the Arctic (Table 3.5, more details in Annex, Table A4). Four of them had exposed fish to MP. Two studies with shorter exposure times of one month and nine weeks, respectively, found no effects on rainbow trout (*Oncorhynchus mykiss*; Rummel et al. 2016) or limited effects (increased feeding and condition factor) on white sturgeon (*Acipenser transmontanus*; Rochman et al. 2017). Size distributions and MP concentrations from these studies can be considered environmentally relevant. However, it is extremely difficult to estimate an environmentally realistic MP concentration for Arctic waters. Recent publications reported 1.24 MP/l (Wu et al. 2024) and 11 MP/l on average, with higher concentrations at the coast (Emberson-Marl et al. 2023). In a sea ice core, Peeken et al. (2018) found several thousand MP particles per liter, potentially causing exposure in a case of melting. A recent study on fathead minnow (*Pimephales promelas*) exposed to MP for six months (a long exposure time) found many and severe effects at early life stages (Bucci et al. 2024). In this study, 100 and 2000 MP/l were employed and may not exceed concentrations over safety margins when considering more contaminated areas. Another study, in which Atlantic cod was exposed to a PE-containing diet (1% PE added to the feed, size range 300–600 µm), showed effects on gonadal and liver expression of steroidogenic enzyme 20β-hydroxysteroid dehydrogenase (20β-hsd) and vitellogenin1 (vtg1), which are indicators of disrupted sexual maturation (Fernández-Míguez et al. 2023). In the work of Fernández-Míguez et al. (2023), the concentration of 1% PE in the diet fed was high but compared to some reported levels in mussels and amphipods, not far off (Iannilli et al. 2019, Teichert et al. 2021). However, the size of the applied MP, 300–600 µm, was entirely in the larger range of environmental sizes found in animals (Fernández-Míguez et al. 2023). This study exposed the fish for almost a year starting prior to spawning and underlines the importance of chronic exposure studies. One study exposed fathead minnow to NP of polystyrene with 100 mg/l for just 1 h, which stressed the innate immune system (Greven et al. 2016). Concentrations of NP in water or prey are not known, so it is not possible to evaluate if the study design was representative of environmental concentrations.

3.2.3e Summary: microplastics in Arctic fish

In summary, there are indications of a possible translocation of small particles from the GIT to fillets, although studies from the Arctic are sparse. This would be of concern in the context of human exposure to MP. Studies with fish and chronic exposures to MP showed indications of effects on sexual maturation in Atlantic cod, as well as on growth, lipid storage, and endocrine disruption impacting egg production and viability, and malformation in fathead minnows. However, the number of studies is low and partially carried out for short periods with particle sizes that might not be representative for environmental exposure. However, given the uncertainties of MNP quantification, it is difficult to decide on typical environmental concentrations.

Table 3.5 Micro- and nanoplastic effects on Arctic fish.

Species	Particle size in µm, polymer type, dose	Duration	Exposure route and effects	Reference
Fathead minnow-	41 nm (PS) 159 nm (PC). 100 mg/l	1 h	Increase in stressors to the innate immune system.	Greven et al. 2016
	150–500 µm PE 100 MP/l and 2000 MP/l	6 months, from egg stage to reproduction	Impact growth, lipid storage, external coloration, suggesting a food dilution effect. Environmental MP: endocrine disrupting impacts: later egg production, eggs less viable, malformation.	Bucci et al. 2024
Rainbow trout-	212–250 µm, 40% w/v, PE	9 weeks	No difference for depuration rate for organic chemicals, condition factors, or growth rate.	Rummel et al. 2016
White sturgeon~	12–704 µm, 2.8–4.2 mg/l; PET, PVC, PS, PE	1 month	Exposed fish fed more and had higher condition factor than control fish. No difference between polymers.	Rochman et al. 2017
Atlantic cod~	300–600 µm, PE, weathered, 1% of feed	June–May, starting nine months prior to spawning	No effect on digestion or fish biometrics. Effects on steroidogenic enzyme 20β-hydroxysteroid dehydrogenase (20β-hsd) and vitellogenin 1 (vtg1): indicators of disrupting sexual maturation.	Fernández-Míguez et al. 2023

MP = microplastic, PC = polycarbonate, PVC = polyvinyl chloride, PS = polystyrene, PE = polyethylene, PET = polyethylene terephthalate; ~ = marine, - = freshwater. Blue, no effects found for the measured endpoints, red, effects shown.

3.2.4 Arctic invertebrates

3.2.4a All studies found microplastics – limited comparability

The occurrence of MP in Arctic invertebrates has been previously reviewed (Grøsvik et al. 2023a). For occurrence, twelve additional studies have been included in this report; for effect studies, seven additional studies have been included. In the Arctic, MP contamination was assessed in bivalves (one freshwater and two marine species), amphipods, copepods, crabs, starfish, and whelks (Table 3.6; more details including methods in Annex, Table A5). All studies found MP in at least some of the studied organisms. From the Pechora Sea, 22% to 35% of snow crabs (*Chionoecetes opilio*) contained MP in their stomachs (Gebruk et al. 2021). In a comprehensive study on benthic organisms from the Bering/Chukchi Sea, including starfishes, whelks, and bivalves, the mean abundances of MP

Table 3.6 Microplastic occurrence in Arctic invertebrates.

Species	N	Lowest size limit MP	Amount, FO %, polymer type	Matrix	Reference
Freshwater pearl mussel (*Margaritifera margaritifera L.*)	110	100 µm	0–11 fibers/g tissue. FO 92%	Soft tissue	Doucet et al. 2021
Copepods (*Calanus hyperboreus*, *C. glacialis*/*C. finmarchicus*) Amphipods (*Themisto libellula*, *T. abyssorum*)	177 1229 5 5	6.25 µm	Copepods: 0.01–0.21 MP/ind. Amphipods: 1–1.8 MP/ind.	Whole body	Botterell et al. 2022
Greenland smoothcockle (*Serripes groenlandicus*) Snail (*Buccinum* spp.)	13 12	10 µm	FO 69% 1.2 ± 1.1 MP/ind. FO 75% FO 2.0 ± 1.7 MP/ind.	Soft tissue	von Friesen 2018
Shells (*Mytilus* spp., *Macoma balthica*, *Abra nitida*, *Thyasira* spp., and *Hiatella arctica*)	332	10 or 50 µm	< LOD–3.14 MP/ind.	Soft tissue	Bråte et al. 2019
Blue mussel (*Mytilus edulis, M. trossulus*, *M. galloprovincialis*, and hybrids)	332	70 µm	Mean 1.5 MP/ind.; 0.97 MP/g, max 7.9 MP/g. FO 0–100% (site spec.)	Soft tissue	Bråte et al. 2018a
Mytilus spp.	332	70 µm	1.5 MP/ind. 0.97 MP/g	Soft tissue	Bråte et al. 2018b
Blue mussel Polychetes	N = 3 samples per station, 3 stations N = 2 sample, 1 station	50 µm – 300 µm	Py-GC/MS 20 and 25 mg/g d.w. tire wear particles. MP mussels: 1.69, 20, and 20 MP/g d.w., respectively. Lab blank around 5 MP per sample. MP polychetes: 15 and 2 MP per sample, LOD 12.5 MP	Soft tissue	Alling et al. 2023
Wrinkled rock-borer or Arctic saxicace (Bivalve, *Hiatella arctica*)	12	10 µm	Up to 184 MP per bivalve	Soft tissues	Teichert et al. 2021
Amphipod (*Gammarus setosus*)	20	3 µm	FO 72.5%	Digestive tract	Iannilli et al. 2019
Snow crab, great spider crab (*Hyas araneus*), (*Pagurus pubescens*)	23 3 43	ND	FO 27 (22–35)%	Stomach content	Gebruk et al. 2021
Starfishes: (*Asterias rubens, Ctenodiscus crispatus, Leptasterias polaris*) Brittle star (*Ophiura sarsii*) Shrimp (*Pandalus borealis*) Snow crab Whelks (*Retifusus daphnelloides*, *Latisipho hypolispus*) Moon snail *(Euspira nana*) Bivalves (*Astarte crenata*, *Macoma tokyoensis*)	19, 64,15 90 21 59 26 19 28 28, 29	100 µm	0.04–1.67 MP/ ind., 0.02–0.46 MP/g	Whole body	Fang et al. 2018
Sea anemone (Actiniidae) Deposit-feeding starfish (*Ctenodiscus crispatus*) Snow crab	100 100 70	30 µm	0.2–1.7 items/ind. 0.1–1.4 items/ind. 0.0–0.6 items/ind.	Whole body	Fang et al. 2021

MP = microplastic, ind. = individual, FO = frequency of occurrence, LOD = level of development, d.w. = dry weight.

ranged from 0.04 to 1.67 items/individual (Fang et al. 2018). Frequency of occurence of up to 100% was found for soft tissues of mussels, depending on the sampling site (Bråte et al. 2018). However, because both the reporting and the size ranges of the analyses varied, comparability is limited and affects the overall summary of the results. None of the studies investigated NP.

3.2.4b High numbers in some bivalves

Some studies counted high MP numbers: in drilling bivalves (*H. artica*) collected along the northern coast of Svalbard; 1–184 MP particles were found, mainly polyethylene, polyethylene terephthalate, and polypropylene (Teichert et al. 2021), with an average MP number of 8.1 at 27 m depth and 112.8 at 40 m depth. The amphipod *G. setosus* from Svalbard had an average of 72.5 MP per specimen (range 65–90; Iannilli et al. 2019).

Harris et al. (2023) recently estimated high levels of MP in the ocean water column as a transitory sink for MP to the seabed. This could impact filtrating pelagic organisms, e.g., appendicularians, which filter large water volumes, discard their netlike houses regularly as a means of filtering food and contamination, and inhabit all oceans. They are shown to effectively capture and ingest small MP and submicron particles. Fecal pellets and their discarded houses can be important transport of carbon and MP to benthic ecosystems (Katija et al. 2017, Choy et al. 2019). More knowledge is needed to understand how they may be affected by MP exposure or if they could be suitable as indicators. Benthic filter feeders have been previously suggested as suitable indicators because they take up MP particles from the seafloor (Grøsvik et al. 2023b).

3.2.4c First recovery studies

For invertebrate studies, seven studies had method validation steps using recovery tests of known polymers. Although this is an important QA/QC improvement, these tests would further benefit from including challenging polymer types such as polycarbonate, polyethylene terephthalate, and polyamide, which are more difficult to determine (Hove et al. 2023). In addition, recovery experiments could be further improved by studying more comprehensive size ranges, including small MP particle sizes (see Annex, Table A5).

3.2.4d Effects of microplastics on invertebrates – changes in tissue, fecundity, and feeding rates shown with high concentrations or long exposure – importance of end points analyzed

We found 13 articles that addressed effects on species other than blue mussels (*Mytilus* sp.), however, the applied methods and results varied widely rendering their comparisons difficult (Table 3.7; more details including methods and lowest detectable size limits in Annex, Table A6).

For blue mussels, the available global database is too large for the scope of this report. Because most of the data on blue mussels have been reviewed before, readers are, for example, referred to the systematic list of lowest observable effect concentrations (LOEC) published by Gomes et al. (2020). In summary (also see Table 3.7 and Annex, Table A6) the studies on blue mussels found that MP concentrations starting at 8 ng/l caused antioxidant enzymatic activity. Concentrations of this order of magnitude are certainly environmentally relevant (Pakhomova et al. 2022, Hamilton et al. 2024, Wu et al. 2024). Larval malformations were found at concentrations of 0.42 μg/l. Concentrations of 25–50 μg/l led to neurotoxicity, altered gene expression and growth, mortality, phagocytic activity, histopathological alterations, production of reactive oxygen species (ROS) and lipid peroxidation, byssus production, and immune deficiency (Gomes et al. 2020). These concentrations are higher than what has been reported in water but might occur regionally. One study found changes in feeding behavior at concentrations of 3000 MP/l for PET fibers. These concentrations were roughly 1.5–3 orders of magnitude greater than the fiber concentrations (2–203 fibers/l) in the wastewater effluent of Longyearbyen (Herzke et al. 2021). The study finding lysosomal membrane stability to be affected (Avio et al. 2015), reviewed by Gomes et al. (2020), used 1.5 g for exposure, but it investigated nanoparticles that have unknown concentrations in the environment.

Seven studies on the effects of MP exposure were identified for *Calanus* spp. One study did not find any effects on the survival of adult females, while the other studies reported a change in feeding behavior and prey selectivity, reduced fecal pellet production, earlier molting, reduced fecundity, and stress-induced spawning, although sizes, shapes, polymer type, doses, and duration of the exposure studies varied (Table 3.7). Effects were often observed at high concentrations (> 50,000 MP/l), which do not seem representative of Arctic waters. Reduction in feeding rates in *Calanus* were exacerbated when co-exposed to oil droplets (Almeda et al. 2021).

One study found a temperature-dependent effect of MP on *Daphnia magna* and reported that fragments were over 70 times more toxic than beads (Na et al. 2023). The MP concentrations of this study were 4 and 27 mg/l, which are high concentrations compared to concentrations found in the environment.

Two studies investigated dose-response effects in amphipods/isopods. Respiration and metabolic rates were negatively affected by fibers but not by fragments in the amphipod *Orchomene* sp., while chronic ingestion of MP had no distinct adverse effects on survival, intermolt duration, and growth on the isopod *Idotea emarginata* (Granberg et al. 2020).

Two studies with Norway lobster (*Nephrops norvegicus*) exposed to PE or PS microspheres for three weeks did not find ingestion of MP or effects on the nutritional state (Devriese et al. 2017), wheras a long-term, eight-month study with five larger fibers/feeding did report effects on the reduction of feeding rate, body mass, and lipid storage (Welden and Cowie 2016). In a short-term study with acorn barnacles (*Balanus glandula*), no effect was found on feeding rates (Davies 2021). Again, the data point toward the importance of long-term studies and a careful selection of relevant endpoints to be studied, together with the importance of the study design regarding type and size of MP.

Table 3.7 Effects of micro- and nanoplastics in Arctic invertebrate species.

Species	Particle size, polymer type, dose	LOEC (lowest observed effect concentration)	Exposure route and effects	Reference
Bivalves				
Blue mussels (*Mytilus edulis*; *M. galloprovincialis*)			Endpoints for which significant effects were recorded:	Gomes et al. 2020 and references therein
	PE fragments > 500 µm	25 µg/l	Byssus production and immune deficiency	
	PS spheres 1–9 µm	32 µg/l	Mortality, concentration, and phagocytic activity of circulation hemocytes, histopathological alterations, ROS production, and lipid peroxidation	
	mixture PE and PP fragments, 200–500 µm	8 ng/l, 10 µg/l, respectively	Antioxidant enzymatic activity and genotoxicity	
	PET fibers 200 to > 500 µm	3000 MP/l	Feeding behavior	
	PE and PLA fragments 1 to 50 µm	30 µg/l	Alterations in gene and protein expression, growth	
	PS spheres, 1–9 µm	0.42 µg/l	Larval malformations	
	PE and PS fragments size range from < 0.05 to 99 µm	1.5 g/l	Lysosomal membrane stability	
	PS spheres 0.1–0.99 µm	50 µg/l	Neurotoxicity	
Species	Particle size, polymer type, dose	Duration	Exposure route and effects	Reference
Copepods				
Centropages typicus	1.7–31 µm PS 4,000,000 MP/l	24 h	Gut blockage and increased gut retention times leading to reduced feeding function	Cole et al. 2013
Calanus helgolandicus	20 µm PS 75,000 MP/l	24 h and 9 d	Reduced fecundity	Cole et al. 2015
C. finmarchicus	15–30 µm PS, spheres and fragments 50,000 and 500,000 MP/l	11 d	No effects observed on survival of adult females	Vroom et al. 2017
C. finmarchicus	PA fibers (10 × 30 µm) granules (10–30 µm) 50,000 MP/l	6 d	Nylon fibers can affect prey selectivity in *C. finmarchicus*, both nylon fibers and granules caused earlier molting	Cole et al. 2019
C. helgolandicus	20.7 µm PE 100,000 MP/l	24 h	Changes in feeding behavior and selectivity for feeding on algae	Coppock et al. 2019
Calanus hyperboreus	20.7 µm PE 20,000 MP/l	5 d	Copepods stopped feeding. Reduced fecal pellet production rates after co-exposure to oil and MP	Almeda et al. 2021
C. finmarchicus, *C. glorious*, and *C. hyperboreus*	20 µm PE 20,000 MP/l	6 d	MP did not negatively affect fecal pellet production rates. MP exposure can cause stress-induced spawning	Rodríguez-Torres et al. 2020
Crustaceans				
Isopod Idotea emarginata	1–1000 µm Microbeads, fragments, and fibers PE 0.3 mg MP per g food	6 weeks	Chronic ingestion of MP had no distinct adverse effects on survival, inter-molting duration, and growth of the isopods	Hämer et al. 2014
Amphipod *Orchomene* spp.	Fibers 5 µm × 50 µm, fragments 10–100 µm; PET 0; 5000; 50,000; 500,000, and 5,000,000 MP/kg d.w. sediment.	24 h	Respiration and metabolic rates were negatively affected by PET fibers, although not egested, but not fragments. Significantly lower movement response (500,000 MP/kg; all p's < 0.05)	Granberg et al. 2020
Daphnia magna	MP fragments (42 µm) induced over 70 times higher lethal toxicity than MP beads (44.50 µm) median effective concentrations (EC50) of 4 and 276 mg/l	2 d	Elevated temperature significantly increased ($p < 0.05$) the lethal and sublethal (lipid peroxidation and total antioxidant capacity) toxicity in those exposed to MP fragments compared to those at the reference temperature	Na et al. 2023

Species	Particle size, polymer type, dose	LOEC (lowest observed effect concentration)	Exposure route and effects	Reference
Norway lobster *(Nephrops norvegicus)*	Length: 3000–5000 µm × 200 µm PP 5 fibers/feeding total: 360 MP per ind.	8 months	Reduced feeding rate, body mass, and lipid storage	Welden et al. 2016
N. norvegicus	6 µm or 500–600 µm PE, PS microspheres 155 mg by feed ca. 30 g seafood and 25 g water/gelatin, approx. 3 mg/g.	3 weeks	The ingestion of MP did not affect nutritional state	Devriese et al. 2017
Acorn barnacle (*Balanus glandula*)	Unstandardized fiber lengths PES 70,000 MP/l	24 h	There was no detectable short-term effect of microfiber ingestion on feeding rate	Davies et al. 2021

PES = polyester, PS = polystyrene, PE = polyethylene, PET = polyethylene terephthalate; ind. = individual, ROS = reactive oxygen species. Blue = no effects shown; red = effects shown.

3.2.4e Summary: microplastics in Arctic invertebrates

In summary, MP occur widely in invertebrates, although the question of representative environmental concentrations remains. Studies on blue mussels elsewhere than in the Arctic have been consistent stating that effects do occur at concentrations considered environmentally relevant. These effects include immune deficiency, increased mortality, oxidative stress, and neurotoxicity. Based on these results, we suggest that Arctic blue mussels may be at risk for biological effects resulting from MNP exposure. However, these suggestions come with a high degree of uncertainty due to low study numbers and limited comparability across studies. For all other species, some effect studies have reported results. However, as short exposure periods might not represent environmentally realistic conditions, long-term studies seem necessary to pick up the effects of realistic concentrations. Effects observed outside the Arctic (see introduction) in combination with the wide MP occurrence in Arctic invertebrates indicate some effects might occur that remain to be documented.

3.2.5 Effects related to the size of the exposure material

Figure 3.5 summarizes the results on reported effects according to the MP size used in the different studies. No trends according to the size of the exposure material could be discerned from the sparse material available on Arctic species. Apart from blue mussels, only one size category was studied for most species,

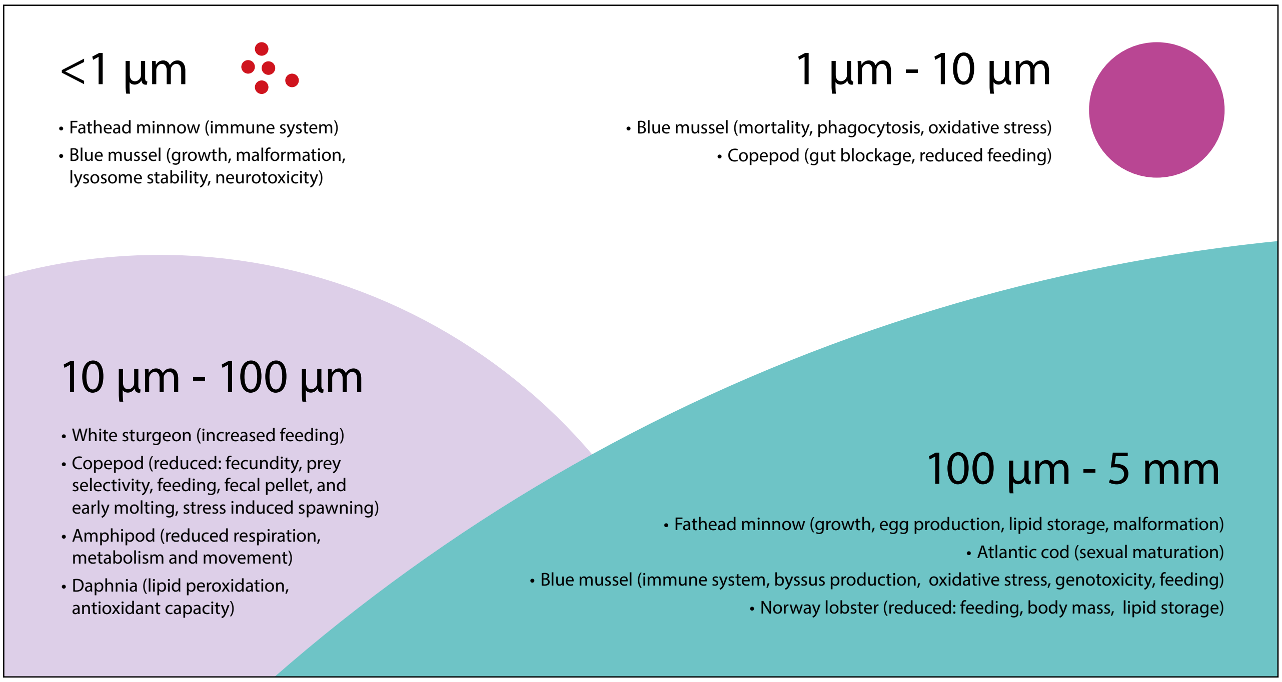


Figure 3.5 Reported effects of MNPs from exposure studies of animals with Arctic occurrence. Details and references in Tables 3.4, 3.6, and in Annex, Tables A4, and A6 sorted according to size categories of the exposure particles. When exposed with particles of mixed sizes with a reported range, the lower end was used for categorization. Illustrating spheres/sections of spheres reflect the size categories.

Table 3.8 Summary of the occurrence and effects of microplastics on Arctic biota.

	Data on MP occurrence	Effect studies	Lowest size limit of MP; number of studies in the following groups		
				Occurrence	Effects
Mammals	Sparse, mostly GIT, one study including blubber, none for terrestrial mammals	No information	1–49 µm	2	0
			50–299 µm	4	0
			> 300 µm (incl. "visible" and "handled with forceps")	4	0
Birds	Common: GIT	No information	1–49 µm	0	0
			50–300 µm	0	0
			> 300 µm (including "visible")	28	0
Fish	Sparse, mostly on GIT, one study on muscle	Sparse	< 1 µm	0	1
			1–49 µm	7	1
			50–299 µm	4	2
			> 300 µm (including not defined)	5	1
Invertebrates	Sparse and varied	Sparse	< 1 µm	0	0
			1–49 µm	6	10
			50–299 µm	6	3
			> 300 µm (including not defined)	1	1

analyzed endpoints, and/or plastic type. For blue mussels, Gomes et al. (2020) reported that "the highest number of studies" (12 in total) used PS particles between 1 and 9 µm. Studies with PE particles used size ranges of 20–49 µm and 50–99 µm with five studies each, along with PS particles with sizes 0.1–0.99 µm and 20–49 µm. All the other particle size distributions had less than five studies each. Most of the reviewed studies only reported effects for particles above 1 µm, with only a small number showing impacts with particles within nano-range, more specifically PS and PE (Table 3.8). This is the reflection of the size-dependent threshold commonly associated with the particle-selection feeding behavior characteristic of most of the species included in this taxonomical group.

3.3 Conclusions

3.3.1 Key findings

3.3.1a Arctic mammals

- Data on MP in Arctic mammals are sparse, with a total absence of information on terrestrial mammals. Data on terrestrial animals would also be relevant from a human exposure point of view, particularly for caribou/reindeer.
- Although most data stem from stomachs and intestines, one study detected MP in blubber, which is an important subsistence resource for Arctic peoples. More information on the MNP content in blubber and other tissues used for human consumption is needed.
- Preliminary data from scat samples indicate that these could serve as non-invasive indicators for plastic exposure, in contrast to stomach analysis. The information conveyed from scat and stomach analyses should be carefully evaluated prior to any changes in sampling strategies.
- Preliminary indications show higher MP levels in whales than in seals. This will need confirmation.
- A power analysis showed that sample sizes of N = 36 and 10 (for walrus and seals, respectively) were too low to determine a significant relationship between the presence of MP > 0.5 or 1 mm in stomachs, as compared to procedural blanks.
- Data on lower MP sizes are urgently needed, with adequate sample sizes to obtain sufficient statistical power.
- No data were available on effects of MP in Arctic mammals except mice. Mice exposed to MP showed pathological changes in the gut and liver, which should also be investigated for other Arctic mammals.

3.3.1b Arctic birds

- Seabirds constitute the most studied Arctic vertebrate group for plastic occurrence, with the Northern Fulmar being the most studied, highly contaminated species and a defined bioindicator. Short-tailed Shearwater, Fork-tailed Storm Petrel, and Parakeet Auklet are also potentially highly contaminated.
- Differences in plastic content in different species are likely due to feeding strategies. Relevant data to reflect differences in ecology, in particular feeding habits, should be obtained.
- Data on plastic in bird tissues consumed by humans should be acquired and related to human exposure.
- Including smaller MP would be highly relevant and could also provide data on indirect ingestion through contaminated prey.
- Most studies relied on visual identification only. For at least a subset of particles, the polymer type should be identified.
- Additional tissues and organs would be relevant to study for a better understanding of MP passage through the animal and for more information to assess human exposure via consumption of Arctic birds.
- Regarding effects, no study on Arctic seabirds could be found.

It is important to note that the current monitoring of Northern Fulmars is not related to effects. A correlation between the number of ingested plastics and the extent of fibrosis in the stomach lining, the spleen, the kidney, and the proventriculus accompanied by inflammation was reported for birds from outside the Arctic. These sparse data encourage further investigation of effects on Arctic birds.

3.3.1c Arctic fish

- Most studies only addressed MP occurrence in the GIT, although indications exist of MP transferring to muscle tissue/fillets. Because this would be of concern in the context of human exposure to MP, there is a need to investigate translocation and bioaccumulation of MP in tissues.
- Several studies relied on visual identification only. For at least a subset of particles, the polymer type should be identified.
- No effects were found in the studies with shorter exposure times (< 9 weeks), except for increased feeding. In contrast, studies with long exposure times (> 6 months) observed various effects, including endocrine disruption and effects on sexual maturation, growth, lipid storage, and malformation. The data are sparse, and more studies are required to draw conclusions.
- Data on smaller size classes are of high interest because FO% seems to be influenced by the lowest MP size determined in the study. For comparability, data should be reported in more detail and with relevant metadata.

3.3.1d Arctic invertebrates

- Blue mussels may be at risk for biological effects by exposure to local environmental MNP loads.
- All studies found MP in Arctic invertebrates. However, details were not always provided (e.g., on MP detection frequencies, total and relative polymers abundances, and particle size distribution).
- Effect studies on invertebrates from the Arctic and elsewhere are limited, with both negative and positive findings. Exposure duration seems to be a determining factor. Few studies reported on long-term exposures (> 3 weeks). More knowledge from chronic exposures is needed.
- Microplastics caused structural changes in gills and the digestive glands of mussels as well as necrosis in other tissues. High concentrations of MP were found to reduce feeding rates. This effect was exacerbated in a case of co-exposure to other contaminants.
- Copepods were shown to stop feeding and displayed stress-induced spawning. This needs better investigation because copepods are an important basis of the Arctic food chain.
- Amphipods had lower movement rates when exposed to MP, and high contamination levels were shown in one study. Metabolic rates and respiration of amphipods were negatively affected by PET fibers.

3.3.1e Methodological considerations

- Comparability between studies is extremely limited because different organs, tissues, and sample types (stomach, intestine, GIT, scat, fecal precursors, muscle, whole animal) were analyzed (see Tables and Figure 3.3).
- Organs and tissues other than GIT should also be studied, for the following reasons: (1) our scientific knowledge of MP translocation is limited; (2) other matrices are important from a human exposure point of view; and (3) non-invasive techniques should be further developed.
- Methods with different lower limits in terms of particle size are used, and data are not reported in harmonized groups, for example, size range of polymer types. The publication of all metadata and detailed data on particle resolution should be promoted because accessibility would allow data users to make purpose-adjusted categories. Efforts toward standardized reporting should be prioritized.
- Harmonization is necessary for trend analysis and other combinations of data, and further efforts in this field should be prioritized, for example, through intercalibrations.
- The standardized bioindicator fulmar, focusing on MP > 1 mm might have led to an omittance of the analysis of smaller MP in this matrix. Research and monitoring efforts should not be limited to standardized indicators but should report on the standardized indicator at a minimum.
- The MP particle size range of the study, in particular the lowest detectable level, should always be reported because it is crucial for the correct interpretation of the data. Data on smaller size classes are of high interest because FO% seems to be influenced by the lowest sized MP determined in the study.
- Positive and negative controls should be analyzed in sufficient numbers and be representative of the analyte type and size. This means that recovery tests should be conducted and appropriate blanks should be included (field blanks and laboratory blanks).
- None of the occurrence studies measured NP. Effect studies of NP were only carried out for two species of Arctic relevance (fish and blue mussel). The lack of data on NP occurrence is common for all regions and is caused by a lack of suitable methodology. Therefore, the occurrence in Arctic animals is unknown. Effects of NP on aquatic animals are better studied in non-Arctic animals.
- Temperature changes more rapidly in the Arctic than in other areas of the world. Indications exist of increasing toxic MNP effects with increasing temperature (Lins et al. 2022, Sulukan et al. 2022, D'Avignon et al. 2023, Hasan et al. 2023, Na et al. 2023), but for the Arctic, this has only been shown for *Daphnia pulex*.

3.3.1f Considerations on risk assessment

So far, the existing data are too sparse to allow even first steps toward a risk assessment. We consider it premature to build complex Bayesian models for risk assessment for human health in the Arctic such as in Saeed et al. (2022) or to pursue an ecotoxicological risk assessment without a sufficient quality-assured data basis. Models need to contain defined empirically testable, affordable checkpoints as well as quality assured, reproducible data collected using best practice standards (Cowger et al. 2020). An attempt to define and investigate quality criteria necessary for risk assessment has been made in the FARE project (https://fare.grida.no). The toolbox from FARE was used on a selection of the articles (max 10 per animal group) presented in this review, and it showed that the diversity of sampling, processing, and analytical steps between studies could compromise the suitability of data for risk assessments. A pre-defined scoring matrix assigns scores to criteria included (or excluded) in scientific articles. For example, scores are available depending on the number of relevant criteria (defined based on size of plastics and purpose of the study: exposure assessment vs. hazard assessment). Scores for exposure assessments are grouped by sampling methods, laboratory contamination, and processing and analysis. Hazard assessment scores are grouped by particle exposure/characterization, test species information, toxicity test design, relevance for hazard assessment, and data reporting/QA. The maximum scores available for each criteria varied and papers could score n. a. in which a criterion was not relevant. The results are presented as averages, thus, the higher the percentage, the higher the score.

Of the exposure papers included in the review, 36 were included in this assessment. The average score for all papers was 59.1%, although individual papers scored between 29.2% and 91.2%. Laboratory contamination was the weakest criterion for all biota groups (except for birds in which this was not assessed). The strongest criterion was processing and analysis (Table 3.9).

Eleven publications on effects were included in this assessment (6 invertebrates, 5 fish). Ten of the papers were *in vivo* assessments, whereas one was performed as *in vitro*. The overall average score was 52.9% (N = 11). The minimum score was 36.5% whereas the maximum score was 70.0%. Invertebrates scored on average 69.6%, whereas the fish publications scored 56.9% (Table 3.10).

3.3.2 Limitations

Any articles not containing microplastic* OR nanoplastic* OR TWP (short for Tyre Wear Particles) in the parts defined as "topic" in Web of Science (or otherwise known to the authors) will not have been included in our analysis. Nevertheless, the authors agree that the status quo of the research field of microplastic occurrence in or effects on Arctic biota will not be significantly misrepresented.

3.3.3 Knowledge gaps and recommendations

Tables 3.11-3.13 summarize knowledge gaps and recommendations for future work derived from our gap analysis, which could pave the way toward a risk assessment of Arctic MNP pollution. To achieve a risk analysis, it is pivotal to match the types of plastic in terms of shape, size, and composition analyzed in occurrence studies with those administered for effect studies.

To enable such comparisons, occurrence data need to be published in greater detail, preferably on a per particle basis with metadata. Effect studies should employ realistic concentrations and long-term exposures. Currently there is no common database for the collection of data on MP occurrence in biota. Work is underway to integrate marine biota (blue mussels) within the ICES (International Council for the Exploration of the Sea) DOME database, which is used by OSPAR, HELCOM

Table 3.9 Summary results of the FARE Toolbox used to assess data on microplastics in environmental samples from the Arctic. Birds were assessed using the mesoplastic assessment criteria considering that most studies followed well-established guidelines (i.e., OSPAR) whereby the target plastics are > 1 mm. For full scoring criteria please refer to the FARE Toolbox (T. Maes, A. L. Lusher, and L. Sørensen, *unpublished report*).

		Average scores					
Group	N	Sampling methods (max score: 10)	Lab contamination (max score: 8)	Processing and analysis (max score: 16)	Total available score	Achieved score	%
Mammals	9	5.3	5.0	8.8	34	19.1	56.2
Birds	10	4.8	n.a.	9.8	24	14.6	60.8
Fish	10	6.3	3.8	10.8	34	20.9	61.5
Invertebrates	7	5.6	4.3	9.6	34	19.4	57.1
Overall average	**36**	**5.5**	**4.4**	**9.8**			**59.1**

Table 3.10 Summary results of the FARE Toolbox used to assess data on microplastics exposure on species relevant for the Arctic. For full scoring criteria please refer to the FARE Toolbox (T. Maes, A. L. Lusher, and L. Sørensen, *unpublished report*).

		Average scores					
Group	n	Particle exposure/ characterization	Test species	Toxicity test design	Relevance for hazard assessment	Data reporting	%
Fish	**5**	7.8	10.0	15.4	5.0	2.8	56.9
Invertebrates	**6**	6.7	9.0	13.2	5.0	2.7	49.6
Overall average	**11**	**7.2**	**9.5**	**14.2**	**5.0**	**2.7**	**52.9**

Table 3.11 Knowledge gaps and recommendations for future research: occurrence data.

Knowledge gaps	Recommendations for future research
Occurrence of MNP in biota with chemical identity, size, and shape information: 1. Which biota/matrices are most polluted or most sensitive and are suited as additional indicators for ecosystem pollution? 2. Data on terrestrial mammals. 3. Data comparing species. 4. Data on animals that are food items. 5. Data with physiological relevance, i.e., on MNP sizes that enter the respective biota/matrices and have physiological effects. Data on NP are currently lacking. 6. Do MNP bioaccumulate (their uptake rate exceeds the excretion) and biomagnify through the Arctic food chain in the GIT and in tissues? 7. Data on translocation for different animals. 8. Which number of analyses is representative for the population? Biota/matrix-specific.	1. Improved quality of MP analyses. 2. Consolidation and harmonization of analytical methods for the determination of MP, including establishment of proficiency testing. 3. Several studies rely exclusively on visual inspection in MP analysis. Analysis with advanced techniques (e.g. FTIR, Raman) should be performed on at least a subset of samples to ensure distinction between plastics and natural materials and to provide polymer-specific data. 4. Identification of biota at risk of MNP exposure. Example: data on potentially exposed birds other than Northern Fulmar: Great Shearwater, Little Auk/Dovekie. 5. Identification of biota with otherwise high importance, e.g., because it is human food. Example: data on caribou/reindeer stomach, intestine and/or feces, and meat. 6. Analysis of more than one tissue/organ. Examples: (1) intestinal tract, if the species tends to accumulate MNP in the intestinal tract; (2) scat/feces; and (3) tissues/blood. 7. Development of non-invasive sampling methods. 8. Development of suitable methods for MNP analysis, with focus on small particles, including polymer-type identification, and their application to relevant samples. Examples: Py-GC/MS, μ-FTIR, Raman, L-OPTIR, applied to caribou meat or whale/seal blubber. 9. Determination if bioaccumulation or biomagnification takes place, and in which species/tissues. Examples: (1) compare animals of the same species, but different age groups; (2) data on indirect uptake though prey; and (3) food web analysis: comparison of biota on different trophic levels. 10. Consider additional indicator species and matrices to cover environmental monitoring and assessments of human exposure. 11. Power analysis to determine sufficient sample number for representing the population, method /MP size group specific. 12. Start trend (space, time) monitoring.

Table 3.12 Knowledge gaps and recommendations for future research: effect studies.

Knowledge gaps	Recommendations for future research
1. Effect of the concentrations and combinations of MNP found in the environment on (1) the physiology of the organisms and (2) the ecosystem. 2. Effects of multiple stressors, i.e., temperature, salinity, and pH change, changed prey and predators, co-exposure to additives and other pollutants. Synergistic, additive, or antagonistic?	1. Laboratory-based, long-term exposure studies on size, type, and concentrations measured in the efforts described above, and projections thereof, to achieve realistic environmental concentrations. 2. Development of relevant and easy-to-use combinations of exposure and effect biomarkers that can be applied in the field in a coordinated way. 3. Combined investigation of MNP occurrence and physiological markers in the wild. 4. Multiple stressors in interdisciplinary studies. Example: (1) data on additives or other contaminants in addition to MNP; (2) effect studies from co-exposure situations; and (3) data on changes related to climate change.

Table 3.13 Overarching knowledge gaps and recommendations for future research.

1. Raise awareness and prevent plastic pollution despite knowledge gaps. Data should be published according to local and global harmonization suggestions. However, so far this has not led to comparability of published studies or of different size groups. Data are still published in different size groups and polymer-type groups, and grouping without publishing the data on a per-particle basis renders the data largely incomparable. 2. To enable comparison, data on individual MP level and metadata should be made available as part of scientific publications or data repositories. 3. Data should be produced according to quality criteria and assessed for transparent and published quality criteria before being fed into models. Models should contain transparent empirical checkpoints.

(The Baltic Environment Protection Commission), and AMAP in collaboration with EMODnet (European Marine Observation and Data Network). However, it may not be possible to integrate all species within the same database because of the large variety of methodological approaches and target size of particles. Therefore, and meanwhile, such detailed data should be provided as part of scientific publications. The QA/QC should be incorporated more consistently, and results should be reported. Contamination control and the analysis of negative controls (blanks) has improved, however, recovery analysis of all important analytes with respect to size, shape, and composition in the analyzed matrix is still needed to estimate MP loss during sample processing and analysis. Despite the continuing lack of quality data, action is necessary to reduce the acceleration of plastic pollution all over the world. Documented effects from entanglement (Chapter 1), ingestion (Chapter 2), and plastic chemicals (Chapter 4) highlight the need to protect Arctic animals from detrimental effects of plastics. However, even if macroplastic emissions are reduced, MNP pollution will continue from the breakdown of existing macroparticles in the environment. Although MNP effects on Arctic biota are still largely unknown, the precautionary principle should be applied to protect Arctic animals by reducing their exposure to MNP. Action to curb plastic pollution should not await these measures.

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3.5 Annex for Physiological Effects from Ingested Micro- and Nanoplastics in Arctic Animals – a Critical Review and Gap Analysis

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3.5.1 Search string

A systematic literature search was carried out with a Web of Science Topic search on April 3 2023, using a comprehensive search string : (microplastic* OR nanoplastic* OR TWP) AND (biota OR animal* OR organism* OR *fish* OR Cod* OR mussel* OR bivalve* OR invertebrate* OR sculpin OR saithe* OR pollock* OR salmon* OR trout* OR capelin OR whiting OR mackerel OR char OR haddock* OR cusk OR tusk OR shark OR bear* OR fulmar OR *bird* OR cucumber* OR flounder OR cisco OR arenicola OR predator* OR plankton OR whale OR copepod* OR amphipod* OR annelid* OR shrimp* OR lobster* OR crab* OR spec* OR *food OR fillet OR liver OR walrus OR seal* OR asterias OR ctenodiscus OR lepasterias OR pandalus OR chionocetus OR ophiura OR retifusus OR latisipho OR euspira OR macoma OR gammarus OR oweniidae OR galathowenia OR owenia OR actiniidae OR ctenodiscus OR chionoecetes OR *salpa OR iasis OR ihlea OR calanus OR mytilus OR hyas OR borealis OR pagurus OR hiatella) AND (accumulat* OR uptake OR effect* OR pattern* OR damage OR disorder OR swimming OR detection OR contamination OR amount OR characteriz* OR risk* OR fillet OR liver OR identif* OR occurrence OR pattern* OR quanti* OR *toxic* OR risk* OR monitor* OR mapping OR analys* OR concentration* OR food OR endocrine OR oxidat* OR carcinogen* OR reproduct* OR behavi* OR impact* OR physiologic*) AND (Arctic OR Barent* OR Kara* OR Greenland* OR Norw* OR Iceland* OR Beaufort* OR Canad* OR Russia* OR "North pole").

3.5.2 Extended data on Arctic mammals

In summary, 6 publications including 5 species, 239 individuals reported on pinnipeds, including ringed seals (N = 145), fur seals (N = 35), bearded seals (N = 7), harbor seal (N = 1), and walrus (N = 51). Whale analysis consisted of 3 publications, 5 species, 39 individuals, and included fin whales (N = 26), beluga whales (N = 8), gray whales (N = 2), humpback whale (N = 1), and minke whale (N = 1).

Most studies digested the analyzed organic matter with KOH or H_2O_2, with one adding a density separation step (polar bear scat). One study on seal scat used a detergent and a density separation step. For most studies, microscope assisted visual pre-sorted particles were identified by FTIR, while one study used Raman microscopy and mass-based analysis: pyrolysis-gas chromatography/mass spectrometry (py-GC/MS), but on different samples. In this study, instrument failure led to a reduction in samples investigated with py-GC/MS, to only two from the Arctic (Alaska). Where fiber contamination could not be monitored or controlled in the field and laboratory, they were excluded from the results with one exception that did not have field blanks.

Table A3.1 Microplastic occurrence in Arctic mammals.

Region	Species	Lower MP size limit	Amount, FO%, polymer type	Matrix	Method	Reference
Pinnipeds						
Alaska, 2 sites	Northern fur seal (*Callorhinus ursinus*) N = 35 (scat samples)	250 µm	0–86 MP per scat; FO 55%; most PE; no difference to control for fibers. 49% fibers. 80% below 1 mm.	Scat	Detergent rinse, sieving, visual sorting (> 500 µm), H_2O_2, 75 °C, density separation (250–500 µm). FTIR on 2 particles. No recovery test.	Donohue et al. 2019*
Nunavut, Canada	Ringed seals (*Phoca hispida*) N = 135, bearded seals (*Erignathus barbatus*) N = 6, harbor seal (*Phoca vitualina*) N = 1	425 µm	None	Stomach content	Stomach contents flushed through sieve stack (450, 825 µm), visual inspection (microscope). Fibers excluded. No recovery test.	Bourdages et al. 2020
Svalbard	Walrus (*Odobenus rosmarus*), colony of 15, 8 samples	500 µm	34 MP/kg; FO 87.5%; 6 polymer types, mostly PES; 70% fibers	Feces	Density separation and KOH (10%), microscopy, µ-FTIR. No recovery test.	Carlsson et al. 2021
Nunavut, Canada	Atlantic walrus (*Odobenus rosmarus rosmarus*) N = 36	80 µm filter, visible particles	no difference to control for fibers	Stomach	KOH (10%) 3:1, 14 d, microscopy, subsample µ-ATR-FTIR. No recovery test.	Jardine et al. 2023a *
Nunavut, Canada	Ringed seals N = 10	80 µm, visible particles	None; no difference to control for fibers	Stomach	KOH (10%) 3:1, 14 d, microscopy, subsample µ-ATR-FTIR. No recovery test.	Jardine et al. 2023b*
Whales (and one seal)						
Eastern Beaufort Sea, Canada	Beluga whale (*Delphinapterus leucas*) N = 7	20 µm	18–147 MP/GIT; mean 97 ± 42; FO 100%; 9 polymer categories, most PES; 49% fibers. Most < 500 µm.	GIT	KOH (10%), visual inspection stereomicroscope, FTIR. No recovery test.	Moore et al. 2020
Western Iceland	Fin whale (*Balaenoptera physalus*) N = 25	1.2 mm	57 MP/kg krill; FO = n.a.; PE, PP, PS, cellulose acetate, acrylic.	Stomach	Krill samples (20 g) from stomach. 15% H_2O_2, 20:1; 5 d, 65°C, filtration, visual inspection (stereomicroscope), µ-FTIR, no field controls. Recovery on PE, PP, PET of 209 µm – 3 mm in fish GIT and mussel 88.75%, lowest recovery for 200–400 µm.	Garcia-Garin et al. 2021
Alaska	py-GC/MS: gray whale (*Eschrichtius robustus*) N = 1; bearded seal N = 1 Raman: beluga whale N = 1, humpback whale (*Megaptera novaeangliae*) N = 1, gray whale 0.19 MP/g, fin whale N = 1, minky whale (*Balaenoptera acutorostrata*) N = 1	For Raman: handled with forceps, py-GC/MS 1 µm glass filter	Py-GC/MS: gray whale: 9.5 µg/g bearded seal: 0.59 µg/g; six polymer types targeted, found PVC. Raman: beluga whale 0.05 MP/g, humpback whale 0.05 MP/g, gray whale 0.19 MP/g, fin whale, 0.1 MP/g, minke whale 0.05 MP/g (numbers estimated from graph)	Blubber	Sample of 9-27 g w. w.; KOH (20%) 3:1, 50C 2 d, 30 mL 100% ethanol 4h dissolve soap, filtered, 30% H_2O_2, 1 h; Raman or microwave assisted digestion in dichloromethane and py-GC/MS. No recovery test.	Merrill et al. 2023*
Bears						
Nunavut and Northwest Territories, Canada	Polar bear (*Ursus maritimus*) N = 15 (scat samples) + 15 (colon feces)	100 µm	0-1.4 MP/g d.w., mean < 1 MP/g. PP, PE, PET in 8 bears. Lower detectable size limit around 1 MP/g.	Scat, feces	Feces (~4 g), KOH (20%) 4-11 d, 10% H_2O_2 2-3 d, density separation ($CaCl_2$), Raman microscopy. Performed recovery study on MP 40–450 µm: 73.9%. PET > PP > PS.	Iyare et al. 2024*

GIT = gastrointestinal tract (esophagus, stomach, intestines), MP = microplastic, FO = frequency of occurrence, PC = polycarbonate, PE = polyethylene, PES = polyester, PP = polypropylene, PS = polystyrene, PVC = polyvinyl chloride., d. w. = dry weight, w. w. = wet weight. FO partially calculated by authors of this report. * = not from systematic literature search.

3.5.3 Extended data on Arctic birds

Table A3.2 Microplastic occurrence in Arctic birds.

Region	Species, N	Lower MP size limit	Amount, FO%, polymer type	Matrix	Method	Reference
Alaska, USA	37 species 1969–1977; N = 1968	1.2 mm	FO 22.8%, 0–6.17 MP/ind., 0–0.24 g/ind., 15 of 37 species contained plastic.	Stomach content	Visual	Day 1980* Thesis
	Short-tailed Shearwater (*Puffinus tenuirostris*) N = 200, Fork-tailed Storm Petrel (*Oceanodroma furcata*) N = 8, Parakeet Auklet (*Aethia psittacula*) N = 116		FO 75–100%			
	Northern Fulmar (*Fulmarus glacialis*) N = 38, Northern Phalarope (*Phalaropus lobatus*) N = 3		FO 50–74%			
	Leach's Storm petrel (*Oceanodroma leucorhoa*) N = 4, Sooty Shearwater (*Puffinus griseus*) N = 76, Cassin's Auklet (*Ptychoramphus aleuticus*) N = 10, Horned Puffin (*Fratercula corniculata*) N = 148		FO 25–49%			
	Glaucous Gull (*Larus hyperboreus*) N = 33, Black-legged Kittiwake (*Rissa tridactyla*) N = 188, Red-legged Kittiwake (*R. brevirostris*) N = 46, Thick-billed Murre (*Uria lomvia*) N = 138, Least Auklet (*A. pusilla*) N = 89, Tufted Puffin (*Fratercula cirrhata*) N = 348		FO 1–24%			
	Laysan Albatross (*Phoebastria immutabilis*) N = 1, Pelagic Cormorant (*Phalacrocorax pelagicus*) N = 3, Doubled-crested Cormorant (*P. auritus*) N = 4, Red-faced Cormorant (*P. urile*) N = 2, Pomarine Jaeger (*Stercorarius pomarinus*) N = 1, Parasitic Jaeger (*S. parasiticus*) N = 1, Glaucous-winged Gull (*L. glaucescens*) N = 63, Herring Gull (*L. argentatus*) N = 5, Mew Gull (*L. canus*) N = 10, Bonaparte's Gull (*Chroicocephalus philadelphia*) N = 4, Ivory Gull (*Pagophila eburnea*) N = 1, Sabine's Gull (*Xema sabini*) N = 1, Arctic Tern (*Sterna paradisaea*) N = 21, Aleutian Tern (*Onychoprion aleuticus*) N = 8, Common Murre (*Uria aalge*) N = 191, Pigeon Guillemot (*Cepphus columba*) N = 18, Kittlitz's Murrelet (*Brachyramphus brevirostris*) N = 5, Marbled Murrelet (*B. marmoratus*) N = 61, Ancient Murrelet (*Synthliboramphus antiquus*) N = 16, Crested Auklet (*Aethia cristatella*) N = 85, Whiskered Auklet (*A. pygmaea*) N = 5, Rhinoceros Auklet (*Cerorhinca monocerata*) N = 20		FO 0%			
Russia	Parakeet Auklet (*A. psittacula*) N = unknown	Visible	Almost all birds had plastic. 0–40 MP/ind.	Stomach content	Visual	Mikhtaryantz 1981*
Svalbard, Norway	Northern Fulmar N = 14	Visible	FO 36%	Stomach content	Visual	Mehlum and Giertz 1984*
	Common Eider (*Somateria mollissima*) N = 1, Black Guillemot (*Cepphus grylle*) N = 8, Dovekie/Little Auk (*Alle alle*) N = 29, Thick-billed Murre N = 1, Black-legged Kittiwake N = 27, Glaucous Gull N = 2, Ivory Gull N = 6, Long-tailed Jaeger (*Stercorarius longicaudus*) N = 1		FO 0%			
Bear Island,	Northern Fulmar N = 51	Visible	FO 4.5 %	Stomach content	Visual	van Franeker 1985*
Jan Mayen, Norway			FO 4.7 %			
Canada	Dovekie/Little Auk N = 303	Visible	"present"	Stomach content	Visual	Day et al. 1985*
Alaska	White-winged Scoter (*Melanitta deglandi*) N = 11		FO 0%			
	Barrow's Goldeneye (*Bucephala islandica*) N = 17, Greater Scaup (*Aythya marila*) N = 3, Harlequin Duck (*Histrionicus histrionicus*) N = 6, Long-tailed Duck (*Clangula hyemalis*) N = 27, Surf Scoter (*Melanitta perspicillata*) N = 11		FO 0%			
Eastern Svalbard, Norway	Northern Fulmar N = 8	Visible	FO 50%	Stomach content	Visual	Gjertz et al. 1985*
	Black-legged Kittiwake N = 18		Not mentioned			
Hornsund, Svalbard, Norway	Dovekie/Little Auk N = 11	Visible	FO 45%	Stomach content	Visual	Lydersen et al. 1989*
	Thick-billed Murre N = 21		FO 24%			
	Northern Fulmar N = 20		FO 15%			
	Black-legged Kittiwake N = 20		FO 5%			
	Common Eider N = 20, Atlantic Puffin (*Fratercula arctica*) N = 14, Black Guillemot N = 20, Glaucous Gull N = 18		FO 0%			

Region	Species, N	Lower MP size limit	Amount, FO%, polymer type	Matrix	Method	Reference
Greenland	Thick-billed Murre N = 202	Visible	23 pieces of plastic < 10 mm	Stomach content	Visual	Falk et al. 1993*
Russia	Northern Fulmar N = 5	Visible	FO 20%, 0.02 g/ind.	Stomach content	Visual	Weslawski et al. 1994
	Common Eider N = 5, Black Guillemot N = 5, Arctic Tern N = 5, Glaucous Gull N = 5		FO 0%			
Alaska, USA	24 species 1988–1990; N =1851	500 μm	FO 0–93.8%.	Stomach content	Visual	Robards et al. 1995*
	Short-tailed Shearwater N = 5, Fork-tailed Storm Petrel N = 21, Parakeet Auklet N = 208		FO 75–100%			
	Northern Fulmar N = 19		FO 50–74%			
	Sooty Shearwater N = 76, Leach's Storm Petrel N = 64, Cassin's Auklet = 35, Horned Puffin N = 120		FO 25–49%			
	Glaucous Gull N = 33, Black-legged Kittiwake N = 256, Red-legged Kittiwake N = 15, Thick-billed Murre N = 92, Least Auklet (*Aethia pusilla*) N = 13, Tufted Puffin N = 489		FO 1–24%			
	Pelagic Cormorant N = 10, Red-faced Cormorant N = 16, Glaucous-winged Gull N = 63, Common Murre N = 134, Crested Auklet N = 40, Whiskered Auklet N = 22, Ancient Murrelet (*Synthliboramphus antiquus*) N = 68, Kittlitz's murrelet (*Brachyramphus brevirostris*) N = 17, Marbled Murrelet N = 96, Pigeon Guillemot N = 43, Rhinoceros Auklet N = 1, Mew Gull N = 4		FO 0%			
			Combined percentages increased, compared to Day et al. 1980: FO increase of > 20% for Northern Fulmar, Leach's Storm Petrel. Decrease for Pelagic Cormorant, and Mew Gull.			
Bear Island, Norway, Barents Sea	Northern Fulmar N = 22	Visible	FO 82% (1–21 items)	Stomach contents	Visual	Camphuysen et al. 1997*
Greenland	Dovekie/Little Auk N = 104	Visible	FO 8.65%	Stomach contents	Visual	Pedersen and Falk 2001*
Southeastern Bering Sea	Short-tailed Shearwaters N = 330	Visible	FO 84%	Stomach content	Visual	Vlietstra and Parga 2002*
Russia	Herring Gull N = 187	Visible	FO 19.8%	Stomach content	Visual	Turovskaya and Nichkevich 2005*
Nunavut, Canada	Northern Fulmar N = 42	Visible	1.3 MP/ind.; 0.02–0.31 g/ind.; FO 36%	Stomach content	Visual	Mallory 2006*
Nunavut, Canada	Northern Fulmar N = 102	Visible	7.4 MP/ind.; 0.31 g/ind. (in affected fulmars); 31% FO, 50% gizzard, 50% proventriculus, season dependent	Stomach content	Visual	Mallory 2008*
Nunavut, Canada	Northern Fulmar N = 25	Visible	5.6 MP/ind.; 0.094 g/ind.; FO 85%	GIT content	Visual	Provencher et al. 2009*
Cape Vera on northern Devon Island, beside the Hell Gate-Cardigan Strait Polynya, Nunavut	Northern Fulmar N = 233 Black Guillemot N = 45	Visible	None	Stomach content	Visual	Byers et al. 2010*
Bjørnøya, Norway	Great Skua (*Stercorarius skua*) N = pellets from 25 nests		Pieces of plastic in 12 pellets from 5 nests	Regurgitated pellets	Visual	Knutsen 2010*
Nunavut, Canada	Thick-billed Murre N = 186	Visible	0.2 MP/ind.; 0.0016 g/ind.; FO 11%	GIT content	Visual	Provencher et al. 2010*

Region	Species, N	Lower MP size limit	Amount, FO%, polymer type	Matrix	Method	Reference
Bering Sea	Short-tailed Shearwater N = 99	Visible	Mean: 15.1 (± 2.9) MP, 0.23 g/ind. PE > PP > PC/ABS > PS Fragment > resin pellet/sheet > fiber > foam	Stomach content	Visual and near-infrared spectrometry	Yamashita et al. 2011*
Russia	Herring Gull N = 306	Visible	MP “present”	Stomach content	Visual	Tolmacheva 2012*
Faroe Islands	Northern Fulmar N = 647	1 mm	15 MP/ind.; 0.21 g/ind.; FO 44%	Stomach content	Visual	van Franeker 2012
Iceland	Northern Fulmar N = 58	1 mm	6.0 MP/ind.; 0.13 g/ind.	Stomach content	Visual	Kühn and van Franeker 2012*
Newfoundland, Canada	Common Murre N = 71 Thick-billed Murre N = 1591	Visible	Specified per period	GIT content	Visual	Bond et al. 2013
Alaska, USA	Short-tailed Shearwater N = 12	Visible	FO 100% 0.04–0.59 g per bird	Stomach content	Visual, near-infrared spectrometry	Tanaka et al. 2013*
Greenland, Nunavut, Canada, Newfoundland, Canada, Faroe Islands, Norwegian Sea	Northern Fulmar N = 35 Atlantic Puffin N = 52 Thick-billed Murre N = 299 Northern Common Eider (*S. mollissima borealis*) N = 584 Long-tailed Duck N = 27, King Eider (*Somateria spectabilis*) N = 54, Common Eider (*S. mollissima sedentaria*) N = 388, Common Murre N = 33, Razorbill (*Alca torda*) N = 10, Dovekie N = 19, Arctic Tern N = 41, Surf Scoter (*Melanitta perspicillata*) N = 38	1 mm	FO 51% FO 13% FO 2% FO < 1% FO 0%	GIT content	Visual	Provencher et al. 2014*
Russia	Short-tailed Shearwater N = 2, Parakeet Auklet N = 3 Northern Fulmar N = 8, Thick-billed Murre N = 3 Common Murre/Common Guillemot N = 1, Tufted Puffin N = 4, Whiskered Auklet N = 2, Red-legged Kittiwake N = 2	Visible	FO 100% FO 25–49% FO 0%	Stomach content	Visual	Artukhin 2014
Faroe Islands, Norway Iceland	Northern Fulmar N = 27 Northern Fulmar N = 40	1 mm	FO 33.3%, 0.23 ± 0.35 g/ind., 13.9 ± 29 MP/ind., 0–152 FO 90 %, 0.12 ± 0.02 g/ind.	Stomach content	Visual	Trevail et al. 2014
Svalbard, Norway	Northern Fulmar N = 40	1 mm	FO 87.5% 0.08 g/ind.	Stomach content	Visual	Trevail et al. 2015
Greenland	Dovekie/Little Auk N = 44	0.1 mm	8.99 & 9.99 MP/ind.; FO 100%; PVC > PE > PES	Gular pouch content	Visual + FTIR (sub-sample)	Amélineau et al. 2016
Faroe Islands	Great Skua N = 1034	Visible	FO 6%	Regurgitated pellets	Visual	Hammer et al. 2016
Norway	Northern Fulmar N = 75	1 mm	FO 81%	Stomach content	Visual	Herzke et al. 2016
Nunavut, Canada	Northern Fulmar N = 19 Black-legged Kittiwake N = 11 Thick-billed Murre N = 20 Black Guillemot N = 3	1 mm	3.4 MP/ind. 0.18 MP/ind. 0 MP/ind. 0 MP/ind.	GIT content	Visual	Poon et al. 2017
Labrador Sea, Canada	Northern Fulmar N = 30	1 mm	1.9 MP/ind.; FO 47%	Last 10 cm intestine, fecal precursor	H_2O_2, visual	Provencher et al. 2018a
Nunavut, Canada	Northern Fulmar N = 70	1 mm	11.6 MP/ind.; FO 79%	Stomach content	Visual	Avery-Gomm et al. 2018*
Newfoundland, Canada	No original data, data from (Avery-Gomm, Provencher et al. 2018b)					Provencher et al. 2018b
Russia	Short-tailed Shearwater N = 1	Visible	FO 100%, 6 MP/ind.	Stomach content	Visual	Golovnyuk et al. 2019*

Region	Species, N	Lower MP size limit	Amount, FO%, polymer type	Matrix	Method	Reference
Russia	Short-tailed Shearwater N = 2	Visible	FO 100%	Stomach content	Visual	Solovyeva et al. 2020*
Nunavut, Canada	Northern Fulmar N = 44	1 mm	1.7 MP/ind. FO 72%	Stomach content	Visual	Baak et al. 2020*
	Black-legged Kittiwake N = 20		0.2 MP/ind. FO 15%			
	Thick-billed Murre/Brünnich's Guillemot N = 80		FO 0%			
	Black Guillemot N = 20		FO 0%			
Faroe Islands	Northern Fulmar N = 86	1 mm	FO 87.3%, 0.18 g/ ind., 11.4 MP/ind.	Stomach content	Visual, FTIR	Ask et al. 2020
Greenland			FO 90.3%, 0.06 g/ind., 6.16 MP/ind.			
Alaska, US	Glaucous-winged Gull N = 2, Crested Auklet N = 7, Northern Fulmar N = 4, Tufted Puffin N = 19	Visible	FO 50–74%	Stomach content	Density separation, visual	Padula et al. 2020
	Black-legged Kittiwake N = 5, Red-faced Cormorant N = 10, Pelagic Cormorant N = 10, Horned Puffin N = 3		FO 25–49%			
	Common Murre N = 6		FO 1–24%			
	Pigeon Guillemot N = 8		FO 0%			
Nunavut, Canada	Northern Fulmar N = 27	800 µm	1.8 MP/ind.;	Intestines	KOH 50 °C, visual, Raman	Bourdages et al. 2021
	Thick-billed Murre N = 30		0 MP/ind.; PES > PE = PA + dyes			
Canadian Arctic	No original data, data from (Bourdages et al. 2021 and Baak et al. 2020)					Hamilton et al. 2021
Canadian Arctic	No original data, data from (Poon et al. 2017 and Avery-Gomm et al. 2018)					Sühring et al. 2022
Svalbard, Norway	Glaucous Gull N = 21	1 mm	FO 14.3% 0.14 MP/ind. 0.008 g/ind.	GIT content	Visual, microscope, FTIR	Benjaminsen et al. 2022*
Greenland	Northern Fulmar N = 145	1 mm	13.5 MP/ind.; 0.14 g/ind.; FO 86%.	Stomach content	Visual	van Franeker et al. 2022*
Svalbard, Norway	Northern Fulmar N = 43	1 mm	10.3 MP/ind.; 0.07 g/ind.; PE > PP > PS.	Stomach content	KOH 20 °C, visual, FTIR	Collard et al. 2022a
Faroe Islands	Northern Fulmar N = 20	1 mm	12.4 MP/ind.; 0.15 g/ind. FO 95%; PE > PP > PS > PES.	Stomach content	KOH 20 °C, visual, FTIR	Collard et al. 2022b *
Svalbard, Norway	Northern Fulmar N = 39	1 mm	36.1 MP/ind.; 0.21 g/ind.; 0 (1 piece in total); PE > PP > PS.	Stomach intestine content	KOH, visual, FTIR	Tulatz et al. 2023*
Nunavut, Canada	Black-legged Kittiwake N = 5 (2008), Prince Leopold Island		FO 20%, 0.004 g/ ind., 2 MP/ind.;	GIT content	Visual, microscope	Baak et al. 2024*
	Black-legged Kittiwake N = 20 (2021), Qikiqtarjuaq		FO 17%, 0.003 g/ ind., 2.7 MP/ind.			
	Black-legged Kittiwake N = 19 (2021), Pond Inlet, Black Guillemot N = 1 (2021), Qikiqtarjuaq, Black Guillemot N = 22 (2022), Pond Inlet		FO 0%			

GIT = gastrointestinal tract (esophagus, stomach, intestines), MP = microplastic, FO = frequency of occurrence, PA = polyamide, PC = polycarbonate, PE = polyethylene, PES = polyester, PP = polypropylene, PS = polystyrene, PVC = polyvinyl chloride, ind. = individual. *not from systematic literature search. None of the studies had carried out a recovery analysis.

3.5.4 Extended data on Arctic fish

The following species have been investigated from Arctic marine waters: Atlantic cod (*Gadus morhua*; 5 studies, N = 1627, Norway, Iceland, Canada), saithe/pollock (*Pollachius virens*; 1 study, N = 46, Iceland), Greenland cod (*Gadus ogac*; 1 study, Greenland, N = 9), polar/Arctic cod (*Boreogadus saida;* 3 studies, N = 157, Norway, Alaska), Atlantic salmon (*Salmo salar*; 1 study, Canada, N = 69), capelin (*Mallotus villosus*; 2 studies, Canada and Alaska N = 378) saffron cod (*Eleginus gracilis*; 1 study, Alaska, N = 26), Arctic cisco (*Coregonus autumnalis*; 1 study, Alaska, N = 26) four-horn sculpin (*Myoxocephalus quadricornis*; 1 study, Alaska, N = 7), bigeye sculpin (*Triglops nybelini*; 1 study, Greenland, N = 71), Greenland shark (*Somniosus microcephalus*; 1 study, Greenland, N = 30) Atlantic mackerel (*Scomber scombrus*; 1 study, Iceland, N = 50) and from Arctic freshwater: Siberian dace (*Leuciscus baicalensis*; 1 study, Russia, N = 40), common dace (*Leuciscus leuciscus* L.; 1 study, Russia, N = 13), white sucker (*Catostomus commersonii*; 1 study, Canada, N = 173), and common carp (*Cyprinus carpio*; 1 study, Canada, N = 58).

Table S3 Microplastic occurrence in Arctic fish.

Region	Species	Lower size limit MP	amount, FO%, polymer type, observations	Matrix	Method	Reference
East, West, Southwest Greenland ~	Greenland shark N = 30	1 mm	0–1 MP/fish, FO 3.33%	Stomach content	Stomach content, visual examination. No chemical identification or recovery study.	Nielsen et al. 2014*
Norway, Six locations ~	Atlantic cod N = 302	3.2 mm	0–8 MP/fish, FO 18.8% PES > PP, PVC > PS, Teflon, Nylon, PE, SAN, PBMS	Stomach content	202 full, 100 empty stomachs. Picking items from stomach, stereomicroscope, FTIR. No recovery study.	Bråte et al. 2016
Newfoundland, Canada ~	Atlantic cod N = 71	1mm	0–2 MP/fish, FO 2.4%	GIT content	Visual sorting No chemical identification or recovery study.	Liboiron et al. 2016
Arctic European basin sea ice around Svalbard ~	Juvenile polar cod N = 72	650* 400 µm 590* 170 µm	2 MP, FO 2.8% Non fibrous: epoxy resin, PMMA. Non fibrous plastic in 2 individuals. 2 of 8 particles found were plastic.	GIT content	Stereo microscope, µ-FTIR Fibers excluded because of contamination. No recovery study.	Kühn et al. 2018
Northeast Greenland ~	Bigeye sculpin N = 71, polar cod N = 85	90 µm	1–5 mm (23) > 500 µm–1 mm (13) > 90 µm–499 µm (11). 1–2 MP/fish Sculpin FO 34% polar cod FO 18% PES > acrylic > PA > PE > EVA 1 fragment, 5 fibers. Demersal > pelagic.	GIT	20 ml 1 M NaOH, 24 h, 0.7 µm glass filter, stereo study microscope, particles > 700 µm FTIR. No recovery study.	Morgana et al. 2018*
Newfoundland, Canada ~	Atlantic cod N = 1010, Atlantic salmon N = 69, capelin N = 350	1 mm	0–2 MP/fish, salmon FO 0%, capelin FO 0%, cod FO 1.68% fibers > fragment > sheet PE > PVC > ABS, PET, PC MP > mesoplastic. Food presence and sex related to FO%, higher in females. Salmon and capelin lower food presence than cod.	GIT	Color sorted. Microscope, according to maximum length. No chemical identification or recovery study.	Liboiron et al. 2019
Iceland ~	Atlantic cod N = 39, Saithe/pollock N = 46	50 µm, FTIR set to 80 µm	0.23 MP per cod 0.28 MP per saithe cod: FO 20.5%; saithe: FO 17.4 % fibers (N) > fragments (N) Difference in fish length between fish containing MP and fish without. Color sorted; fewer colors in cod than in saithe.	GIT	KOH digestion (1:6 sample/KOH), 60°C, 24 h, citric acid neutralization, filtered, visual identification, FTIR. No recovery study.	de Vries et al. 2020
Iceland ~	Greenland cod N = 9	20 µm	12 ± 6 MP/fish, FO 100%	GIT	Enzymatic digestion, visual and FTIR on selected particles. No recovery study.	Granberg et al. 2020*
Newfoundland, Canada ~	Atlantic cod N = 205	1 mm	0–1 MP/fish, FO 1.4%	GIT	Visual sorting, suspected by Raman No recovery study.	Saturno et al. 2020

Region	Species	Lower size limit MP	amount, FO%, polymer type, observations	Matrix	Method	Reference
Yenisei tributary of Nizhnyaya (lower) Tunguska river (Tom River) Siberia, Russia -	Common dace N = 13	No lower detectable size limit provided, filtration: 0.45 µm	204 ± 28.7 MP /fish, most (80%) < 150 µm, most (70% fragments), FO 100% Amount (N): Smaller than 150 > 150–300 > 300–2000 > 3000–4000. Fragments > spheres > films > fibers.	GIT	25 ml HNO_3 (22.5 M), 12 h; room temperature, 2 h boiling water; density separation, 20% NaCl, 12 h; filtered 0.45 µm, rinsed with 2% KOH, stereomicroscope. No chemical identification or recovery study.	Frank et al. 2020
Iceland ~	Atlantic mackerel N = 50	100 µm	1.3 MP/fish, FO 23%	GIT	Alkaline digestion, filtration, visual examination→ Raman. No recovery study.	Malinen 2021*
Beaufort Sea North of Alaska ~	Arctic/polar cod N = 20, saffron cod N = 26, Arctic cisco N = 26, four-horn sculpin N = 7, capelin N = 28	15 µm	0.37 ± 0.16 MP/fish, FO 21% 78% fibers, PES/PET > PA > acrylic > PE > PU > POM > PP 0.15 MP/fish, FO 15% 0.35 MP/fish, FO 34.3% 0.18 MP/fish, FO 19.3% 0.44 MP/fish, FO 42.9% 0.7 MP/fish, FO 7.1% Estimated ingestion 3800–145,000 MP annually.	GIT	10% KOH, 3 x volume of GIT for 2 weeks, occasional swirling, filtered 20 µm, detergent and Tergazyme® enzymes, optical microscope, picked particles ATR-FTIR. No recovery study.	Moore et al. 2022
Freshwater: Thames River Ontario, Canada -	white sucker N = 173, common carp N = 58	34.5 µm	485 MP in total. White sucker: 0–14 MP/fish, FO 44%; common carp: 0–128 MP/fish, FO 31%. MP total/per fish: common carp: 49/0.85, white sucker: 102/0.59 Black unknown particles (suspected tire wear) > acrylic paint > PVC, PP, PE, urban > rural; fragment > fiber. Fish and sediment abundance of MP correlated. Body mass positively correlated to MP/fish.	GIT	20% KOH, 45 C°, 48 h, filtered 10 µm PC filter, stereo microscope, FTIR on 10% subsample, color sorted. No recovery study.	Wardlaw et al. 2022
Yenisei tributary of Nizhnyaya (lower) Tunguska River, Siberia, Russia -	Siberian dace N = 40	150 µm	1.55 ± 1.95 (1–2.1) MP/ fish, 43.1 ± 52.5 MP/kg, FO 60% 300–1000 > 1000–2000 > 150–300 > 2000–5000 PP > PET > PE > PS > PVC No difference between male and female. Smaller and younger fish had higher MP concentrations/kg.	GIT	KOH 55 °C, 2 d, density separation NaCl, upper phase 96% ethanol (10% of sample volume), 0.45 µm cellulose Ester membrane filter, visual identification, µRaman. No recovery study.	Frank et al. 2023
Bering Sea ~	Alaska pollock N = 80	26 µm	2.7 ± 2.8 (0–14) MP/ fish, 85% FO, older fish ingest more, according to age groups: 3 + (58%) to 11 + (100%). Fiber (80.5%) > 100–500 µm (34.4%) > 1000–5000 µm (20.9%) > 26–100 µm (17.2%). Rayon (56.7%), PE (15.3%), PET (13.5%), followed by PES (4.2%), PVC (3.7%), and PVA, PP, PC, PEI, PAM, nylon, cellophane, a copolymer with PVC and polyvinyl alcohol (PVC + PVA), and a copolymer with polyvinylidene chloride and polyacrylonitrile (PVDC + PAN).	GIT	10% KOH, 60°C, FTIR. No MP in matrix free procedural blanks found. No recovery study.	Ding et al. 2023*
Nunavut, Canada ~	Arctic char N = 20 lake trout N = 7	1 µm	73.8 ± 200.79 µg/g (char), 6.01 ± 6.97 µg/g (lake trout); FO 100% PMMA, PP, PVC, PS, PE, PC	Muscle	20% KOH, MAE extraction, py-GC/MS. Recoveries > 90% for all polymer types.	Hamilton 2023*
Nunavut, Canada~	Arctic char N = 20	45 µm	26 ± 19 MP/fish; FO 95% anthropogenic cellulose, PET, Acrylic, PU, PA, PE, PP	GIT	20% KOH, visual characterization; Raman.	Hamilton et al. 2024

GIT = gastrointestinal tract, MP = microplastic, FO = frequency of occurrence, PES = polyester, PP = polypropylene;, PVC = polyvinyl chloride, PS = polystyrene; Teflon, PA = polyamid 6; 6.6 (nylon, PA), PE = polyethylene, SAN = styrene acrylonitrile resin, PBMA = Poly(n-butyl methacrylate, PET = polyethyleneterephthalate, PVA = polyvinyl alcohol, PC = polycarbonate, PEI = polyether-imide, PAM = polyacrylamide, ind. = individual. *not from systematic literature search; ~ marine; - freshwater.

Table A3.4 Micro and nanoplastic effect studies in Arctic fish.

Species	Particle size in µm, Polymer type, dose	Duration	Exposure route and effects	Reference
Fathead minnow -	41 nm (PS, labeled) 159 nm (PC), sonicated. 100 mg/l	1 h	In vitro, adult plasma: increase in degranulation of primary granules and neutrophil extracellular trap release (stressors to the innate immune system).	Greven et al. 2016*
	150–500 µm PE (pre-consumer and environmental) 100 MP/L and 2000 MP/L	6 months, from egg stage to reproduction	Environmentally reasonable dose. Impact adult growth, lipid storage, and external coloration, suggesting a potential food dilution effect. Environmentally sourced MP, but not pre-consumer MP: endocrine disrupting impacts on parental generation and offspring in low concentration: later egg production, eggs less viable, offspring: higher rates of malformation.	Bucci et al. 2024*
Rainbow trout -	212–250 µm, 40% w/v PE	9 weeks	Through feed. No difference for PCB depuration rate, condition factors or growth rate.	Rummel et al. 2016*
White sturgeon ~	12–704 µm, 2.8–4.2 mg/l, PET, PVC, PS, PE	1 month	Exposure through food web: MP fed to Asian clams, clams fed to fish. Exposed fish fed more and had higher condition factor than control fish. No difference between polymers.	Rochman et al. 2017*
Atlantic cod ~	300–600 µm, PE powder, weathered in Oslo fjord for 4 months, 1% of feed	June-May, starting 9 months prior to spawning	Through feed. Effects of weathered polyethylene MP ingestion on sexual maturation, fecundity, and egg quality in maturing broodstock. No effect on digestion or fish biometrics (e.g., body weight, HIS, and GSI), gonadal development, fecundity or egg quality, BPG-axis gene expression, plasma steroids. Effects on gonadal and liver expression of steroidogenic enzyme 20β-hydroxysteroid dehydrogenase (20β-hsd) and vitellogenin1 (vtg1): could be indicators of exposure to contaminants that disrupt sexual maturation.	Fernández-Miguez et al. 2023

MP = microplastic, PC = polycarbonate, PVC = polyvinyl chloride, PS = polystyrene, PE = polyethylene, PET = polyethyleneterephtalate, GSI = gonadosomatic index, HIS = hepatosomatic index, ~marine, - freshwater. *not from systematic literature search.

3.5.5 Extended data on Arctic invertebrates

Table A3.5 Microplastic occurrence in Arctic invertebrates.

Region	Species	Size limit MP	Amount, FO %, polymer type	Matrix	Method	Reference
Saint John River, Canada	Freshwater pearl mussel (*Margaritifera margaritifera L.*) N = 110	100 µm	0–11 fibers/g tissue. FO 92%. Close to wastewater treatment plant Microfibers. Did not investigate polymer types. Blue fibers dominated (44 %).	Soft tissue	10% KOH, 24 h, 60°C. Sieved at 90 µm, repeated digestion. Visual inspection under stereomicroscope. No method validation performed.	Doucet et al. 2021
Fram Strait	Amphipods (*Themisto libellula*) N = 5, *Themisto abyssorum* N = 5, *Apherusa glacialis* N = 1; Copepods (*Calanus hyperboreus* N = 177, *Calanus glacialis*, *C. finmarchicus* N = 1229)	6.25 µm	Amphipods: 1.8, 1, and 1 MP/ind. Copepods: 0.21, 0.01, MP/ind. per species, respectively. PU, PES, acrylic, PVDC, PS, PE. Most particles below 50 µm.	Whole body	Enzymatic digestion, FTIR Perkin Elmer Spotlight 400, software: SIMPLE. For recovery testing of filtration, fluorescent PS (20 µm) and 19*250 µm nylon fibers, stained, were added to samples in triplicates. 97%.	Botterell et al. 2022
Kongs-fjorden and Rijpfjorden, Svalbard	Greenland smoothcockle (*Serripes groenlandicus*) N = 13, *Buccinum* spp. N = 12	20 µm	FO 69% 1.2 ± 1.1 MP per ind. FO 75% 2.0 ± 1.7 MP/ind.	Soft tissue	Enzymatic digestion with Creon 40,000 (lipase, amylase, protease) 37.5°C, 126 rpm, visual analysis, FTIR. Method validation: recovery of PE, 200–600 µm: 90–95%	von Friesen 2018*
Northern Norway	*Mytilus* spp., *Macoma balthica, Abra nitida, Thyasira* spp., and *Hiatella arctica* N = 332	10 or 50 µm	< LOD-3.14 MP/ind.	Soft tissue		Bråte et al. 2019*

Region	Species	Size limit MP	Amount, FO %, polymer type	Matrix	Method	Reference
Norwegian coastal waters, including Bodø and Varanger peninsulas	Blue mussels (*Mytilus edulis*, *M. trossulus*, *M. galloprovincialis*, and hybrids of these) N = 332	70 μm	Mean 1.5 (± 2.3) MP/ ind.; 0.97 (± 2.61) MP/g, max 7.9 MP/g. FO 0–100% per individual depending on site, and in 94% of sites. Fibers accounted for 83 % of particles. Cellulosic most common, black rubber the second. Most < 1 mm.	Soft tissue	10% KOH (10 times v/v), 60 °C, 140 rpm, 24h. Procedural controls. Method validation: recovery test with beads > 2 mm. Recovery 100%, material loss not assessed. Subsample with μ-FTIR.	Bråte et al. 2018a*
Norwegian coastal waters	Mussels (*Mytilus* spp.) N = 332	70 μm	1.5 MP/ind. 0.97 MP/g ww Cellophane (63.9%), EVA, and parking lot tar (18.7%), PET (9.9%)	Whole body (without foot and byssus filaments)	10% KOH (60 °C), visual observation, μ-FTIR. Method validation: recovery of PP, PET, PS, PA-66, and LDPE (nurdles, 2–5 mm), and cloth (acrylic, PES, CA) were tested. Recovery 100%, material loss not assessed.	Bråte et al. 2018b*
Northern Svalbard	Wrinkled rock-borer or Arctic saxicave (bivalve, *Hiatella arctica*) N = 12	10 μm	9% of particles belong to size class 10–300 μm; PE, PS, PET, PVC, PAN, EVAC, PA, PP; 10% of particles belong to 300–1000 μm and 1% to 1000–5000 μm size class. Up to 184 MP per bivalve.	Soft tissues	Enzymatic digestion, FTIR. No method validation or recovery study performed.	Teichert et al. 2021
Ålesund; Skallneset Varangerfjorden; Brashavn (mussel) Bugeøynes (polychaetes) Varangerfjorden	Blue mussel; N = 3 samples per station, 3 stations Polychaetes; N = 2 samples, one station	50 μm–300 μm	Py-GC-MS. 20 and 25 mg/g d.w. tire wear particles. MP mussels: 1.69; 20 and 19.99 MP/g d.w., respectively. Lab blank around 5 MP per sample. MP polychetes: 15 and 2 MP per sample, LOD 12.5 MP? PP > PE > cellulose acetate > PS > PES > PA	Soft tissue	Tire wear particles: py-GC/MS. Recovery for py-GC/MS 83–104%. MP: 10% KOH 1:10. 48 h, 40 °C, and acetic acid, μ-FTIR. Recovery on MP of PS, 500 μm: 90 and 98%, in spiked sample and blank, respectively; PE 125 μm. Recovery 54 and 63%, respectively.	Alling et al. 2023*
Svalbard	Amphipod (*Gammarus setosus*) N = 20	3.25 μm (no chem id)	95 % fragments, 5% fibers; size range: 3.25–369.3 μm, 76% < 30 μm	Digestive tract	H_2O_2, 30%, 24 h, 60 °C Nile red and μ-FTIR Nicolet iN10. No method validation or recovery study.	Iannilli et al. 2019
Pechora Sea	Snow crab (*Chionoecetes opilio*) N = 23, great spider crab (*Hyas Araneus*) N = 3, and (*Pagurus pubescens*) N = 43	Not determined	Frequency of MP occurrence (FO 35%, FO 22%, FO 26%) of diet components in stomachs: 22–35%	Stomach content	Visual identification. No chemical identification, no method validation or recovery study.	Gebruk et al. 2021
Shelf of Behring and Chukchi Seas	Starfishes: (*Asterias rubens* N = 19; *Ctenodiscus crispatus* N = 64; *Leptasterias polaris* N = 15). Northern shrimp (*Pandalus borealis* N = 21); snow crab N = 59; brittle star (*Ophiura sarsi*); whelks (*Retifusus daphnelloides* N = 26; *Latisipho hypolispus* N = 19); bivalves (*Astarte crenata*, N = 28).	100 μm	0.04–1.67 items/individual, 0.02–0.46 items/g 34% 100–800 μm; 33% 800 mm–1.5 mm; 18% 1.5–2.1 mm, 17% > 2.2 mm PA (46%), PE (23%), PET (18%), CP (13%)	Whole body	Alkaline digestion followed by flotation, visual observation and μ-FTIR. Method validation: recovery of two positive controls of PE (90–95%) and PP (95–115%) of 500 μm.	Fang et al. 2018
Chukchi Sea	Sea anemone (Actiniidae und. N = 100); deposit-feeding starfish (*Ctenodiscus crispatus* N = 100); and snow crab (N = 70)	30 μm	0.03 item/g w.w., 0.09 item/ind. 0.72 item/g w.w., 0.56 item/ind. 0.02 item/g w.w., 0.027 item/ind. PES (33%), PA (11%), PET (11%)	Whole body	10% KOH, density separation (saturated NaCl), visual observation, μ-FTIR. Method validation: four positive controls with PS and PP of 50 and 100 μm, recoveries 84%–86% (50 μm), 96%–100% (Reference in reference).	Fang et al. 2021*

PES = polyester, EVA = ethylene-vinyl acetate, PP = polypropylene, PVC = polyvinyl chloride, PS = polystyrene, Teflon, polyamide 6; 6.6 (nylon), PE = polyethylene, PET = polyethylene terephthalate, SAN = styrene acrylonitrile resin, PBMA = poly(n-butyl methacrylate), ind. = individual, w.w. = wet weight., v/v = volume per volume, LOD = limit of detection. *not from systematic literature search.

Table A3.6 Effects of microplastics in Arctic invertebrate species.

Species	Particle size, polymer type, dose	LOEC (lowest observed effect concentration)	Exposure route and effects	Reference
Blue mussels (*Mytilus edulis*, *Mytilus galloprovincialis*)			Endpoints for which significant effects were recorded:	Gomes et al. 2020*
	PE fragments > 500 µm	25 µg/l	Byssus production and immune deficiency.	
	PS spheres 1–9 µm	32 µg/l	Mortality, concentration and phagocytic activity of circulation hemocytes, histopathological alterations, ROS production, and lipid peroxidation.	
	Mixture PE and PP fragments, 200–500 µm	8 ng/l, 10 µg/, respectively	Antioxidant enzymatic activity and genotoxicity.	
	PET fibers 200 to > 500 µm	3000 MP/l	Feeding behavior.	
	PE and PLA fragments 1 to 50 µm	30 µg/l	Alterations in gene and protein expression, growth.	
	PS spheres, 1–9 µm	0.42 µg/l	Larval malformations.	
	PE and PS fragments size range from < 0.05 to 99 µm	1.5 g/l	Lysosomal membrane stability.	
	PS spheres 0.1–0.99 µm	50 µg/l	Neurotoxicity.	

Species	Particle size, polymer type, dose	Duration	Exposure route and effects	Reference
Copepods				
Centropages typicus	1.7–31 µm PS 4,000,000 MP/l	24 h	Gut blockage and increased gut retention times leading to reduced feeding function.	Cole et al. 2013*
Calanus helgolandicus	20 µm PS 75,000 MP/l	24 h and 9 d	Reduced fecundity.	Cole et al. 2015*
C. finmarchicus	15–30 µm PS, spheres and fragments 50,000 and 5,00,000 MP/l	11 d	Not observed any effects on survival of adult females.	Vroom et al. 2017*
C. finmarchicus	PA fibers (10 × 30 µm) granules (10–30 µm) 50,000 MP/l	6 d	Nylon fibers can affect prey selectivity in *C. finmarchicus*; both nylon fibers and granules caused earlier molting.	Cole et al. 2019*
C. helgolandicus	20.7 µm PE 100,000 MP/l	24 h	Changes in feeding behavior and selectivity for feeding on algae.	Coppock et al. 2019*
C. hyperboreus	20.7 µm PE	5 d	Influence of PE on polycyclic aromatic hydrocarbon (PAH) bioaccumulation and oil toxicity. Up to 30% of the copepods stopped feeding. Reduced fecal pellet production rates after co-exposure to oil (1 µl/l) and MP (20 MPs/ml), MP did not influence bioaccumulation of PAHs in copepods or their fecal pellets, but chemical dispersant increased bioaccumulation. At high concentrations (20 MP/ml), MP can trigger behavioral stress responses (e.g., feeding suppression) to oil pollution in zooplankton.	Almeda et al. 2021*
C. finmarchicus, *C. glorious*, *and C. hyperboreus*	20 µm PE 20,000 MP/l	6 d	MP did not affect negatively fecal pellet production rates in any of the species at the studied exposure concentrations. However, egg production rates of copepods exposed to MP were 8 times higher compared with the controls, which suggests that MP exposure can cause stress-induced spawning in Arctic copepods.	Rodríguez-Torres et al. 2020
Crustaceans				
Isopod Idotea emarginata	1–1000 µm Microbeads, fragments and fibers PE 0.3 mg MP per g food	6 weeks	Chronic ingestion of microplastics had no distinct adverse effects on survival, intermolt duration, and growth of the isopods.	Hämer et al. 2014*

Species	Particle size, polymer type, dose	Duration	Exposure route and effects	Reference
Amphipod *Orchomene* spp.	Fibers 5 μm x 50 μm, fragments 10–100 μm; PET 0; 5,000; 50,000; 500,000, and 5,000,000 MP/kg d.w. sediment.	24 h	Sediment from two beaches, Blomstrandhalvøya in Kongsfjorden, Svalbard and amphipods. Nile Red stained, then biofilms from seawater on MP. Respiration and metabolic rates were negatively affected by PET fibers, but not fragments, significant for fibers without biofilm. Dose-response. Significantly lower movement response (all p's < 0.05).	Granberg et al. 2020*
Daphnia magna	MP fragments (42 μm) induced over 70 times higher lethal toxicity than MP beads (44.50 μm) median effective concentrations (EC50) of 4 and 276 mg/l.	2 d	Elevated temperature significantly increased ($p < 0.05$) the lethal and sublethal) lipid peroxidation and total antioxidant capacity) toxicity in exposed to MP fragments compared to those at the reference temperature.	Na et al. 2023*
Norway lobster (*Nephrops norvegicus*)	Length: 3000–5000 μm × 200 μm PP 5 fibers/feeding total: 360 MP per ind.	8 months	Reduced feeding rate, body mass, and lipid storage.	Welden and Cowie 2016*
N. norvegicus	6 μm 500–600 μm PE, PS microspheres 155 mg	3 weeks	The ingestion of MP did not affect nutritional state.	Devriese et al. 2017
Acorn barnacle (*Balanus glandula*)	Unstandardized fiber lengths PES 70,000 MP/l	24 h	Exposure through seawater in a tank. There was no detectable short-term effect of microfiber ingestion on feeding rate.	Davies et al. 2021

PES = Polyester, PS = polystyrene, PE = polyethylene, PET = polyethylene terephtalate, ind. = individual. *not from systematic literature search.

4. Occurrence and Effects of Plastic Additive Chemicals on Arctic Biota

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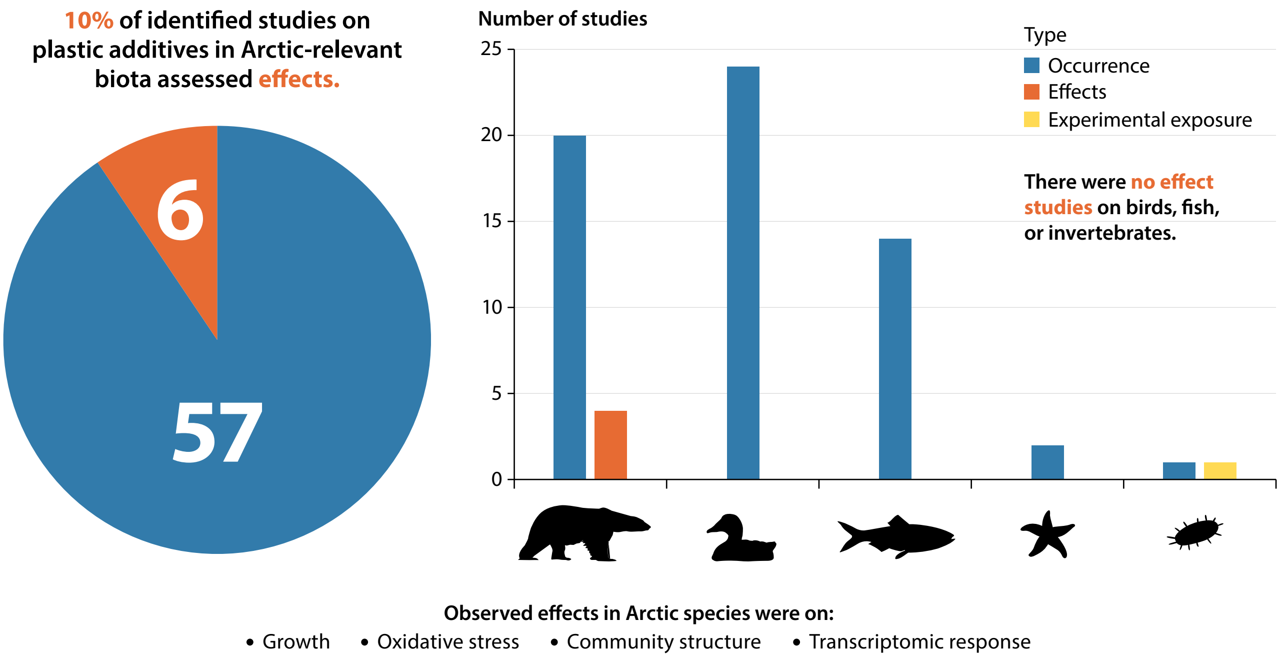

4.0 Introduction

Plastic production has rapidly increased since the 1950s (Geyer et al. 2017) leading to high plastic emissions, projected to continually increase even under the most aggressive waste reduction scenarios (Borrelle et al. 2020). Plastic pollution, including microplastics (< 5 mm), has been designated as an emerging concern in the Arctic (AMAP 2017, PAME 2019) due to its widespread presence in the northern environment (Zhang et al. 2020, Collard and Ask 2021, Bergmann et al. 2022, Citterich et al. 2023, Martin et al. 2023). Plastic pollution is a complex contaminant because it varies in size, morphology, and chemical ingredients (Rochman et al. 2019) and can elicit various effects (see AMAP 2025, Chapters 1, 2, and 3). Although there has been considerable progress on assessing the physical effects of plastic pollution (Bucci et al. 2020, MacLeod et al. 2021), the potential impacts of plastic additives have recently gained attention (Hermabessiere et al. 2017, Hahladakis et al. 2018, Pinto da Costa et al. 2023).

Plastic additive chemicals (hereafter plastic additives) include metals, organic compounds, and/or inorganic compounds that are "intentionally added to plastics to achieve a physical or chemical effect during the processing of a material or to impart functional properties to meet the requirements of the final products" (ECHA 2019). Plastic additives are mixed into the plastic polymer to give it desired characteristics or functions, such as flexibility, color, ultraviolet (UV) protection, and durability (e.g., Hermabessiere et al. 2017, Hahladakis et al. 2018). In some plastic products, additives, such as plasticizers, flame retardants, UV stabilizers, pigments, and antioxidants (Kühn et al. 2020) may account for nearly 60% of the total mass of the material (Net et al. 2015). These additives are usually not bound to the polymeric matrix, thus they can leach from plastic materials into their surrounding environment (Lithner et al. 2011, Hermabessiere et al. 2017, Hahladakis et al. 2018), depending on various physical-chemical properties (Fauser et al. 2022).

Plastic additives have been found in environmental matrices for several decades, with elevated levels of plastic additives near point sources dating back to the 1960s (e.g., UV-benzotriazoles, UV-BZTs, in sediments in the eastern United States; Cantwell et al. 2015). Further, historical deposition records identified in high Arctic ice cores have supported the presence of plastic additives in remote environments for the last several decades, e.g., organophosphate esters (OPEs) and plasticizers in high Arctic ice cores (De Silva et al. 2023). Indeed, additives previously used in plastics, e.g., polybrominated diphenyl ethers (PBDEs), hexabromocyclododecane (HBCD), and hexabromobiphenyl (HBB) but now regulated as persistent organic pollutants (POPs), have been part of monitoring programs in the Arctic for the last three decades (de Wit et al. 2010, AMAP 2017), whereas studies on other plastic additives are more limited and research-based (Hamilton et al. 2022). Current-use plastic additives (i.e., non-regulated chemical additives), like various halogenated flame retardants, have been found in Arctic soils (e.g., Lee et al. 2019), waters (Lusher et al. 2015, Martin et al. 2023), and air (e.g., Choi et al. 2020, Ademollo et al. 2021), as well as in Arctic wildlife (e.g., Vorkamp et al. 2019 and references therein).

The majority of published literature on current-use plastic additives in the Arctic has primarily focused on seabirds. For example, substituted diphenylamine antioxidants (SDPAs), UV-BZTs, (e.g., Lu et al. 2019), and phthalates (e.g., Padula et al. 2020, Routti et al. 2021) have been detected in seabirds, with potential implications for seabird health. Phthalates, commonly used as plasticizers, are one of the most studied groups of plastic additives (Meeker et al. 2009). Studies have demonstrated that phthalates can leach out of plastic products, particularly when exposed to environmental conditions like sunlight and seawater (Net et al. 2015). These chemicals have been associated with endocrine disruption, reproductive abnormalities, and developmental issues in marine organisms, including fish, crustaceans, and mollusks (Oehlmann et al. 2009, Net et al. 2015, Gunaalan et al. 2020).

Another prevalent plastic additive, bisphenol A (BPA), is used in the production of polycarbonate plastics and epoxy resins. Bisphenol A is known to leach out of plastic containers, especially when exposed to heat or acidic conditions (e.g., Krishnan et al. 1993). It can mimic estrogen in organisms, leading to reproductive and developmental abnormalities in wildlife (e.g., Crain et al. 2007, Flint et al. 2012). Research indicates that BPA exposure can impact the growth, behavior, and physiology of marine organisms, including fish and invertebrates (e.g., Crain et al. 2007). Additionally, antioxidants and UV stabilizers, used to prolong the lifespan of plastic products, can also leach out and accumulate in marine environments. Although research on their ecological impacts is ongoing, preliminary studies suggest potential adverse effects including endocrine disruption and oxidative stress on aquatic organisms (e.g., Fent et al. 2014, Liang et al. 2019).

Further, a new report by Wagner et al. (2024) synthesized data on more than 16,000 chemicals used or present in plastic products. They highlighted that only 6% of these chemicals were subject to international regulations. Wagner et al. (2024) also identified more than 4,200 plastic chemicals of concern due to persistent, bioaccumulative, mobile, and/or toxic (PMBT) properties, with over a quarter of known plastic chemicals lacking fundamental toxicity information. Thus, there is a great research need, both above and below the Arctic Circle, to understand the occurrence and effects of plastic additives in our global ecosystems.

Indeed, information on the effects of plastic additives in the Arctic is limited at a time when anthropogenic activity is growing, leading to a likely increase in plastic additive abundance and exposure in Arctic ecosystems. As climate change continues to dramatically influence Arctic environments (Rantanen et al. 2022), adding additional stress on Arctic species, it is crucial to improve our understanding of the effects of these chemicals in Arctic wildlife to better inform conservation and management efforts.

4.1 Methods

A systematic literature search was carried out using Web of Science topic search on 24 April, 2023 with the following search string:

(*arctic* OR Barents* OR Kara* OR Beaufort* OR Greenland* OR Iceland* OR Norw* OR Russia*) AND (microplastic* OR *plastic* OR polymer* OR rubber* OR *siloxane* OR tire* OR tyre* OR paint* OR TWP) AND (additive* OR bisphenol* OR phthalate OR UV-stabilizer OR organophosphate ester* OR tire leachate* OR antioxidants* OR antifouling* OR SPDAs*) AND (effect* OR accumulat* OR endocrine OR physio* OR carcinogenic OR reproductiv* OR behav* OR feed* OR *toxic* OR impact* OR risk*) AND (biota OR animal* OR organism*OR fish* OR cod* OR mussel* OR bivalve OR invertebrate* OR shrimp* OR plankton* OR sculpin* OR saithe* OR pollock* OR salmon* OR trout* OR capelin* OR whiting* OR mackerel* OR char* OR haddock* OR flounder* OR cisco* OR whitefish* OR cucumber* OR cusk OR tusk OR shark OR fulmar OR bird* OR seabird* OR duck* OR mammal* OR whale* OR caribou* OR reindeer* OR Arenicola OR predators* OR spec* OR *food).The literature search focused on a selected group of intentionally added substances (IAS), i.e., plastic additives and did not prioritize already regulated chemicals such as PBDEs, dechloranes, short-chained chlorinated paraffins, and selected PFAS, which are now regulated as POPs. For these legacy IAS, a considerable body of literature is available, including extensive Arctic wildlife monitoring and toxicological effects data (e.g., AMAP 2016, 2018, Dietz et al. 2019, Rigét et al. 2019). For IAS, such as phthalates and bisphenols, a large body of toxicological data is available, but rarely on Arctic organisms. The chemicals not covered by the applied search string, i.e., IAS present in car tire rubber, or other chemicals not referred to as additives in scientific papers on environmental pollution originally not connected to plastic pollution are a limitation of the approach applied here.

4.2 Results

This search yielded 341 articles. Upon removing reviews and irrelevant articles (e.g., manuscripts on manufacturing techniques), pairs or groups of 3 screened and sorted the remaining 36 articles into the 2 categories: studies on effects and/or studies on occurrence. Studies were included if they met the following inclusion criteria: (1) must include plastic chemical target compounds and (2) must be in a naturally occurring Arctic biota within an Arctic distribution defined by AMAP (Reiersen et al. 2024). Following this, 16 publications on occurrence and 1 publication on effects were included. The main findings of the publications are summarized in Tables 1–5, according to species groups: birds, mammals, fish, invertebrates, and protozoa. Importantly, the literature search failed to retrieve some publications known to the authors. These additional articles along with published or in-preparation theses and other gray literature were included post-hoc. These additional references on occurrence were differentiated from the results of the literature search with an asterisk (*) in each table.

The majority of studies document the presence of current-use additives in Arctic biota rather than their potential effects. Chemical additives have been shown to be present in a variety of mammals, birds, and fish, with a few additional studies on invertebrates (Tables 4.1–4.3). However, very few studies were designed to connect the presence of chemical additives to their potential source—plastic particles.

Table 4.1 The main findings of the publications from the literature search regarding birds in the Arctic. Additional literature included post-hoc is denoted by an (*). [(a)] includes banned polybrominated diphenyl ethers (PBDEs) and non-regulated, current-use halogenated flame retardants (HFRs); [(b)] regulated under the Stockholm Convention on POPs.

Species	Region	Study type	Target chemicals	Endpoints	Matrix	Reference
Black-legged Kittiwakes (*Rissa tridactyla*)	Canada	Occurrence	Substituted diphenylamine antioxidants (SDPAs), Ultraviolet (UV)-benzotriazoles (BZTs)	Spatial and species-related trends	Liver, egg	Lu et al. 2019
Northern Fulmar (*Fulmarus glacialis*)	Canada	Occurrence	SPDAs, UV-BZTs	Spatial and species related trends	Liver, egg	Lu et al. 2019
Northern Fulmar	Laboratory study	Experimental exposure	organophosph ate esters (OPE), phthalates/terephthalates, bumetriazole, phenols (antioxidants), phenyl benzoate (preservative) transformation products, and precursors, etc.	Leaching of plastic additives	Stomach oil	Kühn et al. 2020
Common Eider (*Somateria mollissima*)	Norway	Occurrence	OPEs, current-use halogenated flame retardants (HFRs), bromo- and alkylphenols, siloxanes, phthalates, organotins[a]	Trends	Eggs	Huber et al. 2015
European Shag (*Phalacrocorax aristotelis*)	Norway	Occurrence	OPEs, current-use halogenated flame retardants (HFRs), bromo- and alkylphenols, siloxanes, phthalates, organotins, chlorinated paraffins[a]	Trends	Eggs, tissues	Huber et al. 2015
European Herring Gull (*Larus argentatus*)	Norway	Occurrence	OPEs, current-use HFRs, bromo- and alkylphenols, siloxanes, phthalates, organotins[a]	Trends	Eggs, tissues	Huber et al. 2015
Northern Fulmar	Canada	Occurrence	OPEs, current-use HFRs, phthalates	Bioaccumulation, co-occurrence w/plastic particles	Muscle, liver, egg, brain, fat	Sühring et al. 2022
Black-legged Kittiwake	Canada	Occurrence	OPEs, current-use HFRs, phthalates	Additive co-occurrence w/ plastic particles	Eggs, liver	Sühring et al. 2022
Northern Fulmar	Canada	Occurrence	SDPAs, UV-BZTs, organic UV filters, phenolic antioxidants	Bioaccumulation and temporal trends	Eggs	Provencher et al. 2022
Black-legged Kittiwake	Canada	Occurrence	SDPAs, UV-BZTs, organic UV filters, phenolic antioxidants	Bioaccumulation and temporal trends	Eggs	Provencher et al. 2022
Thick-billed Murre (*Uria lomvia*)	Canada	Occurrence	SDPAs, UV-BZTs, organic UV filters, phenolic antioxidants	Bioaccumulation and temporal trends	Eggs	Provencher et al. 2022
Black Guillemot (*Cepphus grylle*)	Canada	Occurrence	SDPAs, UV-BZTs	Spatial and species patterns	Liver	*Granados-Galvan et al. 2024
Northern Fulmar	Canada	Occurrence	SDPAs, UV-BZTs	Spatial and species patterns	Liver	*Granados-Galvan et al. 2024
Thick-billed Murre	Canada	Occurrence	SDPAs, UV-BZTs	Spatial and species patterns	Liver	*Granados-Galvan et al. 2024
Peregrine Falcon (*Falco peregrinus*)	Greenland	Occurrence	Current-use HFRs[a]	Bioaccumulation	Eggs	Vorkamp et al. 2018
Black guillemot (*Cepphus grylle*)	Greenland	Occurrence	Current-use HFRs	Bioaccumulation observed	Eggs	*Vorkamp et al. 2015
Glaucous Gull (*Larus hyperboreus*)	Greenland	Occurrence	Current-use HFRs	Bioaccumulation observed	Liver	*Vorkamp et al. 2015
Black Guillemot (*Cepphus grylle*)	Faroe Islands	Occurrence	Current-use HFRs	Bioaccumulation observed for some of the studied compounds	Eggs	*Schlabach et al. 2011
Black-legged Kittiwake	Svalbard, Norway	Occurrence	OPEs, phthalates, siloxanes, BPA	Bioaccumulation observed for some compounds (DEHP, some OPEs and siloxanes), not for BPA	Liver	*Evenset et al. 2009
Common Eider	Svalbard, Norway	Occurrence	OPEs, phthalates, siloxanes, BPA	Bioaccumulation observed for some compounds (DEHP, some OPEs and siloxanes), not for BPA	Liver	*Evenset et al. 2009
Black-legged Kittiwake	Svalbard, Norway	Occurrence	OPEs	Bioaccumulation observed for some compounds and species, no OPE detection in Brünnich's Guillemot	Liver (Kittiwake); eggs (others)	*Hallanger et al. 2015

Species	Region	Study type	Target chemicals	Endpoints	Matrix	Reference
Brünnich's Guillemot /Thick-billed Murre	Svalbard, Norway	Occurrence	OPEs	Bioaccumulation observed for some compounds and species, no OPE detection in Brünnich's Guillemot	Eggs	*Hallanger et al. 2015
Glaucous Gull	Svalbard, Norway	Occurrence	OPEs	Bioaccumulation observed for some compounds and species, no OPE detection in Brünnich's Guillemot	Eggs	*Hallanger et al. 2015
Glaucous Gull	Canada	Occurrence	OPEs, current-use HFRs	Bioaccumulation observed	Liver	*Verreault et al. 2018
Northern Fulmar	Faroe Islands	Occurrence	UV-filters	Occurrence	Eggs	*Schlabach et al. 2022

Most studies quantified the compounds without elucidating the source of exposure.

Thus, there is a general knowledge gap in understanding the sources of microplastic and plastic additives (via long-range transport or local emissions) in the Arctic, as well as the biological and ecological fate of these compounds within the ecosystem.

Few studies have examined multiple taxa in a systematic way to better understand biological fate and effects of plastic additives. For example, a recent study in the Canadian Arctic carried out an analysis of two groups of plastic additives (UV absorbents and industrial antioxidants) in a suite of animal livers, including mammals, fish, and seabirds (Granados-Galvana et al. 2024). The authors found that although seabirds accumulated higher levels of these contaminants compared to the other species groups, the levels of UV absorbents and industrial antioxidants in seabirds did not correlate with the level of plastic ingestion.

4.2.1 Birds

Seabirds are one of the most widely studied animal groups for physical plastic ingestion in the Arctic (Bergmann et al. 2022). Although few studies addressed the co-occurrence of plastic particles and chemical plastic additives, several studies have analyzed chemicals that are current-use plastic additives in tissues and eggs of seabirds (Table 4.1). Relevant halogenated flame retardants (HFRs; e.g., Dechlorane plus) were also found in eggs of one terrestrial bird species (Peregrine Falcon, *Falco peregrinus*; Vorkamp et al. 2018). Although the egg samples were collected at the breeding grounds in Greenland, their concentrations likely reflect the long-range migration of the Peregrine Falcons, targeting the same wintering grounds in South America every year (Vorkamp et al. 2018). In general, there are many studies that report additive occurrence in non-Arctic seabird populations, offering insights into composition and distribution for seabirds in general for some of these species (e.g., Tanaka et al. 2019,Yamashita et al. 2021).

Interestingly, the long-term trends established for current-use HFRs in Greenland Peregrine Falcon eggs and for UV filters and antioxidants in Arctic Black-legged Kittiwakes (*Rissa tridactyla*), Thick-billed Murres (*Uria lomvia*), and Northern Fulmars (*Fulmarus glacialis*), indicated the presence of these non-regulated chemical additives dating back to the 1970s (Vorkamp et al. 2018, Provencher et al. 2022a). In fact, Provencher et al. (2022a) showed a decrease of 2,6-di-tert-butyl-4-methylphenol (BHT, an antioxidant) over time, which was also reported for 2,4,6-tribromophenyl-2,3-dibromopropyl ether (DPTE;Vorkamp et al. 2018). Given the increasing production of plastic, this was unexpected. In the case of BHT, because of its wide range of applications, including its use as a food additive (Williams et al. 1999), an observed decline could not be directly related to plastic production. Indeed, this underlines the complexity of the large variety of plastic additives and their history of use.

Kuhn et al. (2020) conducted an in vitro experiment in which various additives, including UV stabilizers and flame retardants, i.e., acetophenone, dibutylphenol, phenyl benzoate, bumetrizole, and tris(2-chloroethyl) phosphate (TCEP), and plasticizers, including di(2-ethylhexyl) phthalate (DEHP), leached from plastic particles into the stomach oil of Northern Fulmars. This DEHP persisted in the stomach oil for up to three months or more. Various additional studies conducted experiments on the leachability of brominated flame retardants (BFRs; e.g., PBDEs, dechloranes) from ingested plastic into the gizzard of Northern Fulmars (Turner 2018, Neumann et al. 2021, Collard et al. 2022) or conducted in vivo feeding experiments for BFR and UV-stabilizers (Tanaka et al. 2019, 2020). All these studies found high accumulation rates of the investigated additives in exposed tissues.The question of persistence is a crucial one because phthalates, OPEs, and other compounds used as plastic additives are usually rapidly hydrolyzed and/or metabolized, raising questions about suitable matrices and biomarkers for analysis.

The biological matrices studied in seabirds have varied across research, including muscle, liver, stomach oil, eggs, and brain tissue. This is mostly due to the availability of different biological samples in contaminant monitoring programs, as well as some studies that targeted specific tissues most likely to accumulate the contaminants of interest. Future work should consider these types of studies so that tissue concentrations can be compared to provide more insight into toxicokinetics of additives in birds and other wildlife. These studies can also help to identify the most suited indicator tissue to monitor specific compounds. Similarly, past work on contaminants in Arctic seabirds has focused on the maternal transfer of contaminants to eggs (Bustnes et al. 2003, Verreault et al. 2005, 2006). Indeed, plastic additives have been detected in seabird eggs (Table 4.1) suggesting that maternal transfer takes place even though no systematic partitioning studies have been conducted. We suggest combining more detailed studies on toxicokinetics with other research (e.g., physical plastic ingestion, analyses of other contaminant groups) when possible.

Northern Fulmars have been proposed as a Priority 1 indicator for plastic monitoring in the Arctic (AMAP 2021, Provencher et al. 2022b), thus monitoring the activities of these birds offers the possibility of linking the occurrence of plastic particles in the bird stomachs to the occurrence of plastic additives in the bird tissue. Combined with ongoing studies into leaching behavior from plastic particles, these approaches would improve the understanding of plastic particles as a direct exposure source to chemical additives in Arctic birds and could then be used to assess the potential impacts of these additives on the birds.

4.2.2 Mammals

Although marine mammals are widely monitored for contaminants in the Arctic, with a general focus on POPs (AMAP 2016, Rigét et al. 2019), there are few studies examining current-use plastic additives in both terrestrial and marine mammals (Table 4.2). Granados-Galvan et al. (2024) found that concentrations of UV absorbents and industrial antioxidants in walrus (*Odobenus rosmarus*) and ringed seals (*Pusa hispida*) varied between tissue types. Specifically, in walrus, UV-BZTs were significantly different between tissues (blubber > muscle > liver). These findings have direct relevance for designing monitoring programs and effects studies because they provide insight into environmentally relevant concentrations and tissue distribution, which could inform future research on potential impacts on both animal and human health.

Few papers have reported plastic additives in terrestrial mammals (Table 4.2). Although current-use HFRs (including, PBDEs, pentabromotoluene, hexabromobenzene, hexabromycylododecane) were detected in the lichen (*Cladina rangiferina* and *Cetraria nivalis*)-caribou (*Rangifer tarandus*)-wolf (*Canis lupus*) food-chain in a Canadian study (Morris et al. 2018), there was little evidence of biomagnification. However, given the importance of terrestrial mammals, such as caribou (reindeer), as a food source and livelihood to many communities and Arctic residents', this is a gap that should be addressed in future work. Various terrestrial mammals are currently monitored for other contaminants in the Arctic (e.g., mercury or lead; Gamberg and Braune 1999, Aastrup et al. 2000, Gamberg et al. 2005, AMAP 2021, Morris et al. 2022) and additional campaigns have been conducted (e.g., for perfluoroalkyl substances, PFAS; Bossi et al. 2015, Muir et al. 2019), thus we suggest adding plastic additive analyses to these projects when possible. For example, in the Canadian Arctic, there is an ongoing project in collaboration with several Indigenous communities to obtain liver or muscle samples from caribou (which are monitored for contaminants such as lead) to inform safe harvest (Gamberg and Braune 1999, Gamberg et al. 2005). The AMAP Core monitoring program in Greenland (Rigét et al. 2016) also includes collections of reindeer, offering possibilities of connections with plastic-related studies in Canada or elsewhere.

The current literature largely focuses on flame retardants, with current-use HFRs frequently added to PBDE analyses. Although HFRs and OPE have been found in marine mammals, levels remain low, and biomagnification does not seem to occur (e.g., Hallanger et al. 2015). Similarly, Routti et al. (2021) did not observe bioaccumulation of phthalates in Arctic marine mammals (i.e., polar bear; blue, fin, and bowhead whales; Table 4.2) or in their metabolites, suggesting metabolization and excretion from the body. Consequently, monitoring of non-persistentent plastic additives needs careful consideration regarding the suitable biomarkers of exposure and target tissues/organs for analysis.

Although information on exposure to plastic additives is sparse for marine mammals, there is even less information on observed effects. Given the potential toxicity of compounds of interest, (Routti et al. 2021), effects on Arctic mammals can be assumed, but evidence is lacking. Di(2-ethylhexyl) phthalate was found to modulate the transcriptional activity of whale thyroid hormone receptor beta (THRB) in in vitro gene assays (Routti et al. 2021), but the applied exposure concentrations were higher than those measured in Arctic whales (Table 4.2), and no effects were found in other assays or for other species.

4.2.3 Fish

Although fish are widely monitored for various contaminants across the Arctic, our literature search revealed few publications on plastic additives in Arctic fish (Table 4.3). Granados-Galvan et al. (2024) found higher detection frequencies and concentrations of UV-BZTs in landlocked Arctic char (*Salvelinus alpinus*) compared to sea-run Arctic char. This suggests that freshwater Arctic fish may be more susceptible to accumulating UV-BZTs, which is similar to observations made about other contaminants such as mercury and PFAS (Swanson et al. 2011, Muir et al. 2019). Additionally, Hamilton (2023) identified 27 individual OPEs in sea-run Arctic char liver tissues from the central Canadian Arctic, reporting a higher proportion of non-Cl-OPEs than Cl-OPEs, which agreed with results of Sühring et al. (2022) who reported similar patterns in seabirds. However, Hamilton (2023) did not observe a relationship between Arctic char age and OPE concentrations, suggesting that the analyzed set of OPEs did not bioaccumulate. Because both land-locked and sea-run Arctic char are important subsistence foods for many Indigenous communities, understanding the presence and possible physiological effects of potentially harmful chemicals, including plastic additives, in Arctic char and other Arctic fish is a priority. Indeed, adverse effects on these fish or human exposure associated with contaminants in fish may have important implications on human health and/or wellbeing, thus future research in this area is strongly encouraged.

Evenset et al. (2009) reported several types of plastic additives, including OPEs, DEHP, siloxanes, and BPA, in three fish species from Svalbard in the Norwegian Arctic. Di(2-ethylhexyl) phthalate was found occasionally, while BPA was undetectable in all samples. Of the siloxanes, only the cyclic molecules D4, D5, D6 could be detected in fish. Moreover, the authors detected 8 of the 13 OPEs in the analyzed fish, with highest levels being for 2-ethylhexyl-diphenylphosphate (EHDPP), which is thought to bioaccumulate in organisms (Muir et al. 1988). However, EHDPP was below detection limits in a study on lake trout (*Salvelinus namaycush*) in Canadian Arctic/sub-Arctic lakes, including Great Bear Lake and Kusawa Lake,

Table 4.2 The main findings of the publications from the literature search regarding mammals in the Arctic. Additional literature included post-hoc is denoted by an (*). [(a)] includes banned polybrominated diphenyl ethers (PBDEs) and non-regulated, current-use halogenated flame retardants (HFRs); [(b)] regulated under the Stockholm Convention on POPs.

Species	Region	Study type	Target chemicals	Endpoints	Matrix	Reference
North Atlantic fin whale (*Balaenoptera physalus*)	West Iceland	Occurrence	Phthalates	Bioaccumulation and time trends addressed, but not observed	Muscle	Garcia-Garin et al. 2022
North Atlantic fin whale	West Iceland	Occurrence	Organophosphate esters (OPEs)	Bioaccumulation/ biomagnification addressed, but not observed	Muscle	Garcia-Garin et al. 2020
Ringed seal (*Pusa hispida*)	Canada	Occurrence	Substituted diphenylamine antioxidants (SPDAs), Ultraviolet (UV)-benzotriazoles (BZTs)	Spatial and species-related trends	Liver	Lu et al. 2019
Blue whale (*Balaenoptera musculus*)	Svalbard Archipelago, Norway	Occurrence and effects	Phthalates	Transcriptional activity of peroxisome proliferator-activated receptor gamma (PPARG), glucocorticoid receptor (GR) and the thyroid hormone receptor beta (THRB)	Blubber, plasma	Routti et al. 2021
Fin whale	Svalbard Archipelago, Norway	Occurrence and effects	Phthalates	Transcriptional activity of PPARG, GR, and THRB	Blubber, plasma	Routti et al. 2021
Bowhead whale (*Balaena mysticetus*)	Svalbard Archipelago, Norway	Occurrence and effects	Phthalates	Transcriptional activity of PPARG, GR, and the THRB	Blubber, plasma	Routti et al. 2021
Polar bear (*Ursus maritimus*)	Svalbard Archipelago, Norway	Occurrence and effects	Phthalates	Transcriptional activity of PPARG, GR, and THRB	Adipose tissue, plasma	Routti et al. 2021
North Atlantic fin whale	Off West Iceland	Occurrence	OPEs, HFRs, and short-chain chlorinatedparaffins (SCCPs)[b]	Transplacental transfer	Blubber, fetal tissue from dorsal fin	Sala et al. 2022
Polar bear	Greenland	Occurrence	OPEs	Metabolism	Adipose tissue	Strobel et al. 2018
Ringed seal	Greenland	Occurrence	OPEs	Metabolism	Blubber	Strobel et al. 2018
Polar bear	Greenland	Occurrence	Current-use HFRs	Bioaccumulation	Blubber/adipose tissue	*Vorkamp et al. 2015
Ringed seal	Greenland	Occurrence	Current-use HFRs	Bioaccumulation	Blubber/adipose tissue	*Vorkamp et al. 2015
Caribou (*Rangifer tarandus groenlandicus*)	Canada	Occurrence	Current-use HFRs	Bioaccumulation/ biomagnification addressed, but no biomagnification observed	Muscle, liver	*Morris et al. 2018
Wolf (*Canis lupus*)	Canada	Occurrence	Current-use HFRs	Bioaccumulation/ biomagnification addressed, but no biomagnification observed	Muscle, liver	*Morris et al. 2018
Arctic fox (*Vulpes lagopus*)	Svalbard, Norway	Occurrence	OPEs	Bioaccumulation/ biomagnification addressed, but no biomagnification observed	Liver (Arctic fox), blubber (ringed seals), plasma (others)	*Hallanger et al. 2015
Ringed seal	Svalbard, Norway	Occurrence	OPEs	Bioaccumulation/ biomagnification addressed, but no biomagnification observed	Blubber	*Hallanger et al. 2015
Harbour seal (*Phoca vitulina*)	Svalbard, Norway	Occurrence	OPEs	Bioaccumulation/ biomagnification addressed, but no biomagnification observed	Plasma	*Hallanger et al. 2015
Polar bear	Svalbard, Norway	Occurrence	OPEs	Bioaccumulation/ biomagnification addressed, but no biomagnification observed	Plasma	*Hallanger et al. 2015
Ringed seal	Canada	Occurrence	Current-use HFRs	Bioaccumulation	Blubber	*Houde et al. 2017
Reindeer	Finland, Sweden	Occurrence	Chlorinated paraffins, dechlorane	Occurrence	Muscle	*Schlabach et al. 2022
Beluga (*Delphinapterus leucas*)	Canada	Occurrence	Current-use HFRs	Occurrence	Blubber	*Tomy et al. 2008

Table 4.3 The main findings of the publications from the literature search regarding fish in the Arctic. Additional literature included post-hoc is denoted by an (*). [(a)] includes banned polybrominated diphenyl ethers (PBDEs) and non-regulated, current-use halogenated flame retardants (HFRs); [(b)] regulated under the Stockholm Convention on POPs.

Species	Region	Study type	Target chemicals	Endpoints	Matrix	Reference
Greenland shark (*Somniosus microcephalus*)	Greenland	Occurrence	4-Nonylphenol (4-NP), 4-NP mono- and diethoxylate precursors; BPA	Bioaccumulation observed	Muscle, liver	Ademollo et al. 2018
Arctic char (*Salvelinus alpinus*)	Canada	Occurrence	Substituted diphenylamine antioxidants (SDPAs), Ultraviolet (UV)-benzotriazoles (BZTs)	Spatial and species patterns	Liver	*Granados-Galvan et al. 2024
Arctic char	Canada	Occurrence	Organophosphate esters (OPEs)	No bioaccumulation observed	Muscle, liver	*Hamilton, 2023
Capelin (*Mallotus villotus*)	Svalbard, Norway	Occurrence	OPEs	Bioaccumulation observed	Whole fish	*Hallanger et al. 2015
Lake trout	Canada	Occurrence	OPEs	Bioaccumulation observed for some compounds	Whole fish	*McGoldrick et al. 2014
Walleye (*Sander vitreus*)	Canada	Occurrence	OPEs	Bioaccumulation observed for some compounds	Whole fish	*McGoldrick et al. 2014
Atlantic cod (*Gadus morhua*)	Svalbard, Norway (Arctic char: Bjørnøya, Norway)	Occurrence	OPEs, phthalates, siloxanes, BPA	Bioaccumulation observed for some compounds of each group (not for BPA)	Liver	*Evenset et al. 2009
Polar cod (*Boreogadus saida*)	Svalbard, Norway (Arctic char: Bjørnøya, Norway)	Occurrence	OPEs, phthalates, siloxanes, BPA	Bioaccumulation observed for some compounds of each group (not for BPA)	Liver, whole fish	*Evenset et al. 2009
Arctic char	Svalbard, Norway (Arctic char: Bjørnøya, Norway)	Occurrence	OPEs, phthalates, siloxanes, BPA	Bioaccumulation observed for some compounds of each group (not for BPA)	Muscle	*Evenset et al. 2009
Atlantic cod	Faroe Islands; Iceland	Occurrence	Current-use halogenated flame retardants (HFRs)	Bioaccumulation observed for some, but not all compounds	Liver	*Schlabach et al. 2011
Arctic char	Faroe Islands	Occurrence	Current-use (HFRs)	Bioaccumulation observed for some, but not all compounds	Muscle	*Schlabach et al. 2011
Atlantic cod	Faroe Islands	Occurrence	UV-filters, Chlorinated paraffins, declorane	Occurrence	Liver	*Schlabach et al. 2022
Brown trout (*Salmo trutta*)	Faroe Islands	Occurence	Chlorinated paraffins, declorane	Occurence	Muscle	*Schlabach et al. 2022

where TCEP was the dominant OPE observed (McGoldrick et al. 2014). Importantly, the potential effects of these additives were not assessed in any of these fish species. Previous studies have reported translocation of plastic particles into liver and muscle tissue in Arctic fish (e.g., salmonids; Hamilton 2023), which raises the question as to whether there is a relationship between the concentration of physical plastic pollution and the chemical additives in Arctic fish tissues and other wildlife of ecological and subsistence importance.

The Greenland shark (*Somniosus microcephalus*) is a particularly interesting species because it is high in the food web, can reach an age of centuries, and thus can be a strong indicator of bioaccumulating chemicals (Adomello et al. 2018). Nonylphenol (4-NP, an antioxidant), its ethoxylate precursors, and BPAs are endocrine disrupting compounds that were all detected in Greenland sharks from waters around Greenland (Adomello et al. 2018). Concentrations in the liver were generally higher than in the muscle. In the liver, concentrations of ethoxylates were highest, followed by 4-NP and BPA. Adomello et al. (2018) proposed continuous and chronic exposure possibly related to marine plastic debris because these compounds are not considered persistent. As a long-lived species, studies on the potential impact of the bioaccumulation of these and other additives in Greenland sharks are encouraged.

4.2.4 Invertebrates

There are very few studies that have assessed plastic additives in Arctic invertebrates (Table 4.4). Garcia-Garin et al. (2020) evaluated OPEs in krill (*Meganyctiphanes norvegica*) as the main feeding source of North Atlantic fin whales (*Balaenoptera physalus*). Out of 19 OPEs included in the study, 7 and 5 were detected in fin whales and krill, respectively. The concentrations were comparable on a lipid basis, leading Garcia-Garin et al. (2020) to conclude that although OPEs were detected in both krill and whales, there was no evidence of biomagnification. Tri-n-butyl phosphate (TNBP), isopropylated triphenyl phosphate (IPPP), and triphenylphosphine oxide (TPPO) were present in both krill and fin whale samples, with TNBP at the highest concentrations. This study is one of few to report OPEs in krill,

Table 4.4 The main findings of the publications from the literature search regarding invertebrates. Additional literature included post-hoc is denoted by an (*).

Species	Region	Study type	Target chemicals	Endpoints	Matrix	Reference
Krill (*Meganyctiphanes norvegica*)	West Iceland	Occurrence	Organophosphate esters (OPEs)	Bioaccumulation / biomagnification not observed	Whole body	Garcia-Garin et al. 2020
Blue mussels (*Mytilus edulis*)	Iceland; Northern Norway	Occurrence	Current-use halogenated flame retardants (HFRs)	Bioaccumulation observed	Whole body	*Schlabach et al. 2011

thus further research is needed to determine the potential effects of these concentrations on krill and other invertebrate species.

Current-use HFRs were widely detected in blue mussels (*Mytilus edulis*) from the Arctic, but at low concentrations (Schlabach et al. 2011). This is consistent with the general findings of low concentrations of this diverse group of plastic additives in Arctic biota, with few exceptions (Vorkamp et al. 2019). However, it is important to note that mussels might represent water or sediment contamination as a result of their preferred habitat and filter feeding behavior. Because of this, routes of contaminant exposure may differ from that of fishes and may provide additional information on contaminant levels in the environment. Given their low trophic position, they have not been monitored extensively for POPs in the Arctic, but are frequently used in other pollutant monitoring programs, such as the Coordinated Environmental Monitoring Program of the Oslo-Paris Commission for the Protection of the Northeast Atlantic (OSPAR; Beyer et al. 2017), and have proven useful for the monitoring of plastic pollution (Bråte et al. 2018).

Sørensen et al. (2023) applied non-target and suspect screening to an array of Arctic marine zooplankton from the Kongsfjord in Svalbard, Norway and identified various chemical compounds, including plastic additives, antioxidants, antimicrobials, and flame retardants. Pelagic amphipods (*Themisto abyssorum*, *T. libellula*), copepods (*Calanus finmarchicus*, *C. glacialis*, *C. hyperboreus*), arrow worms (*Parasagitta elegans*), and krill (*Thysanoessa inermis*) as well as benthic scavenging amphipods (*Onisimus* sp., *Anonyx* sp.) were among the species analyzed. The benthic amphipods and lipid-rich copepods contained the highest abundance of identified and tentative compounds in comparison to the other test species (Sørensen et al. 2023). Only five compounds (4-methyl-5-thiazoleethanol, H-indole3-carboxaldehyde, dicyclopentadienediepoxide, benzeneacetamide, and 2-phenylpropenal) were identified in > 50% of the samples (Sørensen et al. 2023). Several compounds were found only in pelagic invertebrate samples, including flame retardants and UV-stabilizers, several of which are currently undergoing assessment as persistent bioaccumulative toxic (PBT) chemicals. Of the compounds tentatively identified in samples by the non-target screening protocol, phthalide, benzothiazole, benzophenone, 2-(methylthio)benzothiazole, and n-butylbenzenesulfonamide were quantified and found in levels > limit of detection (LOD) in only one sample. Triclosan and BPA were quantified in almost all invertebrate species. The concentrations reported in invertebrate samples were several orders of magnitude above what would be expected based on simple bioconcentration from water (Sørensen et al. 2023), suggesting that plastic may be an important exposure mechanism for these species. Importantly, the potential effects of these chemicals were not assessed for any of the species in this study, again emphasizing the need for future research in this area. Further investigation of additive chemical loads in invertebrates is needed, and the use of non-target approaches is encouraged.

4.2.5 Protoctista and prokaryota

Two publications were identified in our literature search on additive occurrence and effects in Arctic protoctista and prokaryota; one on microalgae (*Chaetoceros muelleri*; Wang et al. 2021) and one on the composition of bacterioplankton communities (Martinez-Varela et al. 2021; Table 4.5). Wang et al. (2021) investigated the role of polystyrene (PS) and triphenyl phosphate (TPhP) on growth, photosynthesis, and oxidative stress on microalgae (*Chaetoceros meulleri)* and determined that PS inhibited the growth of the algae and caused dose-dependent oxidative stress, leading to cell damage. Combined exposure led to a decrease in cell damage, suggesting antagonistic effects, yet cell damage was still noticeable and not fully alleviated or reversed. Martinez-Varela et al. (2021) investigated the influence of anthropogenic dissolved organic carbon (ADOC), including n-alkanes, polycyclic aromatic hydrocarbons (PAHs), and OPEs, on the structure and metabolic functioning of marine bacterioplankton communities that may play a role in modulating the fate of additives through degradation. Martinez-Varela et al. (2021) observed gene expression profiles that showed the presence of stress-response strategies to cope with ADOC toxicities and also observed that microbial communities can be influenced by background concentrations of ADOC (Martinez-Varela et al.

Table 4.5 The main findings of the publications from the literature search regarding algae/bacteria. Additional literature included is denoted by an (*).

Species	Study type	Target chemicals	Endpoints	Reference
Microalgae (*Chaetoceros meulleri*)	Experimental exposure	Polystyrene (PS) and triphenyl phosphate (TPhP)	Growth; oxidative stress	Wang et al. 2021
Bacterioplankton communities	Occurrence	Alkanes, polycyclic aromatic hydrocarbons (PAHs), Organophosphate esters (OPEs)	Communities' structure and metabolic functioning	Martinez-Varela et al. 2021

2021). Northern microbial communities exhibited the greatest changes after exposure, whereas southern communities were more resilient (Martinez-Varela et al. 2021). Outside the Arctic, the results indicate potential effects on microbial communities from chemicals associated with plastics, which may add to stressors in Arctic ecosystems and warrant further investigation.

4.3 Discussion

Plastic additives represent a chemically diverse group of more than 16,000 compounds (Weisinger et al. 2021, Wagner et al. 2024), which vary in their physical-chemical properties, their amount in the polymer, their leaching behavior, and in their toxic effects. Most monitoring efforts so far have approached specific groups of chemical additives as "chemicals of emerging Arctic concern (CEAC)" from a history of POP monitoring. Indeed, many studies use samples originally collected for POP monitoring or extend existing analytical methods to related compounds. For example, based on established PBDE monitoring, it was an obvious next step to screen for current-use HFRs and/or OPEs as the most important replacement compounds. This common approach of expanding current monitoring programs can explain some of the observations in this review:

- Most studies only included one or a few compound groups rather than a broad range of plastic additives.
- Microplastics/plastic pollution as an identified source of the plastic additives detected in an animal is still somewhat rare.
- Targeted parent compounds are often the focus of studies because methods for determining metabolites are not always available.
- Often only one organ or tissue was analyzed, involving assumptions of toxicokinetics.

The commonly studied non-POP plastic additives include flame retardants (HFRs and OPEs), UV filters (benzotriazoles and other groups, except UV-328, which has been regulated under the Stockholm Convention), antioxidants, and plasticizers (phthalates). All these compounds have been found in Arctic animals, albeit often at low concentrations. These observations are likely related to chemical transformation processes (e.g., via metabolism) following ingestion and/or exposure, which are not yet well-understood. A better understanding of transformation and toxicokinetic processes is therefore a priority for choosing the most suitable matrices and biomarkers for monitoring efforts (Hamilton et al. 2022).

Although slim, the results outlined above indicate a presence of plastic additives in a range of Arctic biota, highlighting a major research gap in the monitoring of the occurrence and effects of plastic additives across the circumpolar North. Current knowledge points toward an abundant presence of some additives in some Arctic wildlife and not in others, e.g., SDPAs (substituted diphenylamine antioxidants), OPEs, suggesting there may be differences in exposure, uptake, and metabolization among species. These species-specific differences are largely unknown for current-use plastic additives and need to be explored further to find suitable organisms for monitoring. The obvious connection to plastic monitoring in Northern Fulmars (as a Priority 1 indicator in the Arctic and included in OSPAR), combined with relatively high levels of some plastic additives in their tissues and/or eggs (e.g., OPEs, UV-stabilizers), might favor their use as an indicator species for monitoring. However, our review also shows that additional monitoring priorities and programs are required to better understand the occurrence and effects of plastic additives in the Arctic. For example, deploying suspect-screening and non-targeted analysis in an organism that ingests high levels of plastics (e.g., Northern Fulmars) can provide data to inform chemical monitoring priorities across different monitoring programs. Further, adding different monitoring species, such as mussels, fish, and/or marine and terrestrial mammals to plastic additive monitoring (targeted or non-targeted) will provide information for human health risk assessments.

Monitoring of POPs, CEACs (some of which are plastic additives), and heavy metals is carried out through national programs with the resulting national data used in regular circumpolar assessments, often coordinated by AMAP. An important factor is the need for comparable data quality across national studies to allow robust spatial and temporal comparisons. Although analytical methods undergo rapid developments, for example toward lower detection limits, robust quality assurance/quality control (QA/QC) programs and interlaboratory comparison studies must be implemented to ensure data comparability.

Along with methodological developments for targeted analysis, suspect and non-targeted screening techniques are likely to gain importance in Arctic studies. These techniques are widely applied in research, with many efforts toward more harmonization and standardization of QA/QC procedures and expansions of databases. Large compound lists, such as those of plastic additives, are well-suited for suspect screening approaches mainly to identify compounds rather than quantify them, with initial attempts of semi-quantification (Hollender et al. 2023).

The primary aim of this review was to compile knowledge of effects of current-use plastic additives on Arctic biota. However, few studies to date have addressed effects of these chemicals in Arctic biota. Consequently, the current state of knowledge remains similar to that outlined in previous AMAP reviews (2017). Below, some specific points regarding chemical additives in the Arctic are discussed in depth, with a focus on knowledge gaps and considerations for future effects studies and monitoring.

4.3.1 Fate and transport

Given the environmental degradability of most current-use plastic additives, the detection of these chemicals in Arctic regions suggests plastic particles as a transport vector (either oceanic or atmospheric; Andrade et al. 2021). However, the current literature is limited in characterizing the environmental fate of these plastic additives, including their long-range transport with plastic particles and their exposure to wildlife. For some non-persistent plastic additives, local emission sources exist in the Arctic, such as untreated wastewater effluents (Gewurtz et al. 2020), which can be a direct point source of the chemicals as well as a source of microplastic particles and fibers (von Friesen et al. 2020, Herzke et al. 2021).

Other local sources of plastics in the Arctic include landfills (Haar et al. 2023), fishing (Liboiron et al. 2021), and shipping activities (Hamilton et al. 2021). In addition, plastic particles can be transported to the Arctic from more southern regions by ocean currents (Cózar et al. 2017), riverine transport (Zhang et al. 2023), wind (Bullard et al. 2021), atmosphere (Tatsii et al. 2023), and biotransport by transient wildlife species (Bourdages et al. 2021). Bourdages et al. (2021) showed that Arctic-breeding seabirds can act as vectors of microplastic transport from sea to land. The retention time of plastics in seabirds can be days to months (Ryan 2015), and it is likely that these microplastics and their associated additives are acquired in more southern regions throughout migration.

4.3.2 Leaching behavior of plastic additive chemicals

The size, shape, and surface area of plastic items and particles have an impact on the leaching of plastic additive chemicals into water or the tissues and fluids of organisms (Gaylor et al. 2013, Sun et al. 2019). Of these, a higher surface area (including microporous structures) facilitates greater interaction with the surrounding water or biota tissue/fluid, enabling additives to leach more readily (Fu et al. 2008, Lithner et al. 2011). Smaller plastic particles generally have a larger surface area relative to their volume, providing more sites for chemical interactions and leaching to occur. As a result, microplastics exhibit large surface areas per unit volume, intensifying their leaching potential compared to macroplastic litter (> 5 mm). In a study investigating the release of brominated flame retardants, Sun et al. (2019) found that leaching increased significantly with a reduction in particle size from 50–100 μm to 10–20 μm.

Irregularly shaped plastic particles have more surface roughness (e.g., sharp edges) creating additional crevices and grooves where chemicals can be released, again linked closely to the available surface area. In contrast, spherical particles will typically have a smaller surface area and a corresponding lower rate of leaching resulting from a reduced number of contact points with its surroundings. It is worth noting here that the degree of plastic degradation or weathering can significantly alter the size, shape/morphology, and surface area of a plastic item in the marine environment. In particular, mechanical abrasion/degradation and UV degradation can fragment larger plastic items into smaller, irregular-shaped particles with an increased surface area for leaching (Gaylor et al. 2013, Liu et al. 2020a, Sait et al. 2021, Sørensen et al. 2021, Luo et al. 2022). Furthermore, “fresh” polymer material and surfaces revealed through fragmentation may contain higher concentrations of plastic additives relative to already leached surfaces, which also drives the leaching process.

A broad range of intrinsic plastic additive properties, intrinsic physicochemical polymer properties, and extrinsic environmental factors all influence the leaching behavior of additives from plastic and elastomer materials present in the natural environment (Suhrhoff and Scholz-Böttcher 2016, Wei et al. 2019, Xu et al. 2020, Bridson et al. 2023, Henkel et al. 2023). The process of additive leaching from microplastics into an aqueous solution is a combination of both internal (intraparticle diffusion, IPD) and external diffusion (aqueous boundary layer diffusion, ABLD; Luo et al. 2020, Henkel et al. 2023). Depending on the physicochemical properties of the chemical and the environmental conditions, either diffusion in the plastic matrix or diffusion in the plastic-water boundary layer, or both, could control the overall mass transfer rate (Sun et al. 2019, Do et al. 2022); the slower of the two processes limits the overall diffusion of the additive (Grathwohl 1998). Although ABLD has been seen as important for sorption/desorption processes (relevant for interaction of plastics and environmental contaminants), IPD appears to be more important for the leaching of additives (Sun et al. 2019). The plastic additive properties do not change, however, external conditions can also impact the physical properties of the polymeric materials. Table 4.6 provides a detailed overview of the intrinsic and extrinsic parameters known to impact the leaching of additives from plastic materials in aquatic environments. Parameters with the greatest influence on plastic additives are discussed in further detail, including from an Arctic perspective. Finally, following ingestion of plastic items, additive leaching directly into exposed organisms is addressed.

4.3.2.1 Plastic additive chemical molecular weight and size

The molecular weight of plastic additives can have a strong impact on their leaching behavior (Poças et al. 2008, Hahladakis et al. 2018, Sun et al. 2019). The molecular weight range of known plastic additive substances is wide, estimated to be from 200–2000 g mol^{-1} (Hansen et al. 2013, Hahladakis et al. 2018). Higher molecular weight/larger molecule additives

Table 4.6 Intrinsic and extrinsic factors influencing the leaching of chemical additives from plastic. Properties highlighted in gray are considered to play a key role in controlling additive chemical leaching from plastics specifically in Arctic aquatic environments.

Intrinsic property of chemical additive	Intrinsic polymer and plastic properties	Extrinsic conditions
Mass transfer coefficient % in plastic	Polymer composition	Density
Molecular weight	Size	Composition/salinity
Octanol-water coefficient (K_{ow})/polarity	Shape/morphology	Surfactancy
Volatility	Surface area	pH
Solubility	Crystallinity	Temperature
Diffusion coefficient in leachate medium	Glass transition temperature	UV exposure
Diffusion coefficient in polymer		Turbulence/energy
Concentration		

(e.g., polyethylene glycol) tend to have stronger interactions/bonds (e.g., Van der Waals forces, hydrogen bonds, ionic bonds) within the plastic matrix, but this cannot necessarily be generalized. The specific bonding mechanism depends on the chemical properties of both the additive and the polymer. Plastic additives that form stronger bonds with the polymer matrix leach more slowly into the surrounding environment. In contrast, smaller plastic additives with lower molecular weights (e.g., dioctyl phthalate) often have weaker interactions with the plastic matrix, making them more susceptible to leaching (Groh et al. 2019). It is also worth highlighting that molecular configuration plays a role in controlling the rate of leaching, with more spherical/compact molecules diffusing through the plastic matrix more slowly than elongated molecules of the same molecular weight (Berens and Hopfenberg 1982, Rusina et al. 2010a). Similarly, additives that fit more tightly into the polymer pore spaces have a reduced capacity to migrate from the polymer (Teuten et al. 2007).

4.3.2.2 Plastic additive molecular polarity and water solubility

The leaching behavior of chemical additives from plastic into water is highly influenced by their water solubility and polarity, as measured by their octanol-water partition coefficient (K_{ow}). Additives with a preference for nonpolar environments tend to leach less readily into water from plastics, especially if the plastic itself is nonpolar (e.g., polyethylene). In this scenario, the shared nonpolar nature reduces the solubility of the plastic additive in water, limiting leaching. Similarly, additives that favor polar environments, i.e., chemical polarity, are better retained in polar plastics such as polyvinyl chloride (PVC) when they come into contact with water. Diffusion across the plastic-water boundary layer (ABL) is usually a rate-limiting step for hydrophobic organic compounds (HOCs; i.e., compounds that are not water soluble) with a high octanol-water coefficient in planar passive samplers. (Rusina et al. 2007, Lohmann 2012). For example, Lee et al. (2018) observed that the desorption of HOCs with low plastic-water partition coefficients from polyethylene (PE) and polypropylene (PP) films was dominated by IPD while chemicals with higher plastic-water partition coefficients was determined by ABL diffusion. Still, there is a need to better understand these processes as they pertain to plastic additives, which are diverse in their physical chemical properties (e.g., range in polarity, hydrophobicity, etc.). The plastic material/water partition ratio is important in understanding leaching behavior of additive chemicals in the environment. Although the plastic material/water partition ratio may be correlated to the octanol/water partition coefficient for many combinations of plastic types and additives, this will not always be the case.

4.3.2.3 Plastic additive chemical concentration

The concentration or mass fraction of additives in plastic materials can significantly influence their leaching behavior into surrounding water (Hansen et al. 2013, Do et al. 2023). The rate of leaching decreases as the concentration gradient between the polymer and the surrounding environment becomes smaller (Fauser et al. 2022). The plastic material effectively acts as a reservoir of the chemicals, where a larger reservoir has the capacity to lead to faster and more extensive leaching. In reality, plastics rarely contain a single additive; rather, they are intricate blends designed to meet specific performance criteria. These mixtures often contain multiple additives with very different properties and at different relative concentrations. The interaction between different additives can also affect their leaching behavior; synergistic effects enhance the leaching of one or several plastic additives, while antagonistic effects reduce the leaching tendencies. These complex interactions can make it very difficult to accurately predict the absolute and relative leaching of plastic additives.

It is important to note that the plastic to water ratio (often referred to as loading) can also play an important role in determining the composition and concentration of the resulting leachate. Beiras et al. (2019) previously showed that loading and toxicity are not linearly related, suggesting that at higher plastic loadings the solubility limits of certain toxic chemicals are reached and no further increase in leachate concentrations are obtained. Similarly, no difference was observed in the leaching of low-solubility PAHs with increasing car tire rubber loading (Halsband et al. 2020). Furthermore, a near 10:1 ratio of the more soluble benzothiazole was observed between 10 and 1 g/L leachates, while the ratio between 100 and 10 g/L leachates was closer to 2:1, likely meaning that saturation had been reached. A key issue is that most reported leaching studies have used particle/plastic concentrations that are orders of magnitude above those found or estimated to occur in aquatic environments (Delaeter et al. 2022). For practical reasons, these leachate “stocks” are often diluted to obtain the concentration ranges of “leachates” for use in toxicity studies (Gunaalan et al. 2020), which may end up underestimating the toxicity of leachates.

Although the focus of Gunaalan et al. 2020’s work is specifically on the leaching of plastic additives present in the matrix of polymer materials, it is worth highlighting a key difference between the leaching/desorption of plastic additive chemicals and pollutant chemicals in the environment that adsorb to the surface of plastic material. Plastic additive chemicals are generally uniformly dispersed throughout the polymer matrix, and thus plastic additives are already at the polymer/water interface. Therefore, diffusion in the ABLD must be a rate-limiting step. However, the leaching process causes a gradient of higher concentrations in the core of the material to lower concentrations near the surface. This scenario drives IPD as the system slowly attempts to reach an equilibrium. In contrast, chemical pollutants sorbed from the surrounding environment can be present at the surface layers of the plastic material. Sun et al. (2019) suggested that the greater abundance of sorbate molecules at the particle surface would render diffusion in the plastic matrix (IPD) less important (i.e., there are already enough sorbate molecules at the plastic-water boundary layer). A number of other studies have also argued that IPD is the rate-limiting step in the transfer of plastic additives to water and biota (Teuten et al. 2007, 2009, Koelmans et al. 2013, Valderrama et al. 2016).

4.3.2.4 Polymer composition and glass transition temperature

The leaching of plastic additives is strongly influenced by the type of polymer and the glass transition temperature (Tg) of a particular polymer material (Sun et al. 2019). Different polymer types have distinct chemical properties, affecting how additives interact within the material. For example, polymers with higher polarity such as polyethylene terephthalate (PET) often retain additives more effectively due to their cohesive nature. In contrast, nonpolar polymers like PE have looser structures, potentially allowing additives to migrate more readily. The leaching of plastic additives is also influenced by the molecular weight of the polymers, where higher molecular weight polymers (e.g., longer polymer chains), exhibit lower leaching rates compared to lower molecular weight counterparts. This is due to the stronger interaction of the chemicals with the polymer and their reduced mobility within longer chains.

The reduction in diffusivity in polymers with increasing *T*g has been understood for decades (Sun et al. 2019). This means that rubbers (low *T*g values) would be expected to exhibit higher additive diffusivity than most thermoplastics (apart from PE and PP), particularly at environmentally relevant temperatures. For example, the diffusion of PAHs and polychlorinated biphenyls (PCBs) through silicone rubbers, which have some of the lowest plastic *T*g values, was higher than in low density PE (LDPE; Rusina et al. 2007, 2010a). A study predicting the leaching/diffusion of different brominated flame retardants in a range of common thermoplastics demonstrated the importance of *T*g on the leaching/diffusion rate (Sun et al. 2019). The diffusion coefficient of a common PBDE flame retardant (BDE-209) was estimated to be lower (i.e., faster release) for polymers with low *T*g values (e.g., LDPE, PP, and silicon rubber) compared to those with high *T*g values, e.g., polyamide (PA), PET, PS, and polymethyl methacrylate (PMMA). The authors also concluded that brominated flame retardants generally exhibited extremely high half-lives that ranged from tens to thousands of years (assuming the polymer structure remained intact for such lengths of time). With the exception of vehicle tire rubber materials, rubber materials have generally been poorly studied in terms of plastic chemical leaching into the natural environment and therefore should be the focus of greater research.

4.3.2.5 **Crystallinity**

The degree of polymer crystallinity significantly impacts the rate and extent at which plastic additives leach from plastic into aqueous matrices (Satoto et al. 1997, Hansen et al. 2013, Sun et al. 2019). Crystalline regions in plastic material comprise densely packed and ordered polymer chains, which can act as effective barriers to impede the migration and leaching of additives (Sun et al. 2019). Conversely, amorphous regions lack the ordered structure found in crystalline polymers and there is a greater distance between the polymer chains, allowing additives to diffuse and move more freely (Hartmann et al. 2017). Consequently, the leaching of plastic additives occurs at a comparatively slower pace in plastics containing a higher degree of crystalline polymer regions (e.g., PE) than those with a higher level of amorphous regions (lower crystallinity, e.g., PVC) due to limited diffusion pathways. It is important to note that most plastic materials do not exhibit uniform crystallinity, but rather regions of higher and lower crystallinity (Yang et al. 2018). As such, most plastic items present in aquatic environments may undergo plastic additive leaching at different rates from different regions of the material.

4.3.2.6 **Extrinsic environmental conditions**

As shown in Table 4.6, there are several extrinsic environmental parameters with the potential to influence the leaching of plastic additives from plastic materials present in the environment. Although many of these (e.g., salinity, turbulence, and UV exposure) are as relevant in an Arctic environment as in any other environment found globally, the temperatures that plastic materials are subjected to in cold environments can impact chemical additives behavior. There remains a paucity of data on the leaching of chemicals from microplastic under environmentally realistic conditions (Do et al. 2022), meaning there are knowledge gaps about the partitioning of additives between plastic and environmentally relevant waters.

Temperature plays a key role in chemical behavior and environmental processes, including the leaching of chemical additives from plastic materials (Nam et al. 2010, Hansen et al. 2013, Ye et al. 2020, Dhavamani et al. 2022). Temperature may well be one of the most important parameters influencing plastic additive leaching, especially in low temperature environments like the Arctic. For example, the leaching of dibutyl phthalate from PVC was investigated at temperatures of 4, 25, and 45 °C, with higher levels of leaching occurring at the highest temperature (Ye et al. 2020). Higher temperatures can facilitate the breakage of chemical bonds in the polymer (lysis) and accelerate the aging process, thus allowing more additive chemical molecules to leach from inside the material. Further, higher temperatures can cause swelling of the polymer and an increase in the free volume inside (Alin and Hakkarainen 2010). This can lead to more of a chemical additive migrating freely to the surrounding water because the additives are present in a larger free volume that provides more opportunities for the molecules to move out of the polymer matrix (Liu et al. 2013). Due to the cold temperatures in the Arctic, there may be less additive leaching; however, as climate change continues to raise environmental temperatures, additive leaching may increase. Thus, understanding leaching behavior under different temperature scenarios is paramount in understanding exposure and effects and how this may impact Arctic biota in the future.

Salinity may play a role in controlling the leaching of organic chemical additives, but different studies have indicated different trends (i.e., increased or decreased leaching with increasing salinity; Rani et al. 2015, Suhrhoff and Scholz-Böttcher 2016, Paluselli et al. 2018, Dhavamani et al. 2022, Henkel et al. 2023). For example, the leaching of DEHP is reduced at higher levels of salinity due to salting out (Henkel et al. 2023). Similarly, the leaching of phthalic acid esters was found to decrease with increasing salinity (Dhavamani et al. 2022). In contrast, additives present in PE (bisphenol A, phthalates, citrates, and Irgafos® 168 phosphate) were found to leach more readily into salt water than into deionized water (Suhrhoff and Scholz-Böttcher 2016). In general, it seems that the influence of salinity is related to the specific chemical properties of an individual additive and does not appear to be generalized for compound groups (Suhrhoff and Scholz-Böttcher 2016, Henkel et al. 2023). Moreover, other processes appear to have a much stronger influence on chemical additive leaching and will most likely dominate any effect that might derive directly from differences in salinity of the surrounding medium. Although salinity levels are generally lower in Arctic marine waters compared to other oceanic basins because of heavy freshwater inputs and low evaporation rates, this could shift because of climate change. Therefore, like temperature effects, it is important to conduct experiments under different salinity scenarios to tease apart leaching behavior differences.

Exposure of plastic materials to natural sunlight (UV irradiation) typically causes embrittlement and, subsequently, increased fragmentation (e.g., Gewert et al. 2015, Da Costa et al. 2018, Sørensen et al. 2021). However, plastic additives can be both active and passive under UV exposure. The Arctic experiences far more sunlight than most locations during the summer months and far less during the winter season; natural sunlight seasonality and variation should be considered in experimental studies. One class of plastic additives, UV stabilizers, is specifically incorporated into the polymer to help prevent or slow down the rate of UV degradation. Such chemicals work by adsorbing UV irradiation and protecting the polymer. The presence of other types of additives (e.g., colorants) has also been shown to reduce degradation rates (Liu et al. 2021), possibly through shielding and/or competition for electrons and reactive oxygen species generated during the UV exposure (Gewert et al. 2018). However, the fragmentation process facilitated by UV exposure can significantly increase the available plastic surface area in contact with the surrounding matrix. As a result, UV degradation actively promotes the leaching of chemical additives (Sait et al. 2021, Sørensen et al. 2021). It is worth noting that UV stabilizer chemical additives themselves can leach directly from plastic materials but are rapidly degraded in the natural environment if they are exposed to sunlight/UV (Suhrhoff and Scholz-Böttcher 2016, Sørensen et al. 2021). Importantly, UV degradation of the polymer material itself also leads to the formation of small molecular degradation products, which can readily leach into the surrounding media. For example, UV degradation of PET results in the formation of ethylene glycol and terephthalic acid degradation products (Hurley and Leggett 2009, Sarno et al. 2021).

There is strong evidence that plastic materials exposed to varying degrees of turbulence (e.g., water flow) in aquatic environments can be subject to increased leaching of additives in multiple polymer types (Suhrhoff and Scholz-Böttcher 2016, Gulizia et al. 2023, Henkel et al. 2023). The effect of the turbulence prevents concentration gradients on the polymer surface during the experiment due to the constant mixing of the water, as well as reducing the thickness of the ABL. As a result, a steady state equilibrium is unable to develop at the polymer-water interface, allowing chemical additives with relatively low molecular weights to continuously diffuse into solution. The degree of increased leaching, however, varies between different additive and polymer types depending on their respective physicochemical properties. For example, Almeda et al. (2023) saw no difference in the composition and concentration of additives in leachates generated under shaking conditions of 1 and 60 rpm. For those compounds that are highly soluble already, turbulence will have a much lower impact than for chemicals that are less soluble (Suhrhoff and Scholz-Böttcher 2016).

4.3.2.7 Chemical additives leaching directly into organisms

Studying the direct uptake of additives from plastic ingested by organisms is difficult, especially in aquatic organisms. In general, exposures need to be conducted in aqueous environments that require the addition of the plastic test materials to the aqueous media where they can be ingested by the test organisms. Such an approach naturally leads to the leaching of chemical additives into the surrounding water, making it challenging to truly determine the mechanisms of additive exposure (i.e., surrounding water or directly from the particle). In many cases, the test plastic particles/materials are spiked with chemical additives, which does not necessarily provide a fair representation of naturally occurring exposures and processes. As a result, it is difficult to infer whether the leaching of plastic additives in the gut of aquatic organisms is an important exposure route (Hermabessiere et al. 2017).

For example, Chua et al. (2014) exposed a marine amphipod (*Allorchestes compressa)* to PBDE in the presence or absence of microbeads with adsorbed PBDEs. Both microbead ingestion and PBDE transfer via the microbeads were demonstrated at the end of the exposure. However, concentrations of PBDEs were lower in amphipods exposed to PBDE adsorbed on microbeads than in amphipods exposed to PBDEs without microbeads suggesting that the transfer of PBDE adsorbed on microplastics can occur, but at a lesser extent than the transfer via water. In a study performed using LDPE pellets incubated in natural seawater for two months, Rochman et al. (2013) observed that subsequent laboratory exposure of Japanese medaka (*Oryzias latipes*) to the pellets resulted in the accumulation of significant amounts of PBDEs that had adsorbed onto the pellets from the surrounding seawater.

There have been a number of attempts at modeling the exposure of aquatic organisms to chemical additives leaching into the digestive tract following the ingestion of plastic items. In general, such approaches have found that this exposure route is most likely to be negligible in many scenarios. In a biodynamic modeling approach, it was shown that the potential leaching of 4-NP and BPA in the intestinal tracts of lugworm (*Arenicola marina*) and North Sea cod (*Gadus morhua*) was possible, but most likely negligible compared to the overall exposure the organisms were likely to experience (Koelmans et al. 2014). The conservative analysis showed that this exposure route might be relevant for lugworms under some specific circumstances. Similarly, Bakir et al. (2016) demonstrated, using a one-compartment model, that microplastics do not provide an additional pathway for the transfer of plastic chemicals, including DEHP and perfluorooctanoic acid (PFOA), from seawater to marine organisms even if plastic transits through the gut of organisms.

Seabirds in the order Procellariiformes, such as fulmars, petrels, and shearwaters, have the highest rates of plastic ingestion by seabirds across the globe (Acampora et al. 2016, Roman et al. 2016, 2019). These birds retain plastic in the stomach along with the oily components of their prey, creating a unique stomach oil that offers a potential route for additive leaching to occur (Kühn et al. 2020). In a recent study using a microplastic mixture produced from collected beach litter, additives including plasticizers, antioxidants, UV-stabilizers, flame retardants, and preservatives were found to variously partition from the plastic material into the oily stomach fluid of Northern Fulmars (Kühn et al. 2020). Collected from beaches, Northern Fulmars that contained > 0.1 g of plastic in their stomachs were found to have higher concentrations of PBDE chemical additives in their livers than "bycatch" Northern Fulmars containing almost no plastic in their stomachs (Neumann et al. 2021). Similarly, a study on Short-tailed Shearwaters (*Puffinus tenuirostris*) showed that plastics ingested by these birds at sea transferred flame retardants (PBDEs) to the birds, including BDE-209, which is specific to plastic (Tanaka et al. 2013). Although not an Arctic species, Tanaka et al. (2020) observed accumulation of UV-328 from ingested plastics in various tissues of Streaked Shearwater (*Calonectries leucomelas*) chicks via feeding experiments (Tanaka et al. 2020), further supporting that ingested plastics may be an important exposure mechanism of harmful plastic additives to exposed organisms. Moreover, Sühring et al. (2022) found higher concentrations of additives in Northern Fulmars and Black-legged Kittiwakes with higher plastic loads; thus, highlighting the importance foraging behavior has on contaminant burdens (Sühring et al. 2022). Combined, these studies provide strong evidence for chemical additive leaching and uptake in the digestive tract of exposed birds.

As stated previously, there is conflicting information available about the viability and relevance of additive leaching in the gut of marine organisms following ingestion. Further studies, especially those that use real plastic materials (i.e., not spiked) at environmentally relevant concentrations, and which develop and employ robust exposure approaches, are needed to gain a more accurate understanding of this potential exposure route for additives. Modeling approaches indicate that direct release of chemical additives in the digestive tract of an organism is likely to be a negligible exposure route for additives , especially within the context of other exposure routes for the same chemicals (e.g., directly from the surrounding water or via food; Koelmans et al. 2014). However, for certain organisms, such as marine worms and seabirds, ingestion of plastic may represent a viable and important exposure and uptake route for plastic additives.

4.3.3 Effects of plastic additives in exposed organisms

Although there are very few studies assessing the effects of plastic additives on Arctic species, their effects on some marine invertebrate and fish species (not specific to the Arctic) have recently been reviewed in the scientific literature (Gunaalan et al. 2020, Pires et al. 2022). Studies have revealed that plastic additives can exert a range of biological impacts on marine organisms. Phthalates, for example, have been linked to endocrine disruption in marine species, affecting reproductive processes and development. Research by Rochman et al. (2014) found that exposure to phthalates led to decreased reproductive success and altered hormone levels in fish. Similarly, BPA, another common plastic additive, has been shown to disrupt endocrine function in marine organisms, potentially impairing reproduction and growth (Teuten et al. 2009).

Furthermore, plastic phthalate congeners and BPA cause significant toxic effects, including endocrine disruption, impairment of reproductive systems, feminization effects, larval developments and impaired behavioral vigilance, and predatory avoidance in invertebrates (Gunaalan et al. 2020, Delaeter et al. 2022). Other additives, such as flame retardants (e.g., OPEs; Alzualde et al. 2018) and tire leachate (Chibwe et al. 2022), have been shown to have developmental effects, such as focal abnormalities, i.e., pin-eyes and structural cranial deformities in fathead minnow (*Pimephales promelas*), which can lead to disrupted foraging efficiency (Wirt et al. 2018). More recently, several studies have reported lipid metabolism alteration in the cell lipidome and generation of reactive oxygen species (ROS) as negative effects of plastic additives exposure to aquatic organisms (Pérez-Albaladejo et al. 2020). Furthermore, toxico-proteomic studies have demonstrated that plastic additives, including phthalate esters, flame retardants, ultraviolet stabilizers, and synthetic phenolic antioxidants, can lead to reproductive toxicity, hepatotoxicity, and neurotoxicity in invertebrates and fish by disrupting detoxification, oxidative stress, hormone modulation, signal transduction, and apoptosis (Liu et al. 2020b). It is becoming increasingly evident that current-use additives can lead to toxic effects in exposed organisms. Indeed, there are large knowledge gaps when it comes to Arctic species exposed to additives via uptake of plastic, and there are very few studies assessing plastic additive effects on Arctic relevant species.

From an Arctic perspective, it is critical to understand the cumulative effects of current-use additives on Arctic biota, especially because the organisms may be co-exposed to relatively high concentrations of POPs and heavy metals as well as experiencing multiple stressors simultaneously as the North continues to undergo rapid environmental change.

4.4 Knowledge gaps and future directions

Our review shows most of the studies addressing current-use additives in Arctic biota only document their occurrence, with very few studies investigating the potential effects on these animals. Indeed, we show that the effects on Arctic wildlife are largely unknown, both for single compounds and mixtures. There is a need for studies investigating important toxicological endpoints, which are required for robust ecological risk assessments (Box 4.1). As a result of this, our targeted research recommendations (Box 4.2) work toward filling these knowledge gaps to better understand how these plastic additives may impact Arctic biota. Furthering our knowledge on these subjects will allow us to improve our understanding of the potential effects on Arctic biota, including cumulative effects, and help inform circumpolar chemical policy and management decisions (Box 4.3).

4.4.1 Recommendations

Plastic additives are an inherently complex mixture; ultimately, they do not fit into a single category or class of chemicals (Hamilton et al. 2022). Long-range transport pathways play a role in introducing current-use additives (of chemicals as well as plastics) to the Arctic, while local sources exist as well

Box 4.1 List of knowledge gaps (in no particular order) on the effects of plastic additives on Arctic biota.

1. **Sources:** The sources of plastic additives need to be better understood and (semi-) quantified/weighted for mitigation actions, including local emissions of plastics vs. long-range transport of plastics.
2. **Long-range transport:** The long-range transport of chemical additives, both non-persistent and persistent, in plastics is not well-studied. Local emission sources of plastic additives in the Arctic need to be better understood. [The topic of local sources vs. long-range transport of chemicals is currently being addressed by AMAP.]
3. **Exposure:** Assessing occurrence and monitoring concentrations of current-use additives across the Arctic is needed to better understand local, regional, and pan-Arctic exposure for wildlife and human populations alike.
4. **Chemical properties**: Identities and quantities of chemical additives in different polymers are not well-known.
5. **Leaching behavior:** The leaching behavior of chemical additives needs further study, both in seawater and in tissues/body fluids of organisms. The role of plastic weathering will be important to consider in this process, as well as the environmental conditions of the Arctic.
6. **Bioaccumulation and transformation:** Species-specific bioaccumulation vs. transformation of plastic additives is not known, including identities of important transformation products.
7. **Effects:** The effects on Arctic wildlife are largely unknown, both for single compounds and mixtures.

(Andrade et al. 2021). Indeed, the literature suggests that plastic particles likely play an important role in the long-distance transport of current-use, environmentally unstable additives. There is a need to co-monitor plastic particles and their plastic additives (e.g., Sühring et al. 2022) by building on monitoring infrastructure that already exists (e.g., POPs monitoring) across the circumpolar Arctic. Occurrence monitoring is important to understand wildlife exposure, and thereby, organismal, population, and ecosystem level effects. We (1) highlight the need to fill the knowledge gaps; (2) call for an Arctic-wide, current-use additive monitoring plan; and (3) describe the need for effects measurements on Arctic species to ultimately inform monitoring priorities, risk assessments, and policy decisions.

4.4.1.1 Coordinated effort to fill current knowledge gaps

Although there are individual studies assessing the occurrence of current-use plastic additives in the Arctic, there is a need to monitor these chemicals in the circumpolar North in a coordinated way because it is critical to understanding wildlife and human exposure. As previously emphasized, the current-use additive manufacturing and use is complex (i.e., there are thousands of chemicals used and produced). Therefore, this diverse group of chemical additives should be more clearly defined, including chemical identities, use categories and

Box 4.2 List of recommendations for future research on the effects of plastic additives on Arctic biota.

1. Planned work in AMAP on CEAC (e.g., update of AMAP 2017) should consider plastic additives. Ideally, experts should liaise with LMEG to ensure relevant inputs on (micro) plastics and associated chemicals. This focus and strategy also applies to planned work in AMAP on temporal trends of POP and CEAC.
2. The diverse substance group of plastic additives should be more clearly defined, including chemical identities, use categories, and quantities and hazard characterization. Ongoing research in this field should be noted and used for planning future work by AMAP LMEG.
3. Ongoing monitoring efforts should consider ways of expanding for screening and/or monitoring of plastic additives. There are multiple pathways: (1) Existing Arctic monitoring programs for POPs and/or heavy metals could incorporate plastic additives. (2) Specimen banks and sample collections can be used for retrospective analyses or screening. (3) (Micro-)plastic monitoring targeting Arctic animals could incorporate a chemical component, for example in fulmars, fish, etc. (4) Community-based approaches can direct monitoring efforts, for example from a food-safety perspective.
4. Suspect/non-target screening approaches are useful techniques in identifying suspected or unknown compounds in a sample. Compound lists of chemical additives should be compiled (see point 2) as a basis for suspect screening efforts. These approaches should also be applied to identify potential metabolites (metabolomics).
5. Besides chemical additives, research and monitoring should consider residual monomers and other residues in the plastic particle that might also be toxicologically relevant.
6. Targeted analytical methods should be further developed to allow accurate, precise, and robust quantification of a wide range of chemical additives (and relevant transformation products), potentially at low concentrations and in complex biotic matrices. These developments must include a strong focus on QA/QC, including interlaboratory comparisons, to ensure comparability across studies, also with a view to future circumpolar AMAP assessments. This also links with point 1 above.
7. Research should be intensified for a better understanding of the environmental fate of chemical additives, including (1) near and distant sources; (2) long-range environmental transport of plastic particles; (3) leaching processes under various conditions, including the weathering of plastics; (4) transformation processes in the environment and in vivo.
8. Effect-based methods should be applied, preferably in concert with chemical analyses, for a better understanding of links between the presence of chemicals and observed effects.

Box 4.3 List of policy implications on the effects of plastic additives on Arctic biota.

1. The transport of chemical additives incorporated in a plastic particle to the Arctic has been recognized as long-range environmental transport in the regulation of UV-328 under the Stockholm Convention. This reasoning will likely apply to many other chemical additives.
2. The regulation of UV-328 should be followed up by monitoring efforts in the Arctic to ensure the effectiveness of the regulation.
3. Plastic additives are receiving attention by several fora (e.g., UNEP) for which Arctic data will be relevant for future risk assessments and potential regulatory actions.
4. Plastic additives (and related compounds such as residual monomers) bridge between plastic and chemical monitoring in the Arctic, requiring more integrative approaches to monitoring and assessments.
5. Risk assessments of the high number of chemicals used as plastic additives are challenging. New approaches, for example via groups of chemicals, could be considered (Fauser et al. 2022). Science can assist in these processes with relevant data, but also new conceptual developments.

quantities, and hazard characterization. Besides chemical additives, research and monitoring should consider residual monomers and other residues in plastic particles that might also be toxicologically relevant.

Targeted and non-targeted analytical methods should be developed further to allow identification and accurate, precise, and robust quantification of a wide range of chemical additives (and relevant transformation products), potentially at low concentrations and in complex biotic matrices. These developments should include a strong focus on robust QA/QC protocols, including interlaboratory comparisons, to ensure comparability across studies, but also with a view to future circumpolar AMAP assessments. Research should be intensified for a better understanding of the environmental fate of chemical additives in the Arctic, including (1) near and distant sources; (2) long-range environmental transport of plastic particles; (3) leaching processes under various conditions, including the weathering of plastics; (4) transformation processes in the environment and in vivo.

Novel approaches to monitoring the occurrence of current-use additives can be used to determine contaminant burdens across species. For example, Zahaby et al. (2021) developed a toxicogenomics approach (ToxChip) to understand the contaminant burden of polycyclic aromatic compounds in seabird populations in the Baffin Bay-Davis Strait region of the Canadian Arctic Archipelago. This ToxChip approach was created to identify/monitor avian populations following oil spills. Zahaby et al. (2021) determined contaminant burdens in the livers of Thick-billed Murres and Black Guillemots (*Cepphus grylle*) and successfully distinguished between the two distinct colonies of seabirds based on their gene expressions known to be associated with contaminant exposure. Novel approaches to effects monitoring in wildlife are important to develop robust and consistent monitoring efforts across the pan-Arctic region.

4.4.1.2 Develop an Arctic-wide additive monitoring plan

To further assess the toxicological effects of plastic additives on Arctic biota, a pan-Arctic additive monitoring plan should be developed. This can be connected with ongoing activities of e.g., POP monitoring in Arctic biota, which has traditionally included screening studies of CEACs and additions of relevant compounds to systematic monitoring efforts. Because there are over 10,000 chemical additives in existence, with more being produced each year (Wiesinger et al. 2021), it is unrealistic to monitor all additives in Arctic biota and their environments. Instead, both chemical additives and species of focus will need to be selected based on current knowledge, need, and feasibility. The recent PLASTCHEM report led by Wagner et al. (2024) can serve as a valuable basis to connect occurrence data in the Arctic with effect data of a broad range of additional additives and non-intentionally added substances and help prioritize effects testing on Arctic species. Non-targeted screening analyses are a useful technique to improve the understanding of the occurrence, semi-quantitative levels, and chemical scenarios of leached compounds. This information, along with targeted analysis, can then be used to propose specific compound groups and species of focus and should include the consideration of species of subsistence importance, which could vary regionally. Importantly, the development of this monitoring plan should be done collaboratively and cross-linked with ongoing initiatives, such as those in the field of POP monitoring.

Ongoing monitoring efforts should consider ways of expanding for screening and/or monitoring plastic additives in a harmonized way through the production of AMAP monitoring guidelines for plastic additives. There are multiple pathways to execute this, as outlined in Hamilton et al. 2022: (1) existing Arctic monitoring programs for POPs or heavy metals could incorporate plastic additives; (2) specimen banks and sample collections can be used for retrospective analyses or screening; (3) (micro-)plastic monitoring targeting Arctic animals could incorporate a chemical component, for example in fulmars, fish, etc.; and (4) community-based approaches can direct monitoring efforts, for example from a food-safety perspective.

4.4.1.3 Effects measurements and risk assessment for Arctic relevant species

Highlighted coordinated, pan-Arctic monitoring efforts will help inform targeted effects measurements and testing. To understand how additives may affect different Arctic species, environmental concentrations need to be monitored. Further, biotic and abiotic concentrations need to be assessed and compared to those reported in dose-response studies to determine whether environmental concentrations are above threshold levels of effects. Toxicogenomic and toxico-proteomic information can help elucidate which mechanisms of toxicities are involved and whether adverse outcome pathways (AOP) may be initiated (Ankley et al. 2010).

Data from RNA-sequencing analysis can either be mapped to a reference genome/transcriptome, or they can be used to assemble a de novo transcriptome (i.e., creation of a transcriptome without a reference genome), which the data are then mapped to (Conesa et al. 2016). Although available annotated transcriptomes for many Arctic species are still sparse (Lenz et al. 2021), de novo assemblies exist for several species in the Arctic, such as copepods (e.g., *C. finmarchicus* and *C. glacialis*; Lenz et al. 2014, Bailey et al. 2017). Increasing the number of reference transcriptomes from Arctic organisms can be a useful way forward to address responses to plastic additives and other environmental stressors. By carefully comparing transcriptomic data from animals in laboratory experiments with data obtained from animals in the field, it is possible to better understand if/how organisms are affected by plastic chemicals in nature (Grøsvik et al. 2023).

Suspect/non-target screening approaches are useful techniques for identifying suspected or unknown compounds in a sample, which can then be isolated and used for toxicity testing. A compound list of chemical additives should be compiled as a basis for suspect screening efforts. These approaches should also be applied to identify potential metabolites (metabolomics) and degradation products, which can be included in effects testing.

Key species of food webs typical for different regions of the Arctic are of interest to test for effect responses in dose-response experiments to establish threshold levels for effects related to concentrations taken up by the organism. Such species could be benthic or pelagic suspension feeding invertebrates (e.g., bivalves and copepods). Organisms with other feeding strategies, such as deposit feeding organisms, would give information on the importance of uptake of sedimented material, while effect studies of selected fishes at different developmental stages would give important information on various species of subsistence and ecological importance.

Although in vitro studies on the effects of plastic additives are important to obtain a baseline understanding of a contaminant, exposure to only one contaminant is not the reality in the environment; there are a multitude of contaminants that may have combined or have cumulative effects on the organisms or ecosystem in which they are present. Monitoring plastic additives in combination with other contaminants of interest can lead to an ecologically relevant view of contaminant exposure to wildlife. Further, while assessing the toxicological effects of plastic additives in different environmental matrices, it is important to consider existing local environmental conditions such as water or sediment/soil composition, light, microorganisms, and weathering forces to understand the effects of plastic additives in environmental conditions. Hereby, the higher sensitivity of Arctic ecosystems, as compared to lower latitudes and multiple stressor effects, must be taken into consideration. Thus, using samples to examine plastic additives as well as other contaminants under varying environmental conditions can help us better understand the cumulative effects of current-use plastic additives at varying levels of biological organization.

4.5 Conclusion

We reviewed the current state of knowledge on the physiological effects of a selected group of current-use plastic additives on Arctic biota. Although various studies address the occurrence of these plastic additives in various Arctic species, few examine the potential physiological effects on these animals. We highlighted a general lack in our understanding of the chemical additives found in plastic pollution, the processes of plastic additives leaching into the Arctic environment and exposed organisms, concentrations of plastic additives in biological tissues, and the link between concentrations and specific effect(s). Moving forward, we recommend future work be focused on: (1) filling the extensive knowledge gaps outlined in this review; (2) developing a pan-Arctic plastic additive monitoring plan in coordination with existing monitoring efforts; and (3) conducting effects measurements and risk assessments for species that are distributed in or are relevant to the Arctic. As climate change impacts and human activities continue to increase in the Arctic and across the globe, thus increasing plastic pollution and chemical additives in the Arctic environment, harmonized monitoring and measuring effects of current-use additives on Arctic biota will be paramount in informing local, regional, and international monitoring priorities, management, and policy decisions.

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5. Conclusions and Outlook

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This technical report on the effects of plastic pollution on biota in the Arctic covered four major themes:

1. Effects of macroplastics in terms of entanglement;
2. Effects of macroplastics in terms of ingestion;
3. Physiological effects from microplastics; and
4. Effects from chemical additives in plastics.

It highlights the many ways in which plastics can affect different organisms, some of which are clearly visible (e.g., large whales entangled in plastic ropes), while others can be difficult to detect without physiological assays and advanced analytical tools (e.g., how gut microbiota vary with levels of plastic ingestion).

A previous AMAP LMEG report summarized and discussed plastic ingestion by animals (AMAP 2021). The report summarized dozens of publications on plastic ingestion by mammals, birds, fish, and invertebrates in the Arctic region. However, much fewer studies were found that specifically addressed effects of plastics on Arctic animals.

In addition to a lack of information at the individual level, negative effects at the population level have hardly been addressed. Chapter 1 focused on entanglement and provided a discussion on how population-level effects' analyses may be achieved. The results of this and the other chapters highlighted that for many species, we simply do not have the data required to assess effects at the population level. Therefore, before entering discussions on specific indicators that could be recommended for measurements of effects, the report recommends a better alignment of studies and data and focusing research and monitoring efforts on data collection for future effect assessments.

Among other suggestions, Chapters 2 and 3 continue to recommend the adoption of harmonized protocols and metrics to be recorded for plastic occurrence and effect observations in organisms. This could include animals collected and dissected in subsistence harvests in the Arctic. The plastic monitoring procedures outlined by AMAP's LMEG could be extended to include additional matrices to study plastic occurrence, such as mammals scat samples, as non-invasive indicators. For birds, susceptibility to microplastic uptake and retention is associated with their feeding behavior and stomach structure, and thus some species are more susceptible to plastic ingestion and retention within the gut than others. The Northern Fulmar (*Fulmarus glacialis*) is well established as an indicator for large microplastics (1–5 mm), however other birds and smaller MP should be considered. For fish, Atlantic cod was the most investigated species but not established as the most meaningful indicator in terms of occurrence or susceptibility.

Effect studies or studies on the translocation from the gastrointestinal tract to tissue are lacking, therefore hampering the selection of indicator species for effects monitoring. To assess effects, studies on occurrence must be paired with physiological metrics or necropsies that can assess physical changes in the individuals.

Although an increasing number of peer-reviewed papers report on the occurrence of chemical additives in plastic, no information was available on the effects of current-use plastic additives on Arctic species. Effects of harmful chemicals on Arctic wildlife have been assessed by AMAP previously (AMAP 2018), showing exposure of some species to toxic chemicals levels that can cause risks. Some of these chemicals were previously used as plastic additives but are banned today because they meet the criteria of persistent organic pollutants (POPs). Current-use plastic chemical additives cover a wide range of chemical identities with different physical-chemical properties that fulfil different functions in plastic polymers, creating complex challenges for measurements and risk assessment. In addition, they add to, and may aggravate the effects of multiple stressors, such as climate change, habitat loss, or infections. Therefore, beyond the analytical quantification of plastic additives in biota, further effect-based assays are needed to gain knowledge. This is true for Arctic species, but also globally. In addition, mechanisms of exposure to plastic additives contain knowledge gaps. Because plastic additives can leach out of the polymer into the surrounding environment, exposure sources exist in the environment (water, prey species) in addition to the potential uptake of plastic particles and the leaching of chemicals in the gut of an organism. These processes will also depend on the physical-chemical properties of the chemicals and might vary with different degrees of weathering of plastic particles.

This technical gap analysis provides an overall picture of the state of information on biological effects from plastic pollution in the Arctic. Its intention was to work toward the development of indicators for plastic-related effects on Arctic organisms. However, given the substantial gaps in the current knowledge, this seems premature. Instead, recommendations include:

- harmonized and interoperable databases for entanglement and ingestion
- community-based studies taking advantage of animals available from subsistence hunts
- controlled effect studies with different types of microplastics at environmentally relevant concentrations
- improved understanding of the leaching of chemical additives and resulting exposure of Arctic animals

The Intergovernmental Negotiating Committee (INC) represents the international community in a global effort toward combating plastic pollution through a legally binding agreement. A global instrument should also contribute to reductions in plastic waste levels in the Arctic, considering both local and distant emission sources of plastic to the environment. The indicators that AMAP's LMEG proposed for plastic monitoring in the Arctic can help to follow changes in environmental plastic levels over time. Expanding them to include effect indicators would complement the indicator pool. However, more knowledge is needed for the development of suitable indicators.

Table 5.1 Highest priority gaps toward understanding the risk of plastic pollution in the Arctic (from each section).

Negative effect from plastic pollution	Description of most important research gaps to be filled toward understanding this effect in the Arctic	Recommendations for future research
Entanglement	Systematic mapping of entanglement. Effects at the population level.	Increased work to combine and work with existing databases that report entanglement in plastic pollution to assess effects across the range of species.
Physical effects on animals from ingested plastics	Extent and severity of the problem is not known for the Arctic.	Extend data collection on plastic ingestion necropsies to include collecting information on metrics that assess the impacts from plastics (e.g., abrasions, lesions, etc.)
Effects on hormone and immune systems, oxidative stress, lipid storage and metabolism, neuro- and reproductive functions, feeding and motion behavior, malformation, growth, and mortality	1. Effects of those concentrations and combinations of MNP found in the environment on a) the physiology of the organisms b) the ecosystem. 2. Effects of multiple stressors, i.e., temperature, salinity and pH change, changed prey and predators, co-exposure to additives, and other pollutants? Synergistic, additive, or antagonistic?	Laboratory-based long-term exposure studies to realistic size, type, and environmental concentrations. Development of relevant and easy-to-use combinations of exposure and effect biomarkers that can be applied in the field in a coordinated way. Combined investigation of MNP occurrence and physiological markers in the wild. Multiple stressors in interdisciplinary studies. Example: 1) Data on additives or other contaminants in addition to MNP. 2) Effect studies from co-exposure situations. 3) Data on changes related to climate change.
Adverse effects from additive chemicals in plastic	No studies available on Arctic animals regarding current-use plastic additives.	Mechanisms of leaching and exposure. Types of adverse effects for different categories of species.

5.1 References

Arctic Monitoring and Assessment Programme (AMAP). 2018. AMAP assessment 2018: biological effects of contaminants on Arctic wildlife and fish. AMAP, Oslo, Norway. https://www.amap.no/documents/doc/amap-assessment-2018-biological-effects-of-contaminants-on-arctic-wildlife-and-fish/1663

Arctic Monitoring and Assessment Programme (AMAP). 2021. Arctic Monitoring and Assessment Programme (AMAP). 2021. Litter and microplastics monitoring guidelines. Version 1.0. Arctic Monitoring and Assessment Programme, Tromsø, Norway. https://www.amap.no/documents/download/6761/inline

Abbreviations and Acronyms

4-NP = nonylphenol, an antioxidant

20β-hsd = 20β-hydroxysteroid dehydrogenase

ABL = plastic-water boundary layer

ABLD = external diffusion = aqueous boundary layer diffusion

ADOC = anthropogenic dissolved organic carbon

ALDFG = abandoned, lost, or otherwise discarded fishing gear

AMAP = Arctic Monitoring and Assessment Programme

AOP = adverse outcome pathway

BDE-209 = decabromodiphenyl ether

BFR = brominated flame retardants

BPA = bisphenol a

BHT = 2,6-di-tert-butyl-4-methylphenol (an antioxidant)

BYCELS = Working Group on By-catch, Entanglements and Live Strandings (under NAMMCO)

BZT = benzotriazoles

CAFF = Conservation of Arctic Flora and Fauna

CEAC = chemicals of emerging Arctic concern

Cl-OPE = chlorinated organophosphate esters

COASST = Coastal Observation and Seabird Survey Team

CWHC = Canadian Wildlife Health Cooperative

DBMP = 2,6-di-tert-butyl-4-methylphenol

DEHP = di(2-ethylhexyl) phthalate

DPTE = 2,4,6-tribromophenyl-2,3-dibromopropyl ether

d.w. = dry weight

EC50 = median effective concentrations

ECHA = European Chemicals Agency

EFSA = European Food Safety Authority

EHDPP = 2-ethylhexyl-diphenylphosphate

EMODnet = European Marine Observation and Data Network

EU = European Union

FARE = Food Access Raises Everyone

FO = frequency of occurrence

FTIR = Fourier transform infrared spectroscopy

GES = good environmental status

GIT = gastrointestinal tract

GR = glucocorticoid receptor

H2O2 = hydrogen peroxide

HELCOM = The Baltic Environment Protection Commission

HBB = hexabromobiphenyl

HBCD = hexabromocyclododecane

HFR = halogenated flame retardants

HOC = hydrophobic organic compounds

HSD = hydroxysteroid dehydrogenase

IAS = intentionally added substances

ICES = International Council for the Exploration of the Sea

IMR = Injury and Mortality Reporting portal of the U.S Fish and Wildlife Service

INC = Intergovernmental Negotiating Committee

IPD = intraparticle diffusion

IPTP(P) = isopropylated triphenyl phosphate

KOH = potassium hydroxide

Kow = octanol-water partition coefficient

LDPE = low density polyethylene

LMEG = Litter and Microplastics Expert Group

LOD = level of detection

LOEC = lowest observable effect concentrations

L-O-PTIR = laser optical photothermal infrared spectroscopy

µ-FTIR = micro Fourier Transform Infrared Spectroscopy

MLDB = European Marine Litter Database

MNPs = micro- and nanoplastics

MPs = microplastics

MSFD = European Union Marine Strategy Framework Directive

N (n) = number

ng/l = nanograms/liter

NAMMCO = North Atlantic Marine Mammal Commission

NGO = non-governmental organization

NOAA = National Oceanic and Atmospheric Administration

NPs = nanoplastics (1 nm to 1 µm)

OPE = organophosphate esters

OSPAR = Oslo/Paris Convention for the Protection of the Marine Environment of the Northeast Atlantic

PA = polyamide

PAH = polycyclic aromatic hydrocarbons

PAME = Protection of the Arctic Marine Environment

PBDE = polybrominated diphenyl ethers

PBT = persistent bioaccumulative toxic chemicals

PC = polycarbonate

PC/ABS = thermoplastic alloy (or blend) of polycarbonate (PC) and acrylonitrile-butadiene-styrene (ABS)

PCB = polychlorinated biphenyls

PE = polyethylene

PET = polyethylene terephthalate

PFAS = per- and polyfluoroalkyl substances

PFOA = perfluorooctanoic acid

PMBT = persistent, bioaccumulative, mobile, and/or toxic

PMMA = polymethyl methacrylate

PNEC = predicted no effect concentrations

POP = persistent organic pollutants

PP = polypropylene

PPARG = peroxisome proliferator-activated receptor gamma

PS = polystyrene

PVC = polyvinyl chloride

py-GC/MS = Pyrolysis gas chromatography – mass spectrometry

QA/QC = quality assurance/quality control

Raman = a light scattering technique

RNA = ribonucleic acid

ROS = reactive oxygen species

RPM = rotations per minute

SCCP = short-chain chlorinated paraffins

SDPA = substituted diphenylamine antioxidants

SWOT = strengths, weaknesses, opportunities, threats

TBBPA = 2,4,6-tribromophenyl-2,3-dibromopropyl ether/tetrabromobisphenol

TCEP = tris(2-chloroethyl) phosphate

Tg = glass transition temperature

THRB = thyroid hormone receptor beta

TIDES = Trash Information and Data for Education and Solutions

TNBP = tri-n-butyl phosphate

ToxChip = toxicogenomics approach

TPP = triphenyl phosphate

TPPO = triphenylphosphine oxide

TV = threshold value

TWP = tire wear particle

UN = United Nations

UNEP = United Nations Environment Programme

UNEP/MAP = United Nations Environment Programme Mediterranean Action Plan

UV-328 = 2-(2H-benzotriazol-2-yl)-4,6-di-tert-pentylphenol

UV = ultraviolet

UV-BZT = ultraviolet-benzotriazoles

Vtg1 = vitellogenin1

WHIP = Wildlife Health Intelligence Platform

Arctic Monitoring and Assessment Programme

The Arctic Monitoring and Assessment Programme (AMAP) was established in June 1991 by the eight Arctic countries (Canada, Denmark, Finland, Iceland, Norway, Russia, Sweden and the United States) to implement parts of the Arctic Environmental Protection Strategy (AEPS). AMAP is now one of six working groups of the Arctic Council, members of which include the eight Arctic countries, the six Arctic Council Permanent Participants (Indigenous Peoples' organizations), together with observing countries and organizations.

AMAP's objective is to provide 'reliable and sufficient information on the status of, and threats to, the Arctic environment, and to provide scientific advice on actions to be taken in order to support Arctic governments in their efforts to take remedial and preventive actions to reduce adverse effects of contaminants and climate change'.

AMAP produces, at regular intervals, assessment reports that address a range of Arctic pollution and climate change issues, including effects on health of Arctic human populations. These are presented to Arctic Council Ministers in 'State of the Arctic Environment' reports that form a basis for necessary steps to be taken to protect the Arctic and its inhabitants.

This report has been subject to a formal and comprehensive peer review process. The results and any views expressed in this series are the responsibility of those scientists and experts engaged in the preparation of the reports.

The AMAP Secretariat is located in Tromsø, Norway. For further information regarding AMAP or ordering of reports, please contact the AMAP Secretariat (The Fram Centre, P.O. Box 6606 Stakkevollan, N-9296 Tromsø, Norway) or visit the AMAP website at www.amap.no.

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