Marine Pollution Bulletin 196 (2023) 115586

Contents lists available at ScienceDirect

Marine Pollution Bulletin

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

Occurrence and sources of microplastics on Arctic beaches: Svalbard

Tesni Lloyd-Jones [a], Jonathan J. Dick [a,*], Timothy P. Lane [a], Eoghan M. Cunningham [b], Konstadinos Kiriakoulakis [a]

[a] School of Biological and Environmental Sciences, Liverpool John Moores University, Byrom Street, Liverpool L3 3AF, UK
[b] Queen's University Marine Laboratory, Queen's University Belfast, 12-13 The Strand, Portaferry BT22 1PF, Northern Ireland, UK

ARTICLE INFO

Keywords:
Arctic
Microplastics
Pollution

ABSTRACT

Plastic pollution is recognised as a major global environmental concern, especially within marine environments. The small size of microplastics (< 5 mm) make them readily available for ingestion by organisms in all trophic levels. Here, four beach sites in Adventfjorden on the west coast of Svalbard, were sampled with the aim of investigating the occurrence and abundance of microplastics on beaches to assess potential sources of microplastic pollution. High variability in microplastic amount, type and polymers were found at all sites ranging from means of 0.7 n/g (number) at the remotest site and 2.2 n/g (number) at the site closest to Longyearbyen. Statistical analyses suggested that patterns observed were linked to direct proximity to human activities through land uses and effluent discharge. These findings point to an increased importance of localised factors on driving elevated microplastic pollution in beach sediments over oceanic controls in remote but inhabited Arctic locations and have important implications for our understanding and future assessments of microplastic pollution in such settings.

1. Introduction

Marine plastic pollution is recognised as a major global environmental concern affecting all oceans of the world (Solomon and Palanisami, 2016). With an estimated 4.8 to 12.7 million tonnes of plastic entering them each year (Jambeck et al., 2015), they pose serious increasing risks to biotic environments and human health (Renner, 2018). In 2021, worldwide plastic production reached 390 million tonnes yr^{-1} (PlasticsEurope, 2022). Plastics are ubiquitously used due to their low cost, lightweight and durable nature, the same properties which cause plastic to persist in the environment far beyond their consumer lifetime (Ahmed et al., 2022). Plastic litter in the marine environment has been reported since the early 1970s, when the production of plastic was only a fraction of what it is today (Cole et al., 2011). Marine plastic pollution is particularly concerning, and often attributed to the increased manufacturing of single-use plastic, insufficient waste management and the increasing global population (Scott, 2019). Microplastics in oceans can originate from both land and ocean-based sources as marine litter which is then slowly degraded over time via fragmentation of larger plastics, because of chemical and physical processes such as wave action and UV-radiation induced degradation (Masura et al., 2015; Sharma, 2019) forming microplastics. Microplastics are defined as particles between 1 nm to <5 mm in diameter (Frias and Nash, 2019). They can also be released to the ocean as MP particles themselves, through terrestrial processes and effluent discharge, either direct or indirect via river systems (Ross et al., 2021).

The presence of microplastics in the marine environment is becoming an increasing concern, and they are now estimated to account for 92.4 % of marine plastic pollution, predominantly comprising of pellets, fragments, and fibres (Eriksen et al., 2014). They are readily bioavailable, can bioaccumulate within individual organisms, and be transferred up the food chain (Halsband and Herzke, 2019; Sathish et al., 2019). Microplastic ingestion has potential physiological and mechanical effects which have consequent negative impacts on growth and reproduction (Li et al., 2019, and Jiang et al., 2022). Additionally, they can absorb persistent organic pollutants, which may facilitate the entry of toxins into the food chain, potentially harming both wildlife and humans (Andrady, 2011; Li et al., 2019). Microplastics have been detected throughout the marine environment particularly within marine sediments, water columns and marine biota (Yu et al., 2016). Reports of plastic pollution have occurred in even the most remote and pristine areas of the world, including the deep marine environments (e.g., Kane et al., 2020), the Southern Ocean (e.g. Cunningham et al., 2020), the Arctic (e.g. Jaskolski et al., 2018), the Antarctic (Aves et al., 2022) and

* Corresponding author.
E-mail address: j.j.dick@ljmu.ac.uk (J.J. Dick).

https://doi.org/10.1016/j.marpolbul.2023.115586
Received 16 March 2023; Received in revised form 21 September 2023; Accepted 23 September 2023

8440 m.a.s.l. on Mt. Everest (Napper et al., 2020). Despite its remoteness and low population density, the Arctic is not exempt from microplastic pollution (Hallanger and Gabrielsen, 2018), which has been detected within Arctic waters (0.7 particles m^{-3}; Kanhai et al., 2018), sea ice (1.1 (±0.8) x 10^6 to 1.2 (±1.4) x 10^7 particles m^{-3}; Obbard et al., 2014; Peeken et al., 2018), deep-sea sediment (42–6595 particles kg^{-1}; Bergmann et al., 2017b) and biota (e.g. Fang et al., 2018).

In Svalbard, microplastics have been reported in surface and subsurface waters in the south (0.34–2.68 particles m^{-3}; Lusher et al., 2015), and Kongsfjorden, north-western Svalbard (112 ± 53 particles m^{-3}; Scott, 2019). In the Krossfjord-Kongsfjord system, surface sediments contained abundances of 721–783 particles kg^{-1} (Choudhary et al., 2022). The lack of studies in Svalbard makes it difficult to assess sources of Svalbard microplastics, though Lusher et al. (2015) suggest they derive from long-range transport. Bao et al. (2022) suggest likely contributions from multiple sources, including sea ice melting, long-range atmospheric deposition, and deposition from vessels. In Svalbard beach settings, the majority of plastic studies have focused on macroplastics, finding extensive plastic pollution in North and West Svalbard (Bergmann et al., 2017a; Jaskolski et al., 2018). At present, only three studies investigate microplastics within Svalbard beaches, in Breibogen, North Svalbard (111 particles/l above the high tide mark, 5–8 particles/l below; Sundet et al., 2017), Barentsburg (Granberg et al., 2019), and Longyearbyen (0 to 6.3 particles kg^{-1}; Sundet et al., 2016). Knowledge of microplastic uptake by organisms in Svalbard is limited, but thus far been seen in Northern Fulmars (Trevail et al., 2015), benthic amphipods (Iannilli et al., 2019), and juvenile cod (Kühn et al., 2018).

While numerous studies have confirmed the presence of microplastics within both Arctic marine and terrestrial environments (Lusher et al., 2015, Scott, 2019, and Bergmann et al., 2017a), few have focussed

Fig. 1. Svalbard and the location of the study area at Longyearbyen. Map insert shows location of the Svalbard archipelago within the northern Atlantic Ocean. Map data © OpenStreetMap, and is licensed under the Creative Commons Attribution-ShareAlike 2.0 license (CC BY-SA 2.0).

on investigating occurrence of microplastic pollution on Arctic beaches (Sundet et al., 2017), and fewer still on specific sources. Better quantification of microplastics in beach sediments is vital, as these sediments are considered long-term sinks for microplastics (Kanhai et al., 2018), with the potential to damage a diverse and vulnerable ecosystem (Halsband and Herzke, 2019). A greater understanding of microplastic occurrence and the sources will enhance our regional and global picture of microplastic pollution and provide an insight into how to mitigate ecosystem hazard (PAME, 2019).

This study aims to investigate the occurrence and abundance of microplastics on beaches in Adventfjorden, western Svalbard, to assess potential sources of microplastic pollution. The specific objectives were to:

- Characterize the presence, type and distribution of beach microplastics
- Investigate the sources of microplastics to those beaches

2. Methodology

2.1. Study site

Svalbard is a Norwegian Archipelago in the Arctic, located between mainland Norway and the North Pole at 74 - 81°N, surrounded by the Barents, Greenland and Norwegian Seas (Fig. 1). The archipelago is composed of nine main islands covering 61,299 km^2, with a 3500 km long coastline (Aamot, 2016). Svalbard is characterized by a mountainous landscape, glacially eroded fjords, and glaciers which cover 60 % of the land surface, and hosts terrestrial and marine wildlife including polar bears, whales, and walruses (Ingólfsson, 2004). The archipelago is influenced by two major currents: the warm West Spitsbergen Current and the cold East Current. The warm current passes the west coast of Svalbard, causing the west coast to be predominately ice free (Jaskolski et al., 2018). Longyearbyen is the northernmost permanently inhabited settlement and is located in the south coast of Adventfjorden, a 7 km long and 4 km wide bay running south-eastwards from the southern side of Isfjorden on the west coast of Spitsbergen, the largest island of the Svalbard Archipelago (Fig. 2). Adventfjorden is influenced by fjord dynamics, with sediment and freshwater inputs from glaciers and river systems, including the Adventelva and Longyearelva rivers. Both Isfjorden and Adventfjord shores vary between tidal flats, rocky coasts and sandy-gravel shores (Weslawski et al., 2011). Four sand-gravel beaches in Adventfjorden and Isfjorden (Fig. 2) were sampled in July 2019, with 10 samples of beach sediments collected at regular intervals along each beach (Fig. 4). Site 1 is the furthest from an inhabited settlement, adjacent to Isfjorden, between Platåberget and Pilarberget, close to the Bjørndalselva River. Site 2 is on the northern shore of Adventfjorden, opposite Longyearbyen and close to Advent City, which was abandoned in 1917 (Kruse et al., 2016). Sites 3 and 4 are closest to human activity; near the Kullkaia harbour (Site 3) and in Longyearbyen (Site 4).

2.2. Microplastic analysis

2.2.1. Sediment sampling, preparation and contamination control

5–10 g of surficial sediment was collected every 30 m along the strandline of the four sites as microplastic abundance is shown to be concentrated there (Sartain et al., 2018); 10 samples were collected at each site (Fig. 4). The samples were collected using a metal trowel and

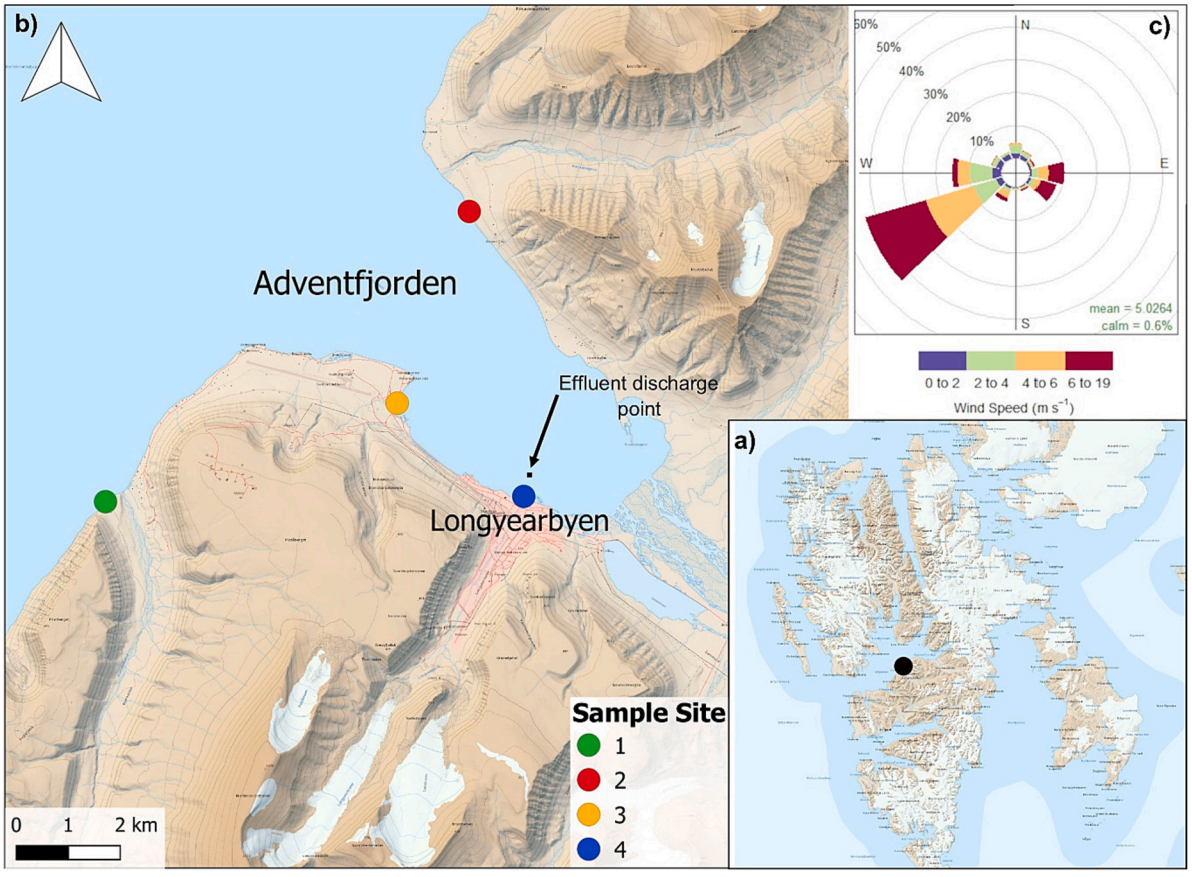

Fig. 2. Location data for the sampling sites around the shores of Adventfjorden, Svalbard. Map shows: a) location of sampling relative to Svalbard, b) higher scale view of Adventfjorden and the 4 sampling sites and location of effluent discharge point, c) wind rose showing percentage frequency of wind direction for Adventfjorden. Map data from The Norwegian Polar Institute under the Creative Commons Attribution 4.0 International (CC BY 4.0) license (Norwegian Polar Institute, n. d). Wind data from The Norwegian Meteorological Institute.

immediately wrapped in aluminium foil, to minimize contamination, then labelled. On return to the laboratory, samples were dried in a mild (40 °C) passive oven and then sieved through a 2 mm metal sieve to remove larger sediment particles before the digestion step. The contents of the sieve were studied for microplastics, to identify any microplastic particles over 2 mm so as not to exclude from analysis (c.f. Identification and quantification of microplastics). Once sieved, the dry weight of the samples was recorded, and they were transferred into glass beakers and labelled.

2.2.2. Digestion

Organic matter and calcium carbonate were removed by sequential treatment with 30 % H_2O_2 (at 50 °C for 1 h, then left to stand at room temperature for 24 h to remove organic matter) and 2 N HCl (at room temperature for 24 h; then at 60 °C for 1 h to remove carbonates), respectively for all samples and 4 procedural blanks (purified and pre-sieved sand; Martin et al., 2017). Glassware was carefully rinsed with deionised water in between analyses and at the end, ensuring minimal losses and removing excess H_2O_2 and HCl. Then the samples were centrifuged (3 min; 3000 rpm) and excess water was syphoned off once samples had settled (Ball, 2019). These steps were necessary to prevent interference during identification and quantification of microplastic particles (Hidalgo-Ruz et al., 2012).

2.2.3. Density separation and filtration

Each sample and procedural blank was transferred into a pre-washed 50 ml falcon tube and labelled. They were then topped up to 45 ml with filtered deionised water and centrifuged (HERMLE Z 446) at 3000 rpm for 3 min for further washing. The water was then replaced and the sample agitated using a vortex mixer for 30 s before repeating the washing process. Once washed, the water was removed using a 50 ml pipette and 30 ml of sodium polytungstate (SPT; 1.6 g cm^{-3}) was added before further agitation (Zhang et al., 2018). The samples and procedural blanks were then centrifuged (3000 rpm; 20 min) to float any potential microplastics from the sediment. Finally, the supernatant was decanted and subsequently vacuum filtered, using a three piece Hartley pattern filter funnel and 25 mm GF/F filter paper. The funnel wall and universal containers were washed with deionised water to ensure complete transfer of plastic particles (Bridson et al., 2020). Each filter paper was immediately stored and covered in foil cases and dried at 40 °C in a passive oven till further analyses. The remaining SPT solution was removed and filtered for recycling.

2.2.4. Identification and quantification of microplastics

Microplastic particles were detected under a stereomicroscope (ZEISS Stemi DV4) at 32× magnification. Filters were examined systematically in parallel rows to avoid overlapping and overestimation of microplastics. Microplastics were identified following established visual identification protocols (e.g. Norén, 2007; Ball, 2019), i.e. particles that are bright unnatural and homogenously coloured, particles with no visible cellular or organic structures, and fibres that are equally thick and did not taper at the ends. Identified microplastics were classified into three groups: fragments (hard, rough plastic particle), fibres (thin, elongated plastic) and pellets (rounded, solid plastic particles) (Hidalgo-Ruz et al., 2012). The type, colour and size of microplastics were recorded for each sample. Selected plastic-like particles were extracted from filter papers using tweezers and transferred to an FTIR spectrometer (Agilent Cary 630) equipped with a diamond ATR for polymer identification. To ensure that a representative subsample of microplastics was identified for the FTIR analysis, at least one particle per potential microplastic type and colour per sample filter was chosen. This led to a range of 43–56 % of samples being selected and analysed at each site. FTIR spectra were processed using an average of 32 co-scans, a spectral range of 650–4000 cm^{-1} and a resolution of 8 cm^{-1}. These spectra were compared with referenced spectra from the MicroLab Software Library to determine polymer composition. Matches of >70 % were positively identified as microplastics in this study (Sathish et al., 2019; Zheng et al., 2020). Only fibres were detected in the procedural blank samples and were taken into consideration when studying the other samples (Cole et al., 2014; Ball, 2019).

2.2.5. Contamination control

All surfaces were cleaned and inspected for airborne contamination prior to each analytical step. All glassware was washed twice with deionised water and dried at 40 °C before use and covered with glass plates or foil during all procedures to minimize airborne contamination. In the case that plastic containers were used, they were prewashed twice using filtered deionised water and inspected prior to use for contamination under the stereomicroscope. White cotton lab coats, non-synthetic clothing and nitrile gloves were worn throughout the analyses (Bridson et al., 2020). Additionally, to quantify microplastic contamination during the methodological steps and within the general laboratory setting, blanks were analysed. Methodological blanks were analysed using purified sand of equal weight to the subsamples and were used to assess contamination from the air throughout the digestion and filtration stages. Additionally, to directly assess atmospheric microplastic particles in the laboratory, filter papers were moistened and left in Petri dishes exposed to the air.

The contamination assessment identified a mean potential contamination for the procedural blanks of 0.16 n/g (number), indicating a low level of contamination overall. Microplastic contamination was in the form of fibres (Polyester fibres, $n = 3$). Fibres in the procedural blanks were visually different from the other fibres in the sediment samples and therefore no adjustments were made. Atmospheric blanks showed a mean of 2.4 fibres per sample, and thus, coupled with the limited contamination in the procedural blanks, the steps taken to control contamination were considered to be adequate and successful.

2.3. Statistical analysis

To identify significant differences within the sites, a Kruskal–Wallis test was performed on all 4 sites. An adjusted Wilcoxon test was then performed on pairs of sites as a post-hoc test to identify differences between them. *P* values were adjusted to counteract the issue of multiple comparisons using a Bonferroni correction (Holm, 1979). A principal component analysis (PCA) analysis was conducted on the total MP in n/g (number), fraction of each of the three types of MP and the distance of each sample to the nearest potential sources of microplastic. These sources were either urban impacted watercourses, urbanised land use, port land use, airport land use, or the effluent discharge point in Adventfjorden (78° 13.665 ′N, 15° 39.953 ′E, Vasskog et al., 2008, Fig. 2). Land use and potential urban impacted runoff were mapped using and openstreetmap.com within the QGIS software (QGIS 3.16, 2020) with the information at https://www.lokalstyre.no/arealplaner.486570.no.html used to confirm land use classifications. The location of the effluent discharge point was acquired from Vasskog et al. (2008). Additionally, elevation on the beach and bearing of the sample point down the beach to the sea were included to investigate azimuth of the sample down beach, to determine if wind direction impacted the concentration. Data was standardized to avoid the influence of different parameter scales. All statistical analysis was performed in R 4.1.0 (R Core Team, 2021) with a significance level (p) of 0.05 (Yu et al., 2016).

3. Results

3.1. Microplastic abundance and distribution

All sediment samples collected from the four beach sites were found to contain microplastics. The median microplastic concentration of the four sampled beaches was Site 1 (0.7 n/g); Site 2 (1 n/g); Site 3 (1.7 n/g); Site 4 (2.2 n/g) (Fig. 3a). A statistically significant difference (Kruskal-Wallis, $p < 0.001$) was observed between the four beach sites.

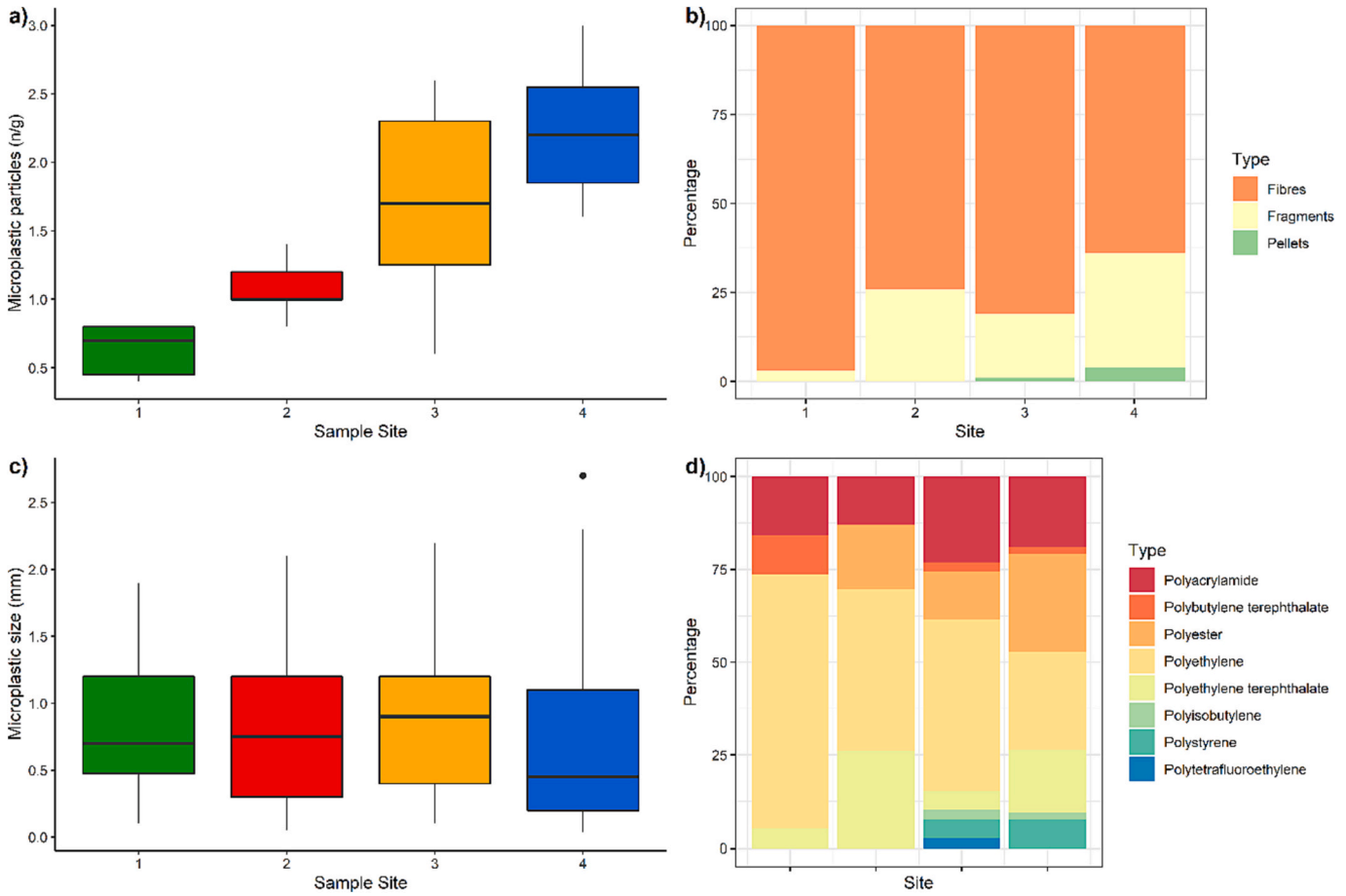

Fig. 3. Microplastic abundance, size, type and polymer for all four sites. a) Box and whisker diagrams of the microplastic abundances in n/g (number) for the 10 samples at each of the four sample sites. Each box and whisker diagram presents the median, 25th and 75th quartiles, minimum, and maximum MP abundances. b) Stacked bar chart comparing the percentage fraction of types of plastic particles (fibres, fragments and pellets) found at each of the sites. c) Box and whisker diagrams of the microplastic size (mm) for the 10 samples at each of the four sample sites. Each box and whisker diagram presents the median, 25th and 75th quartiles, minimum, and maximum MP abundance. d) Stacked bar graph comparing the percentage fractions of the different types of polymers identified at each of the sites.

Pairwise Wilcoxon tests showed significant differences between Site 1 and all other sites (p-adjusted <0.01 in all cases). There were no significant differences between Sites 2 and 3 (p-adjusted = 0.23), and Site 3 and 4 (p-adjusted = 0.44).

The microplastic abundance of each sample site was determined as the mean ± SD (standard deviation) of n/g (number). The mean ± SD abundance of microplastics for Site 1 was 0.64 ± 0.18, Site 2 was 1.08 ± 0.17, Site 3 was 1.68 ± 0.69 and Site 4 was 2.24 ±. 0.47 n/g (number). The highest abundance of microplastics was observed at Site 4 which is the site located at the foot of Longyearbyen town, by the outlet of the Longyear River which drains the town. The lowest abundance of microplastics was found in Site 1 which is situated furthest from inhabited settlements.

Looking within each site, the number of microplastics at Site 1 ranged from 0.4 to 0.8 n/g (number), Site 2 ranged from 0.8 to 1.4 n/g (number), Site 3 ranged from 1.8 to 2.6 n/g (number) and Site 4 ranged from 1.6 to 3 n/g (number). The distribution of microplastics was highly variable across each beach site (Fig. 4), with the lowest heterogeneity (lowest ranged and standard deviations) in concentrations in Sites 1 and 2, which are the most remote from Longyearbyen (Fig. 3a). Sites 3 and 4 and situated within Adventfjorden had the highest heterogeneity (largest ranges and standard deviations. Visually there are spatial correlations between the highest concentrations of microplastic and proximity to an industrial site at Site 3 (Fig. 4, Site 3), and at Site 4 microplastic concentrations increased towards the mouth of the Longyear River, where the highest concentration of 3 n/g (number) was observed (Fig. 4, Site 4).

3.2. Microplastic type and size

The microplastics found in the present study consisted of fibres, fragments, and pellets (Fig. 3b). Fibres were the most abundant particle type comprising of 75 % of all microplastics identified, while pellets were the least abundant microplastic type. Photographs of microplastics were taken (as shown in Fig. 5) to visually illustrate the different plastic particle types (Cox, 2018).

At Site 1, fibres are the most abundant plastic particle type with 97 % ± 0.88 fibres identified and a much lower proportion of fragments (3 % ± 0.32; Fig. 3b). Likewise, Site 2 had a greater abundance of fibres (74 % ± 1.05) and a lower proportion of fragments (26 % ± 1.26). No pellets were detected in either Site 1 or Site 2. Pellets were identified within Site 3 and Site 4 however only at very low proportions. At Site 3 pellets were identified at a percentage of 1 % ± 0.32 and a percentage of 4 % ± 0.52 at Site 4. Fibres were also the most abundant within Site 3 and Site 4, with a percentage of 81 % ± 2.57 (Site 3) and 64 % ± 1.75 (Site 4). A statistical difference was found in the abundance of fibres (Kruskal-Wallis, $p < 0.001$), fragments ($p < 0.001$) between the four sites. Pairwise, there was no significant difference in fibre count between sites 1 and 2 (p-adjusted = 0.36) and site 3 and 4 (p-adjusted = 1). There were significant differences between all other sites, e.g., site 1 and 4 (p-adjusted >0.001) and site 2 and 3 (p-adjusted = 0.04). Fragments showed significant differences between site 1 and the other three sites

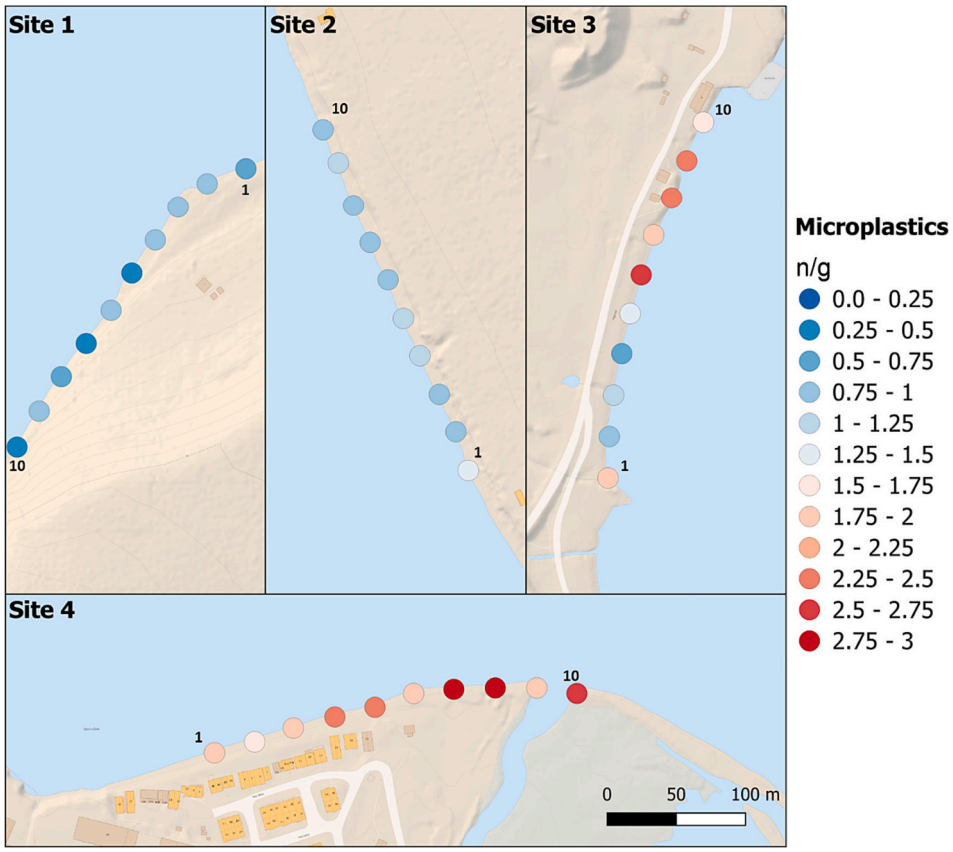

Fig. 4. Beach microplastic sampling sites for the 4 locations showing each of the 10 sampling points along each beach. Each point shows colour on a graduated scale with blue symbols showing MP abundances of 0–1.5 n/g (number), and red symbols showing abundances over 1.5 n/g (number). The first and last sampling locations are labelled (1–10). Map data from The Norwegian Polar Institute under the Creative Commons Attribution 4.0 International (CC BY 4.0) license (Norwegian Polar Institute, n.d). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

(p-adjusted <0.05 for all pairs). Sites 3 and 4 also showed a significant difference (p-adjusted = 0.02). There were no significant differences between the other sites. Pellets were not analysed statistically due to the very low numbers and absence at some sites.

The size of microplastics identified in this study ranged from 0.04 mm to 4 mm (Fig. 3c). Approximately 65 % of the detected microplastics were less than or equal to 1 mm, and the average microplastic size at each site were 0.82 ± 0.44 mm (Site 1), 0.84 ± 0.61 mm (Site 2), 0.91 ± 0.57 mm (Site 3) and 0.40 ± 0.62 mm (Site 4).

3.3. Microplastic polymer type

Eight different polymer types were identified within the samples as presented in Fig. 6, which displays the typical FTIR spectra of each dominant type of microplastic. Microplastic polymer types included polyethylene terephthalate, polyacrylamide, polyester, polyethylene, polyisobutylene, polybutylene terephthalate, polystyrene and polytetrafluoroethylene (Kanhai et al., 2018). The most common polymer types detected were polyethylene and polyacrylamide followed by polyester and then polyethylene terephthalate (Fig. 3d). There were several polymers only identified at Sites 3 and 4, closest to Longyearbyen; these were polyisobutylene, polystyrene, and polytetrafluoroethylene (which was only present at Site 3). There were no specific spatial patterns in the fractions of the other polymers, however the dominance of polyethylene decreased from Site 1 to 4.

3.4. Microplastic sources

Based on results from the PCA (Fig. 7), two principal components (PCs) comprised 72.7 % of the variance. The first principal component (PC1) accounted for 51.1 %, with the variance characterized by decreasing total microplastic concentrations and increasing distance to the effluent discharge point and port and urban land uses. The second principal component (PC2) accounted for 21.6 % of the variance, characterized more loosely by a decreasing fraction of fibres and increasing fraction of fragments and distance to the airport land use and runoff sources. The main contributing variables to the PCA were number of fragments, number of fibres, the overall total microplastic concentration, distance to the effluent source, and nearest measured Port/Urban land use with pellets contributing the least. A strong relationship was observed between the closeness of each site to the Port, Urban land use, and effluent discharge sources and the total microplastic concentration. A similar if slightly reduced relationship was also observed between these sources and the fraction of pellets and fragments. Essentially, proximity to these sources sees a higher amount of microplastic pollution and greater numbers of fragments and pellets. A moderate positive correlation was observed between the fraction of fibres, the effluent discharge point, and to a lesser extent the urban and port land uses. This likely stems from the relative dominance of fibres at the more remote sites (Fig. 3b). The PCA also showed an almost orthogonal relationship, indicating no correlation between azimuth (direction down beach to the sea) and the fraction of fragments and pellets, and a loose negative correlation with total microplastic concentration. Based on their contribution to PC1, as azimuth increases, microplastic concentrations decrease – reversing this, the closer to North the bearing down the beach, the higher the microplastic concentration, however correlations between the fractions of microplastic type and azimuth were unclear. The correlation between microplastic concentration and azimuth is pertinent, as the wind direction for Adventfjorden showed a dominant wind direction between 180° and 270° (Fig. 2), and the azimuth down

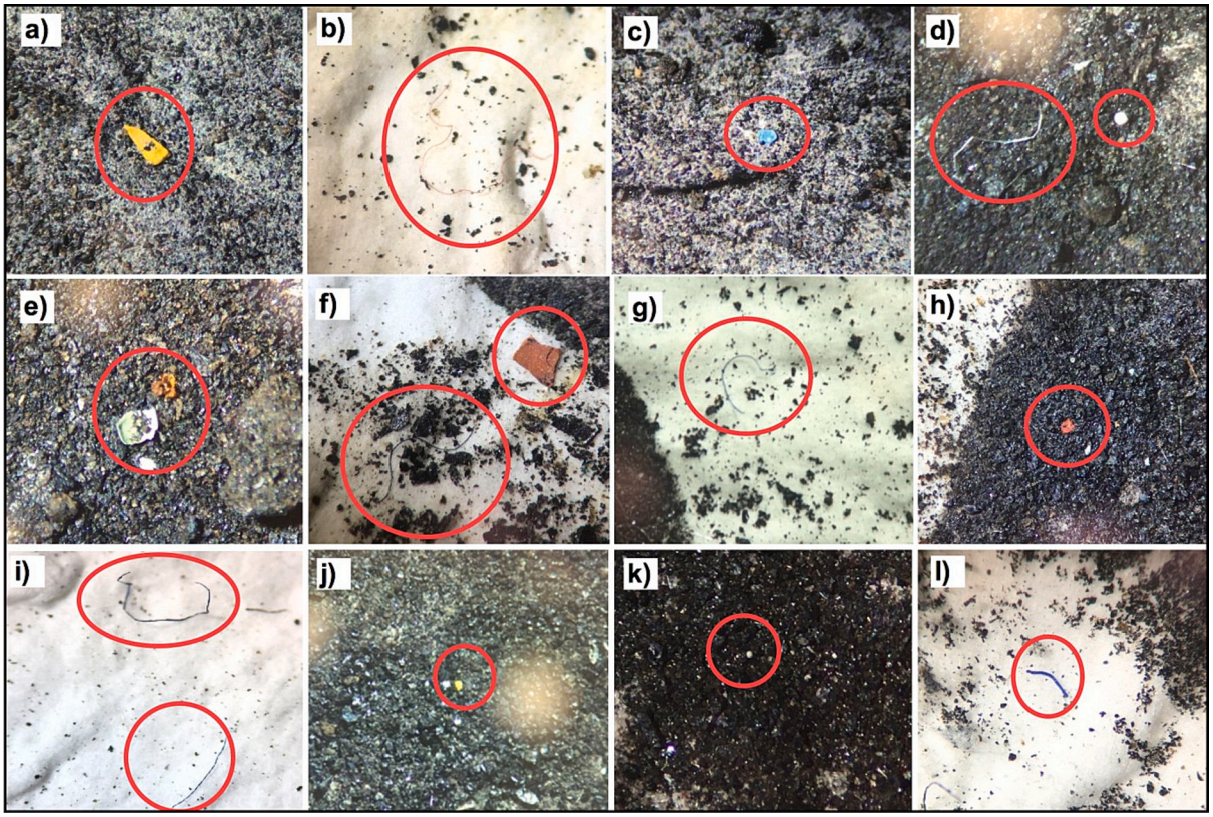

Fig. 5. Photographs of selected microplastics a) yellow fragment from Site 4. b) red fibre from Site 4. b. blue fragment from Site 4. d) transparent fibre and white pellet from Site 4. e) green fragment and orange fragment from Site 4. f) black fibre and red fragment from Site 4. g) blue fibre from Site 3. h) red fragment from Site 4. i) blue fire and black fibre from Site 2. j) yellow fragment from Site 3. k) cream pellet from Site 3. i) blue fibre from Site 1. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

the beach between 0° and 90° for Sites 3 and 4 with the highest microplastic concentrations.

4. Discussion

Microplastic pollution was evident at all four beach sites and every sample analysed contained microplastics. Whilst disappointing, this is unsurprising given the ever-growing understanding of how widespread microplastics are in the Arctic (Bergmann et al., 2022), demonstrating that these remote Arctic environments are far from immune to microplastic pollution (Kanhai et al., 2018). As with most other microplastic studies, we report a high occurrence of fibrous microplastics followed by fragments, while pellets were most infrequent, and as a consequence least contributory to statistical findings (Granberg et al., 2019; Tiwari et al., 2019). The high quantities of fibres and fragments suggests secondary microplastics were the most dominant form of plastic pollution, compared to primary microplastics (Shim et al., 2018). Reported microplastic abundances are similar to those of Choudhary et al. (2022), who found between 0.2 and 1.8 n/g (number) in fjord surface sediments. The presence of microplastics in Arctic beaches is a particular concern to the surrounding marine environment, as beaches are important environments for many species (Varotsos and Krapivin, 2018). Microplastic presence increases the possibility of marine organisms encountering plastic particles, which may have adverse toxicity effects and/or bioaccumulate through the arctic food web (Lusher et al., 2015).

A statistically significant difference was found in microplastic quantities between the four beach sites (specifically between S1 and the other sites), which suggests sources of microplastics differ between sample sites. The variability in microplastic abundance and distribution between the sites is likely to be affected by the variability in surrounding environments at each site, and more specifically the presence of effluent discharge and proximity to urbanised land uses, a finding similar to those of Yu et al. (2018) in the South-eastern USA. Urbanisation and human activity in the local areas increases the likelihood of macroplastic and microplastic pollution entering the environment (Dippo, 2012; Yu et al., 2018; Tibbetts et al., 2018), and despite substantially less urbanisation than other regions of the Earth, it also impacts on microplastic pollution in Arctic regions (Choudhary et al., 2022).

Sites 1 and 2 showed the lowest microplastic abundances (and lowest number of fibres and fragments). The remoteness of the sites, and statistical relationships between increased microplastic abundance and proximity to anthropogenic impacts suggests that the remoteness of a location leads to lower microplastic concentrations, a relationship observed elsewhere (e.g. Zheng et al., 2020). It is likely that the observed microplastic pollution at both sites is derived from ocean-based sources and supported by the dominance of fibres at Site 1 and 2, which compares well with the dominance of fibres observed in ocean waters (Ross et al., 2021). The identified microfibers are likely associated with the breakdown of larger plastic debris from fishing equipment and shipping activity, from both local and/or from polluted areas further south (Halsband and Herzke, 2019; Ross et al., 2021), as the relatively stagnant water of the Adventfjorden may preclude large microplastic particle fluxes from Adventfjorden to Isfjorden and the Arctic Ocean (Granberg et al., 2017). The microplastics fibres at the most remote sampling sites likely accumulated after long periods at sea, as 66 % of the microplastics found were less than or equal to 1 mm, suggesting long-range transport which successively breaks particles down into smaller pieces (Obbard, 2018). Long-range transport of plastic pollution in the ocean is predominantly controlled by oceanic currents such as the West Spitsbergen current, which also influences regional pollution sources (Jaskolski et al., 2018). This carries floating litter from northern Europe and the North Atlantic, as suggested by previous studies

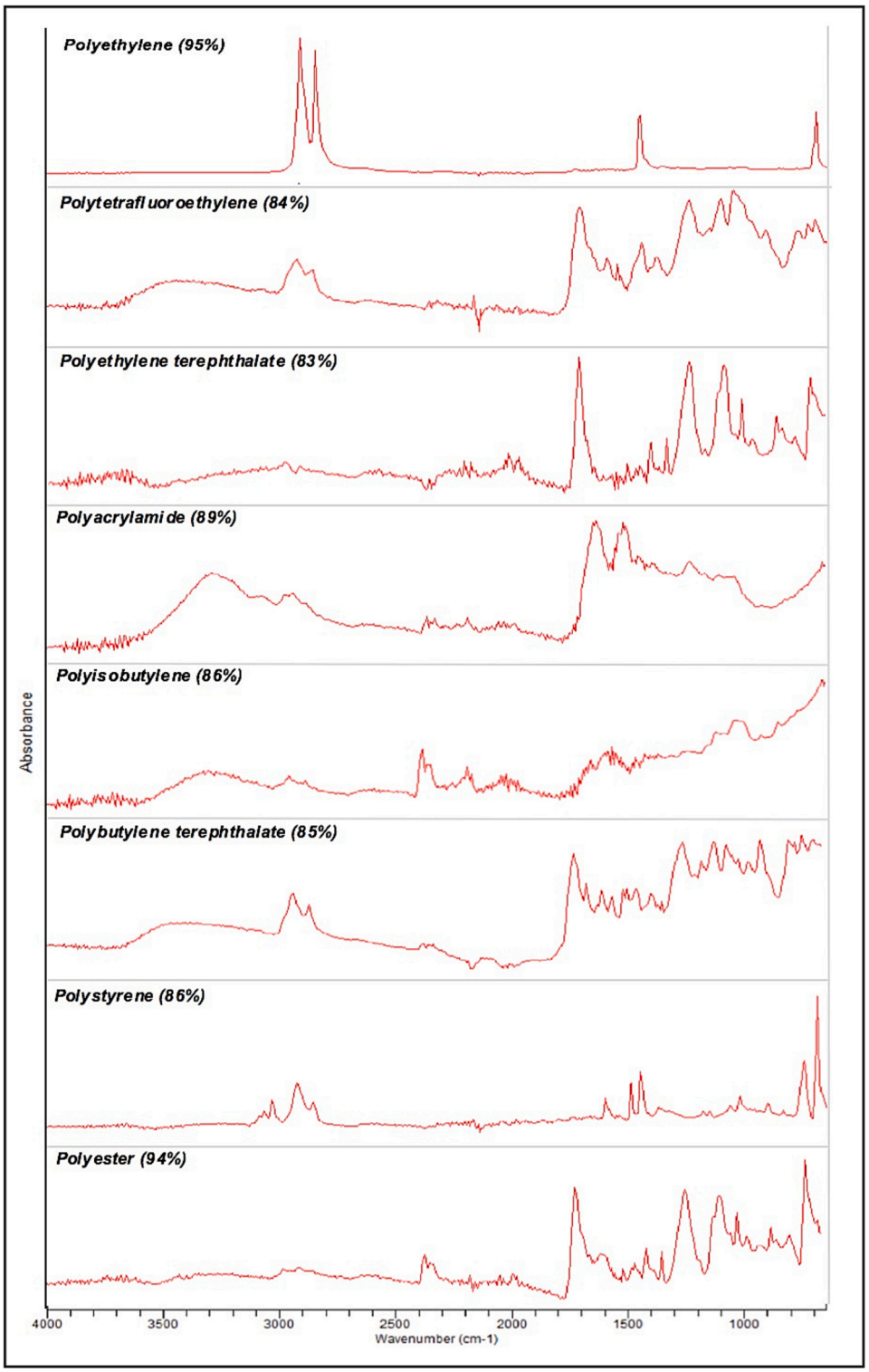

Fig. 6. Example FTIR spectrum of microplastics found in sediment samples and the percentage next to polymer names are the match degrees (%) of detected plastic particles to the MicroLab Software Library.

(Bergmann et al., 2017a; Cózar et al., 2017; Jaskolski et al., 2018).

The substantially higher microplastic concentration observed at the sites located at the foot of Longyearbyen town (Site 4 and to a lesser extent Site 3) indicates that there are different, more substantial sources of pollution at these sites compared to other sample sites (Sathish et al., 2019). Activities such as transportation and construction have also increased in Longyearbyen in recent years (i.e. increased hotel construction) in order to keep up with the demand of increasing tourism (see Jaskolski et al., 2018). These findings are supported by the contributions to the variance and correlations found in the PCA, with a likely source of microplastic pollution to the beaches from human activity at Longyearbyen being associated with the untreated sewage and wastewater that is released directly into the Adventfjord. The annual sewage discharge into Adventfjorden (>25 km2) is approximately 170,000 m3 (Granberg et al., 2017) – equivalent to 68 olympic swimming pools. Previous studies have detected high concentrations of microplastic

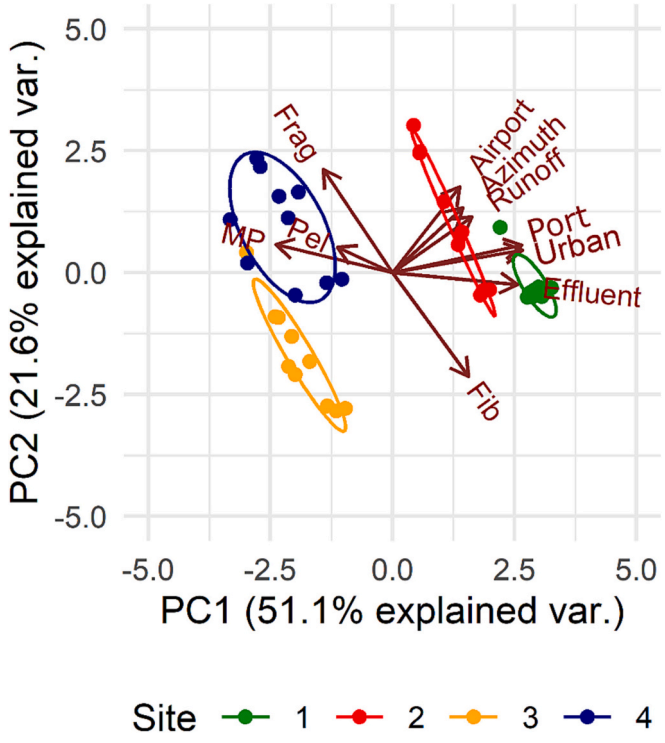

Fig. 7. Principal component analysis of microplastic concentration (MP), the fraction of fibres (Fib), fragments (Frag) and pellets (Pel) along with the distance to the effluent discharge point (Effluent), runoff sources (Runoff) and Airport, Urban, and Port land uses. It also includes the azimuth of the sample down the beach to the sea. Individual samples are indicated in different colours along with their sample site.

fibres within this wastewater (e.g. Sundet et al., 2016). Microplastics are likely to be incorporated in wastewater due to the release of fibres during the washing of synthetic fabrics, especially synthetic fleece garments. The high fibrous concentrations observed in wastewater by Sundet et al. (2016) is therefore not surprising, as the most common synthetic textile used in Longyearbyen is fleece due to the cold environment (De Falco et al., 2019). It should be noted that while the total number of fibres remained dominant on beaches closer to the effluent discharge, the relative fraction of fibres decreased with increases seen in both pellets and fragments. The relationship identified between increasing fractions of fragments and pellets (and total microplastic concentrations) and proximity to effluent discharge is complimentary to the high concentrations of all types of microplastic in effluent discharges to aquatic environments (Uddin et al., 2020). Although the effluent discharge outlet sits several hundred meters offshore (Fig. 2b), factors such as waves, surface currents and in particular the predominant northeasterly winds cause microplastics to mix with sea water and be washed ashore relatively quickly (Sagawa et al., 2018) and eventually to accumulate within beach sediment. This theory is supported by Granberg et al. (2019) who suggests that microplastics accumulate near wastewater discharge points.

Additionally, the relationship between proximity to urbanised land uses and microplastic abundance suggest that at these locations the drivers of increased microplastic pollution relate to the anthropogenic activities within them. Activities include road use (Kole et al., 2017), industrial and residential litter (Dris et al., 2018) and fishing/shipping activity (the breakdown of fishing gear - Jaskolski et al., 2018, and accidental spills - Renner, 2018) as all are locations which are primarily associated with the breakdown of larger plastic items (Ball, 2019), and as such, likely sources of microplastic pollution. At the sampling sites 3 and 4, plastic debris in the form of plastic cups, bottles, and lids, were present. and most likely derived from land sources such as direct deposition of litter or potentially litter brought by wind from other terrestrial sources. This observation supplements the lack of statistical relationship between direction of the beach (azimuth) and fragments, which given the relationship with total microplastic concentrations is surprising. However, the overall importance of azimuth as a variable in the PCA was low, and it's likely that the relationship between the fraction of fragments and the azimuth was complicated by these terrestrial activities as seen in the stronger negative relationship noted between proximity to Port and Urban land uses. This suggestion is also compounded by the dominance of Polyacrylamide (PAM) fragments at sites 3 and 4 (~50 %). PAM has a density above that of seawater (1.11 versus 1.03 g cm-3) and would likely sink to the sea bed (Gupta et al., 2021). However, macro-litter may also have been washed ashore from a combination of landward winds and waves/currents (Shim et al., 2018). The plastic litter is broken down over time into fragments and fibres through direct exposure to ultraviolet radiation from the sun and from the varied Arctic temperatures (Sundet et al., 2016). Other factors such as wind and wave action may also cause weathering of plastics into smaller particles (Tiwari et al., 2019).

The highest microplastic abundance was found at the mouth of the Longyear River. Statistical analysis did not provide a strong link between river activity and microplastic pollution variation. However, the river's close proximity to urbanised areas makes it likely that it acts as a conduit of plastic debris and microplastics from further inland areas of Longyearbyen town, which are then discharged into Adventfjorden, where they can then accumulate on beaches (Yu et al., 2016). These riverine inputs of microplastics may also be enhanced by high river flows associated with snow melt or heavy rain events, which can be observed in Longyearbyen (Shim et al., 2018).

A statistically significant difference in the abundance of fibres, fragments and pellets was also discovered among the four sites. This suggests that the variables determining the deposition and accumulation of the different microplastic types also differ between the different beaches (Ball, 2019). The higher proportion of pellets at the least remote sites further supports the theory that wastewater is a source of microplastic pollution, as microplastics pellets are mostly associated with cosmetic use (Sartain et al., 2018; Ball, 2019). This supports the work of Sundet et al. (2016), who suggested that microplastics pellets within the Adventfjord are from local human activity, either effluent discharge or activities such as industrial air blast cleaning media which can often be associated with machinery and boats at harbours (Browne, 2015). This could explain the pellet pollution at Site 3, in close proximity to the current port.

Another possible source of microplastics at all sites, but potentially dominant at Site 1, is aeolian transport. Aeolian transport is a potential reason for the microplastic pollution at the remote sites, as Bergmann et al. (2019) found atmospheric transport to be notable pathway in both the Alps and the Arctic. Aeolian transport may also be a contributing reason for the statistically significant difference in microplastic abundance between sites. However, there remains a lack of information regarding atmospheric transportation and distribution of microplastics, and so future work is needed to investigate aeolian transported microplastics; both globally and in the arctic where sources may be lower (Dris et al., 2018).

Analysis of polymer type of the observed plastic particles were used to help propose potential origins of the detected microplastics. Polyethylene and polyacrylamide were the most common polymer types detected within the sampled beaches, followed by polyester and polyethylene terephthalate. Those results are unsurprising, as polyethylene and polystyrene are the most common type of plastic produced (Cox, 2018). Polyester, ethylene-propylene copolymer, polyacrylamide, polyamide, polystyrene and polyethylene have been discovered within the Arctic Ocean by previous studies (Booth et al., 2017; Kanhai et al., 2018; Scott, 2019). Polyethylene terephthalate (PET) was most abundant at Site 4, which suggests the main sources being plastic litter and textiles, because PET is often used within clothing and in packaging

materials such as plastic bottles (Kanhai et al., 2018), which were present at the site. A high abundance of polyester was also found at Site 4 which again likely originates from textiles and effluent discharge (Sartain et al., 2018). The prevalence of polyethylene (PE) over polyester in beach sediments has been documented by several studies (e.g., Frias et al., 2010; Bridson et al., 2020), including this study, where polyethylene fibres dominated at all but the most urbanised beaches (Site 4). Interestingly, polyester fibres typically dominate microplastics water samples in the remote Arctic Ocean through mixing of Atlantic waters (Ross et al., 2021), which suggests further work is needed into understanding linkages between ocean water and beach deposition. The microplastic pellets at Site 4 were identified as both polyethylene and polystyrene which are the most common polymer type of pellets used in cosmetic products (Cole et al., 2011). Polyester, PET and polyethylene was observed by Granberg et al. (2019) within the wastewater of Ny-Ålesund in North-Western Spitsbergen, again indicating possible links between Site 4 and anthropogenic impacts and effluent discharge. Both PE and polyacrylamide were found within all the sampled beach sites, predominantly at Site 3, which most likely originates from fishing gear, such as fishing tackle and netting (Iannilli et al., 2019). High abundance of polyacrylamide was also detected in Svalbard by Iannilli et al. (2019), who found polyacrylamide to be the most dominant polymer type within benthic amphipods. Polystyrene was also observed within samples from the most urban impacted sites (3 & 4) which is a common polymer used within plastic utensil and food containers. Both plastic food containers and utensils are often used in recreational boating especially within tourist trips in Svalbard (Andrady, 2011).

5. Conclusion

This study reports microplastic abundance from remote and anthropogenically developed beach samples in Svalbard. Abundances found were similar to those identified in other areas of Svalbard and the Arctic. A clear link between anthropogenic impacts was identified suggesting human activities in the near vicinity of beach location are an important driver of microplastic pollution in what are typically considered as low human impact areas (Jacobson et al., 2019). While there is background pollution from oceanic waters driven by microplastic sources from regions further south in the Atlantic, local activities have a significant role in contributing microplastic particles to their immediate surroundings from terrestrial activities, and importantly, effluent discharge to oceans and coastal areas. This has implications for:

1. Understanding the causes and sources of microplastic pollution threatening arctic ecosystems
2. Understanding the contribution of local derived microplastic pollution in the food chain
3. The development of effective measures to mitigate arctic microplastic pollution

While this study has clearly identified the anthropogenic sources of microplastic pollution found on high Arctic beaches, more work is needed to quantify pollution from these sources and develop effective mitigation measures.

CRediT authorship contribution statement

Tesni Lloyd-Jones: Conceptualization, Methodology Writing – Original Draft, Methodology, Visualization, Formal analysis. **Jonathan J Dick:** Writing- Original draft preparation, Visualization, Formal analysis. **Timothy P Lane:** Writing- Original draft preparation, Writing - Review & Editing. **Eoghan M Cunningham:** Methodology, Formal analysis, Writing - Review & Editing. **Konstadinos Kiriakoulakis:** Conceptualization, Methodology, Resources, Supervision.

Declaration of competing interest

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

Data availability

Data will be made available on request.

References

Aamot, D.B., 2016. Sense of Svalbard a Study of Place Attachment through Winter Recreation on Svalbard, 1–89. Master's thesis. The Arctic University of Norway.

Ahmed, R., Hamid, A.K., Krebsbach, S.A., He, J., Wang, D., 2022. Critical review of microplastics removal from the environment. Chemosphere 133557.

Andrady, A.L., 2011. Microplastics in the marine environment. Mar. Pollut. Bull. 62, 1596–1605.

Aves, A.R., Revell, L.E., Gaw, S., Ruffell, H., Schuddeboom, A., Wotherspoon, N.E., LaRue, M., McDonald, A.J., 2022. First evidence of microplastics in Antarctic snow. Cryosphere 16 (6), 2127–2145.

Ball, H., 2019. Microplastic in Saltmarshes: Developing Extraction Methods and Investigating Past Accumulation, 1–61. Master's thesis. Liverpool John Moores University.

Bao, M., Huang, Q., Lu, Z., Collard, F., Cai, M., Huang, P., Yu, Y., Cheng, S., An, L., Wold, A., Gabrielsen, G.W., 2022. Investigation of microplastic pollution in Arctic fjord water: a case study of Rijpfjorden, Northern Svalbard. Environ. Sci. Pollut. Res. 1–10.

Bergmann, M., Lutz, B., Tekman, M.B., Gutow, L., 2017a. Citizen scientist reveal: marine litter pollutes Arctic beaches and affects wildlife. Mar. Pollut. Bull. 125, 535–540.

Bergmann, M., Wirzberger, V., Krumpen, T., Lorenz, C., Primpke, S., Tekman, M.B., Gerdts, G., 2017b. High quantities of microplastic in Arctic deep-sea sediments from the HAUSGARTEN observatory. Environ. Sci. Technol. 51, 11000–11010.

Bergmann, M., Mützel, S., Primpke, S., Tekman, M., Trachsel, J., Gerdts, G., 2019. White and wonderful? Microplastics prevail in snow from the Alps to the Arctic. Sci. Adv. 5 (8), 1–10.

Bergmann, M., Collard, F., Fabres, J., Gabrielsen, G.W., Provencher, J.F., Rochman, C.M., van Sebille, E., Tekman, M.B., 2022. Plastic pollution in the Arctic. Nat. Rev. Earth Environ. 3 (5), 323–337.

Booth, A.M., Kubowicz, S., Beegle-Krause, C.J., Skancke, J., Nordam, T., Landsem, E., Throne-Holst, M., Jahren, S., 2017. Microplastic in global and Norwegian marine environments: distributions, degradation, mechanisms and transport. SINTEF Ocean AS M- 918, 1–104.

Bridson, J.H., Patel, M., Lewis, A., Gaw, S., Parker, K., 2020. Microplastic contamination in Auckland (New Zealand) beach sediments. Mar. Pollut. Bull. 151, 1–8.

Browne, M.A., 2015. Sources and pathways of microplastics to habitats. Mar. Anthropogenic Litter 229–244.

Choudhary, S., Neelavanan, K., Saalim, S.M., 2022. Microplastics in the surface sediments of Krossfjord-Kongsfjord system, Svalbard, Arctic. Mar. Pollut. Bull. 176, 113452.

Cole, M., Lindeque, P., Halsband, C., Galloway, T.S., 2011. Microplastics as contaminants in the marine environment: a review. Mar. Pollut. Bull. 62, 2588–2597.

Cole, M., Webb, H., Lindeque, P.K., Fileman, E.S., Halsband, C., Galloway, T.S., 2014. Isolation of microplastics in biota-rich seawater samples and marine organisms. Sci. Rep. 4 (4528), 1–8.

Cox, K., 2018. Distribution, Abundance, and Spatial Variability of Microplastic Pollution in Surface Waters of Lake Superior, 1–96. Master's thesis. University of Waterloo.

Cózar, A., Martí, E., Duarte, C.M., García-de-Lomas, J., Sebille, E.V., 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. 1–8.

Cunningham, E.M., Ehlers, S.M., Dick, J.T., Sigwart, J.D., Linse, K., Dick, J.J., Kiriakoulakis, K., 2020. High abundances of microplastic pollution in deep-sea sediments: evidence from Antarctica and the Southern Ocean. Environ. Sci. Technol. 54 (21), 13661–13671.

De Falco, F., Di Pace, E., Cocca, M., Avella, M., 2019. The contribution of washing processes of synthetic clothes to microplastic pollution. Sci. Rep. 9 (1), 6633.

Dippo, B.M., 2012. Microplastics in the Coastal Environment of West Iceland, 1–53. Master's thesis. University of Akureyri.

Dris, R., Gasperi, J., Tassin, B., 2018. Sources and fate of microplastics in urban areas: a focus on Paris megacity. Freshw. Microplastics 69–83.

Eriksen, M., Lebreton, L.C., Carson, H.S., Thiel, M., Moore, C.J., Borerro, J.C., Galgani, F., Ryan, P.G., Reisser, J., 2014. Plastic pollution in the world's oceans: more than 5 trillion plastic pieces weighing over 250,000 tons afloat at sea. PloS one 9 (12), e111913.

Fang, C., Zheng, R., Zhang, Y., Hong, F., Mu, J., Chen, M., Song, P., Lin, L., Lin, H., Le, F., Bo, J., 2018. Microplastic contamination in benthic organisms from the Arctic and sub-Arctic regions. Chemosphere 209, 298–306.

Frias, J.P., Nash, R., 2019. Microplastics: finding a consensus on the definition. Mar. Pollut. Bull. 138, 145–147.

Frias, J.P.G.L., Sobral, P., Ferreira, A.M., 2010. Organic pollutants in microplastics from two beaches of the Portuguese coast. Mar. Pollut. Bull. 60 (11), 1988–1992.

Granberg, M.E., Ask, A., Gabrielsen, G.W., 2017. Local contamination in Svalbard-overview and suggestions for remediation actions. Norsk Polarinstitutt 044, 1–42.
Granberg, M., Friesen, L.W.V., Bach, L., Collard, F., Strand, J., Gabrielsen, G.W., 2019. Anthropogenic microliter in wastewater and marine samples from Ny-Ålesund, Barentsburg and Signehamna, Svalbard. In: IVL Swedish Environmental Research Institute, 373, pp. 1–27.
Gupta, P., Saha, M., Rathore, C., Suneel, V., Ray, D., Naik, A., Unnikrishnan, K., Dhivya, M., Daga, K., 2021. Spatial and seasonal variation of microplastics and possible sources in the estuarine system from central west coast of India. Environ. Pollut. 288, 117665.
Hallanger, I.G., Gabrielsen, G.W., 2018. Plastic in the European Arctic. Norsk Polarinstitutt 045, 1–16.
Halsband, C., Herzke, D., 2019. Plastic litter in the European Arctic: what do we know? Emerg. Contam. 5, 308–318.
Hidalgo-Ruz, V., Gutow, L., Thompson, R.C., Thiel, M., 2012. Microplastics in the marine environment: a review of the methods used for identification and quantification. Environ. Sci. Technol. 46, 3060–3075.
Holm, S., 1979. A simple sequentially rejective multiple test procedure. Scand. J. Stat. 65–70.
Iannilli, V., Pasquali, V., Setini, A., Corami, F., 2019. First evidence of microplastics ingestion in benthic amphipods from Svalbard. Environ. Res. 179, 108811.
Ingólfsson, Ó., 2004. Outline of Geography and Geology of Svalbard, 1–12. University of Iceland and UNIS.
Jacobson, A.P., Riggio, J., Tait, A., M., Baillie, J., E.M., 2019. Global areas of low human impact ('Low Impact Areas') and fragmentation of the natural world. Sci. Rep. 9 (1), 1–13. Fig 3.
Jambeck, J.R., Geyer, R., Wilcox, C., Siegler, T.R., Perryman, M., Andrady, A., Narayan, R., Law, K.L., 2015. Plastic waste inputs from land into the ocean. Science 347 (6223), 768–771.
Jaskolski, M., Pawlowski, Ł., Strzelecki, M., Zagorski, P., Lane, T.P., 2018. Trash on Arctic beach: coastal pollution along Calypsostranda, Bellsund, Svalbard. Pol. Polar Res. 39 (2), 211–221.
Jiang, W., Fang, J., Du, M., Gao, Y., Fang, J., Jiang, Z., 2022. Microplastics influence physiological processes, growth and reproduction in the Manila clam, Ruditapes philippinarum. Environ. Pollut. 293, 118502.
Kane, I.A., Clare, M.A., Miramontes, E., Wogelius, R., Rothwell, J.J., Garreau, P., Pohl, F., 2020. Seafloor microplastic hotspots controlled by deep-sea circulation. Science 368 (6495), 1140–1145.
Kanhai, L.D.K., Gårdfeldt, K., Lyashevska, O., Hassellöv, M., Thompson, R.C., O'Connor, I., 2018. Microplastics in sub-surface waters of the Arctic Central Basin. Mar. Pollut. Bull. 130, 8–18.
Kole, P.J., Löhr, A.J., Van Belleghem, F.G., Ragas, A.M., 2017. Wear and tear of tyres: a stealthy source of microplastics in the environment. Int. J. Environ. Res. Public Health 14 (10), 1265.
Kruse, F., Nobles, G., Jong, M.d., Bodegom, R.v., 2016. Human impact at Advent City. Post-evacuation Assess. Rep. 1–60.
Kühn, S., Schaafsma, F.L., Werven, B.V., Flores, H., Bergmann, M., Egelkraut-Holtus, M., Tekman, M.B., Franeker, J.A.V., 2018. Plastic ingestion by juvenile polar cod (*Boreogadus saida)* in the Arctic Ocean. Polar Biol. 41 (6), 1269–1278.
Li, J., Lusher, A.L., Rotchell, J.M., Deudero, S., Turra, A., Bråte, I.L.N., Sun, C., Hossain, M.S., Li, Q., Kolandhasamy, P., Shi, H., 2019. Using mussel as a global bioindicator of coastal microplastic pollution. Environ. Pollut. 244, 522–533.
Lusher, A., 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. 1–9.
Martin, J., Lusher, A., Thompson, R.C., Morley, A., 2017. The deposition and accumulation of microplastics in marine sediments and bottom water from the Irish continental shelf. Sci. Rep. 7, 10772.
Masura, J., Baker, J., Foster, G., Arthur, C., 2015. Laboratory Methods for the Analysis of Microplastics in the Marine Environment: Recommendation for quantifying synthetic particles in waters and sediments. NOAA Technical Memorandum NOS-OR&R, 48, pp. 1–37.
Napper, I.E., Davies, B.F., Clifford, H., Elvin, S., Koldewey, H.J., Mayewski, P.A., Miner, K.R., Potocki, M., Elmore, A.C., Gajurel, A.P., Thompson, R.C., 2020. Reaching new heights in plastic pollution—preliminary findings of microplastics on Mount Everest. One Earth 3 (5), 621–630.
Norén, F., 2007. Small Plastic Particles in Coastal Swedish Waters. N-Research Report, Commissioned by KIMO Sweden, pp. 1–11.
Norwegian Polar Institute. (n.d.) TopoSvalbard [online]. Available at. https://toposvalbard.npolar.no. (Accessed 1 March 2020).
Obbard, R.W., 2018. Microplastics in polar regions: the role of long range transport. Curr. Opin. Environ. Sci. Health 1, 24–29.
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.
PAME, 2019. Desktop study on marine litter including microplastics in the Arctic. Arctic Council 1–102.
Peeken, I., Bergmann, M., Gerdts, G., Katlein, C., Krumpen, T., Primpke, S., Tekamn, M., 2018. Microplastics in the Marine Realms of the Arctic with Special Emphasis on Sea Ice. Arctic Programme, p. 1.
Plastics Europe, 2022. Plastic- the Facts 2021: an analysis of European plastics production, demand and waste data [online]. Available at: https://www.plasticseurope.org/application/files/9715/7129/9584/FINAL_web_version_Plastics_the_facts2019_14102019.pdf [Accessed 27th February].
QGIS.org, 2020. QGIS geographic information system 3.16. QGIS association. http://www.qgis.org.
R Core Team, 2021. R: A language and environment for statistical computing. In: R Foundation for Statistical Computing, Vienna, Austria. https://www.R-project.org/.
Renner, K.O., 2018. Particle size analysis, Quantification and Identification of Microplastics in Selected Consumer Products: A critical Comparison of Methods and Analytic Techniques. Brunel University, pp. 1–211. PhD thesis.
Ross, P.S., Chastain, S., Vassilenko, E., Etemadifar, A., Zimmermann, S., Quesnel, S.A., Eert, J., Solomon, E., Patankar, S., Posacka, A.M., Williams, B., 2021. Pervasive distribution of polyester fibres in the Arctic Ocean is driven by Atlantic inputs. Nat. Commun. 12 (1), 1–9.
Sagawa, N., Kawaai, K., Hinata, H., 2018. Abundance and size of microplastics in a coastal sea: comparison among bottom sediment, beach sediment, and surface water. Mar. Pollut. Bull. 133, 532–542.
Sartain, M., Wessel, C., Sparks, E., 2018. Microplastics Sampling and Processing Guidebook. Agricultural Communications Mississippi State University, pp. 1–35.
Sathish, N., Jeyasanta, K.I., Patterson, 2019. Abundance, characteristics and surface degradation features of microplastics in beach sediments of five coastal areas in Tamil Nadu, India. Mar. Pollut. Bull. 142, 112–118.
Scott, N.J.H., 2019. Investigating the Environmental Partitioning of Microplastics in Two Contrasting Marine Ecosystems, 1–89. Master's thesis. University of Exeter.
Sharma, R., 2019. Experimental Analysis of Microplastics in Beach Sediment Samples by Density Separation and Microscopic Examination, 1–39. Degree thesis, ARCADA.
Shim, W.J., Hong, S.H., Eo, S., 2018. Marine microplastics: abundance, distribution, and composition. Microplastics Contam. Aquat. Environ. 1–26.
Solomon, O.O., Palanisami, T., 2016. Microplastics in the marine environment: current status, assessment methodologies, impacts and solutions. J. Pollut. Effects Control 4 (02), 1–13.
Sundet, J.H., Herzke, D., Jenssen, M., 2016. Forekomst og kilder av mikroplastikk I sediment, og konsekvenser for bunnlevende fisk og evertebrater på Svalbard. *Svalbards Miljøvernfond,* RIS-Project No.10495, pp. 1–13.
Sundet, J.H., Herzke, D., Jenssen, M., 2017. Forekomst av mikroplastikk i sjøvann, bunnsedimenter, fjæresediment og i filtrerende bunnorganismer i nære kystområder på Svalbard. *Svalbards Miljøvernfond,* RIS-Project No.10495, pp. 1–10.
Tibbetts, J., Krause, S., Lynch, I., Sambrook Smith, G.H., 2018. Abundance, distribution, and drivers of microplastic contamination in urban river environments. Water 10 (11), 1597.
Tiwari, M., Rathod, T.D., Ajmal, P.Y., Bhangare, R.C., Sahu, S.K., 2019. Distribution and characterization of microplastics in beach sand from three different Indian coastal environments. Mar. Pollut. Bull. 140, 262–273.
Trevail, A.M., Gabrielesen, G.W., Kühn, S., Franeker, J.A.V., 2015. Elevated levels of ingested plastic in a high Arctic seabird, the northern fulmar (*Fulmarus glacialis)*. Polar Biol. 38 (7), 975–981.
Uddin, S., Fowler, S.W., Behbehani, M., 2020. An assessment of microplastic inputs into the aquatic environment from wastewater streams. Mar. Pollut. Bull. 160, 111538.
Varotsos, C.A., Krapivin, V.F., 2018. Pollution of Arctic waters has reached a critical point: an innovative approach to this problem. Water Air Soil Pollut. 229 (11), 1–14.
Vasskog, T., Anderssen, T., Pedersen-Bjergaard, S., Kallenborn, R., Jensen, E., 2008. Occurrence of selective serotonin reuptake inhibitors in sewage and receiving waters at Spitsbergen and in Norway. J. Chromatogr. A 1185 (2), 194–205.
Weslawski, J.M., Gluchowska, M., Kotwicki, L., Szczuciński, W., Tatarek, A.M., Wiktor, J. M., Wlodarska-Kowalczuk, M., 2011. Adventfjorden: Arctic Sea in the Backyard. Institute of Oceanology PAS, Sopot, pp. 1–39.
Yu, X., Peng, J., Wang, J., Wang, K., Bao, S., 2016. Occurrence of microplastics in the beach sand of the Chinese inner sea: the Bohai Sea. Environ. Pollut. 214, 722–730.
Yu, X., Ladewig, S., Bao, S., Toline, C.A., Whitmire, S., Chow, T.A., 2018. Occurrence and distribution of microplastics at selected coastal sites along the southeastern United States. Sci. Total Environ. 613-614, 298–305.
Zhang, S., Yang, X., Gertsen, H., Peters, P., Salánki, T., Geissen, V., 2018. A simple method for the extraction and identification of light density microplastics from soil. Sci. Total Environ. 616, 1056–1065.
Zheng, Y., Li, J., Cao, W., Jiang, F., Zhao, C., Ding, H., Wang, M., Gao, F., Sun, C., 2020. Vertical distribution of microplastics in bay sediment reflecting effects of sedimentation dynamics and anthropogenic activities. Mar. Pollut. Bull. 152 (110885), 1–8.