Science of the Total Environment 792 (2021) 148308

Contents lists available at ScienceDirect

Science of the Total Environment

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

# Understanding the occurrence and fate of microplastics in coastal Arctic ecosystems: The case of surface waters, sediments and walrus (*Odobenus rosmarus*)

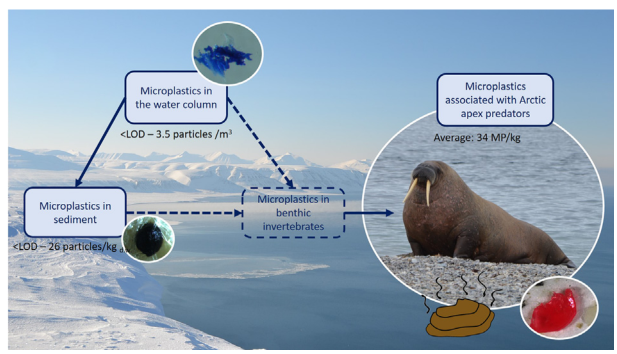
Pernilla Carlsson [a,b], Cecilie Singdahl-Larsen [c], Amy L. Lusher [c,d,*]

[a] *Norwegian Institute for Water Research (NIVA), Framsenteret, Hjalmar Johansens gate 14, 9007 Tromsø, Norway*
[b] *University Centre of the Westfjords, Suðurgata 12, 400 Ísafjörður, Iceland*
[c] *Norwegian Institute for Water Research (NIVA), Gaustadalléen 21, NO-0349, Oslo, Norway*
[d] *Department of Biological Sciences, University of Bergen, 5020 Bergen, Norway*

## HIGHLIGHTS

- Water samples showed no difference in microplastic concentration between fjords.
- Sediment particle concentration revealed similar levels to other Arctic studies.
- No difference in particle concentrations between populated and remote Svalbard fjords
- Non-invasive method to investigate interaction of marine mammals with microplastics
- First observation of microplastics in walrus faeces averaging 34 particles/kg

## GRAPHICAL ABSTRACT




## ARTICLE INFO

*Article history:*
Received 30 March 2021
Received in revised form 23 May 2021
Accepted 3 June 2021
Available online 5 June 2021

Editor: Kevin V. Thomas

*Keywords:*
Plastic pollution
Svalbard
FT-IR
Marine mammals
Manta net

## ABSTRACT

The Arctic ecosystem receives contaminants transported through complex environmental pathways – such as atmospheric, riverine and oceanographic transport, as well as local infrastructure. A holistic approach is required to assess the impact that plastic pollution may have on the Arctic, especially with regard to the unseen microplastics. This study presents data on microplastics in the Arctic fjords of western Svalbard, by addressing the ecological consequences of their presence in coastal surface waters and sediment, and through non-invasive approaches by sampling faeces from an apex predator, the benthic feeder walrus (*Odobenus rosmarus*). Sample locations were chosen to represent coastal areas with different degrees of anthropogenic pollution and geographical features (e.g., varying glacial coverage of catchment area, winter ice cover, traffic, visitors), while also relevant feeding grounds for walrus. Microplastics in surface water and sediments ranged between <LOD (limit of detection)-3.5 particles/m$^3$ and <LOD-26 particles/kg dry weight, respectively. This study shows that microplastics may also enter the Arctic food web as the microplastic concentration in walrus faeces were estimated at an average of 34 particles/kg. Polyester was identified by Fourier transformation infrared spectroscopy (FT-IR) as the most common plastic polymer (58% in water, 31% in walrus), while fibres were the most common shape (65% water, 71% in sediment, 70% walrus). There was no significant difference in microplastic occurrence between water samples from populated or remote fjords, suggesting that microplastics are a ubiquitous contaminant which is available for interaction with Arctic marine animals even at distances from settlements. The present study

* Corresponding author at: Norwegian Institute for Water Research (NIVA), Gaustadalléen 21, NO-0349, Oslo, Norway.
*E-mail address:* amy.lusher@niva.no (A.L. Lusher).

https://doi.org/10.1016/j.scitotenv.2021.148308

contributes to our understanding of microplastics in the remote Arctic ecosystem. It also identifies the potential of non-invasive sampling methods for investigating Arctic pinnipeds. This approach will need further development and standardisation before utilisation to monitor plastic pollution in other marine mammals.


## 1. Introduction

The Arctic – once regarded as a pristine environment – has succumbed to increasing anthropogenic pressures such as legacy contaminants, climate driven changes and anthropogenic impact (AMAP, 2016, 2017). In recent years, many researchers have begun to investigate the sources and quantities of plastics on Svalbard. It appears that almost all investigated locations in the Arctic have displayed some level of contamination from plastics including shorelines, the deep-sea floor, surface waters and biota (Halsband and Herzke, 2019; Tirelli et al., 2020).

The presence of plastic pollution has been posited to be linked to local and distant sources, with distribution driven by a complex system of anthropogenic and environmental processes. Multiple entry routes of plastic to the Arctic environment have been highlighted including routes directly related to human activity and settlements – such as wastewater, dumping and equipment losses. Other entry routes include environmental processes related to atmospheric, riverine and oceanic transport (PAME, 2020 and references therein). Despite several studies investigating pathways of microplastics (<5 mm) into the Arctic environment, it is difficult to understand the mechanisms of transport and fate. Harmonisation of analyses and improvement of data presentation, which includes larger data sets with quality assurances have gained increased focus amongst scientists (Brander et al., 2020; Cowger et al., 2020). There is a need for comparable data with standardised quality assurance to allow us to identify sources and at-risk areas, to develop mitigation strategies, and implement science-based management (Lusher et al., 2020; von Friesen et al., 2020).

Currently, efforts towards understanding plastic and microplastic pollution in the Arctic suggest that the quantities of floating plastic fragments observed in the Barents Sea may correspond to a "plastic conveyor belt" associated with the global thermohaline circulation (Cózar et al., 2017), and those identified in the Kara Sea are driven by riverine input (Yakushev et al., 2021). However, local point sources should not be ruled out, as has been shown in samples collected in the vicinity of wastewater treatment plants in Ny-Ålesund (von Friesen et al., 2020). Prior to 2007, waste generated in the main settlement on Svalbard (Longyearbyen, 2400 habitants) was buried or burnt. After 2007, almost all waste has been shipped to the Norwegian mainland. Wastewater from Longyearbyen and the Russian settlement, Barentsburg (470 habitants), is not treated before it is release into the marine environment. Other local sources may be connected to dumping sites, shipping, fishing, as well as land and ocean-based industries (Granberg et al., 2019). Svalbard is also receiving an increased number of tourists each year and the current infrastructure may not have suitable waste handling facilities to prevent contamination (Granberg et al., 2019). Marine litter on beaches are dominated by fishing gear and packaging materials (OSPAR, 2021).

Microplastics have been found in both coastal and offshore environments of Svalbard (e.g., Kanhai et al., 2020; Lusher et al., 2015) as well as in several other areas of the Arctic Ocean, including Arctic and sub-Arctic biota (PAME, 2020; Tirelli et al., 2020). Samples obtained closer to the coast tend to have elevated microplastic concentrations, especially with regard to proximity to urban settlements and wastewater facilities (von Friesen et al., 2020). Microplastics have also been reported in sea ice, and it has been suggested that the annual formation of ice could contribute to the flux of plastics in and around the Arctic regions (Kanhai et al., 2020; Obbard et al., 2014; Peeken et al., 2018). In addition to these sources, atmospheric transportation of microplastics into the Arctic has recently been suggested as a potential transport mechanism. This has barely been investigated, and the size and importance of atmospheric transport in addition to direct human activity, rivers and long-range oceanic transport is not known (Evangeliou et al., 2020; Zhang et al., 2020).

The presence of microplastics in the environment raises concern for the Arctic ecosystem and it is critical to understand the sources, distribution and fate of these particles, including the consequences on Arctic species (Tirelli et al., 2020). Before studies into the consequences are initiated it is first necessary to determine whether microplastics can be identified in trophic systems. Studies have shown that animals from the Arctic and sub-Arctic are ingesting microplastics, including fish (Bråte et al., 2016; de Vries et al., 2020; Kühn et al., 2018), bivalves (Bråte et al., 2020) and birds (Bourdages et al., 2021; Hamilton et al., 2021). Marine mammals are exposed to microplastics through the ingestion of prey, whether passively through filter feeding, or through what is termed "trophic transfer" by which a prey species is pre-exposed to, and therefore contains microplastics, and is subsequently ingested by a predator (Panti et al., 2019). As marine mammals do not drink seawater, this is not assumed as a major exposure pathway when compared to feeding.

Data on microplastics in marine mammals comes mostly from temperate studies and as observations following strandings and bycatch post-mortems, as well as faecal pellet analysis (reviewed in Zantis et al., 2021). Unfortunately, there is limited evidence of the consequences for the apex predators in the Arctic, like the walrus (*Odobenus rosmarus*). Walrus are predators which feed on benthic primary consumers - predominantly the clam *Mya truncata*. Walrus feed in shallow areas - and have a shorter trophic chain than other Arctic marine mammals (Gjertz and Wiig, 1992; Norwegian Polar Institute, 2021). Since wind and waves may transport microplastics from the upper surface water towards the benthic zone, shallow benthic feeders may be exposed to microplastics with a lower density than sea water. Clams filter particulate matter from the water column which may include microplastics. There is already abundant literature where bivalves collected from sub-littoral areas contain microplastics following water filtration (Bråte et al., 2020). This suggests that if microplastics are within clams when they are preyed upon, trophic transfer could facilitate uptake to the walrus. The shallower feeding and the higher potential of water mixing (winds, waves), which exposes filter feeders to microplastics, suggests that walrus could be exposed through trophic transfer. Furthermore, as walrus feed at the interface between water and sediment, any microplastics which settle on the surface of sediments could also be ingested.

The purpose of the present study was to investigate the effects of microplastics in Arctic fjord ecosystems by (1) quantifying the presence of microplastics in coastal surface waters and sediments, and (2) addressing transfer through the food web to top predators using non-invasive approaches. The secondary aim of the study was to investigate whether differences in microplastic concentration could be related to sources of pollution or environmental characteristics. It was hypothesised that microplastics would be present at higher levels closer to the populated areas than remote fjords. We further hypothesised that water mixing in shallow coastal waters will facilitate microplastics transfer to walrus feeding on benthic invertebrates.

## 2. Materials and methods

### 2.1. Site selection

Svalbard's coastline is characterised by several fjords with glacial input, with around 60% of the archipelago covered by glaciers and snow caps. The Isfjorden system and St. Jonsfjorden, located on west coast of Svalbard, were chosen as the study areas since they contain fjords of comparable size and catchment areas, and have populated and less visited areas within practical distances (Fig. 1). The Isfjorden system consists of Grønfjorden and Adventfjorden in the south. These fjords contain the settlements Barentsburg and Longyearbyen, respectively. Longyearbyen is the Norwegian settlement with a population of around 2400 individuals, while Barentsburg has 455 habitants (Statistics Norway, 2021). The northern fjord arms of Isfjorden consist of Dicksonfjorden, Ekmanfjorden and Nordfjorden. These sites were chosen as remote sites with less summer and winter visitors than the other fjords within Isfjorden. Ekmanfjorden has glaciers draining directly into one bay of the fjord and a river delta from land-based glaciers in the other bay. Dicksonfjorden has two larger river deltas, with land-based glaciers in one, and smaller snow caps and glaciers in the catchment area of the other delta. The river in Oxaasdalen, drains the small glacier on Bollen (756 m above sea level) and enters Dicksonfjorden south of the confluence of the two rivers Huginelva and Nathorstelva. Nordfjorden is situated between the main open part of Isfjorden and the entrance of Ekman- and Dicksonfjorden.

St. Jonsfjorden is situated on the west coast of Svalbard. This fjord was included in the present study as it has a very large glacier-covered catchment area, with few visitors. It is also one of the closest fjords to the resident walrus (*Odobenus rosmarus*) colony on the island of Prins Karls Forland. Thus, St. Jonsfjorden was better suited for water sampling to avoid disturbing the resident walrus. Finally, Poolpynten, located on Prins Karls Forland, to the west of St. Jonsfjorden was chosen for the collection of walrus faecal samples. Walrus are often observed very close to shore in areas shallower than 100 m (90% occurrence), occupying shallow water where molluscs occur (Freitas et al., 2009; Gjertz and Wiig, 1992; Hamilton et al., 2015). At Prins Karls Forland, 57% of the observations of walrus have been in areas with water no deeper than 5 m (Kovacs et al., 2014). Due to logistical challenges (need for larger ship or divers), clams and sediment were not collected from this site. Walrus faeces were possible to collect without disturbance to the colony and hence used in this study.

All sampling was conducted between June and September 2018 (Table 1). A full list of sample types and replicates are presented in the Supplementary Material (Table S1). Isfjorden along with St. Jonsfjorden and Forlandssundet – which are situated between Prins Karls Forland and Spitsbergen – are impacted by the West Spitsbergen Current. This current system drives water into the southern side of Isfjorden which then circulates northwards before exiting on the northern coast of Isfjorden (Nilsen et al., 2016).

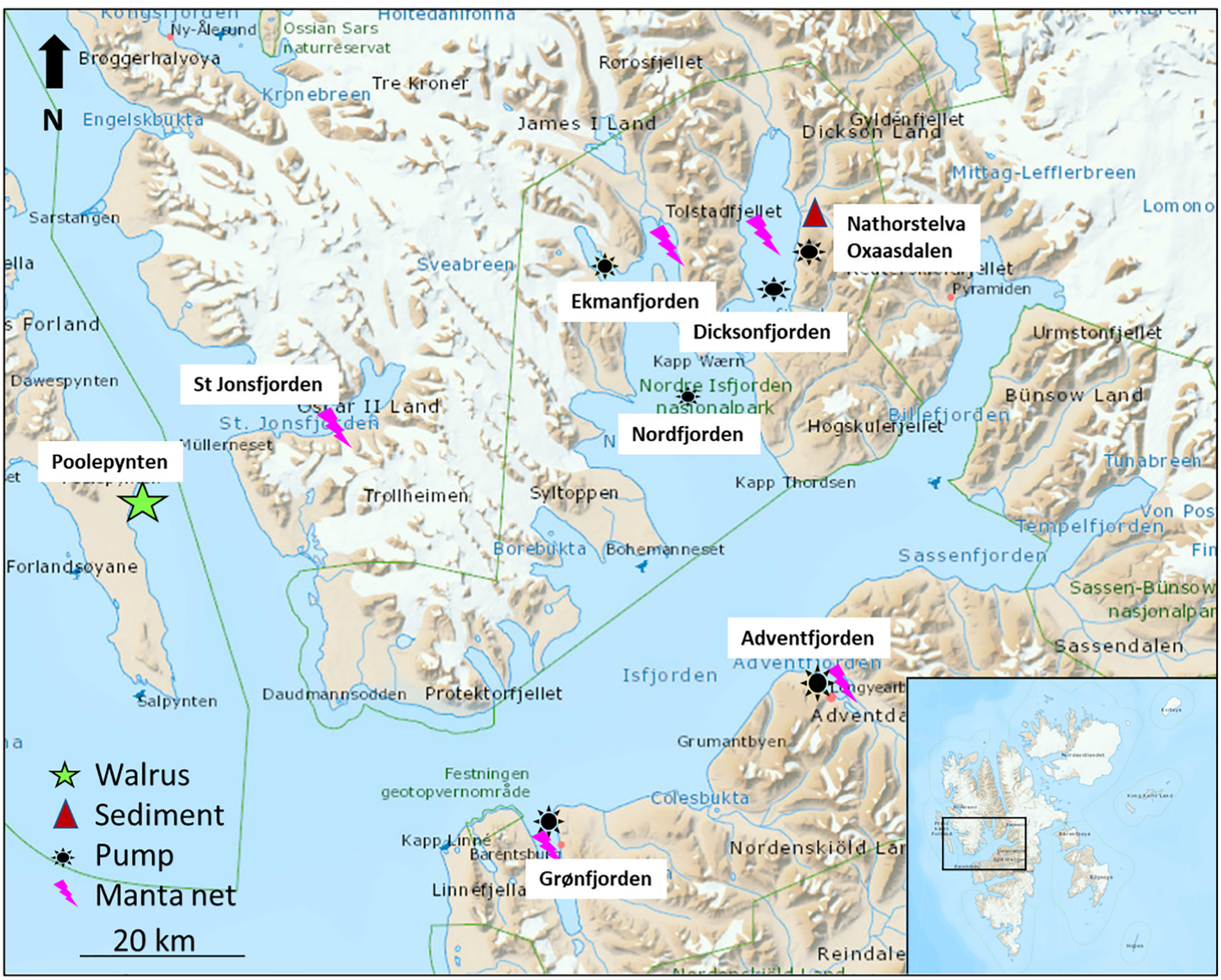



**Fig. 1.** Map of sampling locations during the summer/autumn sampling campaigns in Isfjorden (Grøn-, Advent-, Nord-, Dickson-, Ekman-) and St. Jonsfjorden. The icons represent the different samples collected (walrus, sediment, and water: pump, manta). Map from the Norwegian Polar Institute.

**Table 1**
Samples collected from the western fjord of Svalbard using the manta net and pump (water) and sediment and walrus faeces during the summer/autumn period 2018.

| Site (fjord) and sample type | Replicates | Duration of sampling (minutes, range) | Estimated volume (range) of water ($m^3$)[a] | Sampling date |
|---|---|---|---|---|
| **Manta samples** | | | | |
| Ekmanfjorden | 4 | 40 | 120–130 | 31.08.2018 |
| Dicksonfjorden | 2 | 20–40 | 80–170 | 01.09.2018 |
| Grønfjorden | 3 | 30 | 100–110 | 02.09.2018 |
| Adventfjorden | 5 | 10–20 | 30–70 | 02.09.2018 |
| St. Jonsfjorden | 5 | 15 | 60 | 29.06.2018 |
| **Pump samples** | | | | |
| Ekmanfjorden | 1 | 70 | 2.8 | 31.08.2018 |
| Dicksonfjorden | 1 | 70 | 3.0 | 01.09.2018 |
| Nordfjorden | 1 | 60 | 2.6 | 01.09.2018 |
| Grønfjorden | 1 | 67 | 2.7 | 02.09.2018 |
| Adventfjorden | 1 | 70 | 2.8 | 02.09.2018 |
| Oxaasdalen | 1 | 70 | 3.0 | 01.09.2018 |
| **Sediment samples** | | | | |
| Huginelva/Nathorstelva estuary zone, Dicksonfjorden | 6 | n.r. | 200 g wet sediment | 01.09.2018 |
| **Walrus faeces** | | | | |
| Poolepynten, Prins Karls Forland | 8 | n.r. | 100 g faeces | 30.06.2018 |

[a] Amount of water (average) is given as range per site, estimated from speed of boat, distance sailed and opening of the manta net. Flow rate was measured for pump samples.

### 2.2. Water sample collection and processing

Water sampling was conducted in Adventfjorden, Dicksonfjorden, Ekmanfjorden, Grønfjorden, Nordfjorden and St. Jonsfjorden. Two methods of water collection were performed. Firstly, sea surface water from all fjords, with the exception of Nordfjorden (Table 1), were sampled using a manta net with a metal framed opening ($0.61 \times 0.16$ m) and a cod end net of 333 μm mesh. The net was carefully rinsed with water from the fjord from the outside of the net to avoid cross-sample contamination before and between each sample. Sampling was carried out between two and five times in each fjord depending on weather conditions (Table 1), and the net was towed from 10 to 40 min at 1–2 knots. Each sample collected in the cod-end of the net was transferred into pre-cleaned glass jars, and any visible particles found in the net were included in the sample. Jars were stored and transported cold to laboratory facilities where they were stored frozen (−20 °C) until analysis. Weather and sea conditions were recorded. The volume of water filtered was calculated using vessel speed and tow duration.

For the second sampling method, a stainless-steel stacked filter system (500, 300, 100 μm, KC Research Equipment, Denmark) was used to sample seawater in all fjords, with the exception of St. Jonsfjorden (Fig. 1, Table 1). All components (metal filters, the pump and rubber hoses) were previously rinsed with pre-filtered tap water, and metal filters were wrapped in tin foil while the rest of material was packed in hermetic boxes to avoid any external contamination during transport. Sea surface water was sampled while drifting in 1–2 knots (Table 1). Weather and sea conditions were recorded, although sampling was carried out when optimal conditions occurred (dry, little wind). One sample with 2–3 filters were collected per fjord in addition to one sample from the river in Oxaasdalen which drains into Dicksonfjorden.

Water samples were defrosted over night at room temperature before processing. Each manta sample was flushed through a sieve (100 μm) and the retained material was rinsed into a gravimetric cylinder to which high-saline filtered water (100 g NaCl per 200 mL water) was added. The pump samples were rinsed directly into gravimetric cylinders with the high-saline filtered water. Each sample (irrespective of sampling method) was left covered to separate for 24 h, after which the floating fraction of water was siphoned off before further clean-up and analysis. As several of the samples contained organic matter, they required a further clean-up step with 10% potassium hydroxide (KOH) solution. The solution was prepared following standard procedures with filtered water and re-filtering after preparation (Bråte et al., 2018). The solution was added to each sample with a 3:1 (*v*/v) ratio. Each sample was shaken before heating at 50 °C for at least 24 h with periodic shaking. Once digestion was complete each sample was filtered under vacuum onto GF/F (pore size: 0.7 μm, Ø: 47 mm) or GF/A (pore size: 1.6 μm, Ø: 47 mm) Whatman filters. Filters were sealed in a petri dish until analysis.

The project had a secondary objective to test the application of different methodological approaches for water sample collection, the results of which are presented in the Supplementary Material (Table S1 and Fig. S2).

### 2.3. Sediment sample collection and processing

The Huginelva/Nathorstelva estuary zone in Dicksonfjorden was selected to collect sediment samples for microplastic analysis. This area is important because the river discharges through this estuary, and it is an area relatively close to walrus feeding grounds. A total of six sediment samples of 200 g were collected at low tide using metal spoons, wrapped in tin foil and stored in plastic bags frozen (−20 °C) until analysis.

Sediment samples were defrosted overnight at 40 °C before processing. Three sub-samples were taken from each sediment sample, each weighing around 30 g wet weight. These samples were then dried in the oven at 60 °C overnight and re-weighed once dry. Each sediment sample underwent gravity separation in a similar way to the water samples (density separation with NaCl), and the liquid fraction was siphoned off before a second gravity separation was performed. Both fractions were filtered under vacuum onto GF/F (pore size: 0.7 μm, Ø: 47 mm) and combined as one sample. Filters were sealed in a petri dish until analysis.

### 2.4. Walrus faecal sample collection and processing

Due to economical and logistical reasons, it was not possible to collect benthic clams. Hence, walrus faeces were chosen to represent benthic feeders. Walrus faecal samples were collected from the shoreline haul-out at Poolepynten after recent defecation. The walrus colony at Poolepynten is known to be feeding on clams (*Mya truncata*) in the shallow areas east of Prins Karls Forland, close to their haul-out area. At the time of sampling the walrus colony consisted of about 15 individuals. Faecal samples were identified at a recent, but unoccupied, haul-out location. The walrus were at a second haul-out location >50 m away. Faecal samples were collected with single-use plastic bags previously checked for microplastic contamination and stored frozen (−20 °C) for further analysis. Some of the faecal samples were separated by >10 m suggesting that they may have come from different individuals.

Faecal samples were processed following a protocol available in the literature (Lusher and Hernandez-Milian, 2018), with some modification. Subsamples of ~100 g were obtained from the middle of each sample for microplastic analysis. This was to avoid any potential contamination from sampling or settling of airborne fibres to the exposed surfaces of the faeces. The remaining material was stored for other studies. Samples were rinsed under running filtered water through a nested sieve stack (1 mm, 500 μm, 25 μm). The obtained material was separated between dietary items and any visible plastics. Full dietary analysis was not the purpose of this study, identifiable items were noted but no further research was conducted on the other items. Any material remaining on the sieves were processed through density separation ($CaCl_2$) and digestion (KOH 10%). Samples were filtered under vacuum onto GF/F (pore size: 0.7 μm, Ø: 47 mm) Whatman filters. Filters were sealed in petri dishes until analysis.

### 2.5. Microplastic identification

All samples were visually inspected under two types of stereo microscope (M205 C Leica, <16×/Nikon SMZ745T, 20× magnification), measured (at their longest, length and shortest, width) and photographed (using Infinity 1-3C/INFINITY 1 Lumenera camera and INFINITY ANALYZE and CAPTURE software). Visual identification followed the methods and standards presented in Lusher et al. (2020) regarding microplastics categorization by shape, size and colour. Visual identification was supported by Fourier Transform Infrared spectroscopy (FT-IR; Cary 630, Agilent/Microscope Spotlight 400, PerkinElmer) to determine the type of plastics recovered. Some of the particles were too thin to obtain viable FT-IR spectra for identification and hence, FT-IR was applied in various amounts across sample types and sites. Further details can be found in the Supplementary Material. Results are reported as the total amount of particles detected (visually) but focus on particles confirmed through FT-IR as microplastic. A particle length of >100 μm was set as lowest identifiable length to correspond to the mesh sizes applied in the methods. Therefore, our limit of detection (LOD) is between 5 mm–100 μm. There were 15 particles that were longer (<11.7 mm) than the size definition of microplastics used in the present study (5 mm) and therefore excluded from further analysis. Microplastics were reported per unit volume ($m^3$) and per unit weight (kg).

### 2.6. Quality assurance and quality control

All samples were processed under controlled conditions in laboratories at both Akvaplan-niva (Tromsø, Norway) and the Norwegian Institute of Water Research (Oslo, Norway). To prevent procedural contamination in both laboratories, all surfaces were thoroughly cleaned with pre-filtered alcohol before rinsing with filtered water. All equipment was cleaned with filtered tap or milliQ water (33 μm/0.22 μm).

In total, 38 blanks were collected from the laboratory (air, liquids, equipment, $n = 33$) and in the field ($n = 5$) to understand procedural contamination. Two field blanks were collected by rinsing the manta net and the pump system, respectively with pre-filtered (5 μm) seawater. These were treated in the same way as the samples. Field blanks for the walrus samples consisted of unopened plastic bags (n = 3). Air blanks were not possible to collect during field work, but procedural blanks were collected for each batch of samples in the laboratory. If microplastics were detected in blanks, similar particles (shape, colour, polymer - if confirmed) that were observed in corresponding samples were subtracted from the counts.

To minimise contamination sources in the field, plastic materials in clothes and equipment were avoided to the largest practical extent and samples were handled up-wind from people as long as feasible. Only cotton clothing was worn during laboratory work.

### 2.7. Statistical analysis and data treatment

Due to the small sampling size and types of sampling carried out, it was not feasible to conduct large statistical tests. Furthermore, the uneven application of FT-IR across the samples limits the comparison. Therefore, data analysis was performed on the visual data only. Mann Whitney's and Kruskal Wallis' test were used for statistical significance tests on water samples where there were replicates in all fjords since they can be applied on non-normal distributed data. Manta net samples were then grouped as fjords with similar characteristics to increase replicates/group, Dickson-Ekmanfjorden (remote) were combined as well as Grøn- and Adventfjorden (populated), while St. Jonsfjorden (outside Isfjorden) remained separate, although combined with Ekman- and Dicksonfjorden for the Mann Whitney's test. Descriptions of particle characteristics refer to the data obtained through visual analysis (e.g. shape, size, colour) whereas the polymer differentiation refers to the data collected through FT-IR.

## 3. Results and discussion

Microplastic particles were found in many of the blanks, ranging from 1 to 3 particles/blank, which were primarily fibres (78%). The field blank from the pump contained one blue fibre corresponding to the 500 μm filter. The field blank for the manta net included one black fibre, while the laboratory blanks contained mainly black and blue fibres (0–3 particles/blank processed parallel with each sample batch). No additional particles were recorded in the blank sample bags used during sampling at the walrus haul-out site. All data presented was therefore corrected for the blank values.

### 3.1. Microplastics in water samples from the fjords

Microplastics (visually identified) were found in 79% of all fjord samples using the manta net (ranging <LOD-0.27, average; 0.06 particles/$m^3$) and in 80% of the pump samples (<LOD-3.5, average 1.4 particles/$m^3$). Fig. 2 presents the data for manta net samples collected from all fjords. Advent- and St. Jonsfjorden presented the highest quantities of microplastics, but St. Jonsfjorden was significantly higher compared to the populated fjords (Adventfjorden and Grønfjorden) and the other remote fjords Ekman- and Dicksonfjorden (Kruskal-Wallis test, $p = 0.005$). There was no significant difference between the populated fjords (Adventfjorden and Grønfjorden) compared to remote fjords (St. Jons-, Ekman-, Dickson- and Nordfjorden) (Mann-Whitney test, $p = 1$). Shape composition in manta nets within the fjords (with replicates combined) showed that fibres (25–79%) were more common than beads (0–75%) and fragments (0–40%), although there were differences between sample types (Fig. S1).

Pump samples presented similar data to the manta nets. There was no difference between the two methods applied (see Supplementary Materials and Fig. S2 for more information), although some interesting

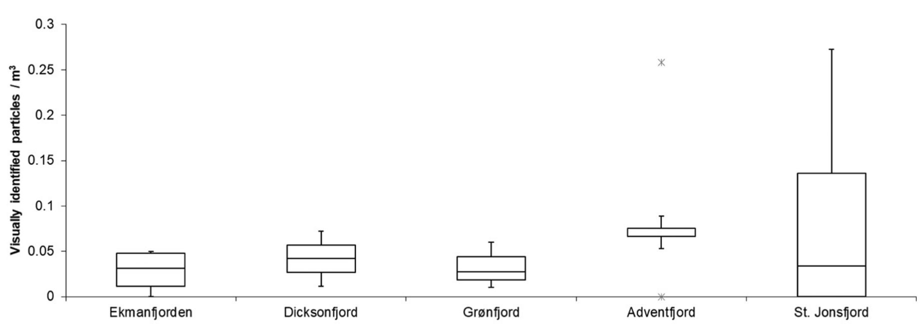



**Fig. 2.** Results from manta net samples collected in all 5 fjord locations. Data presented is visually accepted particles only, after blank correction. Data has not been corrected for polymers due to uneven application of FT-IR (detailed in SI, Table 1).

sampling features were observed. The pump samples in Dicksonfjorden and Grønfjorden contained higher quantities of microplastics than the manta net samples taken in the same fjords (Table S2). Interestingly, the pump sample from the most populated fjord (Advent-) did not contain any particles. All particles collected were fibres except one fragment found in the Nordfjorden sample and one fragment found in the Oxaasdalen river sample. Beads were not detected in any of the pump samples and it is hypothesised that this sampling technique undersampled beads.

FT-IR was applied in varying degrees to the visually identified particles collected from surface waters (more information presented in the Supplementary Materials). Of the particles analysed with FT-IR from Isfjorden, polyethylene (PE) was the most commonly identified polymer (86% of total particles, Fig. 3), followed by polypropylene (PP; 7%) and rubber-like particle (7%). PP was as common as PE in Grønfjorden samples (50% of each polymer). Polyester was the main polymer in St. Jonsfjorden (42%), followed by PE (27%). Acrylic (8%) was found in two out of five samples, and PP (4%) was found in only one out of five samples from St. Jonsfjorden.

It was hypothesised that locality to human activities might influence the abundance and composition of microplastics in the surface waters. Adventfjorden was identified as the area with most traffic (snow mobile, cars, boats), followed by Grønfjorden (Barentsburg settlement, and popular snow mobile tourism area) (personal observations, communications with e.g. tourist guide companies). However, the pump sample from Adventfjorden was the only pump sample without microplastics. The largest quantity of microplastics in manta net samples (0.26 particles/m$^3$) was found in the sample closest to Longyearbyen airport. This sample consisted mostly of fragments ($n = 4$). All particles in this sample could be confirmed with FT-IR as PE and one fragment as a rubber-like particle. Although no significant difference (few samples), there seemed to be slightly more particles in Adventfjorden manta nets than those samples from Grønfjorden, while Grønfjorden was comparable to Ekman- and Dicksonfjorden even though those two fjords lack settlements and have less visitors and traffic. St. Jonsfjorden had a higher number of particles than Isfjorden. The difference is not large and is most likely explained by different weather conditions and local impact. In addition to the single rubber-like particles identified in Adventfjorden, one other rubber-like particles was observed in Dicksonfjorden. Similar particles were not identified in the other fjords. A large source of rubber particles globally, has been identified as car tyres (Kole et al., 2017), however, this location lacks cars and roads, and the wintertime snow mobile traffic is low compared to Advent- and Grønfjorden. Snow mobile belts are made of rubber and could be a source together with e.g. car tyres in harbours (e.g. Bråte et al., 2018), although it would be expected to see greater numbers closer to populated areas. To confirm this theory, more studies should be conducted in Adventfjorden and Grønfjorden, and preferably during or slightly after the winter season. Since the present study was carried out in the autumn, winds, currents, and waves have most likely facilitated the spreading and occurrence of winter-transport related particles from the more populated areas.

Only one pump sample was taken in Nordfjorden and the manta net could not be deployed due to rough sea. Nevertheless, the total amount in Nordfjorden was one of the highest in the Isfjorden fjords; 3.5 particles/m$^3$ (100, 300 and 500 μm combined, one black fibre confirmed with FT-IR) and were similar to Oxaasdalen river sample (1.7 particles/m$^3$; 100 μm and 300 μm filters combined) in amount (Table 1). Fibres dominated (91% of particles) in all pump samples but were generally too thin to be analysed successfully with the available FT-IR instruments.

The microplastics concentration in water samples presented here were low compared to a recent study from Kongsfjorden, which is situated further north on Svalbard. The reported particle concentrations varied from 0.1 particles/L (i.e. 100 particles/m$^3$) in the water column from the fjord mouth and up to 48 particles/L (i.e. 48,000 particles/m$^3$) in surface water from central Kongsfjorden (von Friesen et al., 2020). Several factors such as increased human activity, e.g. presence of recreational and commercial vessels, as well as the discharge of sewage effluents may influence the concentrations of microplastics identified across studies. Nevertheless, methodological differences and weather may also influence the results. This highlights the need for the collection of metadata for sufficient data interpretation. As an example, the number of persons inhabiting Kongsfjorden during summer is normally around 100 but can reach 1000–4000 people with the addition of cruise tourists in the summer months (personal observation). Therefore, sampling at different times of year to monitor the influence of human presence on Svalbard would be beneficial in future studies.

Von Friesen et al. (2020) used Niskin bottles from a ship and sampled small volumes, while manta net and large sampling volumes were used in the present study. In additional to the methodological differences, the weather might also have affected the results. As both studies were performed in the same season (i.e. late June-early September) we can consider the local weather conditions. There were high winds (10–17 m/s; Norwegian Meteorological Institute, 2019) before and during sampling in the present study, which may have led to mixing of water and potentially facilitated the transport/dispersal of particles within the water column. This might have had a greater effect on the pump samples, since water is collected with a small inlet (2 cm) and hence, the pump samples might have underestimated the microplastic concentration in the investigated fjords. On the other hand, the results from the present study were slightly lower/in line with previous reports of microplastics sampled around Svalbard with a manta net (S-SW of Svalbard, 0.34 particles/m$^3$) and pumping systems (Isfjorden area and Kongsfjorden; <LOD-1 particle/m$^3$) (Lusher et al., 2015; Sundet et al., 2017). These Svalbard studies (and the present study) are comparable to/slightly lower than microplastic investigations performed East of

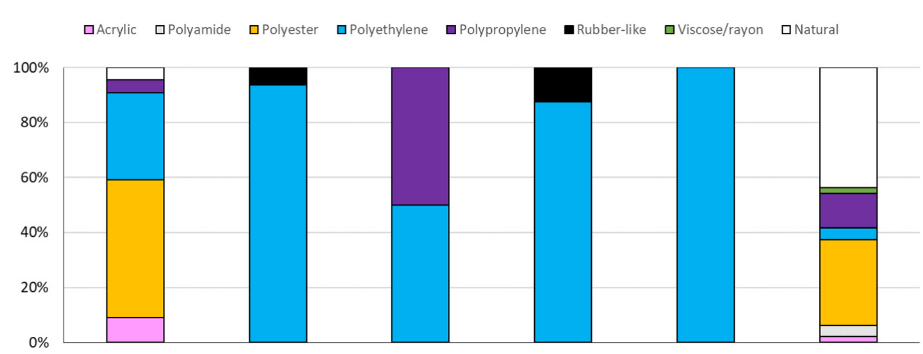



**Fig. 3.** Relative distribution of confirmed polymers in the manta net samples and walrus faeces. It was not possible to get reliable FTIR results from the sediment and thus not discplayed in the figure.

Greenland (range 1–3 MP/m$^3$, average: 2.4 MP/m$^3$; fibres excluded) (Morgana et al., 2018). It is worth mentioning that depth (6 m outside Greenland, surface water around Svalbard) and analytical methodologies vary between these studies. A recent study from the Eastern Arctic (Barents, Kara, Laptev and East Siberian Seas) showed comparable/ slightly lower concentrations than the present study (mean: 0.004 MP/ m$^3$ in surface, 0.8 MP/m$^3$ in sub-surface water) (Yakushev et al., 2021).

### 3.2. Microplastics in sediment from river deltas

The sediment samples were from a remote tidal estuary with no settlements and little traffic. Four of the sediment samples contained between 2 and 26 particles/kg dw sediment (average 11 particles/kg dw) while no particles were observed in the last two samples. Similar concentrations are reported from other Arctic areas, e.g. <LOD-69 particles/kg dw sediment (unknown depth) in Bering-Chukchi Sea (Mu et al., 2019).

Fibres were the dominate particle type (71%) observed in sediment samples followed by fragments (29%). No beads were found. It was not possible to obtain usable FT-IR data from the sediment samples due to size and state of the particles (i.e. no clear results from FT-IR were obtained). The fjord (Dicksonfjorden), where the sediments were collected and the sampled river drains out, is seldom visited and there are no routine tourist tours here. The abandoned settlement of Pyramiden is situated 20 km east across the mountains and could be a potential source by (small-scale) atmospheric transport to snow caps in combination with river run-off from these areas. No studies regarding microplastics have been conducted here, but Billefjorden outside Pyramiden is a popular tourism area and could therefore be a source of microplastics. Earlier studies have shown that Pyramiden is a local source of polychlorinated biphenyls (PCBs) (Pedersen et al., 2011), suggesting that further research is required to investigate the release of microplastics and other anthropogenic particles.

### 3.3. Microplastic in the coastal benthic food web

Even though all of the North Atlantic Arctic pinnipeds can be found around Svalbard (Bengtsson et al., 2021), their food webs are longer and more complex with more trophic levels involved compared to the food web of walrus. Walrus often feed in shallow areas (<50 m in Svalbard populations) and some of the popular feeding areas outside of Poolepynten are situated at only 10–20 m depth (Kovacs et al., 2014; Norwegian Polar Institute, 2021). Hence, microplastics from the upper surface layer could be mixed down to benthic filter feeders.

Eight pieces of walrus faeces (100 g) were analysed for microplastic content. As all particles were processed with FT-IR, the data presented refers only to microplastic particles. Fibres and fragments were found in all except one sample, ranging from <LOD-7 microplastics/sample (fibres) and <LOD-3 microplastics/sample (fragments). This generates an average of 34 MP/kg faeces (FT-IR confirmed particles), while visual ID confirmed particles were on average 69 MP/kg faeces. Benthic filtering organisms - such as clams - can filter substantial amounts of seawater and hence, pose a high possibility for internalising microplastics. There are very few studies on polar marine pinnipeds and faecal microplastics and the feeding behaviour differs between species as well as distance to urban areas. For example, the pelagic-feeding Antarctic fur seal (*Arctocephalus gazella*) from Deception Island, which target mainly mesopelagic fish and zooplankton, did not contain microplastics in their faeces (Garcia-Garin et al., 2020). Antarctica is more remote than the Arctic and the differences in feeding behaviour between Antarctic fur seals and walrus will also affect the amount of microplastics found in faeces. When referring to density of plastic materials alone, most plastic will float on the surface and hence, should (theoretically) not be available for benthic filter feeders who represent the first trophic level above water. Given this, microplastics should not appear at the next trophic level (as represented by walrus in the present study) in a benthic food web, or, only plastics with a density greater than seawater would be found.

The findings in the present study shows relatively low surface water concentrations of microplastics in all fjords (0.03–0.9 particles/m$^3$, visual ID – manta net). The particles which were identified with FT-IR were mainly represented by low-density plastics (e.g. PE, PP). Similarly, the walrus faeces were dominated by polyester and PP, both with a lower density than seawater. Hence, these particles are most likely reaching the benthic areas through surface mixing facilitated by waves and winds which especially impact shallow areas. Investigations of the primary filter feeders (e.g. clams) would illustrate the uptake of microplastics by benthic filter feeders but it was not possible to collect them for the present study. Nevertheless, there is copious research showing microplastics in benthic filter feeders (e.g. Bråte et al., 2018, 2020) which supports the hypothesis that trophic transfer is a viable route of exposure for the walrus.

The walrus faeces presented dietary items including bivalve shell fragments and remnants of clam siphons, which is in agreement with previous research (e.g., Gjertz and Wiig, 1992; Kovacs et al., 2014). The faecal matter had the appearance of green-brown mud, which was dry on the surface but damp to the touch, indicative of the summer collection conditions. The clam *Mya truncata* is the most common prey of these walrus, and further research should be conducted to confirm the assumptions of the present study. There appears to be negligible consequence to the walrus, as microplastics which have been ingested have successfully passed through the digestive tracts to be defecated along with other inedible items. The digestive system of pinnipeds eliminates non-digestible items, as has been evidenced through observations into the entire digestive tracts (e.g. Lusher et al., 2018) as well as faecal analysis (Donohue et al., 2019; Hudak and Sette, 2019). Nevertheless, the study design applied here cannot deduce whether walrus might ingest larger pieces of plastics that could have a longer retention time in their digestive system than microplastics. Such investigation would require necropsies which was not an option in the present study.

The present study does not consider the consequences of inherent additives or sorbed environmental contaminants which may be associated with microplastics or other plastic materials. Some studies have suggested that higher trophic level organisms assimilate these plastic components (e.g. Fossi et al., 2017). During digestion, contaminants from plastics might be released (or absorbed) and their small molecular size allows them to cross gut barriers. However, investigating such a consequence for walrus will require further research. Previous research has investigated the presence of a wide range of contaminants in walrus tissues. These included polychlorinated biphenyls (PCBs), dichlorodiphenyldichloroethylene (DDE), chlordanes, toxaphenes and polybrominated diphenyl ethers (PBDEs), with general trends pointing towards a decrease in PCBs and DDE levels in the Arctic over time (Wolkers et al., 2006; Scotter et al., 2019). The studies showed that the more contaminated individuals tended to be those that were feeding on higher trophic levels (e.g., seals). Regarding the identified contaminants, many of these can also be associated with plastics due to their physical-chemical properties (e.g., Syberg et al., 2020). The combined impact of environmental contaminants and plastic pollution will be an important area for future studies, given that the relationships between PCBs and plastics have been observed for seabird species (e.g., Ryan et al., 1988; Kühn et al., 2020). However, no relationship between persistent organic pollutants (POPs) and plastics was observed for Northern Fulmars from Newfoundland (Provencher et al., 2018) and it has been shown that feed is a more important pathway of POPs and other contaminants to birds than plastics as a vector (Herzke et al., 2016).

This study provides evidence that walrus can defecate microplastics. In doing so, microplastics are re-released into the environment – again becoming available for movement within the Arctic ecosystem – whether to be incorporated into shoreline sediments, be washed back into coastal water bodies, or ingested again by Arctic biota. It is therefore fundamental that the sources of microplastics to the coastal Arctic ecosystem are understood and measures to prevent their release are addressed and implemented.

### 3.4. Local sources of microplastics on Svalbard

With a small data set and without longer sampling periods, it is difficult to draw conclusions regarding sources. The present study found microplastic concentrations in coastal areas (Isfjorden: 0.04 MP/m$^3$ and St. Jonsfjorden; 0.9 MP/m$^3$) comparable to offshore waters (0.34 MP/m$^3$; Lusher et al., 2015). The West Spitsbergen Current transports warm water towards Svalbard and is present in both Isfjorden and Forlandssundet where St. Jonsfjorden and the haul-out sites for the walrus are situated (Nilsen et al., 2016). The plastic polymers identified in this study (Fig. 3) are amongst the most common plastics materials, but as the concentrations were low, they do not indicate large local sources. Potential local sources in and around Svalbard may be fishing vessels and tourism, in addition to the local settlements and wastewater effluents into the fjords. A recent study concluded that Adventfjorden and Kongsfjorden did not show increased concentrations of microplastics in the water compared to offshore areas around Svalbard (Sundet et al., 2017) while other studies indicated that local wastewater is a source of microplastics (Granberg et al., 2019). One should stress that the dilution from, for example, the release of Longyearbyen sewage water (from 2400 habitants at 60 m depth) into Adventfjorden may not result in a measurable increase in the surface water microplastic concentrations and especially not during summer/autumn when riverine run-off is high and covers large parts of the fjord with a low-saline top layer. A fresh/low-saline top layer is also common in fjords with glaciers (e.g. Ekmanfjorden) and/or large/many rivers (Dicksonfjorden, Grønfjorden), which may affect where in the water mass the microplastics can be found.

Like many settlements in the Arctic, Longyearbyen and Barentsburg do not have a sewage treatment system in place. Ny-Ålesund has a small treatment system since 2015, which has proven to be efficient in capturing microplastics, reducing the number of particles from 14,207 to 83 particles/L after treatment (Granberg et al., 2019). The wastewater from Barentsburg drains into Grønfjorden, but information regarding the position of outlets is unclear. Hence, it is not possible to assess the contribution of wastewater effluents as a source of microplastics to Grønfjorden. Even though settlements are potential sources of microplastics, the present study cannot conclude on any increased local occurrence in those fjords. Adventfjorden and Grønfjorden experienced heavy winds two days before sampling and hence, this might also have affected the distribution of microplastics in the fjords. Since Ny-Ålesund has a permanent population of 30–35 habitants and a summer population of 120 (Statistics Norway, 2021), it is expected that the quantities of microplastics released from Longyearbyen and Barentsburg would be much higher. Most likely, the released amounts of microplastics are largely diluted and require different sampling equipment and design to be detected.

#### 3.4.1. Long-range atmospheric transport of microplastics

There have only been a few studies into the atmospheric transport of microplastics, but it seems likely to be a transport route, at least for the smallest and lightest particles (Evangeliou et al., 2020). The river in Oxaasdalen is fed by a small snow cap and contained the second highest quantity of microplastics (1.7 particles/m$^3$) in this study. In addition, some long blue fibres were found in the river, but they were excluded from this investigation as they were > 5 mm. If they would be included, the total particle count here would be 4 microplastics/m$^3$. Whether these fibres come from local (including Isfjorden area) sources or from long-range transport is not known.

## 4. Conclusion

Microplastics were found in all fjord samples (water, sediment, walrus faeces) and the most common polymer was PE (31% of tested particles) followed by polyester (23%). These results show the presence of microplastics in areas once considered pristine and free from anthropogenic pollution. There were no significant differences between the investigated fjords even though there are differences regarding settlement and number of visitors. Walrus are benthic feeders at low trophic levels, but still had several microplastics in their faeces. Whether this type of benthic feeder is more exposed to microplastics than other benthic (and pelagic) feeders should be further investigated together with potential links between different matrices, e.g. sediment and water exposure. Local atmospheric transport may play a role for microplastics distribution, especially in combination with snow melt and runoff.

In general, this baseline investigation shows that, from the low number of samples, there is a need to further investigate microplastics within Arctic ecosystems. The consequences of microplastics may not be extreme for the apex species studied here, but the routes by which the microplastic are ingested and egested will require a focused assessment. Microplastics have been found in several sample types in the Arctic, but to elucidate uptake and transport pathways, it is important that future studies combine different matrices in the same study, utilising reproducible methods, to achieve a holistic approach for ecosystem occurrence and fate of microplastics.

## CRediT authorship contribution statement

Pernilla Carlsson: Conceptualization, investigation, data curation, writing.

Cecilie Singdahl-Larsen: Investigation, data curation, writing.

Amy Lusher: Conceptualization, methodology, investigation, data curation, writing, funding acquisition.

## Declaration of competing interest

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

## Acknowledgements

We acknowledge the Norwegian Ministry of Climate and Environment for funding the project “Mikroplast i polare strøk”, Spitsbergen Guide Service for boat transport and logistics. Sverre Johansen for assisting with fieldwork and Madeleine Purver, MRM student at university Centre in the Westfjords, Iceland. For samples collected in St. Jonsfjorden and Poolepynten, we are grateful for the support from Emilie Kallenbach (NIVA-Denmark) and Tom Foreman (Jason Roberts Productions), as well as Liz Bonnin, Tom Watt-Smith, Ben McGeorge-Henderson and Alistair Livingstone. Gema Hernandez-Milian and two anonymous reviewers are thanked for their input on an earlier iteration of this manuscript.

## Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2021.148308.

## References

AMAP, 2016. AMAP Assessment 2014: Contaminants of Emerging Concern in the Arctic (Oslo, Norway. xvi + 353 pp.).

AMAP, 2017. Snow, Water, Ice and Permafrost in the Arctic (SWIPA) 2017. Arctic Monitoring and Assessment Programme (AMAP), Oslo, Norway. xiv, p. 269.

Bengtsson, O., ton, C.D., Lydersen, C., Andersen, M., Kovacs, K.M., 2021. Distribution and habitat characteristics of pinnipeds and polar bears in the Svalbard Archipelago, 2005–2018. POLAR 40. https://doi.org/10.33265/polar.v40.5326.

Bourdages, M.P.T., Provencher, J.F., Baak, J.E., Mallory, M.L., Vermaire, J.C., 2021. Breeding seabirds as vectors of microplastics from sea to land: evidence from colonies in Arctic Canada. Sci. Total Environ. 764, 142808. https://doi.org/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., Andrews, R.C., Rochman, C.M., 2020. Sampling and quality assurance and quality control: a guide for scientists investigating the occurrence of microplastics across matrices. Appl. Spectrosc. 74, 1099–1125. https://doi.org/10.1177/0003702820945713.

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

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

Bråte, I.L.N., Hurley, R.R., Lusher, A.L., Buenaventura, N., Hultman, M., Halsband, C., Green, N., 2020. Microplastic in Marine Bivalves from the Nordic Environment. Nordic Council of Ministers.

Cowger, W., Booth, A.M., Hamilton, B.M., Thaysen, C., Primpke, S., Munno, K., Lusher, A.L., Dehaut, A., Vaz, V.P., Liboiron, M., Devriese, L.I., Hermabessiere, L., Rochman, C., Athey, S.N., Lynch, J.M., De Frond, H., Gray, A., Jones, O.A.H., Brander, S., Steele, C., Moore, S., Sanchez, A., Nel, H., 2020. Reporting guidelines to increase the reproducibility and comparability of research on microplastics. Appl. Spectrosc. 74, 1066–1077. https://doi.org/10.1177/0003702820930292.

Cózar, A., Martí, E., Duarte, C.M., García-de-Lomas, J., van Sebille, E., Ballatore, T.J., Eguíluz, V.M., González-Gordillo, J.I., Pedrotti, M.L., Echevarría, F., Troublè, R., Irigoien, X., 2017. The Arctic Ocean as a dead end for floating plastics in the North Atlantic branch of the Thermohaline Circulation. Sci. Adv. 3, e1600582. https://doi.org/10.1126/sciadv.1600582.

Donohue, M.J., Masura, J., Gelatt, T., Ream, R., Baker, J.D., Faulhaber, K., Lerner, D.T., 2019. Evaluating exposure of northern fur seals, Callorhinus ursinus, to microplastic pollution through fecal analysis. Mar. Pollut. Bull. 138, 213–221. https://doi.org/10.1016/j.marpolbul.2018.11.036.

Evangeliou, N., Grythe, H., Klimont, Z., Heyes, C., Eckhardt, S., Lopez-Aparicio, S., Stohl, A., 2020. Atmospheric transport is a major pathway of microplastics to remote regions. Nat. Commun. 11, 3381. https://doi.org/10.1038/s41467-020-17201-9.

Fossi, M.C., Baini, M., Panti, C., Galli, M., Jiménez, B., Muñoz-Arnanz, J., Marsili, L., Finoia, M.G., Ramírez-Macías, D., 2017. Are whale sharks exposed to persistent organic pollutants and plastic pollution in the Gulf of California (Mexico)? First ecotoxicological investigation using skin biopsies. Comp. Biochem. Physiol. Part C: Toxicol. Pharmacol. 199, 48–58. https://doi.org/10.1016/j.cbpc.2017.03.002.

Freitas, C., Kovacs, K.M., Ims, R., Fedak, M.A., Lydersen, C., 2009. Deep into the ice: overwintering and habitat selection in male Atlantic walruses. Mar. Ecol. Prog. Ser. 375, 247–261. https://doi.org/10.3354/meps07725.

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

Garcia-Garin, O., García-Cuevas, I., Drago, M., Rita, D., Parga, M., Gazo, M., Cardona, L., 2020. No evidence of microplastics in Antarctic fur seal scats from a hotspot of human activity in Western Antarctica. Sci. Total Environ. 737, 140210. https://doi.org/10.1016/j.scitotenv.2020.140210.

Gjertz, I., Wiig, Ø., 1992. Feeding of walrus Odobenus rosmarus in Svalbard. Polar Record 28, 57–59. https://doi.org/10.1017/S0032247400020283.

Granberg, M., von Friesen, L.W., Bach, L., Collard, F., Gabrielsen, G.W., Strand, J., 2019. Anthropogenic Microlitter in Wastewater and Marine Samples From Ny-Ålesund, Barentsburg and Signehamna, Svalbard. IVL Swedish Environmental Research Institute 978-91-7883-020-6.

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

Hamilton, C., Kovacs, Kit M., Lydersen, Christian, 2015. Year-round haul-out behaviour of male walruses Odobenus rosmarus in the northern Barents Sea. Mar. Ecol. Prog. Ser. 519, 251–263. https://doi.org/10.3354/meps11089.

Hamilton, B.M., Bourdages, M.P.T., Geoffroy, C., Vermaire, J.C., Mallory, M.L., Rochman, C.M., Provencher, J.F., 2021. Microplastics around an Arctic seabird colony: particle community composition varies across environmental matrices. Sci. Total Environ. 773, 145536. https://doi.org/10.1016/j.scitotenv.2021.145536.

Herzke, D., Anker-Nilssen, T., Nøst, T.H., Götsch, A., Christensen-Dalsgaard, S., Langset, M., Fangel, K., Koelmans, A.A., 2016. Negligible impact of ingested microplastics on tissue concentrations of persistent organic pollutants in northern fulmars off coastal Norway. Environ. Sci. Technol. 50, 1924–1933. https://doi.org/10.1021/acs.est.5b04663.

Hudak, C.A., Sette, L., 2019. Opportunistic detection of anthropogenic micro debris in harbor seal (*Phoca vitulina vitulina*) and gray seal (*Halichoerus grypus atlantica*) fecal samples from haul-outs in southeastern Massachusetts, USA. Mar. Pollut. Bull. 145, 390–395. https://doi.org/10.1016/j.marpolbul.2019.06.020.

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

Kole, P.J., Löhr, A.J., Van Belleghem, F., Ragas, A., 2017. Wear and tear of Tyres: a stealthy source of microplastics in the environment. Int. J. Environ. Res. Public Health 14, 1265. https://doi.org/10.3390/ijerph14101265.

Kovacs, K.M., Aars, J., Lydersen, C., 2014. Walruses recovering after 60+ years of protection in Svalbard, Norway. POLAR 33. https://doi.org/10.3402/polar.v33.26034.

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

Kühn, S., Booth, A.M., Sørensen, L., Van Oyen, A., Van Franeker, J.A., 2020. Transfer of additive chemicals from marine plastic debris to the stomach oil of northern fulmars. Front. Environ. Sci. 8, 138. https://doi.org/10.3389/fenvs.2020.00138.

Lusher, A., Hernandez-Milian, G., 2018. Microplastic extraction from marine vertebrate digestive tracts, regurgitates and scats: a protocol for researchers from all experience levels. BIO-PROTOCOL 8. https://doi.org/10.21769/BioProtoc.3087.

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

Lusher, A.L., Hernandez-Milian, G., Berrow, S., Rogan, E., O'Connor, I., 2018. Incidence of marine debris in cetaceans stranded and bycaught in Ireland: recent findings and a review of historical knowledge. Environ. Pollut. 232, 467–476. https://doi.org/10.1016/j.envpol.2017.09.070.

Lusher, A.L., Bråte, I.L.N., Munno, K., Hurley, R.R., Welden, N.A., 2020. Is it or isn't it: the importance of visual classification in microplastic characterization. Appl. Spectrosc. 74, 1139–1153. https://doi.org/10.1177/0003702820930733.

Morgana, S., Ghigliotti, L., Estévez-Calvar, N., Stifanese, R., Wieckzorek, A., Doyle, T., Christiansen, J.S., Faimali, M., Garaventa, F., 2018. Microplastics in the Arctic: a case study with sub-surface water and fish samples off Northeast Greenland. Environ. Pollut. 242, 1078–1086. https://doi.org/10.1016/j.envpol.2018.08.001.

Mu, J., Qu, L., Jin, F., Zhang, S., Fang, C., Ma, X., Zhang, W., Huo, C., Cong, Y., Wang, J., 2019. Abundance and distribution of microplastics in the surface sediments from the northern Bering and Chukchi seas. Environ. Pollut. 245, 122–130. https://doi.org/10.1016/j.envpol.2018.10.097.

Nilsen, F., Skogseth, R., Vaardal-Lunde, J., Inall, M., 2016. A simple shelf circulation model: intrusion of Atlantic water on the West Spitsbergen shelf. J. Phys. Oceanogr. 46 (4), 1209–1230. https://doi.org/10.1175/JPO-D-15-0058.1.

Norwegian Meteorological Institute, 2019. Svalbard Statistics. https://www.yr.no/en/statistics/graph/5-99840/Norway/Svalbard/. (Accessed 5 November 2019).

Norwegian Polar Institute, 2021. Walrus [WWW document]. https://www.npolar.no/arter/hvalross (accessed Jan 12, 2021).

Obbard, R.W., Sadri, S., Wong, Y.Q., Khitun, A.A., Baker, I., Thompson, R.C., 2014. Global warming releases microplastic legacy frozen in Arctic Sea ice. Earth's Future 2, 315–320. https://doi.org/10.1002/2014EF000240.

OSPAR, 2021. OSPAR commission, beach litter [WWW document]. URL. OSPAR Commissionhttps://www.ospar.org/work-areas/eiha/marine-litter/assessment-of-marine-litter/beach-litter. (Accessed 12 January 2021).

PAME, 2020. Desktop study on marine litter including microplastics in the Arctic. http://hdl.handle.net/11374/2389.

Panti, C., Baini, M., Lusher, A., Hernandez-Milan, G., Bravo Rebolledo, E.L., Unger, B., Syberg, K., Simmonds, M.P., Fossi, M.C., 2019. Marine litter: one of the major threats for marine mammals. Outcomes from the European Cetacean Society workshop. Environ. Pollut. 247, 72–79. https://doi.org/10.1016/j.envpol.2019.01.029.

Pedersen, H.R., Kallenborn, R., Ottesen, R.T., Gabrielsen, G.W., Schrum, C., Evenset, A., Ruus, A., Benjaminsen, H., Sagerup, K., Christensen, G., Eggen, O., Carlsson, P., Johansson-Karlsson, E., Polder, A., Lundkvist, Q., 2011. PCB on Svalbard. Norwegian Environment Agency and Governor of Svalbard, Longyearbyen.

Peeken, I., Primpke, S., Beyer, B., Gütermann, J., Katlein, C., Krumpen, T., Bergmann, M., Hehemann, L., Gerdts, G., 2018. Arctic Sea ice is an important temporal sink and means of transport for microplastic. Nat. Commun. 9, 1505. https://doi.org/10.1038/s41467-018-03825-5.

Provencher, J.F., Avery-Gomm, S., Liboiron, M., Braune, B.M., Macaulay, J.B., Mallory, M.L., Letcher, R.J., 2018. Are ingested plastics a vector of PCB contamination in northern fulmars from coastal Newfoundland and Labrador? Environ. Res. 167, 184–190. https://doi.org/10.1016/j.envres.2018.07.025.

Ryan, P.G., Connell, A.D., Gardner, B.D., 1988. Plastic ingestion and PCBs in seabirds: is there a relationship? Mar. Pollut. Bull. 19 (4), 174–176. https://doi.org/10.1016/0025-326X(88)90674-1.

Scotter, S.E., Tryland, M., Nymo, I.H., Hanssen, L., Harju, M., Lydersen, C., Kovacs, K.M., Klein, J., Fisk, A.T., Routti, H., 2019. Contaminants in Atlantic walruses in Svalbard part 1: relationships between exposure, diet and pathogen prevalence. Environ. Pollut. 244, 9–18. https://doi.org/10.1016/j.envpol.2018.10.001.

Statistics Norway, 2021. Befolkningen på Svalbard (Population of Svalbard). https://www.ssb.no/befolkning/statistikker/befsvalbard. (Accessed 11 May 2021).

Sundet, J.H., Herzke, D., Jensen, M., 2017. Forekomst av mikroplastikk i sjøvann, bunnsedimenter, fjæresediment og i filtrerende bunnorganismer i nære kystområder på Svalbard (No. RIS-10495). Institute of Marine Research.

Syberg, K., Knudsen, C.M., Tairova, Z., Khan, F.R., Shashoua, Y., Geertz, T., Pedersen, H.B., Sick, C., Mortensen, J., Strand, J., Palmqvist, A., 2020. Sorption of PCBs to environmental plastic pollution in the North Atlantic Ocean: importance of size and polymer type. Case Stud. Chem. Environ. Eng. 2, 100062. https://doi.org/10.1016/j.cscee.2020.100062.

Tirelli, V., Suaria, G., Lusher, A.L., 2020. Microplastics in polar samples. In: Rocha-Santos, T., Costa, M., Mouneyrac, C. (Eds.), Handbook of Microplastics in the Environment. Springer, Cham https://doi.org/10.1007/978-3-030-10618-8_4-1.

de Vries, A.N., Govoni, D., Árnason, S.H., 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. https://doi.org/10.1016/j.marpolbul.2019.110827.

Wolkers, H., Van Bavel, B., Ericson, I., Skoglund, E., Kovacs, K.M., Lydersen, C., 2006. Congener-specific accumulation and patterns of chlorinated and brominated contaminants in adult male walruses from Svalbard, Norway: indications for individualspecific prey selection. Sci. Total Environ. 370 (1), 70–79. https://doi.org/10.1016/j.scitotenv.2006.06.005.

Yakushev, E., Gebruk, A., Osadchiev, A., Pakhomova, S., Lusher, A., Berezina, A., van Bavel, B., Vorozheikina, E., Chernykh, D., Kolbasova, G., Razgon, I., Semiletov, I., 2021. Microplastics distribution in the Eurasian Arctic is affected by Atlantic waters and Siberian rivers. Commun. Earth Environ. 2, 23. https://doi.org/10.1038/s43247-021-00091-0.

Zantis, L.J., Carroll, E.L., Nelms, S.E., Bosker, T., 2021. Marine mammals and microplastics: a systematic review and call for standardisation. Environ. Pollut. 269, 116142. https://doi.org/10.1016/j.envpol.2020.116142.

Zhang, Y., Kang, S., Allen, S., Allen, D., Gao, T., Sillanpää, M., 2020. Atmospheric microplastics: a review on current status and perspectives. Earth Sci. Rev. 203, 103118. https://doi.org/10.1016/j.earscirev.2020.103118.