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Anthropogenic microlitter in wastewater and marine samples from Ny-Ålesund, Barentsburg and Signehamna, Svalbard

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In cooperation with The Norwegian Polar Institute and Aarhus University, Denmark

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1 Summary

Plastic pollution is recognized as a serious threat to the marine environment by the UN, the EU through the Marine Strategy Framework Directive, the Arctic council, the Nordic council, and national governments worldwide. Marine litter may reach the Arctic with ocean currents from global and regional sources, but may also originate from local emissions related to shipping and fishing activities, runoff from land based industries, dumping sites and wastewater outlets. Wastewater outlets are identified as important sources of microplastics to the marine environment in temperate areas but have received less attention in the Arctic. Wastewater treatment is generally lacking in the Arctic, and in smaller settlements, handling of wastewater (including sewage water) and garbage is comparable to conditions observed in developing countries. In Svalbard, wastewater treatment is absent aside from a small treatment plant in Hornsund and the recently installed treatment plant in Ny-Ålesund.

The aim of this investigation was to quantify and characterize anthropogenic microparticles (AMPs: particles <5 mm of manmade or modified materials, e.g. plastics, paints, rubber and textile fibers) in wastewater from the recently installed treatment plant in Ny-Ålesund (Kongsfjorden), in the marine environment close to the Russian settlement Barentsburg (Grønnfjorden), and in Signehamna (Krossfjorden) far from permanent land-based human impact. Samples of seawater, marine sediments and beach sediments were collected and analyzed for AMPs using optical and spectroscopic techniques.

AMPs were found at all sites and in all matrices investigated in this study. AMPs identified in wastewater and seawater were dominated by fibres (both non-synthetic and synthetic) while AMPs identified in sediment samples were dominated by fragments. Higher concentrations of AMPs and higher polymeric diversity was observed closer to human activities; in sediments close to Ny-Ålesund wastewater outlet and in seawater close to both Barentsburg and Ny-Ålesund, as compared to the remote site at Signehamna.

Based on this investigation, as much as 99 % of the incoming AMPs may be retained by the wastewater treatment plant in Ny-Ålesund. It is thus clear that installation of appropriate wastewater treatment systems can substantially reduce the release of anthropogenic microlitter and potential associated contaminants to the marine environment. However, this investigation is based on a limited number of samples and a comprehensive investigation should be conducted to determine the true efficiency of the wastewater treatment plant in Ny-Ålesund, covering both temporal and spatial variation. The present report should be used as a first step towards resolving the issue of lacking wastewater treatment in Svalbard and in the Arctic as a whole.

2 Sammendrag

Plastforurensning er anerkjent som et alvorlig problem for det marine miljøet av UN, EU gjennom «Marine Strategy Framework Directive», Arktis Råd, og Nordisk Råd samt nasjonale regjeringer fra hele verden. Plastsøppel og mikropartikler kan nå arktiske områder via havstrømmer fra globale og regionale kilder, men kan også komme fra lokale kilder som fiskerier, skipsaktivitet, avrenning fra lokal industri, søppelfyllinger og avløpsvann/kloakk. Avløpsvann og kloakk er en viktig kilde for mikroplast til det marine miljøet i tempererte områder, men får mindre oppmerksomhet i Arktis. Som regel mangler rensing av kloakk og avløpsvann i Arktis. Fra de mindre bosettingene i Arktis er håndteringen av kloakk, avløpsvann og søppel sammenlignbart med tilstandene i utviklingsland. På Svalbard finnes det ikke noe avløpsrensing bortsett fra et lite renseanlegg i Hornsund og et nylig installert anlegg i Ny-Ålesund.

Målet av denne undersøkelsen er å kvantifisere og karakterisere antropogene mikropartikler (AMPs; fragmenter < 5mm fra menneskeskapte eller modifiserte materialer, for eksempel plast, maling, eller tekstile fibere) i avløpsvannet fra det nylig installerte renseanlegget i Ny-Ålesund (Kongsfjorden), fra den russiske bosettingen Barentsburg (Grønnfjord), og fra Signehamna (Krossfjorden) som ligger langt unna menneskelig påvirkning. Prøver av sjøvann, marine sedimenter og strandsedimenter ble innsamlet og analysert for AMPs med hjelp av stereomikroskopiske og FTIR teknikk.

AMPs ble funnet på alle stasjoner og i alle matrikser som ble undersøkt i dette studiet. Partiklene i avløpsvann og sjøvann besto mest av fibere (både syntetisk og ikke syntetisk), mens partiklene fra sedimentprøvene var dominert av fragmenter. Høye konsentrasjoner og høye polymerisk mangfold ble observert i nærheten av bosetningene, i sedimentene ved utløp av avløpsvann i Ny-Ålesund, og i sjøvann fra både Barentsburg og Ny-Ålesund når en sammenlignet med fjerne steder som Signehamna.

Basert på denne undersøkelsen konkluderes det med at opptil 99% av de innkommende AMPs fanges opp av rensingsanlegget i Ny-Ålesund. Dette viser at installasjon av et renseanlegg er viktig for å redusere utslipp av antropogene mikropartikler og assosierte miljøgifter til det marine miljøet i Arktis. Denne undersøkelsen er basert på et begrenset antall prøver. En større og mer omfattende undersøkelse bør gjennomføres for å vurdere den faktiske effekten av rensningsanlegget i Ny-Ålesund, dette gjelder både i tid og rom. Denne rapporten bør bli brukt som et første steg for å løse problemet av den manglende avløpsvannbehandlingen på Svalbard og i hele Arktis.

3 Background and aim of study

Plastic pollution is recognized as a serious threat to the marine environment by the UN (UNEP 2009, 2016), the EU through the Marine Strategy Framework Directive (2008/56/EC), the Arctic council (AMAP/PAME), the Nordic council (Nordisk Miljøhandlingsprogram 2013–2018) and national governments worldwide. Up to 12.7 million tons of plastic waste is estimated to be discharged into the marine environment annually on a global scale, and discharge is expected to increase in the coming years (Eriksen et al. 2014, Jambeck et al. 2015). Recent observations of plastic litter drifting at sea and accumulating in garbage patches are alarming, and the extent of this pollution is only starting to be unraveled. Plastic pollution embraces different sized items or particles from larger marine litter to micro- and nanoplastics (Galgani et al. 2015). While monitoring of plastic macrolitter is comprehensible, biodegradation processes and impacts of, e.g. plastic ingestion and plastic associated contaminants on living organisms is yet largely unknown. Issues regarding micro- (0.001-5 mm) and nanoplastics (<000.1 mm) are even more complex and understudied, and appropriate monitoring methods are currently being developed and harmonized. Because of this, it is at present difficult to evaluate the severity of the problem (Löder 2015).

Microplastic particles originate from a multitude of sources (Lassen et al. 2015, Magnusson et al. 2016a). Some are produced and emitted in the shape of microscopic particles, e.g. industrial plastic pellets, microplastics from hygiene products or plastic granulates from artificial turfs. Others are formed when larger plastic objects are fragmented into smaller pieces as a result of weathering. In all shapes, it has potential of being carriers of contaminants either incorporated during production, or sorbed to the particles during their environmental journey. Negative effects on organisms, populations and ecosystems involve both direct physical effects of the plastic particles themselves and possible exposure to various plastic associated hazardous substances (Teuten et al. 2009, Engler 2012, Herzke et al. 2016).

The Arctic

Due to the remote location of the Arctic, there are still only a limited number of studies on microplastics carried out in this region (GESAMP 2015, Lusher et al. 2015, Trevail et al. 2015b). Recent investigations show that microplastic particles are found throughout the water column in the Arctic central basin (Kanhai et al. 2018), on the sea floor down to 5 500 m depth (Bergmann and Klages 2012, Bergmann et al. 2017), frozen into the lower turbid layer of sea ice from the Arctic Ocean (Obbard et al. 2014, Peeken et al. 2018), and as larger plastic fragments in northern fulmars (*Fulmarus glacialis*) (Trevail et al. 2015a).

Marine litter may reach the Arctic with ocean currents from global and regional sources, but may also originate from local emission related to shipping and fishing activities, runoff from land based industries, dumping sites and wastewater outlets. Wastewater (including sewage, storm water and industrial effluent) outlets are identified as important sources of microplastics to the marine environment in temperate areas. A recent investigation, comparing wastewater treatment systems in Sweden, Finland and Iceland

showed that multi-step wastewater treatment plants retained up to 99% of inflowing microplastic particles while mechanical separation retained 0% of the inflowing microplastics, i.e. leading to 100% emissions (Magnusson et al. 2016b).

Wastewater treatment is generally lacking in the Arctic, and in smaller settlements wastewater and garbage disposal is comparable to conditions observed in developing countries. Important reasons for poor wastewater treatment are likely high costs related to warming and maintenance of treatment ponds and plants, as well as costly and inaccessible shipping routes. Consequently, municipal, industrial and hospital wastewater is often discharged directly into the sea. In the wake of climate change, industrial development and tourism is expected to increase in the Arctic. This leads to temporal population increases in these ecologically sensitive areas with a highly underprovided municipal infrastructure. The relative importance of global, regional and local sources for microplastic pollution is currently unknown, and the impact of microplastics on coastal marine organisms, ecosystems and resources in the Arctic remains yet to be determined.

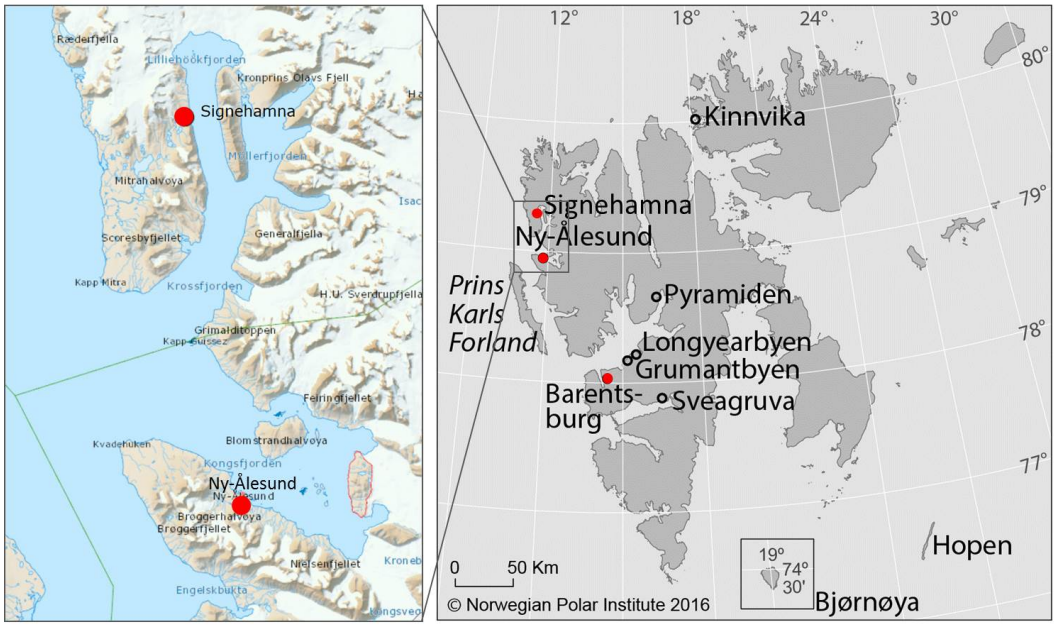

Figure 1. Map of Svalbard with main settlements and sampling sites indicated with red circles. Close-up of the Kongsfjorden-Krossfjorden system with Signehamna and Ny-Ålesund (left figure source: TopoSvalbard, Norwegian Polar Institute).

Aim

The aim of this investigation was to quantify and characterize anthropogenic microparticles (AMPs) in incoming and outgoing wastewater from the recently installed wastewater treatment plant in Ny-Ålesund, as well as close to the wastewater outlet of Barentsburg in Grønnfjorden, Svalbard. AMPs are here defined as particles <5 mm of manmade or modified materials, e.g. plastics, paints, rubber and textile fibers. The aim was further to determine AMP concentrations and characteristics in sediments, beach sand and seawater in the recipients Kongsfjorden and Grønnfjorden, as well as far from local sources in Signehamna, Krossfjorden (Fig. 1).

4 Methodology

4.1 Sites and sample collection

Field sampling sites were situated in Svalbard close to the wastewater outlet in Ny-Ålesund, Kongsfjorden, by the wastewater outlet and dock in Barentsburg, Grønnfjorden, and away from land-based human activities in Signehamna, Krossfjorden (Fig. 1, Table 1).

Table 1. Sampling sites with sample matrix, number of replicates (n) and geographic position.

Site	Region	Sample matrix	n	Position
Ny-Ålesund Incoming	Kongsfjorden	Wastewater	1	-
Ny-Ålesund Outgoing	Kongsfjorden	Wastewater	1	-
Ny-Ålesund WWO	Kongsfjorden	Sediment	2	78°55.720'N, 11°56.975'E
Ny-Ålesund WWO	Kongsfjorden	Seawater	1	78°55.720'N, 11°56.975'E
Signehamna	Krossfjorden	Sediment	3	79°16.243'N, 11°32.036'E
Signehamna	Krossfjorden	Seawater	1	79°16.243'N, 11°32.036'E
Barentsburg WWO	Grønnfjorden	Sediment	3	78°03.975'N, 14°11.850'E
Barentsburg WWO	Grønnfjorden	Seawater	1	78°03.975'N, 14°11.850'E
Barentsburg WWO	Grønnfjorden	Beach sediment	1	78°03.975'N, 14°11.850'E
Barentsburg dock	Grønnfjorden	Sediment	1	78°03.521'N, 14°12.227'E

WWO indicates wastewater outlet.

Ny-Ålesund is located in Kongsfjorden (Fig. 1 & 2) and was founded as a mining settlement by Kings Bay AS in 1917 and terminated as such in 1963. It is now run exclusively as an international research facility, hosting ~50 persons in winter and ~170 in the summer. The community of Ny-Ålesund, including the research facilities and infrastructure, is operated by Kings Bay AS under the Norwegian Ministry of Climate and Environment. As one of the first settlements on Svalbard, Kings Bay AS installed a wastewater treatment plant in Ny-Ålesund summer 2015. The treatment plant collects all wastewater from the settlement and consists of a sedimentation step and chemical and biological treatment steps. The outgoing wastewater is released into the Kongsfjorden. There have been no previous measurements of microplastic or other AMPs content in wastewater from Ny-Ålesund.

Figure 2. Ny-Ålesund remotely situated in North-Western Spitsbergen, here seen from the Kongsfjorden. Photo: Maria Granberg.

Barentsburg is an active coal mining site located at the mouth of Grønnfjorden (Fig. 1 & 3). The settlement harbors 4-500 inhabitants including children. Barentsburg is together with Longyearbyen the most extensive functional coal mining settlement in Svalbard. Several studies have been conducted to determine environmental pollution and risks at these sites. For an overview and references, see Granberg et al. (2017).

Figure 3. Barentsburg. Wastewater and storm water runoff (A) and sampling of beach sand in the runoff stream (B). Photo: Maria Granberg.

The bay **Signehamna** is located in Lilliehöökfjorden, i.e. the North-Western arm of the Krossfjorden close to the Lilliehöökbre glacier front (Fig. 1 & 4). A meteorological station was established by the German Air Force during World War II in the uninhabited wilderness of Signehamna/Signedalen.

Figure 4. Field sampling in Signehamna. Photo: Ingrid Gabrielsen.

All samples were collected during the summer of 2017 in collaboration with the Norwegian Polar Institute and Aarhus University in conjunction with a two-year project financed by the Nordic Council of Ministers aiming to understand the role of local pollution sources of microplastics in coastal benthic systems of Svalbard and Arctic Greenland. Beach sand was collected using a metal shovel and stored frozen in glass jars until analysis. Sediments were collected from a small boat using a hand operated Van Veen grab. Sediment samples were collected from ~10-30 m depth depending on site. The uppermost bioturbated 5 cm of the sediment was carefully retrieved and stored frozen in 1 liter glass jars until analysis.

Water samples were collected using an electrical water pump, pumping up to 1 m^3 seawater through sequential filters (100 and 50 µm mesh size) *in situ* (Fig. 5). Prior to pumping, pre-cleaned filters were mounted in a metal holder, which was submerged in a vertical position to a depth of 1 m with the filters facing upward (Fig. 5B). During pumping the water was sucked in at a slow flowrate across the filters. The pumping speed was recorded at least three times during pumping and the volume was derived by integration using pumping rates and total pumping time. An average volume of 690 L seawater was filtered.

Wastewater samples were collected from the incoming (before treatment) and outgoing (after treatment) wastewater supplies at the wastewater treatment plant in Ny-Ålesund. All collected samples were packed and shipped to the IVL laboratory at Kristineberg Marine Research and Innovation Center, Sweden for extraction and visual analyses. Spectroscopic analyses were performed by IVL at Aarhus University, Denmark.

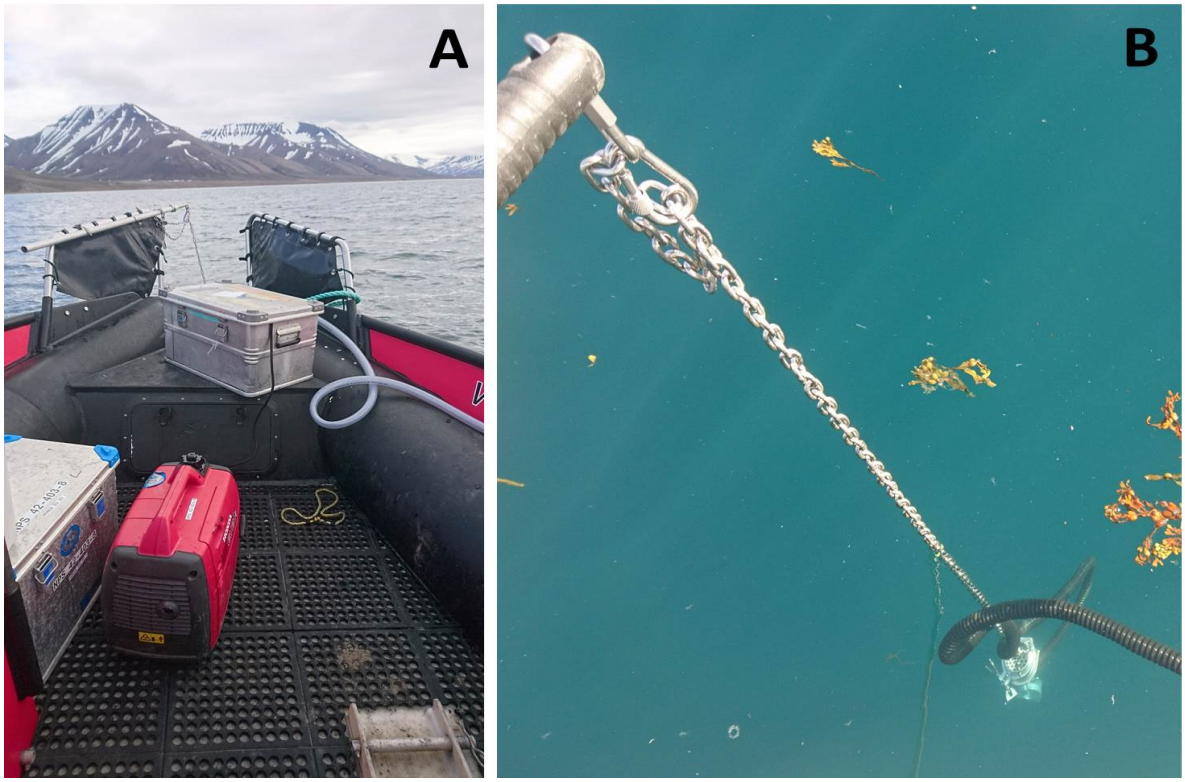

Figure 5. *In situ* pumping for sampling of anthropogenic microparticles in seawater. A) setup of pumping equipment in the field, B) submerged metal filter holder. Photo: Maria Granberg.

4.2 Sample processing and AMP extraction

4.2.1 Sediment and beach sand

The extraction of AMPs from sediment and beach sand was made by density separation in a set-up designed by IVL Swedish Environmental Research Institute and University of Gothenburg for this purpose, originally based on a design by Imhof et al (2012). The device consists of three main parts; I) an engine driving a rotating device situated in the bottom of a sediment container, II) a high conically shaped standpipe and III) a device with a ball valve which enables the division of the sample and sequenced filtering (Fig. 6). Sediment samples were homogenised, subsampled (450-650 g wet weight (WW)) and added to the density separator together with 4.5 L of saturated NaCl prepared with 0.22 µm filtered Milli-Q water (18.2 M Ω cm TC, 0.22 µm, Millipore). When density separation was completed, the solution containing extracted AMPs was vacuum filtered through nylon filters (Sefar Nitex) with the smallest mesh size of 20 µm. Filters were then stored individually in petri dishes. To determine the water content and establish the dry weight (DW) to WW ratios, subsamples (n=3) of each sediment sample were dried in pre-weighed aluminium containers at 105 °C until constant weight was reached. AMP abundance in sediment was reported as numbers (i.e. counts) per unit mass (DW) rather than volume, as suggested by Hanvey et al (2017).

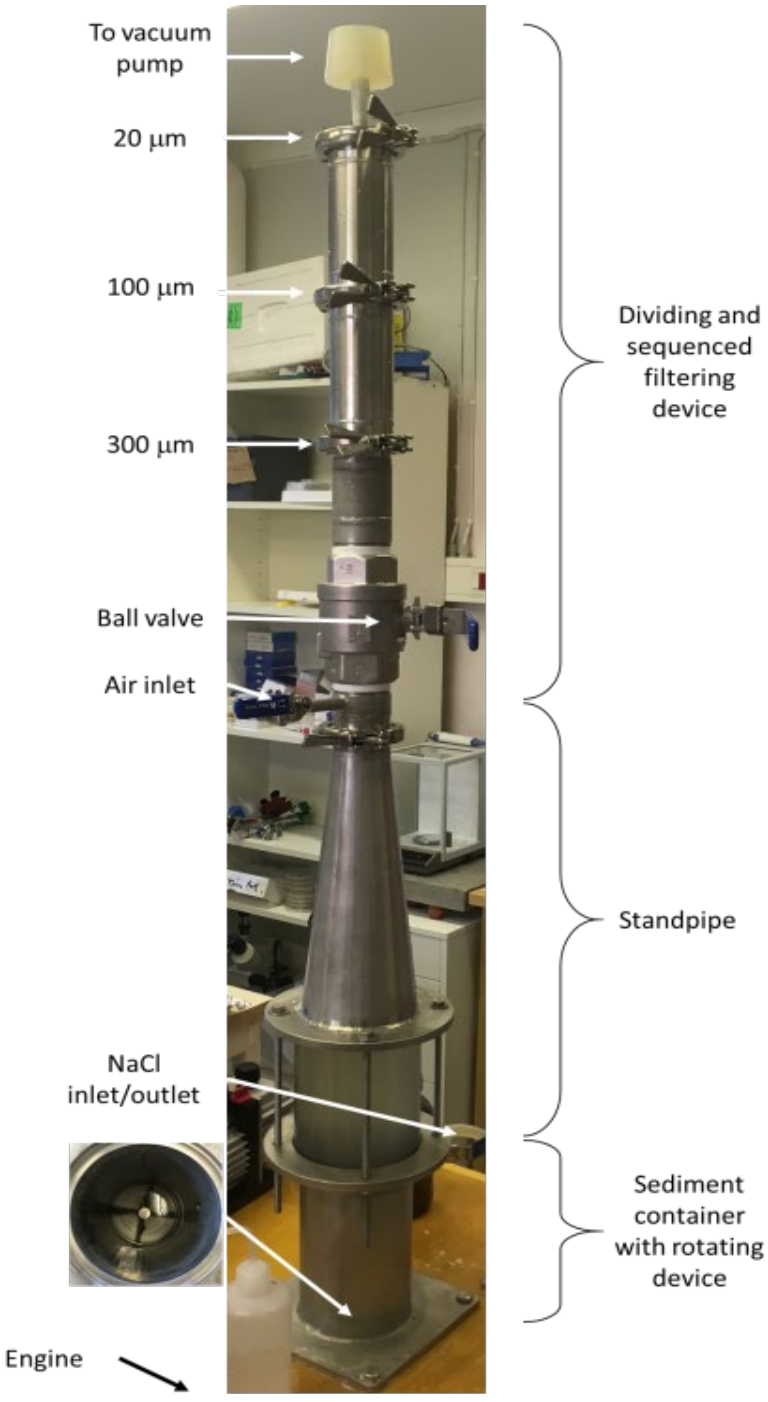

Figure 6. Set-up for density separation and extraction of AMPs from sediment and beach sand.

4.2.2 Seawater

Seawater samples were not subjected to any pre-treatment during sampling or analysis.

4.2.3 Wastewater

A gentle and effective digestion protocol developed at IVL with pancreatic enzymes (Creon® 40 000, Abbott Laboratories GmbH, Germany, Mylan) was applied for the extraction of AMPs from wastewater (von Friesen et al. Submitted manuscript). The pancreatic enzymes used originate from porcine pancreas and contain lipase (40000 Ph.Eur), amylase (25000 Ph.Eur) and protease (1600 Ph.Eur) as active substances. The enzymes were added together with Tris-hydrochloride solution (Trizma, pH 8.0, 1 M, 0.2 µm filtered, Sigma-Aldrich, T3038, USA). Tris-hydrochloride solution was added until the sample pH reached the optimal performing range of pancreatic enzymes (8 ± 0.1), determined with pH indicator strips (pH-Fix 7.0-14.0, Macherey-Nagel). Samples were incubated in 37.5 °C on 145 rpm overnight (Innova 40, Incubator Shaker Series, New Brunswick Scientific). After digestion, the solution was vacuum filtered through nylon filters (Sefar Nitex) with the smallest mesh size of 20 µm. Filters were then stored individually in petri dishes until analysis.

Both incoming and outgoing wastewater from the wastewater treatment plant in Ny-Ålesund was treated with the pancreatic enzyme digestion protocol. Outgoing water was treated after filtration (4710 ml) directly on the filters (100 and 20 µm) (0.7 g pancreatic enzymes, 10 ml tris-hydrochloride solution) whereas incoming water was treated before filtration (150 ml) due to its high contents of organic matter (2.1 g pancreatic enzymes, 100 ml tris-hydrochloride solution).

4.3 Analysis of anthropogenic microparticles

4.3.1 Visual analysis

The filters were examined under a stereomicroscope (Leica M205C) with a maximum magnification of 160x where potential AMPs were classified based on shape (evenness, roundness), colour (homogeneity, shininess, unnatural) and texture (stiffness). All suspected AMPs were photographed with a camera (Leica DFC420C) mounted on the stereomicroscope and processed in Leica Application Suite (Version 4.8.0) for measurements of size. AMPs were assorted in three main visual categories; synthetic fibres, non-synthetic fibres and fragments. Combustion particles were excluded due to the uncertainty of their nature. White/transparent non-synthetic fibres were not quantified due to their ubiquitous presence also in procedural contamination controls, possibly originating from the use of cotton lab coats (data not presented). Before opening the individual petri dishes, a swift visual scan for larger particles (i.e. airborne fibres that can easily contaminate) was performed. Then a few drops of MilliQ water were added, and filters were visually analysed. Due to the high presence of synthetic and non-synthetic fibres in incoming wastewater to the treatment plant in Ny-Ålesund, a subarea corresponding to one fourth of the filter was analysed. With regard to fragments, the whole filter area was analysed.

4.3.2 FTIR analysis

Subsamples of particles visually identified as suspected AMPs were further analysed with Fourier-Transform Infrared (FTIR) spectroscopy for validation of the visual classification along with polymer specific identification. The FTIR technique used in the present study was focal plane array (FPA, 128 x 128) transmission µFTIR (Agilent Technologies, Cary 600 Series FTIR Microscope, Cary 620 FTIR) run with a liquid Nitrogen cooled detector, resolution of 8 cm^{-1} and a scan range of 3800-850 cm^{-1}. 120 background scans were collected before 30 sample scans to adjust for background noise. Suspected AMPs were moved onto a ZnSe disc (Zinc Selenide, Ø 13 mm, thickness: 2 mm) of which an initial photograph was taken in order to correctly set the area for assembly of a mosaic scan with an IR pixel size of 5.5 x 5.5 µm. Obtained spectra were matched (MineIt, KnowItAll Informatics System, vibrational spectroscopy edition) to licensed commercial libraries of polymers (ATR-IR Polymers Bio-Rad Sadtler and IR- Polymers Hummel-BioRad Sadtler). Spectra were also matched to locally produced libraries at the Department of Bioscience, Aarhus University, Denmark, additionally containing both weathered and natural materials. In addition, wool fibres were added into the library in order to minimize the risk of incorrect identification of polyamide, due to their spectral similarities. However, this may have led to the underestimation of polyamide fibres and therefore they are grouped together as wool/polyamide in the present study.

Correlative matching rates to library reference spectra were generated with in-program optimized corrections, including baseline corrections. The results generated by library search were carefully observed to ensure concordant key peaks. An unknown particle category is included in the present study that was visually classified as anthropogenic but did not produce identifiable spectra, and a category called 'synthetic undefined' for clearly synthetic spectra but lacking polymer specific identification. Non-synthetic fibres with an uncertain visual appearance in combination with the FTIR match of cellulose were discarded as natural organic material, but when showing clear visual AMP characteristics retained as cotton fibres. Particles classified as rubber were only subjected to visual analysis due to the limited possibility to receive reliable FTIR spectra of black particles.

4.4 Contamination avoidance

Precaution was taken during all steps to mitigate eventual contamination. All tools, jars and equipment that were used in contact with samples, were rinsed in MilliQ water and kept covered in aluminium foil. White 100 % cotton lab coats were used at all times when samples were handled. A clean air cabinet (Clean Air Techniek B.V.) was used whenever possible and when not, the positioning in laboratories was placed away from ventilation and doors, and surfaces carefully cleaned beforehand. Upon choosing working location, an estimation of background airborne contamination levels in four potential working environments was performed. Three dampened 20 µm filters were air exposed in each environment for two hours followed by visual analysis at 25x magnification. The lowest contamination was 0.7 ± 0.6 fibres per filter and subsequently the position chosen.

Upon filtration, filters were mounted in sequence to minimize air exposure and handling. Filters were thoroughly rinsed in tap water beforehand, and visually inspected under a stereomicroscope where eventual contaminating particles were removed. Each filter was separately stored in a new petri dish (polystyrene) and from there on kept closed at all times until further analysis. Blanks were performed as procedural contamination controls for wastewater (with MilliQ water) and sediment (with density separation solution: saturated NaCl).The contamination controls (n=2 for sediment, n=1 for wastewater) were exposed to the same treatment and analysis as actual samples.

5 Results

5.1 Evaluation of analysis

Due to the economic limitations of the project, only a subsample of the particles visually categorized as AMPs could be further analysed with FTIR to verify polymeric identity. The percentage of FTIR-analysed particles ranged from 0.2 % of the non-synthetic fibres in wastewater to 21.5 % in sediment (Table 2). Subsampling of particles for FTIR analysis was performed to optimize distribution and information, i.e. to include particles from all replicates within a station and matrix, to represent all three visual categories (fibres, non-synthetic fibres and fragments) and to represent both frequently occurring as well as rare particle types. The percentage of particles visually identified as AMPs, which subsequently were verified as such by FTIR was very high, varying between 69 % for wastewater fragments to 100 % for wastewater non synthetic-fibres (Table 2). This indicates high accuracy in the visual examination procedure.

Table 2. Percentage of visually identified particles analysed with FTIR and subsequently identified as anthropogenic microparticles (AMPs). NS-fibres: Non synthetic-fibres.

Matrix	% particles analysed with FTIR	% identified as AMPs
Sediment	21.5	77.0
Seawater	17.0	93.0
Wastewater (fragments)	17.8	69.0
Wastewater (fibres)	4.3	92.0
Wastewater (NS-fibres)	0.2	100.0

5.2 Wastewater

The lowest limit of particle detection in wastewater was 20 µm as restricted by the mesh size of the filter. The concentrations of AMPs dropped dramatically as the wastewater passed through the different steps of the treatment process. Incoming wastewater contained 14 207 AMPs L^{-1} (n=1) while the outgoing wastewater contained 83 AMPs L^{-1} (n=1), representing a 99.4 % decrease of AMPs in outgoing compared to incoming wastewater (Fig. 7). The contamination control sample (Milli-Q water) contained only one AMP L^{-1} (n=1) represented by a black fibre (Fig. 7 & 8). The majority (98 %) of the AMPs in the incoming wastewater consisted of fibres, mostly non-synthetic and identified by FTIR as cotton and wool (Fig. 8, Table 3). In outgoing wastewater 93% of the particles were identified as fibres, thus containing a higher fraction of fragments than incoming wastewater.

The FTIR analysis revealed a higher diversity of polymers in the outgoing wastewater with non-synthetic fibres of wool and cellulose, and fibres and fragments of low-density polyethylene (LDPE), polyethylene terephthalate (PET), polyurethane (PU) and polyester (Table 3). Although varying immensely in total AMP counts, the incoming and outgoing wastewater had very similar AMP colour distributions dominated by black and blue particles (Fig. 8).

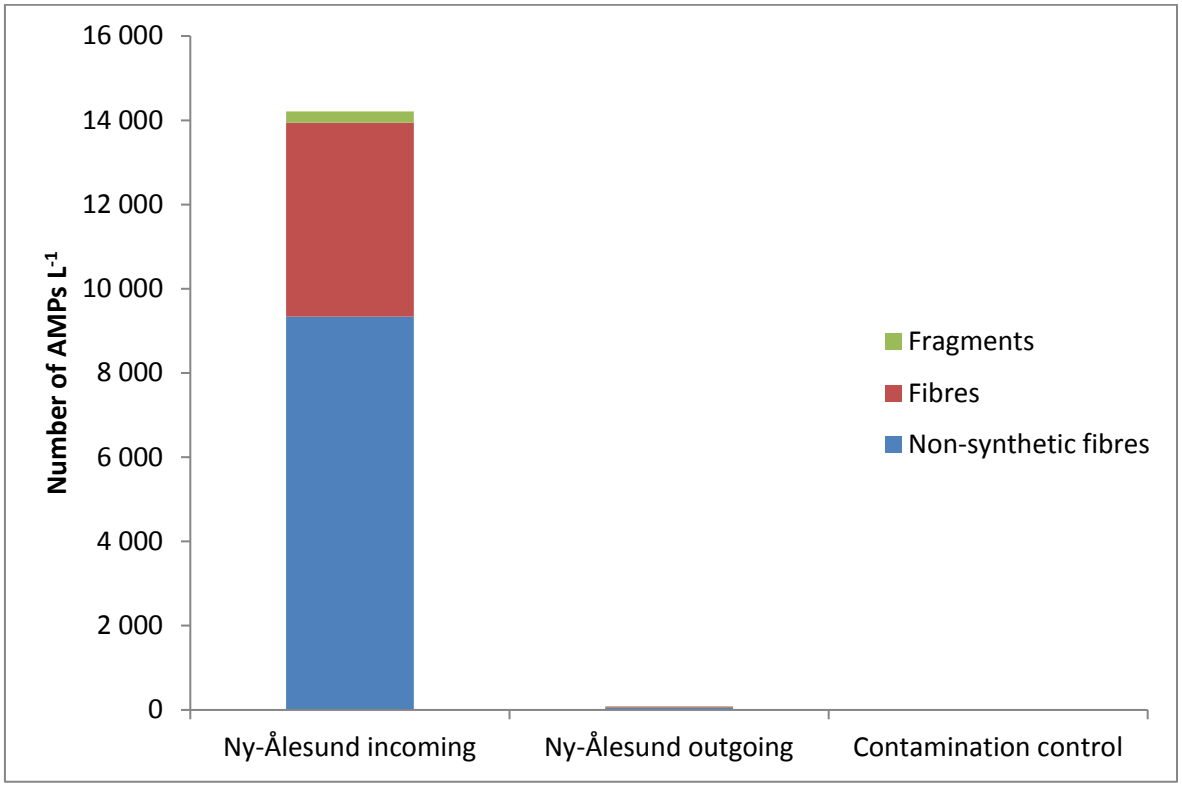

Figure 7. Numbers and types of identified AMPs L^{-1} >20 µm in each sample (n=1).

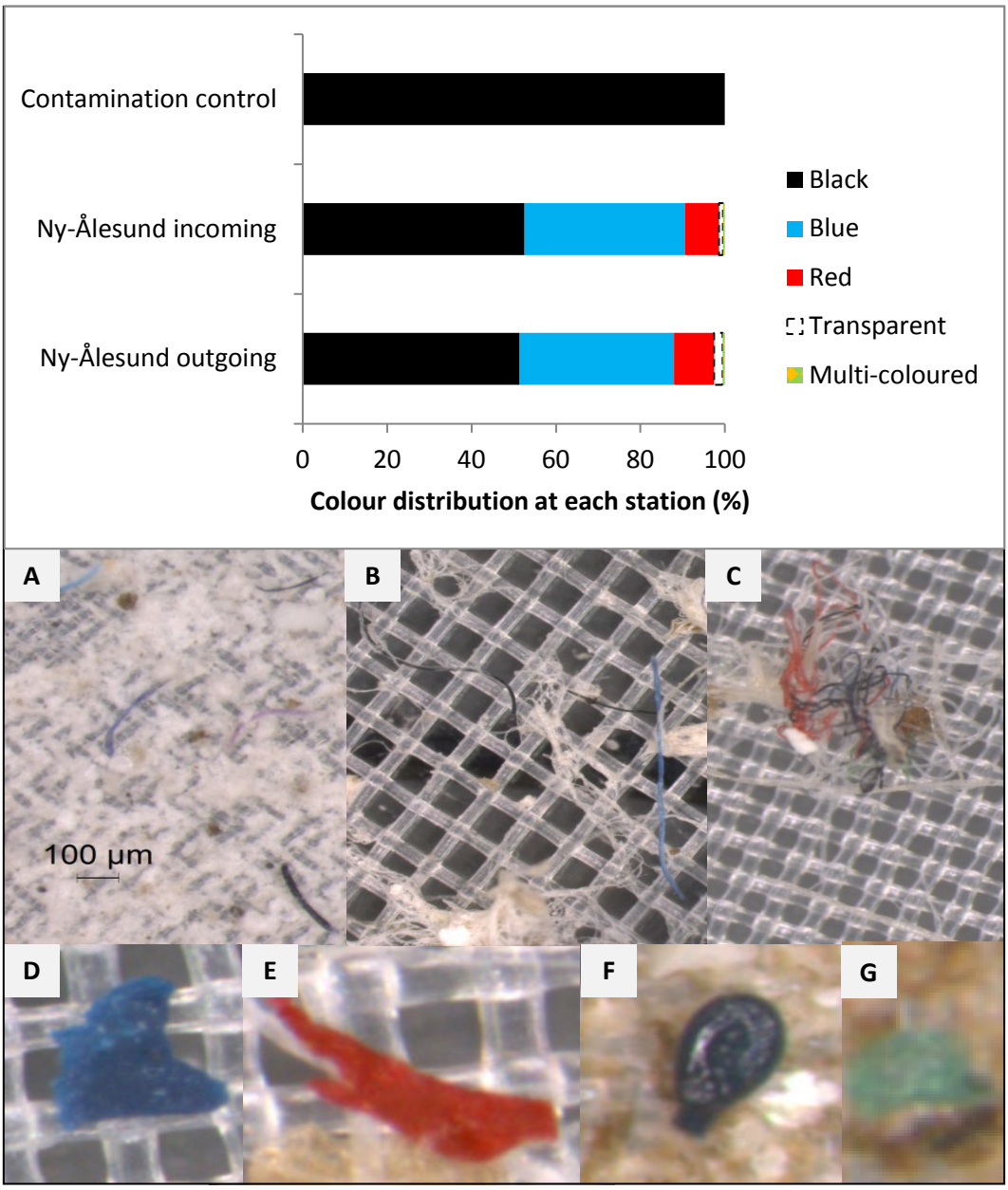

Figure 8. Colour distributions (top) and photographs (bottom) of AMPs >20 µm identified in incoming and outgoing wastewater including contamination control. Photographed particles; A: fibers (non-synthetic and synthetic) in incoming wastewater, B: fibers (non-synthetic and synthetic) in incoming wastewater, on a 300 µm filter, C: fibers (non-synthetic and synthetic) in outgoing wastewater, on a 100 µm filter, D & E: fragments in outgoing wastewater, on a 100 µm filter, F: fragment in outgoing water, ~80 µm, G: fragment in outgoing water, ~50 µm.

5.3 Seawater

The lowest limit of particle detection in seawater was 50 µm as restricted by the mesh size of the filter. The highest concentration of AMPs was found outside the wastewater outlet (WWO) in Ny-Ålesund (61.2 AMPs m^{-3}, 0.06 AMPs L^{-1}) followed by near the wastewater outlet in Barentsburg (51.8 AMPs m^{-3}, 0.05 AMPs L^{-1}) and the reference site at Signehamna (18.6 AMPs m^{-3}, 0.02 AMPs L^{-1}) (Fig. 9). The largest fraction of particles at Signehamna and near the Ny-Ålesund WWO consisted of non-synthetic fibres and the only materials identified at the reference site Signehamna were cotton and wool (Fig. 9, Table 3). Seawater near Barentsburg WWO contained relatively more fragments than seawater from the other sampled sites (Fig. 9). The highest diversity of polymers and particle colours were found near the Ny-Ålesund WWO, where identified AMPs, aside from non-synthetic materials, were paint, PET, polypropylene (PP) and other undefined synthetic materials (Fig. 10, Table 3). A lower diversity of materials was identified outside Barentsburg; PP, polychloroprene (PCP) and cotton (Table 3).

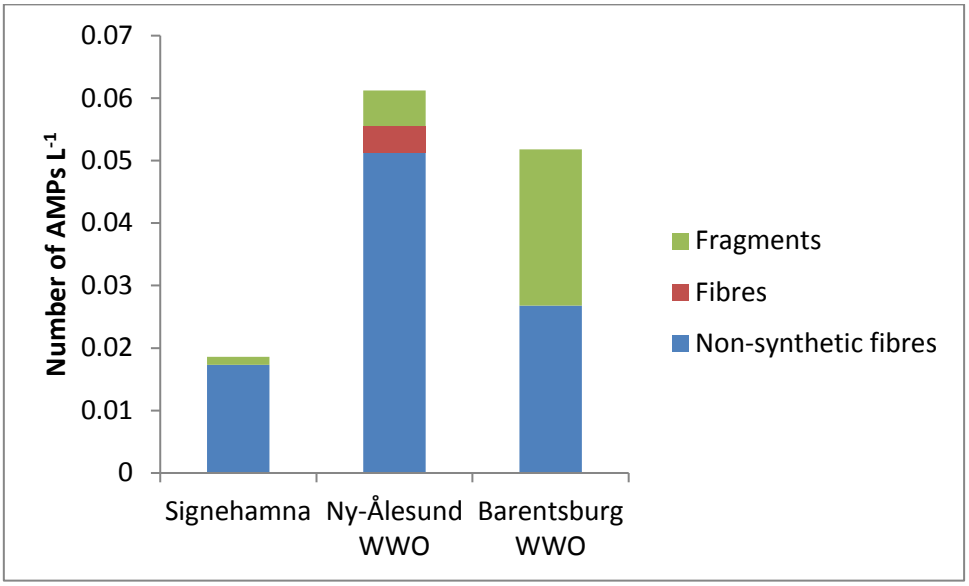

Figure 9. Numbers and types of AMPs L^{-1} >50 µm of *in situ* pumped seawater from different sites (n=1).

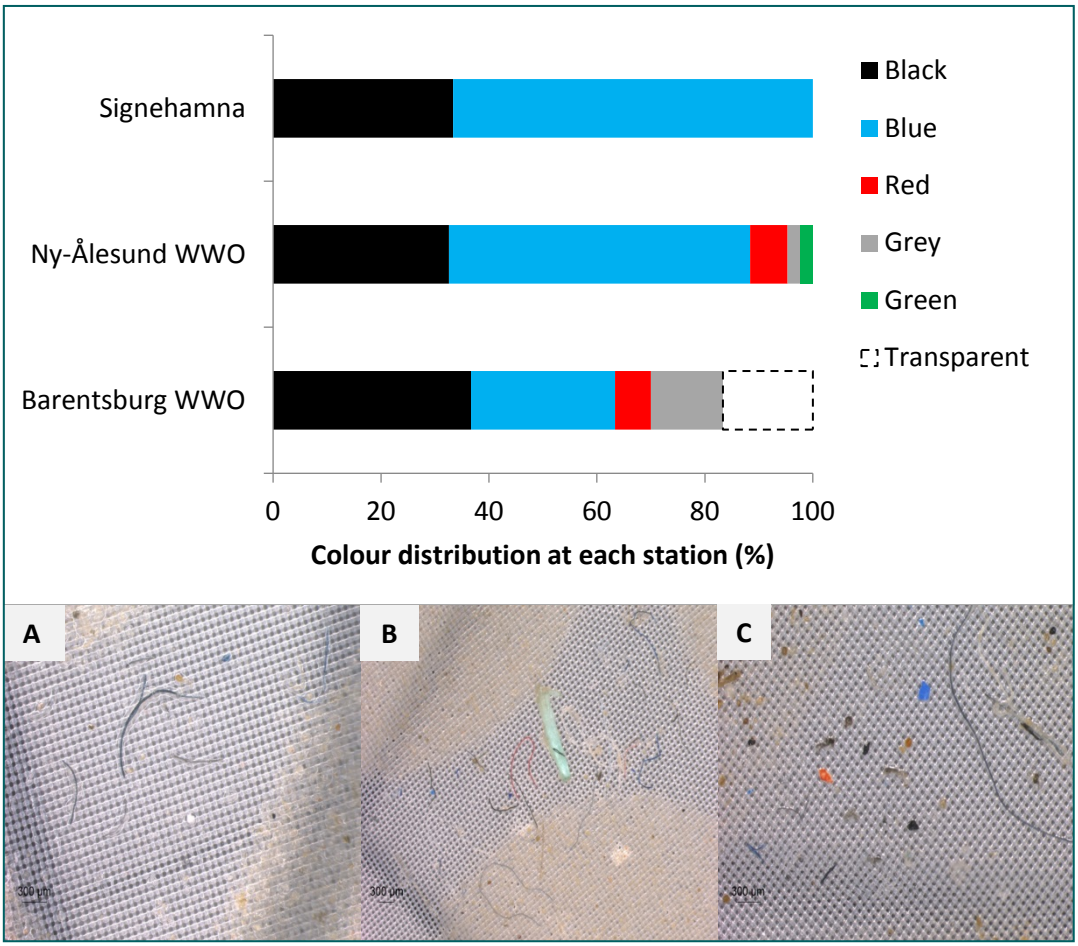

Figure 10. Colour distribution (top) and photographs (bottom) of AMPs >50 µm identified in seawater from A) Signehamna, B) Ny-Ålesund WWO and C) Barentsburg WWO.

5.4 Sediment and beach sediment

The lowest limit of particle detection in seawater was 20 µm as restricted by the mesh size of the filter. The highest (134 AMPs kg^{-1} DW) and the lowest (0 AMPs kg^{-1} DW) AMP concentrations were found in sediment collected at the same site outside the WWO in Ny-Ålesund (Fig. 11). Concentrations in sediments from the other sites were similar and low, varying between 7 and 36 AMPs kg^{-1} DW for individual samples (Fig 11).

Fragments commonly dominated AMPs in the sediment samples and a high diversity of colours was found in sediments near the Barentsburg WWO (Fig. 11 & 12). Non-synthetic AMPs were found in sediments at all sites (Table 3). At Ny-Ålesund WWO, AMPs of polylactic acid (PLA) and PP were identified. At the Barentsburg WWO, PP was identified while the only identified polymer from Barentsburg beach sample was cellophane. At Signehamna AMPs of ester gum and PET were found along with wool and cellulose/

cotton. In the beach sample collected near Barentsburg WWO, the composition of AMPs was found to be almost identical to that in sediment from the same location, but concentrations were one third (11 AMPs kg^{-1} DW) of those found in marine sediments.

Contamination control samples were found to contain low amounts of AMPs, dominated by fragments comprised by rubber and styrene copolymer resin. The two synthetic polymers found in the control samples were never identified in the field samples and could be expected to originate from seals of the density separation device itself.

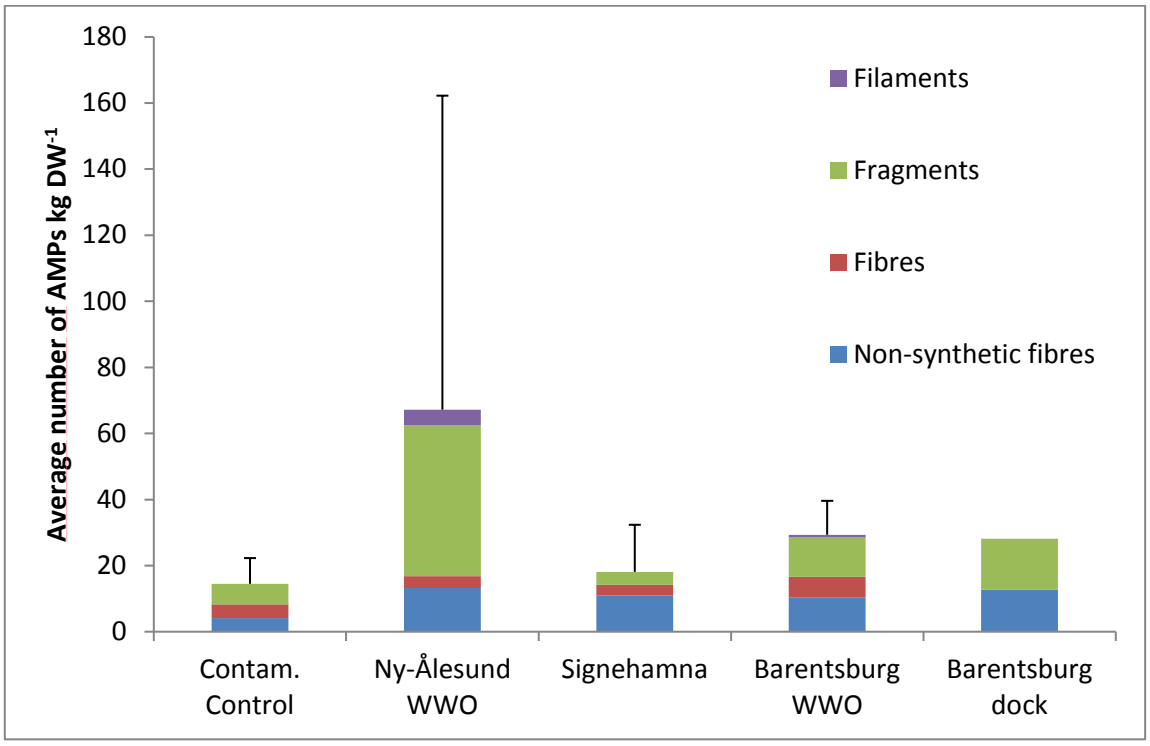

Figure 11. Average numbers (+SD) and types of AMPs kg^{-1} DW >20 µm in marine sediments from different sites.

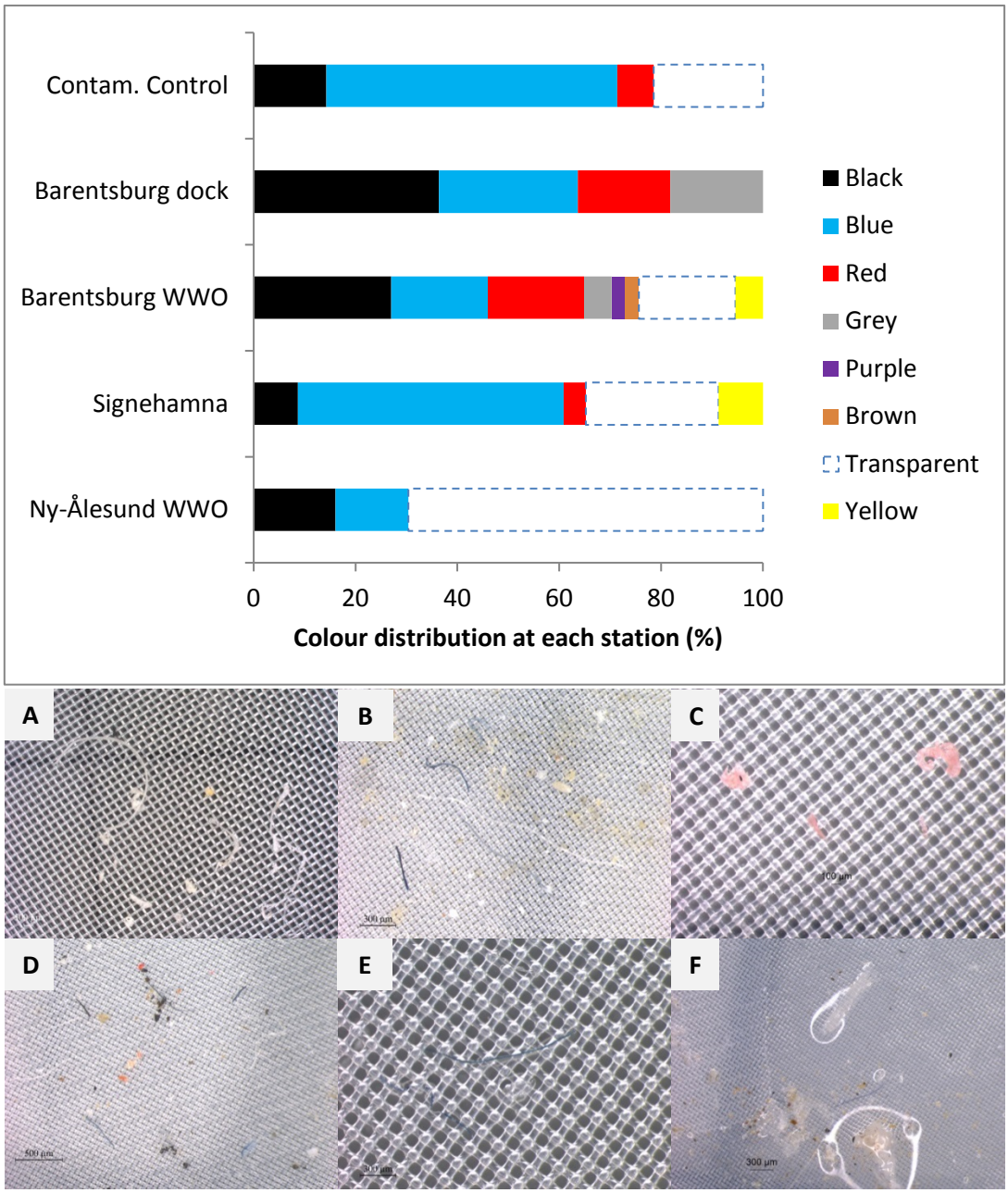

Figure 12. Colour distribution (top) and photographs (bottom) of AMPs >20 µm identified in marine sediments (and beach sediment for photos) from sampled sites. Photographs show AMPs identified A) in sediment at Ny-Ålesund WWO, B) in sediment at Signehamna, C) in sediment at Barentsburg WWO, D) in sediment at Barentsburg dock, E) in the procedural contamination control (blank) sample, F) in beach sediment from near Barentsburg WWO .

5.5 Polymeric composition

Due to the limited subsample size analysed with FTIR, it is important to note that the FTIR results cannot be interpreted quantitatively. There may be other polymers present in the samples from all sites, which were not selected for analysis. A wide variety of polymers were identified in the different matrices (Table 3, section 5.2, 5.3 & 5.4). It is indicated that higher polymer richness (number of different polymers) is detected close to the wastewater outlets (Table 3).

Table 3. Results of FTIR analyses based on a selection of visually identified AMPs. Dark orange refers to sampling sites close to potential point sources (WWO: wastewater outlet), light orange refers to the reference site of Signehamna and grey refers to the procedural contamination controls (blank samples).

	Sediment				Beach	Seawater			Wastewater	
	Blank	Ny-Å WWO	Signe-hamna	Barents WWO	Barents WWO	Ny-Å WWO	Signe-hamna	Barents WWO	Ny-Å IN	Ny-Å OUT
Acrylic										
Cellophane										
Cellulose/cotton[a]										
Ester gum										
LDPE										
Paint										
PCP										
PET										
PLA										
Polyester										
PP										
PU										
Rubber										
Styrene copolymer resin										
Synthetic undefined[b]										
Unknown[c]										
Wool/Polyamide[d]										
Polymer richness	3	4	4	4	1	6	2	3	4	6

[a]A match to cellulose/cotton was only accepted for AMPs exhibiting clear visual anthropogenic properties, i.e. unnatural colour.

[b]Synthetic undefined were a group of particles where no polymer specific identification confidently could be made based on the FTIR spectral libraries, but a synthetic origin was clear.

[c]The category unknown were particles that could neither confidently be rejected as natural material nor accepted as synthetic.

[d]Wool and polyamide were grouped together due to their similarity in spectral appearance.

6 Discussion

Anthropogenic microparticles (AMPs) were found at all sites and in all matrices investigated in this study. Samples of wastewater and seawater were dominated by fibres (both non-synthetic and synthetic) whereas sediment samples were dominated by, or had equal amounts of, fragments compared to fibres. AMPs released with wastewater will have different intrinsic properties such as density and shape, which consequently affect their environmental journeys, e.g. whether they will float or sink to the bottom (Bagaev et al. 2017). Such particle characteristics will influence the fate of AMPs in the marine environment already at the point of release, acting in combination with abiotic factors such as temperature, salinity and currents (Critchell and Lambrechts 2016) as well as biotic factors such as biofouling (Lobelle and Cunliffe 2011). The pattern observed in this study with relatively more fragments in sediments and more fibres in the water column may be a result of such acting forcers. No pattern related to particle density was observed in this study. This suggests that it is a complex combination of factors influencing and determining the distribution of AMPs in the marine environment.

AMPs released with wastewater may be rapidly transported with prevailing currents and thus, accumulation zones distant from the WWO discharge point could exist. It has been suggested that a transport of AMPs away from the immediate vicinity of the wastewater outlet of Longyearbyen is taking place, with AMPs mainly staying buoyant rather than becoming deposited in the sediment (Sundet et al. 2016). This complicates the detection and quantification of actual impact from the point source in the recipient, and further stresses the importance of studying and characterizing local pollution sources and their spatial and temporal dynamics.

Higher concentrations of AMPs were observed closer to human activities both in sediments close to Ny-Ålesund WWO and in seawater close to both Barentsburg and Ny-Ålesund as compared to the remote site at Signehamna. Increasing concentrations of microplastics closer to a point source have previously been found in a similar remote polar settlement of Antarctica (Reed et al. 2018). The large variation between replicates in sediment outside Ny-Ålesund WWO indicates that there are elevated AMP concentrations within this area, but with great variation on small spatial scales. This scenario would require a larger sample size (i.e. number of replicates and sampling sites) to understand the acting forcers and further enable statistical analyses. There was also a higher diversity of AMPs, both in terms of colour (seawater and sediment) and polymer types (seawater) detected in the recipients close to the WWOs as compared to the remote site of Signehamna. This was particularly clear in seawater samples from Ny-Ålesund WWO where five different colours and six different materials were detected, while only two different colours and two materials of non-synthetic origin where recorded in Signehamna. This pattern shows traces of local contamination, and also gives an indication of the level of background concentration potentially from global or regional diffuse sources in this area.

Based on our investigation, as much as 99 % of the incoming AMPs may be retained through the different processing steps of the wastewater treatment plant in Ny-Ålesund and will thus be prevented from being released into Kongsfjorden. Similar measurements have been obtained from wastewater treatment plants in temperate and sub-Arctic areas (e.g. Magnusson et al. 2016b, Murphy et al. 2016). Albeit high retention efficiencies, the concentration of AMPs in the effluents from wastewater treatment plants can be of significant nature just as seen in the present study (83 AMPs L^{-1}). Since the wastewater treatment plant of Ny-Ålesund is not designed for retaining AMPs in incoming wastewater, the observed AMP reduction can rather be seen as a positive side effect. Additional positive side effects of retaining AMPs is that plastic additives and other contaminants which may be associated with the particles (Hermabessiere et al. 2017) are also prevented from entering the marine environment. Thus, through addressing one pollution problem others can simultaneously be resolved. It is important to ensure appropriate disposal and/or destruction of the remaining sludge, to avoid the spread of contaminants in the environment.

Although the wastewater treatment plant is in place in Ny-Ålesund, we do, however, still detect elevated concentrations of AMPs in the recipient both in seawater and in sediment. Aside from the current continuous input of AMPs from outgoing wastewater to Kongsfjorden, wastewater has been released untreated up until 2015. Accumulation zones are most probably dynamic and AMPs stored in sediments could be resuspended to the water column by e.g. currents and sediment dwelling organisms (Galloway et al. 2017). Additionally, AMPs could likely originate from other sources in Ny-Ålesund as well. It could originate from deposition of waste on melting snow and ice, and potentially from snow and glacial meltwater, concentrating AMPs from atmospheric deposition (Dris et al. 2016). An assessment of potential local sources of macro- and microlitter is recommended and would help managing plastic and associated contaminant pollution in Ny-Ålesund and Kongsfjorden, as well as other Arctic communities.

7 Conclusion and recommendation

This investigation shows that local pollution sources of anthropogenic microlitter are significant from the small communities on Svalbard. Further, it shows that installation of appropriate wastewater treatment systems substantially can reduce the release of anthropogenic microlitter to the marine environment. However, it is important to note that our results are based on only one sample of incoming and outgoing wastewater, collected at one occasion during summer. A higher temporal and spatial resolution along with proper replication is needed in order to statistically evaluate and accurately determine the true efficiency of the wastewater treatment plant.

The results do, however, provide a strong indication of the importance even of small-scale wastewater treatment plants for reducing microlitter pollution and potentially associated contaminants from local sources to the marine environment. It should also be noted that although the concentration of microlitter in outgoing wastewater was substantially lower (99.4 %), there is a continuous release of microlitter to the recipient Kongsfjorden.

Therefore, efforts of reducing microlitter in wastewater should additionally be made upstream in e.g. households. This report should be considered as a step towards resolving the issue of local contamination in the Arctic as a whole, with increasing importance in pace with the rapid expansion of tourism and industrial development observed in this region of the world.

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