Environmental Science Advances

ROYAL SOCIETY OF CHEMISTRY

CRITICAL REVIEW

View Article Online
View Journal

Cite this: DOI: 10.1039/d5va00261c

Industrial and public infrastructure as local sources of organic contaminants in the Arctic

Roland Kallenborn, *[a] Geir Wing Gabrielsen,[b] Katrin Vorkamp, [c] Lars-Otto Reiersen,[d] Anita Evenset,[e] Kristin B. Pedersen,[e] Simonetta Corsolini, [fg] Nicoletta Ademollo,[g] Yifan Li,[hi] Zi-Feng Zhang,[i] Håkon Austad Langberg, [j] William F. Hartz, [kl] Frank von Hippel,[m] Derek Muir,[n] Cynthia A. de Wit, [o] Maria J. Gunnarsdottir, [p] Pernille Erland Jensen, [q] Gunvor Marie Kirkelund,[q] Gijsbert Breedveld,[rj] Susan Bengtson Nash,[s] Jan Ludvig Lyche[t] and Elena Barbaro[g]

Arctic pollution has been a focal point in environmental research over the past five decades. Recently, the number of pollutants identified as relevant to the Arctic has significantly increased. Consequently, the expert group on Persistent Organic Pollutants (POPs) and Chemicals of Emerging Arctic Concern (CEACs) of the Arctic Monitoring and Assessment Programme (AMAP) has prepared a series of assessments of contaminants in the Arctic, including influences of climate change. This review addresses local sources of Arctic organic pollutants associated with infrastructure in the Arctic. Industrial, military, and public infrastructures, including domestic installations, sewage treatment, solid waste management, and airports, were identified as significant local pollution sources. Additionally, operational emissions (*e.g.*, from shipping, transportation, heating, and power production) contribute to the overall local pollution profile. Based on currently available scientific information, elevated POP and CEAC levels are mostly found in close proximity to identified local pollution sources. To date, hazardous effects have only been confirmed for a few selected chemicals, such as polycyclic aromatic compounds (PAC) and certain pharmaceutical residues. However, studies are biased in the sense that they often focus on well-known contaminants, at a risk of overlooking CEAC and their effects. The review identifies several measures to reduce human impacts on local Arctic environments, including (i) using local indicator pollutants in ongoing national monitoring schemes, (ii) harmonizing emission reduction policies and licensing of industrial activities in the region to minimize exposure risks and environmental pollution, (iii) encouraging local municipalities, industries, and related stakeholders to coordinate their activities to minimize pollutant emissions.

Received 8th August 2025
Accepted 20th October 2025

DOI: 10.1039/d5va00261c

rsc.li/esadvances

[a]Faculty of Chemistry, Biotechnology and Food Sciences (KBM), Norwegian University of Life Sciences (NMBU), Ås, Norway. E-mail: roland.kallenborn@nmbu.no
[b]Norwegian Polar Institute, Tromsø, Norway
[c]Department of Environmental Science, Aarhus University, Roskilde, Denmark
[d]Arctic Knowledge AS, Oslo, Norway
[e]Akvaplan-niva, Tromsø, Norway
[f]Department of Physical, Earth and Environmental Sciences, University of Siena, Siena, Italy
[g]Institute of Polar Sciences – National Research Council (ISP-CNR), Venice, Italy
[h]Dalian Maritime University (DMU), Dalian, Peoples Republic of China
[i]School of Environment, Harbin Institute of Technology (HIT), Harbin, Peoples Republic of China
[j]Norwegian Geotechnical Institute (NGI), Oslo, Norway
[k]NILU-Climate and Environmental Institute, Kjeller, Norway
[l]Man-Technology-Environment Research Centre (MTM), Örebro University Örebro, Sweden
[m]Mel & Enid Zuckerman College of Public Health, University of Arizona, Tucson, Arizona, USA
[n]School of Environmental Sciences, University of Guelph, Guelph, ON, Canada
[o]Dept. of Environmental Science (ACES), Stockholm University, Stockholm, Sweden
[p]Faculty of Civil and Enviromental Engineering, University of Iceland, Reykjavik, Iceland
[q]Department of Environmental and Resource Engineering, Technical University of Denmark (DTU), Lyngby, Denmark
[r]Arctic Technology Dept., University Center in Svalbard (UNIS), Longyearbyen, Svalbard, Norway
[s]School of Environmental Science, Griffith University, Brisbane, Australia
[t]Faculty of Veterinary Medicine (VET), Norwegian University of Life Sciences (NMBU), Ås, Norway

Environmental significance

The Arctic is known to receive contaminants from long-range transport, yet this review reveals that local infrastructure—including domestic, industrial, and military installations—is a significant and overlooked source of organic pollution. Drawing on decades of monitoring data and recent research, the manuscript demonstrates that Persistent Organic Pollutants (POPs) and Chemicals of Emerging Arctic Concern (CEACs) are frequently emitted from infrastructure-related activities such as heating, waste management, transportation, and firefighting. These pollutants include polycyclic aromatic hydrocarbons (PAHs), per- and polyfluoroalkyl substances (PFAS), pharmaceuticals, flame retardants, and plastic additives, many of which persist under cold Arctic conditions. The review highlights how climate change amplifies the environmental mobility and ecological risks of these contaminants, particularly through permafrost thaw and altered hydrological cycles. It identifies distinct pollution profiles for different infrastructure types and emphasizes the need for integrated monitoring and policy frameworks that account for both long-range and local sources of contamination. This work provides a critical shift in perspective, urging researchers and policymakers to recognize and mitigate the growing impact of local infrastructure on Arctic ecosystems and communities, thereby contributing to more effective environmental protection strategies in a rapidly changing region.

Introduction

For many decades, the Arctic was considered a region without significant local pollution sources, while many legacy pollutants have been proven to undergo atmospheric or ocean borne long-range transport. Hence, the Stockholm Convention on persistent organic pollutants (POPs) evaluates long-range transport as an important criterion for POP classification and states that "the occurrence and presence of chemicals of emerging concern in Arctic/polar environments" is an indication of long-range environmental transport.[1] In order to provide sound advice on the origin of chemical contamination in the Arctic, an existing focus on the long-range transport of POPs and chemicals of emerging Arctic concern (CEACs) to the Arctic[2] requires parallel investigations into occurrence and significance of local sources of pollution in the Arctic.[3] CEACs are considered as those compounds of concern in the Arctic that are not regulated under the Stockholm Convention.[4–6]

As early as the 1990s, a clear association between particle emissions and elevated soot levels in air around settlements was found for several locations in the Arctic.[7,8] Also, pollution near settlements in the Arctic has been documented since the 1980s for polycyclic aromatic hydrocarbons (PAHs), POPs and trace metals.[9–12] During a pilot study in Svalbard (Norwegian Arctic), the characteristic emission profiles for selected power plants in Longyearbyen, Svea, Barentsburg and Pyramiden were investigated for PAH emissions.[13] Furthermore, a complementary study on emissions from cargo and cruise ships revealed a considerable contribution to pollution levels in harbours of Svalbard.[14] Based on these investigations, locally emitted aerosols are assumed to contribute significantly to the overall atmospheric deposition of anthropogenic pollution onto Svalbard surfaces. Similar results were reported from Greenland, Canada, and Alaska as well as the Russian Arctic.[15–19]

Recently, local sources of selected per- and polyfluoroalkyl substances (PFAS) were detected in the Western Arctic including the European Arctic.[20] As for POPs that might be emitted locally, these sources add to the long- and medium-range transport that is known for PFAS, also involving precursors of environmentally stable PFAS, and oceanic and riverine long-range transport.[21–23] A considerable potential for local pollution was identified for several CEACs,[24] besides unregulated PFAS including current-use flame retardants, pharmaceuticals and personal care products (PPCPs), siloxanes, phthalates and other chemicals. However, to date, for most of the CEACs no science-based evidence is available with respect to:

- Local contamination source characterisation.
- Mechanisms for short- and medium-range transport.[25]
- Water-based environmental mobility.
- Information on deposition properties under Arctic conditions.
- Uptake by and exposure profile of Arctic biota.
- Hazardous consequences and direct effects of relevant local pollutants in Arctic biota.

This review presents and discusses local pollution associated with domestic, public, industrial, and military infrastructures in the Arctic including solid waste handling. The term "domestic infrastructures" covers all supporting installations in settlements and cities associated with life support of the local population, such as electric power supply, drinking water supply, wastewater treatment, solid waste treatment, heating, and transport-related infrastructure, such as harbors and airports. Industrial infrastructure includes the exploitation of natural resources such as oil, gas (on land and offshore), and minerals as well as fisheries and aquaculture. Military infrastructure includes installations associated with military activities, currently or in the past, such as airports, early warning sites, dedicated shipyards, and bases. Here, also military vessels and large-scale exercises are included. The focus of this review will be on organic environmental pollutants, *i.e.* POPs and CEACs, as opposed to *e.g.* heavy metals, radionuclides or nutrients. Based on the available scientific literature, monitoring data and national reports, this review addresses the following questions:

(1) How important are infrastructure-related emissions as local sources for the overall contamination in the Arctic with organic pollutants?

(2) Can spatial and temporal trends be identified for infrastructure contribution to local pollution in the Arctic?

(3) Do characteristic pollution patterns and profiles exist for infrastructure-related pollutant sources in the Arctic?

(4) How is Arctic climate change influencing local infrastructure and their associated chemical emissions?

Transport, fate and effects of POPs and CEACs

For centuries, polar regions were considered pristine and unaffected by anthropogenic pollution. However, from the 1500s onwards, local pollution sources might already have been

introduced when natural resources in polar regions were vigorously exploited. In Iceland, the walrus populations were already extinct by foreign hunting companies in the 17th century.[26,27] The observations of abundant natural resources during the expeditions by Willem Barents in the 1590s opened the Svalbard region and Eastern Greenland for large-scale commercial whaling and seasonal hunting of walrus.[28,29] During the following centuries, both right whales (*Eubalaena*) and walrus (*Odobenus rosmarus*) populations in the Barents Sea were brought to the brink of extinction.[30–36] Already during the 1600s, large seasonal settlements were established by European whaling companies in various high Arctic locations for processing the whale and walrus carcasses before shipping the products to the European markets.[31,35,37] These early European settlements in the Arctic presumably produced their own local sources of contaminants, mainly from combustion processes (*e.g.* open fires for whale blubber cooking and heating of poorly insulated houses) leading to emissions of polycyclic aromatic compounds (PACs) including polychlorinated-*p*-debenzo dioxins and furans (PCDD/Fs) and probably also heavy metals such as lead (Pb) and mercury (Hg). The long-range transport of pollutants has been observed for a long time by local and indigenous people. The first observation of long-range transport by foreigners was reported by Nansen *et al.* after their expedition to the Arctic in the period 1893–1896 where they observed dark sea-ice/snow which they interpreted as a result of long-range transport of soot from Europe to the Arctic.[38]

Today it is known that long-range transport is the major entrance route for POPs into the Arctic environment.[2,39–42] The first comprehensive assessment of POPs in the Arctic was prepared under the Arctic Monitoring and Assessment Programme (AMAP) in the 1990s and established the long-range transport from industrial regions to the Arctic as the main source of Arctic POP pollution.[43] Given the semi-volatile nature of most known POPs, the main environmental transport identified for these contaminants is by air.[44] However, for some compounds, such as β-hexachlorocyclohexane (β-HCH) and PFAS, ocean transport is considered as particularly important.[45–50] Since this early AMAP assessment, the knowledge of long-range atmospheric transport as the main pollution source of POPs in the Arctic has been updated repeatedly,[51–57] and the different long-range environmental transport processes were reviewed by Hung *et al.*[58] A historical review of the role of AMAP in the coordinated efforts of circum-Arctic pollutant monitoring and effect evaluation can be found in a recent publication.[59]

Given their physical–chemical properties, a significant proportion of POPs are associated with particles during their transport in the atmosphere.[60,61] The particle composition in northern atmospheric environments is expected to reflect a combination of emissions from local sources (household combustion, fossil fuel combustion, minor industrial releases) and long-range transport.[62,63] However, the role of local atmospheric pollution and its effect on human and environmental health in the Arctic is still only sparsely investigated as stated in a recent review.[64] An earlier model-based assessment predicts increasing trends for emissions of particulate materials including black carbon from local domestic and industrial sources as well as increased ship traffic in the Svalbard region.[65] Therefore, both increasing atmospheric particulate matter, and a changing composition will have direct effects on transport and deposition properties of POPs and other airborne pollutants such as PAHs in the North.

We observe today that the list of CEACs is constantly growing. AMAP recently published a list of around 600 priority CEACs.[66] New *in silico* and non-target screening approaches aiming at identifying new or overlooked chemicals have the potential to extend this list considerably.[67] Their physical–chemical properties might be different from those of well-studied POPs, with implications for transport patterns to and in the Arctic. Furthermore, many of these substances are associated with anthropogenic activities and consumer products, potentially increasing the importance of local emission sources. Therefore, scientific focus needs to be on identifying and characterizing sources of these CEACs and their fate in the Arctic environment.

A group of CEACs for which local sources have been clearly identified includes pharmaceutical residues and chemicals used in personal care products (*i.e.* PPCPs). PPCPs are mainly emitted to the aquatic environment.[68] Although PPCPs are usually not considered persistent (with few exceptions), their high-volume releases into aquatic environments often exceed transformation rates, leading to their 'pseudo-persistent' occurrence in Arctic recipients. It can also be assumed that pollutants, including pharmaceuticals, are more stable at the low ambient temperatures of the Arctic, enhancing their pseudo-persistent properties.[69–71] It is apparent for POPs and PAHs that environmental conditions affect physical–chemical properties and degradation pathways, resulting in an environmental behaviour of contaminants that might be different from that at lower latitudes.

Pharmaceuticals are designed to execute specific biochemical and physiological functions during medical treatment. These biochemical properties expressed under uncontrolled environmental conditions have the potential to result in toxicological effects on Arctic fauna.[72] For the majority of those pharmaceutical agents, effect thresholds in ecosystems are not known, but it can be assumed that extended exposure to these pharmacological active chemicals may harm non-target aquatic organisms. Today more than 100 PPCPs have been reported from Arctic locations, and some of these, such as surfactants, have been found in concentrations up to the high $\mu g\ L^{-1}$ (sewage) or $\mu g\ g^{-1}$ (contaminated soil) range. Most PPCPs were found in the vicinity of settlements mainly associated with sewage related emissions.[66]

Recent climate reports[41,73–79] have identified significant climate-related changes in Arctic environments, which are also likely to influence the short- and long-distance transport and fate of organic pollutants in Arctic environments.[77,78] The cryosphere plays a key role in the distribution and transformation of pollutants in the Arctic[76,77] and is currently changing in an unprecedented way in geological history.[80–82] Release profiles, distribution pathways as well as the inter-compartmental exchange (atmosphere–soil/water; water–sediment; soil/water–

biota) will be influenced by the temperature rise and changes in the Arctic environment, with potential consequences for human and environmental exposure. These changes will likely also affect emission profiles and strength of local pollution sources in the Arctic as discussed in detail in the review by Muir *et al.*[83] reported in this special issue.

Public infrastructure

According to the Arctic council (AC, http://arctic-council.org), approximately 4 million people are living in the Arctic, based on the area defined by AMAP. Except for a small number of larger cities in the Arctic region such as Tromsø (Norway, approximately 70 000 inhabitants), Fairbanks (Alaska, USA, approximately 95 000 inhabitants), Reykjavik (Iceland, approximately 140 000 inhabitants) and Murmansk (Russian Federation, approximately 300 000 inhabitants), settlements in the Arctic are usually isolated small towns and villages with a size of 100–1000 inhabitants.[84] Basic public infrastructure such as water supply systems, power plants and waste handling facilities must be operational under the conditions of the harsh Arctic environment. Regardless of their size, all settlements need to be continuously supplied with goods and services, usually over large distances. This involves complex logistic operations including long distance air cargo, ship-based transportation, and passenger transportation. The challenges related to running the required infrastructure, which might include outdated, limited, or low-technology solutions, create an elevated exposure risk (organic and inorganic pollutants) for local environments.[85–88] Organic contaminants can arise from various sources (Fig. 1):

Energy and heating: infrastructures such as electric power plants, power generators, and heating systems are essential to maintain livelihood and comfort for the daily life in Arctic communities. However, operating these infrastructures may contribute to the emission of potentially harmful pollutants into the adjacent Arctic environment.[64,66]

Sewage and wastewater: inadequate treatment of sewage and wastewater from human settlements can introduce organic pollutants into Arctic waterways. Besides potentially harmful chemicals, these discharges can contain nutrients like nitrogen and phosphorus, which can lead to eutrophication, causing algae blooms and disrupting aquatic ecosystems,[89] and the biogeochemical cycles, including contaminants.[90] In addition, the local impact of wastewater releases from a settlement is also reflected by the presence of personal care products (such as fragrance materials) in coastal seawater.[91] Wastewater emissions as a potential local source of organic chemicals are addressed in the review by Jensen *et al.*[92]

Landfills and waste management: improper disposal of solid waste can result in the release of pollutants into the environment. In the Arctic, where waste management infrastructures may be limited, landfills can contribute to the pollution of the surrounding soil and water, especially as they are often comprised of waste dumped on top of the permafrost and hence readily mobilized.[93] Furthermore, the incineration of waste is common practice in remote communities with insufficient waste collection systems, with the risk of emissions of particulate materials (flyash), and harmful chemicals to air.[94–97] Furthermore, remaining residues (bottom ash) containing pollutants can penetrate into the surface soil and may even contaminate groundwater aquifers after permafrost has retreated.

Accidents: oil spills from transportation activities, such as shipping and land-based transportation, pose a serious threat to the Arctic environment. These spills can introduce organic pollutants into the water, affecting marine life and ecosystems for years.[98]

Maintenance of infrastructure: chemicals used in technical maintenance and operational support, such as road deicers, pesticides, and industrial chemicals, are also identified as local pollutants.[20,99–101]

Use of consumer products: modern societies use many consumer products for lifestyle and health, including cosmetics, surfactants, pharmaceuticals, food additives, preservation agents, plasticizers, flame protection agents and many more. Potentially problematic chemicals can be released from these materials during use and after disposal and may cause potential effects on the local Arctic environment.[102–113]

Some examples of local Arctic pollution sources are depicted in Fig. 1.

In recent years, acknowledging the potential negative consequences of infrastructure-related pollution, some municipalities have addressed these issues by supporting transitions to cleaner energy sources, improvements of energy efficiency, implementation of better waste management practices, and sustainable development in general. Additionally, international cooperation and policies aimed at reducing emissions and pollution in general are crucial in protecting the Arctic environment.[114–117]

Pollution from life supporting infrastructure in operation

Power plants in the Arctic usually rely on fossil fuels such as coal and diesel and cause emissions of pollutants such as particulate matter, polycyclic aromatic chemicals (PACs), volatile organic chemicals (VOCs), nitrogen oxides, sulfur dioxide, and carbon dioxide into the atmosphere.[118–120] These emissions contribute

Fig. 1 Selected examples of potential Arctic local pollution sources.

to local atmospheric pollution and haze and might also contribute to acidification of aqueous environments. "Arctic haze" events can be caused by pollutants from long-range transport as well as local emissions.[121] These events usually contain organic pollutants such as PAHs and PCBs which will ultimately accumulate in snow and ice.[17,122–127] Such Arctic haze events have been reported and scientifically documented since the early 1950s. Pollutants associated with Arctic Haze have also been shown to accumulate in wildlife and contaminate water sources, with potential effects on human exposure and health.[5,124,125,128–132] Besides energy generation, petroleum products also play an important role as petrol for vehicles and jet-fuel for airplane operations. Most petroleum products can be degraded by micro-organisms in soil and groundwater although at considerably lower degradation rates in the Arctic compared to those reported for low latitude environments.[133–136] Heating infrastructure is necessary in the Arctic all year around. In some cases, these infrastructures operate on inefficient or outdated technologies,[137–139] potentially emitting larger amounts of organic pollutants into the air than state-of-the-art systems would. Arctic infrastructure is often the source of a diverse array of contaminants. In a report from Ny-Ålesund oil contamination from an oil tank, which supplied oil to the power station, was the main source of pollution of water, soil and vegetation in the area below the oil tanks.[140] A recent survey in surface soils from an abandoned mining location in Svalbard revealed elevated trace metal and PAH levels.[140] This former mining town, Pyramiden in Billefjord (Svalbard), has been closed and abandoned since 1998. After a first statistical evaluation of the results, the combination of industrial (mining) and domestic sources (fossil fuel driven powerplant and heating circulation) were assumed to be the main sources of these elevated levels in surface soils.[141] Longyearbyen, the largest settlement on Svalbard, has also been investigated regarding potential local emissions from community supporting infrastructures (power plant and heating). The short term emission profiles for PAHs and their major nitrogen- and oxygen-containing transformation products in air indicated a significant contribution from power plant emissions and local traffic.[142] Furthermore, the PAC profiles in air showed a seasonal pattern that reflected emissions from the local fossil fuel driven vehicles, heating and the coal-fired local power plant.[143] The profile of the nitrogen- and oxygen-containing PAH transformation products is strongly influenced by photochemical processes as this recent survey revealed.[142,143] Hence, the atmospheric levels measured for oxy- and nitro-PAHs during the polar night season are significantly higher compared to the midnight sun season (Fig. 2). However, the emission profile of parent PAHs does not seem to be affected by seasonal patterns.

Fossil fuel driven vehicles as pollution sources in the Arctic

For PAHs, seasonal distribution patterns have been identified in surface snow at various locations in Svalbard. Especially retene is found as indicator for local contamination sources.[144] In Arctic remote regions, with an average of 8-months of snow and ice coverage annually and a very limited network of roads,

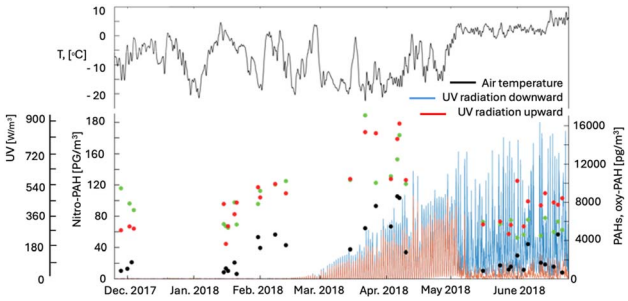

Fig. 2 Atmospheric temperature, UV radiation, and daily total (gaseous + particulate) concentrations of polycyclic aromatic hydrocarbons (PAHs), *i.e.* Σ_{22} PAHs, Σ_{29} oxy-PAHs, and Σ_{35} nitro-PAHs, in the urban air of Longyearbyen, figure reproduced with permission from Drotikova *et al.*[143] under the Creative Commons Attribution 4.0 International licence (CC BY 4.0).

snowmobiles are often the preferred means of transport. Most snowmobiles are operated on 2-stroke engines, which have a low combustion effectivity and emit unburned fuel, VOCs, PAHs and other chemicals.[145] A study in Longyearbyen (Svalbard) investigated the traffic-related emission profile from snow scooters in 2007 (Fig. 3).[146,147] The daily concentrations of benzene, toluene and xylene (BTX) in air were approximately 100 times higher during the main snowmobile season (April–May) compared to the Arctic summer (June–August). These differences were attributed to emissions from snowmobiles. The levels during the late snowmobile season were found to be comparable to BTX monitoring data reported from large western cities (Fig. 3). Similar results have been reported from other Arctic regions where snowmobiles are frequently used for transportation.[148–155]

A survey on the spatial distribution of VOCs and PAHs in the vicinity of Longyearbyen conducted in 2010 showed several volatile and persistent emissions from traffic- and power plant-related sources in Svalbard[157] (Fig. 4). Elevated PAH concentrations were found in soil and snow samples collected at known hotspots (power plant and gas station). The VOC concentrations (sampling April–May 2010) in air at Longyearbyen were lower compared to a previous screening (2007) as depicted for benzene in Fig. 3. These differences are likely due to technical advancement and the larger proportions of 4-stroke engine driven snowmobiles in Longyearbyen in 2010.

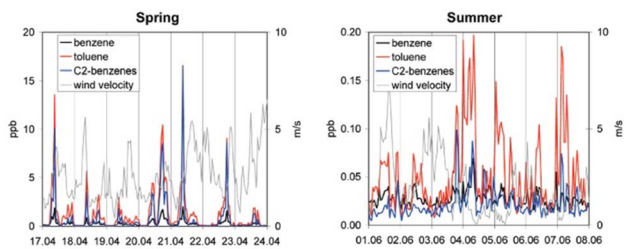

Fig. 3 Concentrations [parts per billion = ppb] of benzene, toluene and C2-benzenes incl. wind velocity in spring (17.04–24.04.2007) and in summer (01.06–08.06.2007), figure redrawn with permission from Reimann *et al.*[156]

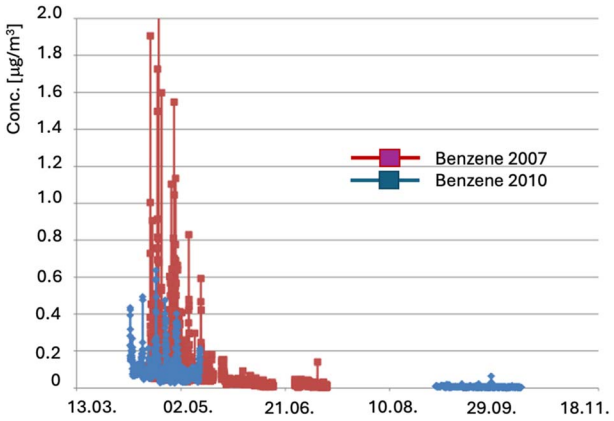

Fig. 4 Benzene level comparison in Longyearbyen air between the 2007 (red) and 2010 (blue) sampling campaign, figure reproduced with permission.[157]

Furthermore, a transect of soil samples crossing over the most frequently used snowmobile route in Svalbard close to Sassendalen valley was characterised by elevated PAH levels with the highest levels found for $\sum_{16}$EPA PAHs (*i.e.* expressed as the 16 priority PAHs of the US Environmental Protection Agency) in the centre of the transect. These characteristic patterns are most probably a result of emissions from snowmobile use during the previous winter season. Increased locally induced air pollution was also reported for the usage of studded tires under Arctic climate conditions.[158]

Arctic settlements as local sources of pollution

Pharmaceuticals and chemicals in personal care products (PPCPs) are identified as relevant contaminants released from domestic, public and hospital-related sources. Most of these chemicals are released with wastewater and/or inappropriate waste handling.[159–161] For Arctic environments, sewage-related release of contaminants is considered an important source of PPCP in freshwater and coastal environments.[68,162,163] This has been further detailed in the review by Jensen *et al.*[92] The low technological standards (primary treatment) or absence of sewage treatment in combination with low ambient temperatures in the receiving waters are considered as main factors for the high emission rates and environmental stability observed for PPCP residues in the Arctic.[68] This was demonstrated in a Norwegian study from 2007 where sewage emissions from Oslo (60° N), Tromsø (69° N) and Longyearbyen (78° N) were compared (Table 1).[68] The Arctic sewage effluents contain elevated concentrations of residues that would likely be retained in modern sewage treatment facilities such as those often operated in lower latitude cities (<66° N latitude). These chemicals include the non-steroid anti-inflammatory agents (=NSAIDs) ibuprofen, diclofenac, and the stimulant caffeine.[68] Introduced into the aquatic environments, PPCPs are usually readily degraded under lower latitude environmental conditions but show a surprisingly high recalcitrance and environmental stability under cold Arctic conditions, as discussed above. This stability was confirmed in a field study in Tromsø and Longyearbyen on five selected serotonin-reuptake inhibiting (SSRI-) antidepressants and their respective transformation products. Both parent compounds and their major transformation products were found in all samples with the highest levels in effluents from Longyearbyen (sum concentrations 5 SSRI max. 3 ng L^{-1}).[164]

The group of PFAS includes several thousand individual compounds.[166] Hence the list of relevant PFAS detected in the Arctic is also constantly increasing. PFAS have been identified as local contaminants in the Arctic mainly associated with the application of aqueous film forming foams (AFFFs),[20,167] open waste dumps,[20,168] sewage and wastewater[169] as well as outdoor activities.[170–173] PFAS pollution in the Arctic has been reviewed previously, with a focus on non-regulated PFAS compounds.[66,174] Recently the presence of 36 different PFAS were confirmed in surface snow in and around Longyearbyen on Svalbard. Several PFAS were associated with local sources in Longyearbyen, whilst the presence of several others was mainly due to their atmospheric formation from precursor compounds. This showed that both a combination of direct local emissions and atmospheric transformation processes contributed as major sources in this area. A seasonal deposition profile was identified by collecting ten precipitation events (snow samples) between January and August 2019 on the Foxfonna ice cap in Spitsbergen (Svalbard). The transition between the polar night season (24 h darkness) into midnight sun (24 h sunlight) is reflected in the seasonal trend of several PFAS (Fig. 5) indicating a significant contribution from the photochemical transformation of precursor PFAS as parent compounds (*e.g.* fluorotelomer alcohols, hydrofluorocarbons and perfluoroalkane sulfonamides). Perfluorooctane sulfonate (PFOS), perfluorobutane sulfonamide (FBSA), hexafluoropropylene oxide dimer acid (HFPO-DA; GenX) as well as C_2–C_{11} perfluorocarboxylic acids (PFCAs, which include trifluoroacetic acid (TFA), perfluorooctanoic acid (PFOA) *etc.*) all correlated with solar flux and had deposition fluxes 7.6–190 times higher during 24 h daylight. Trifluoromethanesulfonate (TFMS) was also detected ubiquitously in surface snow in this study, but no local or long-range source could be assigned.

A temporal trend study into the historical atmospheric deposition of PFAS, including ultra-short chain compounds, was conducted using an ice core from the Lomonosovfonna ice cap on Svalbard.[176] This confirmed that the C_2–C_4 PFCAs and in particular TFA are among the individual PFAS with the highest annual deposition fluxes in Svalbard (and the Arctic) with annual fluxes up to 8–200 ng per m^2 per year.[21] In this study, TFA represented *ca.* 71% of the total C_2–C_{11} PFCAs. Comparisons of PFCA ratios with those in samples taken at the Longyearbyen landfill and airport indicated that the landfill and airport were not a source of C_6–C_{11} PFCAs to the remote Lomonosovfonna ice cap (79 km away).[21]

In Longyearbyen (population *ca.* 2200), surface snow samples showed sporadic variations in PFAS deposition fluxes during January–May 2019, which might be explained by changes in several factors (direct local emissions, snow cover/temperature, transformation of precursors from local source). These findings underscore the importance of PFAS as widespread Arctic

Table 1 Concentrations of pharmaceuticals and personal care products (PPCP) in samples from the Pharmafate pilot study in 2007.[165] Abbreviations: LOQ = limit of quantification, n.a. = not analyzed, n.d. = not detected. Reproduced under the Creative Commons Attribution 4.0 International licence (CC BY 4.0)[a]

Conc. (ng L^{-1})	Oslo		Tromsø		Longyearbyen	
Target PPCPs	Effluent ($n = 1$)	Seawater ($n = 2$)	Effluent ($n = 8$)	Seawater ($n = 8$)	Effluent ($n = 5$)	Seawater ($n = 2$)
Ibuprofen	10	n.d.–52	448	n.a.	30–403	0.4–1
Hydroxy-ibuprofen	126	188–243	3614	n.a.	8–1398	2–34
Carboxy-ibuprofen	42	109–213	70 170	n.a.	41 –34 028	6–26
Diclofenac	25	n.d.–48	78	n.a.	30–1074	1–4
Triclosan	11	n.d.	350	n.a.	28–803	2–2.3
Caffeine	23	5–96	n.a.	n.a.	501–50 704	24–41
Citalopram	238	n.a.	63–102	<LOQ	<LOQ	n.d.
Desmethyl-citalopram	310	n.d.	118–215	<LOQ	<LOQ	n.d.
Didesmetyl-citalopram	10	n.a.	6 – 10	n.d.	n.d.	n.d.
Fluoxetine	8	n.a.	1–5	n.d.	n.d.	n.d.
Norfluoxetine	2	n.a.	0.7–2.5	n.d.	n.d.	n.d.
Fluvoxamine	1	n.a.	0.8–1.7	n.d.	n.d.	0.5–0.8
Sertraline	8	n.a.	8–90	n.d.	n.d.	<LOQ
Desmethylsertraline	6	n.a.	n.d.	n.d.	n.d.	n.d.
Paroxetine	4	<LOQ	3–13	n.d.	n.d.	0.6–1.4
Tetracycline	n.d.	n.d.	n.a.	n.a.	0.6–1.1	n.d.
Trimethoprim	0.8–0.9	n.d.	n.a.	n.a.	0.07–0.15	n.d.
Sulfamethoxazole	0.2–0.3	n.d.	n.a.	n.a.	n.d.	n.d.

[a] n.d. = not detected, table reproduced from ref. 68.

pollutants, which could originate from both local and long-range sources. The sources of TFMS are still unclear.

PFAS was also measured and reported in consecutive sediment surveys in Svalbard from seven locations in the coastal waters of Isfjorden, western Spitsbergen (Fig. 6 and 7). The same samples were also analyzed for polybrominated diphenyl ethers (PBDEs) and other POPs in 1998, 2005 and 2009. PFOS was detected in 6 of 8 analyzed samples, while perfluorononanoic acid (PFNA) was only detected in sediment from Advent-fjorden.[177] The PFAS pattern was dominated by PFOS (0.1–0.5 μg per kg dw) and perfluorooctanoic acid (PFOA) (<0.1–1.3 μg per kg dw), but also perfluorohexane sulfonate (PFHxS) (<0.01–0.03 μg per kg dw) and PFNA (<0.1–0.4 μg per kg dw) were detected in some of the samples. All other PFAS were below the detection limit.

Polycyclic aromatic hydrocarbons (PAHs) are considered as fingerprint chemicals for all types of fossil fuel-based combustion processes.[178–181] PAHs may originate either from

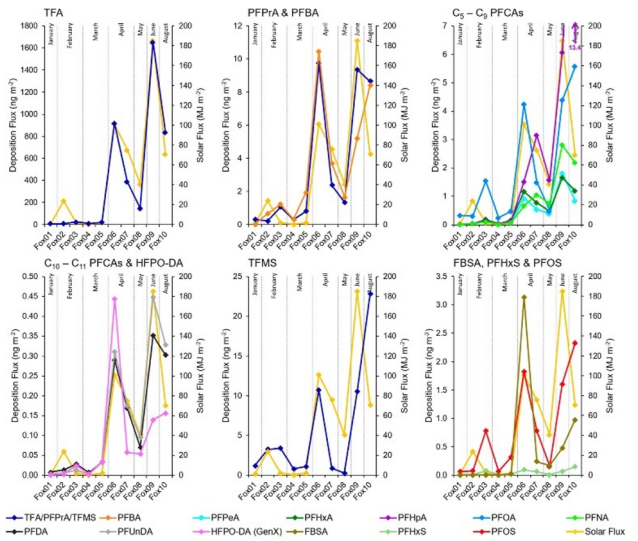

Fig. 5 Deposition fluxes per precipitation event (ng m^{-2}) of trifluoromethane sulfonate (TFMS), perfluorohexane sulfonate (PFHxS), perfluorooctane sulfonate (PFOS), perfluorobutane sulfonamide (FBSA), hexafluoropropylene oxide dimer acid (HFPO-DA; GenX) and C_2–C_{11} perfluorocarboxylic acids (PFCAs) in surface snow on the Foxfonna ice cap during January to August 2019, plotted alongside the solar flux (MJ m^{-2}).[175,176] Reproduced under the Creative Commons Attribution 4.0 International licence (CC BY 4.0).

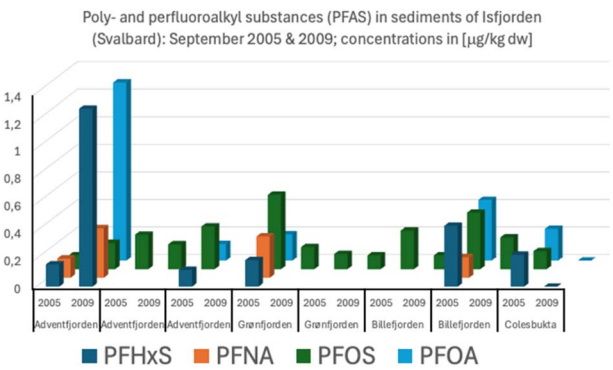

Fig. 6 Perfluoroalkyl substances (PFAS, μg per kg dry weight = dw) in sediment from Isfjorden, September 2005 and 2009.[199] PFHxS = perfluorohexane sulfonate; PFNA = perfluorononanoic acid; PFOS = perfluorooctane sulfonate; PFOA = perfluorooctanoic acid.

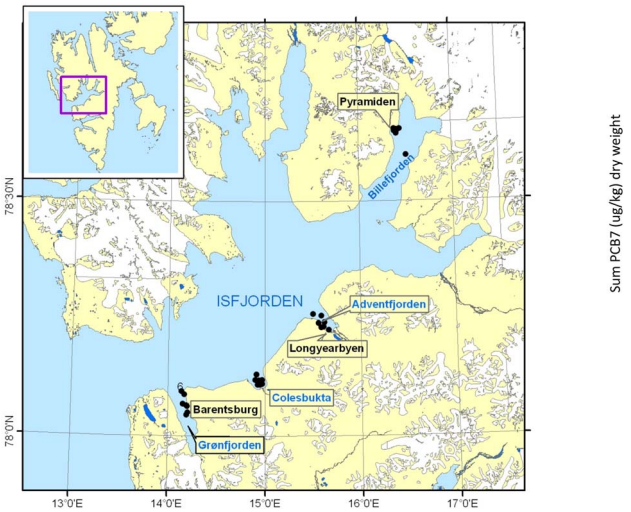

Fig. 7 Sampling stations for sediment in Isfjorden, Svalbard, figure reproduced with permission from ref. 197.

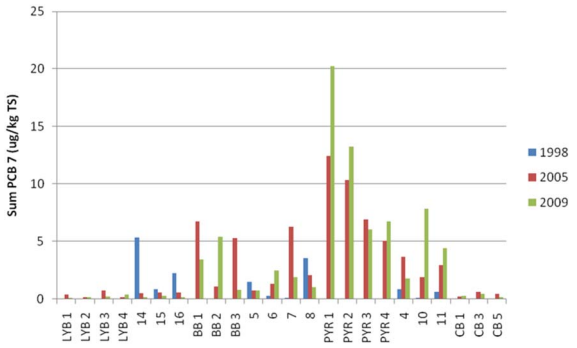

Fig. 8 Concentrations in three years of the sum of 7 congeners of polychlorinated biphenyls (PCBs) (Σ_7PCB = PCB 28, 52, 101, 118, 138, 153, 180) in sediment from Isfjorden, Svalbard. Data from 1998 from ref. 200 data from 2005 (ref. 201) and data from 2009.[202] LYB1–16 = stations in Adventfjorden, BB1–8 = stations in Grønfjorden, PYR1–11 = stations in Billefjorden, CB1–CB5 = stations in Coles Bay, figure reproduced with permission from ref. 177.

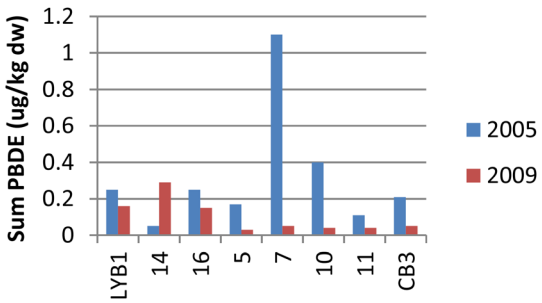

Fig. 9 Levels of polybrominated diphenyl ethers (PBDEs), shown as the sum of PBDE 47, 99, 100, 138, 153, 154, 183, 209, in sediment from Isfjorden, Svalbard (in μg per kg dry weight = dw). Data from 2005 and 2006 from Evenset *et al.*[203] LYB1, 14 and 16 = stations in Adventfjorden, 5 and 7 = stations in Grønfjorden, 10 and 11 = stations in Billefjorden, CB3 = station in Coles Bay, figure reproduced with permission form ref. 177.

incineration processes (fuel burning or biomass burning) or from natural deposits (petrogenic). Hence, pyrogenic PAHs are used to elucidate the anthropogenic impact of fuel burning on the Arctic environments. PAHs are ubiquitously distributed in the Arctic as shown in ongoing long-term monitoring activities.[132,182–188] The annual trend of PAHs in Arctic surface snow demonstrates seasonal variability with significant contributions from long-range atmospheric transport during the winter, while retene shows a local input from local coal burning in autumn and spring.[144] Increased re-emissions of PAHs deposited in Arctic cryosphere sinks are expected to occur and contribute to elevated secondary PAH emissions due to the cryosphere loss in the Arctic,[189] as also discussed by Muir *et al.*[83]

Muir *et al.*[83] also addressed new opportunities for agricultural production in the North in a warming Arctic.[190–193] Consequently, direct emission sources from local applications of pesticides may potentially move closer towards or even into the Arctic region.[194] Currently, levels of modern current-used pesticides (CUPs) found in Arctic environments are at low to medium level (ppb-range) in the cryosphere (ice, snow), soil, water, and biota and considered reflective of long-range transport from their use outside of the Arctic. However, especially low-trophic level organisms show surprisingly high levels indicating possible local sources close to application areas.[195]

Several examples exist of local emissions of POPs from settlements in the Arctic without a clear identification of specific infrastructure sources. Higher PCB levels near human settlements than at remote locations were shown for Greenland.[196] Several PCB congeners, including the low-chlorinated PCB-5, PCB-11 and PCB-52 were found to be co-synthesized during various pigment production processes. These congeners have recently been identified at elevated levels in Arctic snow, soil and even biota, with sources likely including domestic emissions from treated surfaces in housings and waste handling.[197] However, no knowledge is currently available on specific local sources of these unintentionally produced PCBs in the Arctic. In 2005 and 2009 regular monitoring of surface sediments (0–1 cm) was conducted on 24 stations close to active and abandoned settlements along the coast of Isfjorden on Spitsbergen where Longyearbyen, Barentsburg, Pyramiden and Coles Bay are located. Seven of these stations had also been sampled in 1998.[198] All samples were analysed for Σ_7PCB, hexachlorobenzene (HCB), DDTs, total hydrogen content (THC), PAHs and metals, in addition to grain size and TOC (Fig. 7–10).

In 1998 the lowest levels of Σ_7PCB were measured in sediment from Billefjorden (outside Pyramiden, 0.03–0.8 μg per kg dw, $n = 2$), with slightly higher levels in Grønfjorden (outside Barentsburg, 0.02–3.5 μg per kg dw, $n = 3$) and Adventfjorden (outside Longyearbyen, 0.8–4.3 μg per kg dw, $n = 3$ (Fig. 8). These differences at a relatively small geographical scale indicate variations in PCB emissions, likely related to local sources. For a more in-depth analysis, a larger number of stations were included in the 2005 and 2009 surveys. In 2005 and 2009 very low levels were measured in sediment from Adventfjorden (0.1–0.7 μg per kg dw, $n = 7$). Higher concentrations were measured

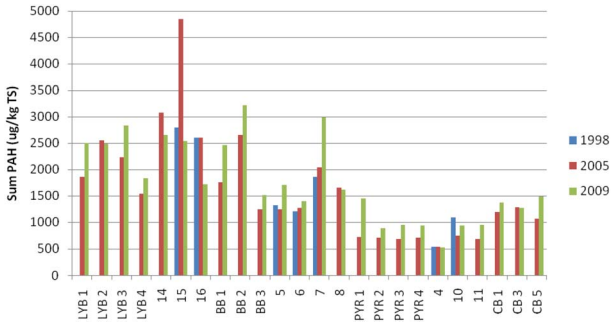

Fig. 10 Time trends for polycyclic aromatic hydrocarbons (PAHs) (sum 16 EPA PAHs, μg per kg dry weight) in sediment from different stations in Isfjorden, Svalbard. Data from 1998 from Cochrane *et al.*,[204] data from 2005 from Evenset *et al.*[203] LYB1 – st. 16 = stations in Adventfjorden, BB1 – st. 8 = stations in Grønfjorden, PYR1 – st. 11 = stations in Billefjorden, CB1–CB5 = stations in Coles Bay, figure reproduced with permission from ref. 177.

in sediment from Grønfjorden (0.7–6.25 μg per kg dw, $n = 7$) and Billefjorden (1.8–20.2 μg per kg dw, $n = 7$). The geometric mean concentrations of Σ_7PCB increased in Billefjorden from 1998 (2 μg kg^{-1}) to the two next measurements in 2005 (4 μg kg^{-1}) and 2009 (8 μg kg^{-1}) (Fig. 8). In contrast, there was a significant decrease in concentration of Σ_7PCB in Adventfjorden from 1998 to 2009 (from 6 to 1 μg kg^{-1}). High levels of PCBs were measured in soil samples from Pyramiden and it is likely that a flood event in 2005 transported contaminated soil to Billefjorden. In Coles Bay levels of Σ_7PCB were low in both 2005 and 2009 (0.2–0.6 μg per kg dw, $n = 3$). The HCB levels were generally low in the investigated fjords. However, slightly elevated levels (max 1 μg per kg dw) were measured at some stations in all areas, except Coles Bay. The highest DDT-levels were measured outside the two Russian settlements; Barentsburg (0.8–7.0 μg per kg dw) and Pyramiden (0.6–1.2 μg per kg dw). Sediment from Adventfjorden had the lowest DDT-levels in 2005 and 2009 (0.06–0.3 μg per kg dw), but higher levels (0.4–2.3 μg per kg dw) were measured in 1998. Generally, DDT that has been transported over long distances will be present as DDE, a stable metabolite of DDT. However, *p,p'*-DDD and *p,p'*-DDT were the dominant DDT-compounds in the sediment samples collected outside the Russian settlements on Svalbard, suggesting past use of DDT in these areas. In Coles Bay, DDT-levels were low to moderate both in 2005 and 2009 (0.2–0.6 μg per kg dw).

Samples collected in 2006 were analysed for tributyltin (TBT). TBT was only detected at two stations: one in Adventfjorden (1.8 μg per kg dw) and one in Grønfjorden (3.6 μg per kg dw). Hence, these results indicate minor contribution from local ships and boat traffic. Organo-metal compounds such as organotin compounds are among the most used chemicals for antifoulant coating of ships.

Levels of PBDEs (PBDE 47, 99, 100, 138, 153, 154, 183, 209) were measured in a few sediment samples in 2005 and 2009. In 2005 the highest levels were measured in Grønfjorden and Billefjorden, while highest levels were measured in Adventfjorden in 2009 (Fig. 9). The steep increase of PBDE in the vicinity of the settlements indicates local contamination sources.

In general, levels of PAHs were highest in Adventfjorden, followed by Grønfjorden, Billefjorden/Coles Bay (Fig. 10). Coal is a likely source for some of these PAHs in Adventfjorden and Barentsburg, but diesel oil and combustion products are other possible local sources. Analysis of organophosphate esters (OPEs) in sediments from Adventfjorden, Grønfjorden and Kongsfjorden showed interesting results. This compound group was detected in both fjords, but concentrations were significantly higher in sediment from Grønfjorden (ΣOPE (sum of 14 compounds) from 6.46–74.0 μg per kg dw) than in sediment from Adventfjorden (0.04–10.5 μg per kg dw). OPEs were not found in sediment from the reference station in Kongsfjorden. This indicates that local sources influence the environmental status of Adventfjorden and Grønfjorden. However, the reference station had lower levels of organic carbon and coarser sediment (lower share of fine-grained materials) than sediments from Adventfjorden and Grønfjorden, and these factors could have contributed to lower levels of OPEs at Kongsfjorden.

However, elevated levels, together with differences in contaminant patterns (relative proportions of different compounds), provide a strong indication that local sources are important, especially in Grønfjorden.[205–207] Some samples of spider crabs (*Hyas araneus*) and fish including shorthorn sculpin (*Myoxocephalus scorpius*) and Atlantic cod (*Gadus morhua*) were also collected in the fjords. The levels of OPEs in great spider crabs and fish were low, both in Adventfjorden (max ΣOPE 0.51 ng per g ww) and Grønfjorden (max ΣOPE 1.01 ng per g ww). In Grønfjorden triphenyl phosphate (TPP) was detected in shorthorn sculpins caught close to the landfills (Biota G1).

The highest detection frequency for OPEs occurred in the samples collected close to the sewage outlet in Grønfjorden (Biota G2). In this area tris(2-chloroethyl) phosphate (TCEP), tris(2-chloroisopropyl) phosphate (TCIPP), TPP and 2-ethylhexyl diphenyl phosphate (EHDP) were detected in shorthorn sculpins, and TCEP and TCIPP in one spider crab sample. Thus, the high levels measured in sediment from Grønfjorden were not reflected in elevated levels in organisms, except for biota samples collected close to potential point sources such as the sewage outlet. The observation of elevated levels in organisms collected near the sewage outlet in Barentsburg, compared to other areas, supports the indication from the sediment samples that this may be a local source for OPEs in Grønfjorden. However, biological processes are likely to change the OPE occurrence in biota compared to sediments and OPEs have been shown to be degradable.[208]

A recent study indicated local emission differences in PFAS levels in Svalbard.[23] A first surface seawater screening along the Isfjord coast in Spitsbergen revealed an interesting feature. The PFAS pattern in surface seawater samples influenced by the Russian mining town Barentsburg (Grønfjorden) in Svalbard was dominated by perfluorobutanoic acid (PFBA), whereas the surface seawater samples from Norwegian settlement mining town Longyearbyen (Adventfjorden) were predominantly contaminated with perfluorooctanoic acid (PFOA) (Fig. 11).[23] These different patterns likely originate from technical mixtures applied for specific applications at the respective locations.

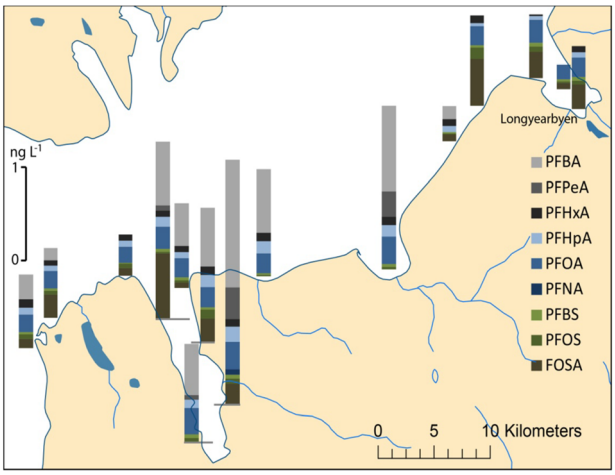

Fig. 11 Screening of per- and polyfluoroalkyl acids (PFAAs) in seawater along the Icefjord (Isfjord) coastal region in Spitsbergen (Svalbard, Arctic). The samples were collected in June 2016, figure reproduced with permission from ref. 23. PFBA = perfluorobutanoic acid; PFPeA = perfluoropentanoic acid; PFHxA = perfluorohexanoic acid; PFHpA = perfluoroheptanoic acid; PFOA = perfluorooctanoic acid; PFNA = perfluorononanoic acid; PFBS = perfluorobutane sulfonate; PFOS = perfluorooctane sulfonate; FOSA: perfluorooctane sulfonamide.

Another study was able to identify the terrestrial extent of PFAS contamination from Longyearbyen using surface snow sampling. For TFMS, PFHxS, PFOS, HFPO-DA (GenX) and C_2–C_{11} PFCAs, the concentration of surface snow in Longyearbyen was compared with other sites around Svalbard: a nearby high elevation ice cap (Foxfonna), high elevation reference locations on several other glaciers and a remote high elevation ice core from Lomonosovfonna (Fig. 12).

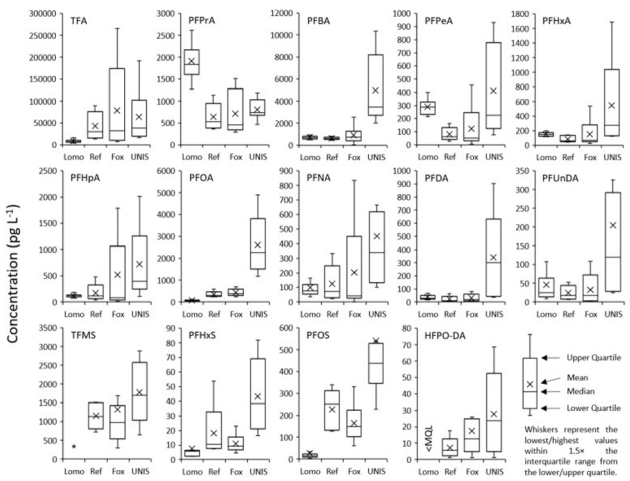

Fig. 12 Boxplots showing the concentrations (pg L^{-1}) of trifluoromethane sulfonate (TFMS), perfluorohexane sulfonate (PFHxS), perfluorooctane sulfonate (PFOS), hexafluoropropylene oxide diamer acid (HFPO-DA; GenX) and C_2–C_{11} perfluorocarboxylic acids (PFCAs) from the Svalbard cryosphere: in an ice core from Lomonosovfonna (Lomo, 2006–2019), and surface snow from the settlement of Longyearbyen (UNIS), Foxfonna (Fox) – a nearby high elevation glacier (Fox) and remote reference sites on Spitsbergen glaciers.[176] Reproduced under the Creative Commons Attribution 4.0 International licence (CC BY 4.0).

When comparing snow from Longyearbyen with snow from the reference sites, no significant difference was found for TFA, perfluoropropanoic acid (PFPrA) and TFMS. This suggested a common source, such as the atmospheric degradation of hydrofluorocarbons (in the case of TFA and PFPrA). However, the concentrations of PFHxS, PFOS and HFPO-DA (GenX) and C_4–C_{11} PFCAs were 2.3–22 times higher in Longyearbyen indicating a local source. This could be from the firefighting training site at the local airport, nearby historic landfills, and/or diffuse sources in the local settlement.[20] Bjornsdotter *et al.*[175] also found that surface snow in Longyearbyen had elevated detection frequencies of perfluoroethane sulfonate (PFEtS) and perfluorobutane sulfonate (PFBS) compared to these same remote reference sites.[167,168,209] Together these studies showed that PFAS concentrations in surface snow at the nearby icecap of Foxfonna were indistinguishable from other remote glacier sites, indicating that PFAS contamination from Longyearbyen was contained within at least 16 km and below 800 m.a.s.l. (for the terrestrial environment). However, contamination from Longyearbyen's landfill and airport is more mobile in the marine environment (Fig. 11).[23]

In the Canadian Arctic, there is a long record of POP surveys near the hamlet of Resolute Bay (also known as Qausuittuq) on Cornwallis Island (73–75 °N, 92–95 °W).[210–212] Spatial analysis of water, sediment, and biota from lakes on Cornwallis Island indicate local sources of PFAS and OPE flame retardants. Stock *et al.*[213] reported elevated concentrations of PFHxS, PFOS, perfluoroheptanoic acid (PFHpA) and PFOA in water and sediment in Resolute Lake that were up to 60-fold higher than corresponding levels in Amituk and Char Lakes in 2003-04 [213] (Fig. 13). The local contamination that was observed in these lakes was attributed to historical use of AFFFs at the Resolute airport as well as raw wastewater discharge from North Base from 1949 until the 1990s[213] (Fig. 12). Lescord *et al.*[214] performed PFAS analysis in biota from lakes on Cornwallis Island. In addition to confirming high PFAS concentrations in water and sediment from Meretta and Resolute Lakes, much higher PFOS concentrations were also measured in Arctic char (*Salvelinus alpinus*) of these lakes, *i.e.* 287 ± 273 ng g^{-1} and 445 ± 545 ng g^{-1} for Meretta and Resolute Lake, respectively. In contrast, in char from other lakes located on Cornwallis Island, PFOS concentrations ranged from 5.3 to 14 ng g^{-1}. Similarly, PFOS in juvenile char whole body homogenate was reported to contain 181 ± 50 ng g^{-1} and 224 ± 491 ng g^{-1} in Meretta and Resolute Lakes, whereas PFOS concentrations ranged from 0.001 to 1.5 ng g^{-1} in the same species from the other four lakes. Even in more recent water sampling from 2014, PFOS, PFOA and other PFAS continued to be higher in Resolute and Meretta Lakes compared to other lakes, although levels are declining (Fig. 13). Resolute Airport served as an air force base in the 1950s and as the main airport in the Canadian High Arctic subsequently, with a staff of up to 150 people in the 1950s.[210] Wastewater from cooking, washing, and sewage was held in holding tanks and released without treatment into the upper catchment of Meretta Lake every day between 1949 and 1998, entering the lake through a series of small watercourses.[211] The eutrophication of the lake was first described in the 1970s.[210] Water from Meretta

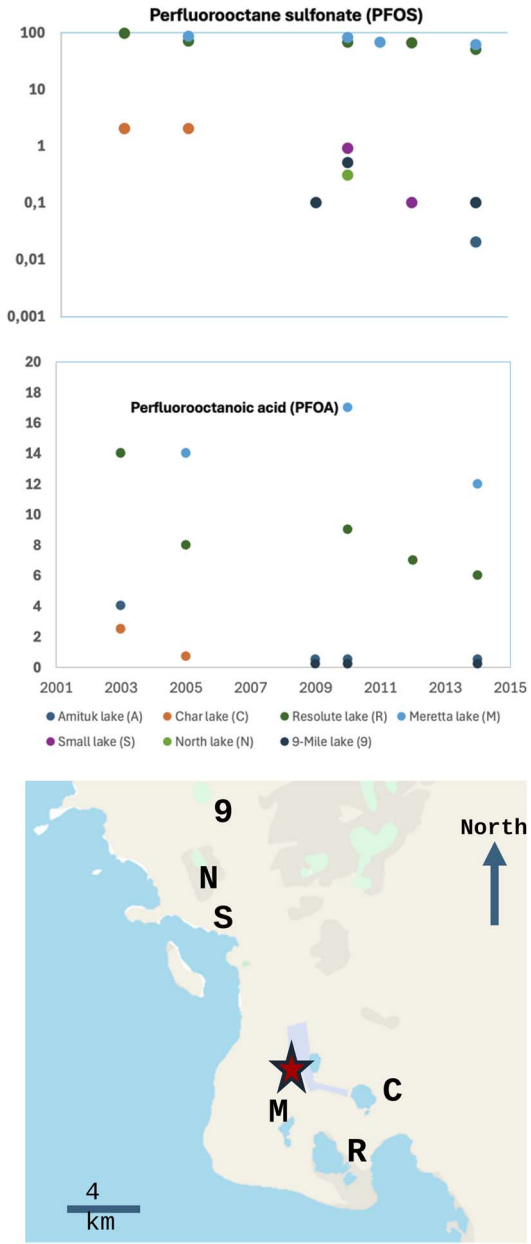

Fig. 13 Perfluorooctane sulfonate (PFOS) and perfluorooctanoic acid (PFOA) in water sampled in lakes near Resolute Bay with higher concentrations [ng L^{-1}] due to local emission sources from the airport (red star). Letters indicate location of lakes: Meretta (M), Resolute (R), Char (C), North (N), 9-Mile (9) and Small (S). Not shown on map is Amituk Lake on the west coast of Cornwallis Island (75.05 N, −93.81 W). Data from MacInnis *et al.*[215] and De Silva unpublished data.

Lake flows into Resolute Lake and thus both lakes have been contaminated. The actual timing of use and emissions of PFAS at the airport is unknown but could have been from routine fire training as well as accidental releases. Cabrerizo *et al.*[212] also found elevated PFAS levels in soils collected near Meretta Lake, with PFOS (7.60 ng per g dry weight) accounting for 90% of total perfluoroalkyl sulfonates ΣPFSAs. PFOS concentrations in Meretta catchment soils were more than 10-fold higher than in soils from the catchment of Resolute Lake.[215]

OPEs are widely used in consumer products, building materials *etc.* and might thus be emitted from the use and disposal of these products in the Arctic. OPEs were found in elevated concentrations in Resolute and Char Lake compared to the other lakes in the vicinity.[216] In water, total OPE (Σ_{14}OPE) were 101 ± 3 ng L^{-1} and 49 ± 2 ng L^{-1} from Meretta and Resolute Lakes sampled in 2017[216] with a predominance of tri-*n*-butyl phosphate (TNBP). This results corresponds with earlier reported OPE concentration in land-based air samples in Resolute from 2012 which had median TNBP corresponding to 416 pg m^{-3} (<LOD to 2052 pg m^{-3}, $n = 10$), whereas TNBP was below detection limits in ship-based air sampling 20 km south of Resolute in 2013.[217] A recent report proposed that TNBP was related to the use of aircraft hydraulic fluid at the local airport. Bartley *et al.*[218] studied PCBs, PAHs, and PFAS in marine sediment cores from Frobisher Bay, Nunavut (Canada). One of the core locations was in Koojesse Inlet, with the city of Iqaluit at its northwest corner. Iqaluit is the most populous city in Nunavut (population *ca.* 8500) and has experienced a rapid population growth and development in recent decades. Historically, it hosted a military air base (1942–1963) and was part of the Distant Early Warning (DEW) Line. The sedimentary record indicated elevated PCB levels due to Aroclor 1260 usage in the 1950s. Similarly, the profile of PAHs was attributed to fossil fuel combustion in and around Iqaluit. Significant PFAS contamination is reported from the 1980s onwards. Local activity also influenced the PFAS profile in sediment cores with compounds specific to AFFFs used in fire-fighting. In contrast, contaminants in the sediment cores from outer Frobisher Bay demonstrated a different pattern. For example, less chlorinated PCBs were predominant, and diagnostic ratio analysis of PAHs revealed petrogenic signatures consistent with long-range transport. Taken together, the results are likely to integrate over different local sources associated with the urban settlement of Iqaluit as well as the military infrastructure. In addition, chlorinated paraffins (CPs) were detected in samples collected near Iqaluit, indicating local sources in an urban settlement. CPs are a mixture of technical chemicals with a very high production volume. It is estimated that a total of 30 million metric tons of CPs have been produced globally until today.[219] Short-chain CPs (SCCPs: C_{10}–C_{13}) and medium-chain CPs (MCCPs: C_{14}–C_{17}) were found in abiotic and biotic Arctic samples.[220–222] In the study from Iqaluit,[223] tissues from Arctic char and ninespine stickleback (*Pungitius pungitius*) were analysed, along with water and sediments from a creek running past the airport and a small stream draining a former military landfill and dumpsite. SCCPs in the water from Airport Creek had a concentration of 13.5 ± 12.3 ng L^{-1} (range 3.7 and 37.2 ng L^{-1}) while MCCPs were higher than SCCPs, averaging 88.6 ± 90.4 ng L^{-1}. The highest levels of MCCPs were found in water emerging from below an old military dumpsite indicating a long-term and continuous source. Elevated levels of SCCPs and MCCPs were found in ninespine sticklebacks collected from the creek. However anadromous Arctic char ($n = 36$) collected at the outflow of the Sylvia Grinnell River (Iqaluit) had relatively low CP concentrations (SCCPs 7.8 ± 17.0 ng per g lipid weight (lw); MCCPs 6.8 ± 11.1 ng per g lw) compared to POPs in landlocked char in the Canadian Arctic. However, there are only few data for

SCCP/MCCPs in fish in the Canadian Arctic for comparison. Char collected at a site about 10 km west of Iqaluit had lower CP concentrations (1.6 ± 2.8 ng per g lw and 3.7 ± 1.8 ng per g lw for SCCPs and MCCPs, respectively). However, the sample size was small ($n = 3$). Significant concentrations of CPs in sediment and water from some sites indicate ongoing local sources of contamination and a lagoon associated with municipal waste disposal had high levels of SCCP/MCCPs. A general overview on SCCP and MCCP levels in Canadian fish is presented in a previous publication.[224] SCCPs are also reported as environmentally mobile and long-range transported into the Arctic earlier.[225] They are classified as POPs, *i.e.* their long-range transport to the Arctic has been confirmed in the POP assessment under the Stockholm Convention. Domestic use of various technical and electronic products, local waste handling and industrial applications have also been described as potential local contamination sources of hexachlorobutadiene (HCBD).[24,66,226–229] HCBD is a byproduct of various technical products and was found in elevated levels in Arctic environments. However, given the volatility of the compound, long-range atmospheric transport is a likely source of HCBD in the Arctic, which is supported by the relatively high levels found in Arctic air.

Plastics, including micro- and nanoplastics, have emerged as significant local pollutants in the Arctic environment in recent years, related to domestic, industrial, and military infrastructures.[230–233] Local sources of plastic pollution include waste handling, aquaculture operations, transportation activities, and consumer product usage. In remote Arctic communities, landfills and waste incineration can release plastic particles into the surrounding environment.[234–236] These landfills, frequently located on permafrost, are vulnerable to thawing, which can mobilize previously trapped contaminants, including plastic debris and associated additives.[237–239] Incineration, especially when uncontrolled, contributes to the release of microplastics and toxic compounds such as dioxins and furans.[240–243] Aquaculture has been identified as a growing source of plastic pollution, with infrastructure components like nets, ropes, and feeding systems made from synthetic polymers that degrade into micro- and nanoplastics. Studies have found elevated levels of polypropylene (PP), polyethylene (PE), and polyvinylchloride (PVC) in mussels collected near aquaculture sites in Northern Norway, indicating direct exposure and accumulation in marine organisms.[237,244,245] These plastics can also act as vectors for POPs, enhancing their mobility and bioavailability in Arctic ecosystems.[246,247] Transportation activities, particularly the widespread use of snowmobiles and fossil fuel-driven vehicles, contribute to plastic pollution through tire wear, fuel combustion, and maintenance operations.[248,249] Consumer products such as cosmetics, detergents, and packaging materials also contribute to the release of microplastics through wastewater discharges. Inadequate sewage treatment in Arctic settlements exacerbates this issue, allowing plastic particles and associated chemicals to enter freshwater and marine environments.[250] Seasonal variations, such as snowmelt and increased UV radiation during the midnight sun, influence the transport and transformation of these pollutants. Atmospheric deposition has also been recognized as a pathway for microplastic transport into the Arctic, with fibers and fragments detected in snow and ice samples.[249] This suggests that long-range transport complements local emissions, creating a complex pollution profile. Climate change further amplifies the problem by altering the cryosphere, increasing the mobility of contaminants, and potentially enhancing degradation rates of plastic materials into smaller particles. Monitoring data from locations such as Longyearbyen and Ny-Ålesund have confirmed the presence of microplastics in soil, water, and biota, with concentrations varying based on proximity to pollution sources.[248,251–254] The interaction between microplastics and Arctic fauna raises concerns about ecological and human health, as ingestion by low-trophic organisms can lead to biomagnification through the food web.[255,256] Despite the growing body of evidence, many micro- and nanoplastics remain undetected due to limitations in analytical techniques and research biases toward known pollutants. The persistence of microplastics in cold Arctic conditions, combined with limited degradation and removal mechanisms, underscores the need for improved waste management, infrastructure design, and pollution mitigation strategies. International cooperation, harmonized policies, and community engagement are critical to addressing the multifaceted challenges posed by plastic pollution in the Arctic.[257–259] Urban settlements have also been associated with local emissions of plastic litter, besides sources related to fisheries.[235,236] Transport also occurs by sea-ice from Russian rivers.[260,261]

Airports as local pollution sources in the Arctic

Soil, drainage water and snow samples in the vicinity of the local airports in Longyearbyen and Ny-Ålesund in Spitsbergen showed surprisingly high PFAS concentrations with PFOS as the major constituent.[23,168] The commercial airports in Norway are owned and operated by Avinor which is a wholly owned state limited company under the Norwegian Ministry of Transport and Communications. The military airbases in Norway are owned by the Norwegian Defense Estates Agency. Avinor phased out the use of PFOS-based AFFF in 2001,[262] and such foam was phased-out from all of Norway in 2007.[263] Firefighting foams containing fluorotelomer-based surfactants (such as 6:2 fluorotelomer sulfonate (6:2 FTS) and/or related products) were used as replacements,[264] however, the use of all PFAS-based foams was discontinued in 2012 and between 2012–2015 at commercial and military airports, respectively.[264] The Norwegian Environmental Agency has demanded an investigation of the extent of PFAS pollution at Norwegian airports. In the following, some illustrative case-study sites from the Arctic Norwegian mainland are discussed regarding the type of pollution and potential consequences.

PFAS-containing AFFFs have been used for firefighting since the 1960s.[265] Extensive use of AFFFs at firefighting training areas, military sites, and airports has resulted in significant PFAS contamination at these sites and nearby environments.[266–269] The long distances in the Arctic in combination with the lack of road and railroad connections make airports critical infrastructure for transport and local accessibility in this region. There are approximately 1300 airports and heliports in the Arctic, however most of them are small both in

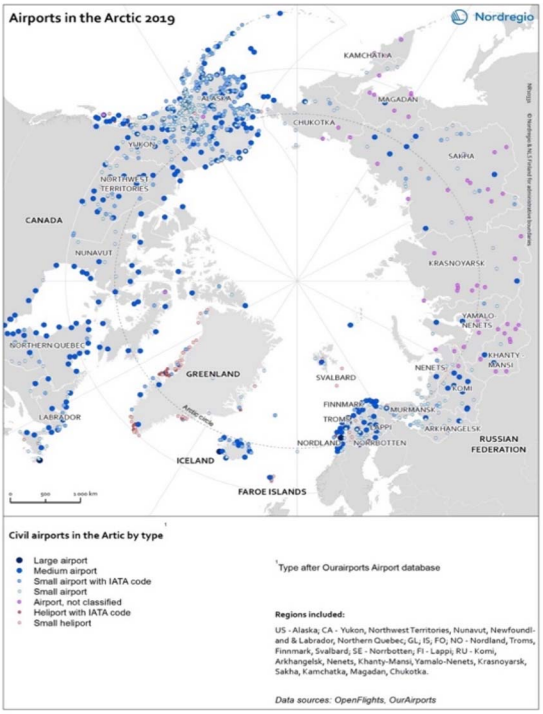

Fig. 14 Civil airports in the Arctic. This figure is reproduced from Nordregio (for more details see https://nordregio.org/maps/airports-in-the-arctic-2019/, downloaded 05.08.2025).

physical extent as well as number of flights (Fig. 14). In 2019, there were seven large and *ca.* 260 medium sized airports in the Arctic region (registered by the International Air Transport Association (IATA)). In addition, there were 265 small airports with an IATA code, and 500 small airports without an IATA code. To service incoming flights, several chemicals are regularly applied at airports in large volumes, including petroleum products, de-icing chemicals and firefighting products.[270] Thus, airports can be a significant local source of pollution, resulting from fuel spills, de-icing routines, and firefighting exercises using AFFFs that contain PFAS.

At Bodø Airport (northern Norway) PFAS were used in firefighting operations and, hence, were found in stormwater and soil leachate which drain into the surrounding marine environment. Both PFOS- and fluorotelomer-based foams were used at the airport. PFOS was the dominating PFAS in stormwater at the airport, while 6:2 FTS dominated in leachate where firefighting training had been performed during the last years before the phase-out of all PFAS-based foams. Sediments and biota (fish and invertebrates) were investigated at the military site at Bodø Airport.[264] Higher concentrations of PFOS were reported in biota near the Air Station compared to a reference site on the other side of the fjord (*e.g.*, Atlantic cod liver concentrations of 6.5 *versus* 1.6 μg kg^{-1}). Interestingly, almost no accumulation of 6:2 FTS was observed in fish. This is in line with the scientific literature reporting that fish biotransform 6:2 FTS into PFAS with shorter alkyl-chains.[271–273] However, in invertebrates, a significant accumulation of 6:2 FTS was observed, possibly reflecting a lower biotransformation potential in invertebrates.[264] Although the concentrations in fish were relatively low (the Environmental Quality Standard, = EQS, of the European Water Framework Directive is 9.1 μg kg^{-1} for PFOS in fish), the higher concentrations in biota near the Air Station compared to the reference station show that PFAS pollution from airports may significantly affect biota concentrations in the local environment. Recently, it has been reported that consumption of even limited amounts of fish with concentrations well below the EQS will lead to exceedance of recent tolerable intake or reference dose values in the EU and the USA.[274] C_4–C_8 perfluoroalkylic acids (PFAAs) were also taken up by benthic organisms in the local coastal marine food web close to Longyearbyen airport. The source of this contamination was suspected to be due to the continuous release of drainage water from the local firefighting training site. However, a direct correlation between the local contamination and top predators (glaucous gulls, *Larus hyperboreus*) could not be established.[275] In addition, in the same region, Warner *et al.*[276] identified three Svalbard settlements (Longyearbyen, Barentsburg, and Pyramiden) as local sources of contamination of eggs of snow buntings (*Plectrophenax nivalis*) with PCBs, OCPs and PFAS.[277]

Marine organisms are important food sources for local communities and wildlife in the Arctic. Associations between the consumption of fish from waters affected by PFAS pollution from Harstad/Narvik Airport in Northern Norway and serum PFAS concentrations in humans have been reported.[278] PFOS concentrations in the muscle of brown trout (*Salmo trutta*) from one of the lakes near the airport were in the range of 13.6–24 μg kg^{-1}. A significant, positive association was seen between consumption of fish from the impacted waters and human serum PFAS concentrations (geometric mean of 28 ng mL^{-1} in the high consumption group *versus* 10 ng mL^{-1} in non-consumers). This shows that local sources of PFAS, from use at airports in the North, may lead to a significantly increased exposure of local populations.

Svalbard airport (Longyearbyen) has one active and one abandoned firefighting training site. PFAS concentrations in runoff-/leachate water were 365 ng L^{-1} and 57 ng L^{-1} for Σ_{14}-PFAS at the active and the abandoned site, respectively and determined as a major local PFAS sources for biota in the surrounding environment.[168] At the research station of Ny-Ålesund, high PFAS concentrations were reported in runoff water (113–119 ng L^{-1} for Σ_{14}-PFAS) and soil (211–800 μg per kg dw for Σ_{14}-PFAS) close to the firefighting training site of the small airport serving the research station.[20] Local sources of PFAS on

Table 2 PFAS (Sum C_4–C_{12} perfluoroalkylic acids (PFAAs)) in soil samples (in ng per g dry weight (dw)) from airport firefighting training sites. Table modified from ref. 20

Location	Levels [ng per g dw]	References
Oslo Airport, Gardemoen, Norway	2600	279
Ellsworth Air Force Base, USA	2400	280
Svalbard, Ny-Ålesund Airport	1100	20
Svalbard, Longyearbyen Airport	1600	23

Svalbard, including Svalbard airport, have been determined to pose a potential risk for top predators and humans due to contamination of surface water and local food.[23]

The PFAS concentrations in Svalbard were approximately 50% of the concentrations reported for corresponding soil samples near Norway's largest airport in Gardermoen, Oslo (Table 2). These results confirm that the environmental PFAS contamination associated with AFFFs from airport firefighting training sites is not directly correlated with the size of the airport but influenced by other factors, for example the retention technologies used to avoid environmental contamination.

As demonstrated for PFAS, Arctic airports can be important local pollution sources. It has been estimated that as many as 152 to 420 different airport locations in Canada have PFAS-impacted surface waters[268] (Fig. 14). Hence, airports and airbases in the Arctic may be significant sources of PFAS to the local environments, with the risk of food web accumulation and subsequent exposure of humans. This is of concern, especially in light of current tolerable intake estimates of only 4.4 ng per week per kg body weight for the sum of PFOS, PFOA, PFNA and PFHxS, as published by the European Food Safety Authority.[281,282]

Jetfuel 1A used in aviation is generally considered to be easily degradable.[283] Although low temperatures and freezing conditions can strongly delay these natural processes,[284] hydrocarbons will be degraded over time, as long as the soil thaws during summertime. A major oil spill that occurred at Ny-Ålesund on Svalbard during the 1980s is at present no longer detectable, based on soil and groundwater measurements.[285,286] The main environmental threat associated with oil contamination is its spreading to the freshwater and the marine environment where animals will be affected.[287] The de-icing of airplanes and runways is common practice in the cold climate of the Arctic. Aircraft de-icing/anti icing fluids (ADFs) are principally based on propylene glycol (PG) while runway de-icing chemicals are based on potassium acetate, formic acid or urea. In general, these are all easily degradable compounds as long as the environmental conditions are favourable.[288] The main concern associated with de-icers is eutrophication resulting in anoxic conditions in both surface and groundwater.[289] However, to improve the performance of de-icers and limit the impact on the materials used in airplanes, additives are added that have been shown to be persistent in the environment.[284] Elevated levels of TNBP in Arctic air were found at some locations in the Canadian Arctic.[241] TNBP was associated with the use in aircraft hydraulic fluids.[242] Furthermore, an airport at Iqaluit was considered as one of the potential sources of SCCPs in the local environment.[191]

Petroleum and de-icing related contamination is relatively easy to reduce by proper management to reduce spills. If spills occur, natural degradation processes can reduce these contaminants over time. However, due to their persistence, local PFAS sources are a serious concern for the Arctic environment, where airports play a major role. However, there is a great variation between countries in available information about PFAS pollution from airports. While relatively much is known about PFAS pollution at Norwegian airports, information about potential PFAS pollution at airports in *e.g.* Greenland and Iceland is limited. However, it has been reported that livers of caribou (*Rangifer tarandus groenlandicus*) from Akia-Maniitoq and Kangerlussuaq, both located near the largest airport in Greenland, along with a former military base, have high concentrations of PFAS (Σ_{12}PFAS of 101 ng g^{-1} and 45 ng g^{-1}, respectively).[290] Furthermore, other chemicals might be used in the aircrafts themselves, for example as lubricants, and emitted during use and maintenance. A recent survey on PFAS contaminated surface waters in Iceland revealed also the domestic airport in Reykjavik as a PFAS source.[291]

The AFFF formulations used at airports have changed over time at some locations.[292,293] Other PFAS than PFOS, including fluorotelomer-related PFAS, have been released into the environment in significant concentrations. These replacement compounds might have different physical–chemical properties and will be important to consider for future sampling campaigns and estimations of human exposure.

In a screening study conducted in Adventfjorden, Grønfjorden and Kongsjorden (Spitsbergen, Svalbard) in 2015,[206,294] several PFAS were detected in surface sediment collected close to potential local sources (Fig. 15). Besides the airport in Longyearbyen, other potential local sources included landfills and sewage outlets were identified. The PFAS that were detected included 6:2 FTS, perfluorodecanoic acid (PFDcA), PFHpA, perfluorohexanoic acid (PFHxA), PFNA, PFOA, PFOS, perfluorotetradecanoic acid (PFTeA) and perfluoroundecanoic acid (PFUnA). Of these, PFDcA, PFNA, PFOA and PFUnA were detected at all sites, including the reference site in Kongsfjorden (Fig. 15). 6:2 FTS was only detected at the reference site. No significant differences were found in ΣPFAS between the three sampling areas, although the levels were generally higher in Grønfjorden than in Adventfjorden and lowest at the reference station. Different PFAS-compounds dominated in the three areas, *i.e.* PFOS in Adventfjorden, PFHpA in Grønfjord and 6:2

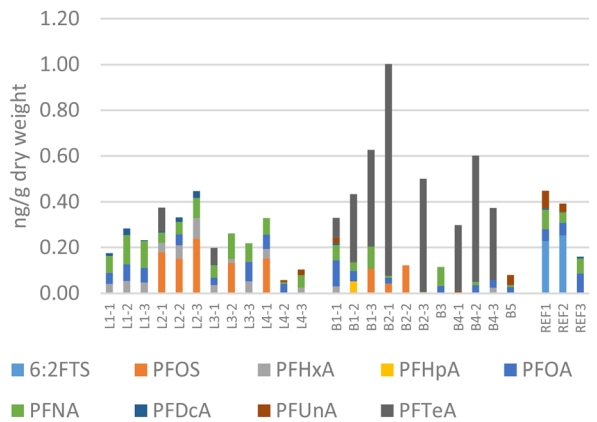

Fig. 15 PFAS concentration [ng per g dry weight] in sediment-samples from Adventfjorden (L), Grønfjorden (B), and Kongsfjorden, September 2015, figure reproduced with permission from ref. 23. 6:2 FTS = 5:2 fluorotelomer sulfone; PFOS = perfluorooctane sulfonate; PFHxA = perfluorohexanoic acid; PFHpA = perfluoroheptanoic acid; PFOA = perfluorooctanoic acid; PFNA = perfluorononanoic acid; PFDcA = perfluorodecanoic acid; PFUnA = perfluoroundecanoic acid, PFTeA = perfluorotetradecanoic acid.

FTS at the reference station. High concentrations of PFOS have previously been reported in soil-samples from Svalbard airport.[23] However, the levels of PFAS were not higher in sediment collected outside the airport than in sediment from other stations in Adventfjorden.[23]

Besides firefighting sites at airports, other AFFF uses might have caused other locally pollution at other sites. For example, in a waste dump in the former mining town Svea, high levels of PFAS were measured in soil and run-off water.[295] Elevated levels were also found in an area outside the settlement. Both areas had been used for firefighting training activities. PFAS-containing AFFF was also used for firefighting in mine 7 on Spitsbergen, and high concentrations of PFOS are still found in soil/dust inside the mine (*pers. Communication*, *Dr Gijs Breetveld*, *NGI&UNIS*).

Solid waste handling as a local pollution source

Waste management in remote Arctic communities is influenced by the low population density, logistic challenges with large distances between settlements and harsh climate conditions, including permafrost. Traditionally, the amount of waste produced in the Arctic has been low, but societal development and economic growth have led to an increase in waste generation. Almost all goods and materials are transported to the Arctic from southern regions. When these goods and materials have reached the end of their lifetime, they typically stay in the Arctic. This means that accumulation of waste, including legacy waste, is a problem that is typically left to be solved locally. Landfilling and incineration are the most common waste treatment practices in the Arctic, while reuse, composting and recycling rates are low.[96] Landfilling varies from unorganized open dumps without any protection of soil or groundwater, to controlled landfills for instance using cover material and dividing different waste types into designated zones, to sanitary landfills with leachate and gas collection.[296] The latter type of landfill is not common in the Arctic. Incineration practices range from sporadic open burning or contained open burning without any control of the incineration process or environmental emissions, to waste-to-energy plants. Waste-to-energy plants have flue gas treatment, which reduces the level of contaminants emitted through the flue gas and utilizes the heat to produce energy for the local community. The use of waste-to-energy plants is common in Greenland, where district heating nets are installed in the towns.[297] Both landfilling and waste burning can be local sources of contamination.

Atmospheric landfill emissions are usually gas, including heat, and leachate. Landfill gas is mainly CH_4 and CO_2 produced by anaerobic digestion of organic waste, an exothermic process, which heats the waste. Furthermore a variety of VOCs and other organic pollutants are reported as landfill emissions not only in the Arctic.[298,299] The amount of landfill gas produced depends on the amount and the composition of the organic waste in the landfill but has not been quantified. Landfills in the Arctic receive a mixture of waste types, including municipal, construction, hazardous and industrial waste, and in some areas also human waste.[296] This mix of waste results in leachate containing dissolved organic matter, inorganic macro components, and CEACs.[300] The amount of leachate is dependent on the net infiltration, the top cover of the landfill and the surface area utilized for the landfill. Additionally, there is drainage water from landfills during snowmelt. The geological location of the landfill will influence the emission paths. Landfills can be placed in natural or excavated trenches, directly in soil (with or without permafrost) or on cliffs close to the coast. The placement of landfills or dumps in the Arctic has mainly been in easily accessible areas, rather than areas which are appropriate for landfilling and many dumps are close to residential areas.

Only a few studies have been conducted on the impacts of landfills in terms of Arctic pollution. Permafrost has been an effective barrier for leachate emissions from Arctic landfills. However, the heat produced in the degradation of organic waste as well as salt release from waste can thaw the permafrost and form talik zones under landfills, activate thermokarst and thermal erosion. Permafrost degradation under landfills is still not investigated well.[301] However, indications exist of thawing permafrost in a warming Arctic, with the potential consequence of contaminant releases from waste storage in permafrost.[302] Due to the cold winters, the degradation processes in waste are expected to be slowed down in Arctic landfills compared to temperate regions. Waste temperatures measured in May 2004 and 2007 at the landfill in Anchorage, Alaska showed low temperatures, even below 0 °C, in the top layers of 0–2 year old waste, which increased to above freezing temperatures after 3 years.[303] Another part of the landfill with 1–5 year old waste had higher temperatures, however, the waste temperatures at the landfill in Anchorage were lower than in more temperate climates.[303] The degradation of organic waste in a landfill proceeds in several phases (acidic phase, initial methanogenic phase and stable methanogenic phase), and under temperate conditions these stages take at least 30 years before the landfill can be considered closed and inert.[300] Arctic landfills can therefore be expected to be a source of local contamination for a much longer time.

In general, the formation of leachate is low under Arctic conditions due to reduced infiltration as the ambient temperature is rather low throughout the year. A recent study showed a stable chemical oxygen demand (COD), reflecting the level of organics in the waste, in column tests with waste during winter and an increase in COD after thawing period in the spring. Lund and Young[304] conducted long-term Li^+ tracer experiments at three different sites with varying soil conditions in the Canadian Arctic. The study indicated that during the snowmelt period in the spring, significant amounts of contaminants could be rapidly removed, even without being dissolved, as the snowmelt also means a significant transport of particles. There is a concern for the release of contaminants from coastal waste disposal sites since they can accumulate in the marine food web and lead to exposure of local populations. Leachate samples taken from three Greenlandic waste disposal sites showed release of both organic and inorganic contaminants, which were discharged directly to the recipients.[305,306] The leachate measurements showed concentrations of PCBs between 0.003 and 0.202 $\mu g\ L^{-1}$, 0.038 and 1.9 $\mu g\ L^{-1}$ for PAHs and International Toxicity Equivalents (I-TEQ) between 0.001 and 0.111 $ng\ L^{-1}$ for PCDD/Fs in the leachate. Brominated flame retardants, oil compounds (like PAHs, *n*-alkanes, aromatic

substances *etc.*) and phthalates were also detected, whereas pesticides were not found. The high chlorinated dioxin level in one of the leachate samples was linked to the practice of open burning that took place periodically at that disposal site.[306] Investigations of the marine recipients including seaweed (*Phaeophyceae*), sculpin (*Myoxocephalus scorpius*), blue mussel (*Mytilus edulis*) and sediment from these three disposal sites showed elevated PCB concentrations compared to background levels, but still relatively low concentrations. For other organic contaminants, the concentrations were either very low or below the detection limit.[305]

Residuals from waste incineration include flue gas (particles, acids, organic and inorganic contaminants) and the fly and bottom ashes produced. Open burning can take place directly at the landfill, in barrels, cages, containers and outdoor furnaces. This type of burning results in incomplete combustion due to low combustion temperatures, and emissions include particles, CO, PAHs, Pb, Hg, POPs, including PCDD/Fs, VOCs and others. The PCDD/F emissions from open burning of mixed municipal solid waste could be 17 times higher compared to controlled incineration.[296] In Greenland, the six biggest towns and settlements currently send part of the waste to simple waste-to-energy incinerators with fly ash collection systems.[307] Dioxin I-TEQ concentrations up to 100 higher in the fly ash/air pollution control residues were seen than in modern Danish incinerators which also have extensive flue gas treatment.[308]

Most of these small facilities (mostly supporting less than 20 households) are facing capacity problems resulting in the accumulation of combustible waste in the landfills[297] and cannot clean the flue gas to sufficient health standards. The EU limit for dioxin emissions is 0.1 ng I-TEQ per m^3 and concentrations of 3.3 ng I-TEQ per m^3 have been measured in the flue gas at the current incinerator in Sisimiut.[309] In smaller settlements where waste is burned without flue gas treatment, PCDD/F concentrations up to 90 ng I-TEQ per m^3 have been measured in the flue gas.[310] To improve the Greenlandic waste management system and reduce environmental emissions, two new larger, state-of-the-art waste incinerators have been constructed for operation in Nuuk and Sisimiut. These two incinerators will treat all combustible waste in Greenland. The waste will be packed locally and shipped to the nearest incineration plant,[311] where the energy will be utilized for heat and the flue gas will be treated to meet EU regulations on incineration emissions. The fly ash is collected at the incineration plants in Greenland and sent to Norway for treatment at special hazardous waste facilities. Bottom ashes are disposed of at the local landfill. Ideally, after complete combustion, they should only contain incombustible larger particles and pieces of metal, soil, ceramic, glass or similar materials.

Spontaneous landfill fires also occur in the cold Arctic. These spontaneous landfill fires can be either surface or subsurface fires. Surface fires are less dangerous, as they can be easily extinguished since they are started by a heat source in contact with the surface of the waste. Subsurface fire occurrence is of a more complex nature and starts from the produced heat in the waste mass. Subsurface landfill fires are typically smoldering at low temperatures and can propagate into a large waste mass and, thus, are very difficult to extinguish. Dioxins in concentrations up to 4.95 pg m^{-3} (not I-TEQ) were recorded in ambient air during a 3-months uncontrolled subsurface landfill fire in Iqaluit, Canada, with mean dioxin concentrations 66 times higher than after the fire was extinguished.[312] Since landfills are typically placed near residential areas, the prevailing wind directions can send the smoke towards the inhabitants.

Contained incineration of waste instead of open dumps has been shown to improve wildlife conditions for non-breeding glaucous gulls in Barrow, Alaska, where the amount of waste in their diet was reduced from 43% to 28%.[313] A study from Alaska has also indicated risk for lower birth weight and growth retardation risks for babies and toddlers in Alaskan native villages from the use and exposure to contaminants from open and uncontrolled dumpsites with open incineration.[314] Furthermore, incineration can possibly increase the risk of cancer for people living in the close vicinity of the incinerator due to elevated air emissions,[315] and incineration also produces particulate residues such as ashes, which are toxic.

In a study from Svalbard, elevated PFAS concentrations were found in soil and meltwater samples contaminated from the abandoned open garbage dump in Adventdalen valley,[23] indicating that open garbage dumps can also be significant local sources of PFAS in the Arctic.

Monitoring and source elucidation of pollutants in marine Arctic biota

Contributions from local sources of contaminant exposure to high trophic level marine species are difficult to evaluate due to the animals' large habitats, foraging ecology, and diversity of prey. A case study (see Houde *et al.*[316]) was conducted based on long-term contaminant data in ringed seals (*Pusa hispida*) generated in the monitoring programs of Canada, Greenland and Svalbard. These monitoring programs have run since the 1990s. Potential local sources were identified at each site of the monitoring sites, *i.e.*, local community infrastructure (*e.g.*, mines, airports, solid and liquid waste treatment facilities, proximity to DEW sites). For ringed seals collected at the Canadian sites Sachs Harbour (Northwest Territories), Arviat and Resolute Bay (Nunavut), and Nain (Labrador), influences of these potential local sources on the long-term contaminant POP data were studied. The models indicated small, but notable contributions from these potential local sources for PCBs and PBDEs. Associations were found with population size, power consumption, mines and airports, accounting for 17% (PCBs) and 3% (PBDEs) relative to the contribution from long-range transport. The study showed that community population size, airport and power sources were positively associated with concentrations of the quantified PBDEs. For PCBs, positive relationships were found between several PCB homologue groups and population size, power consumption, mines and airports. No clear local source effect was seen in the ringed seals for OCPs or PFAS. This study suggests that local source contribution to PFAS, OCP and PBDE exposure of the seals from these communities is very low and fairly low for PCBs relative to the contribution to the environment from long-range environmental transport.[317] However, in a recent survey in gull eggs and fish samples from the Canadian Arctic elevated levels of selected OPE flame retardants were found.[318]

Research stations as contaminant sources

As a Norwegian contribution to the last International Polar Year (IPY 2007–2009), the joint Norwegian-Swedish research station in Kinnvika (Rypfjorden, Nord-Austlandet Island, Svalbard, 78 N latitude) was reactivated in 2007 and refurbished after the station was abandoned and left unused and unchanged since the end of the earlier geophysical Polar Year in 1957–1959. The station consists of a total of 11 buildings and has its own airstrip. The station facilities were not utilized for roughly 50 years until 2007. During the inspection of the station prior to reinstallation, technical wastes, leaking barrels and other waste piles were identified which had not been removed after the station was abandoned in the late 1950s. A first screening of the adjacent soil, drainage water and snow samples revealed elevated concentrations of PAHs and PCBs.[319] A recent follow up study confirmed possible local pollution in the Kinnvika.[9]

Polychlorinated naphthalenes (PCNs) were found to be elevated near the Ny Ålesund research station on Svalbard.[320] However, PCNs were phased out long ago, so their origin might be related to secondary sources or general diffusive occurrence.

Organic contaminants in Antarctic research locations

As a reference region, and to demonstrate the regional differences in pollutant profiles, a section on Antarctic source elucidation is included here as an Arctic comparison. The isolation of Antarctic scientific stations from local domestic and life-supporting infrastructure provides a clearer analysis of these as local sources.

There is no current industrial or public infrastructure in Antarctica. The human footprint in terms of construction includes several research stations used for different purposes. There are more than 60 seasonal (on pack ice) or permanent (on land) runways and heliports, docks, and scientific structures. The scientific activities and related logistics, including the use of machinery and motorized vehicles, produce a variety of waste. Some chemicals and fuel may be kept in containers subject to deterioration or may have been released directly into the environment.[146] Protective activities are carried out under the Antarctic Treaty System (ATS)[321] which guarantees the conservation of the Antarctic ecosystems through its Conventions. The Parties of the AT identified the need for an on-site treatment of waste or waste removal from Antarctica. The recommendation VIII-11 was adopted in 1975 and contained the first agreed guidance for the appropriate management and disposal of waste generated by scientific expeditions and stations, with the aim of reducing the human impacts on the Antarctic environment (https://www.ats.aq/devAS/Meetings/Measure/111). In 1989, the Parties adopted Recommendation XV-3 (https://www.ats.aq/devAS/Meetings/Measure/172) for more stringent waste management and disposal, for managing wastes deriving from present and future activities, and for programs to clean up existing waste and abandoned work sites.

The current specification on waste management and disposal in Annex III to the Environment Protocol, on Waste Disposal and Waste Management (http://www.ats.aq/documents/recatt/Att010_e.pdf) reflects many of those specifications included in Recommendation XV-3. In the past, waste in Antarctica was burned or disposed of in dumps, and abandoned facilities were simply left to deteriorate, causing serious impacts on the environment, such as the accumulation of physical debris (*e.g.* building materials, machinery, vehicles, general rubbish) and the leakage of fuel and chemicals. All waste degrades very slowly under the extreme weather conditions of the Antarctic. In 2019, the Resolution 1 of the Antarctic Treaty Consultative Meeting XLII adopted the Revised Antarctic Clean-Up Manual (Annex III, https://www.ats.aq/devAS/Meetings/Measure/701). While it is difficult to assess the volume of the waste in Antarctica, estimations at well documented sites and further extrapolations suggested that more than 1 million m^3 of waste and a similar volume of petroleum-contaminated sediment could be present in Antarctica.[322–324] This waste, and the associated contaminated sites are concentrated in the very rare deglaciated areas, consisting of approximately 1% of the continent.[325] These ice-free areas host very sensitive ecosystems[326] of endemic flora and fauna (*e.g.* Cape Hallet, East Antarctica) (Fig. 16). The impacts of

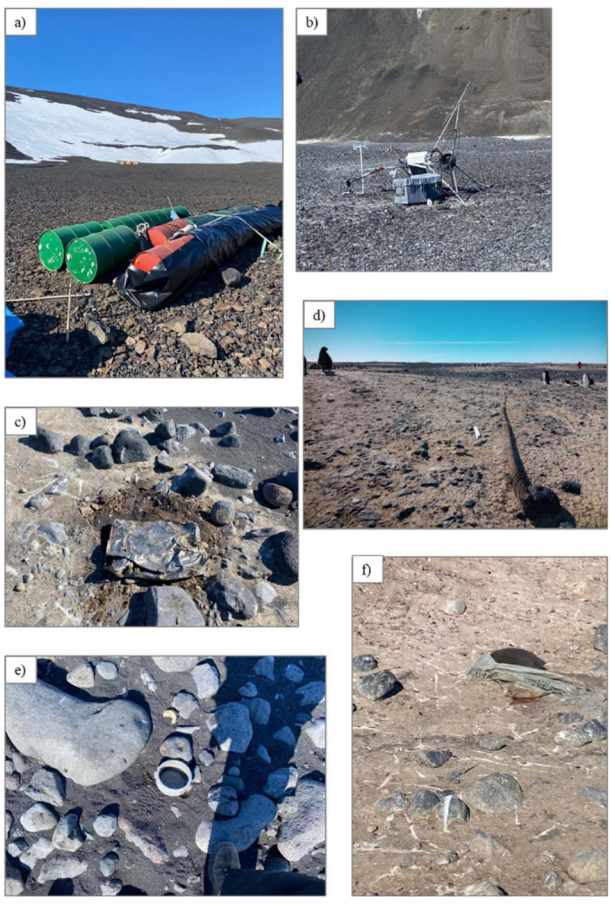

Fig. 16 Cape Hallet, East Antarctica: (a) an example of fuel drums in a helicopter fuel cache; (b) an abandoned weather station; (c, d and f) abandoned waste from an old station destroyed by a fire in 1964; (e) plastic waste on the beach in front of an Adèlie penguin rookery; (Photo N. Ademollo @ Italian National Antarctic Research Program).

waste on terrestrial and marine ecosystems are expected to increase over time: containers can degrade, even if it happens slowly, and release chemicals to the environment.[327]

Several previous studies have reported on pollutants from airports and stations as local sources of contaminants in Antarctica. Sediments, soils, and biota from coastal lakes along the Victoria Land were analysed for PCBs and PBDEs at different distances from the scientific station (Zucchelli Research station, Terra Nova Bay) along with samples from a long time field camp in a penguin rookery and from an airstrip with stored fuel drums.[328] The detection of lower chlorinated PCBs suggested that distant sources from long-range atmospheric transport were the main pathway. However local contamination from application of surface paint (containing *e.g.* PCB-11 and PCB-52) cannot be excluded. The authors hypothesized that the presence of a seasonal field camp near one of the lakes could represent a local contamination source; moreover, a nearby seabird colony could present a secondary source of contaminants in this area. Seabird colonies were suggested as a local secondary source of contamination through biovector-based pollutant transport.[329,330] An airstrip and the refuelling point between two other lakes was ascribed as a local source of PBDEs used during air operations.

The survey indicates that long-range atmospheric transport is mainly responsible for POPs distribution in Antarctica and that local sources (from *e.g.* stations, airstrips, refuelling points) affect mostly the local environment near the research station. Although the presence of seabirds may be a local contaminant input due to the biomagnification of contaminants and further release through excreta and dead organisms, the mechanism is different. It is a secondary source that does not directly result from the emission of chemicals in use. Instead, it is a concentrated input of contaminants that are transported to Antarctica with long-range transport. The transport of contaminants with migratory animals is included in the long-range transport mechanism according to the Stockholm Convention. While the effect on the local environment may be similar, it is relevant for the regulatory context involving long-range transport as a regulation criterion. Similar results were reported earlier for the same lakes[331] where the highest concentrations of higher chlorinated PCB and PBDE congeners were ascribed to local sources, *i.e.* stations as primary emission sources and seabird colonies as secondary sources. These principles were also described for Arctic lakes and ponds.[332,333]

Combustion of biomass and coal due to fires in South America was recognized as the main source for organic contaminants in Antarctica in several atmospheric surveys. However, elevated levels of naphthalene and its homologues (indicators of pyrogenic PAH) suggested that stations and related activities are local pyrogenic PAH sources. Also, higher PAH levels were detected around the Polish Arctowski station which were ascribed to incidental oil spills while using diesel fuel.[334] The local and long-range sources of some POPs and PAHs were investigated in continental and coastal surface snow of Antarctica.[331] The PAH composition suggested that a major source was regional and related to activity at the research stations, ship traffic, and fuel combustion, with an additional contribution from long-range transport. These authors reported about a considerable time lag between the emission of POPs from source regions in the southern hemisphere and the corresponding deposition in Antarctic snow; thus, they suggested that POP concentrations were likely to be ascribed to local or regional sources, while the long-range transport from other continents was hypothesized to play an indirect role.[331]

The distribution of PPCPs in Antarctica is still far from being well documented. A pilot study was conducted to evaluate the presence of organic UV filters in Antarctica.[310] 2-Ethylhexyl 4-methoxycinnimate (EHMC, octinoxate) was detected for the first time in Antarctic surface snow, in snowmelt and ponds with concentrations between 0.4-3.1 ng L^{-1}. Activities at research stations are a potential primary local source of EHMC but the results also suggested the possibility of long-distance sources. Nevertheless, according to the current state of the science, local anthropogenic influence is considered the main possible source of EHMC, both in midlatitude and polar regions.

Microplastics detected in Antarctic snow near research stations have also been attributed to local sources.[335] However, observations of homogenous spatial distributions of microplastics also highlight the likely long-range transport of microplastics to Antarctica.[336]

In summary, levels of long range-transported organic contaminants seem to be quite homogeneously distributed in the different sectors of Antarctica, while locally released contaminants usually show decreasing concentrations with distance from the local source. In this context, climate change-related increasing temperatures and consequent alteration of biogeochemical cycles[337] may affect the distribution of contaminants from local sources. Terrestrial coastal environments in Antarctica (where most stations are located) are linked to marine ecosystems, being both nesting sites for seabirds and resting or reproductive sites for seals. Terrestrial and marine ecosystems are interconnected, and changes, for example resulting from melting ice, may affect all these Antarctic ecosystems. Abandoned waste at continental sites distant from coasts may be initially/theoretically less impacted by climate change since temperatures here are extremely low.

In contrast to the Arctic, Antarctica's lack of industrial infrastructure and its controlled human activity—primarily through scientific stations—provides a relatively clean baseline for identifying pollutant sources. POPs, PAHs, and microplastics in Antarctica originate both from long-range atmospheric transport and local sources such as fuel spills, station operations, and seabird colonies acting as biovectors. These findings underscore the importance of distinguishing between primary emissions and secondary sources such as biovectors. This concept is equally relevant for the Arctic. Moreover, the Antarctic Treaty System's progressive waste management protocols, including on-site treatment and removal strategies, offer a possible model for Arctic governance as well. The Revised Antarctic Clean-Up Manual and Annex III of the Environmental Protocol demonstrate how structured international cooperation can mitigate environmental damage, even in remote regions. The detection of UV filter chemicals and microplastics near stations also highlights the need for monitoring emerging contaminants, which are increasingly relevant in the Arctic due to rising tourism and

research activities. Additionally, the observed influence of climate change on contaminant distribution—through melting ice and altered biogeochemical cycles—mirrors Arctic conditions, reinforcing the urgency for adaptive management strategies. By applying lessons from Antarctic pollutant pathways, regulatory frameworks, and ecological impacts, Arctic stakeholders can better anticipate contamination risks, improve waste handling practices, and strengthen cross-regional environmental protections. Thus, Antarctic findings serve not only as a scientific reference but also as a blueprint for sustainable pollutant management in the Arctic.

Industries as a pollution source in the Arctic

Some of the possible direct and indirect consequences of local industrial pollution in the Arctic include:

Pollutant disruption of marine ecosystems: Pollutants can enter the Arctic Ocean through a variety of local industrial activities. These contaminants can accumulate in the tissues of marine organisms, affecting their health and reproduction. This disruption can have effects on the entire food chain.

Contamination of freshwater sources: local industrial activities can lead to the release of toxic chemicals into freshwater systems. This contamination can impact the health of aquatic organisms and pose risks to communities that rely on these water sources and ecosystems for drinking water, food supply and recreational purposes.

Air pollution and human health risks: regional industrial activities release pollutants into the atmosphere, which can be carried over long distances by air masses. These pollutants after uptake and possible accumulation in the food web, can affect the health of Arctic residents and wildlife. Near emission sources, they can potentially cause respiratory problems and other health issues in local populations.

Disruption of indigenous livelihoods: local and indigenous communities in the Arctic depend on hunting, fishing, and gathering for their subsistence as a part of their social and cultural identity. Pollution-induced disruptions to the natural environment may have severe socio-economic impacts on these communities. Industrial emissions and discharges are identified as a major local source of organic contaminants in the Arctic environment.[101,338–341] Large-scale fisheries, fossil resource extraction as well as Arctic mining are identified as relevant local pollution sources.[89,262,342,343] All these industrial activities require extensive logistical support for allowing effective operations. This includes transportation, maintenance requirements and accommodation for operators.[344–347] To illustrate the different dimensions of these industrial pollutant sources in different Arctic regions, selected examples will be discussed in the following sections. Local sources associated with applications in Arctic industries have also been reported for PCNs, a group of chemicals applied in a variety of applications similarly to the use of PCBs.[348–350] However, the majority of PCN contamination in the Arctic is assumed to be from atmospheric long-range transport from mid-latitude regions.[349,351,352]

Oil and gas extraction

Oil and gas extraction is a major industrial activity in the Arctic. Arctic installations are currently delivering 10% of oil and 25% of natural gas supply globally.[115] According to an earlier survey of the US Geological Survey (USGS), conducted in 2008, the Arctic was estimated to contain about 90 billion barrels of undiscovered, technically recoverable oil and 1669 trillion cubic feet of technically recoverable natural gas.[353] This makes the Arctic one of the globally largest regions for oil- and gas extraction today.

Off-shore and gas extraction industries

The largest marine gas and oil deposits are currently expected in the Eastern Barents Sea coast regions, from the White Sea to the Chukchi Sea. In fact, the gas deposits in the Russian Arctic are considered to account for roughly 50% of the overall Arctic gas deposits.[354] The Russian Federation is, hence, the largest oil- and gas-producing Arctic nation.

With respect to marine oil and gas exploitation, oil and gas deposits in the Barents Sea, the Pechora region and the Siberian basins are specifically highlighted in terms of prospects and logistics and economic feasibility.[116,346,355] Both Russian and Norwegian authorities have recently opened new claims for oil and gas exploration in the Central and Eastern Barents Sea.[355]

Potential environmental consequences and exposure risks of Arctic marine oil and gas exploitation as well as impacts on local communities were comprehensively reported (for the first time) in a technical AMAP report in 2007.[356] Both operation-related pollution releases (drilling fluids, platform operations *etc.*) and accidental releases (oils spills, gas release *etc.*) were highlighted as threats to people and the environment. The report presented a variety of examples demonstrating the serious impact of oil spills on local environments and inhabitants of the North. Besides the potential effects of oil spills, remediation strategies in ice-infested waters were discussed.

Since this first report, elevated pollution levels and associated effects on marine wildlife have been reported in connection with Arctic marine oil and gas extraction. For example, pollution related to oil and gas production in the Barents Sea is reported regularly (for details see: https://www.barentsportal.com/barentsportal/index.php/en/). Early reports showed the direct uptake of oil-related hydrocarbons (*n*-alkanes and PAHs) in fish caught close to offshore installations in the Barents Sea (Fig. 17).

The recent Joint Barents Sea report in the joint report series (2016) showed elevated levels of hydrocarbons at the Stockman oil and gas production facilities. The levels in surface waters along the transect from Murmansk to the Stockman offshore production site exceeded the official maximum permissible concentration limits of 0.05 mg L^{-1} for hydrocarbons (Fig. 18).

High PAH levels were reported in water samples close to the Stockman gas condensate field with levels in the range 260–330 ng L^{-1}. Furthermore, a recent survey from the Barents and Kara Sea region highlighted the environmental consequences of the ongoing oil and gas production sites on the Arctic marine ecosystem and showed that levels of PAHs in the regions were similar to levels reported for Baltic Sea bottom sediments,

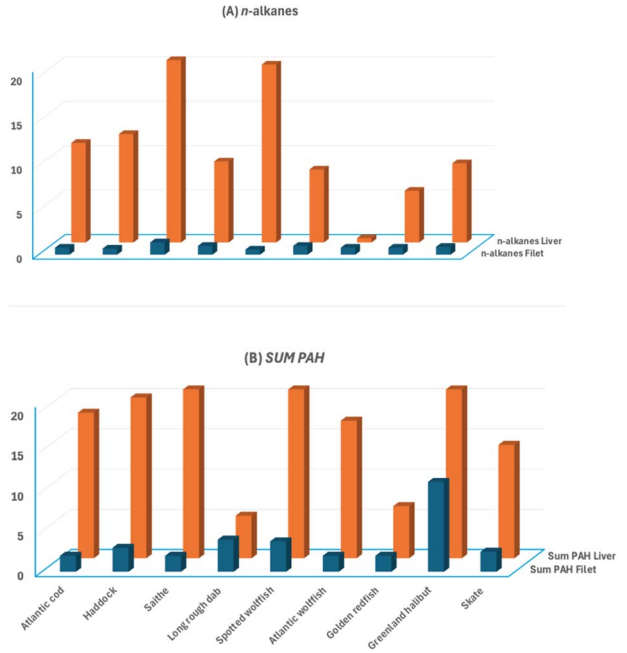

Fig. 17 Concentration levels [μg g^{-1}] of *n*-alkanes (A) and sum of polycyclic aromatic hydrocarbons ((B) Σ_{16} EPA-PAHs) in marine fish collected close to offshore installations. Data retrieved from report IMR/PINRO; Joint report Barents Sea 2007.

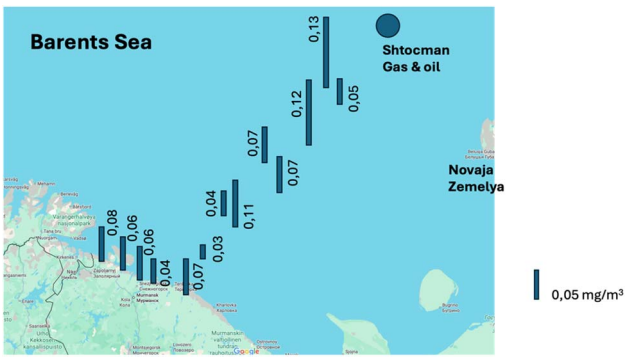

Fig. 18 Distribution of petroleum hydrocarbons in the surface waters [mg m^{-3}] along the projected Stockman pipeline. Report IMR/PINRO; Joint report Barents Sea 2007.

which are assumed to be highly impacted by anthropogenic sources. However, a study by Nemirovskaya and Khramtsova[357] concluded that, in general, the anthropogenic contribution of PAHs and aromatic hydrocarbons in the vicinity of off-shore gas and oil production units was minor compared to the geogenic contribution of aromatic hydrocarbons in the top sediments investigated.

The Exxon Valdez oil spill was a major environmental disaster that occurred on March 24, 1989, in Prince William Sound, Alaska, USA. It remains one of the largest coastal marine oil spills in U.S. Arctic history. The incident had severe consequences for the local ecosystem, wildlife, and the economy of the region. Thousands of birds, mammals, and fish were killed and local fisheries collapsed. The effects of the Exxon Valdez spill are still felt in the region today. Some species have not fully recovered, and the spill's long-term impact on the ecosystem is an ongoing topic of research. Therefore, the incident served as a catalyst for improved safety and environmental regulations in the maritime industry.[358–360]

Over the past three decades, numerous experiments have been conducted to gain insights into the behavior, fate, and weathering processes of oil spills under Arctic conditions. One notable project, known as the Baffin Island Oil Spill (BIOS) project, aimed to investigate the natural weathering processes of oil in the Arctic. The project involved the release of 45 m^3 of medium gravity crude oil in a typical Canadian Arctic shoreline environment at Cape Hatt, located on the northern end of Baffin Island, between May 1980 and August 1983.[361,362] In 1986, the first experimental oil spill in broken ice was conducted in the offshore area of Cape Breton Island, Canada.[363] In these experiments, drums containing oil were dispersed within the pack ice. The spreading of oil in pack ice, oil drift, oil evaporation, and emulsification were examined.

In summary, offshore oil and gas have been identified as important local pollutant sources in the Arctic. Although the focus has been on PAHs, PCBs, and other organic pollutants might also be emitted. Similar pollutant profiles as described above were reported for most regions where offshore installations are operated in the Arctic.[364–367]

Furthermore, PFAS has been associated with various functions in off-shore oil extraction operations. AFFFs are used for fire fighting training, fire combating. PFAS is also applied as enhancing agent for oil recovery[368,369]

Land-based oil and gas extraction industries as Arctic pollution sources

Arctic Canada has a long history of oil pollution, which dates to World War II. Oil contamination arose from weathering tests conducted on oil, oil transportation, and the abandonment of military facilities. These spill events were spread across the Canadian Arctic.

Petroleum hydrocarbons (PHCs) play a crucial role in the Canadian Arctic, where they are extensively utilized. However, the storage, movement, and transportation of fuels have led to numerous spills at both civilian and military sites in the northern regions. The Canola pipeline project, originally built in 1942 to facilitate oil supply to Alaska and Norman Wells, was shut down after just one year of operation. It is estimated that approximately 4 million liters of crude oil were spilled at 81 different locations along the pipeline route. Additionally, approximately 9.5 million liters of crude oil were left behind in the pipeline and storage tanks. More than 1200 m^3 soil was impacted.[369,370]

In the early AMAP report on oil and gas pollution,[356] the Komi oil spill (1994) was shown as an example of large-scale and long-term environmental effects associated with an oil spill in the Arctic.[371,372] Especially the effects on vegetation, terrestrial food webs and freshwater organisms were highlighted. Throughout the past decades several accidents in connection with land-based oil and gas productions have been reported.[89,373–375] Other large accidents reported until today include the following:

Trans-Alaska pipeline system incidents: various incidents have occurred along the Trans-Alaska Pipeline System during the past decades. These incidents have included oil spills, pipeline leaks, and equipment failures[376,377]

Prudhoe Bay oil spill (2006): in Prudhoe Bay, Alaska (USA), a corroded pipeline leaked approximately 267 000 gallons of crude oil, making it one of the largest spills in Alaskan history.[378,379]

The combination of aging infrastructure and loss of cryosphere/permafrost is considered a driving factor for increased pipeline-associated accidents[380] and the release of pollutants. For all these large-scale accidents, severe consequences for wildlife, ecosystems and local human populations have been registered and reported.

Tar sand surface oil extraction

In the North American Arctic and sub-Arctic regions (*e.g.*, Alaska, Canadian Alberta), tar sand deposits are mined and refined as resources for petroleum products. Tar sands, also known as oil sands, are a type of petroleum surface deposit.[381–383] They consist of a mixture of clay, sand, water, and a dense, viscous form of petroleum called bitumen.[384] Bitumen is so thick that it cannot be pumped in its natural state. Therefore, it needs to be extracted and processed before it can be refined into usable products like gasoline and diesel. The petroleum deposits are usually extracted either by surface mining of the bitumen deposits or *in situ* extraction in cases when the bitumen layers are deeper underneath the soil surface.[385]

The extraction of the bitumen layers is causing severe landscape changes and loss of habitat in the Arctic ecosystems with significant effects on biodiversity and ecosystem services for the local people.[386–388]

The following major environmental consequences are identified for tar sand-based petroleum production:

- *Water usage and contamination*: the extraction process requires large amounts of water, especially in surface mining operations. This water is often sourced from nearby rivers or aquifers. The used water, now contaminated with various chemicals, may be stored in tailings ponds, which can pose risks to local water quality and wildlife.[389–391]
- *Greenhouse gas emissions*: extracting and processing tar sands is energy-intensive, which can lead to substantial greenhouse gas emissions. The production of bitumen from tar sands can result in higher emissions compared to conventional oil production methods.[392,393]
- *Air pollution*: the extraction and processing of tar sands can release various air pollutants, including PACs, VOCs, sulfur dioxide, and nitrogen oxides, which can contribute to smog and other air quality issues.[394,395]
- *Tailings ponds*: the leftover materials from the extraction process, known as tailings, are stored in large ponds. These tailings can contain a mixture of water, sand, clay, residual bitumen, and chemical additives. If not managed properly, these ponds can pose risks to the local water quality and wildlife.[396–399]
- *Long-term land reclamation*: once a tar sands site is exhausted, efforts are made to reclaim and restore the land. However, this process can be challenging and may not fully restore the original ecosystem, even after long-term restoration.[400,401]

Significant environmental consequences have been monitored for several of the tar sand production sites.[390,402–404]

For the Alberta tar sand industry, impacts on the fragile local ecosystem have been reported. Furthermore, increased erosion in the open mining areas as well as processing lead to haze and dust events in the nearby communities and may also lead to increased exposure risks for volatile organic industrial chemicals.[405] Elevated levels of chemicals such as PAHs,[406] VOCs,[407] and PFAS[408] were found in environmental samples taken in the vicinity of the extraction sites. Even adverse health effects in the local populations and workers have been frequently reported.[409–411]

Long-term impacts of oil spills in Northern Canada

An investigation on spilled oil weathering in the legacy of civilian and military sites showed restricted mobility of spilled oil. Starting in 1977, site reconnaissance programs were initiated to investigate 11 specific oil spill sites in the Canola pipeline trial. Locating these spill sites proved challenging. The programs observed that the spilled oil had a tar-like consistency and appeared to be stable. Analysis of the PHC content indicated that the primary PHC fractions present were the heavier fractions. There was limited evidence of free oil or oil residues in the water table. The programs did not find any active seepage faces of crude oil or sheens along banks or in sediments at major drainage areas or water bodies, nor was there any evidence of downstream impacts.[370] Pack ice experiments in 1986 revealed a significant reduction in the spreading of oil within the ice. Unlike in spills occurring in more moderate temperatures, the oil in pack ice did not drift in relation to the surrounding floes.[363]

Meanwhile, vertical migration of spilled fuel contamination into permafrost was observed at multiple sites at the Canadian Forces Station Alert and Isachsen High Arctic Weather Stations.[412] Gravity drainage and thermal contraction-induced fissures in the soil were believed to be the major causes of vertical migration. Gravity drainage occurred when the spilled fuel contamination filled interconnected air voids in the soil. These voids acted as conduits, allowing the contaminants to flow downward under the influence of gravity. Thermal contraction could lead to shrink and development of fissures, which provided pathways for the spilled fuel to infiltrate deeper into the permafrost layers.[412]

Long-term natural attenuation of oil in sediments was analyzed in frequent visits to the BIOS sites.[413–415] Oil that was deposited on the sediment surface underwent weathering and was eventually naturally removed. After 20 years, 87% of the total hydrocarbons and 92% of the saturates were degraded.[414] Biodegradation and photooxidation were suggested to play important roles in the removal process. Oil that was soaked into the sediment to a depth of approximately 2 to 5 cm could be sheltered from weathering. A total of 99 PHCs, including PAHs, *n*-alkanes, branched alkanes, alkylcycloalkanes, hopane and sterane biomarkers, and alkylbenzenes were analysed for in surface (0–2 cm) and subsurface (5–10 cm) sediment samples at different sites around Cape Hatt (Baffin island) after 39 years

from when the BIOS project commenced. The results showed that 14 of the 16 priority PAHs were detected in concentrations surpassing the marine sediment quality guideline limits set by the Canadian Council of Ministers of the Environment. The Toxic Equivalency Quotient values for these PAHs ranged from 1.40 to 270 and 1.70 to 350 μg kg^{-1} within the surface and subsurface sediments, respectively.[415] Samples of Lagomedio oil from the intertidal zone were found to be heavily weathered and biodegraded, while the sample from the backshore beach showed minimal alteration after 39 years. It was suggested that biodegradation of oil in the intertidal and backshore beach sediment zones occurred at a very slow pace and could be limited by nutrient availability. While tidal and wave action greatly influenced the persistence of oil in the Arctic environment, temperature variations and microbiology play significant roles in the removal of petroleum.[415]

History and future for arctic oil spill responses

Attempts have been made to remediate oil spills in Arctic regions. At the Canadian Forces Station Alert, two remediation approaches of "new spill" and weathered diesel-contaminated soils were tested in treatment plots. Direct exposure of "new spill" soil was able to achieve 94% removal rate after one year while 2 years were required to remove a similar percentage of diesel in weathered soil. Tilling the soil was found to be advantageous compared to the addition of amendments. It was recommended to land farm amend and combine it with periodic tilling to remediate the contaminated sites at the Canadian Forces Station Alert. These findings have been incorporated into the Alert Environmental Management System.[416] Meanwhile, for offshore oil spill response, there is a lack of effective techniques for recovering spilled oil in ice-covered environments.[417]

The lack of infrastructure and the remote nature of the Arctic exacerbate the challenges associated with spill response, which could take days or even weeks to initiate. For example, an estimated response time for offshore oil spill equipment in the Arctic is around 48 hours for spills up to 150 tons and up to one week for spills up to 1000 tons.[418] Therefore, early warning of oil spill is of great importance. Oil spill reporting systems in Arctic Canada have been developed since the early 1970s. The Northwest Territories-Nunavut Spills Working Agreement was established in 1981 and is still in effect, which serves the purpose of providing a streamlined approach to reporting hazardous materials spills and sharing information related to spills in the Northwest Territories (NT) and Nunavut. By functioning as a single-window approach, the agreement facilitates efficient spill reporting and information dissemination throughout the NT and Nunavut. This approach benefits both the regulatory agency and the responsible party by promoting a consistent and coordinated approach to regulating spill management operations. In summary, the Spills Working Agreement plays a vital role in ensuring effective spill response and promoting cooperation among agencies in the NT and Nunavut. It serves as a model for inter-agency collaboration and contributes to the overall goal of environmental protection.

The exploration, drilling, and offshore oil production activities in the Arctic region pose significant threats to the fish and marine mammals that are crucial for the livelihoods of Arctic Indigenous communities. The Government of Canada announced a ban on issuing new offshore oil and gas licenses in Canadian Arctic Waters in 2016. The designation should be reviewed every five years through a climate and marine science-based life-cycle assessment of Canadian Arctic Waters with the joints of Inuvialuit Regional Corporation, Nunavut Tunngavik Incorporated and the governments of Yukon, the Northwest Territories and Nunavut.

With the melting of Arctic sea-ice, the Northwest Passage and Arctic waters in Canada are becoming increasingly accessible. These routes offer significant advantages over traditional shipping routes in reduced shipping costs and transportation times.[419] However, shipping in the Canadian Arctic still poses significant risks and challenges due to difficult navigation conditions, poor weather and visibility, limited charting, and the difficulty of detecting sea ice. As the melting of sea ice continues, shipping activities are increasing in the region, raising concerns about the potential for oil spills that could harm the fragile Arctic ecosystem and the communities and wildlife that rely on it.[418] Therefore, development of effective oil spill response policies that aim at Arctic shipping is of great importance. Several strategies can improve effectiveness of oil spill response.[420–422]

- Including Arctic Indigenous peoples in decision-making processes is essential.
- Shipping lanes should be determined based on information about subsistence activities and environmentally sensitive habitats.
- Prohibit the use of heavy fuel oil.
- Response plans should address logistical challenges in the Arctic region and ships should carry equipment for initial spill response.
- Equal protection of indigenous communities in the North.
- Establish local training organizations and increase capacities for oils spill response.
- Prioritize Arctic Canada's Oceans Protection Plan.

Arctic mining industries as pollution sources

In this context, we refer to Arctic mining as the extraction of minerals and resources from the Arctic region, which includes the oceans. Arctic mining can have significant environmental impacts, both locally and globally. Here are some ways in which Arctic mining can contribute to Arctic local pollution:

Operation related chemical contaminants: mining operations often involve the use of chemicals to extract valuable minerals from the ground. These chemicals can include technical chemicals, PACs, acids, solvents, and other compounds. If not managed properly, these chemicals can contaminate the local environment, including soil and water.[141,423–426]

Tailings and waste disposal: mining generates a substantial amount of waste material, known as tailings. These tailings often contain leftover minerals, chemicals, and other potentially harmful substances. If not properly contained and managed, they can leach into the surrounding environment, polluting soil and water.[426–429]

Air pollution: mining operations can release various pollutants into the air, including dust, particulate matter, and

potentially harmful gases. This can have health impacts on local communities and wildlife.[429–432]

Transportation and infrastructure: building and maintaining the necessary infrastructure for mining, such as roads, ports, and processing facilities, can have associated pollution impacts. For example, the construction of roads can lead to habitat fragmentation and the release of pollutants from construction equipment.[156,423,433–436]

Oil and fuel spills: in addition to mining activities, transportation of mined materials can also lead to pollution risks. As discussed above, spills from ships or other transportation vessels can release oil and fuel into the surrounding waters, harming marine life and ecosystems.[135,437–441]

Effects on ecosystems: mining operations often require significant alterations to the landscape. This can involve removing vegetation, altering waterways, and disrupting natural habitats. These changes can have cascading effects on local ecosystems and the species that depend on them.[135,425,437,442–447]

Efforts to mitigate the environmental impacts of Arctic mining include the use of advanced technologies, thorough environmental impact assessments, and strict monitoring and remediation measures. Additionally, some argue for the importance of transitioning towards more sustainable and responsible mining practices, as well as considering alternative approaches to meet resource needs, such as recycling and the development of alternative materials.[448–453]

Metal refining industry as local source for organic pollutants in the arctic

Metallurgical industries have been identified as major local pollutant sources in Arctic regions since early screening studies in the 1960s.[454–457] In the beginning, in addition to exceptionally high levels of local trace metal contamination, non-ferrous metal smelters were identified as a major source of medium- and long-range atmospheric transport of components contributing to acid rain, such as sulphur dioxide, and mobile trace metals incl. mercury across the Arctic.[458–462] Metallurgical industries such as nonferrous metal smelter are usually established in connection with mining activities.[463] The Arctic regions are known to be rich in metal deposits, with abundant minable metal deposits documented from virtually all Arctic regions.[463] In addition, the current Arctic environmental changes are accelerating the loss of the Arctic cryosphere. These environmental changes in the Arctic will allow easy and profitable access to mining locations and provide economies for new refinement and metallurgical industries in the region.[464,465]

The first monitoring of pollutant emissions from metallurgical industry was established at the Norwegian–Russian border where significant atmospheric pollution from the metallurgical industries at the Kola peninsula was identified. The metal smelter industry in Nikel, Zapoljarnyi and Montsjegorsk (Murmansk Oblast, Russian Federation) were identified as significant pollution sources. In addition to local soil contamination with metal residues[466] also atmospheric sulfur dioxide and metal residues deposited in Norway were associated with these industries. In the early 2000s, the Petchenganikel smelters were considered Europe's largest sulfur emitters at that time.[467,468] However, the metal smelters in Nikel have stopped production after 2020. Instead, the main production was moved to Montsjegorsk.

PAHs and PCDD/Fs are created during high temperature metal smelting and emitted from metallurgical industries in the Arctic.[89] The first confirmation of PAH and PCDD/F emissions and atmospheric transport from a metal smelter in Northern Norway (Syd-Varanger, Kirkenes, Norway) was documented in the early 1990s. Considerable local PCDD/F contamination in soils, sediments and benthic organisms were confirmed.[469] The metal production at the Syd-Varanger facilities in Northern Norway was discontinued in 1996. The first comprehensive AMAP report documented POP emissions from smelters as local contamination sources and showed that magnesium smelters in Arctic Norway emitting PCDD/Fs.[470] PAH emissions from metallurgical industries in the Arctic were also reported. It should be noted that after more than 30 years of atmospheric PAH monitoring in the Arctic, the overall atmospheric PAH levels are still not declining despite major international regulative efforts to reduce PAH emissions in combustion driven infrastructures in the North. However, PAH emissions from metal production are reported to be declining due to improved process techniques and effective emission clean up procedures.[471]

Aquaculture as a source for organic pollutants in the Arctic

Aquaculture, the controlled cultivation of aquatic organisms, including fish, shellfish and algae, plays a crucial role in the growing demand for food on a global basis. As agriculture faces limitations due to restricted availability of fertile land to meet the demand for food by the increasing human population, aquaculture offers solutions to enhance food security and economic growth. Nevertheless, the release of organic pollutants from aquaculture facilities poses risks to local ecosystems and may affect ecosystem services. Local populations may experience pollutant exposure through direct contamination of the local marine fauna.

The environmental impact of freshwater and marine-based aquaculture is well documented.[472–475] Usually, extensive impact of nutrients on the adjacent aquatic environment are reported as a major concern.[472,476,477] However, elevated pathogen occurrence, high escape of caged specimens, environmental pollution by chemicals used in animal treatment as well as technical aids used for maintaining the infrastructures are important environmental consequences of large-scale aquaculture.[478–480]

The following pollution sources, processes and entrance paths have been associated with aquaculture:

Fish feed: fish feed may contain a variety of chemicals applied as stabilizers, additives and supporting agents, which might be of environmental concern. Furthermore, nutrients (nitrogen and phosphorous) released from excess of fish food and from excrement will lead to eutrophication, algal blooms, and oxygen depletion.

Application of chemicals: antibiotics and other veterinary medicines for disease control may be released into the local

marine environments. Furthermore, pesticides, and antifouling agents are also used in aquaculture and can contaminate water and sediments in the local marine environment.

Wastewater discharge: aquaculture facilities release wastewater containing organic matter, nutrients, and chemicals.

Sediment resuspension: daily activities around the installations such as feeding, harvesting, and maintenance disturb sediments, releasing and remobilizing organic pollutants previously accumulated in the seabed.

Until today, emissions of process chemicals for maintaining functionality of the infrastructure as well as bioactive substances regularly applied for disease prevention and treatment are less frequently investigated, although potential exposure risks to consumers need to be comprehensively evaluated.[481–483]

Large-scale aquaculture production takes place in many locations of the Arctic coastal regions. Productions include salmon (*Salmo salar*), Atlantic cod, catfish (*Anarchichas minor*), sea urchin (*Echinus esculentus*) and even kelp (*Ascophyllum nodosum*) for the Asian markets.[484] Aquaculture is an important industry in virtually all circum-Arctic countries and stands for roughly 2–3% of the world's aquaculture production.[484] Currently, Norway is the largest aquaculture nation in the Arctic with focus on the production of salmonid fish species. The current climate-change related water temperature increases in Arctic coastal waters (estimated 0.5–5 °C) is expected to have a positive effect on the production conditions in Arctic aquaculture. Hence an expansion of aquaculture production in the Arctic is expected due to a rapidly increasing demand of these products on the global market,[484] and increasing environmental consequences must be expected.

The multifaceted impact of large-scale aquaculture is documented in a variety of reports and studies worldwide.[477,485,486] Consequently, the environmental impact of aquaculture in the Arctic is also documented and openly discussed in the public.[487–491] A better understanding of pollutant sources, pollutant fate and their impact on the local marine environments is needed to develop a sustainable future strategy for aquaculture. This is of specific importance for the Arctic since water temperatures are low and the breakdown of organic compounds is slower compared to coastal regions and mid-latitudes. This may lead to enhanced accumulation of anthropogenic pollutants in the environment. Aquaculture was recently identified as an important local pollution source of macro-, meso- and microplastic materials.[235] Elevated exposure of farmed organisms (fish, mollusks, echinoderms, *etc.*) to these polymer materials can be expected. Recently, elevated levels of meso- and microplastic particulate materials were found in blue mussels collected in the close vicinity of production units in Northern Norway.[478] The focus of the study was on the presence of PP, high density polyethylene (HDPE), polyethylene terephthalate (PET), and PVC. It is, hence, expected that a continuous increase of the aquaculture industry in the North will also increase the release of plastic-associated wastes and related pollutants (additives) in the local aquatic environment.[492]

Indications were found that microplastics released from aquaculture infrastructure enhanced the mobility of selected POPs in the nearby environment.[478]

A first screening of POPs in commercial feeding materials for aquaculture as well as in salmon smolts in British Columbia (Canada) was reported from four locations. Also here PCCD/Fs and PCBs but also selected organochlorine pesticides (OCPs) were investigated.[493] Local specific pattern differences in POP concentrations were found in hatchery-fed salmon smolt with predominant contribution of industrial chemicals (PCDD/Fs and PCBs). However, the concentration levels were generally low.[235]

Elevated levels of POPs including PCBs, OCPs, PBDEs but also PFAS were found in wild marine fish caught close to large coastal fish farms in Northern Norway.[494] In this study, particularly, Atlantic cod and saithe (*Pollachius virens*) were collected close to salmon farms. 45% of the wild fish caught were found with salmon feed in their intestinal tracts. POPs and PFAS concentrations were significantly higher in 50% of the farm-associated fish compared with the control group (not associated to fish farming).[494] Several other recent reports identified contaminant levels in farmed organisms as well as direct impact of organic pollutants on the local Arctic ecosystems close to aquaculture infrastructures.[481,493,495–498]

Military infrastructures as local pollution sources in the Arctic

The Global North faces a paradoxical challenge: a region of extreme environmental vulnerability is also of high military significance, leading to permanent military installations as well as large-scale operations in a vulnerable environment.

Recent geopolitical tensions and environmental risk

After a period of détente, following the cold war, military interest in the Arctic has intensified in recent years, driven by strategic imperatives and the opening of new shipping lanes due to melting ice and environmental changes. Especially the large Arctic nations, the USA, Canada and the Russian Federation have currently increased their military presence in the Arctic. The recently proposed US Arctic Commitment Act aims for a year-round naval and coast guard presence in the region, responding to military competition with the Russian Federation (https://www.congress.gov/bill/117th-congress/senate-bill/4736). The Danish Government also recently issued the Danish Defense Agreement, in collaboration with the governments of Greenland and the Faroe Islands, aiming at stronger presence in the Arctic.[499]

Environmental risks: this military buildup of virtually all Arctic nations, however, also poses new environmental risks. As tensions escalate between the U.S. and Russia, a military presence could be perceived as provocation, potentially leading to increased Russian and NATO military exercises in the regions.[500–503] This increased frequency of large-scale military exercises in combination with the establishment of new bases and the reopening of previously operational military

infrastructures will inevitably lead to increased risk of local contamination of the local Arctic environment. Increased shipping for military and civil purposes can generally be associated with the release of certain chemicals, for example antifouling agents applied to ship hulls, such as organotin substances.

Climate change: the Arctic, already warming nearly four times faster than the global average, suffers from decreasing sea ice coverage, rising ambient temperatures, and altered ecosystems. The rising number of military infrastructures in the Arctic contribute to these environmental challenges.[218,501,504] A detailed account on climate change related scenarios for Arctic local pollution has been given in a dedicated review within this special issue by Muir *et al.*[83] Details can also be found in the recent AMAP report on CEACs and POPs in a changing Arctic climate.[24]

Military pollution sources

As earlier discussed for industrial and domestic infrastructures as local pollution sources, also military sites, their life supporting infrastructures and activities associated with maintenance and operation may contribute to pollution of the local Arctic environments as reported for the majority of the Arctic nations.[374,505–507] Several source types have been identified; however, they are largely similar to public and industrial installations. Military pollution sources include airports,[268] training and exercise grounds,[508–512] large scale exercises,[513–516] fixed infrastructures and supporting technologies.[517,518] In general, the polluting impact of military infrastructures and activities is well documented. However, the Arctic region, although currently a geo-political focus area, is only scarcely investigated when it comes to the environmental impact of regional military presence and activities.[89,507,519,520] Pollution sources from military installations and activities were already highlighted in the first AMAP reports.[132,470,506] However, military sources were mainly associated with radioactivity and trace metal pollution rather than organic contaminants in an Arctic context.[506] Nevertheless, recent reports also related high levels of organic pollution to military infrastructures and related activities in the Arctic.[508,515,521,522] This section will discuss military installations and activities as sources of pollution in a historical context. The Second World War was aptly named as conflict spread across most of the planet, including the Arctic. The USA and Japan fought in the Aleutian Islands campaign from 1942–1943, which led to the displacement of Indigenous populations in concert with the militarization of the Aleutian Archipelago.[523] The Cold War greatly expanded the military footprint in the Arctic as both NATO countries and the Soviet Union constructed thousands of military installations across all Arctic countries.[524] When they reached obsolescence, these installations were typically abandoned without regard to containment or removal of on-site contaminants. Therefore, military installations throughout the Arctic, including formerly used defense (FUD) sites, are an important source of contaminants including POPs, petroleum products, and toxic metals and metalloids.

St. Lawrence Island (Sivuqaq), the largest island in the Bering Sea (Fig. 19), illustrates many of the problems of polluted

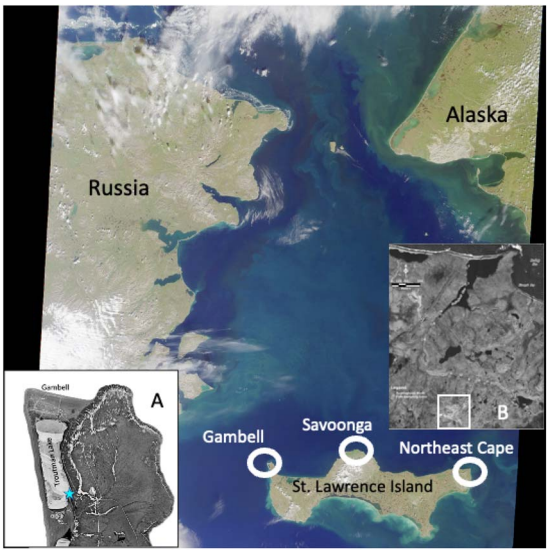

Fig. 19 St. Lawrence Island is the largest island in the Bering Sea, and part of the United States. Because of its proximity to the former Soviet Union, the U.S. military established White Alice radar installations in Gambell and Northeast Cape. Residents from the village of Savoonga still conduct subsistence activities at Northeast Cape. Inset (A): the village of Gambell and the adjacent Troutman Lake. Inset (B): Northeast Cape, including long-term monitoring stations on the Suqi River. The main Formerly Used Defense (FUD) site complex and barrel storage area was located within the white box; contaminants from this site migrated into the Suqi River.

military sites in the Arctic. The island has only two villages, Gambell and Savoonga. As of the 2020 U.S. Census (https://www.census.gov/), the island had a population of 1475 residents, nearly all of whom were Siberian Yupik Alaska Natives. Subsistence hunting, fishing, and gathering are critical to the food security and culture of residents, including in and around their villages as well as at sea. Savoonga residents also conduct subsistence activities at Northeast Cape, which was the site of a Yupik village prior to occupancy by the U.S. military.[525,526] The U.S. military established radar surveillance stations at both Gambell and Northeast Cape during the Cold War, and they remained operational until rendered obsolete by military satellites.

The U.S. Air Force obtained Northeast Cape in 1952 *via* Public Land Order 790 and built an Aircraft Control and Warning Station (AC&WS) in 1957.[527,528] The station transitioned to a White Alice (Alaska Integrated Communications and Electronics) site to detect Soviet aircraft and missile attacks during the Cold War.[528] The facility closed in 1972 and left a legacy of at least 30 contaminated areas across 19 km^2.[528] Contaminants include petroleum products, PCBs, pesticides, solvents, and metals.[528–532] The cost of site cleanup between 1985 and 2014 amounted to US$120 million, the most expensive remediation program to date in Alaska.[528,533,534] Despite extensive cleanup of Northeast Cape, levels of PCBs and Hg, among other contaminants, remain a health concern.[531,532] Freshwater fish collected downstream of the FUD site have significantly higher concentrations of PCBs than do upstream fish, and show induction of vitellogenin (a biomarker of estrogenic contaminants) and

differential gene expression.[532] Dolly Varden (*Salvelinus malma*), an important subsistence fish for residents, contain levels of PCBs and Hg that exceed U.S. EPA health advisories for fish consumption, and these contaminants derive primarily from the FUD site.[531] The Northeast Cape FUD site is also associated with OCPs in freshwater fish, similar to the profiles observed in the blood serum of Yupik residents; these contaminants include chlordanes, DDT compounds, mirex, and HCB.[529] The U.S. operated the surveillance station at Gambell from 1948–1965 on the northern shoreline of Troutman Lake adjacent to the village.[535,536] The FUD site extends over ~7 km^2 in Gambell and along Troutman Lake, including waste disposal sites.[535,536] Remediation of the Gambell FUD site included both burial and removal of hazardous wastes such as PCBs.[536]

Despite remediation, low trophic level fish in Troutman Lake have high concentrations of many organic contaminants, including (in descending order of concentration) PBDEs, PCBs, PFAS, OPEs, ΣDDT, HCB, OPE metabolites, Σchlordane, and ΣHCH.[537–539] These fish also display developmental pathologies and altered gene expression.[537] Yupik residents also have elevated concentrations of many of these same contaminants, which derive primarily from their subsistence diet of high trophic-level long-lived marine mammals that accumulate atmospherically-deposited contaminants,[538,540,541] though the FUD sites also appear to be a source of exposure.[529,542] Concentrations of at least some of these contaminants, including PFAS and PBDEs, are associated with altered concentrations of thyroid axis hormones in Yupik residents.[543]

The contaminant profiles of POPs in the same fish species differ between Troutman Lake and Northeast Cape, illustrating the importance of the local contaminant sources.[529,531,532,538,539,544] Furthermore, the concentrations of POPs in fish at FUD sites on St. Lawrence Island are much higher and contain heavier-molecular weight congeners (for example, for PCBs) than do the concentrations of POPs in fish at nearby reference sites.[531,532] In sum, research from St. Lawrence Island illustrates the importance local sources of contaminants from FUDs in exposure of people and wildlife to pollutants, in addition to exposure from global distillation of POPs. These contaminants pose an on-going threat to human health and wildlife conservation. The St. Lawrence island FUD was an integrated part of the US-Canadian DEW line, established in the 1950s and consisting of software-controlled and coordinated radar and observation units across the western Arctic from Alaska to Greenland and Iceland[506,545–548] (Fig. 20). These previous military activities as well as leakage of technical chemicals from decommissioned technical installations are considered one of the main sources for this type of local organic pollution.[549–553]

The DEW Line was a series of radar stations constructed during the 1950s across the Arctic region, spanning from Alaska through Canada, over Greenland, and reaching Iceland. This military and industrial initiative, outlined in the Abandoned Military Site Remediation Protocol, left behind a significant environmental footprint. Hundreds of trucks, graders, and construction equipment were abandoned, along with approximately 60 476 barrels of oil in the pipelines.[546] Furthermore, it is

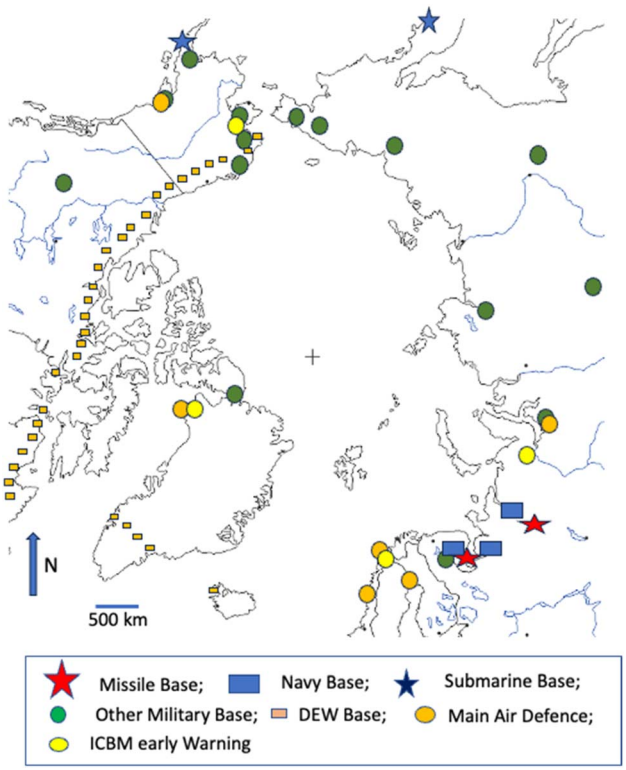

Fig. 20 Military installations reported after the cold war period (1993). This figure is based on information from a combination of openly accessible reports.[554–557]

believed that around 108 857 barrels of oil were spilled, specifically in Hooper Bay, Cape Romanzof, and Point Hope, resulting in an estimated 80 000 gallons (303 m^3) of petroleum leakage into the environment.[558] The Canadian Forces Station Alert, located in the northernmost permanent settlement in the world, relied heavily on petroleum oil and lubricants for heating, power generation, and vehicle transportation. In 1999, a diesel pipeline break resulted in a petroleum leakage incident, causing soil contamination. The total petroleum hydrocarbon concentration in the affected soil was found to be as high as 19 000 parts per million (ppm), involving approximately 1450 m^3 of contaminated soil.[416,559]

Already in the early 1990s, the abandoned DEW line stations, were identified as significant environmental pollution sources in the Arctic (Fig. 20). Pollutants include PFAS, PCBs, VOCs, PAHs and many other contaminant groups.[178,447,506,519,546,560,561] However, this reported mapping of military installations in the North should not be considered as an up-to date version, since several DEW line stations were established later outside the US-Canadian corridor as is reported in the case study from a typical DEW line site in Iceland, reported in this special issue.[562]

In Canada, multiple studies have been conducted on the PCB contamination of sediments, invertebrates, fish, seabirds, and ringed seals from the surrounding of a military radar station in Saglek Bay, Labrador.[550,551,553,563] The recovery process after the cleanup of the source was also assessed and showed decreases in PCB concentrations in sediment and biota after cleanup.[563,564]

Pond sediments adjacent to an old military dump at Iqaluit were also found to be contaminated with PCBs and PAHs at levels exceeding environmental quality guidelines.[565] In a remediation plan for the area, the City of Iqaluit has identified other old dump sites that may be continuing to emit contaminants (Nunatsiaq News 2017; see: https://nunatsiaq.com/stories/article/65674feds_to_clean_up_historic_iqaluit_dumpsites).

The NATO radar station on Jan Mayen was established in the early 1950s. In 1993 and 1994 high THC (13.400 μg per g dw) and PCB (0.2 μg per g dw) concentrations were detected at the military radar station dumpsite at the settlement Olonkinbyen. The PCBs originated from transformer oil, which had been deposited at the now closed dumpsite at Trollsletta. The dumpsite is located on a slope, 2–5 m from the seashore. Environmental investigations established that PCB concentrations in the soil of this dumpsite ranged between 0.06 and 35.8 μg per g dw (average 3 μg per g dw) while samples collected ~20 m from the dumpsite contained PCB concentrations between 0.002 and 0.06 μg per g dw.[566] Samples were also collected from various seabird species. Congener patterns differed between different bird species and concentrations increased with increasing trophic position.[566] Body burdens of PCBs in kittiwake (*Rissa tridactyla*) and glaucous gull corresponded with levels found at other Arctic Norwegian locations, indicating background contamination, thus excluding the dumpsite as a potential PCB source to seabirds. Arctic char was sampled from the freshwater lagoon on the island, showing high concentrations of PCBs. The PCB fingerprint did, however, not correspond with that of the dumpsite and the source connection with the dumpsite was again rejected. Marine fish were sampled along a transect perpendicular to the Trollsletta shore, only showing elevated PCB concentrations in sole (*Hyppoglossoides platessoides*). After additional geological investigations, the risk assessment established that the dumpsite land masses should not be moved, but instead be covered with uncontaminated soil to avoid wind-driven contaminated particle transport to surrounding land and waters.[566] This decision was based on the low average PCB concentrations (3 μg per g dw) measured in soil corresponding to Tier I soils (1–5 μg per g dw) according to the Canadian DEW Line clean-up protocol of the previous Department of Indian and Northern Affairs Canada (INAC).[567] This protocol was used since guidelines for contaminated soil were lacking for the Norwegian Arctic. According to the Canadian protocol Tier I soils are to be deposited in a non-hazardous landfill and covered with clean soil. In the wake of these investigations, guidelines for PCB-contaminated soil were established for Jan Mayen.[568] Similar guidelines are still lacking for Svalbard.[569]

As mentioned above, military airports in the Arctic are currently in the scientific focus as local pollution sources.[570–573] Especially after military operations ceased and the airport infrastructure was transferred to civil operation, many airports in the North were found to be contaminated and in need of often costly remediation actions.[519,521] Earlier, mainly trace metals were reported as operation-related pollutants in Arctic military airports.[574] Recently several organic chemicals emitted from aircraft operations, used for other operational procedures, maintenance of technical equipment and safety operations have been identified as local pollutants in Arctic military airports.[575]

Among others, PFAS were found in soil, water and biota released from firefighting training sites where PFAS-containing AFFFs were regularly used for firefighting drills at military but also civilian airport installations.[576] Elevated PFAS in local caribou was attributed to pollution from local military bases in Greenland (Kangerlussuaq) and other locations.[290] Also in the Norwegian Arctic, elevated levels of PFAS from a military air base are reported in a local coastal food web close to Bodø (Norway).[264] In an early assessment PCBs were already found as priority pollutant with potential for local pollution.[577] Even more than 50 years after the usage of PCBs as technical agent was banned and 20 years after the Stockholm Convention for the global regulation was enforced, PCBs still belong to the POPs found at the highest levels released from technical installations including FUD stations and military air fields.[212]

In addition to PCBs, PAHs are found in high levels near Arctic air fields, mainly in association with emissions for technical equipment in operation, heating and power production at the site.[212,521,574,578] Elevated PAH levels in connection with military operations and sites including airports have been reported from all Arctic counties.[578,579]

A new group of organic pollutants associated with local contamination from Arctic military air operations are OPEs used as flame retardants or additives in hydraulic fluids. In a recent study, elevated levels of OPEs were reported from the Resolute Bay military airport in Arctic Canada.[579,580]

Military exercises have also been recognized as an important local pollution source in the Arctic.[581–584] Especially large scale operations like marine joint navy exercises[585] and air combat simulations[586] are expected to affect the local environment. Firing ranges are sources of metal pollution,[587–589] and the direct impact of explosive residues on the vulnerable Arctic vegetation from military small- and large-scale exercises at shooting ranges has been reported repeatedly.[511,590,591]

Conclusions

Four initial questions were presented in the introductory section of this review as a guiding framework for this overview on infrastructure as local pollution sources in the Arctic. Based upon this literature survey, we conclude with the following answers to our questions.

(1) How important are infrastructure-related emissions as local sources for the overall contamination in the arctic with organic pollutants?

The currently available information confirms that domestic, industrial and military infrastructures are important local pollution sources for many organic pollutants, including POPs and CEACs. Impacts of elevated emissions on the local environment, organisms and inhabitants of various Arctic regions have been documented in numerous scientific reports. Life

supporting infrastructures such as power plants, heat supply, water supply and waste handling have been identified as important and nearly constant pollutant sources in a domestic context. Solid waste handling and uncontrolled waste dumps are identified as particularly important pollutant sources. The long-lasting continuous drainage of contaminants from wastes into adjacent surface and ground water system contributes to the environmental distribution of these pollutants. Local and regional industrial installations including oil exploitation, Arctic mining, metal smelters and associated support and transportation infrastructures, as well as military activities and installations have also been identified as important local sources for a variety of POPs and CEACs. Especially abandoned decommissioned locations, earlier operated as early warning systems (*i.e.* DEW) or large training sites were found to contribute to local contamination with organic pollutants. The literature review generally shows that while local inputs can be substantial, they are usually limited to the surroundings of the source. However, this does not exclude impacts on local populations who depend on natural resources of the specific area. For POPs, bioaccumulation through the food chain still has a strong exposure signal that is also based on long-range transport of contaminants. However, for less bioaccumulative and biomagnifying compounds, exposure from local sources through *e.g.* drinking water can be a concern.

(2) Can spatial and temporal trends be identified for infrastructure contribution to local pollution in the Arctic?

Although information exists on the identification, characterisation and emission profiling of Arctic organic pollution from local domestic, industrial as well as military sources, the regional and temporal resolution of the currently available scientific information is insufficient for spatial or temporal trend estimations. For trend studies, long-term contaminant monitoring needs to be established based on the pollution source, using validated sampling and analytical protocols, ideally coordinated with the ongoing monitoring of the long-range transport of POPs to the Arctic. Initial evidence suggests that most hydrocarbon-based pollutants can be degraded, even under the extreme conditions of the Arctic, although time scales will exceed those at temperate latitudes. However, the FUD examples show that many compounds are still present decades after their use. In general, given the low temperatures, the absence of light for long times, the vulnerability of the Arctic ecosystems and the dependence of local populations on natural resources, any emissions of harmful chemicals should be avoided. Pollution and exposure risks should be considered in any new infrastructure development and ensure the local perspective in environmental impact assessments, as discussed in detail in Muir *et al.*[592]

(3) Do characteristic pollution patterns and profiles exist for infrastructure-related pollutant sources in the Arctic?

According to the scientific information provided here, the major local sources for organic pollution can be associated with characteristic pollution emission profiles. The following indicator chemicals were identified:

- *Power plants and domestic heating*: PAHs, metals (Hg, Cd, Pb *etc.*), VOCs.
- *Vehicles and transportation (fossil fuel combustion)*: PAHs, other aromatic compounds, VOCs, metals.
- *Airports*: PFAS, flame retardants, technical chemicals.
- *Industry (mining, refining, fisheries, offshore oil and gas production)*: plastics, polymers, metals, PAHs, industrial chemicals such as PCBs, flame retardants, PFAS, VOCs, surfactants, anticorrosive chemicals, surfactants, *etc.*
- *Aquaculture (including infrastructures)*: plastics, pesticides, PFAS, PCBs, brominated flame retardants, veterinary pharmaceuticals, hygienic ingredients, surfactants, *etc.*
- *Military installations*: plastics, metals, PCBs, PFAS, PAHs, OPEs, surfactants.
- *Other municipal installations*: anti-corrosives, pesticides, PFAS, PCBs, flame retardants, pharmaceuticals and personal care to Arctic local pollution in the following products (PPCPs), surfactants, cosmetics, bioactive compounds, food preservatives, plastics, polymers, additives, pesticides, metals.

It should be noted that these lists of indicators are not necessarily complete, but subject to a research bias. This means that only those compounds can be identified that are selected for analysis. Consequently, other chemicals can be emitted from these sources that remain undetected because they were not included in the respective study. Emerging techniques within non-target screening and wide-scope analyses are increasingly applied in environmental research and would allow for a broader detection of potential CEACs than those typically analysed so far.

(4) How is Arctic climate change influencing local infrastructure and their associated chemical emissions?

This review has provided strong indications for the direct influence of Arctic climate change on the source strength as well as mobility of the emitted pollutants. More details are provided in a review on future developments and climate change.[592]

Perspectives and recommendations

A first comprehensive review on contributions of local sources to the presence of organic pollutants in the Arctic is presented. Domestic, industrial, military infrastructures as well as solid waste handling and storage are documented and confirmed as local pollution sources for POPs and CEACs, as documented above. To minimize potential exposure and associated hazardous effect for the environmental and the people of the North, we recommend the following:

(1) Harmonize and implement sustainable emission reduction policies for industrial activities incl. mining, shipping, tourism and energy production in the Arctic region.

(2) Encourage the adoption of cleaner technologies, such as low-sulfur fuels for ships and energy-efficient practices for industrial processes.

(3) Establish and maintain comprehensive pollutant monitoring systems to track emissions from various sources.

(4) Encourage collaboration with local communities, research institutions, and industry stakeholders to collect accurate data on emissions.

(5) Encourage sustainable practices in resource extraction, transportation, and tourism.

(6) Support research and development of alternative energy sources to reduce reliance on fossil fuels.

(7) Develop guidelines and regulations for properly managing waste generated by industrial activities, residential areas, and tourism and implement recycling programs and ensure safe disposal of hazardous materials.

(8) Encourage collaboration between neighboring countries and international organizations to address transboundary pollution.

(9) Encourage to actively participate in initiatives like the Arctic Council's Arctic Contaminants Action Program (ACAP).

(10) Invest in research to better understand local emission sources and their impacts on climate, ecosystems, and human health.

Author contributions

RK and GWG coordinated the contributions from all co-authors and wrote the draft versions. KV, LOR, AE, KBP, SC, NA, HAL, WFH, FH, DCGM, CdW, MjG, REJ, GMK, GB, SBN, JLL, YFL, ZFZ and EB contributed with text, expertise and comments on the numerous draft versions of the manuscript.

Conflicts of interest

The authors confirm that "there are no conflicts to declare".

Data availability

No new data were generated as part of this critical review. Any additional information or clarification on the reported presented here will be provided by the corresponding author upon request.

Acknowledgements

We appreciate the support and encouragement of many institutions and colleagues who contributed with data and discussions. We thank the Arctic Monitoring and Assessment Programme (AMAP) for technical and financial support. R. Kallenborn's contribution was partially supported by the EU-project "Arctic pollution in a One Health perspective – from complex challenges to sustainable solutions (ArcSolution)"; Grant No. 101135051. K. Vorkamp received funding from the program "Miljøstøtte til Arktis" (no. 2022-86245) of the Danish Environmental Protection Agency. S. Corsolini and N. Ademollo received funding from the Italian National Programme of Research in Antarctica (PNRA18-00097-B2/PNRA2018/B2Z1.02).

References

1 H. Fiedler, R. Kallenborn, J. d. Boer and L. K. Sydnes, *Chem. Int.*, 2019, **41**, 4–11.
2 H. Hung, R. Kallenborn, K. Breivik, Y. Su, E. Brorstrom-Lunden, K. Olafsdottir, J. M. Thorlacius, S. Leppanen, R. Bossi, H. Skov, S. Mano, G. W. Patton, G. Stern, E. Sverko and P. Fellin, *Sci. Total Environ.*, 2010, **408**, 2854–2873.
3 Y. F. Li, R. Kallenborn and Z. Zhang, *Environ. Sci. Ecotechnology*, 2024, **18**, 100332.
4 Z. Xie, P. Zhang, Z. Wu, S. Zhang, L. Wei, L. Mi, A. Kuester, J. Gandrass, R. Ebinghaus, R. Yang, Z. Wang and W. Mi, *Sci. Total Environ.*, 2022, **835**, 155376.
5 C. Sonne, R. Dietz, B. M. Jenssen, S. S. Lam and R. J. Letcher, *Trends Ecol. Evol.*, 2021, **36**, 421–429.
6 J. C. Gibson, *Sci. Total Environ.*, 2020, **708**, 134538.
7 S. Weinbruch, D. Wiesemann, M. Ebert, K. Schütze, R. Kallenborn and J. Ström, *Atmos. Environ.*, 2012, **49**, 142–150.
8 S. Weinbruch, N. Benker, K. Kandler, K. Schütze, K. Kling, B. Berlinger, Y. Thomassen, T. Drotikova and R. Kallenborn, *Atmos. Environ.*, 2018, **172**, 47–54.
9 M. Granberg, A. Ask and G. W. Gabrielsen, *Local Contamination in Svalbard- Overview and Suggestions for Remediation Actions*, Ecotoxicology Report 44, Norwegian polar institute, Tromsø, Norway, 2019.
10 M. Gamberg, B. Braune, E. Davey, B. Elkin, P. F. Hoekstra, D. Kennedy, C. Macdonald, D. Muir, A. Nirwal, M. Wayland and B. Zeeb, *Sci. Total Environ.*, 2005, **351–352**, 148–164.
11 S. G. Donaldson, J. Van Oostdam, C. Tikhonov, M. Feeley, B. Armstrong, P. Ayotte, O. Boucher, W. Bowers, L. Chan, F. Dallaire, R. Dallaire, E. Dewailly, J. Edwards, G. M. Egeland, J. Fontaine, C. Furgal, T. Leech, E. Loring, G. Muckle, T. Nancarrow, D. Pereg, P. Plusquellec, M. Potyrala, O. Receveur and R. G. Shearer, *Sci. Total Environ.*, 2010, **408**, 5165–5234.
12 S. G. Donaldson, M. S. Curren, B. Adlard, J. Provost, T. Leech, C. Tikhonov, M. Feeley, S. Tomlinson and R. Shearer, *Int. J. Circumpolar Health*, 2013, **72**, 23049.
13 S. Weinbruch, T. Drotikova, N. Benker and R. Kallenborn, *Particulate and Gaseous Emissions of Power Generation at Svalbard (AtmoPart)*, Sysselmannen på Svalbard, Longyearbyen, 2015.
14 S. Eckhardt, O. Hermansen, H. Grythe, M. Fiebig, K. Stebel, M. Cassiani, A. Bæcklund and S. A., *Atmos. Chem. Phys.*, 2013, **13**(16), 8401–8409.
15 C. Schroder, N. Reimer and P. Jochmann, *Ambio*, 2017, **46**, 400–409.
16 J. C. Raut, K. S. Law, T. Onishi, N. Daskalakis and L. Marelle, *Environ. Pollut.*, 2022, **298**, 118832.
17 O. Popovicheva, E. Diapouli, A. Makshtas, N. Shonija, M. Manousakas, D. Saraga, T. Uttal and K. Eleftheriadis, *Sci. Total Environ.*, 2019, **655**, 924–938.

18 A. Paxian, V. Eyring, W. Beer, R. Sausen and C. Wright, *Environ. Sci. Technol.*, 2010, **44**, 1333–1339.
19 C. L. Friedman, Y. Zhang and N. E. Selin, *Environ. Sci. Technol.*, 2014, **48**, 429–437.
20 J. S. Skaar, E. M. Raeder, J. L. Lyche, L. Ahrens and R. Kallenborn, *Environ. Sci. Pollut. Res. Int.*, 2019, **26**, 7356–7363.
21 W. F. Hartz, M. K. Bjornsdotter, L. W. Y. Yeung, A. Hodson, E. R. Thomas, J. D. Humby, C. Day, I. E. Jogsten, A. Karrman and R. Kallenborn, *Sci. Total Environ.*, 2023, **871**, 161830.
22 C. J. Young, V. I. Furdui, J. Franklin, R. M. Koerner, D. C. Muir and S. A. Mabury, *Environ. Sci. Technol.*, 2007, **41**, 3455–3461.
23 L. Ahrens, J. Rakovic, S. Ekdahl and R. Kallenborn, *Chemosphere*, 2023, **345**, 140463.
24 AMAP, *POPs and Chemicals of Emerging Arctic Concern: Influence of Climate Change*, Arctic Council of Ministers, Tromsø, Norway, 2021.
25 AMAP, *AMAP Assessment 2016: Chemicals of Emerging Arctic Concern. Arctic Monitoring and Assessment Programme (AMAP)*, Arctic monitoring and Assessment Programme, Oslo, 2017.
26 B. Bergsveinn, *Den Svarte Vikingen*, Spartacus, Oslo, 2014.
27 E. Trausti, *Hvalveiðar Við Ísland 1600-1939*, Bókaútgáfa Menningarsjóðs, Reykjavík, 1987.
28 Ministerie van Landbouw Naturbeheer en Visserij and Rossiĭskii nauchno-issledovatel'skiĭ institut kul'turnogo i prirodnogo nasledii ̆a ̆, *Conserving Our Common Heritage of the Arctic : Abstracts of the Willem Barents Memorial Arctic Conservation Symposium, Moscow, Russia, 10-14 March 1998*, Russian Research Institute of Cultural and Natural Heritage of the Ministry of Culture of the Russian Federation and the Russian Academy of Sciences, Moscow, 1998.
29 A. d. Bruijne, B. G. Baljé and W. F. J. Mörzer Bruyns, *De eerste tocht van de Willem Barents naar de Noordelijke I Jszee 1878 : de dagboeken van Antonius de Bruijne en Bastiaan Gerardus Balj'e*, Walburg Pers, Zutphen, 1985.
30 Y. van den Hurk, F. Sikstrom, L. Amkreutz, M. Bleasdale, A. Borvon, B. Ephrem, C. Fernandez-Rodriguez, H. M. B. Gibbs, L. Jonsson, A. Lehouck, J. M. Cedeira, S. Meng, R. Monge, M. Moreno, M. Nabais, C. Nores, J. A. Pis-Millan, I. Riddler, U. Schmolcke, M. Segschneider, C. Speller, M. Vretemark, S. Wickler, M. Collins, M. J. Nadeau and J. H. Barrett, *R. Soc. Open Sci.*, 2023, **10**, 230741.
31 S. Mulvad, *Katastrofen i Ishavet 1777 : beretninger fra hval- og sælfangsten ved Grønland*, Fiskeri- og Søfartsmuseet, Esbjerg, 2002.
32 X. Keighley, S. Palsson, B. F. Einarsson, A. Petersen, M. Fernandez-Coll, P. Jordan, M. T. Olsen and H. J. Malmquist, *Mol. Biol. Evol.*, 2019, **36**, 2656–2667.
33 L. Hacquebord and W. H. Vroom., *Walvisvaart in de gouden eeuw : opgravingen op Spitsbergen*, De Bataafsche Leeuw, Amsterdam, 1988 Rijksmuseum (Netherlands) and Rijksuniversiteit Groningen. Arctic Centre.
34 D. Francis, *Arctic Chase : a History of Whaling in Canada's North*, Tops'l Books, Sulhamstead United Kingdom, 1984.
35 D. Eber, *When the Whalers Were up North : Inuit Memories from the Eastern Arctic*, D.R. Godine, Boston, 1st edn, 1989.
36 M. Castellini, *Comp. Biochem. Physiol., Part A: Mol. Integr. Physiol.*, 2000, **126**, 153–159.
37 D. d. Jong, H. Kobell, M. Salieth and C. G. Zorgdrager, *Nieuwe beschryving der walvisvangst en haringvisschery*, ed. J. Roos, Amsteldam, 1791.
38 F. Nansen and O. N. Sverdrup, *Farthest North; Being the Record of a Voyage of Exploration of the Ship "Fram" 1893-96*, Harper & brothers, New York and London, Popular edn, 1898.
39 F. Guglielmo, I. Stemmler and G. Lammel, *Environ. Pollut.*, 2012, **162**, 475–481.
40 C. L. Friedman and N. E. Selin, *Environ. Sci. Technol.*, 2012, **46**, 9501–9510.
41 J. M. Ma, H. L. Hung, C. Tian and R. Kallenborn, *Nat. Clim. Change*, 2011, **1**, 255–260.
42 J. Ma, *Atmos. Chem. Phys.*, 2010, **10**, 7303–7314.
43 AMAP, *AMAP Assessment Report: Arctic Pollution Issues*, AMAP, Oslo, 1998.
44 F. Wania and D. Mackay, *Environ. Sci. Technol.*, 1996, **30**, A390–A396.
45 Y. F. Li, R. W. Macdonald, L. M. Jantunen, T. Harner, T. F. Bidleman and W. M. Strachan, *Sci. Total Environ.*, 2002, **291**, 229–246.
46 Y. F. Li and R. W. Macdonald, *Sci. Total Environ.*, 2005, **342**, 87–106.
47 Z. Zhao, Z. Xie, A. Moller, R. Sturm, J. Tang, G. Zhang and R. Ebinghaus, *Environ. Pollut.*, 2012, **170**, 71–77.
48 X. Zhang, R. Lohmann and E. M. Sunderland, *Environ. Sci. Technol.*, 2019, **53**, 12348–12356.
49 L. W. Y. Yeung, C. Dassuncao, S. Mabury, E. M. Sunderland, X. Zhang and R. Lohmann, *Environ. Sci. Technol.*, 2017, **51**, 6735–6744.
50 N. Yamashita, S. Taniyasu, G. Petrick, S. Wei, T. Gamo, P. K. Lam and K. Kannan, *Chemosphere*, 2008, **70**, 1247–1255.
51 H. Hung, A. A. Katsoyiannis, E. Brorstrom-Lunden, K. Olafsdottir, W. Aas, K. Breivik, P. Bohlin-Nizzetto, A. Sigurdsson, H. Hakola, R. Bossi, H. Skov, E. Sverko, E. Barresi, P. Fellin and S. Wilson, *Environ. Pollut.*, 2016, **217**, 52–61.
52 F. Riget, K. Vorkamp, R. Bossi, C. Sonne, R. J. Letcher and R. Dietz, *Environ. Pollut.*, 2016, **217**, 114–123.
53 H. Hung, A. A. Katsoyiannis, E. Brorstrom-Lunden, K. Olafsdottir, W. Aas, K. Breivik, P. Bohlin-Nizzetto, A. Sigurdsson, H. Hakola, R. Bossi, H. Skov, E. Sverko, E. Barresi, P. Fellin and S. Wilson, *Environ. Pollut.*, 2016, **217**, 52–61.
54 S. Donaldson, B. Adlard and J. O. Odland, *Int. J. Circumpolar Health*, 2016, **75**, 33807.
55 R. Kallenborn, L. O. Reiersen and C. D. Olseng, *Atmos. Pollut. Res.*, 2012, **3**, 485–493.
56 M. S. Olsen, T. V. Callaghan, J. D. Reist, L. O. Reiersen, D. Dahl-Jensen, M. A. Granskog, B. Goodison,

G. K. Hovelsrud, M. Johansson, R. Kallenborn, J. Key, A. Klepikov, W. Meier, J. E. Overland, T. D. Prowse, M. Sharp, W. F. Vincent and J. Walsh, *Ambio*, 2011, **40**, 111–118.

57 D. C. G. Muir and C. A. de Wit, *Sci. Total Environ.*, 2010, **408**, 3044–3051.

58 H. Hung, C. Halsall, N. Ademollo, P. W. Bartlett, S. Bengtson Nash, K. Breivik, S. Corsollini, T. Gouin, R. Guardans, K. M. Hansen, T. Harner, M. Hermanson, D. Herzke, K. Koziol, I. S. Krogseth, G. Lammel, A. Lebedev, L. Li, R. Lohmann, J. Ma, T. Huang, M. Muntean, M. McKinney, K. Pozo, J. Schuster, R. Sühring, D. Szuminska, K. Vorkamp, F. Wania, Z. Zie and Y.-M. Hsu, *Environ. Sci.: Adv.*, 2025, in preparation.

59 L. O. Reiersen, K. Vorkamp and R. Kallenborn, *Environ. Sci. Ecotechnology*, 2024, **17**, 100302.

60 H. Hung, P. Blanchard, C. J. Halsall, T. F. Bidleman, G. A. Stern, P. Fellin, D. C. Muir, L. A. Barrie, L. M. Jantunen, P. A. Helm, J. Ma and A. Konoplev, *Sci. Total Environ.*, 2005, **342**, 119–144.

61 C. J. Halsall, A. J. Sweetman, L. A. Barrie and K. C. Jones, *Atmos. Environ.*, 2001, **35**, 255–267.

62 Y. Xie, Z. Li, L. Li, R. Wagener, I. Abboud, K. Li, D. Li, Y. Zhang, X. Chen and H. Xu, *Sci. Rep.*, 2018, **8**, 9376.

63 J. Paatero, A. Ioannidou, J. Ikonen and J. Lehto, *J. Environ. Radioact.*, 2017, **172**, 10–14.

64 K. S. Law, A. Roiger, J. L. Thomas, L. Marelle, J. C. Raut, S. Dalsoren, J. Fuglestvedt, P. Tuccella, B. Weinzierl and H. Schlager, *Ambio*, 2017, **46**, 453–463.

65 V. Vestreng, R. Kallenborn and E. Økstad, *Climate Influencing Emissions, Scenarios and Mitigation Options at Svalbard*, Norwegian Climtae and Pollution Directorate, Oslo, 2009.

66 AMAP, *AMAP 2017 AMAP Assessment 2016: Chemical of Emerging Arctic Concern*, Arctic Monitoring and Assessment programme, Oslo, 2017.

67 D. Muir, X. Zhang, C. A. de Wit, K. Vorkamp and S. Wilson, *Emerging Contam.*, 2019, **5**, 201–210.

68 R. Kallenborn, E. Brorstrom-Lunden, L. O. Reiersen and S. Wilson, *Environ. Sci. Pollut. Res. Int.*, 2018, **25**, 33001–33013.

69 M. Bergheim, T. Helland, R. Kallenborn and K. Kummerer, *Chemosphere*, 2010, **81**, 1477–1485.

70 American Chemical Society, National Meeting (238th : 2009 : Washington DC), R. U. Halden, American Chemical Society. Division of Environmental Chemistry and American Chemical Society, Contaminants of emerging concern in the environment: ecological and human health considerations, American Chemical Society; Distributed by Oxford University Press, Washington DC New York, 2010.

71 Y. Zhao, L. Ye and X. X. Zhang, *Water Environ. Res.*, 2018, **90**, 1301–1322.

72 K. Kummerer, *J. Environ. Manage.*, 2009, **90**, 2354–2366.

73 M. Pucko, G. A. Stern, R. W. Macdonald, L. M. Jantunen, T. F. Bidleman, F. Wong, D. G. Barber and S. Rysgaard, *Sci. Total Environ.*, 2015, **506–507**, 444–452.

74 K. M. Hansen, J. H. Christensen and J. Brandt, *Int. J. Environ. Res. Public Health*, 2015, **12**, 11254–11268.

75 N. L. Zhu, K. W. Schramm, T. Wang, B. Henkelmann, X. Y. Zheng, J. J. Fu, Y. Gao, Y. W. Wang and G. B. Jiang, *Environ. Pollut.*, 2014, **191**, 166–174.

76 A. M. Grannas, C. Bogdal, K. J. Hageman, C. Halsall, T. Harner, H. Hung, R. Kallenborn, P. Klan, J. Klanova, R. W. Macdonald, T. Meyer and F. Wania, *Atmos. Chem. Phys.*, 2013, **13**, 3271–3305.

77 P. Carlsson, K. Breivik, E. Brorstrom-Lunden, I. Cousins, J. Christensen, J. O. Grimalt, C. Halsall, R. Kallenborn, K. Abass, G. Lammel, J. Munthe, M. MacLeod, J. O. Odland, J. Pawlak, A. Rautio, L. O. Reiersen, M. Schlabach, I. Stemmler, S. Wilson and H. Wohrnschimmel, *Environ. Sci. Pollut. Res. Int.*, 2018, **25**, 22499–22528.

78 R. Kallenborn, C. Halsall, M. Dellong and P. Carlsson, *J. Environ. Monit.*, 2012, **14**, 2854–2869.

79 AMAP, *AMAP Assessment 2020: POPs and Chemicals of Emerging Arctic Concern: Influence of Climate Change*, AMAP, Oslo, 2021.

80 K. Sundseth, J. M. Pacyna, A. Banel, E. G. Pacyna and A. Rautio, *Int. J. Environ. Res. Public Health*, 2015, **12**, 3579–3599.

81 E. Post, U. S. Bhatt, C. M. Bitz, J. F. Brodie, T. L. Fulton, M. Hebblewhite, J. Kerby, S. J. Kutz, I. Stirling and D. A. Walker, *Science*, 2013, **341**, 519–524.

82 B. Bodenhorn and O. Ulturgasheva, *Philos. Trans. R. Soc., A*, 2017, **375**, 1–13.

83 D. Muir, M. J. Gunnarsdóttir, K. Koziol, F. A. von Hippel, D. Szumińska, N. Ademollo, S. Corsolini, A. De Silva, G. Gabrielsen, R. Kallenborn, Ż. Polkowska, E. Krümmel and K. Vorkamp, *Environ. Sci.: Adv.*, 2025, **4**, 355–408.

84 T. Heleniak, *Polar Geogr.*, 2021, **44**, 136–152.

85 H. A. Conley, D. L. Pumphrey, T. M. Toland and M. David, *Arctic Economics in the 21st Century: The Benefits and Costs of Cold*, Center for Strategic and International Studies; Rowman & Littlefield, Washington, D.C., 2013.

86 S. p. Pézard, A. Tingstad, K. Van Abel, S. Stephenson and National Defense Research Institute (U.S.), *Maintaining Arctic Cooperation with Russia : Planning for Regional Change in the Far North*, RAND Corporation, Santa Monica, Calif., 2017.

87 J. Hjort, O. Karjalainen, J. Aalto, S. Westermann, V. E. Romanovsky, F. E. Nelson, B. Etzelmuller and M. Luoto, *Nat. Commun.*, 2018, **9**, 5147.

88 R. G. Barry and T.-Y. Gan, *The Global Cryosphere: Past, Present and Future*, Cambridge University Press, Cambridge, United Kingdom ; New York, N.Y., 2021.

89 V. Gordeev, *Reg. Environ. Change*, 2002, **3**, 88–98.

90 B. Schneider, O. Dellwig, K. Kuliński, A. Omstedt, F. Pollehne, G. Rehder and O. Savchuk, in *Biological Oceanography of the Baltic Sea*, Springer, 2017, pp. 87–122.

91 M. Vecchiato, E. Barbaro, A. Spolaor, F. Burgay, C. Barbante, R. Piazza and A. Gambaro, *Environ. Pollut.*, 2018, **242**, 1740–1747.

92 P. E. Jensen, D. Boratto, P. M. Rossi, M. Velmitskaya, I. B. Øverjordet, H. Ó. Andradóttir, L. T. Hansen, I. Herrmann, R. Mortensen and K. Hoydal, *Environ. Sci.: Adv.*, 2025, **4**, 1373–1402.
93 T. Lloyd-Jones, J. J. Dick, T. P. Lane, E. M. Cunningham and K. Kiriakoulakis, *Mar. Pollut. Bull.*, 2023, **196**, 115586.
94 E. M. Miholits, *Tech Rep Arct Aeromed Lab US*, 1961, 61-9, p. 112.
95 M. H. Hermanson, E. Isaksson, D. Divine, C. Teixeira and D. C. G. Muir, *Chemosphere*, 2020, **243**, 125324.
96 C. Burns, R. W. Orttung, M. Shaiman, L. Silinsky and E. Zhang, *Waste Manage.*, 2021, **126**, 340–350.
97 J. M. Pacyna, *Sci. Total Environ.*, 1995, **160–161**, 39–53.
98 M. N. Alekseeva, L. I. Svarovskaya and I. G. Yashchenko, *AIP Conf. Proc.*, 2019, **2167**, 1–6.
99 P. N. Neitlich, J. M. Ver Hoef, S. D. Berryman, A. Mines, L. H. Geiser, L. M. Hasselbach and A. E. Shiel, *PLoS One*, 2017, **12**, e0177936.
100 M. Babayev, S. L. Capozzi, P. Miller, K. R. McLaughlin, S. S. Medina, S. Byrne, G. Zheng and A. Salamova, *Environ. Pollut.*, 2022, **305**, 119246.
101 J. Sondergaard and A. Mosbech, *Sci. Total Environ.*, 2022, **812**, 152373.
102 M. Sahu, S. Hu, P. H. Ryan, G. Le Masters, S. A. Grinshpun, J. C. Chow and P. Biswas, *Sci. Total Environ.*, 2011, **409**, 2642–2651.
103 C. Oliveira, N. Martins, J. Tavares, C. Pio, M. Cerqueira, M. Matos, H. Silva, C. Oliveira and F. Camoes, *Chemosphere*, 2011, **83**, 1588–1596.
104 L. Lin, Z. H. Fan, X. Zhu, L. H. Huang and L. J. Bonanno, *J. Air Waste Manage. Assoc.*, 2011, **61**, 631–639.
105 M. M. Maricq and H. Maldonado, *J. Air Waste Manage. Assoc.*, 2010, **60**, 1165–1176.
106 L. Lin, M. L. Lee and D. J. Eatough, *J. Air Waste Manage. Assoc.*, 2010, **60**, 3–25.
107 S. Fuselli, M. De Felice, R. Morlino and L. Turrio-Baldassarri, *Int. J. Environ. Res. Public Health*, 2010, **7**, 3792–3803.
108 Y. Wang, H. Xu, Y. Li, N. Lin and P. Xu, *Chemosphere*, 2023, **340**, 139818.
109 A. Paletto, S. Bernardi, E. Pieratti, F. Teston and M. Romagnoli, *Heliyon*, 2019, **5**, e02070.
110 M. Athar, M. Ali and M. A. Khan, *Environ. Monit. Assess.*, 2010, **166**, 625–639.
111 M. S. Reddy, S. Basha, H. V. Joshi and B. Jha, *J. Hazard. Mater.*, 2005, **123**, 242–249.
112 N. Barbalic, G. Marijan and M. Maric, *Arh. Hig. Rada Toksikol.*, 2000, **51**, 217–233.
113 N. Attar and A. Onen, *J. Oral Rehabil.*, 2002, **29**, 791–798.
114 J. P. Dudley, E. P. Hoberg, E. J. Jenkins and A. J. Parkinson, *EcoHealth*, 2015, **12**, 713–725.
115 O. S. Pokrovsky, S. N. Kirpotin and A. I. Malov, *The Arctic: : Current Issues and Challenges*, Nova Science Publishers, Hauppauge, 2020.
116 M. H. Nordquist, J. N. Moore, A. E. Chircop, R. n. Long and University of Virginia. Center for Oceans Law and Policy, *Conference, the Regulation of Continental Shelf Development : Rethinking International Standards*, Martinus Nijhoff Publishers, Leiden, 2013.
117 R. Kallenborn, *Implications and Consequences of Anthropogenic Pollution in Polar Environments*, Springer, 2016.
118 K. A. Sanchez, M. Foster, M. J. Nieuwenhuijsen, A. D. May, T. Ramani, J. Zietsman and H. Khreis, *Environ. Int.*, 2020, **142**, 105826.
119 Y. Zhang, H. Zheng, L. Zhang, Z. Zhang, X. Xing and S. Qi, *Environ. Pollut.*, 2019, **246**, 319–327.
120 D. Yan, S. Wu, S. Zhou, G. Tong, F. Li, Y. Wang and B. Li, *Environ. Pollut.*, 2019, **248**, 804–814.
121 J. Kuttippurath, V. K. Patel, R. Roy and P. Kumar, *Environ. Sci. Pollut. Res.*, 2024, **31**, 1621–1636.
122 W. Zhang, J. Chen, K. Ungar and M. Cooke, *J. Environ. Radioact.*, 2015, **141**, 123–129.
123 M. H. Hermanson, E. Isaksson, S. Forsstrom, C. Teixeira, D. C. Muir, V. A. Pohjola and R. S. van de Wal, *Environ. Sci. Technol.*, 2010, **44**, 7405–7410.
124 K. S. Law and A. Stohl, *Science*, 2007, **315**, 1537–1540.
125 N. Z. Heidam, J. Christensen, P. Wahlin and H. Skov, *Sci. Total Environ.*, 2004, **331**, 5–28.
126 A. V. Polissar, P. K. Hopke and J. M. Harris, *Environ. Sci. Technol.*, 2001, **35**, 4214–4226.
127 R. A. Kerr, *Science*, 1979, **205**, 290–293.
128 C. Y. Huo, W. L. Li, L. Y. Liu, Y. Sun, J. Q. Guo, L. Wang, H. Hung and Y. F. Li, *Sci. Total Environ.*, 2023, **863**, 160852.
129 H. Routti, T. C. Atwood, T. Bechshoft, A. Boltunov, T. M. Ciesielski, J. P. Desforges, R. Dietz, G. W. Gabrielsen, B. M. Jenssen, R. J. Letcher, M. A. McKinney, A. D. Morris, F. F. Riget, C. Sonne, B. Styrishave and S. Tartu, *Sci. Total Environ.*, 2019, **664**, 1063–1083.
130 M. A. McKinney, S. J. Iverson, A. T. Fisk, C. Sonne, F. F. Riget, R. J. Letcher, M. T. Arts, E. W. Born, A. Rosing-Asvid and R. Dietz, *Glob. Change Biol.*, 2013, **19**, 2360–2372.
131 R. J. Letcher, J. O. Bustnes, R. Dietz, B. M. Jenssen, E. H. Jorgensen, C. Sonne, J. Verreault, M. M. Vijayan and G. W. Gabrielsen, *Sci. Total Environ.*, 2010, **408**, 2995–3043.
132 Arctic Monitoring and Assessment Programme, *AMAP Assessment Report : Arctic Pollution Issues*, The Programme, Oslo, Norway, 1998.
133 D. Camenzuli and B. L. Freidman, *Polar Res.*, 2015, **34**, 24492.
134 M. Ji, A. F. Smith, J. E. Rattray, W. E. England and C. R. J. Hubert, *Mar. Pollut. Bull.*, 2023, **196**, 115557.
135 L. Schreiber, B. Hunnie, I. Altshuler, E. Gongora, M. Ellis, C. Maynard, J. Tremblay, J. Wasserscheid, N. Fortin, K. Lee, G. Stern and C. W. Greer, *Environ. Res.*, 2023, **233**, 116421.
136 S. C. Chen, F. Musat, H. H. Richnow and M. Kruger, *J. Environ. Sci.*, 2024, **146**, 283–297.
137 M. Guggisberg, P. A. Hessel, D. Michaelchuk and M. Atiemo, *Int. J. Circumpolar Health*, 2003, **62**, 120–129.
138 A. Cavalerie, D. Dumas and L. Gosselin, *Build. Environ.*, 2024, **253**, 111300.

139 M. Kotol, C. Rode, G. Clausen and T. R. Nielsen, *Build. Environ.*, 2014, **81**, 29–36.
140 A. Jadwiga Krzyszowska, *Polar Res.*, 1989, **7**, 119–131.
141 M. Marques, J. Sierra, T. Drotikova, M. Mari, M. Nadal and J. L. Domingo, *Environ. Res.*, 2017, **159**, 202–211.
142 T. Drotikova, A. M. Ali, A. K. Halse, H. C. Reinardy and R. Kallenborn, *Atmos. Chem. Phys.*, 2020, **20**, 9997–10014.
143 T. Drotikova, A. Dekhtyareva, R. Kallenborn and A. Albinet, *Atmos. Chem. Phys.*, 2021, **21**, 14351–14370.
144 M. Vecchiato, C. Barbante, E. Barbaro, F. Burgay, W. R. L. Cairns, A. Callegaro, D. Cappelletti, F. Dallo, M. D'Amico, M. Feltracco, J.-C. Gallet, A. Gambaro, C. Larose, N. Maffezzoli, M. Mazzola, I. Sartorato, F. Scoto, C. Turetta, M. Vardè, Z. Xie and A. Spolaor, *Environ. Pollut.*, 2024, **340**, 122864.
145 Society of Automotive Engineers, *Evaluations in Two-Stroke Engines and Their Emissions*, Society of Automotive Engineers, Warrendale, PA, 1995.
146 S. Reimann, R. Kallenborn and N. Schmidbauer, *Environ. Sci. Technol.*, 2009, **43**, 4791–4795.
147 M. P. Nuti, *Emissions from Two-Stroke Engines*, Society of Automotive Engineers, Warrendale, PA, 1998.
148 D. D. Shively, B. M. Pape, R. N. Mower, Y. Zhou, R. Russo and B. C. Sive, *Environ. Manag.*, 2008, **41**, 183–199.
149 J. S. Meldrum, *Clean Snowmobile Challenge : the Revival of the 2-stroke Engine and Studying Flex Fuel Engines*, SAE International, Warrendale, PA, 2017, vol. 2.
150 J. J. White, J. N. Carroll and Montana, *Emissions from Snowmobile Engines Using Bio-Based Fuels and Lubricants: Final Report*, Department of Environmental Quality, Southwest Research Institute, (6220 Culebra Road, P.O. Box Drawer 28510 San Antonio 78228-0510) San Antonio, Texas, 1998.
151 K. J. Walus, L. Wargula, P. Krawiec and J. M. Adamiec, *Environ. Sci. Pollut. Res. Int.*, 2018, **25**, 3243–3259.
152 T. D. Gordon, D. S. Tkacik, A. A. Presto, M. Zhang, S. H. Jathar, N. T. Nguyen, J. Massetti, T. Truong, P. Cicero-Fernandez, C. Maddox, P. Rieger, S. Chattopadhyay, H. Maldonado, M. M. Maricq and A. L. Robinson, *Environ. Sci. Technol.*, 2013, **47**, 14137–14146.
153 G. A. Bishop, R. Stadtmuller, D. H. Stedman and J. D. Ray, *J. Air Waste Manage. Assoc.*, 2009, **59**, 936–942.
154 R. C. Musselman and J. L. Korfmacher, *Environ. Monit. Assess.*, 2007, **133**, 321–334.
155 K. Eriksson, D. Tjarner, I. Marqvardsen and B. Jarvholm, *Chemosphere*, 2003, **50**, 1343–1347.
156 S. Reimann, R. Kallenborn and N. Schmidbauer, *Environ. Sci. Technol.*, 2009, **43**, 4791–4795.
157 R. Kallenborn, N. Schmidbauer and S. Reimannn, *Volatile Environmental Pollutants on Svalbard (VETAPOS)*, Report ISBM 978-82-425-2408-9, Norwegian institute for Air Research (NILU), Longyearbyen, 2011.
158 B. C. Barr, H. Ó. Andradóttir, T. Thorsteinsson and S. Erlingsson, *Sustainability*, 2021, **13**, 9607.
159 J. Cangola, F. K. Abagale and S. J. Cobbina, *Sci. Total Environ.*, 2024, **911**, 168633.
160 H. T. Duong, K. Kadokami, D. T. Nguyen, H. T. Trinh, N. H. Doan, H. Mizukawa and S. Takahashi, *Environ. Sci. Pollut. Res. Int.*, 2023, **30**, 34814–34826.
161 X. Tong, S. Mohapatra, J. Zhang, N. H. Tran, L. You, Y. He and K. Y. Gin, *Water Res.*, 2022, **217**, 118418.
162 Y. Choi, K. Kim, D. Kim, H. B. Moon and J. Jeon, *Environ. Pollut.*, 2020, **258**, 113792.
163 S. Huber, M. Remberger, L. Kaj, M. Schlabach, H. O. Jorundsdottir, J. Vester, M. Arnorsson, I. Mortensen, R. Schwartson and M. Dam, *Sci. Total Environ.*, 2016, **562**, 13–25.
164 T. Vasskog, T. Anderssen, S. Pedersen-Bjergaard, R. Kallenborn and E. Jensen, *J. Chromatogr. A*, 2008, **1185**, 194–205.
165 R. Kallenborn, T. Eggen, O. Bergersen, T. Vasskog, E. Jensen, A. Längin, K. Kümmerer, C. Dye, M. Schlabach and E. S. Heimstad, *Pharmaceutical Residues in Norwegian Sewage Treatment Plants and the Adjacent Aqueous Environment under Different Climate Regimes (Pharmafate); Final Report to the Research Council of Norway (RCN)*. Norwegian Institute for Air Research (NILU), 2008.
166 J. Glüge, M. Scheringer, I. T. Cousins, J. C. DeWitt, G. Goldenman, D. Herzke, R. Lohmann, C. A. Ng, X. Trier and Z. Wang, *Environ. Sci.: Processes Impacts*, 2020, **22**, 2345–2373.
167 A. M. Ali, M. Sanden, C. P. Higgins, S. E. Hale, W. M. Alarif, S. S. Al-Lihaibi, E. M. Raeder, H. A. Langberg and R. Kallenborn, *Environ. Pollut.*, 2021, **280**, 116935.
168 A. M. Ali, H. A. Langberg, S. E. Hale, R. Kallenborn, W. F. Hartz, A. K. Mortensen, T. M. Ciesielski, C. A. McDonough, B. M. Jenssen and G. D. Breedveld, *Environ. Sci.: Processes Impacts*, 2021, **23**, 588–604.
169 K. M. Stroski, K. H. Luong, J. K. Challis, L. G. Chaves-Barquero, M. L. Hanson and C. S. Wong, *Sci. Total Environ.*, 2020, **708**, 134494.
170 M. Shoeib, T. Harner, M. Ikonomou and K. Kannan, *Environ. Sci. Technol.*, 2004, **38**, 1313–1320.
171 J. Garcia-Barrios, M. Drysdale, M. Ratelle, E. Gaudreau, A. LeBlanc, M. Gamberg and B. D. Laird, *Int. J. Hyg. Environ. Health*, 2021, **235**, 113754.
172 R. Grønnestad, B. P. Vázquez, A. Arukwe, V. L. B. Jaspers, B. M. Jenssen, M. Karimi, J. L. Lyche and Å. Krøkje, *Environ. Sci. Technol.*, 2019, **53**, 13390–13397.
173 M. Chropeňová, P. Karásková, R. Kallenborn, E. K. Gregušková and P. Čupr, *Environ. Sci. Technol.*, 2016, **50**, 9487–9496.
174 D. Muir, R. Bossi, P. Carlsson, M. Evans, A. De Silva, C. Halsall, C. Rauert, D. Herzke, H. Hung, R. Letcher, F. Rigét and A. Roos, *Emerging Contam.*, 2019, **5**, 240–271.
175 M. K. Bjornsdotter, W. F. Hartz, R. Kallenborn, I. E. Jogsten, J. D. Humby, A. Karrman and L. W. Y. Yeung, *Environ. Sci. Technol.*, 2021, **55**, 15853–15861.
176 W. F. Hartz, M. K. Bjornsdotter, L. W. Y. Yeung, J. D. Humby, S. Eckhardt, N. Evangeliou, I. Ericson Jogsten, A. Karrman and R. Kallenborn, *Environ. Sci. Technol.*, 2024, **58**, 21817–21828.

177 A. Evenset, G. N. Christensen and R. Palerud, *Miljøgifter I Marine Sedimenter I Isfjorden*, ed. E. Svalbard, Akvaplan-niva, Tromsø, 2009.
178 R. W. Macdonal, L. A. Barrie, T. F. Bidleman, M. L. Diamond, D. J. Gregor, R. G. Semkin, W. M. Strachan, Y. F. Li, F. Wania, M. Alaee, L. B. Alexeeva, S. M. Backus, R. Bailey, J. M. Bewers, C. Gobeil, C. J. Halsall, T. Harner, J. T. Hoff, L. M. Jantunen, W. L. Lockhart, D. Mackay, D. C. Muir, J. Pudykiewicz, K. J. Reimer, J. N. Smith and G. A. Stern, *Sci. Total Environ.*, 2000, **254**, 93–234.
179 D. C. Muir, R. Wagemann, B. T. Hargrave, D. J. Thomas, D. B. Peakall and R. J. Norstrom, *Sci. Total Environ.*, 1992, **122**, 75–134.
180 B. R. Simoneit, *Environ. Sci. Pollut. Res. Int.*, 1999, **6**, 159–169.
181 K. Hylland, *J. Toxicol. Environ. Health, Part A*, 2006, **69**, 109–123.
182 I. A. Nemirovskaya and A. V. Khramtsova, *Mar. Pollut. Bull.*, 2022, **185**, 114229.
183 Y. Ma, C. J. Halsall, Z. Xie, D. Koetke, W. Mi, R. Ebinghaus and G. Gao, *Environ. Pollut.*, 2017, **227**, 498–504.
184 M. A. Nguyen, L. Ahrens, S. Josefsson, J. Gustavsson, H. Laudon and K. Wiberg, *Environ. Pollut.*, 2023, **333**, 121992.
185 D. E. Lakhmanov, A. Y. Kozhevnikov, S. A. Pokryshkin, I. P. Semiletov and D. S. Kosyakov, *Mar. Pollut. Bull.*, 2022, **180**, 113741.
186 A. Pouch, A. Zaborska, M. Mazurkiewicz, A. Winogradow and K. Pazdro, *Mar. Pollut. Bull.*, 2021, **164**, 111980.
187 J. Luo, Y. Han, Y. Zhao, Y. Huang, X. Liu, S. Tao, J. Liu, T. Huang, L. Wang, K. Chen and J. Ma, *Environ. Pollut.*, 2020, **261**, 114186.
188 Y. Yu, A. Katsoyiannis, P. Bohlin-Nizzetto, E. Brorstrom-Lunden, J. Ma, Y. Zhao, Z. Wu, W. Tych, D. Mindham, E. Sverko, E. Barresi, H. Dryfhout-Clark, P. Fellin and H. Hung, *Environ. Sci. Technol.*, 2019, **53**, 2375–2382.
189 J. E. Balmer, H. Hung, Y. Yu, R. J. Letcher and D. C. G. Muir, *Emerging Contam.*, 2019, **5**, 128–142.
190 H. Bradley and S. Stein, *Econ. Anthropol.*, 2022, **9**, 207–222.
191 A. Unc, D. Altdorff, E. Abakumov, S. Adl, S. Baldursson, M. Bechtold, D. J. Cattani, L. G. Firbank, S. Grand, M. Guðjónsdóttir, C. Kallenbach, A. J. Kedir, P. Li, D. B. McKenzie, D. Misra, H. Nagano, D. A. Neher, J. Niemi, M. Oelbermann, J. Overgård Lehmann, D. Parsons, S. Quideau, A. Sharkhuu, B. Smreczak, J. Sorvali, J. D. Vallotton, J. K. Whalen, E. H. Young, M. Zhang and N. Borchard, *Front. Sustainable Food Syst.*, 2021, **5**, 1–18.
192 T. Klöffel, E. H. Young, N. Borchard, J. D. Vallotton, E. Nurmi, N. J. Shurpali, F. Urbano Tenorio, X. Liu, G. H. F. Young and A. Unc, *Agric. Syst.*, 2022, **203**, 103507.
193 M. Grimes, J. L. Carrivick, M. W. Smith and A. J. Comber, *Sci. Rep.*, 2024, **14**, 3120.
194 E. Uleberg, I. Hanssen-Bauer, B. van Oort and S. Dalmannsdottir, *Clim. Change*, 2014, **122**, 27–39.
195 J. E. Balmer, A. D. Morris, H. Hung, L. Jantunen, K. Vorkamp, F. Rigét, M. Evans, M. Houde and D. C. G. Muir, *Emerging Contam.*, 2019, **5**, 70–88.
196 M. Glasius, J. H. Christensen, J. Platz and K. Vorkamp, *J. Environ. Monit.*, 2005, **7**, 127–131.
197 P. W. Bartlett, E. Isaksson and M. H. Hermanson, *Emerging Contam.*, 2019, **5**, 9–14.
198 A. Evenset, G. N. Christensen and R. Palerud, *Miljøgifter I Marine Sedimenter, Isfjorden, Svalbard*, Akvaplan-niva, Tromsø, 2006.
199 G. W. Gabrielsen, A. Evenset, S. Frantzen, J. P. Gwynn, I. Gammelsæter, R. Kallenborn, K. A. Pfaffhuber, H. Routti and K. Sagerup, *MOSJ Status Report for Environmental Pollutants in 2011*, Report ISBN 978-82-7666-309-9, Norwegian polar institute Tromsø, 2011.
200 M. Jartun, T. Volden and R. T. Ottesen, *PCB fra lokale kilder in Barentsburg, Pyramiden og Longyearbyen på Svalbrd*, Norwegian Geologiske Undersøkelser (NGU), Trondheim, Norway, 2007.
201 H. Pedersen, R. Kallenborn, R. Ottesen, G. Gabrielsen, C. Schrum, A. Evenset, A. Ruus, H. Benjaminsen, K. Sagerup and G. Christensen, *PCBs in Svalbard: Status of knowledge and management*, Report 1/2008, Governor of Svalbard and Norwegian Pollution Control Authority, Longyearbyen, Norway, 2008.
202 A. Evenset and R. Ottesen, *Environmental contaminants in marine sediments from Isfjorden and Adventfjorden, Svalbard*, Akvaplan-niva Report 3943.01, Akvaplan-niva, Tromsø, Norway, 2009.
203 A. Evenset, G. N. Christensen and R. Palerud, *Miljøgifter I Marine Sedimenter, Isfjorden, Svalbard 2005*, Akvaplan-niva, Tromsø, 2006.
204 S. Cochrane, K. Næs, J. Caroll, H. C. Trannum, R. Johansen and S. Dahle, *Marin Miljøundersøkelse Ved Bosetningene Barentsburg, Longyearbyen, Og Pyramiden I Isfjorden, Svalbard.*, Akvaplan-niva, Tromsø, 2001.
205 P. Carlsson, B. Vrana, J. Sobotka, K. Borgå, P. Bohlin Nizzetto and Ø. Varpe, *Investigation of New Brominated and Organophosphorous Flame Retardants in Svalbard Benthic Marine Food Web*, Norqwegian Institute of Water Research (NIVA), Oslo, 2018.
206 A. Olsson, *Master Thesis*, Lund University, 2016.
207 K. Wiberg, P. L. Andersson, H. Berg, P. E. Olsson and P. Haglund, *Environ. Toxicol. Chem.*, 2006, **25**, 1465–1473.
208 J. Fu, K. Fu, B. Hu, W. Zhou, Y. Fu, L. Gu, Q. Zhang, A. Zhang, J. Fu and G. Jiang, *Environ. Sci. Technol.*, 2023, **57**, 1919–1929.
209 K. Y. Kwok, E. Yamazaki, N. Yamashita, S. Taniyasu, M. B. Murphy, Y. Horii, G. Petrick, R. Kallerborn, K. Kannan, K. Murano and P. K. Lam, *Sci. Total Environ.*, 2013, **447**, 46–55.
210 D. Schindler, J. Kalff, H. Welch, G. Brunskill, H. Kling and N. Kritsch, *J. Fish. Board Can.*, 1974, **31**, 647–662.
211 M. S. V. Douglas and J. P. Smol, *Hydrobiologia*, 2000, **431**, 193–204.

212 A. Cabrerizo, D. C. G. Muir, A. O. De Silva, X. Wang, S. F. Lamoureux and M. J. Lafrenière, *Environ. Sci. Technol.*, 2018, **52**, 14187–14197.
213 N. L. Stock, V. I. Furdui, D. C. Muir and S. A. Mabury, *Environ. Sci. Technol.*, 2007, **41**, 3529–3536.
214 G. L. Lescord, K. A. Kidd, A. O. De Silva, M. Williamson, C. Spencer, X. Wang and D. C. Muir, *Environ. Sci. Technol.*, 2015, **49**, 2694–2702.
215 J. J. MacInnis, I. Lehnherr, D. C. G. Muir, R. Quinlan and A. O. De Silva, *Sci. Total Environ.*, 2019, **666**, 414–422.
216 Y. Sun, A. O. De Silva, K. A. St Pierre, D. C. G. Muir, C. Spencer, I. Lehnherr and J. J. MacInnis, *Environ. Sci. Technol.*, 2020, **54**, 2734–2743.
217 R. Sühring, M. L. Diamond, M. Scheringer, F. Wong, M. Pućko, G. Stern, A. Burt, H. Hung, P. Fellin, H. Li and L. M. Jantunen, *Environ. Sci. Technol.*, 2016, **50**, 7409–7415.
218 M. C. Bartley, T. Tremblay, A. O. De Silva, C. Michelle Kamula, S. Ciastek and Z. Z. A. Kuzyk, *Environ. Sci. Ecotechnology*, 2024, **18**, 100313.
219 C. Chen, A. Chen, F. Zhan, F. Wania, S. Zhang, L. Li and J. Liu, *Environ. Sci. Technol.*, 2022, **56**, 7895–7904.
220 N. Facciola, S. Pedro, M. Houde, A. T. Fisk, S. H. Ferguson, H. Steer, D. C. G. Muir and M. A. McKinney, *Environ. Toxicol. Chem.*, 2021, **40**, 2990–2999.
221 L. Girones, Y. Guida, A. L. Oliva, J. P. Machado Torres, J. E. Marcovecchio, W. Vetter and A. H. Arias, *Chemosphere*, 2023, **328**, 138575.
222 H. Li, J. Fu, W. Pan, P. Wang, Y. Li, Q. Zhang, Y. Wang, A. Zhang, Y. Liang and G. Jiang, *MethodsX*, 2018, **5**, 939–943.
223 T. A. Dick, C. P. Gallagher and G. T. Tomy, *World Rev. Sci. Technol. Sustain. Dev.*, 2010, **7**, 387–401.
224 L. Saborido Basconcillo, S. M. Backus, D. J. McGoldrick, D. Zaruk, E. Sverko and D. C. G. Muir, *Chemosphere*, 2015, **127**, 93–100.
225 H. Li, J. Fu, W. Pan, P. Wang, Y. Li, Q. Zhang, Y. Wang, A. Zhang, Y. Liang and G. Jiang, *Sci. Total Environ.*, 2017, **590–591**, 163–170.
226 H. Zhang, Y. Shen, W. Liu, Z. He, J. Fu, Z. Cai and G. Jiang, *Environ. Pollut.*, 2019, **253**, 831–840.
227 J. E. Balmer, H. Hung, K. Vorkamp, R. J. Letcher and D. C. G. Muir, *Emerging Contam.*, 2019, **5**, 116–122.
228 H. L. Halvorsen, K. A. Pfaffhuber, M. Nipen, P. Bohlin-Nizzetto, T. F. Berglen, V. Nikiforov and W. F. Hartz, *Monitoring of Environmental Contaminants in Air and Precipitation. Annual Report 2022*, E. Chemistry Report ISBN978-82-425-3133-9, Norwegian Institute for Air research (NILU), Kjeller, Norway, 2023.
229 AMAP, *AMAP Assessment 2009:Human Health in the Arctic*, Arctic Monitoring and Assessment programme (AMAP), Oslo, 2009.
230 S. Pakhomova, A. Berezina, I. Zhdanov, O. Mekhova, A. Ilinskaya, A. Golyakov, T. Polivanova, A. Gebruk, A. L. Lusher and E. Yakushev, *Mar. Pollut. Bull.*, 2024, **207**, 116803.
231 J. Mu, L. Qu, F. Jin, S. Zhang, C. Fang, X. Ma, W. Zhang, C. Huo, Y. Cong and J. Wang, *Environ. Pollut.*, 2019, **245**, 122–130.
232 M. L. Mallory, J. Baak, C. Gjerdrum, O. E. Mallory, B. Manley, C. Swan and J. F. Provencher, *Sci. Total Environ.*, 2021, **783**, 146971.
233 M. L. Haarr, L. Bach, C. P. Chambers, J. Falk-Andersson, T. Juul-Pedersen, R. D. Metcalfe, A. Sinisalo, J. Strand, H. Svendsen, J. E. Baak, H. H. Bjornsdottir, E. M. K. Brenner, S. Christiansen, C. Delattre, M. Gauthier, R. H. Georgiou, L. Gunther, F. Hagg, U. Markussen, K. B. Parga Martinez, A. le Pevedic, A. Refosco, B. K. R. Tandberg, F. Tulatz, S. Van Broeck, N. T. Visser and L. Wittwer, *Mar. Pollut. Bull.*, 2023, **191**, 114914.
234 N. Filimonova and S. J. Birchall, *J. Environ. Manage.*, 2024, **371**, 123111.
235 M. Bergmann, F. Collard, J. Fabres, G. W. Gabrielsen, J. F. Provencher, C. M. Rochman, E. van Sebille and M. B. Tekman, *Nat. Rev. Earth Environ.*, 2022, **3**, 323–337.
236 O. Latva and N. Tynkkynen, in *Cold Waters: Tangible and Symbolic Seascapes of the North*, ed. M. Lehtimäki, A. Rosenholm, E. Trubina and N. Tynkkynen, Springer International Publishing, Cham, 2022, pp. 3–17, DOI: 10.1007/978-3-031-10149-6_1.
237 Z. Li, Y. Yang, X. Chen, Y. He, N. Bolan, J. Rinklebe, S. S. Lam, W. Peng and C. Sonne, *Chemosphere*, 2023, **313**, 137637.
238 L. Sorensen, S. Schaufelberger, A. Igartua, T. R. Storseth and I. B. Overjordet, *Sci. Total Environ.*, 2023, **864**, 161056.
239 C. J. Rhodes, *Sci. Prog.*, 2018, **101**, 207–260.
240 D. S. Kosson, H. A. van der Sloot and T. T. Eighmy, *J. Hazard. Mater.*, 1996, **47**, 43–75.
241 Y. Xu, X. Xu, L. Wen and Y. Wang, *Waste Manag.*, 2025, **203**, 114898.
242 Z. Guo, S. Sun, R. Cao, N. Jiang, N. Liang, Z. Wang, J. Guo, M. Li, X. Liu, N. Geng and J. Chen, *J. Hazard. Mater.*, 2025, **498**, 139833.
243 S. Chen, H. Wang, D. Zhang, M. Lan, Z. Sun, Y. Gao and H. Cao, *Sci. Rep.*, 2025, **15**, 4169.
244 M. Skirtun, M. Sandra, W. J. Strietman, S. W. K. van den Burg, F. De Raedemaecker and L. I. Devriese, *Mar. Pollut. Bull.*, 2022, **174**, 113178.
245 A. Kenzhegaliyeva and H. B. Lund, *Geoforum*, 2024, **150**, 103969.
246 J. P. Rodrigues, A. C. Duarte, J. Santos-Echeandía and T. Rocha-Santos, *TrAC, Trends Anal. Chem.*, 2019, **111**, 252–260.
247 L. M. Ziccardi, A. Edgington, K. Hentz, K. J. Kulacki and S. Kane Driscoll, *Environ. Toxicol. Chem.*, 2016, **35**, 1667–1676.
248 T. Noss, *FME NTRANS Annual Report 2025*, Norwegian University of Science and Technology (NTNU), Trondheim, Norway, 2025.
249 E. Evans, L. Jantunen and J. Aherne, *PeerJ*, 2025, **13**, e20237.
250 R. Gunnarsdóttir, P. D. Jenssen, P. Erland Jensen, A. Villumsen and R. Kallenborn, *Ecol. Eng.*, 2013, **50**, 76–85.

251 T. Lloyd-Jones, J. J. Dick, T. P. Lane, E. M. Cunningham and K. Kiriakoulakis, *Mar. Pollut. Bull.*, 2023, **196**, 115586.
252 N. Singh, M. Granberg, F. Collard, G. Caruso, Z. Lu, T. Kögel and G. W. Gabrielsen, *The State of Environmental Science in Svalbard–An Annual Report*, 2021, pp. 118–141.
253 M. Granberg, L. W. von Friesen, L. Bach, F. Collard, J. Strand and G. W. Gabrielsen, *Anthropogenic microlitter in wastewater and marine samples from Ny-Ålesund, Barentsburg and Signehamna*, Report C 373, IVL Swedish Environmental Research Institute, Svalbard, 2019.
254 B. Rosso, F. Scoto, I. G. Hallanger, C. Larose, J. C. Gallet, A. Spolaor, B. Bravo, C. Barbante, A. Gambaro and F. Corami, *J. Hazard. Mater.*, 2024, **467**, 133723.
255 A. L. Lusher, J. F. Provencher, J. E. Baak, B. M. Hamilton, K. Vorkamp, I. G. Hallanger, L. Pijogge, M. Liboiron, M. P. T. Bourdages, S. Hammer, M. Gavrilo, J. C. Vermaire, J. F. Linnebjerg, M. L. Mallory and G. W. Gabrielsen, *Arct. Sci.*, 2022, **8**, 1217–1235.
256 M. P. T. Bourdages, *Plastic ingestion, retention, and transport in animals from the eastern Canadian Arctic*, Carleton University, 2020.
257 V. Shruti and G. Kutralam-Muniasamy, *Trends Environ. Anal. Chem.*, 2023, **38**, e00203.
258 J. F. Provencher, S. Aliani, M. Bergmann, M. Bourdages, L. Buhl-Mortensen, F. Galgani, A. Gomiero, M. Granberg, B. E. Grøsvik and B. M. Hamilton, *Arct. Sci.*, 2022, **9**, 209–226.
259 E. Farmen, J. F. Provencher, S. Aliani, J. E. Baak, M. Bergmann, A. M. Booth, M. P. T. Bourdages, L. Buhl-Mortensen, L. Feld and G. W. Gabrielsen, *AMAP Litter and Microplastics: Monitoring Guidelines. Version 1.0, Arctic Monitoring and Assessment Programme (AMAP)*, 2021, p. 266.
260 B. M. Hamilton, L. Jantunen, M. Bergmann, K. Vorkamp, J. Aherne, K. Magnusson, D. Herzke, M. Granberg, I. G. Hallanger, A. Gomiero and I. Peeken, *Arct. Sci.*, 2022, **8**, 1116–1126.
261 I. Peeken, S. Primpke, B. Beyer, J. Gütermann, C. Katlein, T. Krumpen, M. Bergmann, L. Hehemann and G. Gerdts, *Nat. Commun.*, 2018, **9**, 1505.
262 S. Gulas, M. Downton, K. D'Souza, K. Hayden and T. R. Walker, *Mar. Pol.*, 2017, **75**, 53–61.
263 J. Seow, *Fire Fighting Foams with Perfluorochemicals-Environmental Review*, Hemming Information Services, London, UK, 2013.
264 H. A. Langberg, G. D. Breedveld, H. M. Grønning, M. Kvennås, B. M. Jenssen and S. E. Hale, *Environ. Sci. Technol.*, 2019, **53**, 10951–10960.
265 K. Prevedouros, I. T. Cousins, R. C. Buck and S. H. Korzeniowski, *Environ. Sci. Technol.*, 2006, **40**, 32–44.
266 S. Banzhaf, M. Filipovic, J. Lewis, C. J. Sparrenbom and R. Barthel, *Ambio*, 2017, **46**, 335–346.
267 X. C. Hu, D. Q. Andrews, A. B. Lindstrom, T. A. Bruton, L. A. Schaider, P. Grandjean, R. Lohmann, C. C. Carignan, A. Blum, S. A. Balan, C. P. Higgins and E. M. Sunderland, *Environ. Sci. Technol. Lett.*, 2016, **3**, 344–350.
268 S. A. Milley, I. Koch, P. Fortin, J. Archer, D. Reynolds and K. P. Weber, *J. Environ. Manage.*, 2018, **222**, 122–131.
269 M. E. Andersen, B. Hagenbuch, U. Apte, J. C. Corton, T. Fletcher, C. Lau, W. L. Roth, B. Staels, G. L. Vega, H. J. Clewell 3rd and M. P. Longnecker, *Toxicology*, 2021, **459**, 152845.
270 L. M. Nunes, Y. G. Zhu, T. Y. Stigter, J. P. Monteiro and M. R. Teixeira, *J. Environ. Monit.*, 2011, **13**, 3026–3039.
271 M. Xu, J. Legradi and P. Leonards, *Sci. Total Environ.*, 2023, **887**, 163770.
272 Z. S. Ulhaq, D. A. T. Boncan, T. F. Chan and W. K. F. Tse, *Sci. Total Environ.*, 2023, **904**, 166833.
273 N. Hamid, M. Junaid, M. Sultan, S. T. Yoganandham and O. M. Chuan, *Water Res.*, 2024, **250**, 121044.
274 H. A. Langberg, G. D. Breedveld, R. Kallenborn, A. M. Ali, S. Choyke, C. A. McDonough, C. P. Higgins, B. M. Jenssen, M. Jartun and I. Allan, *Environ. Int.*, 2024, **190**, 108844.
275 H. A. Langberg, S. E. Hale, G. D. Breedveld, B. M. Jenssen and M. Jartun, *Environ. Sci.: Processes Impacts*, 2022, **24**, 330–342.
276 N. A. Warner, K. Sagerup, S. Kristoffersen, D. Herzke, G. W. Gabrielsen and B. M. Jenssen, *Sci. Total Environ.*, 2019, **667**, 638–647.
277 N. A. Warner, K. Sagerup, S. Kristoffersen, D. Herzke, G. W. Gabrielsen and B. M. Jenssen, *Sci. Total Environ.*, 2019, **667**, 638–647.
278 S. Hansen, R. Vestergren, D. Herzke, M. Melhus, A. Evenset, L. Hanssen, M. Brustad and T. M. Sandanger, *Environ. Int.*, 2016, **94**, 272–282.
279 S. Hale, H. Arp, S. GA, W. EJ, B. K, G. Breetveld, S. F, M. Moe and A. Høisæter, *Chemosphere*, 2017, **171**, 9.
280 E. F. Houtz, C. P. Higgins, J. A. Field and D. L. Sedlak, *Environ. Sci. Technol.*, 2013, **47**, 8187–8195.
281 D. Schrenk, M. Bignami, L. Bodin, J. K. Chipman, J. Del Mazo, B. Grasl-Kraupp, C. Hogstrand, L. Hoogenboom, J. C. Leblanc, C. S. Nebbia, E. Nielsen, E. Ntzani, A. Petersen, S. Sand, T. Schwerdtle, C. Vleminckx and H. Wallace, *EFSA J.*, 2020, **18**, 391.
282 E. P. o. C. i. t. F. Chain, D. Schrenk, M. Bignami, L. Bodin, J. K. Chipman, J. Del Mazo, B. Grasl-Kraupp, C. Hogstrand, L. R. Hoogenboom, J. C. Leblanc, C. S. Nebbia, E. Nielsen, E. Ntzani, A. Petersen, S. Sand, C. Vleminckx, H. Wallace, L. Barregard, S. Ceccatelli, J. P. Cravedi, T. I. Halldorsson, L. S. Haug, N. Johansson, H. K. Knutsen, M. Rose, A. C. Roudot, H. Van Loveren, G. Vollmer, K. Mackay, F. Riolo and T. Schwerdtle, *EFSA J.*, 2020, **18**, e06223.
283 G. D. Breedveld, G. Olstad and P. Aagaard, *Biorem. J.*, 1997, **1**, 77–88.
284 M. Børresen, G. D. Breedveld and A. G. Rike, *Cold Reg. Sci. Technol.*, 2003, **37**, 137–149.
285 A. J. Krzyszowska, *Polar Res.*, 1989, **7**, 12.
286 F. Steenhuisen and M. van den Heuvel-Greve, *Environ. Monit. Assess.*, 2021, **193**, 499.
287 F. Fahd, M. Yang, F. Khan and B. Veitch, *Mar. Pollut. Bull.*, 2021, **166**, 112164.

288 Y. Jia, L. Ehlert, C. Wahlskog, A. Lundberg and C. Maurice, *Environ. Monit. Assess.*, 2018, **190**, 4.
289 H. K. French, M. C. Hansen, K. G. Moe and J. Stene, *Water*, 2023, **15**, 985.
290 A. M. Roos, M. Gamberg, D. Muir, A. Kärrman, P. Carlsson, C. Cuyler, Y. Lind, R. Bossi and F. Rigét, *Environ. Sci. Pollut. Res.*, 2022, **29**, 23721–23735.
291 M. J. Gunnarsdóttir, H. Ó. Andradóttir, K. Ólafsdóttir, Á. Ó. Hlödversdóttir, R. Kallenborn, E. M. Ræder, J. L. Lyche, J. Becanova and R. Lohmann, *Environ. Sci.: Adv.*, 2025, **4**, 1427–1443.
292 D. R. Katz, J. C. Sullivan, K. Rosa, C. L. Gardiner, A. R. Robuck, R. Lohmann, C. Kincaid and M. G. Cantwell, *Environ. Pollut.*, 2022, **300**, 118963.
293 B. J. Ruyle, H. M. Pickard, D. R. LeBlanc, A. K. Tokranov, C. P. Thackray, X. C. Hu, C. D. Vecitis and E. M. Sunderland, *Environ. Sci. Technol.*, 2021, **55**, 3686–3695.
294 A. Evenset, A. Olsson, M. Harju and G. W. Gabrielsen, *Settlements on Svalbard as Sources for Emerging Contaminants*, Akvaplan-niva, Tromsø, 2018.
295 K. B. Pedersen and A. Evenset, *Miljøtekniske Undersøkelser I SveaMiljøtekniske Undersøkelser i Svea*, Akvaplan-niva, Tromsø, 2018.
296 R. M. Sebastian and J. Louis, *J. Environ. Chem. Eng.*, 2022, **10**, 106930.
297 R. Eisted and T. H. Christensen, *Waste Manage. Res.*, 2011, **29**, 1064–1070.
298 Q. Pan, Q.-Y. Liu, J. Zheng, Y.-H. Li, S. Xiang, X.-J. Sun and X.-S. He, *Environ. Int.*, 2023, **174**, 107886.
299 D. C. Manheim, N. Yeşiller and J. L. Hanson, *J. Indian Inst. Sci.*, 2021, **101**, 625–657.
300 P. Kjeldsen, M. A. Barlaz, A. P. Rooker, A. Baun, A. Ledin and T. H. Christensen, *Crit. Rev. Environ. Sci. Technol.*, 2002, **32**, 297–336.
301 V. I. Gebenets, V. A. Tolmaniv, F. D. UIurov and P. Y. Groisman, *Environ. Res. Lett.*, 2021, **16**, 11.
302 M. Langer, T. S. von Deimling, S. Westermann, R. Rolph, R. Rutte, S. Antonova, V. Rachold, M. Schultz, A. Oehme and G. Grosse, *Nat. Commun.*, 2023, **14**, 1721.
303 L. Hanson James, N. Yeşiller and K. Oettle Nicolas, *J. Environ. Eng.*, 2010, **136**, 804–814.
304 K. E. Lund and K. L. Young, *Permafr. Periglac. Process.*, 2005, **16**, 195–207.
305 G. Asmund, *Recipientundersøgelse Ved Grønlandske Lossepladser*, Aaarhus Universitet, Aarhus Universitet, 2007.
306 R. Eisted, *PhD thesis*, DTU, 2011.
307 R. Eisted, *Miljøvurdering Af Grønlands Affaldssystem*, Technical University of Denmark, 2011.
308 C. Dias-Ferreira, G. M. Kirkelund and P. E. Jensen, *Chemosphere*, 2016, **148**, 380–387.
309 U. Zielke, M. Kriegbaum and K. U. Knudsen, *Emissioner fra affaldsforbrænding i Grønland*, Grønland Hjemmestyret, Nuuk, 2002.
310 M. D'Amico, A. Gambaro, C. Barbante, E. Barbaro, L. Caiazzo and M. Vecchiato, *Microchem. J.*, 2022, **183**, 108060.
311 ESANI, Greenland`s national waste management company.
312 S. Weichenthal, D. Van Rijswijk, R. Kulka, H. You, K. Van Ryswyk, J. Willey, R. Dugandzic, R. Sutcliffe, J. Moulton, M. Baike, L. White, J.-P. Charland and B. Jessiman, *Environ. Res.*, 2015, **142**, 46–50.
313 E. L. Weiser and A. N. Powell, *Arctic*, 2011, **64**, 220–226.
314 S. Gilbreath and P. H. Kass, *Am. J. Epidemiol.*, 2006, **164**, 518–528.
315 D. Porta, S. Milani, A. I. Lazzarino, C. A. Perucci and F. Forastiere, *Environ. Health*, 2009, **8**, 60.
316 M. Houde, N. Facciola, H. Routti, K. Vorkamp, J. Søndergaard and D. Muir, *Environ. Sci.: Adv.*, 2025, **4**, 990–1013.
317 L. A. Harwood, T. G. Smith and J. C. Auld, *Arctic*, 2012, **65**, 35–44.
318 M. Barghi, E. Fries, R. Chowdhury, J. Provencher, M. L. Mallory, B. M. Hamilton and R. Sühring, *Front. Environ. Chem.*, 2025, **6**, 1549292.
319 R. Kallenborn, K. Borgå, J. H. Christensen, M. Dowdall, A. Evenseth, J. Ø. Odland, A. Ruus, K. Aspmo Pfaffhuber, J. Pawlak and L.-O. Reiersen, *AMAP 2011: Combined Effects of Selected Pollutants and Climate Change in the Arctic Environment*, Arctic Monitoring and Assessment Programme (AMAP), Oslo, 2011.
320 T. K. Gebri, Y. Li, C. Dong, Y. Yang, R. Yang, Q. Pei Zhang and G. Jiang, *Sci. Total Environ.*, 2023, **878**, 163023.
321 G. D. Triggs and British Institute of International and Comparative Law, *The Antarctic Treaty Regime : Law, Environment, and Resources*, Cambridge University Press, Cambridge, Cambridgeshire ; New York, 1987.
322 J. S. Stark, P. A. Corbett, G. Dunshea, G. Johnstone, C. King, J. A. Mondon, M. L. Power, A. Samuel, I. Snape and M. Riddle, *Water Res.*, 2016, **105**, 602–614.
323 I. Snape, S. H. Ferguson, P. M. Harvey and M. J. Riddle, *Chemosphere*, 2006, **63**, 89–98.
324 I. Snape, C. E. Morris and C. M. Cole, *Cold Reg. Sci. Technol.*, 2001, **32**, 157–174.
325 S. T. Brooks, J. Jabour, J. van den Hoff and D. M. Bergstrom, *Nat Sustainability*, 2019, **2**, 185–190.
326 P. Convey, *Antarct. Sci.*, 2020, **32**, 425.
327 I. Hodgson-Johnston, A. Jackson, J. Jabour and A. Press, *Restor. Ecol.*, 2017, **25**, 135–139.
328 S. Corsolini, A. Metzdorff, D. Baroni, J. L. Roscales, B. Jiménez, E. Cerro-Gálvez, J. Dachs, C. Galbán-Malagón, O. Audy, J. Kohoutek, P. Přibylova, M. Poblete-Morales, R. Avendaño-Herrera, E. Bergami and K. Pozo, *Environ. Res.*, 2021, **196**, 110344.
329 G. T. Yogui and J. L. Sericano, *Environ. Pollut.*, 2009, **157**, 975–980.
330 S. Wild, I. Eulaers, A. Covaci, R. Bossi, D. Hawker, R. Cropp, C. Southwell, L. Emmerson, G. Lepoint, P. Eisenmann and S. B. Nash, *Environ. Pollut.*, 2022, **292**, 118358.
331 M. Vecchiato, E. Argiriadis, S. Zambon, C. Barbante, G. Toscano, A. Gambaro and R. Piazza, *Microchem. J.*, 2015, **119**, 75–82.

332 A. Evenset, J. Carroll, G. N. Christensen, R. Kallenborn, D. Gregor and G. W. Gabrielsen, *Environ. Sci. Technol.*, 2007, **41**, 1173–1179.
333 K. L. Foster, L. E. Kimpe, S. K. Brimble, H. Liu, M. L. Mallory, J. P. Smol, R. W. Macdonald and J. M. Blais, *Environ. Sci. Technol.*, 2011, **45**, 10053–10060.
334 J. Potapowicz, M. Szopińska, D. Szumińska, R. J. Bialik and Ż. Polkowska, *Chemosphere*, 2022, **288**, 132637.
335 A. R. Aves, L. E. Revell, S. Gaw, H. Ruffell, A. Schuddeboom, N. E. Wotherspoon, M. LaRue and A. J. McDonald, *Cryosphere*, 2022, **16**, 2127–2145.
336 K. Jones-Williams, E. Rowlands, S. Primpke, T. Galloway, M. Cole, C. Waluda and C. Manno, *Sci. Total Environ.*, 2025, **966**, 178543.
337 A. Cabrerizo, J. Dachs, D. Barceló and K. C. Jones, *Environ. Sci. Technol.*, 2013, **47**, 4299–4306.
338 C. A. de Wit, K. Vorkamp and D. Muir, *Environ. Sci.: Processes Impacts*, 2022, **24**, 1530–1543.
339 J. Beyer, A. Goksoyr, D. O. Hjermann and J. Klungsoyr, *Mar. Environ. Res.*, 2020, **162**, 105155.
340 E. M. Krummel and A. Gilman, *Int. J. Circumpolar Health*, 2016, **75**, 33822.
341 J. Zheng, B. Chen, W. Thanyamanta, K. Hawboldt, B. Zhang and B. Liu, *Mar. Pollut. Bull.*, 2016, **104**, 7–19.
342 J. Schmale, S. Arnold, K. S. Law, T. Thorp, S. Anenberg, W. Simpson, J. Mao and K. Pratt, *Earth's Future*, 2018, **6**, 1385–1412.
343 T. Y. Sorokina, in *Global Arctic: an Introduction to the Multifaceted Dynamics of the Arctic*, Springer, 2022, pp. 229–253.
344 Arctic Council and Protection of the Arctic Marine Environment Working Group, *Arctic Offshore Oil & Gas Guidelines*, PAME, 2002.
345 F. d. r. Lasserre and O. Faury, *Arctic Shipping : Climate Change, Commercial Traffic and Port Development*, Routledge, Lodi ; New York, 2020.
346 C. c. Pelaudeix and E. M. Basse, *Governance of Arctic Offshore Oil and Gas*, Routledge, Taylor & Francis Group, London ; New York, 2018.
347 K. Hossain, J. M. Roncero Martin and A. Petrétei, *Arctic law and governance: the role of law in the Arctic*, ed. K. Hossain and J. M. Roncero, Lapin yliopisto, Arktinen keskus, Rovaniemi, 2018.
348 B. M. Braune, P. M. Outridge, A. T. Fisk, D. C. Muir, P. A. Helm, K. Hobbs, P. F. Hoekstra, Z. A. Kuzyk, M. Kwan, R. J. Letcher, W. L. Lockhart, R. J. Norstrom, G. A. Stern and I. Stirling, *Sci. Total Environ.*, 2005, **351–352**, 4–56.
349 B. M. Braune and D. C. Muir, *Environ. Sci. Technol.*, 2017, **51**, 3802–3808.
350 T. F. Bidleman, P. A. Helm, B. M. Braune and G. W. Gabrielsen, *Sci. Total Environ.*, 2010, **408**, 2919–2935.
351 T. F. Bidleman, P. A. Helm, B. M. Braune and G. W. Gabrielsen, *Sci. Total Environ.*, 2010, **408**, 2919–2935.
352 P. A. Helm, T. F. Bidleman, H. H. Li and P. Fellin, *Environ. Sci. Technol.*, 2004, **38**, 5514–5521.
353 D. L. Gautier, K. J. Bird, R. R. Charpentier, A. Grantz, D. W. Houseknecht, T. R. Klett, T. E. Moore, J. K. Pitman, C. J. Schenk, J. H. Schuenemeyer, K. Sorensen, M. E. Tennyson, Z. C. Valin and C. J. Wandrey, *Science*, 2009, **324**, 1175–1179.
354 K. Keil, *Cooperat. Conflict*, 2014, **49**, 162–190.
355 F. Vidal, in *Global Arctic: an Introduction to the Multifaceted Dynamics of the Arctic*, ed. M. Finger and G. Rekvig, Springer International Publishing, Cham, 2022, pp. 389–405, DOI: 10.1007/978-3-030-81253-9_20.
356 AMAP, *Arctic Oil and Gas 2007*, Arctic monitoring and Assessment Programme (AMAP), Oslo, 2007.
357 I. A. Nemirovskaya and A. V. Khramtsova, *Fluids*, 2021, **6**, 456.
358 N. W. Dudiak, Alaska. Department of Fish and Game. Habitat Division., Chronology of Exxon Valdez oil spill activities in the Homer spill zone, 1989, 1990, and 1991, Habitat Division, Alaska Dept. of Fish and Game, Juneau, Alaska (P.O. Box 25526, Juneau 99802-5526), 1992.
359 K. R. Middleton, Alaska. Department of Fish and Game, Habitat Division, Alaska Department of Fish and Game Exxon Valdez oil spill response operations report : Habitat Division, 1989-1992, The Division, Juneau, Alaska (P.O. Box 25526, Juneau 99802-5526), 1992.
360 U. Varanasi, *Assessment of Oil Spill Impacts on Fishery Resources : Measurement of Hydrocarbons and Their Metabolites, and Their Effects, in Important Species*, Environmental Conservation Division, Northwest Fisheries Science Center, National Marine Fisheries Service, National Oceanic & Atmospheric Administration, Seattle, WA, 1995.
361 G. A. Sergy, *Int. Oil Spill Conf. Proc.*, 1985, **1**, 5.
362 Q. Feng, C. An, Y. Cao, Z. Chen, E. Owens, E. Taylor, Z. Wang and E. Saad, *J. Environ. Inform. Lett.*, 2021, **5**, 39–47.
363 S. L. Ross, *Field Research Spills to Investigate the Physical and Chemical Fate of Oil in Pack Ice*, Environmental Research Lid and Dickins Associates Ltd, Toronto, 1987.
364 A. Chen, X. Wu, S. L. M. Simonich, H. Kang and Z. Xie, *Environ. Pollut.*, 2021, **268**, 115963.
365 V. I. Pavlenko and E. K. Glukhareva, Environmental Impact of Oil And Gas Production In the Russian Arctic, Abstract, *International Ocean and Polar Engineering Conference (ISOPE-2007)*, 2007.
366 E. Arzaghi, R. Abbassi, V. Garaniya, J. Binns and F. Khan, *Mar. Pollut. Bull.*, 2018, **135**, 1117–1127.
367 J. H. Trefry, K. H. Dunton, R. P. Trocine, S. V. Schonberg, N. D. McTigue, E. S. Hersh and T. J. McDonald, *Mar. Environ. Res.*, 2013, **86**, 35–45.
368 Y. Wang, H. Chen, Q. Xing and X. Xu, *Environ. Res.*, 2025, 121865.
369 C. Chen, Y. Wang, F. Chen, X. Wang, Q. Zhang, J. Sun, S. Li, Q. Chen, F. Shang and H. Zhang, *Water*, 2025, **17**, 149.
370 C. R. Gravelle, C. Brown and G. Wiatzka, *Risk Management and Remedial Action Plan for the Canol Trail*, NT, PWGSC, Ottawa, 2016.
371 A. B. Novakovskiy, V. A. Kanev and M. Y. Markarova, *Sci. Rep.*, 2021, **11**, 4888.

372 V. P. Murygina, M. Y. Markarova and S. V. Kalyuzhnyi, *Environ. Int.*, 2005, **31**, 163–166.
373 M. Fingas, *In-situ Burning for Oil Spill Countermeasures*, CRC Press, 2018.
374 J. S. Poland, M. J. Riddle and B. A. Zeeb, *Polar Rec.*, 2003, **39**, 369–383.
375 R. Ahtuangaruak, *Politics, Groups, and Identities*, 2015, **3**(4), 673–677.
376 R. Gillette, *Science*, 1971, **171**, 1130–1132.
377 M. Gillard, M. Jones and A. Rowell, *New Solut.*, 2000, **10**, 167–183.
378 J. A. Massoud, D. M. Boje, E. Capener and M. Marcillo, *Int. J. Organ. Anal.*, 2019, **27**, 1562–1577.
379 S. Basharat and K. Oien, Accidents and Emergency Response in the Arctic Sea, in *OTC Arctic Technology Conference*, Offshore Technology Conference, Houston, Texas, 2014, DOI: 10.4043/24609-MS.
380 T. Nordam, D. A. E. Dunnebier, C. J. Beegle-Krause, M. Reed and D. Slagstad, *Ambio*, 2017, **46**, 442–452.
381 Interstate Oil Compact Commission, *Major Tar Sand and Heavy Oil Deposits of the United States*, Interstate Oil Compact Commission, Oklahoma City, Okla, 1984.
382 F. J. Hein, D. Leckie, S. Larter and J. R. Suter, *Heavy-oil and Oil-sand Petroleum Systems in Alberta and Beyond*, American Association of Petroleum Geologists, 2013, vol. 64, DOI: 10.1306/St641337ISBN.
383 A. S. Kozub, *The Oil Sands of Alberta Showing Bituminous and Oil Sand Leases*, Maclean-Hunter : available from the Lombard North Group, Calgary, Alta, 1975.
384 M. F. Fingas, *The Chemistry of Oil and Petroleum Products*, De Gruyter, Berlin ; Boston, 2022.
385 S. T. Fisher, *Plentiful and Cheap Energy from Fossil Fuel Deposits by Electrical Induction Heating : Proposals for In-Situ Exploitation of Coal, Lignite, Oil-Sand, Oil-Shale and Viscous Oil*, F. T. Fisher's Sons, Montreal, 2d edn, 1976.
386 M. C. Roy, L. Foote and J. J. Ciborowski, *J. Environ. Manage.*, 2016, **172**, 18–28.
387 E. A. Johnson and K. Miyanishi, *Ann. N. Y. Acad. Sci.*, 2008, **1134**, 120–145.
388 S. N. Sarma, L. E. Kimpe, V. C. Doyon, J. M. Blais and H. M. Chan, *Environ. Res.*, 2019, **178**, 108680.
389 L. M. Hewitt, J. W. Roy, S. J. Rowland, G. Bickerton, A. DeSilva, J. V. Headley, C. B. Milestone, A. G. Scarlett, S. Brown, C. Spencer, C. E. West, K. M. Peru, L. Grapentine, J. M. E. Ahad, H. Pakdel and R. A. Frank, *Environ. Sci. Technol.*, 2020, **54**, 1522–1532.
390 A. C. Scott, W. Zubot, C. W. Davis and J. Brogly, *Sci. Total Environ.*, 2020, **712**, 134558.
391 A. A. Holden, R. B. Donahue and A. C. Ulrich, *J. Contam. Hydrol.*, 2011, **119**, 55–68.
392 C. Wei, S. M. Jafari Raad and H. Hassanzadeh, *PNAS Nexus*, 2023, **2**, pgad260.
393 P. M. Fedorak, D. L. Coy, M. J. Salloum and M. J. Dudas, *Can. J. Microbiol.*, 2002, **48**, 21–33.
394 N. Jariyasopit, T. Harner, C. Shin and R. Park, *Environ. Pollut.*, 2021, **282**, 117014.
395 J. R. Graney, M. S. Landis, K. J. Puckett, W. B. Studabaker, E. S. Edgerton, A. H. Legge and K. E. Percy, *Chemosphere*, 2017, **184**, 700–710.
396 J. M. E. Ahad, H. Pakdel, P. R. Gammon, B. Mayer, M. M. Savard, K. M. Peru and J. V. Headley, *Environ. Sci. Technol.*, 2020, **54**, 2790–2799.
397 L. D. Brown and A. C. Ulrich, *Chemosphere*, 2015, **127**, 276–290.
398 E. Lari, E. Mohaddes and G. G. Pyle, *Sci. Total Environ.*, 2017, **605–606**, 824–829.
399 C. Andre, M. Pilote, C. Gagnon and F. Gagne, *Comp. Biochem. Physiol., Part C: Toxicol. Pharmacol.*, 2022, **251**, 109193.
400 M. Noah, M. Lappe, B. Schneider, A. Vieth-Hillebrand, H. Wilkes and J. Kallmeyer, *Sci. Total Environ.*, 2014, **499**, 297–310.
401 D. P. Khasa, M. Fung and B. Logan, *Bioresour. Technol.*, 2005, **96**, 857–864.
402 C. Abou-Khalil, R. C. Prince, C. W. Greer, K. Lee and M. C. Boufadel, *Environ. Sci. Technol.*, 2022, **56**, 8124–8131.
403 S. E. Ruffell, R. A. Frank, A. P. Woodworth, L. M. Bragg, A. E. Bauer, L. E. Deeth, K. M. Muller, A. J. Farwell, D. G. Dixon, M. R. Servos and B. J. McConkey, *Ecotoxicol. Environ. Saf.*, 2016, **133**, 373–380.
404 A. B. Avagyan, *Environ. Sci. Pollut. Res. Int.*, 2017, **24**, 20241–20253.
405 M. S. Landis, J. Patrick Pancras, J. R. Graney, E. M. White, E. S. Edgerton, A. Legge and K. E. Percy, *Sci. Total Environ.*, 2017, **584–585**, 105–117.
406 K. P. Timoney and P. Lee, *Open Conserv. Biol. J.*, 2009, **3**, 35–81.
407 M. A. Bari and W. B. Kindzierski, *Environ. Pollut.*, 2018, **235**, 602–614.
408 R. J. Wenning and L. Martello, in *Environmental Forensics for Persistent Organic Pollutants*, ed. G. O'Sullivan and C. Sandau, Elsevier, Amsterdam, 2014, pp. 357–390, DOI: 10.1016/B978-0-444-59424-2.00008-6.
409 N. C. Swart and A. J. Weaver, *Nat. Clim. Change*, 2012, **2**, 134–136.
410 G. M. Irvine, J. M. Blais, J. R. Doyle, L. E. Kimpe and P. A. White, *Environ. Health*, 2014, **13**, 7.
411 M. L. Finkel, *Curr. Opin. Environ. Sci. Health*, 2018, **3**, 52–55.
412 K. W. Biggar, S. Haidar, M. Nahir and P. M. Jarrett, *J. Cold Reg. Eng.*, 1998, **12**, 84–104.
413 G. A. Sergy, The Baffin Island oil spill (Bios) project: a summary, in *Proceedings of the 1985 Oil Spill Conference (Prevention, Behavior, Control, Cleanup)*, ed. J. O. Ludwigson, American Petroleum Institute Publication, Los Angeles, California, 1985, vol. 4385, pp. 571–575.
414 R. C. Prince, E. H. Owens and G. A. Sergy, *Mar. Pollut. Bull.*, 2002, **44**, 1236–1242.
415 B. Hunnie, Master of Science, University of Manitoba, 2023.
416 C. Mauchan, Master of Science, 2003.
417 A. Dickins, Advancing oil spill response in ice-covered waters, Prince William Sound oil spill recovery institute Cordova, Alaska and United States Arctic Research Comission, Arlington USA, 2004.

418 M. Mirbach, Oil spill response capacity in Nunavut and the Beauford Sea, World Wildlife Fund (WWF), 2017.
419 S. Somanathan, P. C. Flynn and J. K. Szymanski, *Marit. Econ. Logist.*, 2007, **9**, 324–334.
420 V. Mirbach, *WWF-Canada: Oil spill response capacity in Nunavut and the Beaufort Sea: Responding to Arctic shipping oil spills – risks and challenges*, WWF-Canada, Toronto, 2017, pp. 1–56, Retrieved from https://cleanarctic.org/wp-content/uploads/2021/10/170405___oilspillresponsecapacitynunavut_web.pdf.
421 N. P. Ventikos, E. Vergetis, H. N. Psaraftis and G. Triantafyllou, *J. Hazard. Mater.*, 2004, **107**, 51–58.
422 J. W. Doerffer, *Oil Spill Response in the Marine Environment*, Elsevier, 2013.
423 M. Jartun, R. T. Ottesen, T. Volden and Q. Lundkvist, *J. Toxicol. Environ. Health, Part A*, 2009, **72**, 284–294.
424 W. Chen, S. G. Leblanc, P. H. White, A. Patenaude, K. Clark, B. Croft, J. S. Pellissey, L. Meinert, J. Boulanger and A. Gunn, *Environ. Monit. Assess.*, 2021, **193**, 560.
425 J. B. Korosi, D. C. Eickmeyer, J. R. Thienpont, M. J. Palmer, L. E. Kimpe and J. M. Blais, *Sci. Total Environ.*, 2016, **553**, 96–106.
426 M. Perrett, B. Sivarajah, C. L. Cheney, J. B. Korosi, L. Kimpe, J. M. Blais and J. P. Smol, *Environ. Pollut.*, 2021, **278**, 116815.
427 S. J. Brooks, C. Harman, M. T. Hultman and J. A. Berge, *Sci. Total Environ.*, 2015, **524–525**, 104–114.
428 Y. Zhang, W. Shotyk, R. Pelletier, C. Zaccone, T. Noernberg, G. Mullan-Boudreau and J. W. Martin, *Environ. Int.*, 2023, **182**, 108335.
429 Y. Zhang, W. Shotyk, C. Zaccone, T. Noernberg, R. Pelletier, B. Bicalho, D. G. Froese, L. Davies and J. W. Martin, *Environ. Sci. Technol.*, 2016, **50**, 1711–1720.
430 M. A. Naeth and S. R. Wilkinson, *J. Environ. Qual.*, 2008, **37**, 1675–1684.
431 K. J. Fernie, S. C. Marteinson, D. Chen, A. Eng, T. Harner, J. E. G. Smits and C. Soos, *Sci. Total Environ.*, 2018, **624**, 250–261.
432 J. K. Schuster, T. Harner, K. Su, C. Mihele and A. Eng, *Environ. Sci. Technol.*, 2015, **49**, 2991–2998.
433 K. Schütze, T. Drotikova, M. Marquès, M. Nadal, S. Weinbruch and R. Kallenborn, Characterization of local Polycyclic Aromatic Hydrocarbon sources in the Arctic: Emission and deposition from Svalbard (Norway) settlements, *Abstract, International Conference of Chemistry in the Environment*, Leipzig, Germany, 2015.
434 G. A. Stern, C. R. Macdonald, P. C. Carvalho, T. Wolfe and F. Ferraz, *Sci. Total Environ.*, 2023, **855**, 158718.
435 X. Ji, E. Abakumov, V. Polyako, X. Xie and W. Dongyang, *Environ. Pollut.*, 2019, **255**, 113239.
436 T. D. Prowse, C. Furgal, R. Chouinard, H. Melling, D. Milburn and S. L. Smith, *Ambio*, 2009, **38**, 272–281.
437 S. Lofthus, I. Bakke, C. W. Greer and O. G. Brakstad, *Mar. Pollut. Bull.*, 2021, **172**, 112823.
438 S. Lofthus, I. Bakke, J. Tremblay, C. W. Greer and O. G. Brakstad, *Mar. Pollut. Bull.*, 2020, **154**, 111090.
439 T. R. Walker, J. O. Habeck, T. P. Karjalainen, T. Virtanen, N. Solovieva, V. Jones, P. Kuhry, V. I. Ponomarev, K. Mikkola, A. Nikula, E. Patova, P. D. Crittenden, S. D. Young and T. Ingold, *Ambio*, 2006, **35**, 220–228.
440 P. P. Sartz, M. I. Hasan and S. Aggarwal, *Sci. Total Environ.*, 2023, **892**, 163860.
441 P. Sharma and S. Schiewer, *Environ. Sci. Pollut. Res. Int.*, 2016, **23**, 14881–14888.
442 J. Lee, Y. Kim, J. Cha, D. Kim, K. Jang, J. H. Kim, S. I. Nam and S. Hong, *Mar. Pollut. Bull.*, 2023, **189**, 114740.
443 M. V. Bykova, A. V. Alekseenko, M. A. Pashkevich and C. Drebenstedt, *Environ. Geochem. Health*, 2021, **43**, 2331–2346.
444 M. Vecchiato, E. Barbaro, A. Spolaor, F. Burgay, C. Barbante, R. Piazza and A. Gambaro, *Environ. Pollut.*, 2018, **242**, 1740–1747.
445 J. Sondergaard, G. Asmund, P. Johansen and F. Riget, *Mar. Environ. Res.*, 2011, **71**, 331–341.
446 W. L. Lockhart, R. Wagemann, B. Tracey, D. Sutherland and D. J. Thomas, *Sci. Total Environ.*, 1992, **122**, 165–245.
447 D. J. Thomas, B. Tracey, H. Marshall and R. J. Norstrom, *Sci. Total Environ.*, 1992, **122**, 135–164.
448 N. Pelletier, J. Chetelat, B. Cousens, S. Zhang, D. Stepner, D. C. G. Muir and J. C. Vermaire, *Environ. Pollut.*, 2020, **259**, 113888.
449 N. R. Haddaway, A. Smith, J. J. Taylor, C. Andrews, S. J. Cooke, A. E. Nilsson and P. Lesser, *Environ. Evid.*, 2022, **11**, 30.
450 A. Tolvanen, P. Eilu, A. Juutinen, K. Kangas, M. Kivinen, M. Markovaara-Koivisto, A. Naskali, V. Salokannel, S. Tuulentie and J. Simila, *J. Environ. Manage.*, 2019, **233**, 832–844.
451 M. Ellis, I. Altshuler, L. Schreiber, Y. J. Chen, M. Okshevsky, K. Lee, C. W. Greer and L. G. Whyte, *Mar. Pollut. Bull.*, 2022, **174**, 113288.
452 B. Elberling, J. Sondergaard, L. A. Jensen, L. B. Schmidt, B. U. Hansen, G. Asmund, T. B. Zunic, J. Hollesen, S. Hanson, P. E. Jansson and T. Friborg, *Environ. Sci. Technol.*, 2007, **41**, 2407–2413.
453 S. Jackson, G. Poelzer, G. Poelzer and B. Noble, *Environ. Manag.*, 2023, **72**, 37–52.
454 Z. Jaworowski, *Pollution Ofthe Norwegian Arctic: A Review*, Norwegian Polar institute, Oslo, 1989.
455 L. A. Barrie, *Atmos. Environ.*, 1986, **20**, 643–663.
456 N. Z. Heidam, *Atmos. Environ.*, 1984, **18**, 329–343.
457 B. Ottar and J. M. Pacyna, in *Arctic Air Pollution*, ed. B. Stonehouse, Cambridge University Press, Cambridge, 1987, pp. 53–68, DOI: 10.1017/CBO9780511565496.007.
458 S.-M. Li and J. W. Winchester, *J. Geophys. Res.: Atmos.*, 1990, **95**, 1797–1810.
459 L. A. Barrie and M. J. Barrie, *J. Atmos. Chem.*, 1990, **11**, 211–226.
460 J. M. Pacyna, B. Ottar, U. Tomza and W. Maenhaut, *Atmos. Environ.*, 1985, **19**, 857–865.
461 B. Ottar, J. M. Pacyna and T. C. Berg, *Atmos. Environ.*, 1986, **20**, 87–100.

462 H. Li, W. Zhang, H. Yan and P. Gao, *Environ. Pollut.*, 2024, **351**, 124024.
463 R. Boyd, T. Bjerkgård, B. Nordahl and T. Schiellerup, *Mineral Resources in the Arctic – An Introduction*, Geological Survey of Norway, Oslo, 2016.
464 E. C. H. Keskitalo, *The Politics of Arctic Resources : Change and Continuity in the "Old North" of Northern Europe*, Routledge Taylor & Francis Group, London ; New York, NY, 2019.
465 I. Bay-Larsen, B. Dale and B. Skorstad, *The will to drill: Mining in Arctic communities*, Springer, Cham, 2018, DOI: 10.1007/978-3-319-62610-9.
466 G. M. Kashulina, *Eurasian Soil Sci.*, 2018, **51**, 467–478.
467 A. Kammerud, *Air Pollution Effects in the Norwegian-Russian Border Area - A Status Report*, E. monitoring Report ISBN: 82-7655-446-6, Statens Forurensningstilsyn (SFT), Oslo, 2002.
468 T. Norseth, *Sci. Total Environ.*, 1994, **148**, 103–108.
469 M. Schlabach and T. Skotvold, *Undersøkelse Av PCDD/PCDF I Omgivelsene Rundt Pelletsverket Ved Aktieselskapet Sydvaranger*, Report ISBN 82-425-0745-7, Norwegian Institute for Air Research (NILU), Kjeller, 1996.
470 Arctic Monitoring and Assessment Programme, *Arctic Pollution Issues : a State of the Arctic Environment Report*, AMAP, Oslo, 1997.
471 K. Arnesen, *PhD Cummulative*, Norwegian University of Science and Technology (NTNU), 2023.
472 T. V. R. Pillay, *Aquaculture and the Environment*, Halsted Press, New York, 1992.
473 Food and Agriculture Organization of the United Nations. and FAO Fisheries and Aquaculture Dept. Aquaculture Management and Conservation Service, *Environmental Impact Assessment and Monitoring in Aquaculture : Requirements, Practices, Effectiveness and Improvements*, Food and Agriculture Organization of the United States, Rome, 2009.
474 S. Marcheggiani, E. D'Ugo, C. Puccinelli, R. Giuseppetti, A. M. D'Angelo, C. O. Gualerzi, R. Spurio, L. K. Medlin, D. Guillebault, J. Baudart-Lenfant, W. Weigel, K. Helmi and L. Mancini, *Int. J. Environ. Res. Public Health*, 2015, **12**, 5505–5527.
475 L. Metzeling, F. McKenzie and R. StClair, *The Effect of Fish Farming on the Water Quality and Invertebrate Fauna of Two Upland Rivers*, Environment Protection Authority, State Govt. of Victoria, Melbourne, Vic, 1996.
476 Q. Zhou, K. Li, X. Jun and L. Bo, *Bioresour. Technol.*, 2009, **100**, 3780–3786.
477 J. O. Olaussen, *Mar. Pol.*, 2018, **98**, 158–163.
478 I. S. Abihssira-García, T. Kögel, A. Gomiero, T. Kristensen, M. Krogstad and P. A. Olsvik, *Mar. Pollut. Bull.*, 2022, **180**, 113794.
479 L. Cao, W. Wang, Y. Yang, C. Yang, Z. Yuan, S. Xiong and J. Diana, *Environ. Sci. Pollut. Res. Int.*, 2007, **14**, 452–462.
480 M. C. M. Beveridge, *Cage and Pen Fish Farming : Carrying Capacity Models and Environmental Impact*, FAO, Rome, 1984.
481 L. D. Kraemer, J. E. Berner and C. M. Furgal, *Int. J. Circumpolar Health*, 2005, **64**, 498–508.
482 D. T. R. Moreau and B. Neis, *Mar. Pol.*, 2009, **33**, 401–411.
483 A. Rico, M. Vighi, P. J. Van den Brink, M. ter Horst, A. Macken, A. Lillicrap, L. Falconer and T. C. Telfer, *Rev. Aquacult.*, 2019, **11**, 969–988.
484 Ø. Hermansen and M. Troell, *Aquaculture in the Arctic - A Review*, Report 36, Nofima, Tromsø, 2012.
485 N. V. Hidayati, L. Asia, I. Khabouchi, F. Torre, I. Widowati, A. Sabdono, P. Doumenq and A. D. Syakti, *Chemosphere*, 2021, **263**, 128372.
486 R. L. Naylor, R. J. Goldburg, J. H. Primavera, N. Kautsky, M. C. M. Beveridge, J. Clay, C. Folke, J. Lubchenco, H. Mooney and M. Troell, *Nature*, 2000, **405**, 1017–1024.
487 J. A. Hutchings, I. M. Côté, J. J. Dodson, I. A. Fleming, S. Jennings, N. J. Mantua, R. M. Peterman, B. E. Riddell and A. J. Weaver, *Environ. Rev.*, 2012, **20**, 220–311.
488 M. Troell, A. Eide, J. Isaksen, Ø. Hermansen and A.-S. Crépin, *Ambio*, 2017, **46**, 368–386.
489 N. W. Ayer and P. H. Tyedmers, *J. Cleaner Prod.*, 2009, **17**, 362–373.
490 G. Helgadóttir, H. Renssen, T. R. Olk, T. J. Oredalen, L. Haraldsdóttir, S. Skúlason and H. P. Thorarensen, *Front. Sustainable Food Syst.*, 2021, **5**, 654117.
491 B. Ö. Smarason, *Magister Scientiarum Report*, University of Iceland, 2013.
492 H. Zhong, M. Wu, C. Sonne, S. S. Lam, R. W. M. Kwong, Y. Jiang, X. Zhao, X. Sun, X. Zhang, C. Li, Y. Li, G. Qu, F. Jiang, H. Shi, R. Ji and H. Ren, *Eco-Environ. Health*, 2023, **2**, 142–151.
493 B. C. Kelly, M. P. Fernandez, M. G. Ikonomou and W. Knapp, *Aquaculture*, 2008, **285**, 224–233.
494 J. O. Bustnes, E. Lie, D. Herzke, T. Dempster, P. A. Bjørn, T. Nygård and I. Uglem, *Environ. Sci. Technol.*, 2010, **44**, 8736–8743.
495 B. L. Townhill, E. Reppas-Chrysovitsinos, R. Suhring, C. J. Halsall, E. Mengo, T. Sanders, K. Dahnke, O. Crabeck, J. Kaiser and S. N. R. Birchenough, *Ambio*, 2022, **51**, 471–483.
496 M. H. G. Berntssen, L. Thoresen, S. Albrektsen, E. Grimaldo, L. Grimsmo, R. D. Whitaker, V. Sele and M. Wiech, *Foods*, 2021, **10**(6), 1265.
497 S. D. Shaw, D. Brenner, M. L. Berger, D. O. Carpenter, C.-S. Hong and K. Kannan, *Environ. Sci. Technol.*, 2006, **40**, 5347–5354.
498 M. H. Berntssen, L. Thoresen, S. Albrektsen, E. Grimaldo, L. Grimsmo, R. D. Whitaker, V. Sele and M. Wiech, *Foods*, 2021, **10**, 1265.
499 D. M. o. Defence, *First Agreement on theArctic and North Atlantic under the 2024 - 2033 Defence Agreement*, Danish Ministry of Defence, Copenhagen, 2023.
500 L. Heininen, *Security and Sovereignty in the North Atlantic*, Palgrave Macmillan, Basingstoke, Hampshire, 2014.
501 J. Kraska, *Arctic Security in an Age of Climate Change*, Cambridge University Press, Cambridge ; New York, 2011.
502 J. N. Markowitz, *Perils of Plenty: Arctic Resource Competition and the Return of the Great Game*, Oxford University Press, 2020.
503 G. Hønneland, *Slavonic East Eur. Rev.*, 2016, **94**(4), 785–787.

504 B. Braune, J. Chetelat, M. Amyot, T. Brown, M. Clayden, M. Evans, A. Fisk, A. Gaden, C. Girard, A. Hare, J. Kirk, I. Lehnherr, R. Letcher, L. Loseto, R. Macdonald, E. Mann, B. McMeans, D. Muir, N. O'Driscoll, A. Poulain, K. Reimer and G. Stern, *Sci. Total Environ.*, 2015, **509–510**, 67–90.

505 S. Neffe, in *Environmental Contamination and Remediation Practices at Former and Present Military Bases*, ed. B. Paukštys, F. Fonnum, B. A. Zeeb and K. J. Reimer, Springer Netherlands, Dordrecht, 1998, pp. 83–92, DOI: 10.1007/978-94-011-5304-1_8.

506 L.-O. Reiersen, K. Vorkamp and R. Kallenborn, *Environ. Sci. Ecotechnology*, 2024, **17**, 100302.

507 C. Zum Brunnen, *Contested Arctic: Indigenous Peoples, Industrial States, and the Circumpolar Environment*, 1997, pp. 88–121.

508 V. Polyakov, W. Kadoya, S. Beal, H. Morehead, E. Hunt, F. Cubello, S. M. Meding and K. Dontsova, *Sci. Total Environ.*, 2023, **866**, 161434.

509 K. Gebka, J. Beldowski and M. Beldowska, *Environ. Sci. Pollut. Res. Int.*, 2016, **23**, 23103–23113.

510 K. M. Dontsova, S. L. Yost, J. Simunek, J. C. Pennington and C. W. Williford, *J. Environ. Qual.*, 2006, **35**, 2043–2054.

511 S. J. Doherty, K. S. Messan, R. R. Busby and R. A. Barbato, *Int. J. Phytorem.*, 2019, **21**, 958–968.

512 G. Bordeleau, R. Martel, G. Ampleman and S. Thiboutot, *J. Environ. Qual.*, 2008, **37**, 308–317.

513 P. J. M. Vertegaal, *Environ. Conserv.*, 1989, **16**, 54–64.

514 M. J. Lawrence, H. L. J. Stemberger, A. J. Zolderdo, D. P. Struthers and S. J. Cooke, *Environ. Rev.*, 2015, **23**, 443–460.

515 P. Broomandi, M. Guney, J. R. Kim and F. Karaca, *Sustainability*, 2020, **12**, 9002.

516 N. G. Svendsen, P. K. Kalita and D. L. Gebhart, Military maneuver effects on water quality and non-point source pollution: implications for training land use, *ASAE Annual Meeting*, Paper No. 062154, American Society of Agricultural and Biological Engineers, pp. 1–12, DOI: 10.13031/2013.21176.

517 D. Shtob, C. Alvarez and N. Theis, *Sociol. Compass*, 2023, e13079.

518 V. Stabnikov, V. Ivanov and A. Vaseashta, in *Bio-Based Materials and Biotechnologies for Eco-Efficient Construction*, ed. F. Pacheco-Torgal, V. Ivanov and D. C. W. Tsang, Woodhead Publishing, 2020, pp. 377–393, DOI: 10.1016/B978-0-12-819481-2.00018-0.

519 J. S. Poland, S. Mitchell and A. Rutter, *Cold Reg. Sci. Technol.*, 2001, **32**, 93–105.

520 T. K. Vlassova, *Mitig. Adapt. Strateg. Glob. Change*, 2006, **11**, 897–909.

521 R. J. Scrudato, J. R. Chiarenzelli, P. K. Miller, C. R. Alexander, J. Arnason, K. Zamzow, K. Zweifel, J. Gologergen, J. Kava, V. Waghiyi and D. O. Carpenter, *J. Local Global Health Sci.*, 2012, **2012**(2), 1–15.

522 G. Hønneland, *Russia and the Arctic: Environment, Identity and Foreign Policy*, I. B. Tauris, London, 2016.

523 B. Garfield, *The Thousand-Mile War*, Doubleday & Company, Inc., Garden City, NY, 1969.

524 F. A. von Hippel, E. J. Trammell, J. Merilä, M. B. Sanders, T. Schwarz, J. H. Postlethwait, T. A. Titus, C. L. Buck and I. Katsiadaki, *Evol. Ecol. Res.*, 2016, **17**, 487–504.

525 USATSDR, U.S. Agency for Toxic Substances and Disease Registry, 2015.

526 R. Jordan-Ward, F. A. Hippel, J. Schmidt and M. P. Verhougstraete, *Integr. Environ. Assess. Manage.*, 2024, **20**, 1420–1431.

527 USBLM, U.S. Bureau of Land Management, 2015.

528 ADEC, Alaska Department of Environmental Conservation Division of Spill Prevention and Response Contaminated Sites Program, 2013, http://dec.alaska.gov/spar/csp/sites/stlawrence.htm.

529 S. Byrne, P. K. Miller, V. Waghiyi, C. L. Buck, F. A. von Hippel and D. O. Carpenter, *J. Toxicol. Environ. Health, Part A*, 2015, **78**, 976–992.

530 R. J. Scrudato, J. Chiarenzelli, P. K. Miller, C. R. Alexander, J. Arnason, K. Zamzow, K. Zweifel, J. Gologergen, J. Kava, V. Waghiyi and D. O. Carpenter, *J. Local Global Health Sci.*, 2012, **2**, 1–12.

531 R. Jordan-Ward, F. A. von Hippel, G. Zheng, A. Salamova, D. Dillon, J. Gologergen, T. Immingan, E. Dominguez, P. Miller, D. Carpenter, J. H. Postlethwait, S. Byrne and C. L. Buck, *Sci. Total Environ.*, 2022, **826**, 154067.

532 F. A. von Hippel, P. K. Miller, D. O. Carpenter, D. Dillon, L. Smayda, I. Katsiadaki, T. A. Titus, P. Batzel, J. H. Postlethwait and C. L. Buck, *Environ. Pollut.*, 2018, **234**, 279–287.

533 ADEC, Alaska Department of Environmental Conservation Division of Spill Prevention and Response Contaminated Sites Program, 2013, http://146.63.9.103/Applications/SPAR/PublicMVC/CSP/SiteReport/207.

534 USACE, U.S. Army Corps of Engineers, 2012, Contract Number W911KB-12-C-003, FUDs Number F10AK0969-03.

535 USACE, United States Army Corps of Engineers, 2008.

536 USATSDR, Agency for Toxic Substances & Disease Registry, 2020.

537 R. Jordan-Ward, F. A. von Hippel, C. A. Wilson, Z. Rodriguez Maldonado, D. Dillon, E. Contreras, A. Gardell, M. R. Minicozzi, T. Titus, B. Ungwiluk, P. Miller, D. Carpenter, J. H. Postlethwait, S. Byrne and C. L. Buck, *Environ. Pollut.*, 2024, **340**, 122765.

538 S. Byrne, S. Seguinot-Medina, V. Waghiyi, P. K. Miller, C. Buck, F. A. von Hippel and D. Carpenter, *Environ. Pollut.*, 2017, **231**, 387–395.

539 G. Zheng, P. Miller, F. A. von Hippel, C. L. Buck, D. Carpenter and A. Salamova, *Environ. Pollut.*, 2020, **259**, 113872.

540 G. Welfinger-Smith, J. L. Minholz, S. Byrne, V. Waghiyi, J. Gologergen, J. Kava, L. Apatiki, E. Ungott, P. K. Miller, J. Arnason and D. O. Carpenter, *J. Toxicol. Environ. Health, Part A*, 2011, **74**, 1195–1214.

541 S. Byrne, S. Seguinot-Medina, V. Waghiyi, E. Apatiki, T. Immingan, P. Miller, F. A. von Hippel, C. L. Buck and

D. Carpenter, *Environ. Sci. Pollut. Res.*, 2022, **29**, 77145–77156.
542 D. O. Carpenter, A. P. DeCaprio, D. O'Hehir, F. Akhtar, G. Johnson, R. J. Scrudato, L. Apatiki, J. Kava, J. Gologergen, P. K. Miller and L. Eckstein, *Int. J. Circumpolar Health*, 2005, **64**, 322–335.
543 S. Byrne, P. Miller, S. Seguinot-Medina, V. Waghiyi, C. L. Buck, F. A. von Hippel and D. O. Carpenter, *Environ. Res.*, 2018, **166**, 537–543.
544 S. C. Byrne, P. Miller, S. Seguinot-Medina, V. Waghiyi, C. L. Buck, F. A. von Hippel and D. O. Carpenter, *Environ. Res.*, 2018, **166**, 537–543.
545 *Establishment in Canada of Warning and Control System against Air Attack*, U. S. Govt. Print. Off., Washington,, United States. and Canada, 1959.
546 J. P. Stow, J. Sova and K. J. Reimer, *Sci. Total Environ.*, 2005, **342**, 107–118.
547 R. Morenus, *Dew Line; Distant Early Warning, the Miracle of America's First Line of Defense*, Rand McNally, New York,, 1957.
548 P. W. Lackenbauer, M. J. Farish and J. Arthur-Lackenbauer, *The Distant Early Warning (DEW) Line: A Bibliography and Documentary Resource List*, The Arctic Institute of America, Washington, USA, 2005.
549 R. Jordan-Ward, F. A. von Hippel, C. A. Wilson, Z. Rodriguez Maldonado, D. Dillon, E. Contreras, A. Gardell, M. R. Minicozzi, T. Titus, B. Ungwiluk, P. Miller, D. Carpenter, J. H. Postlethwait, S. Byrne and C. L. Buck, *Environ. Pollut.*, 2024, **340**, 122765.
550 T. M. Brown, S. J. Iverson, A. T. Fisk, R. W. Macdonald, C. C. Helbing and K. J. Reimer, *Sci. Total Environ.*, 2015, **515–516**, 188–197.
551 T. M. Brown, S. Luque, B. Sjare, A. T. Fisk, C. C. Helbing and K. J. Reimer, *Environ. Sci. Technol.*, 2014, **48**, 13110–13119.
552 T. M. Brown, A. T. Fisk, C. C. Helbing and K. J. Reimer, *Environ. Toxicol. Chem.*, 2014, **33**, 592–601.
553 Z. A. Kuzyk, J. P. Stow, N. M. Burgess, S. M. Solomon and K. J. Reimer, *Sci. Total Environ.*, 2005, **351–352**, 264–284.
554 S. T. Wezeman, *SIPRI Background Paper*, 2016.
555 M. Farish, *Le. Géogr. Can.*, 2006, **50**, 177–196.
556 S. A. Fritz, *DEW Line Passage: Tracing the Legacies of Arctic Militarization*, University of Alaska Fairbanks, 2010.
557 J. Kjellén, *Arctic Review on Law and Politics*, 2022, **13**(1), 34–52.
558 M. Hird, *N. Rev.*, 2016, **42**, 22.
559 L. G. Whyte, L. Bourbonnière, C. Bellerose and C. W. Greer, *Biorem. J.*, 1999, **3**, 69–80.
560 D. A. Bright, W. T. Dushenko, S. L. Grundy and K. J. Reimer, *Sci. Total Environ.*, 1995, **160–161**, 265–283.
561 B. Braune, D. Muir, B. DeMarch, M. Gamberg, K. Poole, R. Currie, M. Dodd, W. Duschenko, J. Eamer, B. Elkin, M. Evans, S. Grundy, C. Hebert, R. Johnstone, K. Kidd, B. Koenig, L. Lockhart, H. Marshall, K. Reimer, J. Sanderson and L. Shutt, *Sci. Total Environ.*, 1999, **230**, 145–207.
562 M. J. Gunnarsdottir, *Environ. Sci.: Adv.*, 2024, **3**, 972–982.
563 T. M. Brown, T. A. Sheldon, N. M. Burgess and K. J. Reimer, *Environ. Sci. Technol.*, 2009, **43**, 7635–7642.
564 A. H. Buckman, C. S. Wong, E. A. Chow, S. B. Brown, K. R. Solomon and A. T. Fisk, *Aquat. Toxicol.*, 2006, **78**, 176–185.
565 B. DiNapoli and M. Jull, *Ambio*, 2024, **53**, 1109–1123.
566 G. W. Gabrielsen, I. G. Alsos and B. Brekke, *Undersøkelser Av PCB I Jord, Fisk Og Sjøfugl I Områder Rundt Avfallsfyllingen På Jan Mayen (FBT Lokalitet 1805001)*. Norwegian Polar institute, Tromsø, Norway, 1997.
567 M. Hird, *N. Rev.*, 2016, 23–45.
568 N. Skjegstad and G. W. Gabrielsen, *Forslag Til Miljømål for Jan Mayen Med Hensyn Til Håndtering Av PCB I Avfallsdeponi På Øya*, Norwegian Polar Institute (NPI), Tromsø, Norway, 1998.
569 M. E. Grandberg, A. Ask and G. W. G. Gabrielsen, *Local Contamination on Svalbard – Overview and Suggestions for Remediation Actions*, Norwegian Polar Institute (NPI), Tromsø, Norway, 2017.
570 Y. Li, S. Xiong, Y. Hao, R. Yang, Q. Zhang, F. Wania and G. Jiang, *J. Hazard. Mater.*, 2022, **434**, 128872.
571 E. M. Adams, F. A. von Hippel, B. A. Hungate and C. L. Buck, *Heliyon*, 2019, **5**, e02989.
572 P. Kikkert and P. W. Lackenbauer, in *The Palgrave Handbook of Arctic Policy and Politics*, ed. K. S. Coates and C. Holroyd, Springer International Publishing, Cham, 2020, pp. 487–505, DOI: 10.1007/978-3-030-20557-7_30.
573 E. C. Winfield, R. J. Rader, A. M. Zhivov, A. Dyrelund, C. Fredeen, O. Gudmundsson and B. Goering, *E3S Web Conf.*, 2021, **246**, 08004.
574 M. Goodsite, H. Skov, G. Asmund, O. Bennike, A. Feilberg, M. Glasius, A. Gross and M. H. Hermanson, Pilot study of contaminants near Station Nord, a military airbase and research station in NE Greenland, in *Sustainable Cities and Military Installations*, ed. I. Linkov, Springer, Dordrecht, 2014, pp. 177–198, DOI: 10.1007/978-94-007-7161-1_10.
575 M. Masiol and R. M. Harrison, *Atmos. Environ.*, 2014, **95**, 409–455.
576 Y. Lin, J.-J. Jiang, L. A. Rodenburg, M. Cai, Z. Wu, H. Ke and M. Chitsaz, *Environ. Int.*, 2020, **139**, 105699.
577 J. D. Singer and J. Keating, *New Polit. Sci.*, 1999, **21**, 325–343.
578 F. Fonnum, B. Paukstys, B. A. Zeeb and K. Reimer, *Environmental Contamination and Remediation Practices at Former and Present Military Bases*, Springer Science & Business Media, 2012.
579 H. Hung, C. Halsall, H. Ball, T. Bidleman, J. Dachs, A. De Silva, M. Hermanson, R. Kallenborn, D. Muir and R. Sühring, *Environ. Sci.: Processes Impacts*, 2022, **24**, 1577–1615.
580 R. Sühring, M. L. Diamond, S. Bernstein, J. K. Adams, J. K. Schuster, K. Fernie, K. Elliott, G. Stern and L. M. Jantunen, *Environ. Sci. Technol.*, 2020, **55**, 304–312.
581 M. Łuszczuk, in *Future Security of the Global Arctic: State Policy, Economic Security and Climate*, ed. L. Heininen,

Palgrave Macmillan UK, London, 2016, pp. 35–54, DOI: 10.1057/9781137468253_3.
582 R. Macdonald and J. Bewers, *ICES J. Mar. Sci.*, 1996, **53**, 537–563.
583 L. Heininen, K. Everett, B. Padrtova and A. Reissell, *Arctic policies and strategies – analysis, synthesis, and trends*, International Institute for Applied Systems Analysis (IIASA), Laxenburg, Austria, 2020, pp. 1–263, DOI: 10.22022/AFI/11-2019.16175.
584 O. R. Young, *Foreign Policy*, 1985, 160–179, DOI: 10.2307/1148707.
585 A. Clarke and C. M. Harris, *Environ. Conserv.*, 2003, **30**, 1–25.
586 P. Whitney Lackenbauer and M. Farish, *Environ. Hist.*, 2007, **12**, 920–950.
587 T. F. Jenkins, J. C. Pennington, T. A. Ranney, T. E. Berry, P. H. Miyares, M. E. Walsh, A. D. Hewitt, N. M. Perron, L. V. Parker and C. A. Hayes, *Characterization of Explosives Contamination at Military Firing Ranges (ERDC/CRREL Report TR-01-5)*, U.S. Army Engineer Research and Development Center, Cold Regions Research and Engineering Laboratory, Hanover, NH, 2001, pp. 1–36.
588 A. O. Fayiga and U. K. Saha, *Environ. Pollut.*, 2016, **216**, 135–145.
589 A. J. Barker, L. E. Mayhew, T. A. Douglas, A. G. Ilgen and T. P. Trainor, *Chem. Geol.*, 2020, **553**, 119797.
590 C. Fernandez-Lopez, R. Posada-Baquero and J.-J. Ortega-Calvo, *Sci. Total Environ.*, 2022, **843**, 157007.
591 W. T. Dushenko, S. L. Grundy and K. J. Reimer, *Sci. Total Environ.*, 1996, **188**, 29–38.
592 D. Muir, M. J. Gunnarsdóttir, K. Koziol, F. A. von Hippel, D. Szumińska, N. Ademollo, S. Corsolini, A. De Silva, G. Gabrielsen and R. Kallenborn, *Environ. Sci.: Adv.*, 2025, **4**, 355–408.