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Current efforts on microplastic monitoring in Arctic fish and how to proceed

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Abstract

In this review, we investigated published data on the occurrence of microplastic in Arctic fish, and the suitability of the data and species for risk assessment and monitoring. As of 11 November 2021, we found nine studies in the peer-reviewed literature, one thesis and one report, confirming the occurrence of microplastic in fishes from multiple Arctic regions. The studies varied in methodology, detection, and quantification limitations, reported categories of size, shape, and chemical identity. All these factors influence the numbers of microplastic reported, thus limiting comparability and hindering integrative analysis. The physiological impacts of the reported microplastic contamination cannot be determined, as all studies targeted stomach/intestine contents and did not use methods with limits of detection low enough to determine particle translocation from the intestine to other organs, tissues, or body fluids within the fish. Furthermore, there is a fundamental lack of understanding the transfer and the effects of plastic additives to Arctic fishes. In addition to discussing methodological challenges and knowledge gaps, we consider ecosystem needs, commercial interests, Indigenous people's subsistence, food safety and food sovereignty concerns, and developed a framework to harmonize and facilitate pan-Arctic microplastic monitoring.

Key words: microplastic, Arctic, fish, monitoring

1. Introduction

Interest in plastic pollution, a contaminant of emerging Arctic concern, is not new (AMAP 2017; Halsband and Herzke 2019). Plastic is now found ubiquitously throughout the Arctic Ocean, from the surface to the seafloor (Lusher et al. 2015; Buhl-Mortensen and Buhl-Mortensen 2018; Grøsvik et al. 2018; Kanhai et al. 2020; Rist et al. 2020). Plastic material enters the Arctic through long-range transport and local sources (Halsband and Herzke 2019; Liboiron et al. 2021; Huserbråten et al. 2022), where the latter may dominate according to recent models (Strand et al. 2021). Plastic pollution (macroplastic > 5 mm > microplastic (MP)) has been identified in a variety of organisms spanning multiple trophic levels including marine mammals (Moore et al. 2020), seabirds (Baak et al. 2020), fish (Kühn et al. 2018; Moore et al. 2022), and invertebrates (Bellasi et al. 2020). The presence of plastic and MP within Arctic water bodies (Martin et al. In press) raises concerns for the exposure of Arctic fishes to this environmental contaminant, and the effects this might have.

Published field studies covering the Arctic ecosystem and adjacent areas (Nielsen et al. 2014; Bråte et al. 2016 ; Liboiron et al. 2016; Kühn et al. 2018; Morgana et al. 2018; Liboiron et al. 2019; de Vries et al. 2020; Granberg et al. 2020) and from other areas demonstrate uptake of MP in fish (Savoca et al. 2021), including fish widely consumed by humans (Danopoulos et al. 2020; Thiele et al. 2021). The prevalence of MP in wild fishes combined with toxicity data from exposure studies (Kögel et al. 2019; Wang et al. 2020; Gomes et al. 2021) suggests a hazard to ecosystems and consumers. However, the risk is not yet quantified. Effect quantification has mostly been performed using round plastic beads, but there are studies that suggest that some of the post-consumer plastic break-down products—not round but diverse in shape and size, and containing various chemical constituents—may have stronger negative effects (Rochman et al. 2014; Peda et al. 2016; Bucci et al. 2021), and therefore need to be evaluated specifically. Furthermore, there is increasing evidence that smaller MP (<10 μm) may cause more negative effects compared to larger MP (Kögel et al. 2019), likely related to size-dependent uptake and translocation barriers (Jeong et al. 2016; Critchell and Hoogenboom 2018; Lehtiemi et al. 2018; Gomiero et al. 2020). MP quan-

tification in field collected organisms is also suffering from methodological limitations. MP sizes that are used in exposure experiments are often below detection limitations in non-laboratory animals, as the MP analytes are not prelabeled with detectable substances such as fluorescent substances or metals. This so far prevents integrated analysis of the two research fields—experimentally found effects of MP cannot be linked to occurrence data as those are missing for the smaller size classes. Also, chemical additives in plastic are an inherent additional hazard linked to MP, lacking sufficient investigation (Espinosa Ruiz et al. 2016; Kwon et al. 2017; Campanale et al. 2020; Fred-Ahmadu et al. 2020).

Fish, used as indicators of ecosystem health (European Parliament 2000), form an important link between trophic levels within Arctic food webs, including humans as top consumers. Fish constitute a significant protein source for human nutrition and are an important cultural food base for Arctic peoples (Ford 2009). Adverse effects of any contaminant affecting fish will therefore have an impact on food safety, security, and sovereignty (Barboza et al. 2018; Smith et al. 2018; Dietz et al. 2019; De-la-Torre 2020; Kögel 2020; Prata et al. 2020). With MP contamination of fish, the achievement of the UN Sustainable Development Goals 2 (SDG2) (zero hunger), SDG3 (good health and well-being), SDG10 (reduced inequality), and SDG14 (life below water) are at stake. Thus, MP contamination is of concern to food safety authorities, regulatory bodies, environmental agencies, food security stakeholders, such as UN organizations, and rightsholders, such as Arctic Indigenous peoples.

The specifics of MP ingested by Arctic fish need to be quantified to enable mitigation of this emerging threat. To this end, we reviewed studies analyzing MP in Arctic fish. We further discussed the methodological pitfalls, constraints, and knowledge gaps, concluding with suggestions on how to move forward with a harmonised approach to facilitate monitoring at local, regional, and pan-Arctic scales.

2. Materials and methods

In this literature study we aimed to gather all available data on MP contamination of fish within the Arctic region as defined by the Arctic Monitoring and Assessment Programme (AMAP), depicted by a gray line in Fig. 1. This definition follows landmarks, instead of a mathematical circle at a certain latitude as the Arctic circle does (currently 66°33′48.9 N). Web of Science (topic) and PubMed (all text) were searched for the keywords microplastic* AND fish* AND (Arctic OR Barent* OR Kara*), last checked on 11 November 2021. References within references, personal contacts, and Google were also investigated. Inclusion criteria were investigations focused on MP occurrence analysis in fish within the Arctic region. Predefined exclusion criteria were laboratory exposure studies, modelling studies, and studies investigating only plastic additives or co-contaminants. For each included study, several variables were extracted and compared including sampling techniques (collection of samples), species examined, region of sampling, extraction of MP, end-point analysis method, and limitations of detection. The latter was not always reported and therefore refers to the lower size threshold of MP reported, or deducted from applied filter sizes, identification method or instrument limitations. If the lowest size reported was larger than the method's restrictions, we commented "reported" (Bråte et al. 2016 ; Table 1). We compared and discussed the studies, but did not evaluate them statistically, as the qualitative and quantitative level of the studies were not sufficient for statistical meta-analysis. We discussed limitations of this collective set of studies, highlighted knowledge gaps, and recommended guidelines intended as a foundation of future monitoring and assessment of MP pollution in the Arctic.

3. Microplastic in Arctic fish

3.1 Investigated species

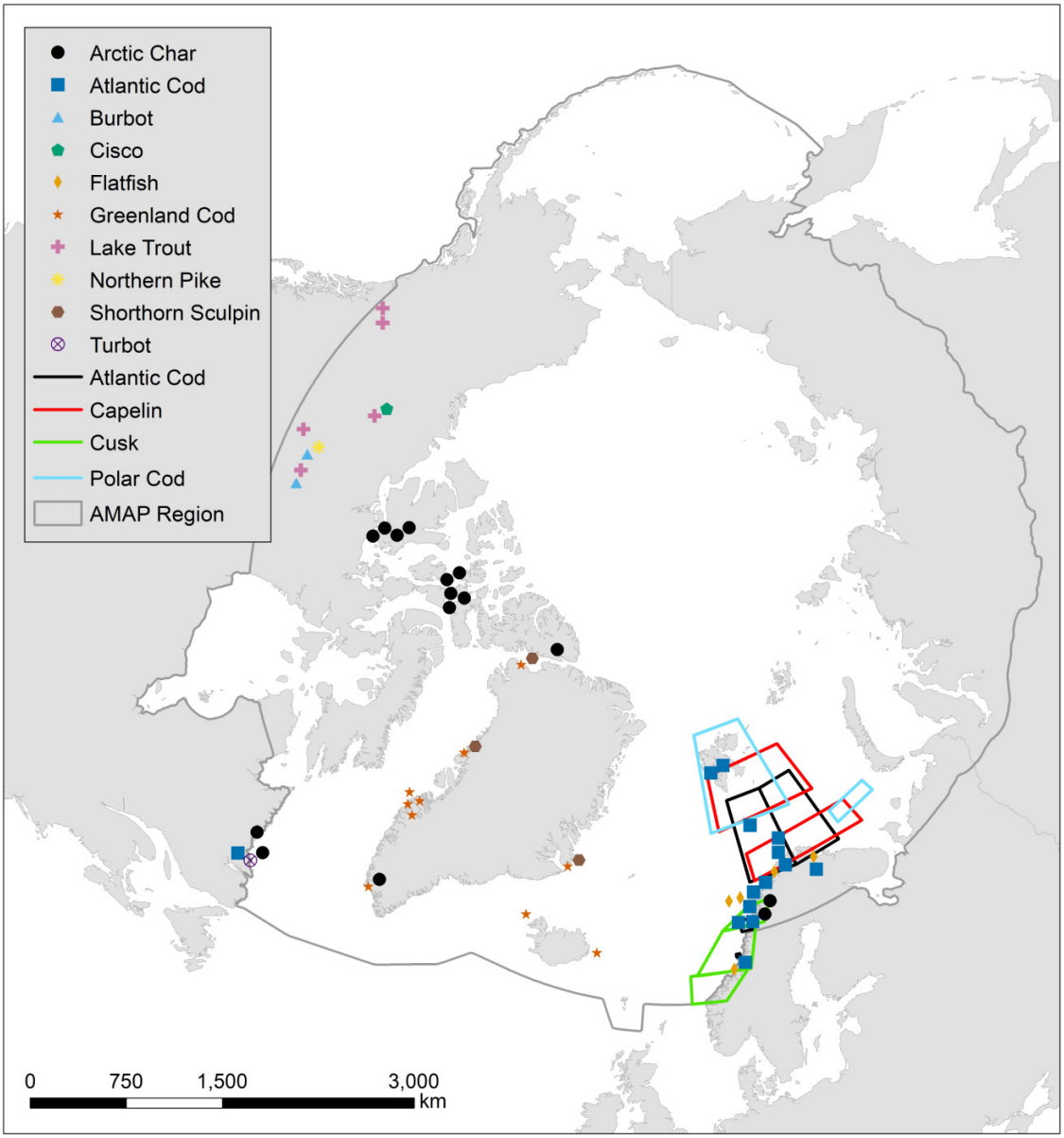
Our literature review uncovered 11 published studies including one thesis and one report that have investigated the ingestion of MP in Arctic fish (Table 1). The studies included a total of 13 species and 2567 individual fish. MP were reported in Arctic/polar cod (*Boreogadus saida*; Kühn et al. 2018; Morgana et al. 2018; Moore et al. 2022), Atlantic cod (*Gadus morhua;*Bråte et al. 2016 ; Liboiron et al. 2016; de Vries et al. 2020; Saturno et al. 2020), Greenland cod (*Gadus ogac*; Granberg et al. 2020), sculpin (*Triglops nybelini*; Morgana et al. 2018), four-horn sculpin (*Myxocephalus quadricornis*; Moore et al. 2022), saithe/pollock (*Pollachius virens*; de Vries et al. 2020), Atlantic salmon (*Salmo salar*; Liboiron et al. 2019), capelin (*Mallotus villosus*; Liboiron et al. 2019; Moore et al. 2022), Greenland shark (*Somniosus microcephalus*; Nielsen et al. 2014), blue whiting (*Micromesistius poutassou*; Malinen 2021), Atlantic mackerel (*Scomber scombrus*; Malinen 2021), Saffron cod (*Eleginus gracilis*; Moore et al. 2022), Arctic cisco (*Coregonus autumnalis*; Moore et al. 2022), and recently, Arctic char (*Salvelinus alpinus*; Hamilton et al. unpublished data, not included in the number summaries in this review). In addition to the scarcity of data on MP occurrence in Arctic fish, the methods applied in the studies varied, compromising comparability among the different studies.

3.2 Microplastic levels and targeted fish matrices

MP contamination levels in fish were often reported as (a) frequency of occurrence (FO) in %, that is, reporting the number of contaminated individuals in the sampled population and (or) (b) the number of MP per fish, ranging from 0 to 12 MP per individual with FO from 0% to 100% (Table 1). Only a single study found more than 2 MP per fish, on average 12 MP per Greenland cod (Granberg et al. 2020). This was also the only Arctic study that reported a FO of 100 % in fish. Published studies from regions outside the Arctic also range over the whole possible FO spectrum from 0% to 100 % (reviewed in Liboiron et al. 2016).

While six studies on Arctic fish only analyzed MP content inside the stomach and intestine, hereafter, called the gastrointestinal tract (GIT; Nielsen et al. 2014; Bråte et al. 2016 ; Liboiron et al. 2016; Kühn et al. 2018; Liboiron et al. 2019; Saturno et al. 2020), five studies assessed the GIT content, including the surrounding tissue of the GIT (Morgana et al.

Fig. 1. Region of interest, which the text referred to as "Arctic" within the circumpolar area. Existing regular fish sampling for other monitoring purposes such as contaminants or fish population monitoring. Repetition interval of sampling is between annually and every third year. Map depicting regular ongoing sampling according to species sampled; see box within figure for symbol coding. As published in the AMAP monitoring guidelines (AMAP et al. 2021). AMAP, Arctic Monitoring and Assessment Programme.



2018; de Vries et al. 2020; Granberg et al. 2020; Malinen 2021; Moore et al. 2022; Table 1), which might play a role for the analysis results. Kühn et al. 2018, who analyzed GIT content only, described a low incidence of MP in Arctic/polar cod compared to Morgana et al. 2018, who included the gut walls, even though Kühn et al. 2018 reported a lower detection limit. No published studies of MP in Arctic fish have analyzed other matrices/tissues than GIT and GIT content, such as liver or muscle/fillet yet.

3.3 Microplastic size detection thresholds

The observation in a global data set that the FO % increases as detection size decreases (Savoca et al. 2021) points towards a relation of the MP size with the accumulation potential. Even though the Arctic data set on MP in fish is too small for a valid meta study, we find the same general pattern there. In Arctic Atlantic cod, four studies had detection thresholds above 1 mm and those had FOs below 2.4 %, while the one study with the lower detection limit of 80 μm found a higher FO of 20.5 % (Table 1). The one study that found an FO of 100 % in Greenland cod (Granberg et al. 2020), had an even lower detection-size limit of 20 μm, as had another study with FO of 7%–43% depending on the fish species (Moore et al. 2022). The latter study filtered through a 20 μm mesh size, but only investigated MP that could be handled by tweezers. For the three studies of Arctic/polar cod, each had a different disad-

Table 1. Overview of available analysis data of MPs in Arctic fish.

Location	Species	Fish (N)	FO (%)	MP per individual (N)	Recovery analysis	Methodology with lower detection limit	Reference
Eurasian Basin, Svalbard, Norway	Arctic/polar cod (*Boreogadus saida*)	72	2.8	0–1	no	Stomach content, visual inspection, suspected MP by FTIR, fibres not included, >35 μm	(Kühn et al. 2018)
North-eastern Greenland	Arctic/polar cod	85	18	1–2	no	GIT and content alkaline digested, visual inspection,	(Morgana et al. 2018)
Northern Greenland	Sculpin (*Triglops nybelini*)	71	34	0–1		>700 μm by FTIR	
Newfoundland, Canada	Atlantic cod (*Gadus morhua*)	205	2.4	0–2	no	GIT content, visual sorting, >1 mm	(Liboiron et al. 2016)
Newfoundland, Canada	Atlantic cod	216	1.4	0–1	no	GIT content, visual sorting, suspected MP by Raman, >1 mm	(Saturno et al. 2020)
Varangerfjord and Lofoten,	Atlantic cod	58	0	n/a	no	Stomach content, visual inspection,	(Bråte et al. 2016)
Northern Norway		56				suspected MP by FTIR, >3.2 mm reported	
Newfoundland, Canada	Atlantic cod	1010	1.68	0–2	no	GIT content, visual sorting, >1 mm	(Liboiron et al. 2019)
	Atlantic salmon (*Salmo salar*)	69	0				
	Capelin (*Mallotus villosus*)	350	0				
Iceland	Atlantic cod	39	20.5	0.23	no	GIT and content alkaline digested, visual inspection → FTIR, >80 μm	(de Vries et al. 2020)
	Saithe/pollock (*Pollachius virens*)	46	17.4	0.28 overall average			
Western Greenland	Greenland cod (*Gadus ogac*)	9	100	12 ± 6	no	GIT and content, enzymatic digestion, visual and FTIR on selected particles, >20 μm	(Granberg et al. 2020), report
East, West, Southwest Greenland	Greenland shark *(Somniosus microcephalus)*	30	3.33	0–1	no	Stomach content, visual examination, >1 mm	(Nielsen et al. 2014)
Iceland	Atlantic mackerel (*Scomber scombrus*)	50	12	1.3	no	GIT and content, alkaline digestion, filtration, visual examination → Raman, >2.7 μm filtration, visually detected particles chemically identified	(Malinen 2021), thesis
	Blue whiting (*Micromesistius poutassou*)	40	6	1			
Beaufort Sea	Polar/Arctic cod	20	15	1 ± 0		GIT and content alkaline digested, microscope aided visual inspection, 20 μm filtration, suspected MP that could be handled with tweezers were analyzed by FTIR	(Moore et al. 2022)

Table 1. Continued

Location	Species	Fish (N)	FO (%)	MP per individual (N)	Recovery analysis	Methodology with lower detection limit	Reference
	Saffron cod *(Eleginus gracilis)*	35	34	1.92 ± 1.19			
		26	19	1.2 ± 0.4			
	Arctic cisco *(Coregonus autumnalis)*	28	7	2 ± 0			
		7	43	1 ± 0			
	Capelin *(Mallotus villosus)* Four-horn sculpin *(Myxocephalus quadricornis)*			0.37 ± 0.16 overall average			

Note: FTIR, Fourier-transform infrared spectroscopy; GIT, gastrointestinal tract; MPs, microplastics.

vantage (Table 1). Moore et al. 2022 who only investigated MPs that could be handled by tweezers despite using a small filtration pore size, found a FO of 15%, similar to the 18% of Morgana et al. 2018, with a 700 μm limit. Both had included the GIT wall. Kühn et al. 2018, with a 35 μm limit, had not included the gut wall and found an FO of only 2.8%. Furthermore, these studies were conducted in different regions and the handling of data differed, as discussed in the end of the following paragraph. As a result, the different observations seen here likely reflect a mixture of methodological differences in detection capacity and data handling, and area or species differences.

3.4 Analysis methods, microplastic identification, and categorization

For quantification of MP in Arctic fish, several approaches were used. Some studies relied on visual identification, with or without a microscope, others added chemical identification by Fourier-transform infrared spectroscopy (FTIR) or Raman spectroscopy. The chemical identification approaches were only applied on suspected particles after visual inspection (Table 1), not on all extracted particles, leaving room for missing particles in the visual inspection step. Amongst those publications providing further information on the reported MP, four studies only found up to five plastic items in total (Nielsen et al. 2014; Liboiron et al. 2016; Kühn et al. 2018; Saturno et al. 2020)—not enough for statistical analysis. One study did not differentiate between Arctic and other regions for color, shape, and polymer type distributions (Morgana et al. 2018) and therefore, these details could not be assessed separately for the Arctic.

According to the shape, two studies found more fragments than fibres in Atlantic salmon and Greenland cod (Liboiron et al. 2019; Granberg et al. 2020), while four found more fibres than fragments in saffron cod, sculpin, four-horn sculpin, saithe, Arctic cisco, capelin and blue whiting (Morgana et al. 2018; de Vries et al. 2020; Malinen 2021; Moore et al. 2022). Atlantic mackerel had a 50/50 distribution between the shape types (Malinen 2021). Polar/Arctic cod contained more fragments (Moore et al. 2022) and more fibres (Morgana et al. 2018), respectively, depending on the study. Moore et al. 2022 probably had a lower detection threshold, as Morgana et al. did not include particles <700 μm. Only one publication considered film an additional category (Liboiron et al. 2019). One study additionally differentiated between fibers and filaments (Granberg et al. 2020). Since not all studies reported on the same shape types, comparison was hampered. Therefore, conclusions on MP numbers per species can only be drawn within, not between studies, due to method differences.

All studies reporting color found high contents of blue, 50% (Atlantic mackerel; Malinen 2021), 49% (Morgana et al. 2018), 34%–38% (de Vries et al. 2020), and 16 % (Liboiron et al. 2019). In two of the studies this was followed by green with 21% (Liboiron et al. 2019) and 33% (de Vries et al. 2020). The latter study categorized an additional 23 % of the MP as black, similar to Atlantic mackerel in Malinen (2021) with 37% black, followed by transparent, while blue whiting contained black, red and green particles with equally shares but no blue. Granberg et al. (2020) found black, blue, red, grey, purple, green, brown, and transparent in this order and did not analyze white or transparant, due to high loads in the controls. In contrast, Liboiron et al. (2019) had reported white MP as the most abundant color. In summary, of the ingested MP particle colors, blue, green, and black were predominant, while also red, transparent, white, grey, purple, and brown were found. The studies reported on different colors, such as that white was excluded in one study, while most abundant in another. Therefore, the color analysis in Arctic fish was not conclusive.

Five studies analyzed more than three particles for chemical identity, of which three found polyester types to be the dominating polymer type (Bråte et al. 2016 ; Morgana et al. 2018; Moore et al. 2022). In addition to polyesters (including PET), nylons (including PA), oleofins (including PE and PP) and acrylics (including paint and PBMA) dominated (Table 2). However, method bias cannot be exculded (see Primpke et al. In press) and the total number of analyzed MP was low. There might be a tendency for benthic fish such as Atlantic and Greenland cod to ingest more heavy polymertypes, such as polyesters and rubber (Table 2). Other than this speculative notion, there was no pattern on polymer types emerging from the available studies. None of the reviewed studies applied chemical analysis to all of the isolated MP, or to all of the particles of the suspected density for plastic.

Only four studies analyzed MP <100 μm in Arctic fish, with a minimum of 20 μm. It is important to be aware of the details in the reporting, such as that one study had a very low filter size of 2.7 μm, and used Raman spectroscopy, but since only microscopically pre-identified plastic particles were subjected to chemical analysis, and no recovery analysis has been performed, the actual detection limit remains unclear and no particles below 100 μm size were described in this study (Malinen 2021).

As a side observation, there were no studies on nanoplastic analysis or the occurrence or effects of chemical additives in Arctic fish species in our returned search results. Information on uncertainty and recovery analysis of the applied methods are generally often lacking in the publications from this research field, and entirely in the data set available for MP in Arctic fish (Table 1). Contamination backgrounds are reported in some of the studies. However, handling background contamination can introduce bias, too. For example, Kühn et al. 2018 did not include microfibres in their analyses to avoid false positives through airborne contamination. Although controlling for false positives is important, the study may have excluded real MP particles from their account.

3.5 Geographical and biological differences

With harmonized data, environmental differences can be compared across studies. In studies on Arctic fish, such comparisons have so far only been achieved within studies. In the three studies with polar/Arctic cod from different locations (Kühn et al. 2018; Morgana et al. 2018; Moore et al. 2022), the data could not be compared because of the reasons discussed previously in this article. Geographical differences were observed within some studies. Bråte et al. 2016

Table 2. Identified polymer types in Arctic fish (Bråte et al. 2016 ; Morgana et al. 2018; Granberg et al. 2020; Malinen 2021; Moore et al. 2022).

(Morgana et al., 2018) MP: N=30 Sculpin and Arctic/polar cod	(Granberg et al., 2020) MP: N=12 Greenland cod	(Malinen, 2021) MP: N=10 Atlantic mackerel and blue whiting	(Brate et al., 2016) MP: N=16 Atlantic Cod	(Moore et al., 2022) MP: N=39 Arctic/polar cod, Saffron cod, Arctic cisco, Capelin and four-horn sculpin
PET/Polyester (N=10)	PA (N=4)	Unidentified (N=5)	PCT (N=6)	Polyester (N=21)
Acrylic (N=7)	Rubber (N=4)	PE, PP (N=4)	PP (N=2)	Nylon (N=7)
Nylon/PA (N=6)	PET/Polyester (N=1)		PVC (N=2)	Acrylic (N=5)
PE (N=5)	Alkyd resin (N=1)	PP (N=1)	PS (N=1)	PE (N=2)
EVA (N=2)	PP (N=1)		Teflon (N=1)	PU (N=2)
	Paint (N=1)		Nylon/PA (N=1)	POM (N=1)
	Unidentified (N=1)		PE (N=1)	PP (N=1)
			SAN (N=1)	
			PBMA (N=1)	

observed MP in Atlantic cod from the harbour of the second largest city of Norway, Bergen, but not from northern Norway or in the vicinity of Norway's capital, Oslo. Greenland cod contained the highest number of MP closer to local pollution sources (Granberg et al. 2020). Finally, some studies point toward species differences. Morgana et al. (2018) found higher MP levels in demersal sculpin compared to pelagic Arctic/polar cod. Liboiron et al. (2019) found MP present only in Atlantic cod, but not in capelin or Atlantic salmon, off the coast of Newfoundland, Canada. The limit of detection for the latter study (e.g., 1 mm) was very high, but may show trends by the large number of individual fish (1010 Atlantic Cod, 350 caplin and 69 Atlantic salmon) investigated.

So far, no general conclusions can be drawn on environmental and biological differences in Arctic fish due to the scarcity of studies. It is not clear if the reported variation is mainly due to the species differences, environmental factors, or the methods applied.

3.6 Sources

In a—hopefully intermediary—state of inability to quantify MP in all fish tissues in a repeatable way, minimizing the potential sources of the MP contaminating the fish might be an area where mitigation could be levered as a preventive measure. However, we know very little about the sources of MP to fish in the Arctic. There are some indications suggesting MP is transported to the Arctic via ocean currents (Cozar et al. 2014; Wichmann et al. 2019; Huserbråten et al. 2022), precipitation (Bergmann et al. 2019), and as waste from boats and ships, including tourism and fishing, that is, fishery gear and products of daily living, oil and gas exploration (UNEP 2009; Nashoug 2017; Bergmann et al. 2017*a*; Falk-Andersson 2019; Eriksen et al. 2020). Also input from wastewater outlets, both with and without treatment, was investigated in the Arctic (Magnusson et al. 2016; Granberg et al. 2019; von Friesen et al. 2020). Furthermore, loss of plastic litter from landfills might be of importance (Granberg et al. 2020). The connectivity between the Arctic Ocean and adjacent southern seas, through the Fram and Bering straits, may play a role. Another possible pathway is transport by marine organisms from more polluted areas into the Arctic (van Franeker 2011; Provencher et al. 2018; Bourdages et al. 2020) or through the food chain (Moore et al. 2022). The relative importance of local and distant pollution sources for MP needs further investigation (PAME 2019).

4. Knowledge gaps and recommendations for microplastic monitoring in the Arctic

4.1 Survey design

To use resources in a meaningful way, monitoring methods of MP pollution in Arctic fish need to meet specific objectives, depending on regions and purposes. Targeted fish sampling can be costly; therefore, planning must be thorough, and the study species and tissues wisely chosen. In the immediate future, MP contamination loads should be compared across different species, tissues, and geographical areas to enable the determination of suitable indicator species and monitoring conditions. To observe trends, several fish species may need to be monitored for MP, as fish are diverse groups across the Arctic.

For the purpose of risk assessment of human consumtion of Arctic fish, the list of species shown to ingest MP includes several species commonly consumed: Atlantic cod (*G. morhua*; Bråte et al. 2016 ; Liboiron et al. 2016; de Vries et al. 2020; Saturno et al. 2020), Greenland cod (*G. ogac*; Granberg et al. 2020), sculpin (*T. nybelini*; Morgana et al. 2018), saithe/pollock (*P. virens*; de Vries et al. 2020), Atlantic salmon (*S. salar*; Liboiron et al. 2019), capelin (*Mallotus villosus*; Liboiron et al. 2019), blue whiting (*M. poutassou*; Malinen 2021), Atlantic mackerel (*S. scombrus*; Malinen 2021), and recently, Arctic char (*S. alpinus*; Hamilton et al. unpublished data). Some industrially caught fish species such as blue whiting are also processed into animal feed without gutting and removing the GIT, so there is a risk of these plastics being fed to domestic animals and fishfarms.

Based on consumption by Arctic residents, primary species to be analyzed in the Arctic are salmonids (e.g., chars, salmon, freshwater whitefish), Arctic/polar cod and sculpin species. Commercial fisheries also catch and export Arctic and At-

lantic cod, saithe, blue whiting, mackerel and salmon. A deep-water fish species that can be regularly assessed for plastic ingestion should be identified in addition. Haddock (*Melanogrammus aeglefinus*), which is of high commercial volume, or cusk/tusk (*Brosme brosme*), known for high accumulation of other toxicants such as mercury and dioxins (Ho et al. 2021) could be options for this purpose. Both have large areas of occurrence, which exceed the Arctic. It should be noted that fish consumption varies greatly with region and culture. For example, a community in the Canadian Arctic Archipelago may consume a higher proportion of a *Salvelinus* spp. (e.g., lake trout and Arctic char) compared to a community located in more interior regions (e.g., the Yukon) where landlocked *Coregonus* species (e.g., whitefish) are abundant. These regional differences should be considered, and specific risk assessments should be done in conjunction with harvest studies that are regionally based and paired with regional data on MP. Other species that should be considered for monitoring are capelin and flounder species. These species represent additional foraging guilds. Capelin is also an important forage fish for dozens of other species in the Arctic region. Including foraging guilds and fish species with occurrence exceeding the Arctic area that are suited for larger region scale comparisons would allow connecting to questions relating to the fate of MP in aquatic ecosystems (AMAP et al. 2021).

Arctic fisheries are providing a considerable share of food sustenance, globally and locally, especially considering protein sources with increased focus on them in the immediate future (https://eatforum.org/). Fishing industry should, therefore, be valuable partners in finding a way to govern this contamination problem and to prevent further escalation. In the case of several Arctic regions, sampling should be carried out in collaboration with local and Indigenous fisheries. For commercially high sale volume species, commercial fishery vessels can be used with the additional benefit of being representative for the market. Such collaboration is for example well established at the Institute of Marine Research in Norway (reference fleet; https://www.hi.no/en/hi/cruises-and-field-work/the-reference-fleet) for regions adjacent to and stretching into the Arctic. Otherwise, samples can be obtained by dedicated cruises, which is often the best way to obtain enough samples with specific characteristics. Also, existing regular cruises, such as those undertaken for population estimations and legacy contaminant surveillance (MILKYS n.d.) should be used for synergy, such as combined use of resources and cruises (collection and sampling of fishes) and correlation studies. Figure 1 shows an overview over areas that are monitored for contaminants in fish in general in the Arctic area (AMAP et al. 2021). For details, refer to the AMAP monitoring guidelines (AMAP et al. 2021).

4.2 Sampling

Caution should be taken when adapting existing fish monitoring for other contaminants to include MP monitoring, as established minimum sample sizes are often designed to assess soluble contaminants. For MP monitoring, by OSPAR and marine framework strategy directive, 50 individuals collected per site for MP analysis is currently recommended (OSPAR 2015), and supported by recent reviews (Hermsen et al. 2018; Dehaut et al. 2019). However, sampling numbers should be adjusted in the context of the questions to be addressed. For example, the number of stations necessary will depend on the mobility of the species in question. The more stationary or restricted by geological boundaries a species is, the more it will reflect local conditions. For example, if inter-lake or inter-fjord comparisons are of interest in a highly mobile species, 50 individuals from each lake or fjord may suffice for this work. If the research question is exploring variability in MP along a fjord, 50 individuals of a stationary fish may be needed from several stations to address this question. However, in many coastal areas these numbers may be excessive. If the spatial scales do not allow for separate sampling of fish, or if the fish population may be highly impacted by taking 50 fish at each station along a single fjord, another environmental compartment should be considered for monitoring. Instead of blindly adhering to 50 individuals, it would ethically make more sense to design monitoring based on pre-existing or pilot data taken under similar conditions. If the goal is to quantify variability in a single area, just enough samples would be needed to approximate the population mean and variance. For a comparison of two or more areas, the number of samples needed to achieve statistical power to detect differences will depend on the variability in the sampled populations.

Some studies on ingested plastic in fish from the Arctic point towards species differences or geographical differences. Liboiron et al. (2019) and Morgana et al. (2018) are suggesting that the MP burden may be species specific and related to foraging as found in seabirds (Poon et al. 2017). If one looks to the global data set on this topic for guidance, one important finding from a recent global meta-analysis of MP in fish is that small planktivorous fish are more likely to have MP accumulated in the GITs than other species (Covernton et al. 2021). How this pattern may hold in the Arctic or other tissues, or not, is yet to be determined, but should be considered when prioritizing species to explore MP ingestion rates. Bråte et al. (2016) observed geographical differences within their study, in which no MP were observed in Atlantic cod from northern Norway or in the vicinity of the capital, Oslo, whereas Atlantic cod from the harbour of the second largest city of Norway, Bergen, contained MP (Bråte et al. 2016). One hypothesis could be that Bergen, with its rough shoreline on the west coast of the European continent, might comb plastics out of the Gulf Stream, as investigated with fish eggs (Furnes et al. 1986; Eriksen et al. 2021; Strand et al. 2021). Another factor is likely to be the body size of the fish, which has been positively correlated with MP abundance in GIT at lower latitudes (McNeish et al. 2018; Jâms et al. 2020; Hamilton et al. 2021; McIlwraith et al. 2021) and should be investigated in the Arctic, too.

Considering the food web, on the one hand, MP in prey organisms, such as plankton, need to be quantified, and on the other hand, accumulation through trophic levels of fish. Further factors increasing data variation are catch season and year. Currently, no recommendation for sampling frequency to establish time trends can be provided because of a lack of data on MP concentrations and determining factors. This is

data that needs to be fed back from results of initial studies leading to planning in an iterative way. Caution should be used with respect to be increasing data variability, otherwise the risk of producing uninterpretable data sets increases.

Sampling can also be biased when not pursuing the same targeted objective under sampling and analysis, in our case, if fish studies are not designed with the purpose of isolating and identifying MP. One of the studies listed in Table 1 was not designed to target MP but was a feeding assessment in which MP content was a side observation (Nielsen et al. 2014). A similar study did not observe any plastic; however, it is not clear whether plastics were not seen or just not reported (Leclerc et al. 2012).

The sampling method applied to an investigation may also influence the detected levels of ingested MP, as has been demonstrated in seabirds by examining gut content, faecal precursor, and guano samples (Provencher et al. 2018). In Arctic fish, Granberg et al. (2020) found the highest numbers of MP per fish GIT in the Arctic. This might be rooted in the low detection size limit or the proximity to a pollution site, but there might be more to it. When catching fish, Granberg et al. (2020) fished with rods and dissected individuals immediately after catching each fish. This likely corresponds to a more "complete" GIT content analysis than usual bulk field collection techniques. When fish from deeper depths are quickly hauled, they invert their stomach and discharge contents, likely including MP, as reported previously (i.e., Lusher et al. 2013Lusher et al. 2013). Apart from this, fish stomachs can also differ in their fullness index depending on seasonal and biological reasons. Sampling only during certain times of the diurnal or annual cycle may lead to an incomplete picture of MP exposure, as some species only feed during part of their life cycle (e.g., some cold-adapted salmonids) or times of the day. When no food content is found in the GIT, it is unlikely that there would be any plastic content either. The other way round, it is likely that there would be found more MP in fish GIT during feeding phases. This factor needs to be considered and controlled for when quantifying average numbers of MP in the GIT. Therefore, rates of empty stomachs and (or) fullness index estimations, provide critical metadata for assessing the exposure of fishes in GIT analysis and should be reported, as has been performed by some studies (Liboiron et al. 2019; Malinen 2021).

A very important factor—often under-communicated in popular scientific dissemination is: Which part of the fish was analyzed? In the case of analysis of MP in Arctic fish, only the content of the GIT or the whole GIT, including the walls have been analyzed, no other organs, tissues, or body fluids.

Fish do not seem to accumulate MP in the GIT over time as seen in some other species, that is, some seabirds (Trevail et al. 2015; Provencher et al. 2018; Bourdages et al. 2020) and crustaceans (Welden and Cowie 2016), which have different gut morphologies. Therefore, the counts of MP in the GIT content of fish likely only represent a snapshot in time for a single organism, generally capturing what enters the GIT before it exits through faeces during the respective stage of the digestive cycle (Peda et al. 2016; Grigorakis et al. 2017; Granberg et al. 2020; Le et al. 2021). Additionally, different species can have vastly different intestine and digestive cycle lengths, adapted to their feeding (Karachle and Stergiou 2010). Nevertheless, monitoring larger MP in the GIT can provide a rough estimate of MP ingestion rates and differences in such rates depending on factors such as species or locations.

It is important to keep in mind that the analysis of one compartment, such as tissue, organ, body fluid, or GIT content, cannot currently be extrapolated to different or mixed compartments. The accumulation potential of those varies, also with different factors, such as MP size (Kögel 2020). For chemical contaminants, inter-tissue extrapolation has been well-studied for some organisms and compounds, and concentrations in one tissue can be related to other tissues through conversion factors (Ackerman et al. 2016), but no such studies have been undertaken to date for MP.

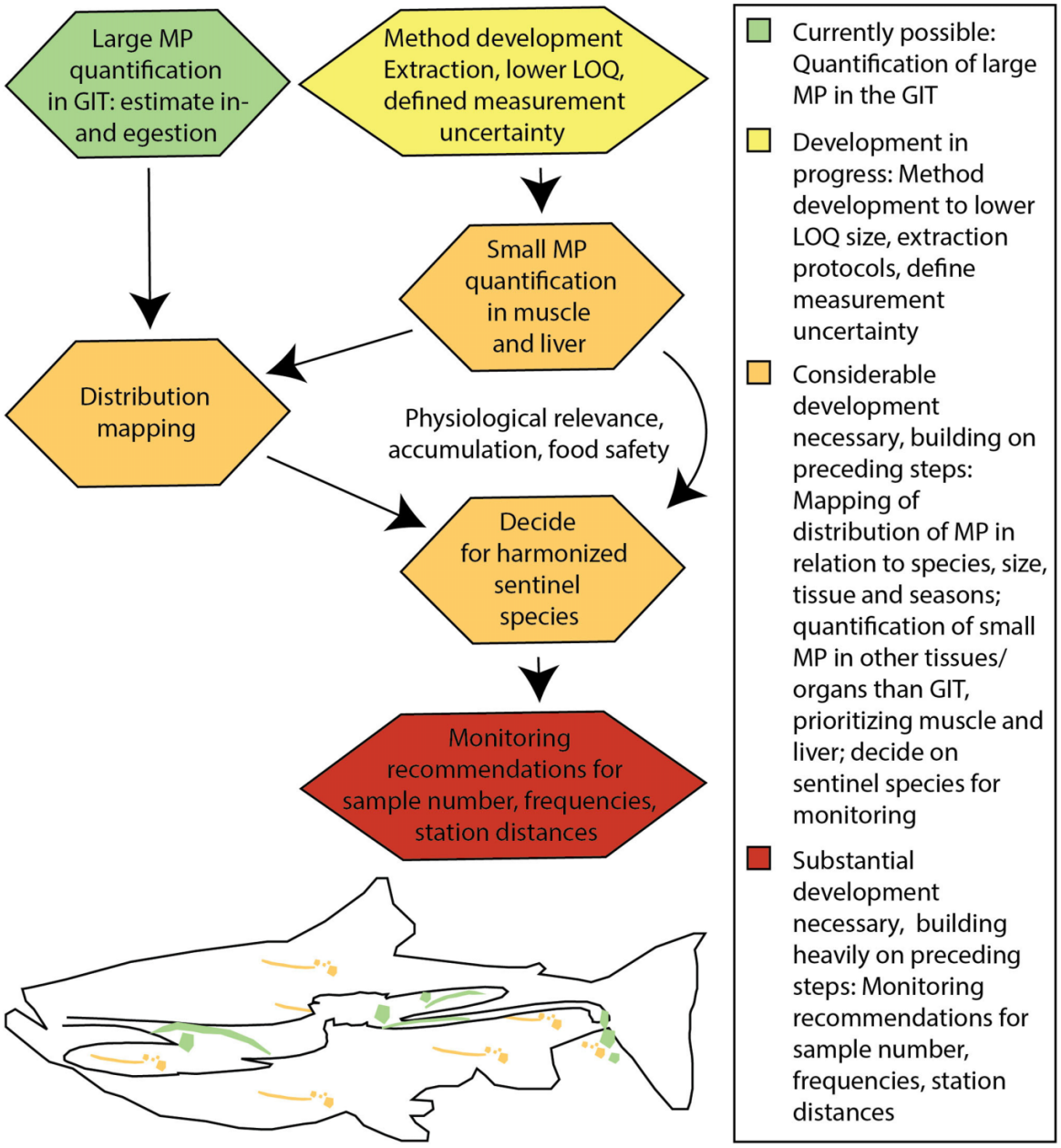
An important finding is that several field reports on non-Arctic fish specimen have shown MP occurrence outside the gut contents, in other fish tissues (Selvam et al. 2021; Ferrante et al. 2022; Makhdoumi et al. 2022; illustrated in Fig. 2). This has not been investigated in Arctic fish yet.

Ideally, end-point analysis should also be considered early in the process, when planning the sampling. Methods such as stereo microscopy and chemical identification (e.g., μ-FTIR, Raman) currently still can have a long processing time and thereby a significant cost per sample. Thus, 50 individuals can be an unrealistic number of samples for quantification, depending on institution capacity and funding. Long-term spatial and temporal monitoring may also require a reduced sample size per sampling event because of the intensity of laboratory processing required for monitoring programs (Bråte et al. 2018). Analysis of pooled samples can reduce the total number of analyses while maintaining representativeness, but comes at the expense of valuable information, such as individual variation and frequency of occurrence.

4.3 Microplastic size and quantification

Besides the different compartments within the animals, size and feeding seasons, there are several factors related to the quantification method that influence the amounts of MP detected to a large extent. Thus, the amounts of MP detected does not necessarily reflect the amounts of MP originally present in the matrices. These factors require thorough characterization in the immediate future. One important factor is the MP size. MP quantification results will only reflect the size range and quality of MP (such as polymer type, color, or shape) that the applied method was capable of detecting (Primpke et al. In press, and Results, this article). The FO % and numbers can only be interpreted in the light of those factors. In abiotic environmental matrices, smaller size fractions of MP consistently occur in higher numbers, down to the detection method's limitations (Mintenig et al. 2017; Bergmann et al. 2017*b*; Peeken et al. 2018; Simon et al. 2018; Haave et al. 2019; Mani et al. 2019; Brandon et al. 2020; Rist et al. 2020). Several studies demonstrated that the incidence of small MP cannot be extrapolated from the incidence of larger MP in a straightforward way with the current available data, neither from macro- to microscale, nor from micro-to nanoscale (Ter Halle et al. 2017; Gomiero et al. 2019; Haave et al. 2019).

Fig. 2. Outline of steps necessary to achieve monitoring of Arctic fish for MP. Fish illustrating that methods for monitoring larger plastics in the intestine are developed (green), while methods for monitoring smaller MP in other tissues require further development (orange). GIT, gastrointestinal tract; MP, microplastic; LOQ, limit of quantification.



Haave et al. and Gomiero et al. have shown that large plastic pieces are transported to different areas of marine sediment as compared to small ones, Ter Halle et al. also found that larger plastic pieces distribute according to their density to different depths of the marine water column, while for smaller plastic pieces, other forces than gravity seem to have a greater influence. More complex extrapolation systems are under development (Koelmans et al. 2020); however, so far they are built on surface water concentrations, not accounting for the MP on sediments which are likely to be ingested by benthic feeding fish. As the authors themselves discuss, testing the general method with the best available data at the time was the primary aim of their study, but those data need to be renewed and expanded with using most recent chemometric procedures to analyze MP spectroscopic data, providing particle number, size, shape and polymer type.

For fish, too, available data points in a similar direction. The size below which MP have been shown to transfer into tissues in significant numbers so far is roughly < 500 μm for pellets and fragments, but up to > 5 mm for fibres (Akhbarizadeh et al. 2018; Gomiero et al. 2020). It should also be noted that there is a mismatch between the recommended minimum size for MP detection in fish monitoring (Box 1), and the feeding particle size preference by various zooplankton organisms, which fish feed on (e.g., for copepods 5–50 μm). The suggested lower size limits for MP monitor-

ing (500 μm) and research (10 μm), respectively, are currently based on instrumental limitations (AMAP et al. 2021) and not the research needs. This hampers the interpretation of MP uptake through food chain transfer at least in pelagic lower trophic level fish. It also hampers the interpretation of exposure studies, which are often designed with MP below 10 μm (Kögel et al. 2019). Therefore, quantifying small MP from fish tissues other than the GIT is a relevant long-term goal for a risk analysis for both seafood safety for human consumption, and the health and population sizes and stability of the fishes themselves (Kögel 2020), but requires further method development (See Primpke et al. In press). In both farmed and wild salmonid livers and fillets from Norwegian areas south of the Arctic, MP below 50 μm occurred more frequently than larger ones. If only larger MP would have been regarded, far fewer MP would have been found. To enable proper risk assessment of MP in fish, necessitating toxicological tests with realistic environmental concentrations, the concentrations of MP, also <50 μm and down into the nanometre range need to be determined (Fig. 2).

The aims and purposes of the fish monitoring should direct the methods; this includes the target MP sizes. When larger MP in fish GITs is the target objective, stake- and rightsholders who have limited access to highly specialized lab equipment, can sample and analyze MP in Arctic fish with the advantage of reducing sampling costs, and results can be rapidly discussed with community members. Importantly, baseline studies across a range of species on the MP size fraction of > 500 μm will provide information necessary for a holistic understanding of how MP impact fish communities. Before monitoring fish on a large scale, methods generally need to be harmonized.

When MP < 500 μm in tissue (such as muscle or liver) are targeted, as, for example, for addressing food safety, method development with high-end instrumentation, clean laboratories, quality assurance (QA) with interlaboratory comparison exercises, and measurement uncertainty determination will be needed. It must be noted that such methods may not be available for the pan-Arctic region in the short term and therefore needs to be incorporated into monitoring at a later timepoint. Due to the considerable method development that is necessary to achieve meaningful monitoring across time and regions, we have divided our recommendations for requirements for the data that need to be collected into two groups, where "Basic" should be feasible to a large number of interest groups and countries. The "Advanced" comprise more cost-intensive goals, which are not feasible or not necessary for all purposes (Box 1).

4.4 Sample processing, microplastic extraction, QA/QC, analysis, and reporting

Depending on the target tissue and aim of each study, different steps are required. There are several prevalent methods for assessing MP in fish. In general, GITs are dissected out and rinsed externally. Then, the GIT content is analyzed with or without including the gut lining. Direct visual inspection or extraction can be performed. Studies focusing on ingestion of larger items of > 500 μm can use a visual sorting method, but limitations include a high detection limit in terms of MP size and increased risks of procedural contamination from extended exposure. When planning studies or interpreting results, one should keep in mind how the GIT dissection is done, such as with or without visual aid. The color of the dissection plate (or other background color) may also lead to a color detection bias. Similarly, depending on the diet of the fish which can be both species-, location-, and season-specific, it may be difficult to distinguish between natural organic material and ingested MP, there may also be a variable color bias, if no efficient digestion method was applied.

In parallel to studies of the MP content in the GIT, MP content in muscle and liver of said indicator species, including small MP < 500 μm, should be analyzed by laboratories with the necessary instrumentation and equipment, and methods should be further optimised (Box 1; advanced, Fig. 2). For such studies, MP extraction is required. A suitable MP extraction protocol requires removal of the tissue, while leaving the MP intact for quantitative analysis, satisfactory recovery percentages and contamination avoidance. The protocol used for digestion will be dependent on the matrix composition (Lusher et al. 2020). Alkaline digestion (Thiele et al. 2021) or enzymatic purification (von Friesen et al. 2019), combined (Sussmann et al. 2021) or combined with oxidation (Loder et al. 2017) are the most prevalent, and successful methods. Temperatures of digesting agents should be kept below 40° C and molarity adjusted for plastic preservation (Thiele et al. 2019). Because some plastic types dissolve with acid digestion, this approach is no longer recommended (Dehaut et al. 2016; Kershaw et al. 2019). At the current stage of the technology, there is still much room for increasing the quality and reducing the time and costs of these protocols.

For mapping and monitoring, harmonized sampling and sample preparation methods in the laboratory are important. This includes protocols to reduce and monitor procedural contamination. Until those are officially established, we suggest the following criteria to enable complementarity of monitoring studies based on existing publications (Lusher et al. 2017; Bessa et al. 2019). For contamination controls, the whole analytical chain from sample preparation to analysis needs to be considered and contamination kept as low as possible through clean laboratory methods and established QA/quality control (QA/QC) measures (Brander et al. 2020; Cowger et al. 2020). For simpler measures, samples should be covered with material other than plastic (e.g., clean aluminium foil) as much of the time as possible (Prata et al. 2021). Equipment and aluminium foil can be heat treated in a muffle oven to disintegrate plastic contamination (Prata et al. 2021). A wet filter or an open water container can be used next to the dissected organism to control for airborne contamination (Prata et al. 2021). The analysis of MP < 500 μm requires clean laboratory methods with air filtration, such as LAF benches (Wesch et al. 2017; Prata et al. 2021). Where in-air filtration is not feasible, a dust box as used at construction sites to reduce airborne particle numbers might be used instead (Bergmann et al. 2019). All digestive agents must be prepared and filtered according to the size-related detection limit to remove impurities and to prevent contamination of the samples (Prata et al. 2021). QA/QC procedures are increas-

Box 1. Required data for monitoring microplastics (MPs) in Arctic fish

Basic
- Name of researcher
- Species
- Location, including longitude and latitude
- Date: day, month, year, time
- Wet weight and total length of fish
- Liver weight
- Tissue(s) sampled; method of gastro-intestinal tract (GIT) lining investigation
- Frequency of occurrence of MP per individual/tissue, including cases of 0 (**rate** of empty GIT)
- For MP > ca. 500 μm: either MP mass or number per tissue weight and particle size group, as mean, with standard deviation and number of samples, median, for individual or defined pooled samples. When reporting for individual fish, include individuals without detected plastic contamination.
- Positive controls and procedural controls
- Collection, extraction, analysis method applied, including equipment, quality assurance/quality control, limit of detection as MP size and (or) mass and measurement uncertainty

Advanced
- Polymer type group (according to (Primpke et al. 2017)
- Shape of the MP, as fibre, fragment, or bead
- Color identification of MP
- For particles < ca. 500 μm: either MP mass or number per tissue weight and MP size group, as mean, with standard deviation and number of samples, median, for individual or defined pooled samples. When reporting for individual fish, include individuals without detected plastic contamination
- Positive, negative, and procedural controls, polymer type and size specific measurement uncertainties
- Sex
- Age of fish
- Depth of collection
- Weather conditions
- Name of fish harvester and boat

ingly applied in the MP research field, including fish studies (Savoca et al. 2021). Ideally, fish should be delivered whole and rinsed with filtered water before cutting and preparing tissue samples inside the clean lab. To avoid contamination from disintegrating inner organs to fillets, frozen fish should not be stored for extended periods (>1 year) even if frozen, be thawed lying on its side, and fillet samples taken from the upper side (Kögel et al.'s personal observation). When preparing samples from muscle, the fish must be rinsed before extraction to remove fish scales because they contain biopolymers, which are very similar to some plastic types and could therefore be mistakenly identified as plastic (Kögel et al.'s personal observation). All instruments must be cleaned between individual samples. Plastic gloves and tools should be avoided or controlled for in the sample results (Prata et al. 2021). All plastic materials used during dissection should be analyzed to provide references for polymer identification. Results of controls, accounting for fibres and other particles of all reported size ranges, and correction calculations should be reported in detail. Raw, uncorrected data should also be made available.

To compare numeric values on plastic contamination between studies, the mesh size and material of filters, the smallest and largest particle sizes that are theoretically measurable, and those that were identified, with mean and median sizes need to be provided. Be aware that the smallest detected particle size does not equal the limit of quantification (LOQ) if not all particles of this size class will be detected or corrected for by the co-analysis of standards, within a defined measurement uncertainty. Preparation steps lead to unequal loss of different types of particles through filtration, by foaming or clogging, to equipment walls or degradation by extraction (Sussmann et al. 2021). Extraction efficiencies, measurement uncertainties, recovery percentages and procedural contamination should therefore be established not only for each fish matrix and method applied, but also respective to size classes, shapes, polymer types and concentration range of the analytes. Such information is generally often lacking in the publications from this research field, entirely in the data set available for MP in Arctic fish (Table 1). If this is not feasible, then the lack of such recovery tests and the reasons for it should be critically discussed with the publication of the data (Cowger et al. 2020). When testing compliance of legacy contaminants with maximum levels, parameters for methods and accuracy, measurement uncertainty, and limit of quantitation (LOQ) are measured and regularly tested by proficiency tests. All of these are defined in accreditation protocols. For MP quantification, such accreditation processes are still in their infancy. Work towards these goals has been started by several initiatives, e.g., from QUASIMEME Laboratory Performance Studies on MP (van Mourik et al. 2021), European Commission JRC/BAM inter-laboratory comparison (proficiency testing) on MP (Belz et al. 2021), and the H2020 Harmonisation Project—EUROqCHARM (CE-SC5-29-2020, www.EUROqCHARM.eu). International and regional standard organisations are working on standard protocols for MP analysis including CEN (TC 249/WG 24 (plastics); TC248/WG 37 (textiles)) and ISO (TC 61, SC 14, WG4 (plastics), will be fused into TC 147/SC2; TC 45 (rubber); TC38, WG 34 (textiles)), and a technical report with the title "Plastics—Environmental aspects—State of knowledge and methodologies" (ISO/TR 21960) has been published and is about to be followed up (ISO/DIS 24187:2021) with a report focussing on the analysis.

The most promising size analysis methods for fish tissue to date are FTIR microscopy and pyrolysis-gas chromatography/mass spectrometry (py-GC/MS) (Fischer and Scholz-Bottcher 2017, 2019; Gomiero et al. 2020) for monitoring pur-

poses (Primpke et al. In press). Raman microscopy is suited for clarification of the size-distribution of MP < 10 μm for selected samples but too time consuming for routine monitoring purposes. Method development for nanoplastic analysis is ongoing but poses large challenges because the particles adhere to all surfaces and are easily dissolved in the attempt to extract them from biotic matrices (Correia and Loeschner 2018).

Regarding MP shape, harmonization of reporting is also important. The studies on MP in Arctic fish were hard to compare, as not the same categories—fragments, fibers, films, and filaments were used by all reports. None of the studies analyzed all detected MP for chemical identity, nor was the representativeness of the chosen fraction for identification investigated. Because plastic polymers have different physiological effects depending on their composition (Avio et al. 2015; Booth et al. 2016; Green et al. 2016; Mattsson et al. 2017; Rochman et al. 2017), polymer type analysis of the total or a representative fraction of all MP in the investigated tissue is meaningful and critical when considering impacts and effects on biota. To achieve better representativeness of presented data, both FO % and mean and median numbers of particles per individual/amount of tissue needs to be presented. Polymer type, sizes, and shapes should be reported if possible. To avoid introducing bias by the contamination data handling, it is important to not only subtract contamination from results, but to report contamination results. For further details, we refer to Box 1 and the AMAP monitoring guidelines (AMAP et al. 2021). For easier data sharing, data distribution tools, such as databases, should be explored for their potential to add MP. At least, extensive supplemental data collections or archiving data in publicly available repositories linked to individual publications can be used for now.

For a detailed synthesis adapted to the Arctic, we refer the readers to the AMAP Guidelines (AMAP et al. 2021) and the article on MP analysis method in this special issue (Primpke et al. In press).

4.5 Additives

One additional aspect to consider, not included in the present harmonization recommendations, is that MP are manufactured containing residual monomers, chemical catalysing agents, reaction by-products, non-intentionally added substances and additives (e.g., flame retardants, biocides, plasticizers, colorants, and stabilizers) and can sorb, and hence be a vector for soluble contaminants (e.g., heavy metals, dioxins, polychlorinated biphnenyls (PCBs), and flame retardants). This topic is discussed in Hamilton et al. In press. The issue of additive chemicals has barely been addressed for seafood organisms, and to our knowledge not at all in Arctic fish. In farmed salmon from the Norwegian area, south of the Arctic, some phthalates might be distributed in a geographically distinct pattern (Gomiero et al. 2020). For monitoring, it will be difficult to differentiate the origin of a soluble contaminant and metabolites between MP and other sources such as prey/food, but the hazard these substances present deserves attention and research with the final goal of risk assessment. Further research incorporating additive chemicals will be necessary to form a clear picture of plastic impacts on Arctic fish.

4.6 Arctic Indigenous peoples and communities

Finally, these recommendations are based on the priorities and insights of an international scientific community. However, this does not mean they include the research needs and priorities of communities and Indigenous peoples in the Arctic, and some of the methods, categories, standards, and research questions in plastic pollution research in the Arctic are skewed towards southern understandings and landscapes (Liboiron et al. 2021; Melvin et al. 2021). The Inuit Tapiriit Kanatami, an organization representing the 65 000 Inuit in the Canadian Arctic, has written in its National Inuit Strategy for Research that, "for far too long, researchers have enjoyed great privilege as they have passed through our communities and homeland, using public or academic funding to answer their own questions about our environment, wildlife, and people. Many of these same researchers then ignore Inuit in creating the outcomes of their work for the advancement of their careers, their research institutions, or their governments. This type of exploitative relationship must end". (ITK 2018). They recommend five priority areas for research in their homelands, including: advancing Inuit governance in research, including being part of funding decisions; enhancing the ethical conduct of research, including strong community partnerships; ensuring Inuit access, ownership, and control over data and information gathered in their homelands, including monitoring data; and building capacity in Inuit research through skill-sharing, equal partnership, and research infrastructure. (ITK 2018). While each Indigenous group and community in the Arctic will be different, many of these principles will hold across the Arctic. We recommend that future monitoring research aligns with these principles with an emphasis on the priorities of local and regional Arctic communities.

5. Summarized conclusions and research gaps

For a risk analysis in general, we need to start with a hazard definition. This has already been achieved: We know that MP are present in Arctic fish (Table 1) and that MP in fish can have negative effects (Kögel et al. 2019). Published information on MP pollution in Arctic fish is scarce and restricted to the GIT. The studies show high variation, both in the applied methods and the results, suggesting a need for method and reporting harmonization and more data in general (AMAP et al. 2021). Smaller MP and MP in tissues other than the GIT are so far not investigated quantitatively. In Arctic fish, MP <500 μm are scarcely analyzed, whilst MP < 20 μm have not been assessed (Table 1). Therefore, the obtained frequency of occurrence and individual MP counts per fish will probably underestimate the real situation in fish as a whole organism and throughout size classes. Future investigations need to also quantify small MP, as well as in other fish tissues than the GIT, starting with liver and muscle.

Little can be concluded about sources, geographical distribution, and species dependency of the levels of pollution with MP in Arctic fish. Such knowledge is valuable for targeted mitigation of MP pollution. To strengthen recommendations for monitoring (e.g., for monitoring of MP in fish GIT) (Fig. 2), the sample numbers, frequency, and station distances necessary to achieve statistical power must be established.

To achieve a full risk assessment, the major knowledge gaps to be filled are the measurement uncertainty of sample preparation and analysis methods which need testing by recovery experiments. We need to quantify the exposure and accumulation of MP and plastic additives in different species and tissues wild fish, throughout the food chain, including information on types and size of plastic particles and analysis of parameters influencing MP ingestion. We need effect studies for the relevant exposure ranges, particle sizes and types found in the environment and, importantly, for long-term exposure (Kögel 2020). Furthermore, suitable indicator species need to be chosen.

We conclude that the Commission Decision on Good Environmental Status statement in the Marine Strategy Framework Directive from the European Commision: "The amount of litter and microlitter ingested by marine animals is at a level that does not adversely affect the health of the species concerned" remains to be proven. In the coming years, more studies will probably use harmonized methods, and thus, the research community will be able to form more specific, evidence-based recommendations to address a series of questions related to monitoring MP in Arctic fish as related to environmental and human health. The research field is in its infancy, leading to many difficulties, but this can also be seized as an opportunity to foster harmonization across the Arctic at this early stage.

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References

Ackerman, J.T., Eagles-Smith, C.A., Herzog, M.P., Hartman, C.A., Peterson, S.H. Evers, D.C., et al. 2016. Avian mercury exposure and toxicological risk across western North America: a synthesis. Sci. Total Environ. **568**: 749–769. doi:10.1016/j.scitotenv.2016.03.071.

Akhbarizadeh, R., Moore, F., and Keshavarzi, B. 2018. Investigating a probable relationship between microplastics and potentially toxic elements in fish muscles from northeast of Persian Gulf. Environ. Pollut. **232**: 154–163. doi:10.1016/j.envpol.2017.09.028.

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

Assessment 2016: Chemicals of emerging concern 2017 xvi+353 Arctic Monitoring & Assessment Programme (AMAP)

Avio, C.G., Gorbi, S., Milan, M., Benedetti, M., Fattorini, D. d'Errico, G., et al. 2015. Pollutants bioavailability and toxicological risk from microplastics to marine mussels. Environ. Pollut. **198**: 211–222. doi:10.1016/j.envpol.2014.12.021.

Barboza, L.G.A., Dick Vethaak, A., Lavorante, B., Lundebye, A.K., and Guilhermino, L. 2018. Marine microplastic debris: an emerging issue for food security, food safety and human health. Mar. Pollut. Bull. **133**: 336–348. doi:10.1016/j.marpolbul.2018.05.047.

Bellasi, A., Binda, G., Pozzi, A., Galafassi, S., Volta, P., and Bettinetti, R. 2020. Microplastic contamination in freshwater environments: a review, focusing on interactions with sediments and benthic organisms. Environments **7**: 30. doi:10.3390/environments7040030.

Belz, S., Bianchi, I.C.C., Emteborg, H., Fumagalli, F., Geiss, O., Gilliland, D., et al. 2021. Current status of the quantification of microplastics in water - Results of a JRC/BAM inter-laboratory comparison study on PET in water. In EUR 30799 EN. Publications Office of the European Union, Luxembourg.

Bergmann, M., Lutz, B., Tekman, M.B., and Gutow, L. 2017a. Citizen scientists reveal: marine litter pollutes arctic beaches and affects wild life. Mar. Pollu.t Bull. **125**: 535–540. doi:10.1016/j.marpolbul.2017.09.055.

Bergmann, M., Mutzel, S., Primpke, S., Tekman, M.B., Trachsel, J., and Gerdts, G. 2019. White and wonderful? Microplastics prevail in snow from the Alps to the Arctic. Sci Adv. **5**: eaax1157. doi:10.1126/sciadv.aax1157.

Bergmann, M., Wirzberger, V., Krumpen, T., Lorenz, C., Primpke, S., Tekman, M.B., and Gerdts, G. 2017b. High quantities of microplastic in Arctic deep-sea sediments from the HAUSGARTEN observatory. Environ. Sci. Technol. **51**: 11000–11010. doi:10.1021/acs.est.7b03331.

Bessa, F., Frias, J., Kögel, T., and Gerdts, G. 2019. Harmonized protocol for monitoring microplastics in biota. BASEMAN - Defining the baselines and standards for microplastics analyses in European waters Technical Report. Ocean best practices repository. Available at https://repository.oceanbestpractices.org/handle/11329/1313

Booth, A.M., Hansen, B.H., Frenzel, M., Johnsen, H., and Altin, D. 2016. Uptake and toxicity of methylmethacrylate-based nanoplastic particles in aquatic organisms. Environ. Toxicol. Chem. **35**: 1641–1649. doi:10.1002/etc.3076.

Bourdages, M.P.T., Provencher, J.F., Baakc, J.E., Mallory, M.L., and Vermaire, J.C. 2020. Breeding seabirds as vectors of microplastics from sea to land: evidence from colonies in Arctic Canada. Sci. Total Environ. **764**: 142808. doi:10.1016/j.scitotenv.2020.142808.

Brander, S.M., Renick, V.C., Foley, M.M., Steele, C., Woo, M., Lusher, A., Carr, S., Helm, P., Box, C. Cherniak, S., et al. 2020. Sampling and quality assurance and quality control: a guide for scientists investigating the occurrence of microplastics across matrices. Appl. Spectrosc. **74**: 1099–1125. doi:10.1177/0003702820945713.

Brandon, J.A., Freibott, A., and Sala, L.M. 2020. Patterns of suspended and salp-ingested microplastic debris in the North Pacific investigated with epifluorescence microscopy. Limnol. Oceanogr. Lett. **5**: 46–53. doi:10.1002/lol2.10127.

Bråte, I.L.N., Eidsvoll, D.P., Steindal, C.C., and Thomas, K.V. 2016. Plastic ingestion by Atlantic cod (*Gadus morhua*) from the Norwegian coast. Mar. Pollut. Bull. **112**: 105–110. doi:10.1016/j.marpolbul.2016.08.034.

Bråte, I.L.N., Hurley, R., Iversen, K., Beyer, J., Thomas, K.V. Steindal, C.C., et al. 2018. *Mytilus* spp. as sentinels for monitoring microplastic pollution in Norwegian coastal waters: a qualitative and quantitative study. Environ. Pollut. **243**: 383–393. doi:10.1016/j.envpol.2018.08.077.

Bucci, K., Bikker, J., Stevack, K., Watson-Leung, T., and Rochman, C. 2021. Impacts to larval fathead minnows vary between preconsumer and environmental microplastics. Environ. Toxicol. Chem. **41**: 858–868. doi:10.1002/etc.5036.

Buhl-Mortensen, P., and Buhl-Mortensen, L. 2018. Impacts of bottom trawling and litter on the seabed in Norwegian waters. Front. Mar. Sci. **5**. doi:10.3389/fmars.2018.00042.

Baak, J.E., Linnebjerg, J.F., Barry, T., Gavrilo, M.V., Mallory, M.L., Price, C., and Provencher, J.F. 2020. Plastic ingestion by seabirds in the circumpolar Arctic: a review. Environ. Rev. **28**: 506–516. doi:10.1139/er-2020-0029.

Campanale, C., Massarelli, C., Savino, I., Locaputo, V., and Uricchio, V.F. 2020. A detailed review study on potential effects of microplastics and additives of concern on human health. Int. J. Env. Res. Public Health **17**: 1212. doi:10.3390/ijerph17041212.

Correia, M., and Loeschner, K. 2018. Detection of nanoplastics in food by asymmetric flow field-flow fractionation coupled to multi-angle light scattering: possibilities, challenges and analytical limitations. Anal. Bioanal. Chem. **410**: 5603–5615. doi:10.1007/s00216-018-0919-8.

Covernton, G.A., Davies, H.L., Cox, K.D., El-Sabaawi, R., Juanes, F., Dudas, S.E., and Dower, J.F. 2021. A Bayesian analysis of the factors determining microplastics ingestion in fishes. J. Hazard. Mater. **413**: 125405. doi:10.1016/j.jhazmat.2021.125405

Cowger, W., Booth, A.M., Hamilton, B.M., Thaysen, C., Primpke, S., Munno, K., et al. 2020. Reporting guidelines to increase the reproducibility and comparability of research on microplastics. Appl. Spectrosc. **74**: 1066–1077. doi:10.1177/0003702820930292 .

Cozar, A., Echevarria, F., Gonzalez-Gordillo, J.I., Irigoien, X., Ubeda, B., Hernandez-Leon, S., et al. 2014. Plastic debris in the open ocean. P. Natl. Acad. Sci. U. S. A. **111**: 10239–10244. doi:10.1073/pnas.1314705111.

Critchell, K., and Hoogenboom, M.O. 2018. Effects of microplastic exposure on the body condition and behaviour of planktivorous reef fish (*Acanthochromis polyacanthus*). PLoS ONE **13**: e0193308.10.1371/journal.pone.0193308.

Danopoulos, E., Jenner, L.C., Twiddy, M., and Rotchell, J.M. 2020. Microplastic contamination of seafood intended for human consumption: a systematic review and meta-analysis. Environ. Health Perspect. **128**: 126002. doi:10.1289/EHP7171.

de Vries, N.A., Govoni, D., Árnason, S.H., and Carlsson, P. 2020. Microplastic ingestion by fish: body size, condition factor and gut fullness are not related to the amount of plastics consumed. Mar. Pollut. Bull. **151**: 110827. doi:10.1016/j.marpolbul.2019.110827.

De-la-Torre, G.E. 2020. Microplastics: an emerging threat to food security and human health. J. Food Sci. Technol. **57**: 1601–1608. doi:10.1007/s13197-019-04138-1.

Dehaut, A. Hermabessiere, L., and Duflos, G. 2019. Current frontiers and recommendations for the study of microplastics in seafood. TracTrends in Analytical Chemistry **116**: 346–359. doi:10.1016/j.trac.2018.11.011

Dehaut, A., Cassone, A.L., Frere, L., Hermabessiere, L., Himber, C., Rinnert, E., et al. 2016. Microplastics in seafood: benchmark protocol for their extraction and characterization. Environ. Pollut. **215**: 223–233. doi:10.1016/j.envpol.2016.05.018.

Dietz, R., Letcher, R.J., Desforges, J.P., Eulaers, I., Sonne, C., Wilson, S., et al. 2019. Current state of knowledge on biological effects from contaminants on Arctic wildlife and fish. Sci. Total Environ. **696**: 110827. doi:10.1016/j.scitotenv.2019.133792

Eriksen, E., van der Meeren, G.I., Nilsen, B.M., von Quillfeldt, C.H., and Johnsen, H. 2021. Særlig verdifulle og sårbare områder (SVO) i norske havområder - Miljøverdi - en gjennomgang av miljøverdier og grenser i eksisterende SVO og forslag til nye områder (Particularly valuable and vulnerable areas (SVO) in Norwegian seas - Environmental values) Havforskningsinstituttet. Available from: https://www.hi.no/templates/reporteditor/report-pdf?id=45755&55653641).

Eriksen, M., Borgogno, F., Villarrubia-Gomez, P., Anderson, E., Box, C., and Trenholm, N. 2020. Mitigation strategies to reverse the rising trend of plastics in polar regions. Environ. Int. **139**: 105704. doi:10.1016/j.envint.2020.105704.

Espinosa Ruiz, C., Esteban, M.A., and Cuesta, A. 2016. Microplastics in aquatic environments and their toxicological implication for fish. In:Toxicology - New Aspects to This Scientific Conundrum. Edited by:Soloneski and Larramendy IntechOpen, London, UK. doi:10.5772/64815 .

Falk-Andersson, J. 2019. Svalbard beach litter deep dive. SALT report number 1033.

Ferrante, M., Pietro, Z., Allegui, C., Maria, F., Antonio, C., Pulvirenti, E., et al. 2022. Microplastics in fillets of Mediterranean seafood. A risk assessment study. Environ. Res. **204**: 112247. doi:10.1016/j.envres.2021.112247.

Fischer, M., and Scholz-Bottcher, B.M. 2017. Simultaneous trace identification and quantification of common types of microplastics in environmental samples by pyrolysis-gas chromatography-mass spectrometry. Environ. Sci. Technol. **51**: 5052–5060. doi:10.1021/acs.est.6b06362.

Fischer, M., and Scholz-Bottcher, B.M. 2019. Microplastics analysis in environmental samples - recent pyrolysis-gas chromatography-mass spectrometry method improvements to increase the reliability of mass-related data. Anal. Methods **11**: 2489–2497. doi:10.1039/C9AY00600A.

Ford, J.D. 2009. Vulnerability of Inuit food systems to food insecurity as a consequence of climate change: a case study from Igloolik, Nunavut. Reg. Environ. Change **9**: 83–100. doi:10.1007/s10113-008-0060-x.

Fred-Ahmadu, O.H., Bhagwat, G., Oluyoye, I., Benson, N.U., Ayejuyo, O.O., and Palanisami, T. 2020. Interaction of chemical contaminants with microplastics: principles and perspectives. Sci. Total Environ. **706**: 135978. doi:10.1016/j.scitotenv.2019.135978.

Furnes, G.K., Hackett, B., and Saetre, R. 1986. Retroflection of Atlantic water in the Norwegian trench. Deep-Sea Res. **33**: 247–265. doi:10.1016/0198-0149(86)90121-4.

Gomes, T., Bour, A., Coutris, A., Almeida, A.C., Bråte, I.G. Wolf, R., et al. 2021. Ecotoxicological impacts of micro- and nanoplastics in terrestrial and aquatic environments. *In Microplastic in the Environment: Pattern and Process*, Edited by.Bank Springer Open Press. Berlin/Heidelberg, Germany

Gomiero, A., Haave, M., Bjorøy, Ø., Herzke, D., Kögel, T., Nikiforov, V., and Øysæd, K.B. 2020. NORCE Reports. Bergen, Norway

Gomiero, A., Oysaed, K.B., Agustsson, T., van Hoytema, N., van Thiel, T., and Grati, F. 2019. First record of characterization, concentration and distribution of microplastics in coastal sediments of an urban fjord in south west Norway using a thermal degradation method. Chemosphere **227**: 705–714. doi:10.1016/j.chemosphere.2019.04.096.

Granberg, M., von Friesen, L., Bach, L., Collard, F., Strand, J., and Gabrielsen, G. 2019. . IVL Swedish Environmental Research Institute. Fiskebäckskil, Sweden. pp. 28.

Granberg, M.E., von Friesen, L.W., Ask, A., Collard, F., Magnusson, K. Eriksson Wiklund, A.-K., et al. 2020. Microlitter in arctic marine benthic food chains and potential effects on sediment dwelling fauna. Nordic Council TemaNord. Denmark, Finland, Iceland, Norway, Sweden, the Faroe Islands, Greenland and Åland **528**, pp. 77.

Green, D.S., Boots, B., Sigwart, J., Jiang, S., and Rocha, C. 2016. Effects of conventional and biodegradable microplastics on a marine ecosystem engineer (*Arenicola marina*) and sediment nutrient cycling. Environ. Pollut. **208**: 426–434. doi:10.1016/j.envpol.2015.10.010.

Grigorakis, S., Mason, S.A., and Drouillard, K.G. 2017. Determination of the gut retention of plastic microbeads and microfibers in goldfish (Carassius auratus). Chemosphere **169**: 233–238. doi:10.1016/j.chemosphere.2016.11.055.

Grøsvik, B.E., Prokhorova, T., Eriksen, E., Krivosheya, P., Homeland, P.A., and Prozorkevich, D. 2018. Assessment of marine litter in the Barents Sea, a part of the Joint Norwegian Russian ecosystem survey. Front. Mar. Sci. **5**. doi:10.3389/fmars.2018.00072.

Halsband, C., and Herzke, D. 2019. Plastic litter in the European Arctic: what do we know? Emerg. Contam. **5**: 308–318. doi:10.1016/j.emcon.2019.11.001.

Hamilton, B.M., Baak, J.E., Vorkamp, K., Hammer, S., Granberg, M., Herzke, D., and Provencher, J.F. In press. Plastics as a carrier of chemical additives to the arctic: possibilities for strategic monitoring across the circumpolar North. Arct. Sci. doi:10.1139/AS-2021-0055.

Hamilton, B.M., Rochman, C.M., Hoellein, T.J., Robison, B.H., Van Houtan, K.S., and Choy, C.A. 2021. Prevalence of microplastics and anthropogenic debris within a deep-sea food web. Mar. Ecol. Prog. Ser. **675**: 23–33. doi:10.3354/meps13846.

Hermsen, E. Mintenig, S.M. Besseling, E. and Koelmans, A.A. 2018. Quality Criteria for the Analysis of Microplastic in Biota Samples: A Critical Review. Environmental Science & Technology **52**: 10230–10240. doi:10.1021/acs.est.8b01611

Ho, Q.T., Bank, M.S., Azad, A.M., Nilsen, B.M., Frantzen, S., Boitsov, S., Maage, A., Kogel, T., Sanden, M. Froyland, L., et al. 2021. Co-occurrence of contaminants in marine fish from the North East Atlantic Ocean: implications for human risk assessment. Environ. Int. **157**: 106858. doi:10.1016/j.envint.2021.106858.

Huserbråten, M., Hattermann, T., Broms, C., and Albretsen, J. 2022. Transpolar drift-pathways of riverine European microplastic. Sci. Rep. **12**: 3016. doi:10.1038/s41598-022-07080-z.

Haave, M., Lorenz, C., Primpke, S., and Gerdts, G. 2019. Different stories told by small and large microplastics in sediment - first report of microplastic concentrations in an urban recipient in Norway. Mar. Pollut. Bull. **141**: 501–513. doi:10.1016/j.marpolbul.2019.02.015.

Inuit Tapiriit Kanatami. 2018. National Inuit Strategy on Research. Canada. Available from: https://www.itk.ca/wp-content/uploads/2020/10/ITK-National-Inuit-Strategy-on-Research.pdf.

Jâms, I.B., Windsor, F.M., Poudevigne-Durance, T., Ormerod, S.J., and Durance, I. 2020. Estimating the size distribution of plastics ingested by animals. Nat. Commun. **11**: 1594. doi:10.1038/s41467-020-15406-6.

Jeong, C.B., Won, E.J., Kang, H.M., Lee, M.C., Hwang, D.S. Hwang, U.K., et al. 2016. Environ. Sci. Technol. **50**: 8849–8857. doi:10.1021/acs.est.6b01441.

Kanhai, L.K., Gardfeldt, K., Krumpen, T., Thompson, R.C., and O'Connor, I. 2020. Microplastics in sea ice and seawater beneath ice floes from the Arctic Ocean. Sci. Rep. **10**: 5004. doi:10.1038/s41598-020-61948-6.

Karachle, P.K., and Stergiou, K.I. 2010. Gut length for several marine fish: relationships with body length and trophic implications. Mar. Biodivers. Rec. **3**: E106. doi:10.1017/S1755267210000904.

Kershaw, P., Turra, A., and Galgani, F. 2019. *Guidelines for the Monitoring and Assessment of Plastic Litter in the Ocean*. GEAMP Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection by United Nations Environment Programme (UNEP). New York, USA

Koelmans, A.A., Redondo-Hasselerharm, P.E., Nor, N.H.M., and Kooi, M. 2020. Solving the nonalignment of methods and approaches used in microplastic research to consistently characterize risk. Environ. Sci. Technol. **54**: 12307–12315. doi:10.1021/acs.est.0c02982.

Kwon, J.H., Chang, S., Hong, S.H., and Shim, W.J. 2017. Microplastics as a vector of hydrophobic contaminants: importance of hydrophobic additives. Integr. Environ. Assess. Manag. **13**: 494–499. doi:10.1002/ieam.1906.

Kühn, S., Schaafsma, F.L., van Werven, B., Flores, H., Bergmann, M. Egelkraut-Holtus, M., et al. 2018. Plastic ingestion by juvenile polar cod (*Boreogadus saida*) in the Arctic Ocean. Polar Biol. **41**: 1269–1278. doi:10.1007/s00300-018-2283-8.

Kögel, T., Bjoroy, O., Toto, B., Bienfait, A.M., and Sanden, M. 2019. Micro- and nanoplastic toxicity on aquatic life: determining factors. Sci. Total Environ. **709**: 136050. doi:10.1016/j.scitotenv.2019.136050.

Kögel, T., Refosco, , and Maage, A. 2020. Surveillance of seafood for microplastics. 1–34 *In Handbook of Microplastics in the Environment*. C. Springer.Rocha-Santos Costa and Mouneyrac Berlin/Heidelberg, Germany

Le, H.T.M.D., Lie, K.K., Etayo, A., Ronnestad, I., and Saele, O. 2021. Physical and nutrient stimuli differentially modulate gut motility patterns, gut transit rate, and transcriptome in an agastric fish, the ballan wrasse. PLoS One **16**: e0247076. doi:10.1371/journal.pone.0247076

Leclerc, L.M.E., Lydersen, C., Haug, T., Bachmann, L., Fisk, A.T., and Kovacs, K.M. 2012. A missing piece in the Arctic food web puzzle? Stomach contents of greenland sharks sampled in Svalbard, Norway. Polar Biol. **35**: 1197–1208. doi:10.1007/s00300-012-1166-7.

Lehtiemi, M., Hartikainen, S., Näkki, P., Engström-Öst, J., Koistinen, A., and Setälä, O. 2018. Size matters more than shape: ingestion of primary and secondary microplastics by small predators. Food Webs **17**: e00097. doi:10.1016/j.fooweb.2018.e00097.

Liboiron, M., Liboiron, F., Wells, E., Richard, N., Zahara, A. Mather, C., et al. 2016. Low plastic ingestion rate in Atlantic cod (*Gadus morhua*) from Newfoundland destined for human consumption collected through citizen science methods. Mar. Pollut. Bull. **113**: 428–437. doi:10.1016/j.marpolbul.2016.10.043.

Liboiron, M., Melvin, J., Richard, N., Saturno, J., Ammendolia, J. Liboiron, F., et al. 2019. Low incidence of plastic ingestion among three fish species significant for human consumption on the island of Newfoundland, Canada. Mar. Pollut. Bull. **141**: 244–248. doi:10.1016/j.marpolbul.2019.02.057.

Liboiron, M., Zahara, A., Hawkins, K., Crespo, C., Neves, B.D., Wareham-Hayes, V., Edinger, E., Muise, C., Walzak, M.J. Sarazen, R., et al. 2021. Abundance and types of plastic pollution in surface waters in the Eastern Arctic (Inuit Nunangat) and the case for reconciliation science. Sci. Total Environ. **782**: 146809. doi:10.1016/j.scitotenv.2021.146809.

Loder, M.G.J., Imhof, H.K., Ladehoff, M., Loschel, L.A., Lorenz, C., Mintenig, S., Piehl, S., Primpke, S., Schrank, I. Laforsch, C., et al. 2017. Enzymatic purification of microplastics in environmental samples. Environ. Sci. Technol. **51**: 14283–14292. doi:10.1021/acs.est.7b03055.

Lusher, A. L. McHugh, M., and Thompson, R.C. 2013. Occurrence of microplastics in the gastrointestinal tract of pelagic and demersal fish from the English Channel. Marine pollution bulletin **67**: 94–9. doi:10.1016/j.marpolbul.2012.11.028

Lusher, A.L., Munno, K., Hermabessiere, L., and Carr, S. 2020. Isolation and extraction of microplastics from environmental samples: an evaluation of practical approaches and recommendations for further harmonization. Appl. Spectrosc. **74**: 1049–1065. doi:10.1177/0003702820938993.

Lusher, A.L., Tirelli, V., O'Connor, I., and Officer, R. 2015. Microplastics in arctic polar waters: the first reported values of particles in surface and sub-surface samples. Sci. Rep. **5**: 14947. doi:10.1038/srep14947.

Lusher, A.L., Welden, N.A., Sobral, P., and Cole, M. 2017. Sampling, isolating and identifying microplastics ingested by fish and invertebrates. Anal. Methods **9**: 1346–1360. doi:10.1039/C6AY02415G.

Magnusson, K., Jörundsdóttir, H., Norén, F., Lloyd, H., Talvitie, J., and Setälä, O. 2016. *Microlitter in Sewage Treatment Systems: A Nordic Perspective on Waste Water Treatment Plants as Pathways for Microscopic Anthropogenic Particles to Marine Systems*. Nordic Council of Ministers. Denmark, Finland, Iceland, Norway, Sweden, the Faroe Islands, Greenland and Åland 56.

Makhdoumi, P., Hossini, H., and Pirsaheb, M. 2022. A review of microplastic pollution in commercial fish for human consumption. Rev. Environ. Health. doi:10.1515/reveh-2021-0103.

Malinen, A. 2021. Microplastic ingestion by Atlantic mackerel and blue whiting in Icelandic waters. University of Akureyri Faculty of Business and Science University Centre of the Westfjords Master of Resource Management: Coastal and Marine Management Ísafjörður, Iceland Available from: https://skemman.is/bitstream/1946/39152/1/Anni%20Malinen.pdf.

Mani, T., Primpke, S., Lorenz, C., Gerdts, G., and Burkhardt-Holm, P. 2019. Microplastic pollution in benthic mid-stream sediments of the Rhine River. Environ. Sci. Technol. **53**: 6053–6062. doi:10.1021/acs.est.9b01363.

Martin, J., Granberg, M., Provencher, J.F., Liboiron, M., Pjogge, L., Magnusson, K., et al. In press. The power of multi-matrix monitoring in the pan-arctic region: plastics in water and sediment. Arct. Sci. doi:10.1139/AS-2021-0056.

Mattsson, K., Johnson, E.V., Malmendal, A., Linse, S., Hansson, L.A., and Cedervall, T. 2017. Brain damage and behavioural disorders in fish induced by plastic nanoparticles delivered through the food chain. Sci. Rep. **7**: 11452. doi:10.1038/s41598-017-10813-0.

McIlwraith, H.K., Kim, J., Helm, P., Bhavsar, S.P., Metzger, J.S., and Rochman, C.M. 2021. Evidence of microplastic translocation in wild-caught fish and implications for microplastic accumulation dynamics in food webs. Environ. Sci. Technol. **55**: 12372–12382. doi:10.1021/acs.est.1c02922.

McNeish, R.E., Kim, L.H., Barrett, H.A., Mason, S.A., Kelly, J.J., and Hoellein, T.J. 2018. Microplastic in riverine fish is connected to species traits. Sci. Rep. **8**: 11639. doi:10.1038/s41598-018-29980-9.

Melvin, J., Bury, M., Ammendolia, J., Mather, C., and Liboiron, M. 2021. Critical gaps in shoreline plastics pollution research. Front. Mar. Sci. **8**. doi:10.3389/fmars.2021.689108.

Miljøgifter i kystområdene (MILKYS). n.d. Environmental Agency Oslo, Norwayhttps://www.miljodirektoratet.no/ansvarsomrader/overvaking-arealplanlegging/miljoovervaking/overvakingsprogrammer/basisovervaking/miljogifter-langs-kysten/.

Mintenig, S.M., Int-Veen, I., Loder, M.G.J., Primpke, S., and Gerdts, G. 2017. Identification of microplastic in effluents of waste water treatment plants using focal plane array-based micro-Fourier-transform infrared imaging. Water Res. **108**: 365–372. doi:10.1016/j.watres.2016.11.015.

Moore, R.C., Loseto, L., Noel, M., Etemadifar, A., Brewster, J.D. MacPhee, S., et al. 2020. Microplastics in beluga whales (*Delphinapterus leucas*) from the Eastern Beaufort Sea. Mar. Pollut. Bull. **150**: 110723. doi:10.1016/j.marpolbul.2019.110723 .

Moore, R.C., Noel, M., Etemadifar, A., Loseto, L., Posacka, A.M., Bendell, L., and Ross, P.S. 2022. Microplastics in beluga whale (*Delphinapterus leucas*) prey: an exploratory assessment of trophic transfer in the Beaufort Sea. Sci. Total Environ. **806**: 150201. doi:10.1016/j.scitotenv.2021.150201.

Morgana, S., Ghigliotti, L., Estevez-Calvar, N., Stifanese, R., Wieckzorek, A. Doyle, T., et al. 2018. Microplastics in the Arctic: a case study with sub-surface water and fish samples off Northeast Greenland. Environ. Pollut. **242**: 1078–1086. doi:10.1016/j.envpol.2018.08.001.

Nashoug, B.F. 2017. "Sources of marine litter" –Workshop report, Svalbard 4th–6th September 2016. Sustainable coastal development (SALT) Svolvær, Norway

Nielsen, J., Hedeholm, R.B., Simon, M., and Steffensen, J.F. 2014. Distribution and feeding ecology of the Greenland shark (*Somniosus microcephalus*) in Greenland waters. Polar Biol. **37**: 37–46. doi:10.1007/s00300-013-1408-3.

OSPAR request on development of a common monitoring protocol for plastic particles in fish stomachs and selected shellfish on the basis of existing fish disease surveys. ICES Special Request Advice Northeast Atlantic and Arctic Ocean. 2015 ICES Copenhagen, Denmark

PAME. 2019. Desktop study on marine litter including microplastics in the Arctic, 11th Arctic Council Ministerial Meeting Rovaniemi, Finland. Akureyri, Iceland. Available from: https://pame.is/index.php/document-library/pame-reports-new/pame-ministerial-deliverables/2019-11th-arctic-council-ministerial-meeting-rovaniemi-finland/423-desktop-study-on-marine-litter-including-microplastics-in-the-arctic/file.

Peda, C., Caccamo, L., Fossi, M.C., Gai, F., Andaloro, F. Genovese, L., et al. 2016. Intestinal alterations in European sea bass *Dicentrarchus labrax* (Linnaeus, 1758) exposed to microplastics: preliminary results. Environ Pollut. **212**: 251–256. doi:10.1016/j.envpol.2016.01.083.

Peeken, I., Primpke, S., Beyer, B., Gutermann, J., Katlein, C. Krumpen, T., et al. 2018. Arctic sea ice is an important temporal sink and means of transport for microplastic. Nat. Commun. **9**: 1505. doi:10.1038/s41467-018-03825-5.

Poon, F.E., Provencher, J.F., Mallory, M.L., Braune, B.M., and Smith, P.A. 2017. Levels of ingested debris vary across species in Canadian Arctic seabirds. Mar. Pollut. Bull. **116**: 517–520. doi:10.1016/j.marpolbul.2016.11.051.

Prata, J.C., da Costa, J.P., Lopes, I., Duarte, A.C., and Rocha-Santos, T. 2020. Environmental exposure to microplastics: an overview on possible human health effects. Sci. Total Environ. **702**: 134455. doi:10.1016/j.scitotenv.2019.134455.

Prata, J.C., Reis, V., da Costa, J.P., Mouneyrac, C., Duarte, A.C., and Rocha-Santos, T. 2021. Contamination issues as a challenge in quality control and quality assurance in microplastics analytics. J. Hazard. Mater. **403**: 123660. doi:10.1016/j.jhazmat.2020.123660.

Primpke, S., Booth, AM., Gerdts, G., Gomiero, A., Kögel, T. Lusher, A., et al. In press. Monitoring of microplastic pollution in the Arctic: recent developments in polymer identification, quality assurance and control (QA/QC), and data reporting. Arct. Sci. doi:10.1139/AS-2022-0006.

Primpke, S., Lorenz, C., Rascher-Friesenhausen, R., and Gerdts, G. 2017. An automated approach for microplastics analysis using focal plane array (FPA) FTIR microscopy and image analysis. Anal. Methods **9**: 1499–1511. doi:10.1039/C6AY02476A.

Provencher, J.F., Vermaire, J.C., Avery-Gomm, S., Braune, B.M., and Mallory, M.L. 2018. Garbage in guano? Microplastic debris found in faecal precursors of seabirds known to ingest plastics. Sci. Total Environ. **644**: 1477–1484. doi:10.1016/j.scitotenv.2018.07.101.

Rist, S., Vianello, A., Winding, M.H.S., Nielsen, T.G., Almeda, R., Torres, R.R., and Vollertsen, J. 2020. Quantification of plankton-sized microplastics in a productive coastal Arctic marine ecosystem. Environ. Pollut. **266**: 115248. doi:10.1016/j.envpol.2020.115248.

Rochman, C.M., Kurobe, T., Flores, I., and Teh, S.J. 2014. Early warning signs of endocrine disruption in adult fish from the ingestion of polyethylene with and without sorbed chemical pollutants from the marine environment. Sci. Total Environ. **493**: 656–661. doi:10.1016/j.scitotenv.2014.06.051.

Rochman, C.M., Parnis, J.M., Browne, M.A., Serrato, S., Reiner, E.J. Robson, M., et al. 2017. Direct and indirect effects of different types of microplastics on freshwater prey (*Corbicula fluminea*) and their predator (*Acipenser transmontanus*). PLoS ONE **12**: e0187664. doi:10.1371/journal.pone.0187664.

Saturno, J., Liboiron, M., Ammendolia, J., Healey, N., Earles, E. Duman, N., et al. 2020. Occurrence of plastics ingested by Atlantic cod (*Gadus morhua*) destined for human consumption (Fogo Island, Newfoundland and Labrador). Mar. Pollut. Bull. **153**: 110993. doi:10.1016/j.marpolbul.2020.110993.

Savoca, M.S., McInturf, A.G., and Hazen, E.L. 2021. Plastic ingestion by marine fish is widespread and increasing. Global Change Biol. **27**: 2188–2199. doi:10.1111/gcb.15533.

Selvam, S., Manisha, A., Roy, P.D., Venkatramanan, S., Chung, S.Y. Muthukumar, P., et al. 2021. Microplastics and trace metals in fish species of the Gulf of Mannar (Indian Ocean) and evaluation of human health. Environ. Pollut. **291**: 118089. doi:10.1016/j.envpol.2021.118089 .

Simon, M., van Alst, N., and Vollertsen, J. 2018. Quantification of microplastic mass and removal rates at wastewater treatment plants applying focal plane array (FPA)-based Fourier Transform Infrared (FT-IR) imaging. Water Res. **142**: 1–9. doi:10.1016/j.watres.2018.05.019.

Smith, M., Love, D.C., Rochman, C.M., and Neff, R.A. 2018. Microplastics in Seafood and the implications for Human Health. Curr. Environ. Health Rep. **5**: 375–386. doi:10.1007/s40572-018-0206-z.

Strand, K.O., Huserbraten, M., Dagestad, K.F., Mauritzen, C., Grøsvik, B.E. Nogueira, L.A., et al. 2021. Potential sources of marine plastic from survey beaches in the Arctic and Northeast Atlantic. Sci. Total Environ. **790**: 148009. doi:10.1016/j.scitotenv.2021.148009.

Sussmann, J., Krause, T., Martin, D., Walz, E., Greiner, R. Rohn, S., et al. 2021. Evaluation and optimisation of sample preparation protocols suitable for the analysis of plastic particles present in seafood. Food Control **125**: 107969. doi:10.1016/j.foodcont.2021.107969.

Ter Halle, A., Jeanneau, L., Martignac, M., Jarde, E., Pedrono, B., Brach, L., and Gigault, J. 2017. Nanoplastic in the North Atlantic subtropical gyre. Environ. Sci. Technol. **51**: 13689–13697. doi:10.1021/acs.est.7b03667.

Thiele, C.J., Hudson, M.D., and Russell, A.E. 2019. Evaluation of existing methods to extract microplastics from bivalve tissue: adapted KOH digestion protocol improves filtration at single-digit pore size. Mar. Pollut. Bull. **142**: 384–393. doi:10.1016/j.marpolbul.2019.03.003.

Thiele, C.J., Hudson, M.D., Russell, A.E., Saluveer, M., and Sidaoui-Haddad, G. 2021. Microplastics in fish and fishmeal: an emerging environmental challenge? Sci. Rep. **11**: 2045. doi:10.1038/s41598-021-81499-8.

Trevail, A.M., Gabrielsen, G.W., Kuhn, S., and Van Franeker, J.A. 2015. Elevated levels of ingested plastic in a high Arctic seabird, the northern fulmar (Fulmarus glacialis). Polar Biol. **38**: 975–981. doi:10.1007/s00300-015-1657-4.

UNEP. 2009. *Marine Litter: a Global Challenge*. United Nations Environment Programme (UNEP), Nairobi. Available from: https://wedocs.unep.org/bitstream/handle/20.500.11822/10744/MarineLitterAglobalChallenge.pdf?sequence=1&isAllowed=y.

van Franeker, J.A. 2011. Reshape and relocate: seabirds as transformers and transporters of microplastics. Fifth International Marine Debris Conference, Honolulu, Hawaii.

van Mourik, L.M., Crum, S., Martinez-Frances, E., van Bavel, B., Leslie, H.A., de Boer, J., and Cofino, W.P. 2021. Results of WEPAL-QUASIMEME/NORMANs first global interlaboratory study on microplastics reveal urgent need for harmonization. Sci. Total Environ. **772**: 145071. doi:10.1016/j.scitotenv.2021.145071.

von Friesen, L.W., Granberg, M.E., Hassellov, M., Gabrielsen, G.W., and Magnusson, K. 2019. An efficient and gentle enzymatic digestion protocol for the extraction of microplastics from bivalve tissue. Mar. Pollut. Bull. **142**: 129–134. doi:10.1016/j.marpolbul.2019.03.016.

von Friesen, L.W., Granberg, M.E., Pavlova, O., Magnusson, K., Hassellov, M., and Gabrielsen, G.W. 2020. Summer sea ice melt and wastewater are important local sources of microlitter to Svalbard waters. Environ. Int. **139**: 105511. doi:10.1016/j.envint.2020.105511.

Wang, W.F., Ge, J., and Yu, X.Y. 2020. Bioavailability and toxicity of microplastics to fish species: a review. Ecotoxicol. Environ. Saf. **189**: 109913. doi:10.1016/j.ecoenv.2019.109913.

Welden, N.A.C., and Cowie, P.R. 2016. Long-term microplastic retention causes reduced body condition in the langoustine, Nephrops norvegicus. Environ. Pollut. **218**: 895–900. doi:10.1016/j.envpol.2016.08.020.

Wesch, C., Elert, A.M., Worner, M., Braun, U., Klein, R., and Paulus, M. 2017. Assuring quality in microplastic monitoring: about the value of clean-air devices as essentials for verified data. Sci. Rep. **7**: 5424. doi:10.1038/s41598-017-05838-4.

Wichmann, D., Delandmeter, P., and van Sebille, E. 2019. Influence of near-surface currents on the global dispersal of marine microplastic. J. Geophys. Res. Oceans **124**: 6086–6096. doi:10.1029/2019JC015328.