Microplastics in the realm of Svalbard: current knowledge and future perspectives (MIRES)

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1. Introduction

Plastic and our society have become inseparable. Almost all aspects of daily life involve plastics. Plastics are found, for example, in electronics, home appliances, vehicles, food/goods packaging material, cosmetics, and a range of textiles. Plastic industries have assessed that in 2018 about ≈ 359 million tons of plastics were manufactured, globally out of which 62 million tons were produced in Europe (Plastics Europe 2019). Resultantly, plastic pollution has become a critical point of concern, on account of the rapid production and disposal rate of plastic combined with poor waste handling and the slow degradation rate of the material. Merely a few decades into the rapid rise of plastic mass production, we have generated a huge volume of plastic debris in the environment, potentially leaving classifiable fossil records for future generations (Zalasiewicz et al. 2015). Furthermore, plastics are mostly produced from carbon-based raw materials, mainly natural gas and crude oil (Plastics Europe 2019). Different reactions of polymerization contribute to regulating the properties of plastics such as its hardness or softness, opacity or transparency, flexibility or stiffness. Chemical additives such as plasticisers, flame-retardants, and dyes are added to plastics to modulate their flexibility, durability, or other characteristics. First used in environmental sciences two decades ago (Thompson et al. 2004), the term "microplastics" (MPs) encompasses plastic fragments 1 µm to 5 mm in size. Sources of MPs overlap with those of meso- (2.5 cm – 0.5 cm) and macro-plastic (1 m – 2.5 cm) in many instances. Based on their origin, MPs can be distinguished into the following two types: primary and secondary. Primary MPs are small-sized and commonly used as exfoliants/scrubbers in cosmetics and industrial abrasives (Cole et al. 2011; Leslie 2014), plastic beads that serve as drug vectors in medicines (Patel et al. 2009), or precursors in the production of other plastic products. Conversely, secondary MPs originate from the fragmentation of meso or macro or even larger plastics under the influence of light, mechanical abrasion, and/or temperature fluctuations. Further fragmentation from MPs to smaller sized (below 1 µm) particles are known as nano plastics (MSFD Technical Group on Marine Litter 2013). The focus of this review is only on the microplastic (MPs) category.

Not only is plastic production and littering on the rise globally but the high buoyancy of many plastics leads to long-distance transport in the ocean. Plastic is widely distributed in almost all zones from the polar region to tropics and habitats from the sea surface and pelagic zone to the benthos and deep sea. In the Arctic, remote from large populations, plastic debris dominates observations of marine litter (Grøsvik et al. 2018) and has been suggested to have negative effects on the Arctic biota and threatening the ecosystem (Halsband and Herzke 2019; Tanaka et al. 2020). The most common fatal interaction of Arctic organisms with plastic litter is entanglement and ingestion (Derraik 2002). Reported observations of entanglement and ingestion of mega/macroplastics by terrestrial and marine organisms are numerous. Entanglement in abandoned fishing gear and other marine litter on the beaches of Svalbard, Norway, by reindeer and seabirds is common (Figure 1) (Hallanger and Gabrielsen 2018; Nashoug 2017; Øritsland 1986). Moreover, ingestion of plastics has been found in organisms at lower and higher levels of the Arctic food web including benthic organisms like starfish, shrimp, and crabs (Fang et al. 2018), fish (Kühn et al. 2018; Morgana et al. 2018), whales (Finley 2001), and seabirds (Poon et al. 2017; Provencher et al. 2017; van Franeker et al. 2011; Trevail et al. 2015). Seabirds, in particular, are vulnerable to marine plastics, mainly on account of their high trophic position and their extensive foraging range. For these reasons, and because their colonies are relatively easily accessed by researchers once a year, seabirds are considered ideal monitoring sentinels for marine plastic pollution in the environment (Herzke et al. 2016; Avery-Gomm et al. 2012; van Franeker et al. 2011).

Figure 1: Interaction of local organisms with plastic litter. Photos: Jon Aars (Reindeer), Susanne Kühn (Fulmar), Governor of Svalbard (Arctic tern, polar bear)

1.1. Adverse effects of MPs

While the adverse effects of larger plastic litter are obvious as the observations mentioned above demonstrate, our knowledge of the adverse effects of MPs is very limited. Biological effects of micro-and nano-plastics have only been observed in laboratory studies. From the available data, the adverse effects of MPs can be divided into mechanical effects, chemical toxicity, and pathogenic microbial toxicity.

Mechanical effects: Unlike macroplastics, the mechanical effects of MPs are hard to assess. Hard and sharp-edged MPs can potentially injure or scratch the soft tissues of the lungs, liver, and gastrointestinal tract (GI) (Hwang 2019; Fry 1987), which may lead to infection and sometimes even death (Duis and Coors 2016). MPs are widely reported to be excreted after ingestion (De Sales-Ribeiro et al. 2020) but they have also been reported to cause obstruction or create a false satiated feeling by residing in the GI tract, leading to death due to starvation in organisms. MPs have been reported to affect the GI microbiome in organisms (Cox et al. 2019; Wright et al. 2017; Fackelmann et al. 2019) and rare observations report that MPs with a diameter of ≤250 µm may translocate into tissues in fish (Gomiero et al. 2020) and also in mammals exposure studies (Volkheimer 1975).

Chemical toxicity: The chemical toxicity of MPs can be mediated by plastic additives or contaminants adsorbed to the plastic particles as they travel through the environment. Additives are chemicals added at the time of production, which can be fillers, plasticizers, flame retardants, colorants, stabilizers, lubricants, foaming agents, and antistatic agents (Groh et al. 2019). Chemicals that sorb onto the plastics particles include persistent organic pollutants (POPs), heavy metals, and pharmaceuticals (Magara et al. 2019; Zhang et al. 2019; Sikdokur et al. 2020; Tang et al. 2020). These chemicals can potentially alter physiological functions after being ingested with the plastic particles (Cole et al. 2011; Watts et al. 2014; Wang et al. 2018). On the other hand, some studies show no correlation between the body burden of organic pollutants and the gut content of MPs in seabirds (Herzke et al. 2016). Whether chemicals associated with or sorbed to plastics play a greater role than other sources of toxic chemicals is yet to be investigated. MPs may not be a net to the most important vector for environmental pollutants other than for those primarily associated with plastics as additives.

Pathogenic microbial toxicity: Plastics, including MPs, are a suitable substrate and vector for pathogenic microbes in the environment (Oberbeckmann et al. 2015; Koelmans et al. 2016; Hartmann et al. 2017). By colonizing and growing on the surface of MPs, microbes form dense biofilms, the so-called "plastisphere" (Kirstein et al. 2019; Zettler et al. 2013). Organisms can get infected by ingesting MPs colonized by pathogenic microbes (Bhattacharya and Khare 2020). Pathogenic antibiotic-resistant bacteria have been found on MPs collected in the intertidal zone in western Norway (Radisic et al. 2020).

2. Overview of existing knowledge

Different studies of MPs pollution in Svalbard have used different methods and units of measurement (Appendix 1), making it difficult to discern trends and draw comparisons. Furthermore, many studies do not satisfactorily assess measurement uncertainties. Harmonised analytical protocols and data reporting, as well as method inter-calibration with proficiency tests, are essential to enable us to compare datasets and observe trends. Advice on plastic monitoring in the Arctic regarding sampling and measurement methods is currently being developed through the Arctic Monitoring and Assessment Programme (AMAP). To our knowledge, two ring test regimes are ongoing on a European basis, organised by Quasimeme and the European Commission Joint Research Centre (JRC)/German Federal Institute for Materials Research and Testing (BAM), respectively.

Reports of MP in sea ice, snow, seawater, beach sand, deep-sea and shallow sediments, invertebrates, fish, and seabirds allow to coinciding that MPs have become ubiquitous in the Svalbard ecosystem (Appendix 1). Knowledge of the ecological and social impacts of this is vital for making informed decisions and policies regarding MPs pollution. Therefore, by exploring the following points, we aim to synthesise the existing knowledge of MPs pollution in and around Svalbard and to identify new insights from current research and gaps and challenges to be addressed by future research.

2.1. Known and potential sources of MPs in Svalbard

MPs have numerous points of entry into the environment, complicating it to pinpoint their particular source. However, based on current knowledge, the sources of MPs in Svalbard can be divided into local and long-range distances (Figure 2). Local sources include industrial activities (e.g. fishing and shipping), tourism, domestic activities such as the washing of synthetic textile clothing, personal care products such as cosmetics containing MPs (e.g. toothpaste, exfoliators, etc.), dumpsites and landfills, sewage, vehicle tyres, and snowmobile belt. Long-distance sources are mostly in similar categories as local sources but on a larger scale. From long-distance sources, MPs travel to Svalbard via atmospheric and ocean currents (Obbard et al. 2018). Before the discovery of atmospheric transportation of MPs, ocean currents were considered as the main pathway for MPs to remote locations such as the polar regions (Cózar et al. 2017). Lately, the detection of MPs in snow, ice and air samples (Bergmann et al. 2019) has highlighted the role of air currents as an important pathway for MPs as well as challenged our current understanding of their global transport. A recent modelling study found the high transport efficiencies of MPs particles produced by road traffic (tyre wear particles [TWP] and brake wear particles [BWP]) to remote regions via the air (Evangeliou et al. 2020).

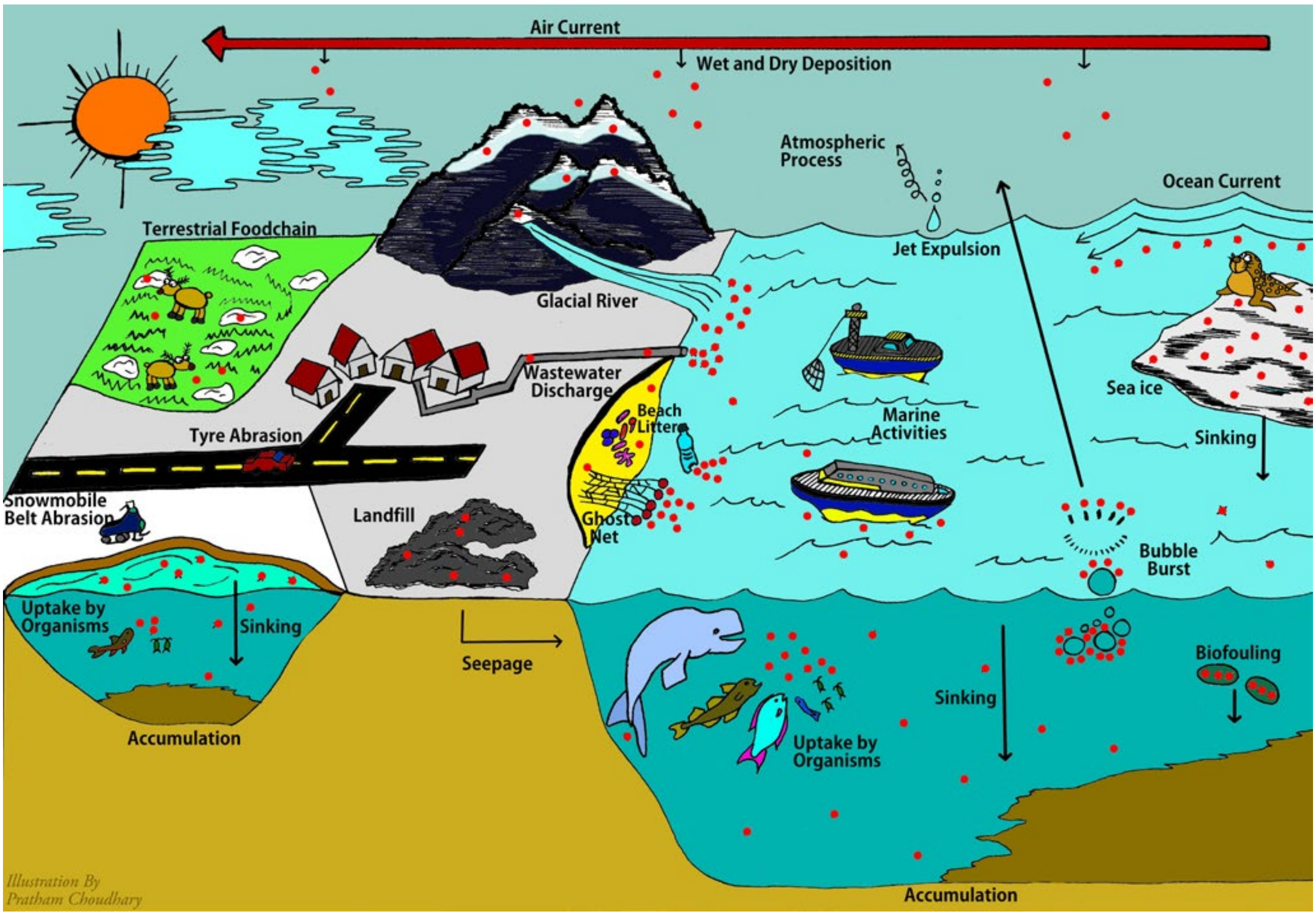

Figure 2: Potential sources and pathways of MPs in Svalbard

2.2. Pathways and movement of MPs between environmental compartments

Glaciers: Glaciers are likely deposition areas for debris and pollutants transported by air, as glacier ice forms through the transformation of accumulated snowfalls, which are particularly efficient at scavenging dust, soot, contaminants, and MPs from the atmosphere (Bergmann et al. 2019). Cryoconite – the dust on a glacier – absorbs solar radiation and forms vertical cylindrical holes. These holes are likely to retain and accumulate atmospherically transported contaminants such as heavy metals (Nagatsuka et al. 2010; Łokas et al. 2016; Baccolo et al. 2017; Singh et al. 2017; Huang et al. 2019), organic compounds (Ferrario et al. 2017; Weiland-Bräuer et al. 2017) and MPs (Ambrosini et al. 2019) from local and distant sources. Seasonal and climate change-induced temperature rise may result in the release of cryoconite/glacial debris-bound MPs to watersheds and the ocean.

Wetlands: Wetlands consist of shallow and perennial lakes and ponds. These environments play an important role in ecosystem structure by sustaining most of the Arctic microbial biodiversity and providing natural refuges and feeding places for wildlife (Walseng et al. 2018). Wetlands are potentially susceptible to the deposition of atmospherically transported MPs from local and distant sources and seabirds (Luoto et al. 2019) to some extent.

Marine environment: The Svalbard ecosystem is mainly influenced by the West Spitsbergen Current, which carries water from the Atlantic northwards and along the western coast of Svalbard (Svendsen et al. 2002), and brings MPs from North Atlantic fisheries to Svalbard (Bergmann et al. 2017). The eastern coasts of Svalbard receive Arctic water from the northeast (Misund et al. 2016), which could be expected to be more influenced by MPs from sea maritime industries and activities (shipping, fishing, cruise ships, and scientific expeditions) from lower latitudes. In addition, MPs may also originate from local sources such as sewage/wastewater discharge, fishing and tourist activities, and dumpsites and landfills among other local sources. After reaching Svalbard, MPs with a higher density than seawater would sink and accumulate in the sediments (Woodall et al. 2014; Alomar et al. 2016), whereas MPs with a lower density would float on surface water (Suaria and Aliani 2014). The density of floating MPs can be altered through 'biofouling', a process in which biofilm that forms on the surface of floating MPs increases the density of MPs, thereby increasing sinking (Caruso 2020). Floating MPs may be distributed by wind and waves to the fjords and/or washed onto the shores. MPs input into fjords can also be expected from glaciers as a result of iceberg calving, surface melting, runoff, or melting under floating ice shelves. Corroborating this hypothesis, the highest concentrations of MPs measured in Arctic sediments were found close to glaciers (Huntington et al. 2020) and sea-ice fronts (Bergmann et al. 2017; von Friesen et al. 2020).

Sea ice plays a triple role for MPs: a temporary sink, a secondary source, and a transport medium (Obbard et al. 2014; Peeken et al. 2018; Kanhai et al. 2020). Forming sea-ice traps MPs from the surrounding water and sediments. Additionally, atmospheric MPs deposits on sea-ice floes. When sea ice drifts, it transports MPs and release trapped MPs by melting caused by seasonal or climate-induced warming. Regardless of transport, sea ice will release previously trapped MPs into the ocean again, due to seasonal and/or climate change-induced melting.

Generally, the ocean acts as a sink and conveyance for MPs from one to the other location; recently, however, an additional role of the ocean as an indirect source of MPs has been highlighted (Allen et al. 2020). According to Allen et al. MPs may enter the atmosphere from the ocean in a similar way as sea salt aerosol and organic matter do under the influence of wave action, adding a new aspect to the environmental plastic cycle.

Terrestrial environments: MPs in the terrestrial environment may enter via atmospheric currents from long-distance/local sources and be deposited on land, vegetation, water bodies, etc. Earlier it was believed that mainly marine organisms are at risk of MPs contamination. However, with the discovery of the pervasiveness of MPs via atmospheric currents, it has become clear that MPs may also enter the terrestrial food chain (Bergmann et al. 2019). This has raised questions about not only the security of terrestrial wildlife but also for people living in Svalbard because terrestrial wildlife (e.g. reindeer) is a culturally and nutritionally important traditional food.

Seabirds can also be a source of MPs to the terrestrial environment since MPs can be contained in their regurgitates and faeces. Plastic ingested by seabirds is thought to be retained in the gizzard and mechanically broken down over time until the small pieces are able to pass through the intestines and be excreted (Reynolds and Ryan, 2018).

2.3. Status of MPs in different environmental compartments

Atmosphere: Atmospheric long-distance transport of MPs to the Arctic was not reported until last year when snow samples from Fram Strait and Svalbard were shown to contain MPs (Bergmann et al. 2019). Although the significance of atmospheric transport has been highlighted in a few studies from Svalbard and other remote locations, no data are available on the route and transportation pattern of MPs, except the recently published atmospheric model-based study (Evangeliou et al. 2020), which suggests that the Arctic may be a receptor region for atmospherically transported MPs.

Ice and snow: There are only very few published reports of MPs in snow and sea ice from Svalbard MPs (Appendix 1). MPs in ice cores from the Fram Strait and north of Svalbard have been observed as a result of local sources (Peeken et al. 2018) and via air currents, transported over long distances to the Arctic (Bergmann et al. 2019). So far, no publication quantifies the deposition of MPs on glaciers in Svalbard. However, there is evidence of MPs in glacial debris from the Forni Glacier (Italian Alps) (Ambrosini et al. 2019), which could be considered instrumental to conduct similar research by the unexplored glaciers of Svalbard. In light of this, we can hypothesise that similar processes may occur for MPs transportation and deposition on glacial debris/cryoconite in Svalbard. MPs records in glacial ice cores could provide indications for temporal variations, in a similar way to lake sediments that act as a lacustrine archive of MPs (Turner et al. 2019).

Open ocean: MPs in Arctic seawater are relatively well studied in comparison to the other compartments (Appendix 1). MPs have been reported in surface waters and sub-surface waters near Svalbard, potentially as a result of the breakdown of larger items (transported over a long distance or originating from local vessels) or derived from sewage and wastewater from coastal areas (Lusher et al. 2015). MPs were detected through the entire water column of the Arctic Ocean (Tekmann et al. 2020), suggesting that the Arctic is an accumulation area for MPs coming from (i) the North Atlantic via the thermohaline circulation, (ii) north of the Fram Strait entrained in sea ice and released during melting, (iii) the Barents Sea, (iv) ships in the vicinity, (v) different directions through the atmosphere and precipitations, and/or (vi) rivers discharge.

Coastal and fjord waters: MPs have been detected in seawater, sea-ice samples from different fjords in Svalbard as a result of summer sea-ice melting (von Friesen et al. 2020), and wastewater outlets near settlements (Sundet et al. 2016; von Friesen et al. 2020); they have also been seen to come from local activities as well as distant ones (Purver 2019) via Atlantic ocean currents (Scott 2019).

Freshwater bodies: Despite their importance for the Svalbard ecosystem, freshwater bodies are not well studied for MPs contamination. To date, two studies of MPs in freshwater from Svalbard are available. A study of MPs in sediment from Lake Revvatnet suggested that the increased little auk (*Alle alle*) population acts as a source for MPs (Luoto et al. 2019). However, in another study, MPs in sediment from Lake Knudsenheia (near the western shore of Kongsfjorden) was discussed as a result of atmospheric deposition from local or distant sources (Gonzalez-Pleiter et al. 2020).

Marine sediment and beaches: Detection of MPs in the deep-sea sediment of the Arctic Basin and from the HAUSGARTEN deep-sea observatory in eastern Fram Strait suggested the Arctic as an accumulation zone for MPs particles transported from long distance and/or local sources (Woodall et al. 2014; Bergmann et al. 2017; Tekman et al. 2020). MPs in sediment from the Barents Sea reported at a depth between 650 and 508 meters were discussed as a result of the accumulation of debris (Møskeland et al. 2018). MPs have been detected in shallow sediment from Svalbard coastal areas, in the Kongsfjord-Krossfjord system, Rijpfjorden, Grønnfjorden, Adventfjorden, and Breibogen and are discussed as a result of local and/or long-distance MPs pollution (Sundet et al. 2016; Granberg et al. 2019; von Friesen 2018; Granberg et al. In press).

Terrestrial environment: Even though plastic pollution mainly originates from terrestrial sources, terrestrial systems have not received much scientific attention. To date, there is no publication describing the MPs distribution in terrestrial surface (soil, vegetation) in Svalbard.

2.4. Variability of MPs characteristics and their spatial distribution

MPs in the environment, including the Svalbard region, show huge variability in shapes, sizes, colours, and polymers (Lusher et al. 2015; Duis and Coors 2016; Peeken et al. 2018; von Friesen et al. 2020) for many reasons. First, plastic materials consist of diverse polymers and additives. Once in the environment, these additives continue to modify the fate of plastics. MPs with UV stabilisers will resist for a longer time to degradation compared to a plastic material without such additives. The fragmentation rate will then be different, which will result in a difference in size. The colour also influences the MPs fate since some animals are attracted by specific colours and specifically ingest some and overlook others (Ory et al. 2018; Roch et al. 2020). MPs have a range of intrinsic densities which commonly range from 0.9 to 2.3 g/cm^3 (Hidalgo-Ruz et al. 2012). The shape and the density of MPs will also play a role in their sinking rate (Kowalski et al. 2016; Kane et al. 2019). Their density determines their fate and increases their vertical ubiquity in the aquatic environments, while the shape influences buoyancy. The more rounded, the faster MPs will sink under the same environmental conditions (Kowalski et al. 2016). MPs can also originate from a tremendous number of sources and all inhabited regions of the world, contributing to spatial variability in the environment. In addition to the numerous sources, oceanic and atmospheric currents transport MPs all around the globe, increasing their occurrence in remote areas such as Svalbard. Those currents affect the fate of MPs at both a large and a small scale (Van Sebille et al. 2020). All these parameters simultaneously determine the spatial and vertical distributions and the fate of MPs, leading to the high variability of MPs occurrence in the Svalbard aquatic environments.

2.5. Effects of climate change on MPs distribution

The most important factors affecting the weathering and breakdown of plastic debris to MPs and nano plastics are UV-light and temperature (Andrady 2011). How Arctic conditions, in particular, affect plastic litter weathering is largely unexplored (PAME 2019). Cold winter temperatures and intense 24-hour summer sunlight are both unexplored in their influence on plastic. The harsh weather conditions on the very exposed coasts are further likely to cause fragmentation of already brittle pieces of macroplastic (Brandon et al. 2016). Due to the complexity of the weathering process, it cannot be concluded by deduction if plastic fragmentation is occurring more rapidly in the Arctic than, e.g. in the tropics until now. However, the Arctic is a

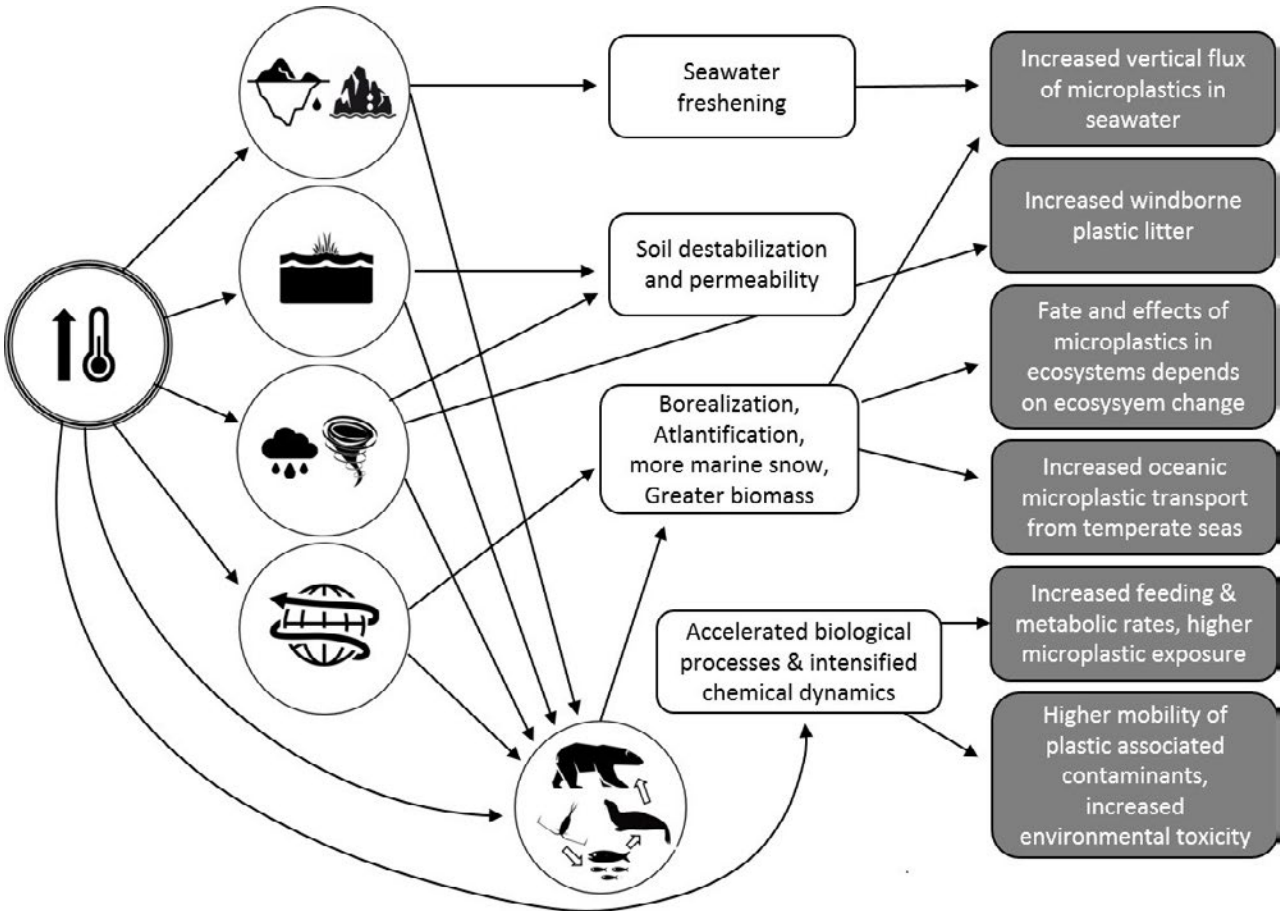

Figure 3: Conceptual illustration showing the potential effects of climate change on plastic pollution in the Svalbard environment

global 'hot-spot' for climate change and is warming more rapidly than other places on the planet due to polar amplification. The term 'polar amplification' refers to a greater temperature change near the poles compared to the rest of the globe for a given global climate force such as a change in greenhouse gasses (Screen and Simmonds 2010; Adakudlu et al. 2019). The major impact of climate change in the Arctic is the loss of sea ice, permafrost thawing, ice melting, glacier reduction, soil destabilization, weather pattern changes, and ocean current alterations (Figure 3). Such climate-induced changes in environmental variables will likely have sizeable impacts on the distribution of MPs in Svalbard and should be investigated.

Changing climate and melting of sea ice and glaciers: The Arctic cryosphere, consisting of ice and snow, is highly sensitive to climate change. Based on satellite images from 1979 to the present, a 40 % decline in Arctic sea ice has been observed (Parkinson and DiGirolamo 2016). Climate models predict that with the current rate of atmospheric CO_2 increase, the Arctic will be ice-free in summer by as early as the 2030s (AMAP 2017). The IPCC further estimates that the earth will pass the threshold of a 1.5°C increase in temperature by 2030 (IPCC 2018), which is considered a 'guardrail' beyond which the effects of climate change will become increasingly severe and difficult to adapt to (Schoolmeester 2019). Increasing warm temperatures accelerate the melting of sea ice and snow and influence MPs distribution. Seasonal expansion and contraction of ice are considered to contribute to the flux of MPs, as particles become trapped as water freezes and released when it melts. It is also projected that the melting of ice may result in the release of the entrained plastics (Obbard et al. 2014). Furthermore, the melting of ice and snow will lead to the freshening of seawater along coasts and sea-ice edges (Woosley and Millero 2020). As freshwater has a lower density than seawater, many floating polymers supported by high saline seawater will now sink and the vertical flux of MPs to deeper waters and sediments will likely increase (Welden and Lusher 2017, Kanhai et al. 2018). As sea ice and glaciers retreat, new shipping routes open up, and resource exploitation will be possible in previously inaccessible areas, making the neo-industrialized Arctic an increasingly important region in the global economy (Avango et al. 2013; AMAP 2017). Plastic, as well as other pollutants, are likely to increase in the Arctic with increased human industrial activity (Granberg et al. 2017). The sources for MPs will thus likely change the distribution and increase as a whole.

Changing climate and melting permafrost: As permafrost melts, the soil becomes destabilized. Old dumping sites in Svalbard and the wider Arctic are placed directly on the permafrost and become undermined as permafrost melts (Granberg et al. 2017). Dumping sites contain a mix of debris and associated pollutants. The increased permeability of land soil leads to increased water-borne transport through the ground, likely carrying all types of pollutants including MPs and its associated chemicals to lakes and coastal waters (Walvoord 2016) (Figure 2).

Changing climate and weather patterns: In the last decades, the frequency of heavy rainfalls and storms has increased in the Arctic and is projected to further increase (IPCC 2018). The Arctic has generally become a wetter and warmer place as indicated by river discharge spikes being heavier and occurring earlier in the spring (Lammers et al. 2001). Future wind patterns will potentially determine the MPs loads deposited from the atmosphere in the Arctic. Heavy rain will also flush MPs from land to the sea at higher rates. Along with this, 'rain-on-snow' events have always occurred in Svalbard as a product of the strong oceanic influence on weather systems in the archipelago (Svendsen et al. 2002) but winter warming is making such events more frequent (Førland et al. 2012). Winter rains may encapsulate pieces of discarded plastics in ice, making the plastics very brittle and liable to break into MPs, which may end up in lakes or the ocean. More and bigger storms will likely lead to greater losses of larger plastic pieces – such as fishing nets and broken snowmobile parts – from ships and dumping sites and land-based activities (Welden and Lusher 2017). Such plastic debris, whether intentionally or unintentionally discarded, are likely to end up in seas, lakes, and other water bodies.

Changed ocean circulation: Ocean circulation and coastal oceanography are undergoing dramatic changes in the wake of climate change. Around Svalbard, Atlantic water thrusts farther north, bringing along the water from more industrialised parts of the world. Lusher et al. (2015) detected higher concentrations of MPs in Atlantic water than in seawater collected closer to glacial outflows, suggesting Atlantic water to function as a vector for MPs pollution to Svalbard and the Arctic. Fibres constituted the majority (95%) of detected MPs. As the Atlantic water input is predicted to increase as climate change progresses (Polyakov et al. 2020), waterborne MPs pollution from southern latitudes is likely to increase in the Svalbard region.

Ecosystem change and biochemical interactions: It is difficult to predict how ecosystems will change in response to global warming and even more difficult to estimate how the distribution and effects of MPs will be altered in relation to ecosystem change. However, as biological borealisation progresses, i.e. the climate change-induced transformation from the Arctic towards more boreal or temperate ecosystems, information gained from scientific investigations in more southern latitudes may become more relevant for the Arctic. In a modelling study, Kvale et al. (2020) predicted an increase in the transport of MPs from surface waters to deeper waters and sediments in the Arctic Ocean as the sea ice melts and exposes more water to sunlight, stimulating increased primary production and microalgal growth. Marine snows (sinking particulate organic matter) and ingestion (and subsequent egestion) by zooplankton (e.g. Cole et al. 2013 and 2016) become vehicles for MPs transport to the ocean depths. This coincides with the predicted increase in marine biomass stimulated by borealisation (Polyakov et al. 2020). However, Lannuzel et al. (2020) predict a decrease in vertical carbon flux close to the sea ice. As first-year ice will replace multiyear ice the developed microalgal ecosystem can no longer be sustained but may instead be replaced by opportunistic microalgal species of low food quality, e.g. *Phaeocystis* sp. (Assmy et al. 2017). This change will cascade through the pelagic food web and smaller grazing zooplankton producing small faecal pellets will replace large sympagic amphipods and *Calanus* species. Small faecal pellets do not sink and vertical flux of organic matter and associated particles will thus slow down and particulates kept in surface waters. Processes near coasts will likely differ from those close to the sea-ice edge (Lannuzel et al. 2020).

Luoto et al. (2019) further showed that the species composition and trophic interactions in Lake Revvatnet, Svalbard, changed with the increased input of turbid glacier meltwater. Colonies of little auks (*Alle alle*) have also grown as the lake remains unfrozen for a longer period. A higher MPs content in lake sediments coincides with these changes, and the authors suggest seabird guano as a vector for MPs pollution, as previously suggested for northern fulmars by Provencher et al. (2018). The birds mistake MPs for food when foraging at sea, which has been observed in little auks in Greenland (Amélineau et al. 2016)

Increased temperatures will accelerate all biological processes and intensify chemical dynamics. For example, ingestion and respiration increase in many invertebrates as temperature rises (Acheampong et al. 2014; Grote et al. 2015; Zhang et al. 2016). Elevated seawater temperatures may thereby lead to increased ingestion of MPs by marine organisms. As the dynamics and mobility of plastic-associated contaminants increase with rising temperatures, the bioavailability of these chemicals may elevate environmental toxicity.

2.6. The ultimate fate of MPs in Svalbard

In aquatic environments, it is expected that the majority of MPs will eventually be transported to the oceans via rivers and coastal areas. The distribution of MPs in the ocean is affected by hydrodynamic forces (waves, tides, and currents) (Rocha-Santos and Duarte 2014). As a result of ocean currents and the wind, ocean garbage patches can be regarded as temporal sinks. The Barents Sea is assumed to become another major convergence zone and, thus, a large sink for plastic debris (Cózar et al. 2017). Svalbard, which borders the Barents Sea on the archipelago's east, would consequently be increasingly exposed to both macro and MPs.

In addition to long-distance transport from other regions of the world, Svalbard will be more exposed to local plastic pollution due to an increase in shipping, tourism, and fishing activities as explained in the previous section (v). The fate of this plastic debris in Svalbard is largely unknown. We expect an accumulation of plastic in different compartments: sediment, beaches, glaciers, and sea ice. Once in the Svalbard region, MPs will be exposed to cold temperatures, entrapment in sea ice (van Sebille et al. 2020), and continuous and absent UV exposure for several months. These parameters will influence the fate of MPs in Svalbard in different ways. Cold temperatures will lower the degradation rate by microorganisms (Bergmann and Klages 2012; Urbanek et al. 2017), resulting in a higher accumulation rate compared to temperate and tropical regions. The cold temperatures and absence of sunlight in winter will inhibit biofilm formation (Chen et al. 2019), and floating MPs will stay longer at the water surface in winter. In the summer, the continuous sunlight might support the formation of a biofilm and the fragmentation of MPs compared to the winter season. However, the average temperature of surface waters around Svalbard remains low even in the summer (for example from 4°C to 7°C in Kongsfjorden) (Tverberg et al. in Hop and Wiencke 2019), limiting the density increase due to biofilm formation. The consequences of MPs entrapment in sea ice are currently unknown. On one hand, the physical abrasion of MPs due to ice formation and movements can be expected; on the other, MPs might be protected from direct exposure to sunlight (Chen et al. 2019).

Despite the influence of these parameters, it can be assumed that MPs will eventually sink to the seafloor, end up on beaches, or be trapped in sea ice temporarily, or might cycle in abiotic and biotic ecosystem compartments. The finding of a great deal of MPs in the deep sea near the HAUSGARTEN observatory (Tekman et al. 2017) suggests that future research should investigate whether under-sea canyons are accumulation locations for plastics. To a certain extent, the Svalbard marine fauna can be considered as a temporary sink and transport medium, but literature is lacking regarding MPs trophic transfer, retention times, whether MPs simply pass through the GI of larger organisms or get accumulated. More data are urgently needed to understand the fate of MPs in cold regions and worldwide in general. Especially, experiments investigating the degradation and sinking rates under Arctic conditions are needed to understand the fate of MPs in Svalbard.

On the seafloor, MPs will be available to many organisms such as snow crabs (*Chionoecetes opilio*) (Sundet 2014) deep-sea starfish (*Hymenaster pellucidus*) (Courtene-Jones et al. 2017) and from sea ice to polar cod (*Boreogadus saida*) due to melting (Kühn et al. 2018). From a food security perspective, the presence of MPs in harvested marine species raises concerns for people.

2.7. Sociological impacts of MPs (e.g. food safety)

As research on MPs has expanded beyond observations highlighting their ubiquity in the oceans, the focus has shifted towards ecological and health risks (GESAMP 2015; Cox et al. 2019). MPs are considered widespread contaminants, ingested directly, or indirectly by trophic transfer through the marine food web (Vandermeersch et al. 2015; Lusher et al. 2015; Desforges et al. 2015; Wright et al. 2013; Nelms et al. 2018). With the identification of MPs in a variety of Arctic organisms (Diepens and Koelmans 2018; Fang et al. 2019; Kühn et al. 2018; Provencher et al. 2018), it has become clear that marine organisms can ingest and even transfer MPs along with the Arctic food webs into top predators including fish and marine mammals. However, little is known today about trophic transfer dynamics, retention times, and whether, and under which conditions, MPs simply pass through the GI of biota or get accumulated in other tissues.

Most of the MPs' research on living organisms has concentrated on marine organisms. Terrestrial organisms have not been regarded as threatened as they are not part of the ocean system, which was considered the main pool of MPs. Recently this perception has changed, and it is now recognised that terrestrial organisms (e.g. reindeer) may be exposed to MPs (Bergmann et al. 2019) and that from them MPs may distribute to their predators. Reindeer (*Rangifer tarandus*) is an important traditional food in Svalbard and is, therefore, a potentially significant contaminant pathway to humans. There is no information available on the MPs ingestion or accumulation in terrestrial harvested wildlife. Therefore, to provide knowledge-based advice for environmental agencies, food security stakeholders (e.g. UN organisations), consumers, and food producers and to enable educated management, there is an immediate need to develop a baseline database of MPs in harvested terrestrial wildlife.

There is ample evidence of MPs ingestion by fish destined for human consumption, including Arctic species (Kühn et al. 2018; Bråte et al. 2016; Morgana et al. 2018; Fischer and Scholz-Bottcher 2017; Leclerc et al. 2012; Nielsen et al. 2014; Welden and Cowie; 2016; Sundet 2014). In bivalves in the Arctic, MPs are reported in blue mussels (Bråte et al. 2018; Halldórsson and Guls 2018; Granberg et al. 2020) and Greenland smoothcockle (von Friesen et al. 2018). Generally, low counts of MPs in seafood species obscured any differences between collection sites, but MPs in fish are commonly analysed in the (GI) only. Often, only larger MPs, such as above 300 µm, are investigated, and sometimes only by visual inspection without chemical identification. Moreover, estimates of larger MPs in the GI can provide rough estimates of MPs ingestion. However, MPs of such size are unlikely to cross the intestinal barrier or to accumulate in any tissue (Pedà et al. 2016; Grigorakis et al. 2017). Additionally, there is evidence from exposure studies that smaller MPs are of higher concern in terms of negative health effects on aquatic organisms and mammals (Kögel et al. 2020). Hence, to investigate if such accumulation is of concern for seafood security at current pollution levels, quantifications of smaller MPs in the consumed tissues are required. The thresholds for translocation or accumulation are not clarified yet, as very little data from the environment is available, and exposure studies do not cover many polymer types and often use round beads instead of the angular fragments and fibres, which are common in the environment. However, a recent report – although not from fish from Arctic regions – describes the occurrence of MPs in fish muscle and liver, but none above 250 µm (Gomiero et al. 2020). Reports including findings of MPs smaller than 50 µm in other seafood species were recently summarized in Kögel et al. 2020a. However, data is scarce, and additionally neither harmonized nor fully quantitative due to methodological challenges (Kögel et al. 2020b). However, the only field studies analysing MPs in seafood species from Arctic regions including MPs below 300µm, point towards that also Arctic seafood species are no exception to containing MPs. Thus, rather low MPs were reported in 11 species of benthic invertebrates sampled from the Bering and Chukchi Seas (>100µm, Fang et al. (2019), in the gastrointestinal tract of Polar cod (Boreogadus saida) (> 35 µm; Kühn et al. 2018) and of Greenland Cod (Gadus ogac) (>20 µm, (Granberg et al. 2020). From exposure studies, there is plenty of evidence that uptake of smaller MPs into other tissues does occur. Nano plastic administered in high doses experimentally reached the brain of fish and nano plastic and MPs lead to altered behavior that may affect the fishes' capability of sustenance (Mattsson et al. 2015; Mattson et al. 2017; Chae et al. 2018; Barboza et al. 2018).

In conclusion, sufficient knowledge of the concentrations per size distribution and uptake and effects of MPs to establish a proper risk assessment for both the environment and human consumers is still lacking (Skåre 2019; Backaus et al. 2018), even though MSFD 2008/56/EC and decision 2017/848/EU require that the situation be documented: "The amount of [...] micro-litter ingested by marine animals is at a level that does not adversely affect the health of the species concerned. [...] Micro-litter shall be monitored [...]". Resultantly, there is an immediate need for an analysis of the occurrence and levels of MPs in seafood organisms as a foundation for knowledge-based advice for food safety authorities, environmental agencies, food security stakeholders, such as UN organisations, and Arctic Inuit authorities and those of other Arctic people, consumers and food producers.

3. Connections and synergies with other SESS report chapters

Many SESS report chapters explore issues that have been identified as gaps related to our understanding of plastics in Svalbard. Figure 4 shows that merging or forming trans-chapter workgroups or task forces must be a way forward for SIOS to utilise the full breadth of competence within the consortium.

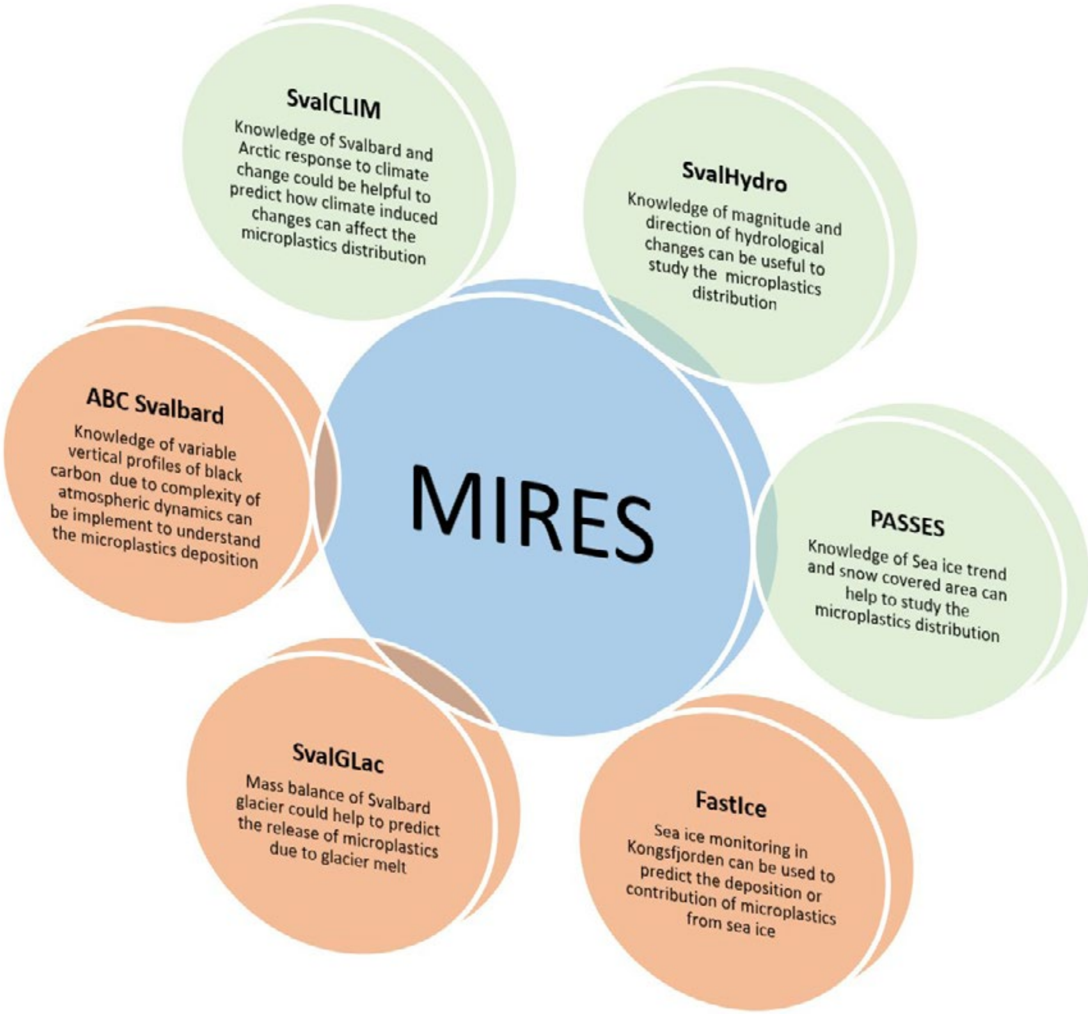

Figure 4: Potential connections with other chapters: SvalCLIM (Gjermundsen et al. 2021), SvalHydro (Nowak et al. 2021), PASSES (Salzano et al. 2021), SvalGlac (Schuler et al. 2020), FastIce (Gerland et al. 2020), and ABC Svalbard (Gilardoni et al. 2020).

4. Unanswered questions

On the basis of the knowledge presented in sections 2.1 - 2.7, we have noted the following knowledge gaps, which are important to be filled for a good understanding of the status of MPs in the realm of Svalbard:

a. *Trends:* The lack of conformity in sampling, analytical protocols, and data units and the paucity of data make it difficult to determine MPs trends in different environmental compartments.
b. *Sources:* There is a lack of information regarding the relative importance of distant and local MPs pollution sources in Svalbard and the Arctic in general.
c. *Toxicological effect:* MPs have been detected in fish and other marine organisms, but there is limited information on the effects MPs cause themselves in environmentally relevant sizes, doses, and combinations and through leakage or uptake of plastic additives (plasticisers and stabilisers) or from the pollutants present in the environment (POPs). As such knowledge emerges, it will be necessary to evaluate to what degree Svalbard deviates from other environments concerning toxicological effects.
d. *Fate:* The long-term transformation and deposition reservoirs of MPs need further investigation in the Svalbard environment. The interactions between physical and biological compartments regarding MPs are poorly understood and effects on ecosystems therefore highly unpredictable.
e. *Effect of climate change:* Global climate change is likely to affect the concentrations, transformation processes, and mobility of MPs in the ecosystem. Such effects are currently mainly speculative and need to be quantified.
f. *Food Security:* MPs concentrations in harvested terrestrial wildlife, marine mammals, fish, and shellfish are poorly documented. It is further unknown whether ingested MPs may have any adverse effect on humans.
g. *Exchange:* Information on MPs levels in the air and their influence on terrestrial environments is lacking.

5. Recommendations for the future

Harmonising methodologies: A workshop is needed to facilitate agreements among international MPs experts on how to start monitoring MPs at the four observatories in Svalbard (Hornsund, Barentsburg, Longyearbyen, Ny-Ålesund). Here, the work currently being finalised by AMAP on MPs monitoring will be highly valuable.

Long-term monitoring: A monitoring programme should be designed to consider societal needs such that the science can provide advice regarding plastic use in Svalbard, wastewater treatment, effects of cruises and other tourism activities, and fishing.

Mapping: MPs in the unexplored parts of Svalbard, which include terrestrial and marine biota needs to be mapped in order to establish a proper risk assessment for both the environment and human consumers.

Collaboration: It is recommended that a Svalbard plastics task force be formed and meet regularly to develop methods and monitoring recommendations to ensure that there is a concerted effort to fill the identified knowledge gaps.

Experiments: Experimental studies of Arctic key species and the possible trophic transfer of MPs under Arctic conditions should be set up.

6. Data availability

This review chapter is an overview of published articles, thesis, and reports. Information about the literature used in this chapter can be found in the reference section.

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Appendix 1

MPs in different abiotic compartments of Svalbard

Location	Year	Medium	Range	Unit	Sampling method	Analytical method	Reference
Ice and Snow							
Svalbard and Fram Strait	2016 – 18	Snow	0 – 14,400	Particles L^{-1}	Surface sample with a spoon	Vacuum filtration (1 mL aliquot) and Fourier-transform infrared microspectroscopy (µFTIR)	Bergmann et al. 2019
Fram Strait, north of Svalbard	2014 – 15	Sea ice	(1.1 0.8) x 10^6 – (1.2 1.4) x 10^7	Particles m^{-3}	Ice coring	Vacuum filtration and µFTIR (FPA detector)	Peeken et al. 2018
Svalbard	2017	Sea ice	0.158	MPs m^{-3}	Boat hook	Visual analysis under stereomicroscope and Fourier transform infrared spectroscopy (FTIR)	Von-Friesen et al. 2020
Water							
Norwegian Sea (Transect from Tromsø up to SW Svalbard (78.07°)	2014	Water (surface)	0 – 11.5	Particles m^{-3}	Manta net (330 µm)	Gravity separation, light microscopy and FTIR	Lusher et al. 2015
		Water (subsurface)	0 – 1.31	Particles m^{-3}	Seawater intake	Vacuum filtration, light microscopy and FTIR	
Svalbard	2017	Water (0–1000m)	0.0007 – 0.048	MPs m^3	Niskin bottles	Visual analysis under stereomicroscope and FTIR	Von-Friesen et al. 2020
Adventfjorden	2018	Water	0 – 3.32	MPs/m^3	manta net and a high-capacity pump system	Visual identification by microscope and FTIR	Purver 2019
Grønfjorden			0.1 – 1.09				
Dicksonfjorden			0.12 – 1.67				
Oxaasdalen			4.0				
Ekmanfjorden			0.36 – 0.83				
Nordfjorden			3.85				
Kongsfjorden	2018	Surface water	23.7 ± SD 11.9 – 74.1 ± SD 43.9	Particles m^{-3}	Surface trawl by using Plankton net	vacuum filtration and FTIR	Scott 2019
Kongsfjorden	2018	Subsurface (160m depth)	53.8 ± SD 3.2 – 92.3± SD 12.8	Particles m^{-3}		vacuum filtration and FTIR	Scott 2019
Svalbard	2015	Water (near wastewater outflow)	97 (Mean)	Particles L^{-1}	n/a	n/a	Sundet et al. 2016

Svalbard	2016	Subsurface water	n.d	Particles L^{-1}	Seawater intake	Vacuum filtration and visual sorting	Sundet 2017
Fram Strait (HAUSGARTEN observatory)	2016	Water	0 – 1287	Particles m^{-3}	High volume pump	Vacuum filtration, visual sorting and FTIR	Tekman et al. 2020
Svalbard	2017	Incoming and outgoing wastewater	Inflow: 14,207 Outflow:83	Particles L^{-1}	Water pump	Enzymatic digestion, vacuum filtration, light microscopy and FTIR	Granberg 2019
		Subsurface water	18.6 – 61.2	Particles L^{-1}	Water pump	Sequential filtration, light microscopy and FTIR	
Sediment							
Knudsenheia lake (western shore of Kongsfjorden)	2018	sediments adhered to rocks	400 (average)	microparticles/m^2		Raman Microscopy, µFTIR and Synchrotron Radiation µFTIR (SR-FTIR)	Gonzalez-Pleiter et al. 2020
Lake Revvatnet	2013	Sediment	7.4	particles/cm^{-1}	Kajak corer	n/a	Luoto et al. 2019
Dicksonfjorden	2018	Sediment	0 –0.63	MP/kg	Sediment sampler	Visual identification by microscope and FTIR	Purver 2019
Fram Strait (HAUSGARTEN observatory)	2015	Sediment	42 – 6595	Particles kg^{-1}	Multicorer	Microplastic sediment separator, Fenton's reagent, visual sorting, FTIR	Bergmann et al. 2017
Barents Sea	2017	Sediment	830 – 3900	Particles kg^{-1}	Grab sample	Density separation, digestion, filtration and visual sorting	Møskeland et al. (2018)
Svalbard	2015	Sediment	0 – 9.2	Particles kg^{-1} DW	Van Veen grab	Density, filtration and visual sorting	Sundet et al. 2016
		Beach sediment	0 – 6.4	Particles kg^{-1} DW	Beach Sand collection		
Svalbard	2016	Sediment	2 – 10	Particles L^{-1}	Van Veen grab	Density separation (NaCl), vacuum filtration and visual sorting	Sundet et al. 2017
Svalbard	2017	Sediment	0 – 134	Particles kg^{-1} DW	Van Veen grab	Density separation (NaCl), vacuum filtration, light microscopy and FTIR	Granberg et al. 2019
		Beach sediment	11 – n/a	Particles kg^{-1} DW	Metal shovel		
Fram Strait (HAUSGARTEN observatory)	2016	Sediment	239 – 13,331	Particles kg^{-1}	Multicorer	Density separation, vacuum filtration, visual sorting and FTIR	Tekmann et al. 2020
Fram Strait	2001 – 2011	Deep-sea sediment	10 – 15	Particles 50 mL	Subsample of a box/ megacore	Density separation (NaCl), visual sorting, FTIR	Woodall et al. 2014