Akvaplan niva

Rapport Report

Literature survey of emerging contaminants in the Barents Sea and climate change effects on contaminant fate

I. Levels of emerging contaminants in the Barents Sea biota
II. Contaminants fate in a warming climate

Photo: J. Bytingsvik, Akvaplan-niva AS

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Akvaplan-niva AS
Rådgivning og forskning innen miljø og akvakultur
Org.nr: NO 937 375 158 MVA
Framsenteret
9296 Tromsø
Tlf: 77 75 03 00, Fax: 77 75 03 01
www.akvaplan.niva.no

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Literature survey of emerging contaminants in the Barents Sea and climate change effects on contaminant fate

Author(s)
Jenny Bytingsvik

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Summary

Based on novel literature (mainly 2012-2014), the main objectives of this report was (I) to give a brief description of the levels of emerging contaminants such as PFASs, novel brominated and chlorinated flame retardants (novel BFRs and CFRs) including chlorinated paraffins (CPs), phosphorous flame retardants (PFRs), current-use pesticides (CUPs), siloxanes, phenols and PAHs in the biotic environment of the Barents Sea area, and (II) to summarize existing knowledge on how climate change may influence the environmental fate of contaminants.

From the examined studies, species and contaminant groups included in this report, the levels in biota were highest for PFASs, siloxanes and CPs. The highest levels of PFASs were found in polar bear mothers and their cubs, ivory gull eggs, harbor seals and northern fulmar chicks, the highest levels of siloxanes were found in sculpin and Atlantic cod (relatively low levels in seabirds and marine mammals), and the highest levels of CPs were found in Arctic charr, little auk, kittiwake and Greenland shark. Levels of novel BFRs, the CFR dechlorane plus, PFRs, CUPs, phenols and PAHs were considerably lower than the levels of PFASs, siloxanes and CPs. .

The way climate change may influence on contaminants fate in the environment is complex. To date, a combination of modelling and observations show that the atmospheric transport, deposition and remobilization, bioavailability, bioaccumulation and biomagnification, degradation, toxicology and toxic effects of contaminants in different ways and through different mechanisms may be affected by the increasing temperatures.

As this literature survey includes contaminant measurements in a wide range of species, tissues, age groups and genders, comparisons and evaluations to give overall conclusions on the contaminant status in the Barents Sea biota is difficult. The increased use of and lack of knowledge concerning emerging contaminants, the increased anthropogenic activity in the Barents Sea, and the potential influence of climate change on contaminants fate in the environment emphasizes the importance and need to conduct well-planned monitoring studies. Future monitoring studies should involve an optimal study group (e.g. species, trophic levels, age, gender) with the aim to increase the knowledge about bioaccumulation and biomagnification of emerging contaminants in the Barents Sea region.

Project manager

Jenny Bytingsvik, Akvaplan-niva AS

Jenny Bytingsvik

Quality controller

Anita Evenset, Akvaplan-niva AS

Anita Evenset

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Preface

As a part of the Joint Norwegian-Russian Commission on Environmental Cooperation focussing on protecting the northern ocean environment, Norway and Russia are a part of ocean environment group. The overall goal of this political collaboration between the Norwegian Ministry of Climate and Environment and Russian Ministry of Natural resources and Ecology, is to maintain the clean and rich environment in the Barents Sea, as this is a part of a joint and highly productive ecosystem. Through this collaboration, the Norwegian Environment Agency (Miljødirektoratet) is involved in a project where one of the objectives is to continuously assess the Barents Sea pollution status (HAV21). Based on novel scientific results, mainly published between 2012 and 2014, the main aims of this literature survey is to give a brief description of the biotic levels of emerging contaminants in the Barents Sea ecosystem, and to describe how climate change may influence on the environmental fate of contaminants. The literature survey was conducted by Akvaplan-niva AS, Tromsø.

Tromsø 28.11.14

Jenny Bytingsvik
Post Doc. /Researcher
Akvaplan-niva AS

1 Assignment

The main objectives of this report is to give a brief description of the levels of emerging contaminants in the biotic environment of the Barents Sea (Part 1), and to summarize existing knowledge on how climate change may influence the environmental fate of contaminants (Part 2). This literature survey is mainly based on available and relevant Norwegian, Russian and other scientific literature and monitoring reports published between 2012 and 2014. Literature published before 2012 is considered to be important if recent publications covering similar topics are not assessable.

The Barents Sea covers the area between the Norwegian and Russian mainland in the south, Franz Josef Land in the north, Spitsbergen, Edge Island and Bear Island in Svalbard in the west, and Novaya Zemlya in the east (Fig. 1). In part 1 of the report comprising the levels of emerging contaminants, the focus will be limited to the biotic environment of the Barents Sea area and the surrounding islands (e.g. Svalbard including Bear Island, Franz Josef Land and Novaya Zemlya). Although Lofoten and the Tromsø-region is located south-west of what most commonly is defined as the Barents Sea, the report have included some central studies from these areas. As the term "emerging contaminants" refer to various compounds in published studies, we chose to focus on the following emerging contaminants which we found relevant for the Barents Sea region: perfluoralkyl substances (PFASs), flame retardants such as novel brominated flame retardants, chlorinated flame retardants (CFRs) such as for instance short-chained chloroparaffins (SCCPs), organophosphorous flame retardants (OPFRs), current-used pesticides (CUPs), siloxanes, phenolic compounds and PAHs. In part 2 of the report comprising the potential influence of climate change on environmental fate of contaminants, a more general description of the influence of climate change on contaminant fate will be given as many of these processes are comparable between various arctic areas. The majority of examples of studies where results have been suggested to reflect a climate effect are from the Barents Sea region.

2 Background

The Barents Sea is a part of the Arctic Ocean with the Norwegian-Russian border stretching from south to north (Figure 1.1). Due to being rich in natural resources, such as fish and petroleum, the Barents Sea is an ecosystem of high economic value.

While the majority of legacy persistent organic pollutants (POPs) show declining levels in the arctic environment, the presence of novel contaminants are increasing in the Arctic. The milder climate and reduced spatial and temporal distribution of sea ice covers, has made the area more accessible for various activities, such as increased petroleum activity as well as tourism. While the presence of many of the emerging contaminants are due to being used as replacement chemicals for phased-out chemicals (e.g. brominated flame retardants and pesticides), the presence of others are related to increased industrial activity and human settlements in northern areas. Climate change may also affect the contaminants fate in the environment, and in some cases cause increasing levels in the biotic environment. Climate change may influence on processes governing the fate of contaminants in the environment, such as wind systems, ocean currents, precipitation, melting of sea ice, glaciers, and ice caps, reducing the permafrost, affect frequency of extreme weather, degradation and transformation of contaminants, degree of re-volatilization of contaminants from secondary sources, partitioning of contaminants between air-water and other phases, biodiversity and biotic transport of contaminants. As the emerging contaminants included in the survey represent a heterogeneous group with regard to chemical structure, physic-chemical properties, and use, and they originate from everything from industrial applications, agriculture, personal care products, as well as being by-products from industrial processes, monitoring the contaminant situation in the Barents Sea, as well as other areas is complex, in particular when taking climate change effects into account. The changes the Arctic environment experience emphasizes the importance of a continuous monitoring of the environment to make sure contaminant levels are kept below threshold levels set to protect our future resources.

Figure 1.1: Barents Sea (Source: Google maps).

3 Emerging contaminants in the Barents Sea

3.1 Perfluoralkyl substances (PFASs)

PFASs are anthropogenic chemicals manufactured since the 1950s (Prevedouros et al., 2006; Buck et al., 2011), with a ubiquitous environmental distribution (Giesy and Kannan 2001; Butt et al., 2010). PFASs are present in the Arctic, yet, their fate in the environment and transport pathways to arctic areas are still not well understood. The PFASs most commonly detected in biota are divided into two groups based on their chemical structure: perfluorinated carboxylic acids (PFCAs) and perfluorinated sulfonic acids (PFSAs) (Butt et al., 2010; Buck et al., 2011). Their physicochemical properties are highly different from legacy POPs such as PCBs, as the fluorocarbon side chains are hydrophobic and the functional groups (carboxylic acid for PFCAs and sulfonic acid for PFSAs) are hydrophilic. Their strong carbon-fluorine (C-F) bonds and their amphipathic nature makes PFASs highly resistant against degradation (by heat, acids, bases, reducing and oxidizing agents) and useful in numerous industrial and commercial applications, such as in surfactants and polymers. PFASs are used in coating of food-contact papers, pots, pans and textiles to obtain stain repellent, non-stick surfaces or waterproof surfaces, in addition to its use in grease-proof, fire-fighting foams, lubricants, insecticides, paint, metal plating and cleaners (Prevedouros et al., 2006; Buck et al., 2011). Compared to the previous use of PCBs (i.e. industrial use), PFASs are more frequently used in everyday products.

Despite the limited number of studies of PFASs in the European Arctic, PFASs have been detected in several arctic species associated with the Barents Sea region (Bytingsvik et al., 2012; Nost et al., 2012; Routti et al., 2014) (Table A.1). PFASs have been detected in black-legged kittiwake chicks (*Rissa tridactyla*) and northern fulmar chicks (*Fulmarus glacialis*) from Kongsfjorden (Svalbard) (Nost et al., 2012), in harbour seals (*Phoca vitulina*) from Svalbard (Routti et al., 2014), and in polar bears (*Ursus maritimus*) and their cubs-of-the-year from Svalbard (Bytingsvik et al., 2012) (Table A.1). Levels were measured in plasma in all these studies, and perfluorooctane sulfonate (PFOS) was the dominating PFAS-compound in all species. The highest levels of PFCAs and PFSAs were found in adult polar bears. Levels of PFCAs in suckling polar bear cubs were comparable to levels in northern fulmar chicks, and adult and juvenile harbour seals. Lowest levels were found in black-legged kittiwake chicks. For PFSAs, the levels in polar bear cubs were comparable to levels in

northern fulmar chicks, but higher than levels in adult and juvenile harbour seal. Also for PFSAs the levels were lowest in black-legged kittiwake. PFASs have also been measured in ivory gull eggs (*Pagopila eburnean*) from the Norwegian and Russian Arctic (Miljeteig et al., 2009), as well as in eggs of common eider (*Somateria mollissima*), European shag (*Phalacrocorax aristotelis*) and herring gull (*Larus argentatus*) from Røst (Nordland, Norway) (Huber et al., 2014), and Atlantic cod (*Gadus morhua*), common eider, herring gull and harbour seals from the northern parts of Norway (Herzke et al., 2013) (Table A.1). Of the measurements done in ivory gull eggs, minor differences were seen between the examined locations in the Norwegian and Russian Arctic. The PFAS-levels in eggs were somewhat higher in the ivory gulls from the Norwegian and Russian Arctic compared to in the herring gulls from the Norwegian mainland, and the levels in ivory gull eggs and most of the herring gull eggs were higher than in the common eider eggs (Table A.1).

Another interesting and in part expected finding was that plasma levels of PFASs in suckling polar bear cubs in general were lower than in the plasma of their mothers. This is opposite of what is seen for highly hydrophobic compounds such as PCBs and is most likely a result of the differences between PCBs and PFASs physic-chemical properties (e.g. hydrophobicity, molecular shape and size). The reason for comparable levels of PFCAs and PFSAs in juvenile and adult harbour seal could be that pups of harbour seal only nurse for a few weeks before starting to feed and consume food comparable to adult seals.

In polar bears, the levels of most PFCAs, and in particular the longer-chained PFCAs, were higher in 2008 compared to 10 years earlier. These findings are in accordance with a recent study on marine mammals, such as ringed seals (*Phoca hispida*), white sided dolphin (*Lagenorhynchus acutus*) and pilot whales (*Globicephala melas*) in the North Atlantic and West Greenland, sampled between 1984 and 2006 (Rotander et al., 2012). Hence, PFASs, and in particular long-chained PFCAs should still be monitored.

3.2 Novel flame retardants

The three main groups of brominated flame retardants (BFRs) previously studied have been polybrominated diphenyl ethers and biphenyls (PBDEs and PBBs), hexabromocyclododecanes (HBCDs) and tetrabromobisphenol A (TBBPA). The pentabrominated PBDEs (penta-BDEs) and octabrominated PBDEs (octa-BDEs) were banned in the EU and in Norway in 2004, and since 2009 these groups of PBDEs have been included in the Stockholm Convention. The use of decabrominated PBDEs (deca-BDE) has been

banned in electrical and electronic applications since 2008, and since 2012 it has been classified as a persistent, bioaccumulating and toxic compounds (PBT-compound) and to be very persistent and very bioaccumulating (vPvB) by the European Chemicals Agency (ECHA). Deca-BDE as well as HBCDs is considered a global risk, and while HBCD was included in the Stockholm Convention in 2013, Norway has suggested to also including deca-BDE. The last decade, the use of other flame retardants, referred to as "novel BFRs" and "chlorinated flame retardants (CFRs)", has increased as they have been used as replacements for these PBDE-groups. Although little is known about use and production volumes of novel flame retardants, it has been estimated that novel BFRs alone account for about 25% of the global production of flame retardants, with a growth of about 5% per year (Covaci et al., 2011).

3.2.1 Novel brominated flame retardants (novel BFRs)

Several novel BFRs have been detected in the Arctic environment indicating long-range atmospheric transport. These compounds are also of concern due to their high octanol-water coefficient (log *Kow*: 4.0-12.6) which is associated with bioaccumulating and biomagnifying abilities (Covaci et al., 2011; Vorkamp and Riget 2014). However, evidence of bioaccumulation is sparse. Some of the most studied novel BFRs are 2,3-dibromopropyl - 2,4,6-tribromophenyl ether (DPTE), allyl-2,4,6-tribromophenylether (ATE) and 2-bromoallyl 2,4,6-tribromophenyl ether (BATE), 1,2-bis(2,4,6-tribromophenoxy)-ethane (BTBPE), decabromodiphenyl ethane (DBDPE), bis(2-ethylhexyl)tetrabromophthalate (TBPH or BEHTBP), 2-ethyl-hexyl-2,3,4,5-tetrabromo benzoate (TBB), pentabromoethylbenzene (PBEB), pentabromophenol (PBP), 2,4,6-tribromophenol (TBP or 2,4,6-TBP), and 2,3,5,6-tetrabromo-*p*-exylene (TBX). While ATE and BATE are believed to be degradation products of BTBPE, DBDPE, TBPH and TBB are the major replacement products for PBDEs (Covaci et al., 2011; Vorkamp and Riget 2014).

Although to a varying extent dependent on compound, novel BFRs have been detected in fish, birds and marine mammals in the Barents Sea (Table A.1) (Reth et al., 2006; Herzke et al., 2013; Strid et al., 2013; Huber et al., 2014) (Table A.1).

An interesting compound is DBDPE. DBDPE has a molecular structure relatively similar to deca-BDE (BDE-209), and hence, the physic-chemical properties are also presumably similar. DBDPE has a low volatility, low water solubility and a high *Kow*-value. Although BDE-209 is hardly detectable in Arctic biota, there are studies indicating that

DBDPE might biomagnify (Vorkamp and Riget 2014). In the Barents Sea region, DBDPE has in a recent screening study been detected in several species representing several trophic levels, such as Atlantic cod, polar cod (*Boreogadus saida*), seabirds, such as kittiwake and glaucous gulls (*Larus hyperboreus*), ringed seal and polar bears, in addition to terrestrial birds (Herzke et al., 2013) (Table A.1). DBDPE and other novel BFRs have also been detected in egg of common eider, European shag and herring gull from another screening study where eggs were sampled for instance at Røst (Nordland, Norway) (Huber et al., 2014) (Table A.1). In general, levels of novel BFRs appear to be low compared to for instance levels of PFASs.

DPTE have been found in concentrations as high as 470 ng/g wet weight (ww) and 340 ng/g ww in blubber and brain, respectively, of harp seals (*Pagophilus groenlandica*) from the Barents Sea/Greenland Sea sampled in the early 1990s, and DPTE-levels exceeded the levels of PBDEs (von der Recke and Vetter 2007). Although the sampling was done in the early 1990s, this compound is of concern both due to the high levels found in biota and because it appears to be the dominating compound in brain tissue of for instance harp seals (von der Recke and Vetter 2007).

TBP have multiple applications in the chemical industry, it is a reactive flame retardant, used e.g. as a fungicide (e.g. wood treatment). It is produced in large amounts in the U.S., China and Japan. Although TBP is known to be of direct anthropogenic origin, it is also a metabolite of PBDEs, hydroxylated- and methoxy-PBDEs, a by-product in the production of other novel BFRs, and might be produced naturally by marine algaes (Covaci et al., 2011). Its presence in the marine environment has been suggested to mainly be a result of natural production (Covaci et al., 2011).

3.2.2 Chlorinated flame retardants (CFRs)

3.2.2.1 Dechlorane plus

Dechlorane plus is a non-regulated and highly chlorinated flame retardant. It was considered a good replacement for deca-BDE. It has also been used as a replacement for the insecticide Mirex. Despite being used since the 1960s, there has been few studies on the distribution and effect of dechlorane plus. A recent review has concluded that dechlorane plus is a global pollutant with the potential for long-range transport (Na et al., 2014; Vorkamp and Riget 2014). Levels of dechlorane plus in seawater, sediments, soil, moss, dung and the atmosphere is lower in Ny-Ålesund (Svalbard) than in more southern latitudes of Europe and Asia (Na et al., 2014). Although it has been detected in black guillemot eggs from the Faroe

Islands (Vorkamp and Riget 2014), and has shown biomagnification in a non-arctic food-chain (Tomy et al., 2007), the evidence of bioaccumulation in the Arctic is sparse. Dechlorane plus have been measured in herring gull eggs from Røst (Nordland, Norway), but not in common eider and European shag eggs from the same location (Huber et al., 2014). However, the levels in the gull eggs were low (<0.08 ng/g ww), and most eggs had levels below the detection limit. Attempts to measure dechlorane plus in the Greenland shark (*Somniosus microcephalus*) from the North-East Atlantic were negative as levels were below detection limits (Strid et al., 2013) (Table A.1). Although the sharks were sampled around Island, their migration capacity (up to 1000 km in 59 days, Lydersen and Kovacs, 2014) indicate that they might have a home range that also covers the Barents Sea region.

3.2.2.2 Chlorinated paraffins (CPs)

CPs is a large group of compounds commonly divided into sub-groups based on their degree of chlorinated and molecular length such as short-chained chlorinated paraffins (SCCPs), medium-chained chlorinated paraffins (MCCPs) and long-chained chlorinated paraffins (LCCPs). SCCPs, and to some extent also MCCPs, are most relevant for the Arctic as these relatively small compounds are known to be transported from industrial areas to the Arctic by long-range transport. SCCPs have mainly been used as lubricants and coolants in metal cutting and metal forming operations, but also as plasticizers and flame-retardant additives in rubber, paints and other coating (EPA, 2009). SCCPs are defined as a vPvB group of compounds, and as a persistent, bioaccumulates (log *Kow*: 5.2-9), and toxic compounds (PBT-compound) by the REACH-regulation. SCCPs are included in the candidate list of the European Union, and it has been suggested to include SCCPs in the Stockholm convention. SCCPs are also listed in the "Long-Range Transboundary Air Pollution" (LRTAP) POPs protocol. Air measurements from 2013 at the Zeppelin station (Ny-Ålesund, Svalbard) confirm the presence of CPs in arctic air (Nizzetto et al., 2014). Although measuring CPs is an analytical challenge and results should be interpreted with care, CPs are found at the same levels as PAHs which is approximately three orders of magnitude higher than the concentrations of legacy POPs (i.e. PCBs) (Nizzetto et al., 2014).

Although CPs have been produced since the 1930s, and their ability to bioaccumulate was known several decades ago, little monitoring data is available for these compounds because standard and well-established methods to analyse CPs remain to be developed. However, in Norwegian Screening Programs, SCCPs and MCCPs have been measured above detection limits in several arctic species such as polar cod (whole fish), Atlantic cod (liver),

common eider (eggs), European shag (eggs), kittiwake (eggs), herring gull (eggs), glaucous gulls (plasma), ringed seals (plasma) and polar bears (plasma) sampled at Svalbard in 2012 (Herzke et al., 2013; Huber et al., 2014) (Table A.1). In the screenings, the highest levels of SCCPs were found in Atlantic cod liver (10.3 ng/g ww), while the highest levels of MCCPs were found in common eider eggs (17.5 ng/ml ww) (Herzke et al., 2013; Huber et al., 2014). For reported levels available only in lipid normalized concentrations, the levels of SCCPs have been found in concentrations as high as 430 ng/g lipid weight (lw) in Greenland shark (Strid et al., 2013), and levels of MCCP as high as 3 700 ng/g lipid weight (lw) in little auk (*Alle alle*) and 1 600 ng/g lw in Arctic charr (*Salvelinus alpinus*) (Table A.1).

3.2.3 Phosphorous flame retardants (PFRs or OPFRs)

Phosphorous flame retardants (PFRs), also referred to as organophosphorous flame retardants (OPFRs), have a broad field of application in addition to being used as flame retardants. PFRs have been used as plasticisers and anti-foaming agents, in particular in polyurethane foam, in addition to in paint, building materials, hydraulic fluids, lacquers, glues, electronics, furniture, textiles and floor finisher waxes. As the novel BFRs and CFRs, PFRs have also been used as replacement chemicals for PBDEs as they were taken off the market, and hence, the use of PFRs has increased the last decade. It is mainly the halogenated PFRs that have been used as flame retardants, and they are additive, meaning they are not covalently bound to the material they are applied to. Additive flame retardants have a greater tendency to leak into the environment than reactive flame retardants. PFRs are persistent, they bioaccumulate, are toxic, and as they are present in the Arctic, they can also undergo long-range transport. Although tris(2-chloroethyl) phosphate (TCEP) is included in the candidate list of the European Union, there are few regulations covering the production and use of novel BFRs.

In the Barents Sea, several Norwegian screening programs have included PFRs (Evenset et al., 2009; Sagerup et al., 2011). In the most recent screening (Sagerup et al., 2011), the following compounds were analysed: tris(isobutyl)phosphate (TiBP), tris(n-butyl)phosphate (TBP), TCEP, tris(chloropropyl) phosphate (sum 3 isomers) (TCPP), dibutylphenyl phosphate (DBPhP), butyldiphenyl phosphate (BdPhP), tris(1,3-dichloro-2-propyl) Phosphate (TDCPP), tris(2-butoxyethyl) phosphate (TBEP), triphenyl phosphate (TPP), ethylhexyl diphenyl phosphate (EHDP), tris(2-ethylhexyl) phosphate (TEHP), tri-o-Tolylphosphate (ooo-TCP), tri-m-Tolylphosphate (mmm-TCP), tri-p-Tolylphosphate(ppp-

TCP), Sum-TCP (all isomers including those that are o,m,p mixed), and tris(2,3-dibromopropyl)phosphate (TBPP).

Although PFR-levels are low in the samples collected at Svalbard between 2007 and 2010, PFRs have shown to be present at several trophic levels represented by capelin (*Mallotus villosus*, whole fish), seabirds, such as black-legged kittiwake (liver), Brünnich guillemot (*Uria lomvia* eggs), glaucous gull (eggs), ringed seal (blubber), harbour seal (plasma), arctic fox (*Vulpes lagopus*, liver) and polar bears (plasma) (Sagerup et al., 2011) (see report for detailed overview of PFR-levels). Briefly, the 2011-report show that seven of the 14 compounds were present in >30% of the samples within at least one of the species. Elevated levels of TBEP were found in Arctic fox (132.4 ng/g ww), while levels of the remaining PFRs in all other species were detected in concentrations <16.3 ng/g ww (see report for detailed overview of PFR-levels). In the report from 2009, PFRs was also detected in Polar cod, Atlantic cod, Arctic charr, and common eider. Recently, egg of common eider, European shag and herring gull sampled in 2012 for instance at Røst (Nordland, Norway) was screened for PFRs (Huber et al., 2014) (Table A.1). The highest levels were found in the European shag eggs (64.5-95.6 ng/g ww). Beside these screening reports from 2009 and 2011, no studies have been conducted on PFRs in the Barents Sea biota.

3.3 Current-use pesticides (CUPs)

Pristine areas, such as the Arctic, have very few or no local sources of agricultural chemicals such as pesticides. Although we know that significant amounts of phased-out legacy pesticides (e.g. DDTs, chlordanes, Mirex, toxaphene) have been transported to Arctic areas by long-range transport (e.g. air, ocean), little is known about how most CUPs are distributed globally. CUPs such as dacthal, trifluralin, pentachlorophenol (PCP), pentachloronitrobenzene (PCNB), pentachloroanisole (PCA), chlorothalonil, chloropyrifos, diazinon, endosulfans, methoxychlor, and dicofol are still found in regions far from their emission sources, and many of them have been detected in air, snow, and ice caps at Svalbard (Norway) (Hoferkamp et al., 2010; Ruggirello et al., 2010; Weber et al., 2010; Vorkamp and Riget 2014).

In a review by Vorkamp and Riget (2014) it was concluded that the data material available on CUPs in the Arctic from recent years have not been substantially extended since reviews by Hoferkamp et al. (2010) and Weber et al. (2010) were published in 2010. The

number of publications covering CUPs in the Arctic biota is low, and hence, even fewer have been published from the Barents Sea.

In 2012, a screening for the occurrence of various novel contaminants in the Norwegian environment, including Svalbard, was conducted (Langford et al., 2012). The screening covered current-use pesticides, such as chlorpyriphos, methoxychlor, trifluralin, dacthal, and pentachlorophenol (PCP). Chloropyrifos and methoxychlor have previously been used as insecticides, but chloropyrifos was prohibited in the U.S. in 2000 and the production of methoxychlor ended about ten years ago (Vorkamp and Riget 2014). Trifluralin is still used in the U.S., India, Japan and China, while it was banned in the EU in 2009 (Vorkamp and Riget 2014). While the levels of chlorpyriphos and methoxychlor have declined in the abiotic arctic environment since the 1980s and 1990s, respectively, trifluralin shows no clear trends (Vorkamp and Riget 2014). PCP is a pesticide currently used as a fungicide and herbicide to protect wood. However, it has also been suggested that PCP found in mammals may be a metabolite of HCB or pentachloroanisole (PCA) of which the latter is an environmental transformation product of PCP, simply a methylated variant of PCP (Hoekstra et al., 2003; Routti et al., 2009; Hoferkamp et al., 2010; Vorkamp and Riget 2014).

The Arctic samples of whole polar cod and capelin, seal blubber, kittiwake and glaucous gull blood/plasma, and eider duck and common guillemot (*Uria aalge*) eggs collected in Svalbard in 2007, 2010 and 2011 contained detectable levels of chlorpyriphos, methoxychlor, trifluralin, dacthal, while levels of PCP were below the limit of detection (Langford et al., 2012) (Table A.1). A single sample of ringed seal blubber contained 1.4 ng/g of chlorpyrifos (Langford et al., 2012) (Table A.1). Also egg of common eider, European shag and herring gull sampled in 2012 for instance at Røst (Nordland, Norway) have been screened for CUPs such as for instance trifluralin and methoxychlor (Huber et al., 2014). However, levels were below detection limits in all eggs except one common eider egg which contained 0.37 ng/g ww methoxychlor.

A recent study show that the plasma levels of PCP (<1 ng/g ww) and 4-OH-HpCS (~0.1 ng/g ww) in harbour seal from Svalbard are relatively low (Routti et al., 2014) (Table A.1). 4-OH-HpCS is a metabolite of the industrial by-product octachlorostyrene, and can be formed in incomplete combustion processes of chlorinated compounds and during manufacturing of chlorinated compounds. Monitoring PCP and 4-OH-HpCS is still relevant, in particular as the decreasing level of HCB in the air at Svalbard and other Arctic areas has

levelled off the last decade and that elevated levels have been measured for some years after 2000 (Hung et al., 2010). It has been suggested that this could be due to increased worldwide use of certain pesticides (e.g. clorothalonil, quintozene) which are contaminated with HCB.

The fungicide pentachloroanisole (PCA) has been measured in amphipods, marine invertebrates, marine fish, and marine mammals in other Arctic areas than the Barents Sea (Vorkamp and Riget 2014). Although findings differed from a typical biomagnification pattern, levels in the amphipods were up to 20 ng/g lw. PCA has never been examined in the Barents Sea biota. The fungicides pentachloronitrobenzene (PCNB) and chlorothalonil and the insecticide diazinon has been measured in the abiotic arctic environment, but never in Barents Sea biota (Vorkamp and Riget 2014). PCNB and chlorothalonil are still used in the U.S. and Europe, while diazinon is banned in the EU, but still used in the U.S.

One of the few studies on levels of methoxychlor in the European Arctic biota from recent years was on ringed seal from the Russian Arctic (Savinov et al., 2011). Levels in the seals blubber were below the detection limit in seals from the Barents Sea and below 8.4 ng/g lw in seals from the White Sea (Savinov et al., 2011) (Table A.1). Methoxyclor has also been detected in glaucous gull liver sampled in the Barentsburg area at Svalbard in 2001 (Females 1.2±1.4 ng/g ww, Males 0.6±0.7 ng/g ww) (Savinov et al., 2005) (Table A.1).

Endosulfan (ES) is an organochlorine pesticide used for the last five decades with a widespread use in most parts of the world. The commercial mixture consists of two isomers, α- and β-ES, of which the α-ES is most common. ES is listed by the Stockholm convention as a POP due to the compounds bioaccumulating character and persistent nature, and listed as a "Priority Hazardous Substance" in the EU Water Framework Directive. Today, India and Israel are the largest producers of ES. The last three decades, levels of ESs have increased in the Arctic abiotic environment and it is one of the main pesticides in Arctic air (Vorkamp and Riget 2014), emphasising the need to monitor this compound further. Although the two isomers differ in their physico-chemical properties, both are acute toxic to aquatic organisms, and they are capable of long-range transport due to their semi-volatile character. Despite this knowledge, few studies have been conducted to examine ES-levels in Arctic biota, in particular in the Norwegian and Russian Arctic.

One of the few studies on level of ESs in the European Arctic biota from recent years was on ringed seal from the Russian Arctic (Savinov et al., 2011). Levels of α- and β-ES in

the seals blubber were below 1.8 ng/g lw in seals from the Barents Sea and below 3.1 ng/g lw in seals from the White Sea (Savinov et al., 2011). Higher levels have been detected in marine fish (50 ng/g lw in capelin) from Greenland (Vorkamp and Riget 2014). Although the few studies on ES in Arctic biota (mainly from Canada) show levels below toxicity thresholds in birds and mammals, there is a clear need to analyse ES in the European Arctic biota. In particular, this concerns measuring ES-levels in lower trophic levels, as well as to examine the bioaccumulation potential of ES in a well-defined arctic food-web.

3.4 Siloxanes

Organosiloxanes are a broad group of chemicals produced in large volumes in Europe. One subgroup of organosiloxanes is the cyclic volatile methyl siloxanes (cVMS). cVMS are persistent, bioaccumulative and toxic compounds widely used in personal care products including lotions, cosmetics and deodorants. cVMS are also used in industrial applications as solvents, lubricants, coatings and sealants (Wang et al., 2013). The wide application of siloxanes is related to their physical/chemical properties such as their low surface tension, high thermal stability, and inert properties. The same properties also cause concern regarding their environmental persistence. The cVMS congeners that are environmental persistent are octamethylcyclotetrasiloxane (D4), decamethylcyclopentasiloxane (D5) and dodecamethylcyclohexasiloxane (D6). Volatilization of cVMS to the atmosphere is the dominant mode of emission to the environment due to their high vapour pressure and low water solubility (log *Kow* 6.5-9.1) (Xu and Kropscott 2012). These properties make cVMS undergo long-range transport, at the same time as they show a low deposition potential.

cVMS have been included in previous screening programs examining their presence in the abiotic and biotic environment at Svalbard (Evenset et al., 2009; Nizzetto et al., 2014). Air measurements from 2013 at the Zeppelin station (Ny-Ålesund, Svalbard) show that these pollutants are found at the same levels as PAHs which is approximately three orders of magnitude higher than the concentrations of legacy POPs (i.e. PCBs) (Nizzetto et al., 2014). Further, no cVMS were found in sediments, and the wet weight levels were higher in fish than in seabirds (opposite in lipid weight) (Evenset et al., 2009) (Table A.1). It was suggested that the lower levels in the seabirds compared to fish could relate differences in respiration (water vs. air), and that birds have a significant loss of cVMS through expiration. In a recent screening of eggs of common eider, European shag and herring gull sampled in 2012 for instance at Røst (Nordland, Norway) (Huber et al., 2014) (Table A.1). cVMS-levels in the

examined eggs were lower than seen in fish species from Svalbard (Table A.1), and hence, confirm the previous statement that levels often are lower in air-breathing animals such as birds, compared to water-breathing species such as fish. In another study by Warner et al., (Warner et al., 2010), examining levels of cVMS in sediments and biota at Svalbard, the levels were also higher in fish (i.e. shorthorn sculpin and Atlantic cod) compared to organisms higher in the food chain, such as bearded seals. Warner concluded that there was a low risk of cVMS accumulation within mammals. Levels in sediments decreased with increasing distance from the settlement in Longyearbyen, and levels were below detection limit in Kongsfjorden and near the less populated settlement Ny-Ålesund. This confirms that the settlement in Longyearbyen is a point source of cVMS at Svalbard.

In Atlantic cod sampled near the community of Tromsø city in northern Norway (Tromsøysundet) D5 was the dominating compound in cod liver, and the levels were up to10 times higher than levels of PCB-153 and -180 which are among the most common PCB-congeners (Warner et al., 2014). At a location 30 km northeast of the city (Nipøya), the levels of D5 were approximately 5 times lower, but also in these cods, D5 dominated over D4 and D6 (Table A.1). This was expected, as cod closer to the city is more impacted by wastewater effluent and other anthropogenic activity where it is likely that cVMS are present. cVMS levels were considerably higher in the cod sampled around Tromsø compared to levels in cod sampled at Svalbard (Table A.1).

3.5 Phenols

In 2012, a screening for the occurrence of contaminants in the Norwegian environment, including Svalbard, was conducted (Langford et al., 2012). The screening covered several groups of contaminants, including alkylphenols and bisphenol A. Levels of 4-n-octylphenol (4nOP, straight chain), 4-tertiary-octylphenol (4tOP, branched), octylphenol monoethoxylates (OP1EO), nonylphenol (4-n-nonylphenol (4nNP), straight chain and technical nonylphenol (tNP), branched chain), nonylphenol monoethoxylate (NP1EO), were below the limits of detection in all the Arctic samples collected in Svalbard in 2010/2011 (whole polar cod and capelin, seal blubber, kittiwake and glaucous gull blood/plasma, and eider duck and common guillemot eggs) (Langford et al., 2012).

BPA is a phenolic brominated flame retardant commercially produced the last 50-60 years. It is mainly used as a component in polymer consumer products, such as plastic bottles, sports equipment, dental filling and household electronics (Rubin 2011). TBBPA is a BPA

derivate mainly used as a flame retardants in electronic equipment. While BPA is broken down rather easily in water, TBBPA is a more persistent compound which also has been detected in arctic air samples (Covaci et al., 2011; Rubin 2011). Both are endocrine disrupting chemicals with the potency to interrupt with reproductive processes. No studies have analysed biota sampled from the Barents Sea for bisphenol A (BPA) or tetrabromobisphenol A (TBBPA) since the screening in 2009 (Evenset et al., 2009). However, the report, which covered various marine fish and seabirds species sampled in the fjords on the north-western parts of Svalbard in 2008, Arctic charr from Bjørnøya sampled in 2004 and sediment samples from the western parts of the Barents Sea, stated that none of the samples contained BPA or TBBPA above the detection limit. However, BPA has previously been detected in sediment sampled from the eastern and southern parts of the Barents Sea (Bakke et al., 2008). Follow-up studies of these findings including biota sampled from the same areas have not been conducted. Besides banning the use of BPA in baby bottles, no further regulations have been initiated in the EU. Hence, BPA is still produced and used in significant amounts. However, BPA is being evaluated both by ECHA and EFSA. Although TBBPA is defined as an environmental pollutant, no regulations in the production and use have come into force. The use of TBBPA has increased after the ban of penta-BDEs and octa-BDEs. BPA and TBBPA should be kept on monitored in the Barents Sea area due to increased anthropogenic activity.

Levels of BPA and TBBPA was also below detection limits in the screening of eggs of common eider, European shag and herring gull sampled in 2012 for instance at Røst (Nordland, Norway) (Huber et al., 2014). The eggs were also screened for several other bromophenols such as 2,4-bromophenols (DBP), TBP (see also paragraph 3.2.1), PBP (see also paragraph 3.2.1), as well as the following alkylphenols; 4-t-pentylphenol, 4-n-pentylphenol, 4-t-octylphenol, 4-n-octylphenol, 4-n-heptylphenol, 4-iso-nonylphenol, 4-t-octylphenol monoethoxylates (4-t-OPEO1), and C10-iso-phenol (Huber et al., 2014). Briefly, DBP was measured in one third of the samples, TBP in all sampled and 4-iso-nonylphenol in half of the samples (Table A.1). Of the detected compounds, 4-iso-nonylphenol was measured in the highest concentrations, and the levels were higher in the common eider and herring gull eggs compared to the European shag (Table A.1).

3.6 PAHs

PAHs originate from incomplete combustion processes of organic material, they travel long distances in the atmosphere, and are toxic to animals and humans. Hence, PAH-emission is still ongoing. Atmospheric transport has been demonstrated to be the main route for PAHs

to reach pristine areas such as the Arctic. Although levels of PAH in the atmosphere still is lower in the Barents Sea area (i.e. Andenes and Ny-Ålesund) compared to the southern parts of Norway (i.e. Birkenes), the PAH-levels measured in 2013 were the highest measured since 2007 and are up to three orders of magnitude higher than levels of legacy POPs (e.g. PCBs) (Nizzetto et al., 2014). Higher levels and more local PAH-emission in the northern regions is expected due to increased industrial activity (e.g. petroleum industry), tourism, and shipping as the climate becomes milder.

In a study combining about 20 000 tissue concentrations of PAHs, a food web bioaccumulation model, and a time-trend analyses demonstrated that the PAH-levels in lower trophic levels have increased 10 to 30 folds the last 25 years (De Laender et al., 2011). Hence, PAHs were classified as emerging contaminants in the Arctic. In a recent monitoring study, PAH-levels were measured in blue mussels (*Mytilus edulis*) sampled at various locations in the North-Atlantic and sub-Arctic coastal environment including one location close to Svolvær (Nordland, Norway) and one location in Varangerfjorden (Finnmark, Norway) (Jorundsdottir et al., 2014). Sampling was conducted in September and November 2011. In general, the levels at the two Norwegian sites were low (Sum 16 PAHs: 58 and 39 ng/g dry weight (d.w.) of soft tissue, respectively) and lower than most of the sites at for instance Greenland, Island and Faroe Islands (28-480 ng/g d.w. soft tissue) (Jorundsdottir et al., 2014). Dominating compounds in the Norwegian samples were phenanthrene, chrysene and benzo(a)pyrene in the samples from Svolvær and phenanthrene, fluoranthene and pyrene in the samples from Varangerfjorden.

In another study, blue mussels and Icelandic scallops (*Chlamys islandica*) were sampled in a fjord in Troms (Norway) in March, June, September and December in 2010 (Nahrgang et al., 2013). Levels of PAHs in the mussels (Sum 16 PAHs: <5 to 22 µg/kg ww) and scallops (Sum 16 PAHs: 5.7 to 17 µg/kg ww) were given in wet weight and could not be compared to the levels in mussels from Svolvær and Varangerfjorden.

PAHs were also included in the screening of eggs of common eider, European shag and herring gull sampled in 2012 for instance at Røst (Nordland, Norway) (Huber et al., 2014). The PAHs included in the screening were: naphtalene, acenaphtalene, acenaphtene, fluprene, phenantrhrene, antracene, fluoranthene, pyrene, benz(a)anthracene, chrysene, benzo(b)fluoranthrene, benzo(k)fluoranthrene, benzo(a)pyrene, indeno(1,2,3-cd)pyrene, dibenz(ac/ah)anthracene, and benzo(ghi)perylene. Naftalene, antracene, fluoranten, pyrene,

chrysene was detected above the detection limit in herring gull eggs, only pyrene was detected in common eider, while no PAHs were detectable in the European shag (Table A.1). PAH-levels were highest in the herring gull eggs.

3.7 Conclusions

Emerging contaminants is an increasing environmental problem due to their abundance and potential toxic effects on biota. It is a challenge both to analyze and to evaluate the impact of emerging contaminants due to the complexity of compounds and the wide range of physico-chemical properties these compounds represent.

As this literature survey includes studies where contaminants have been measured in a wide range of species, tissues (including eggs), age groups, and genders, comparisons and evaluations to give overall conclusions on the contaminant status (and what impact it might have on the Barents Sea ecosystem) in the Barents Sea biota is difficult. Notwithstanding, a simple comparison of the levels of each of the groups of emerging contaminants this literature survey includes show that the highest levels found in the Barents Sea biota were associated with PFASs, cVMS, and CPs.

The highest levels of PFASs (all in wet weight) were found in polar bears mothers (mothers with cubs-of-the-year) from Svalbard, followed by their polar bear cubs, ivory gull eggs from Franz Josef Land, harbor seals from Lofoten and northern fulmar chicks from Kongsfjorden (Svalbard). However, the number of studies on levels of PFASs in the Barents Sea biota is low and studies on lower trophic levels are lacking. Hence, PFASs should be given more scientific attention in future Barents Sea studies.

The highest levels of siloxanes, i.e. D4, D5 and D6 (all in lipid weight), were found in species sampled near Arctic settlements, such as in sculpin from Adventfjorden near Longyearbyen at Svalbard and in Atlantic cod sampled near Tromsø. Lower levels were found in zooplankton, Atlantic cod, seabirds and bearded seals from more rural locations at Svalbard.

The highest levels of CPs, i.e. SCCPs and MCCPs (all in lipid weight), were found in Arctic charr, little auk and kittiwake from Bjørnøya, and in Greenland shark from the North-east Atlantic. Although the levels were lower, SCCPs and MCCPs were also found in polar cod, Atlantic cod, common eider, kittiwake, glaucous gulls, ringed seal, and polar bears from Svalbard.

While PFASs, siloxanes and CPs were the emerging compounds found in the highest concentrations in the examined biota from the Barents Sea region, levels of the remaining

emerging contaminants included in this study, such as novel BFRs, the CFR dechlorane plus, PFRs, CUPs, phenols and PAHs, were considerably lower, or for some compounds studied to a limited extent. The novel BFRs that dominated in biota appeared to be DBDPE, TBPH and TBP, while TBEP, TCP and EHDPP where the most common PFRs, and PCP, 4-OH-HpCS, methoxychlor and ES the most common CUPs.

A food web bioaccumulation model, and a time-trend analyses demonstrated that the PAH-levels in lower trophic levels have increased 10 to 30 folds the last 25 years. The increasing industrial activity, tourism and shipping in combination with the expected PAH time-trends in the Barents Sea region emphasize the need to keep on monitoring PAHs in the environment.

As this report is a literature survey and not a monitoring study, the findings might give insight into what compounds that have been studied in the Barents Sea biota, which compounds that are present, and to some extent also be informative with regard to levels detected in various species representing various trophic levels. The literature survey does also inform the reader about knowledge gaps with regard to emerging contaminants. However, it should be noted that the report might be somewhat limiting with regard to how well the included studies fulfill each other with regard to the species and areas examined, and how well they reflect the actual and overall situation in the Barents Sea. For instance, more knowledge is needed concerning how many of the emerging contaminants are transported to the Barents Sea and the Arctic, and how and if they bioaccumulate, and biomagnify (e.g. novel BFRs, CUPs). We would like to emphasize the importance of conducting monitoring program that is well-defined and though-through when it comes to compound and selected species (e.g. predator-prey, key-species and sensitive species), individuals (e.g. age, gender), and tissue of interest should be emphasized. A proper study design where species represent ecological relevant food webs is necessary to understand the environmental fate of contaminants, bioaccumulation and biomagnification, and to give firm conclusion on the overall impact contaminants might have on ecosystems.

4 Impact of climate change on environmental fate of contaminants

4.1 Climatic change

In the Fifth Assessment Report (AR5) from Intergovernmental Panel on Climate Change (IPCC) published in 2014, it was stated "The human influence on the climate system is clear, and recent anthropogenic emissions of greenhouse gases are the highest in history" (IPCC, 2014). The warming of the climate system is unequivocal, and in brief, the atmosphere and ocean have warmed, the amounts of snow, ice and sea ice have diminished, and sea level has risen. The average global temperature (land and surface temperature data) show a warming of 0.85 (0.65 to 1.06)°C over the period 1880 to 2012. Without additional effort in reducing the emission of greenhouse gases (GHGs), the global mean surface temperature is expected to increase between 3.7 and 4.8°C above the average for 1850-1900 for a median climate response (and exclusive climate uncertainty). It is also concluded that the Arctic regions will continue to warm more rapidly than the global mean. The sea-ice extent has decreased over the period from 1979 to 2012 with a rate at 3.5-4.1% per decade. Since 1979, the extent of the Arctic sea-ice has decreased in every season and every decade, and the most pronounced changes are seen for the decadal mean in summer sea-ice extent. It is expected that the Arctic Ocean is ice-free in the sea-ice minimum in September before mid-century.

A global warming does not only result in the sea-ice to melt, sea-levels to rise, wind systems and ocean currents to change, glaciers and permafrost to melt. Changes in these processes do also have several secondary effects, for instance influencing the fate of contaminants in the environment.

4.2 Climate change effects on contaminant fate

Several thorough reviews on these topics have been published that last years (Macdonald et al., 2005; Noyes et al., 2009; UNEP/AMAP 2011; Kallenborn et al., 2012; Moe et al., 2013; Stahl et al., 2013). Although the expected influence of climate change on processes involved in transport and distribution of contaminants in a warming environment is complex, it can be summed up to six main parameters; 1) use, sources and emission, 2) transport and distribution, 3) bioavailability, 4) environmental stability, 5) transformation and degradation, and 6) toxicity/ecotoxicology.

Several models have been made to predict the effects of climate change on **atmospheric transport** of contaminants (Kallenborn et al., 2012). Based on model studies (BETR-Global), it has been suggested that the circulation pattern referred to as the North Atlantic Oscillation (NAO), which describe wind speed/direction and precipitation rates, most likely will influence the air concentrations in the Northern Europe and the Arctic during winter and spring. That the NAO and a second circulation pattern, the Arctic Oscillations (AO), which is an index of sea-level pressure variability in the Northern Hemisphere, impact the transport of contaminants such as HCHs (e.g. α- and ɣ HCH (lindane)) in the Norwegian Arctic around Svalbard has been demonstrated (Becker et al., 2008; Becker et al., 2012). Model studies have also concluded that secondary emission of contaminants, such as PCBs becomes more and more important in the delivery of contaminants to higher latitudes relative to primary sources (Stemmler and Lammel 2012). By secondary emission one refers to a process where contaminants go through a secondary release (**revolatilization/remobilization**) from primary reservoirs where they were first deposited from environmental reservoirs where they have been deposited for decades. Ma et al. (2011) examined time series of POP air concentrations measured at the Zeppelin and Alert Stations in Svalbard and Canada, respectively, in the attempt to detect evidence of POP revolatilization in the Arctic. Their results show that sea-ice retreat and rising temperatures result in many POPs, including those with lower volatilities, to remobilize into the arctic air. Models have also been made to predict organic compounds potential to be transported to polar latitudes, and several of the parameters in the model (e.g. sea ice cover, latitudinal temperature gradients) are expected to be influenced by global warming (Meyer and Wania 2007). One actual observations in the Barents Sea region believed to be explained by the increasing temperatures, is the increasing air concentrations of HCB observed at Zeppelin, Ny-Ålesund (Svalbard) since 2004 (Hung et al., 2010). The increase in air concentrations is believed to be due to increased evaporation of HCB from the ocean in a period with ice-free winters at the western coast of Spitsbergen (2004-2010). Because the vapour pressure of chemicals increases exponentially with temperature, the flux between air and soil and between air and water (partitioning) will be affected by climate changes (UNEP/AMAP 2011). A temperature increase of 1°C will approximately cause and 10 to 15 % increase in the volatility of for instance a semi-volatile POP such as PCBs (UNEP/AMAP 2011). The temperature in the Arctic is believed to increase two times faster than the global average. Hence, revolatilization of contaminants might be a significant emission source in the future. The permanent ice-cover in the central Arctic is believed to be particularly important to hinder evaporation of volatile compounds

such as HCHs and HBB from secondary sources. Also studies on animals support the notion that climate variability may impact the transport and fate of POPs, and hence, modulate the temporal trends of POPs in the Arctic environment (Bustnes et al., 2010). Bustnes et al. show that the levels of POPs in glaucous gulls breeding in the Norwegian Arctic were relatively higher in breeding seasons following years with high air transport towards the Arctic, and that this association was stronger for HCB and other relative volatile compounds compared to the more heavy compounds such as high-chlorinated PCBs (Bustnes et al., 2010).

Extreme weather is also a parameter that might have impact on the remobilization and thus **bioavailability** of contaminants. The effects of serious flooding on remobilization of legacy POPs have been demonstrated in countries in southern Europe, where deposited pollutants stored in for instance soil were released into nearby rivers and the air (Kallenborn et al., 2012). Hence, contaminants that originally were stored in compartments where they were relatively inaccessible (e.g. soil, sediment, agricultural soil) were made bioavailable. Similar scenarios might be expected also in northern areas and has been predicted to be of significance due to the size of for instance the rivers in northern Russia. Attempts to predict climate change-induced alterations in bioaccumulation of contaminants in a Barents Sea food web show that bioavailability may be influenced by the extent of primary production (Borga et al., 2010). Models have predicted that increased temperatures coupled with increased primary production (particulate organic matter) in water might reduce bioaccumulation of contaminants such as PCB-52- and -153. However, what scenarios that will be relevant in the environment are still hard to predict, but it is suggested that it will be highly regionally specific. It has for instance been reported that the primary production has decreased in the Arctic due to limited nutrient supply caused by increased stratification of the surface oceans (Boyce et al., 2010). Also a reduction in permafrost, snowpack and glacial ice and increased erosion might increase the release of contaminants into air and aquatic systems (Kallenborn et al., 2012). There is some evidence from Antarctica that the release of DDTs has increased as glaciers are melting (Kallenborn et al., 2012). Similar scenarios are expected in the Arctic, where for instance relatively high levels of CUPs have been detected in the top layers of the Svalbard glaciers (Hermanson et al., 2005; Ruggirello et al., 2010). In Russia, melting of the permafrost, followed by an accelerated mobilization of legacy contaminants from abandoned technical waste buried in arctic communities situated in permafrost, have been suggested to explain the increasing levels of PCBs and lead in the arctic indigenous people (Chashchin 2010).

It is expect that the temperature increase and loss of sea-ice to some extent may change the **ecosystem structure** in the Barents Sea and other locations (Noyes et al., 2009; UNEP/AMAP 2011). This might alter trophic structures and biodiversity, food sources, migratory patterns, and feeding behaviour, and hence, influence the **bioaccumulation and biomagnification** of contaminants in food webs and the biological transported of contaminants between latitudes. Changes in food webs have already been reported from some Arctic areas (UNEP/AMAP 2011). It has for instance been observed that boreal zooplankton species (*Calanus finmarchicus* and *Themisto abyssorum*) are more regularly found in fjords at Svalbard (Hop et al., 2002). Further, in a study combining observation data and modelling, it was concluded that the spring bloom and peak seasonal productivity occur progressively earlier in the year in areas such as the Barents Sea (Harrison et al., 2013). The same study also state that the overall phytoplankton productivity will increase, and that cold-water phytoplankton will be displaced by warm-water species (Harrison et al., 2013). Over times, this may influence on food web-structures, and contaminant fate in the environment. It has also been predicted that the food availability for some species may be scarce as a result of increasing temperatures followed by asynchronous spatial or temporal patterns between predators and prey. For some species, this is expected to result in starvation. In for instance common eider, it is reported that eiders at high latitudes metabolize relatively more lipids and release more contaminants to the blood stream than eiders from more southern latitudes during incubation (Bustnes et al., 2012). As limited access to food might have similar effects as minor food intake during incubation, concern exist with regard to what effects warming of the Arctic might have on common eider populations at Svalbard. It has also previously been shown that food availability has a direct influence on condition, contaminant levels and reproductive fitness in marine birds, such as the great black-backed gulls (*Larus marinus*) (Bustnes et al., 2008). Also polar bears in the Barents Sea are expected to undergo periods of starvation and increased contaminant release from fat reservoirs as already observed in the population in Hudson Bay where climate effects is said to be one step ahead of the European Arctic (UNEP/AMAP 2011). A worst-case scenario might be that starvation and loss of habitat affect the populations' reproductive success and that the few polar bears that are able to reproduce have elevated levels of contaminants in their milk which further might harm their cubs.

Recently, Armitage and Wania (2013) published a paper also emphasizing that the amount of particulate organic carbon (POC) (representing primary production and terrestrial

input) in the marine environment can be important for the actual fate of hydrophobic chemicals in the environment. The model outputs between two scenarios (i.e. GCC and GCC + POC) illustrated that enhanced POC in the Arctic marine environment during the primary emission phase can exert a mitigating influence on exposure to hydrophobic chemicals. Authors concluded that the potential changes in POC cycling and sediment-water exchange dynamics linked to GCC therefore were key processes to consider further.

Increasing temperatures will probably result in increased **degradation** of contaminants in for instance soil and sea-water. However, it is not clear whether the capacity of microorganisms will increase or whether they will experience thermal stress under a warming climate. An increased degradation will increase the production of metabolites with an unknown toxic potency. In air, contaminants are expected to be degraded faster due to increased photolytic irradiation and higher concentration of radicals. Effects of climate change on contaminant degradation and transformation is highly uncertain (UNEP/AMAP 2011).

It has been concluded that climate change parameters will impact the **toxicology and toxic effects** in biota (Noyes et al., 2009). This might occur as a result of an enhanced toxicity with increasing temperatures (unknown mechanisms, but probably related to changes in metabolism), some species temperature-tolerance may be impaired with co-exposure to toxicants, and changes in salinity may influence the chemical itself or modulate toxicity and physiological functioning of species. Further, it is also recognized that a warmer climate will affect the toxicokinetics of contaminants within poikilothermic organisms, such as for instance invertebrates and fish, by directly enhancing uptake in gills and the intestine. An increase in metabolic rates is not believed to be of importance, although some fish species have been shown to transform PCBs to the more toxic OH-PCBs (UNEP/AMAP 2011). It has also been predicted that contaminants might impair the ability of animals to respond to environmental changes or stress (e.g. through endocrine disrupting chemicals) (Jenssen 2006). Such effects are regarded as a serious challenge to animals for instance adapted to arctic environmental conditions. The immune system has also shown to be sensitive to contaminant exposure (Bustnes et al., 2004; Lie et al., 2004; Lie et al., 2005), and in environments where new and more pathogens are expected, the situation might represent a serious future challenge.

A warmer climate in the Arctic may also make it economically feasible to explore natural resources such as minerals and petroleum resources in the Arctic. A reduced ice cap

may also provide space for new settlements, industrial activities and agriculture. Increased human activity can be seen at as a new contaminant source, and in particular, the presence of chemicals today associated with more urban areas (e.g. siloxanes) and industrial activity (e.g. metals and PAHs) might increase in the Arctic ecosystems. Another example of changes in use of chemicals is the expected increase in needs for pesticides for disease vector control that might be relevant in other amounts and other areas the earlier (UNEP/AMAP 2011).

Due to the increased atmospheric concentration of carbon dioxide (CO_2), the average pH of the surface waters of the global oceans has decreased from 8.2 to 8.1 since the onset of the industrial revolution. The current ocean acidification is extremely rapid and a lower pH may have consequences for marine life. Of particular concern are organisms that are shell-forming, have exoskeletons and claws, and corals which all rely on calcified structures, as well as deep-sea organisms (UNEP/AMAP 2011). Besides existing knowledge that pH affect metal toxicity in aquatic organisms, there are few reports on how pH actually affects organic contaminants. Actually, studying the influence of complex processes, such as ocean acidification, can only in part be done through observations. Hence, numerical models have become available. Attempts have been made to model the effects of rising atmospheric CO_2 levels and climate change on the ocean's acid-base status in the Barents Sea (Skogen et al., 2014), but results from this single study are too uncertain to be used to predict future situations.

The rationale for employing chiral POPs (e.g.α-HCH, *trans*-chlordane) in the investigation of climate-mediated air-surface exchange and degradation processes have been investigated by Bidleman et al. (2013) Enantiomers of chiral POPs have identical molecular weights, vapour pressures, water solubility and partition coefficient between air, water and octanol. Most chiral compounds are produced as racemates (equal proportions of enantiomers), and nonracemic residues in the environment indicate enantioselective microbial degradation in soil, water or processes in higher organisms (e.g. absoption, metabolism, membrane transport). By separating the enantiomers by chromatography and mass spectrometer detection one can distinguish emission of chemicals from two source types: racemic (newly released or not subject to microbial attack) and nonracemic (enantioselectively degraded by microbial action in soil and water). Results from air monitoring showed seasonal variation that could be under climate control indicating that enantiospecific analysis of POPs can be useful in monitoring programs.

4.3 Conclusions

The ways climate change may influence on contaminant fate in the environment are many and complex. Based on results from recent studies ranging from simple observations to complex modelling, it can be concluded that most of the following processes are affected by climate change: 1) use, sources and emission, 2) transport and distribution, 3) bioavailability, 4) environmental stability, 5) transformation and degradation, and 6) toxicity/ecotoxicology. One example where changes in POP levels in the biota of the Barents Sea region appear to be a result of climate change is the study done on glaucous gulls where the gulls were found to have relatively higher concentrations of the more volatile POPs in the breeding seasons following years with high air transport towards the Arctic. However, as the number of studies from the Barents Sea is relatively low, there are still many knowledge gaps concerning the exact ways climate change may influence on contaminant fate in this region.

In addition to study the way climate change affect contaminant fate, there are also studies demonstrating that the way some contaminants distribute (chiral compounds) may be used as indicators on climate change effects.

It should also be emphasized that climate change in many ways may modulate both the levels and patterns of contaminants animals are exposed to, and hence, the complex mixtures and toxic effects might be different in the future compared to what we see today.

Due to the increased use and complexity of emerging contaminants, and the high levels of legacy POPs deposited in the Arctic environment, and the multiples ways increasing temperatures might have on how contaminants behave in the environment, monitoring the pollution status in the Barents Sea regions should be maintained.

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Appendices

Table A.1: Overview of levels of emerging contaminants in the Barents Sea biota. Dependent on what data that was available, the concentrations are here either presented as average concentrations±standard deviations, or simply the average values. In a few cases where study groups were small, levels are presented as individual values.

Species	Location and sampling year	Sex	Age	Tissue	Concentration [3]		Reference
PFASs							
PFCAs							
Black-legged kittiwake	Kongsfjorden, Svalbard (2006)	-	chicks	Plasma	6.4±2.0	ng/g ww	Nost et al., 2012
Northern fulmar	Kongsfjorden, Svalbard (2006)	-	chicks	Plasma	20.9±9.0	ng/g ww	Nost et al., 2012
Harbour seal, adults	Svalbard (2009/2010)	F	7	Plasma	20 (15-24)	ng/g ww	Routti et al., 2014
Harbour seal, adults	Svalbard (2009/2010)	M	7	Plasma	24 (22-26)	ng/g ww	Routti et al., 2014
Harbour seal, juveniles	Svalbard (2009/2010)	F+M	1	Plasma	22 (16-26)	ng/g ww	Routti et al., 2014
Polar bear (mothers)	Svalbard (2008)	F	13±2	Plasma	92.1±10.5	ng/g ww	Bytingsvik et al., 2012
Polar bear (cubs)	Svalbard (2008)	F+M	<1	Plasma	23.0±2.1	ng/g ww	Bytingsvik et al., 2012
PFSAs							
Black-legged kittiwake	Kongsfjorden, Svalbard (2006)	-	chicks	Plasma	12.7±3.6	ng/g ww	Nost et al., 2012
Northern fulmar	Kongsfjorden, Svalbard (2006)	-	chicks	Plasma	72.7±29.3	ng/g ww	Nost et al., 2012
Harbour seal. adults	Svalbard (2009/2010)	F	7	Plasma	37 (26-40)	ng/g ww	Routti et al., 2014
Harbour seal. adults	Svalbard (2009/2010)	M	7	Plasma	45 (42-55)	ng/g ww	Routti et al., 2014
Harbour seal. juveniles	Svalbard (2009/2010)	F+M	1	Plasma	41 (30-48)	ng/g ww	Routti et al., 2014
Polar bear (mothers)	Svalbard (2008)	F	13±2	Plasma	341.6±40.7	ng/g ww	Bytingsvik et al., 2012
Polar bear (cubs)	Svalbard (2008)	F+M	<1	Plasma	77.5±8.0	ng/g ww	Bytingsvik et al., 2012

PFCAs+PFSAs							
Ivory gull	Svenskøya, Svalbard (2007)	-	-	Eggs	116.0	ng/g ww	Miljeteig et al., 2009
Ivory gull	Nagurskoe, Franz Josef Land (2006)	-	-	Eggs	91.3	ng/g ww	Miljeteig et al., 2009
Ivory gull	Cape Klyuv, Franz Josef Land (2006)	-	-	Eggs	96.9	ng/g ww	Miljeteig et al., 2009
Atlantic cod	Lofoten, Nordland (2012)	F+M	-	Liver	1.89	ng/g ww	Herzke et al., 2013
Common eider	Grindøya, Troms (2012)	-	-	Egg	18.3	ng/g ww	Herzke et al., 2013
Herring gull	Sørøya, Finnmark (2012)	-	-	Egg	72.1	ng/g ww	Herzke et al., 2013
Harbour seal	Lofoten, Nordland (2012)	F+M	-	Liver	85.5	ng/g ww	Herzke et al., 2013
Polar bear	Svalbard (2012)	M	4-25	Plasma	298	ng/ml ww	Herzke et al., 2013
Ringed seal	Kongsfjorden, Svalbard (2010)	F+M	-	Plasma	59.3	ng/ml ww	Herzke et al., 2013
Glaucous gull	Kongsfjorden, Svalbard (2012)	-	-	Plasma	45.5	ng/ml ww	Herzke et al., 2013
Kittiwake	Kongsfjorden, Svalbard (2012)	-	-	Egg	72.8	ng/g ww	Herzke et al., 2013
Common eider	Kongsfjorden, Svalbard (2012)	-	-	Egg	11.2	ng/g ww	Herzke et al., 2013
Atlantic cod	Svalbard (2012)	-	-	Liver	1.13	ng/g ww	Herzke et al., 2013
Polar cod	Svalbard (2012)	-	-	Whole fish	n.d.	ng/g ww	Herzke et al., 2013
Common eider	Røst, Nordland, Norway (2012)	-	-	Egg	23.6-46.4	ng/g ww	Huber et al., 2014
European shag	Røst, Nordland, Norway (2012)	-	-	Egg	19.5-32.8	ng/g ww	Huber et al., 2014
Herring gull	Røst, Nordland, Norway (2012)	-	-	Egg	23.6-41.5	ng/g ww	Huber et al., 2014
Novel BFRs							
BTBPE							
Greenland shark	North-East Atlantic (2001-2003) [1]	F	-	Liver	0.61	ng/g lw	Strid et al., 2013
DBDPE							
Atlantic cod	Lofoten, Nordland (2012)	F+M	-	Liver	4.29±0.70	ng/g ww	Herzke et al., 2013
Common eider	Grindøya, Troms (2012)	-	-	Egg	0.33±0.11	ng/g ww	Herzke et al., 2013

Herring gull	Sørøya, Finnmark (2012)	-	-	Egg	0.44±0.20	ng/g ww	Herzke et al., 2013
Harbour seal	Lofoten, Nordland (2012)	F+M	-	Liver	12.9±6.8	ng/g ww	Herzke et al., 2013
Polar bear	Svalbard (2012)	M	4-25	Plasma	6.98±9.11	ng/ml ww	Herzke et al., 2013
Ringed seal	Kongsfjorden, Svalbard (2010)	F+M	-	Plasma	5.36±1.94	ng/ml ww	Herzke et al., 2013
Glaucous gull	Kongsfjorden, Svalbard (2012)	-	-	Plasma	6.43±2.62	ng/ml ww	Herzke et al., 2013
Kittiwake	Kongsfjorden, Svalbard (2012)	-	-	Egg	1.01±1.55	ng/g ww	Herzke et al., 2013
Common eider	Kongsfjorden, Svalbard (2012)	-	-	Egg	0.82±0.62	ng/g ww	Herzke et al., 2013
Atlantic cod	Svalbard (2012)	-	-	Liver	5.57	ng/g ww	Herzke et al., 2013
Polar cod	Svalbard (2012)	-	-	Whole fish	0.42	ng/g ww	Herzke et al., 2013
TBPH (BEHTBP)							
Atlantic cod	Lofoten, Nordland (2012)	F+M	-	Liver	0.14±0.02	ng/g ww	Herzke et al., 2013
Common eider	Grindøya, Troms (2012)	-	-	Egg	0.04±0.02	ng/g ww	Herzke et al., 2013
Herring gull	Sørøya, Finnmark (2012)	-	-	Egg	1.99±2.65	ng/g ww	Herzke et al., 2013
Harbour seal	Lofoten, Nordland (2012)	F+M	-	Liver	0.10	ng/g ww	Herzke et al., 2013
Polar bear	Svalbard (2012)	M	4-25	Plasma	0.15±0.16	ng/ml ww	Herzke et al., 2013
Ringed seal	Kongsfjorden, Svalbard (2010)	F+M	-	Plasma	0.04	ng/ml ww	Herzke et al., 2013
Glaucous gull	Kongsfjorden, Svalbard (2012)	-	-	Plasma	0.026±0.001	ng/ml ww	Herzke et al., 2013
Kittiwake	Kongsfjorden, Svalbard (2012)	-	-	Egg	0.10±0.09	ng/g ww	Herzke et al., 2013
Common eider	Kongsfjorden, Svalbard (2012)	-	-	Egg	0.06±0.07	ng/g ww	Herzke et al., 2013
Atlantic cod	Svalbard (2012)	-	-	Liver	0.07	ng/g ww	Herzke et al., 2013
Polar cod	Svalbard (2012)	-	-	Whole fish	n.d.	ng/g ww	Herzke et al., 2013
PBEB							
Greenland shark	North-East Atlantic (2001-2003) [1]	F	-	Liver	3.0	ng/g lw	Strid et al., 2013

TBX							
Greenland shark	North-East Atlantic (2001-2003) [1]	F	-	Liver	n.d.	ng/g lw	Strid et al., 2013
PBP							
Atlantic cod	Lofoten, Nordland (2012)	F+M	-	Liver	n.d.	ng/g ww	Herzke et al., 2013
Common eider	Grindøya, Troms (2012)	-	-	Egg	n.d.	ng/g ww	Herzke et al., 2013
Herring gull	Sørøya, Finnmark (2012)	-	-	Egg	n.d.	ng/g ww	Herzke et al., 2013
Harbour seal	Lofoten, Nordland (2012)	F+M	-	Liver	n.d.	ng/g ww	Herzke et al., 2013
Polar bear	Svalbard (2012)	M	4-25	Plasma	n.d.	ng/ml ww	Herzke et al., 2013
Ringed seal	Kongsfjorden, Svalbard (2010)	F+M	-	Plasma	n.d.	ng/ml ww	Herzke et al., 2013
Glaucous gull	Kongsfjorden, Svalbard (2012)	-	-	Plasma	n.d.	ng/ml ww	Herzke et al., 2013
Kittiwake	Kongsfjorden, Svalbard (2012)	-	-	Egg	n.d.	ng/g ww	Herzke et al., 2013
Common eider	Kongsfjorden, Svalbard (2012)	-	-	Egg	n.d.	ng/g ww	Herzke et al., 2013
Atlantic cod	Svalbard (2012)	-	-	Liver	n.d.	ng/g ww	Herzke et al., 2013
Polar cod	Svalbard (2012)	-	-	Whole fish	n.d.	ng/g ww	Herzke et al., 2013
TBP (2.4.6-TBP)							
Atlantic cod	Lofoten, Nordland (2012)	F+M	-	Liver	68.8±35.8	ng/g ww	Herzke et al., 2013
Common eider	Grindøya, Troms (2012)	-	-	Egg	66.2±74.2	ng/g ww	Herzke et al., 2013
Herring gull	Sørøya, Finnmark (2012)	-	-	Egg	62.5±64.8	ng/g ww	Herzke et al., 2013
Harbour seal	Lofoten, Nordland (2012)	F+M	-	Liver	164.0±84	ng/g ww	Herzke et al., 2013
Polar bear	Svalbard (2012)	M	4-25	Plasma	25.7±14.7	ng/ml ww	Herzke et al., 2013
Ringed seal	Kongsfjorden, Svalbard (2010)	F+M	-	Plasma	31.2±32.3	ng/ml ww	Herzke et al., 2013
Glaucous gull	Kongsfjorden, Svalbard (2012)	-	-	Plasma	30.8±9.0	ng/ml ww	Herzke et al., 2013
Kittiwake	Kongsfjorden, Svalbard (2012)	-	-	Egg	52.6±18.8	ng/g ww	Herzke et al., 2013
Common eider	Kongsfjorden, Svalbard (2012)	-	-	Egg	37.8±17.6	ng/g ww	Herzke et al., 2013
Atlantic cod	Svalbard (2012)	-	-	Liver	115.0±61	ng/g ww	Herzke et al., 2013

Polar cod	Svalbard (2012)	-	-	Whole fish	n.d.	ng/g ww	Herzke et al., 2013
Sum Novel BFRs [4]							
Common eider	Røst, Nordland, Norway (2012)	-	-	Egg	0.83-2.27	ng/g ww	Huber et al., 2014
European shag	Røst, Nordland, Norway (2012)	-	-	Egg	1.58-2.65	ng/g ww	Huber et al., 2014
Herring gull	Røst, Nordland, Norway (2012)	-	-	Egg	10.02-14.25	ng/g ww	Huber et al., 2014
PFRs							
Common eider	Røst, Nordland, Norway (2012)	-	-	Egg	16.8-81.3	ng/g ww	Huber et al., 2014
European shag	Røst, Nordland, Norway (2012)	-	-	Egg	64.5-95.6	ng/g ww	Huber et al., 2014
Herring gull	Røst, Nordland, Norway (2012)	-	-	Egg	31.9-47.8	ng/g ww	Huber et al., 2014
CPs							
SCCPs							
Atlantic cod	Northern Norway (2004)	F	-	Liver	17 and 52	ng/g ww	Reth et al., 2006
Atlantic cod	Northern Norway (2004)	F	-	Liver	35 and 140	ng/g lw	Reth et al., 2006
Arctic charr	Bjørnøya, Svalbard (2001)	F	-	Liver	11 and 27	ng/g ww	Reth et al., 2006
Arctic charr	Bjørnøya, Svalbard (2001)	F	-	Liver	89 and 230	ng/g lw	Reth et al., 2006
Arctic charr	Bjørnøya, Svalbard (2001)	F	-	Muscle	7 and 13	ng/g ww	Reth et al., 2006
Arctic charr	Bjørnøya, Svalbard (2001)	F	-	Muscle	300 and 540	ng/g lw	Reth et al., 2006
Little auk	Bjørnøya, Svalbard (2001)	M	-	Liver	18 and 88	ng/g ww	Reth et al., 2006
Little auk	Bjørnøya, Svalbard (2001)	M	-	Liver	190 and 880	ng/g lw	Reth et al., 2006
Little auk	Bjørnøya, Svalbard (2001)	M	-	Muscle	7 and 16	ng/g ww	Reth et al., 2006
Little auk	Bjørnøya, Svalbard (2001)	M	-	Muscle	150 and 430	ng/g lw	Reth et al., 2006
Kittiwake	Bjørnøya, Svalbard (2001)	F+M	-	Liver	6 and 44	ng/g ww	Reth et al., 2006
Kittiwake	Bjørnøya, Svalbard (2001)	F+M	-	Liver	110 and 860	ng/g lw	Reth et al., 2006
Kittiwake	Bjørnøya, Svalbard (2001)	F+M	-	Muscle	5 and 5	ng/g ww	Reth et al., 2006

Kittiwake	Bjørnøya, Svalbard (2001)	F+M	-	Muscle	41 and 95	ng/g lw	Reth et al., 2006
Greenland shark	North-East Atlantic (2001-2003) [1]	F	-	Liver	430	ng/g lw	Strid et al., 2013
Polar bear	Svalbard (2012)	M	4-25	Plasma	3.99±2.91	ng/ml ww	Herzke et al., 2013
Ringed seal	Kongsfjorden, Svalbard (2010)	F+M	-	Plasma	4.96±2.70	ng/ml ww	Herzke et al., 2013
Glaucous gull	Kongsfjorden, Svalbard (2012)	-	-	Plasma	3.95±1.99	ng/ml ww	Herzke et al., 2013
Kittiwake	Kongsfjorden, Svalbard (2012)	-	-	Egg	7.83±8.26	ng/g ww	Herzke et al., 2013
Common eider	Kongsfjorden, Svalbard (2012)	-	-	Egg	3.23±1.77	ng/g ww	Herzke et al., 2013
Atlantic cod	Svalbard (2012)	-	-	Liver	10.3±10.7	ng/g ww	Herzke et al., 2013
Polar cod	Svalbard (2012)	-	-	Whole fish	2.28	ng/g ww	Herzke et al., 2013
MCCPs							
Atlantic cod	Northern Norway (2004)	F	-	Liver	7 and 47	ng/g ww	Reth et al., 2006
Atlantic cod	Northern Norway (2004)	F	-	Liver	14 and 130	ng/g lw	Reth et al., 2006
Arctic charr	Bjørnøya, Svalbard (2001)	F	-	Liver	13 and 43	ng/g ww	Reth et al., 2006
Arctic charr	Bjørnøya, Svalbard (2001)	F	-	Liver	110 and 360	ng/g lw	Reth et al., 2006
Arctic charr	Bjørnøya, Svalbard (2001)	F	-	Muscle	10 and 47	ng/g ww	Reth et al., 2006
Arctic charr	Bjørnøya, Svalbard (2001)	F	-	Muscle	440 and 1 600	ng/g lw	Reth et al., 2006
Little auk	Bjørnøya, Svalbard (2001)	M	-	Liver	48 and 370	ng/g ww	Reth et al., 2006
Little auk	Bjørnøya, Svalbard (2001)	M	-	Liver	500 and 3 700	ng/g lw	Reth et al., 2006
Little auk	Bjørnøya, Svalbard (2001)	M	-	Muscle	17 and 55	ng/g ww	Reth et al., 2006
Little auk	Bjørnøya, Svalbard (2001)	M	-	Muscle	450 and 1 200	ng/g lw	Reth et al., 2006
Kittiwake	Bjørnøya, Svalbard (2001)	F+M	-	Liver	12 and 39	ng/g ww	Reth et al., 2006
Kittiwake	Bjørnøya, Svalbard (2001)	F+M	-	Liver	240 and 730	ng/g lw	Reth et al., 2006
Kittiwake	Bjørnøya, Svalbard (2001)	F+M	-	Muscle	5 and 38	ng/g ww	Reth et al., 2006
Kittiwake	Bjørnøya, Svalbard (2001)	F+M	-	Muscle	41 and 720	ng/g lw	Reth et al., 2006
Polar bear	Svalbard (2012)	M	4-25	Plasma	2.20±1.84	ng/ml ww	Herzke et al., 2013
Ringed seal	Kongsfjorden, Svalbard (2010)	F+M	-	Plasma	2.91±2.39	ng/ml ww	Herzke et al., 2013

Glaucous gull	Kongsfjorden, Svalbard (2012)	-	-	Plasma	8.87±9.8	ng/ml ww	Herzke et al., 2013
Kittiwake	Kongsfjorden, Svalbard (2012)	-	-	Egg	4.91±4.88	ng/g ww	Herzke et al., 2013
Common eider	Kongsfjorden, Svalbard (2012)	-	-	Egg	4.24±4.07	ng/g ww	Herzke et al., 2013
Atlantic cod	Svalbard (2012)	-	-	Liver	0.94	ng/g ww	Herzke et al., 2013
Polar cod	Svalbard (2012)	-	-	Whole fish	1.51	ng/g ww	Herzke et al., 2013
SCCP+MCCP							
Common eider	Røst, Nordland, Norway (2012)	-	-	Egg	0.79-22.3	ng/g ww	Huber et al., 2014
European shag	Røst, Nordland, Norway (2012)	-	-	Egg	0.8-7.12	ng/g ww	Huber et al., 2014
Herring gull	Røst, Nordland, Norway (2012)	-	-	Egg	n.d.-3.85	ng/g ww	Huber et al., 2014
CUPs							
Chlorpyrifos							
Glaucous gull	Svalbard (2011)	F+M	-	Serum	<d.l.	ng/g ww	Langford et al., 2012
Kittiwake	Svalbard (2007)	-	-	Plasma	<d.l.	ng/g ww	Langford et al., 2012
Bearded seal	Svalbard (2011)	F+M	-	Blubber	<d.l.	ng/g ww	Langford et al., 2012
Ringed seal	Svalbard (2011)	M	Adult	Blubber	1.4	ng/g ww	Langford et al., 2012
Eider duck	Svalbard (2010/2011)	-	-	Egg	<d.l.	ng/g ww	Langford et al., 2012
Common guillemot	Svalbard (2010/2011)	-	-	Egg	<d.l.	ng/g ww	Langford et al., 2012
PCP							
Harbour seal, adults	Svalbard (2009/2010)	F	7	Plasma	0.28-0.51	ng/g ww	Routti et al., 2014
Harbour seal, adults	Svalbard (2009/2010)	M	7	Plasma	0.25-0.42	ng/g ww	Routti et al., 2014
Harbour seal, juveniles	Svalbard (2009/2010)	F+M	1	Plasma	0.22-0.67	ng/g ww	Routti et al., 2014
4-OH-HpCS							
Harbour seal, adults	Svalbard (2009/2010)	F	7	Plasma	0.04-0.13	ng/g ww	Routti et al., 2014

Harbour seal, adults	Svalbard (2009/2010)	M	7	Plasma	0.07-0.09	ng/g ww	Routti et al., 2014
Harbour seal, juveniles	Svalbard (2009/2010)	F+M	1	Plasma	0.07-0.15	ng/g ww	Routti et al., 2014
Trifluralin							
Common eider	Røst, Nordland, Norway (2012)	-	-	Egg	n.d.	ng/g ww	Huber et al., 2014
European shag	Røst, Nordland, Norway (2012)	-	-	Egg	n.d.	ng/g ww	Huber et al., 2014
Herring gull	Røst, Nordland, Norway (2012)	-	-	Egg	n.d.	ng/g ww	Huber et al., 2014
Methoxychlor							
Ringed seal	White sea (2001-2005)	F	6±3	Blubber	<d.l.-4.2	ng/g lw	Savinov et al., 2011
Ringed seal	White sea (2001-2005)	M	7±3	Blubber	<d.l.-8.4	ng/g lw	Savinov et al., 2011
Ringed seal	Barents sea (2001-2005)	F	4±1	Blubber	<d.l.	ng/g lw	Savinov et al., 2011
Ringed seal	Barents sea (2001-2005)	M	4±2	Blubber	<d.l.	ng/g lw	Savinov et al., 2011
Common eider	Røst, Nordland, Norway (2012)	-	-	Egg	n.d.-0.37	ng/g ww	Huber et al., 2014
European shag	Røst, Nordland, Norway (2012)	-	-	Egg	n.d.	ng/g ww	Huber et al., 2014
Herring gull	Røst, Nordland, Norway (2012)	-	-	Egg	n.d.	ng/g ww	Huber et al., 2014
α-Endosulfan							
Ringed seal	White sea (2001-2005)	F	6±3	Blubber	<d.l.-0.5	ng/g lw	Savinov et al., 2011
Ringed seal	White sea (2001-2005)	M	7±3	Blubber	<d.l.	ng/g lw	Savinov et al., 2011
Ringed seal	Barents sea (2001-2005)	F	4±1	Blubber	<d.l.	ng/g lw	Savinov et al., 2011
Ringed seal	Barents sea (2001-2005)	M	4±2	Blubber	<d.l.-0.5	ng/g lw	Savinov et al., 2011
β-Endosulfan							
Ringed seal	White sea (2001-2005)	F	6±3	Blubber	<d.l.-3.1	ng/g lw	Savinov et al., 2011
Ringed seal	White sea (2001-2005)	M	7±3	Blubber	<d.l.	ng/g lw	Savinov et al., 2011
Ringed seal	Barents sea (2001-2005)	F	4±1	Blubber	<d.l.	ng/g lw	Savinov et al., 2011

Ringed seal	Barents sea (2001-2005)	M	4±2	Blubber	<d.l.-1.8	ng/g lw	Savinov et al., 2011
Dacthal, PCP, Methoxychlor, and Trifluralin							
Glaucous gull	Svalbard (2011)	F+M	-	Serum	<d.l.	ng/g ww	Langford et al., 2012
Kittiwake	Svalbard (2007)	-	-	Plasma	<d.l.	ng/g ww	Langford et al., 2012
Bearded seal	Svalbard (2011)	F+M	-	Blubber	<d.l.	ng/g ww	Langford et al., 2012
Ringed seal	Svalbard (2011)	M	-	Blubber	<d.l.	ng/g ww	Langford et al., 2012
Eider duck	Svalbard (2010/2011)	-	-	Egg	<d.l.	ng/g ww	Langford et al., 2012
Common guillemot	Svalbard (2010/2011)	-	-	Egg	<d.l.	ng/g ww	Langford et al., 2012
cVMS (siloxanes)							
D4							
Zooplankton	Kongsfjorden, Svalbard (2009)	-	-	Whole org.	n.d.	ng/g lw	Warner et al., 2010
Zooplankton	Liefdefjorden, Svalbard (2009)	-	-	Whole org.	n.d.	ng/g lw	Warner et al., 2010
Atlantic cod	Adventfjorden, Svalbard (2009)	-	-	Liver	n.d.	ng/g lw	Warner et al., 2010
Sculpin	Adventfjorden, Svalbard (2009)	-	-	Liver	n.d.	ng/g lw	Warner et al., 2010
Atlantic cod	Kongsfjorden, Svalbard (2009)	-	-	Liver	n.d.	ng/g lw	Warner et al., 2010
Sculpin	Liefdefjorden, Svalbard (2009)	-	-	Liver	n.d.	ng/g lw	Warner et al., 2010
Bearded seal [2]	Kongsfjorden, Svalbard (2009)	-	-	Blubber	n.d.	ng/g lw	Warner et al., 2010
Atlantic cod	Tromsø, Norway (2011/2012)	-	4-6	Liver	15.7-111	ng/g lw	Warner et al., 2014
Atlantic cod	Nipøya, Norway (2011/2012)	-	3-7	Liver	5.6-15.0	ng/g lw	Warner et al., 2014
Atlantic cod	Kongsfjorden, Svalbard (2008)	-	-	Liver	2.9-3.9	ng/g ww	Evenset et al., 2009
Polar cod	Fjords at Svalbard (2008)	-	-	Whole fish	n.d.-9.2	ng/g ww	Evenset et al., 2009
Kittiwake	Fjords at Svalbard (2008)	-	-	Liver	n.d.-3.5	ng/g ww	Evenset et al., 2009
Common eider	Kongsfjorden, Svalbard (2008)	-	-	Liver	n.d.	ng/g ww	Evenset et al., 2009

D5							
Zooplankton	Kongsfjorden, Svalbard (2009)	-	-	Whole org.	n.d.	ng/g lw	Warner et al., 2010
Zooplankton	Liefdefjorden, Svalbard (2009)	-	-	Whole org.	n.d.	ng/g lw	Warner et al., 2010
Atlantic cod	Adventfjorden, Svalbard (2009)	-	-	Liver	45.5-358	ng/g lw	Warner et al., 2010
Sculpin	Adventfjorden, Svalbard (2009)	-	-	Liver	54.3-2 150	ng/g lw	Warner et al., 2010
Atlantic cod	Kongsfjorden, Svalbard (2009)	-	-	Liver	12.7-29.1	ng/g lw	Warner et al., 2010
Sculpin	Liefdefjorden, Svalbard (2009)	-	-	Liver	n.d.-11.3	ng/g lw	Warner et al., 2010
Bearded seal [2]	Kongsfjorden, Svalbard (2009)	-	-	Blubber	<field blank	ng/g lw	Warner et al., 2010
Atlantic cod	Tromsø, Norway (2011/2012)	-	4-6	Liver	338-2 530	ng/g lw	Warner et al., 2014
Atlantic cod	Nipøya, Norway (2011/2012)	-	3-7	Liver	30-1 260	ng/g lw	Warner et al., 2014
Atlantic cod	Kongsfjorden, Svalbard (2008)	-	-	Liver	2.7-4.6	ng/g ww	Evenset et al., 2009
Polar cod	Fjords at Svalbard (2008)	-	-	Whole fish	n.d.-19.1	ng/g ww	Evenset et al., 2009
Kittiwake	Fjords at Svalbard (2008)	-	-	Liver	n.d.-5.2	ng/g ww	Evenset et al., 2009
Common eider	Kongsfjorden, Svalbard (2008)	-	-	Liver	n.d.	ng/g ww	Evenset et al., 2009
D6							
Zooplankton	Kongsfjorden, Svalbard (2009)	-	-	Whole org.	n.d.	ng/g lw	Warner et al., 2010
Zooplankton	Liefdefjorden, Svalbard (2009)	-	-	Whole org.	n.d.	ng/g lw	Warner et al., 2010
Atlantic cod	Adventfjorden, Svalbard (2009)	-	-	Liver	5.3-13.8	ng/g lw	Warner et al., 2010
Sculpin	Adventfjorden, Svalbard (2009)	-	-	Liver	n.d.-30.6	ng/g lw	Warner et al., 2010
Atlantic cod	Kongsfjorden, Svalbard (2009)	-	-	Liver	10.7-52.8	ng/g lw	Warner et al., 2010
Sculpin	Liefdefjorden, Svalbard (2009)	-	-	Liver	n.d.-9.6	ng/g lw	Warner et al., 2010
Bearded seal	Kongsfjorden, Svalbard (2009)	-	-	Blubber	n.d-1.1	ng/g lw	Warner et al., 2010
Atlantic cod	Tromsø, Norway (2011/2012)	-	4-6	Liver	28.8-139	ng/g lw	Warner et al., 2014
Atlantic cod	Nipøya, Norway (2011/2012)	-	3-7	Liver	4.6-146	ng/g lw	Warner et al., 2014
Atlantic cod	Kongsfjorden, Svalbard (2008)	-	-	Liver	n.d.	ng/g ww	Evenset et al., 2009
Polar cod	Fjords at Svalbard (2008)	-	-	Whole fish	n.d.-10.7	ng/g ww	Evenset et al., 2009

Kittiwake	Fjords at Svalbard (2008)	-	-	Liver	n.d.	ng/g ww	Evenset et al., 2009
Common eider	Kongsfjorden, Svalbard (2008)	-	-	Liver	n.d.	ng/g ww	Evenset et al., 2009
D4+D5+D6							
Common eider	Røst, Nordland, Norway (2012)	-	-	Egg	n.d.	ng/g ww	Huber et al., 2014
European shag	Røst, Nordland, Norway (2012)	-	-	Egg	n.d.-1.4	ng/g ww	Huber et al., 2014
Herring gull	Røst, Nordland, Norway (2012)	-	-	Egg	n.d.-1.5	ng/g ww	Huber et al., 2014
Phenols							
DBP							
Common eider	Røst, Nordland, Norway (2012)	-	-	Egg	n.d.-2.9	ng/g ww	Huber et al., 2014
European shag	Røst, Nordland, Norway (2012)	-	-	Egg	n.d.	ng/g ww	Huber et al., 2014
Herring gull	Røst, Nordland, Norway (2012)	-	-	Egg	n.d.-0.2	ng/g ww	Huber et al., 2014
TBP							
Common eider	Røst, Nordland, Norway (2012)	-	-	Egg	0.29-0.67	ng/g ww	Huber et al., 2014
European shag	Røst, Nordland, Norway (2012)	-	-	Egg	0.14-0.72	ng/g ww	Huber et al., 2014
Herring gull	Røst, Nordland, Norway (2012)	-	-	Egg	0.26-0.34	ng/g ww	Huber et al., 2014
PBP, TBBPA, and BPA							
Common eider	Røst, Nordland, Norway (2012)	-	-	Egg	n.d.	ng/g ww	Huber et al., 2014
European shag	Røst, Nordland, Norway (2012)	-	-	Egg	n.d.	ng/g ww	Huber et al., 2014
Herring gull	Røst, Nordland, Norway (2012)	-	-	Egg	n.d.	ng/g ww	Huber et al., 2014
4-iso-nonylphenol							
Common eider	Røst, Nordland, Norway (2012)	-	-	Egg	n.d.-49	ng/g ww	Huber et al., 2014
European shag	Røst, Nordland, Norway (2012)	-	-	Egg	n.d.-5.9	ng/g ww	Huber et al., 2014

Herring gull	Røst, Nordland, Norway (2012)	-	-	Egg	n.d.-16	ng/g ww	Huber et al., 2014
Other alkylphenols [5]							
Common eider	Røst, Nordland, Norway (2012)	-	-	Egg	n.d.	ng/g ww	Huber et al., 2014
European shag	Røst, Nordland, Norway (2012)	-	-	Egg	n.d.	ng/g ww	Huber et al., 2014
Herring gull	Røst, Nordland, Norway (2012)	-	-	Egg	n.d.	ng/g ww	Huber et al., 2014
PAHs							
SumPAHs							
Blue mussels	Svolvær, Nordland, Norway (2011)	-	-	Soft tissue	58	ng/g dw	Jorundsdottir et al., 2014
Blue mussels	Varangerfjorden, Finnmark, Norway (2011)	-	-	Soft tissue	39	ng/g dw	Jorundsdottir et al., 2014
Blue mussels	Troms, Norway (2010)	-	-	Soft tissue	n.d.-22	ng/g ww	Nahrgang et al., 2013
Icelandic scallop	Troms, Norway (2010)	-	-	Soft tissue	5.7-17	ng/g ww	Nahrgang et al., 2013
Common eider	Røst, Nordland, Norway (2012)	-	-	Egg	n.d.-0.44	ng/g ww	Huber et al., 2014
European shag	Røst, Nordland, Norway (2012)	-	-	Egg	n.d.	ng/g ww	Huber et al., 2014
Herring gull	Røst, Nordland, Norway (2012)	-	-	Egg	n.d.-29.6	ng/g ww	Huber et al., 2014

1 little is known about this species migration pattern and capacity, hence, it could be relevant for the Barents Sea. Study group consist of mature and pre-adult males.

2 Levels were below average field blank of 10.3±12.4 ng/g ww.

3 ww = wet weight and lw = lipid weight.

4 Does not include TBP.

5 4-t-pentylphenol, 4-n-pentylphenol, 4-t-octylphenol, 4-n-octylphenol, 4-n-heptylphenol, 4-t-octylphenol monoethoxylates (4-t-OPEO1), and C10-iso-phenol.