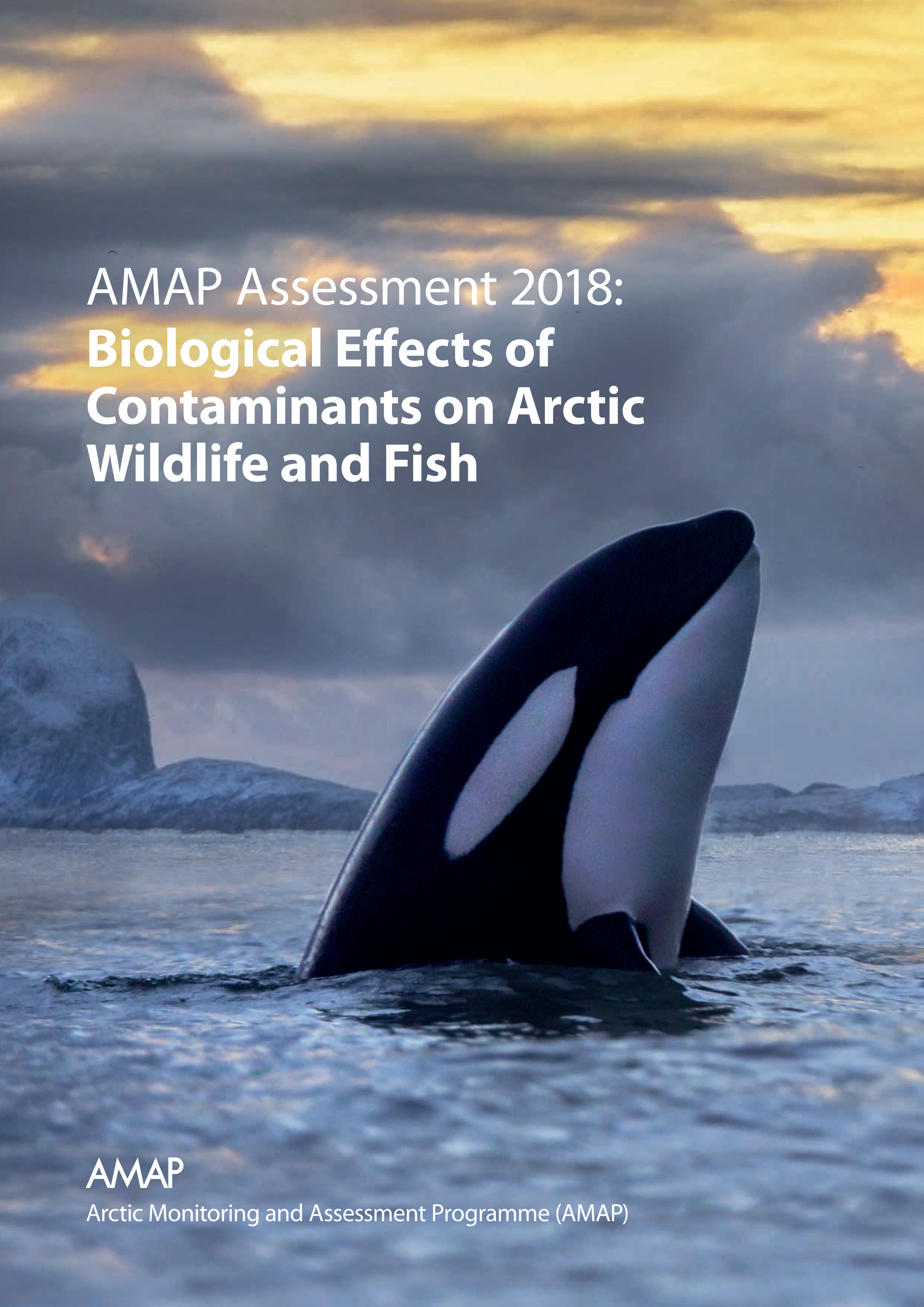

AMAP Assessment 2018:
Biological Effects of Contaminants on Arctic Wildlife and Fish

AMAP
Arctic Monitoring and Assessment Programme (AMAP)

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Contents

Preface

This assessment report presents the results of the 2018 AMAP Assessment of the Biological Effects of Contaminants on Arctic Wildlife and Fish. The assessment updates information presented in the AMAP POPs assessment delivered in 2009, specifically with respect to information on biological effects of organohalogenated compounds (OHCs), and the AMAP assessment of Mercury in the Arctic that was delivered in 2011.

The Arctic Monitoring and Assessment Programme (AMAP) is a Working Group of the Arctic Council. The Arctic Council Ministers have requested AMAP to:

- produce integrated assessment reports on the status and trends of Arctic ecosystems;
- identify possible causes for the changing conditions;
- detect emerging problems, their possible causes, and the potential risk to Arctic ecosystems including indigenous peoples and other Arctic residents;
- recommend actions required to reduce risks to Arctic ecosystems.

This report provides the accessible scientific basis and validation for any statements and recommendations made in related derivative products, including summaries for policy-makers that will be delivered to the Arctic Council Ministers at their meeting in 2019.

The present report includes extensive background data and references to the scientific literature and details the sources for graphics reproduced in summary products. Whereas summary products for policy-makers may contain recommendations that focus on policy-relevant actions concerned with addressing contaminant impacts on Arctic ecosystems, the conclusions and recommendations presented in this report also cover issues of a more scientific nature, such as proposals for filling gaps in knowledge, and recommendations relevant to future monitoring and research work.

This assessment of biological effects of organohalogenated compounds and mercury was conducted between 2016 and 2018 by an international group of experts. The expert group members and lead authors were appointed following an open nomination process coordinated by AMAP. A similar process was used to select international experts who independently reviewed this report. Information contained in this report is fully referenced and based first and foremost on results of research and monitoring undertaken since 2009. It incorporates some new (unpublished) information from monitoring and research conducted according to well established and documented national and international standards and quality assurance/quality control protocols. Care was taken to ensure that no critical probability statements are based on non peer-reviewed materials. Access to reliable and up-to-date information is essential for the development of science-based decision-making regarding ongoing changes in the Arctic and their global implications.

The assessment lead authors have confirmed that both this report and its derivative products accurately and fully reflect their scientific assessment. All AMAP assessment reports are freely available from the AMAP Secretariat and on the AMAP website: www.amap.no, and their use for educational purposes is encouraged.

AMAP would like to express its appreciation to all experts who have contributed their time, efforts and data, in particular the lead authors who coordinated the production of this report. Thanks are also due to the reviewers who contributed to the assessment peer-review process and provided valuable comments that helped to ensure the quality of the report. A list of contributors is included in the acknowledgments at the start of this report and lead authors are identified at the start of each chapter. The acknowledgments list is not comprehensive. Specifically, it does not include the many national institutes, laboratories and organizations, and their staff that have been involved in various countries in contaminants-related monitoring and research. Apologies, and no lesser thanks are given to any individuals unintentionally omitted from the list.

The support from the Arctic countries and non-Arctic countries implementing research and monitoring in the Arctic is vital to the success of AMAP. The AMAP work is essentially based on ongoing activities within these countries, and the countries that provide the necessary support for most of the experts involved in the preparation of the AMAP assessments. In particular, AMAP would like to acknowledge Canada and the Kingdom of Denmark for taking the lead country role in this assessment and to thank Canada, the Kingdom of Denmark, Norway (Ministry of Foreign Affairs) and the Nordic Council of Ministers for their financial support to this assessment work.

The AMAP Working Group is pleased to present its assessment to the Arctic Council and the international science community.

Rune Dietz (Assessment co-lead, Denmark)

Robert J. Letcher (Assessment co-lead, Canada)

Marianne Kroglund (AMAP Chair)

Rolf Rødven (AMAP Executive Secretary)

Tromsø, September 2018

1. Introduction

Authors: **Rune Dietz, Robert J. Letcher**

Within the circumpolar Arctic there has been minimal direct use of chemical substances classified as persistent organic pollutants (POPs), and those that have been used largely comprise chlorinated, brominated and/or fluorinated organohalogen compounds (OHCs). However, POPs, or their precursors, degradation products and metabolites, are carried into the Arctic from more southerly latitudes via long-range atmospheric transport as well as via ocean currents and rivers (Braune et al., 2005; de Wit et al., 2006, 2010). As a result, lipophilic contaminants, such as POPs and methylmercury (MeHg) bioaccumulate in organisms and biomagnify through the marine food web, generating concern for the health of exposed wildlife and fish as well as for those indigenous populations that consume these food items as part of a traditional diet. The present assessment concerns the entire circumpolar Arctic despite a lack of knowledge from the areas of the Russian Federation (Figure 1.1).

The harmful effects of many POPs on human and environmental health have been recognized. In 2001, under a groundbreaking United Nations treaty – the Stockholm Convention – signatory countries agreed to reduce or eliminate the production, use, and/or release of 12 key POPs; the so-called 'dirty dozen' (Kaiser and Enserink, 2000) and in 2017 the Minamata Convention on Mercury entered into force worldwide (Evers et al., 2016). During the past two decades, the concentrations of many of these legacy pollutants in marine biota have declined, although for the last ten years or so concentrations of polychlorinated biphenyls (PCBs) and chlordanes have remained relatively constant and at high levels in wildlife tissues (AMAP, 2016). However, every year thousands of new synthetic chemicals are produced, and recently there have been reports of chemicals of emerging concern (CEC) in humans and marine biota, for example, brominated flame retardants (BFRs) including polybrominated diphenyl ethers (PBDEs) (de Wit et al., 2010) among many others (AMAP, 2017a,b).

Most toxicological and biological effects studies in wildlife have been carried out on marine top predators where levels of legacy and unrestricted OHCs as well as metals such as Hg (predominantly total Hg unless mentioned otherwise), are consistently highest. Examination of correlative relationships between biological endpoint (or biomarker) measurements and contaminant levels, supports and contributes to a 'weight of evidence' for there being contaminant-related biological effects rather than being a direct interpretation as an adverse

Figure 1.1 Regions from which contaminant exposure and effect studies were available for the present assessment.

cause-and-effect. Current knowledge of CECs in the Arctic is detailed in a companion assessment (AMAP, 2017a,b). Many of the CECs that have been reported in Arctic wildlife and fish are generally present at much lower tissue levels than for legacy POPs such as PCBs, organochlorine pesticides (OCPs), and BFRs. In some Arctic species and populations, tissue contaminant levels are of a magnitude expected to exceed putative risk threshold values previously estimated for non-Arctic species (Fisk et al., 2005). In these Arctic species studies, exposure to specific OHCs and Hg have been shown to result in deleterious and observable biological effects via mode(s) of action and mechanisms that are a function of the contaminant type and exposure intensity.

Among the eight Arctic countries collaborating within AMAP (Canada, Denmark/Greenland/Faroes Islands, Finland, Iceland, Norway, Russia, Sweden, USA), most effect studies on wildlife and fish were historically available from the North Atlantic between Greenland and Svalbard (Norway) where tissue concentrations of OHCs and other POPs have been shown to be highest. This was consistently emphasized for the period 1998–2012 in the previous three rounds of AMAP reports on POPs and metals including Hg (AMAP, 1998, 2004, 2016; Letcher et al., 2010; Dietz et al., 2013a). The last AMAP report on POP effects that focused on OHCs (Letcher et al., 2010) gave a detailed review of all health effect categories available at that time for all species of Arctic wildlife and fish. The latest Hg effects assessment (Dietz et al., 2013a) gave similar insights specific to Hg, including transport, geographical trends, links to climate change and human health (Outridge et al., 2011).

Ecotoxicological studies on wildlife living in polar environments remain scarce due to challenging logistics often under harsh conditions and in remote areas. For the period 1998–2012, there were far fewer studies of POP and metal levels and any observed effects in Arctic fish than there were for top predators such as polar bear (*Ursus maritimus*) and glaucous gull (*Larus hyperboreus*). However, this has now changed and as of 2017 there have been many effect studies reported or currently underway in Arctic wildlife, including the measurement of strategic biomarker endpoints, *in vitro* experiments for top predator species, and pathological studies on fish around Arctic mining sites. Nevertheless, for wildlife and fish species endemic to the Arctic, it should be noted that there are also numerous natural (ecological and physiological) and anthropogenic factors, including climate change, invasive species and pathogens, changes in food web dynamics and predator-prey interactions, that can influence and confound the exposure to and effects of contaminants (Macdonald et al., 2003, 2005; UNEP/AMAP, 2011; Jenssen et al., 2015; McKinney et al., 2015).

In addition to reports on a growing list of CECs that are present in Arctic wildlife and fish (AMAP, 2017a,b) including polar bears from Hudson Bay (Letcher et al., 2018), Greenland (Dietz et al., 2013b,c) and Svalbard, many new studies on OHC and Hg exposure mediated effects have been published since the previous assessments (Letcher et al., 2010; Dietz et al., 2013a). Previous AMAP assessments have reported on contaminant levels in Arctic species in order to compare against levels known to elicit detrimental effects or in relation to toxicity thresholds. Such effect levels were either derived from laboratory studies, semi-field studies or observations of affected animals in the wild. The present assessment, however, takes one step further through the use of risk quotient calculations, which makes it possible to summarize the cumulative effects of environmental contaminant mixtures for which critical body burdens can be estimated (Pedersen and Petersen, 1995). Such an approach allows for an improvement in predicting or estimating the effects of contaminants in exposed Arctic wildlife and fish at the individual, population and ecosystem level. The aim of the present assessment is to summarize and update knowledge and understanding of legacy pollutants and chemicals of emerging Arctic concern and associated biological effects in Arctic marine and terrestrial mammal, bird and fish species and populations. For the first time, reporting for OHCs and Hg is combined, as the effects elicited by several individual or congener contaminants are via similar modes and mechanisms. Furthermore, consideration of combined or 'complex mixture' effects advances our understanding of impacts of contaminants as a consequence of environmentally realistic exposure scenarios for Arctic biota and their populations.

2. Contaminant exposure and effects in a changing Arctic

Authors: **Rune Dietz**, Jean-Pierre Desforges, Igor Eulaers, Bjørn Munro Jenssen, Robert J. Letcher, Melissa A. McKinney, Christian Sonne

Exposure to bioaccumulative contaminants has been recognized as a health concern in Arctic marine fish and wildlife for several decades (Letcher et al., 2010; Dietz et al., 2013a; AMAP, 2016, 2017a,b). More recently, global climate change has been recognized to lead to substantial changes in Arctic ecosystems (Post et al., 2009, 2013; Wassmann et al., 2011), and research on the interaction between climate-linked ecological change and other dynamic anthropogenic stressors, such as contaminant exposure, is essential for developing adaptation and conservation measures (Parmesan et al., 2011). Organohalogen compounds (OHCs) and mercury (Hg) typically bioaccumulate in individuals and biomagnify through food chains, and are thus subject to altered pathways as a function of ecosystem change (Macdonald et al., 2003, 2005; McKinney et al., 2015). The Arctic is an important region for research on such changes because sea ice – one of its key habitat types – is sensitive to the warming that is occurring in the Arctic at a rate twice the global average (Serreze and Barry, 2011). This phenomenon of enhanced regional warming is known as 'Arctic amplification' and is in fact thought to be largely driven by sea ice loss (Screen and Simmonds, 2010; Pistone et al., 2014; Pithan and Mauritsen 2014). Indeed, the decline in Arctic sea ice over the past three decades is considered to be one of the most concerning indicators of global climate change (Flato et al., 2013). Arctic sea ice is a key habitat for many species, and declines in its extent and thickness, later freeze-up in autumn and earlier break-up in spring, as well as rising temperatures, less snow, and altered nutrient availability, have all led to major disturbances in Arctic marine ecosystems (Post et al., 2009, 2013; Kovacs et al., 2011; Wassmann et al., 2011).

The Arctic species most likely to be affected by climate change are therefore those that depend on sea ice habitat (Laidre et al., 2008). Of these, ivory gull (*Pagophila eburnea*), Pacific walrus (*Odobenus rosmarus divergens*), ringed seal (*Pusa hispida*), hooded seal (*Cystophora cristata*), narwhal (*Monodon monoceros*), and polar bear (*Ursus maritimus*) have been identified as species that are particularly sensitive (Laidre et al., 2015). Some of these are also high level consumers within the Arctic marine food web, and so tend to have elevated contaminant levels (Letcher et al., 2010). Following the UNEP/AMAP (2011) conceptual report on climate change-induced alterations in POP exposure, McKinney et al. (2015) reviewed climate change-mediated alterations in both OHC and Hg pathways based on the published literature, most of which became available after the publication of the previous AMAP POPs and Hg effects assessments (Letcher et al., 2010; Dietz et al., 2011). Although a growing number of studies have examined the effect of climate change on contaminant exposure in Arctic wildlife, there continues to be a lack of data for Arctic Russia or Alaska (Figure 2.1). Overall, these recent studies provide empirical support for earlier predictions that climate-linked ecological changes would influence contaminant levels and pathways within Arctic ecosystems and beyond (Macdonald et al., 2003, 2005; Noyes et al., 2009; Armitage et al., 2011). However, the small number of studies available makes it difficult to hypothesize how extensive the impacts are and which ecological changes are having the greatest impact on contaminant exposure and associated health effects in Arctic wildlife and fish.

The majority of studies report that changes in POP and Hg exposure for different species and their trophic interactions are the consequence of presumed climate-induced changes in type, abundance, habitat range or accessibility of prey species, mainly in relation to changes in sea ice. Hallanger et al. (2011) reported greater POP biomagnification within the Svalbard marine food web during summer than in other seasons, which resulted in higher overall POP concentrations in the food web. Changes in marine food web dynamics have also been documented in the eastern Canadian Arctic. McKinney et al. (2012) showed that biomagnification of several legacy POPs through the food chain was higher when subarctic and transient species (such as harp seal *Pagophilus groenlandicus*) formed part of the food web than when it comprised resident Arctic marine species (such as ringed seal) only. Dietary reconstructions for polar bear supported this observation, showing that a decrease in dietary proportions of the sea ice-associated ringed seal versus open-water seal species, such as harp seal and hooded seal, resulted in slower declines in polar bear POP burdens (McKinney et al., 2013).

The possible influence of changes in body condition has also been examined. A single study (Bustnes et al., 2012) showed High Arctic colonies of fasting common eiders (*Somateria mollissima*) to have greater increases in circulating POP levels during years when ambient temperatures were lower, possibly associated with higher energy expenditures. Such climate change-mediated alteration of physiology not only occurs in bird species but has also been seen to impact contaminant pathways in fish. Long-term warming of freshwaters in the Arctic has been observed to induce metabolic stress in fish, and to enhance Hg bioaccumulation (Reist et al., 2006a). However, warming is also likely to alter growth rates and thus cause lower Hg concentrations in fish with higher growth rates (Simoneau et al., 2005). Reist et al. (2006b) stressed that cold-adapted species such as Arctic char (*Salvelinus alpinus*) and lake trout (*S. namaycush*) grow less efficiently in warmer waters, which results in higher Hg bioaccumulation. The impact of climate change on contaminant exposure through physiological pathways is highly species-specific. Øverjordet et al. (2015) showed lower Hg exposure in black-legged kittiwake (*Rissa tridactyla*) in years when they fed at a lower trophic position, which coincided with years of lower sea-ice cover and thus less access to high trophic level Arctic cod (*Arctogadus glacialis*) enriched in Hg. In contrast, trophic position did not influence Hg exposure in little auk (*Alle alle*). In the lower Canadian Arctic, increases in dietary intake of lower trophic level capelin (*Mallotus villosus*) versus Arctic cod have been recorded in

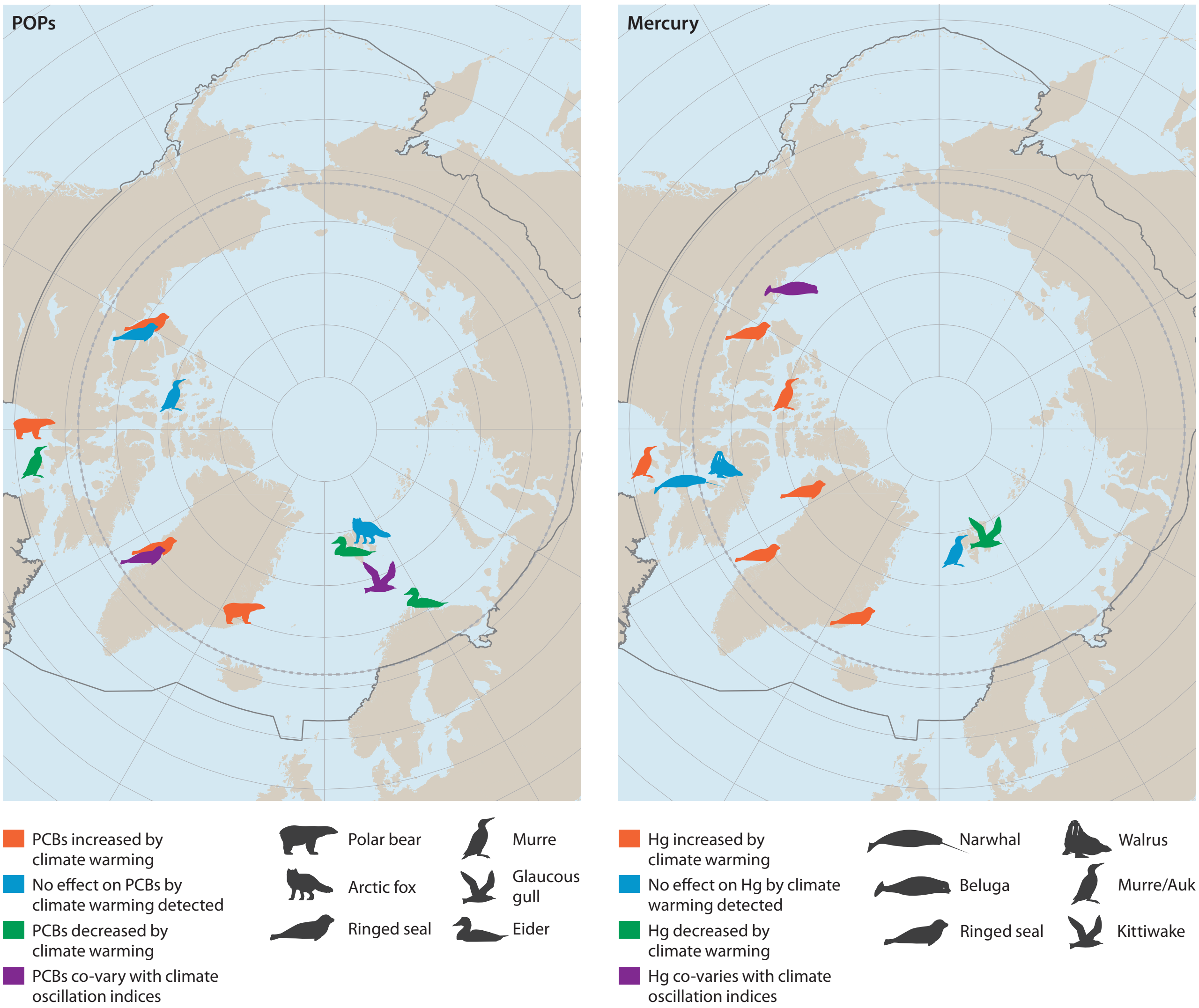

Figure 2.1 Locations and effects of reported studies documenting linkages between global climate change-induced ecological impacts and POP and Hg pathways. Adapted from McKinney et al. (2015). Details of species, locations and compounds are available in Annex 1.

thick-billed murre (*Uria lomvia*) concurrent with sea-ice declines (Braune et al., 2014). Levels of Hg would have been increasing but instead showed non-significant trends possibly related to the concomitant shift to lower trophic level prey. Two other studies on the same colonies showed climate change-mediated shifts in diet were associated with POP exposure (Braune, 2015; Braune et al., 2015).

Finally, climate change has been seen to alter the physical environment and thus alter contaminant pathways (Macdonald et al., 2003, 2005; UNEP/AMAP, 2011). Bustnes et al. (2010) documented that for glaucous gull (*Larus hyperboreus*), concentrations of polychlorinated biphenyls, hexachlorobenzene and oxychlordane declined in relation to the Arctic Oscillation (AO) index. These researchers found that the AO index value for any particular winter was negatively correlated with POP levels, whereas the AO index values for the preceding summer and winter were positively correlated with POP levels. This may indicate indirect effects of global climate change on contaminant exposure, and on possible related health impacts, via altered abiotic transport or the composition of food chains. Additional changes that have been predicted but not empirically tested in Arctic marine ecosystems include altered primary production and food web lipid dynamics, which may influence pathways of lipophilic POPs (Macdonald et al., 2003, 2005; Kainz and Fisk, 2009; UNEP/AMAP, 2011). Recent declines in sea-ice habitat have coincided with increased use of land by polar bears from the southern Beaufort Sea, which may alter the risk of exposure to pathogens and contaminants. Atwood et al. (2017) recently reported that changes in the behavior of polar bears from this population, brought about by climate-induced modifications of the Arctic marine ecosystem, may increase exposure risk to certain pathogens and alter contaminant exposure pathways. Most studies rely on correlative analyses between climate change-metrics and ecological parameters and POP or Hg tissue concentrations. However, it is important to acknowledge that, by their nature, such studies may be unable to address cause-effect linkages. Moreover, the possible interactions between climate change and contaminant dynamics are many and complex (Jenssen, 2006; Jenssen et al., 2015).

3. Biological effects of contaminant exposure in Arctic wildlife and fish

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3.1 Marine and terrestrial mammals

3.1.1 Vitamins and oxidative stress

Vitamins A, D and E are essential nutrients and endo- and exo-hormones, involved in proper function of growth, development, reproduction, bone mineral homeostasis, protection against tissue damage, and immune and endocrine function (Blomhoff, 1994; Debier and Larondelle, 2005). The group of fat soluble vitamin A compounds includes, most notably, retinol, retinyl esters and retinoic acid. Vitamin E refers collectively to several forms of tocopherols and tocotrienols, and these are the most abundant antioxidants in vertebrates. Vitamin D is both endogenously and exogenously acquired, and its metabolism in several organs results in the formation of active metabolites (e.g., 25-hydroxycholecalciferol or calcifediol) which control the homeostasis of calcium, phosphate and other nutrients. Because of their physiological importance, changes in these vitamins have been considered as biomarkers of organohalogen compound (OHC) exposure and effects in wildlife. Multiple studies on both free-ranging and captive experimental terrestrial and marine mammals in the Arctic have been published since the previous AMAP assessment (Letcher et al., 2010), linking tissue contaminant burdens to circulating and tissue residues of vitamins.

In their study of 66 beluga (*Delphinapterus leucas*) in western Arctic Canada (Beaufort Sea), Desforges et al. (2013) measured tissue levels of vitamins A and E as well as blubber OHC concentrations. Despite the confounding influence of biological factors (including age, condition and diet) on vitamin physiology, contaminant exposure was found to be a significant factor in tissue vitamin concentrations; hepatic vitamin levels were negatively correlated with the sum of polychlorinated biphenyl (ΣPCB) concentrations (3093±209 ng/g/g lw), while plasma and blubber vitamin levels increased with ΣPCB concentrations. From these results the authors calculated an integrated toxicity reference value of a 1.6 µg/g lw ΣPCB concentration threshold for vitamin A and E disruption in beluga.

Although just south of the Arctic Circle (but still within the AMAP region) in the North-East Atlantic Ocean, Hoydal et al. (2016) reported on blood plasma and liver concentrations of OHCs in relation to vitamin biomarkers in pilot whales (*Globicephala melas*) from the Faroe Islands. Faroese pilot whales had high body burdens of OHCs such as PCBs, organochlorine pesticides (OCPs) and brominated flame retardants (BFRs) (Hoydal et al., 2015). Vitamins A, D (25-hydroxyvitamin D3) and E (α-tocopherol) were sought in plasma, and vitamins A (total vitamin A, retinol and retinyl palmitate) and E (α- and γ-tocopherol) were sought in liver of Faroese pilot whales (Table 3.1). Multivariate statistical modelling showed that age and sex influenced the relationship between vitamin biomarkers and OHC concentrations. In plasma of juvenile whales, α–tocopherol was also positively correlated with all detected OHCs. Only a few significant correlations were found between single OHCs and retinol and vitamin D in plasma within the age groups studied. There were significant negative relationships between hepatic polybrominated diphenyl ether (PBDE) concentrations and retinol (only for BDE47) and γ–tocopherol (only for BDE49, BDE47, BDE99, BDE100, and BDE153) in liver. The authors concluded that the relationships between OHCs and vitamins suggest that OHCs seem to have only minor effects on vitamin concentrations in Faroese pilot whales.

Tissue levels of vitamins A and D as well as mRNA expression of vitamin A receptors were compared between ringed seals (*Pusa hispida*) from the heavily polluted Baltic Sea and the relatively unpolluted Svalbard region (Routti et al., 2010a). Baltic seals were found to have lower plasma retinol but higher hepatic gene expression of retinoic acid receptor alpha (RARα), while no differences were found in hepatic vitamin A levels. Baltic seals also had higher plasma calcitriol (vitamin D) than the Svalbard seals. Since the reverse trend in calcitriol and significant differences in hepatic vitamin A levels were noted in earlier studies of these populations in the 1990s (Nyman et al., 2003; Routti et al., 2008a,b), the authors suggested that plasma calcitriol and hepatic vitamin A concentrations are no longer depressed due to contaminant exposure in Baltic ringed seals, probably arising from significantly reduced contaminant levels in Baltic ringed seals between 2002 and 2007.

Kanerva et al. (2012) examined differences in antioxidant variables and oxidative stress in ringed seals from the Baltic Sea and Svalbard. To study the possible effects of the temporal decrease in the levels of OHCs in the Baltic seals, the samples originated from two different periods: 1996–1997 and 2002–2007. The results showed that glutathione metabolism was enhanced in the Baltic seals compared to those from Svalbard. However, no signs of oxidative damage were found in either seal population. Seals have evolved high antioxidant capacity as adaptations to prolonged fasting, sleep apnoea and hypoxemia and ischemia/reperfusion which are normally associated with increased systemic or local oxidative damage in mammals (Vazquez-Medina et al., 2012). Thus, the authors suggested that the similarities in oxidative stress levels despite the differences in antioxidant responses between the two populations could be due to a high evolutionary capacity to increase antioxidant defense in seals, and thus a high resistance to oxidative stress.

Although from regions bordering the Arctic, grey seals (*Halichoerus grypus*) from the Isle of May, Scotland were studied during the breeding season in November–December 2008 for contaminant effects on vitamin A (van den Berghe et al., 2013). Concentrations of ΣPCBs, ΣPBDEs, and several individual PCB congeners in female seals were positively correlated

Table 3.1. Summary PCBs- and/or Hg-exposure related changes on several biomarkers in Arctic key wildlife and fish species. Arrows signify a positive (▲), negative (▼), both (▲▼), or no (▶) response in relation to contaminant exposure.

	Thyroid hormones												Steroid hormones								
	Thyroid receptor expression	Deiodinase expression	Blood TT3	Blood FT3	Blood TT4	Blood FT4	Blood TT3:FT3	Blood T4:T3	Blood TT4:FT4	Blood TT4:TT3	Blood FT4:FT3	Blood FT4:TT4	Blood progesterone	Blood prolactin	Estradiol	Blood testosterone	Blood cortisol/corticost	Cortisol/corticosterone	Blood steroid hormones	Blood corticosterone	RAR expression
Marine mammals																					
Polar bear			▶	▼▲	▼	▼	▼▲	▶	▼	▼	▲	▶	▲		▶			▼▲	▼	▼	
Pilot whale			▲	▲	▲	▲	▼	▼	▲	▼	▼	▼									
Beluga	▶		▶	▼	▼	▼	▶	▶	▶	▶	▶	▶									
Ringed seal	▲	▲	▶	▲	▶	▶	▼	▶	▶	▶	▶	▶									▲
Hooded seal			▶	▶	▶	▶	▼	▶	▶	▶	▲	▶									
Baikal seal			▶		▶																
Grey seal																					
Killer whale	▲																				
Narwhal																					
Terrestrial mammals																					
Caribou and moose																					
Seabirds																					
Glaucous gull					▼			▼					▲	▼	▲						
Northern fulmar			▼																		
Ivory gull																					
Common eider																					
Thick-billed murre																					
Black-legged kittiwake			▲		▲												▲	▲			
Great skua																	▼	▼			
Birds of prey																					
White-tailed eagle																					
Fish																					
Greenland shark																					
Arctic char																	▲				
Sculpin																					
Sentinel model species																					
Greenland sledge dogs			▼	▼	▼	▼	▶	▶	▶	▲	▶	▼	▼			▼			▼	▲	
Farmed Arctic fox			▶	▶	▶	▶	▶	▼	▲	▶	▶	▶	▲		▲	▼▲			▼		

Vitamins								Histopathology				Reproduction				Immunotoxicity				Blood biochemistry			Oxidative stress		Genotoxicity
Liver vitamin A	Blood vitamin A	Blubber vitamin A	Liver vitamin E	Blood vitamin E	Blubber vitamin E	Blood vitamin D	Liver vitamin D	Liver pathology	Renal pathology	Thyroid pathology	Bone mineral density	Neurological toxicity	Reproductive performance	Reproductive organs	Eggshell thinning	IL1 expression	Immune system	Antibody response	Lymphocyte response	Liver BCCPs	Kidney BCCPs	Bone BCCPs	CYP-450	ROS	Genotoxicity
▲▼	▼		▶	▶				▲	▲	▶	▼	▲	▼				▼						▲	▼	▲
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with vitamin A concentrations in serum and inner blubber. It was concluded that these findings may suggest enhanced mobilization of hepatic retinoid stores and redistribution in the blubber, a storage site for vitamin A in marine mammals. Furthermore, there was a tendency for a positive relationship between serum concentrations of ΣPCBs and the PCB metabolite 4-hydroxy (OH)-CB-107 with serum vitamin A. These results are in accordance with those for beluga, but are contrary to the trend observed in Baltic Sea versus Svalbard seals, highlighting the complex interactions between contaminants and altered vitamin homeostasis. Furthermore, in the Scottish grey seal study, the animals comprised lactating mothers, a factor which surely confounded both vitamin and contaminant dynamics. For instance, Routti et al. (2010b) described higher concentrations of plasma and hepatic persistent organic pollutants (POP) and vitamin A levels in molting ringed seals indicating seasonal variations. Together, these studies underline the importance of considering life-history and physiological state as confounding factors when studying possible effects of POPs on vitamin status in marine mammals.

In a study on 166 polar bears (*Ursus maritimus*) from East Greenland sampled between 1994 and 2008 (Bechshøft et al., 2016), hepatic, renal and whole blood vitamin A and E were measured and compared with a suite of OHCs. In liver, vitamin A levels were positively correlated with ΣPCB, but negatively correlated with ΣPBDE, ΣDDT, and hexachlorocyclohexane (ΣHCH) concentrations, while no relationships were observed for vitamin E. Kidney cortex vitamin A and E levels correlated negatively with ΣDDTs, ΣPBDEs and/or chlordanes (ΣCHLs), while vitamin E levels also correlated positively with ΣMirex. Lastly, vitamin A concentrations correlated negatively with many levels of OHCs in whole blood and no significant correlations were reported for vitamin E. The varied directionality of effects with different OHCs is difficult to interpret in a mechanistic way, but does suggest that these contaminants are likely to be interfering with the homeostatic processes. This study also reported increased contaminant concentrations during the study period. There are indications that East Greenland polar bears have shown a dietary shift over the past three decades, with ringed seal consumption declining and harp seal (*Pagophilus groenlandicus*) and hooded seal (*Cystophora cristata*) consumption increasing which has increased the contaminant burden of these polar bears (Dietz et al., 2013a,b; McKinney et al., 2013). These results also underline the potential confounding effects of temporal dietary shifts and changing contaminant exposure when evaluating the presence of a true contaminant effect.

In a controlled study where exposed Greenland sledge dogs (*Canis familiaris*) were given a diet containing minke whale (*Balaenoptera acuterostrata*) blubber ‘naturally’ contaminated with POPs (dog ΣOHCs = 5.0 µg/g lw) and a reference group consumed uncontaminated pork fat (dog ΣOHCs = 0.09 µg/g lw), hepatic retinol concentrations were negatively associated with those of ΣDDTs and ΣPBDEs (Kirkegaard et al., 2010). Hepatic vitamin E (tocopherol) levels were lower in exposed dogs compared to control dogs, whereas no difference in vitamin A was observed for the exposed group. Kidney retinol levels were positively correlated with ΣCHL and dieldrin concentrations, while kidney tocopherol was negatively correlated with ΣPCBs. It was concluded that this controlled feeding study shows that even at low exposure levels OHCs appear to be affecting physiological vitamin levels. However, in a complementary study using domesticated Arctic foxes (*Vulpes lagopus*) fed the same minke whale blubber or pork fat diet, there were no observed differences in plasma retinol or tocopherol between feeding groups (Hallanger et al., 2012). The same sledge dog cohort was also used to examine the status of hepatic vitamin D-3 (D3) and hepatic and plasma 25-OH vitamin D-3 (25OHD3) in mothers and their pups (Sonne et al., 2014a). The study showed that in the exposed mothers, the hepatic concentration of 25OHD3 was significantly lower than in the control mothers. No between-group differences were identified for hepatic D3 concentrations or plasma 25OHD3 concentrations in mothers. However, hepatic D3 and plasma 25OHD3 concentrations were significantly higher in the exposed pups compared to the control pups, but hepatic 25OHD3 concentrations showed no difference. When the results from both mothers and pups were pooled, a significant negative relationship between adipose tissue concentration of ΣPCBs and plasma 25OHD3 concentrations was identified, and a similar trend was found for hepatic 25OHD3 versus adipose ΣPCBs. Due to differences in dietary composition of the food provided to the two groups, the results indicate that the homeostasis and metabolism of vitamin D compounds may respond differently to the dietary composition of fatty acids and OHC exposure. It was concluded that it is unclear whether the lower level of 25OHD3 in the liver of exposed dogs would have had any negative effects on immunity and reproduction.

Persistent organic pollutants and their metabolites can induce oxidative stress through the intracellular formation of reactive oxygen species, which can also affect the metabolism of xenobiotic substances. To combat such effects, the antioxidant system in organisms is important. The antioxidant system consists of low molecular weight scavengers and antioxidant enzymes that interact in reducing the harmful biological effects of oxyradicals (Regoli and Giuliani, 2014). Vitamin E is one such group of endogenous molecules that plays an important role in antioxidant defense in organisms (McDowell, 2000). However, several other molecular and cellular mechanisms are involved in tackling the oxidative stress caused by pollutants. Although levels of POPs may be very high in Arctic animals, especially in some marine mammals, there is limited information on the extent to which POPs induce oxidative stress in Arctic mammals and the extent to which the antioxidant system responds to the formation of oxyradicals induced by the exposure. It is therefore important to consider comparative studies on mammals outside the Arctic Circle, which are in closer proximity to pollution sources and so will have higher OHC exposure, such as animals from the Baltic Sea. Baikal seals (*Pusa sibirica*) are another example, since Lake Baikal lies just south of the Arctic Circle, but their life history includes breeding on ice and therefore these seals experience similar environmental conditions to ringed seals and other Arctic seals.

It was recently reported that exposure to OHCs induces production and increased activities of cytochrome P450 (CYP) enzymes in Baikal seals (Hirakawa et al., 2011). In animals sampled in summer 1992 and again in 2005, chronic exposure to polychlorinated-dibenzo-*p*-dioxins and related dioxin-like compounds were related to oxidative stress induction. The expression levels of CYP1A2 were positively correlated with levels of malondialdehyde, a biomarker of lipid peroxidation, and of etheno-dA, a DNA adduct. Thus, in seals OHC exposure may cause enhanced lipid peroxidation through the production of reactive oxygen species triggered by CYP1A2 induction. Furthermore, there was a positive relationship between malondialdehyde concentrations and heme oxygenase activities, suggesting heme degradation by reactive oxygen species. The responses reported for other biomarkers of inflammation, indicated that the increased oxidative stress caused by the induction of CYP1 isoenzymes resulted in increased inflammation. The authors concluded that Baikal seals may suffer from effects of chronic exposure to dioxin-like OHCs caused by increased oxidative stress, heme degradation and inflammation induced through the induction of CYP1 isoenzymes. In another study on Baikal seals sampled in summer 2005, hepatic expression levels of CYP1 genes were positively correlated with the concentrations of OH-PCBs, compounds known to induce oxidative stress (Nomiyama et al., 2014). This indicates that these OH-PCB metabolites may play an important role in inducing oxidative stress in seals.

3.1.2 Endocrinology

According to Jenssen (2006) the effects of global climate change on biodiversity and ecosystem function encompass multiple complex dynamic processes. Climate change and exposure to endocrine-disrupting chemicals are currently regarded as two of the most serious anthropogenic threats to biodiversity and ecosystems. This implies a need for particular attention to the possible effects of endocrine-disrupters on the ability of Arctic marine mammals (and seabirds) to adapt to environmental alterations caused by climate change. Relationships between POPs and hormones in Arctic mammals (and seabirds) imply that these chemicals pose a threat to the endocrine systems of these animals. The strongest relationships have been reported for the thyroid hormone system, but effects are also seen in sex steroid hormones and cortisol (Letcher et al., 2010 and references therein). Although behavioral and morphological effects of POPs are consistent with endocrine disruption, no direct evidence exists for such a cause-effect relationship. Because endocrine systems are important for enabling animals to respond to environmental stress, endocrine-disrupting chemicals may interfere with adaptations to increased environmental stress (Letcher et al., 2010). This is likely to concern adaptive responses regulated by the thyroid, sex steroid, and glucocorticosteroid systems.

Thyroid hormones are an endpoint in studies of contaminant bioaccumulation as well as global climate change. Morphological and pathological changes in thyroid hormone balance can affect reproduction success, growth, thermoregulation and immune competence of neonatal and adult individuals, functions crucial for life in the Arctic (Zoeller et al., 2002; Grandjean and Landrigan, 2006; Klecha et al., 2008).

The previous AMAP assessment report included several studies on associations between POP body burdens and concentrations of thyroid hormones in marine mammals (Letcher et al., 2010). Since 2010, several new articles have been published regarding effects on thyroid hormones in marine mammals as well as mechanisms involved in thyroid disruption. New literature is also available concerning experimental studies on the effects of POPs on thyroid and steroid hormones and vitamin D using Greenland sledge dogs (Kirkegaard et al., 2011; Sonne et al., 2014a,b, 2016), domesticated Arctic foxes (Hallanger et al., 2012; Helgason et al., 2013; Rogstad et al., 2017) as well as vitamin loads from field sampled West Greenland narwhal (*Monodon monoceros*) (Sonne et al., 2010a, 2013a). Since the previous AMAP assessment, endocrine-disrupting chemical-related studies have been reported for seals (Routti et al., 2010a; Wang et al., 2010; Gabrielsen et al., 2011; Villanger et al., 2013; Brown et al., 2014; Imaeda et al., 2014), toothed whales (Buckman et al., 2011; Siebert et al., 2011; Villanger et al., 2011b; Schwacke et al., 2012; Noel et al., 2014) and polar bears (Gutleb et al., 2010; Knott et al., 2011; Simon et al., 2011; Sonne et al., 2011; Villanger et al., 2011a; Bytingsvik et al., 2013; Gabrielsen et al., 2015).

Routti et al. (2010a) reported thyroid receptor beta (TRβ) and deiodinase I hepatic gene expression levels to be higher in ringed seals from the polluted Baltic Sea relative to ringed seals from the less-polluted waters around Svalbard. Free 3,3',5-triiodothyronine (FT3) in plasma was also found to be higher in Baltic seals. The authors concluded that thyroid hormones are a more sensitive effect biomarker of POP exposure in ringed seals than vitamin effects. Thyroid hormone status was evaluated in East Greenland hooded seals and OH-PCBs were found to be the major contaminant class of interest, showing negative correlations with free thyroxine FT4:FT3 and TT3:FT3 ratios in pups only (Gabrielsen et al., 2011; Villanger et al., 2011a). In contrast, Brown et al. (2014) did not find any association of thyroid receptor alpha (TRα) expression with ΣPCBs in ringed seals at a military radar site locally polluted with ΣPCBs on the Labrador coast of Arctic Canada. Similarly, POPs were not significant contributors to thyroid hormone status in Baikal seals (Imaeda et al., 2014).

Buckman et al. (2011) found that ΣPCBs increased expression of the Trβ gene in killer whales (*Orcinus orca*) from British Columbia, Canada, although this was not observed for beluga from the Beaufort Sea (Noel et al., 2014). A study on Svalbard beluga showed that concentrations of the known or suspected thyroid disruptive contaminants BDE28, BDE47, BDE99, BDE100, and BDE154, hexachlorobenzene (HCB), and CB105

Several examples of detrimental health effects due to high PCB concentrations have been documented in killer whales were negatively correlated with circulating levels of total T4 (TT4), FT4 and FT3 (Villanger et al., 2011b). Hoydal et al. (2016) investigated pilot whales from the Faroe Islands for blood plasma and liver concentrations of OHCs in relation to hormone biomarkers. Multivariate statistical modelling showed that age and sex influenced the relationship between biomarkers and OHCs. Some significant positive relationships were found between OHCs and thyroid hormone concentrations in the youngest juveniles. The authors concluded that the OHC exposure observed in the investigated pilot whales seems to have only minor effects on thyroid hormone concentrations.

In southern Beaufort Sea polar bears, negative relationships were reported between PCBs and TT4 while TT3 was positively correlated with mercury (Hg) (Knott et al., 2011). For pollutants in polar bears, it was shown that OH-PCBs (Gutleb et al., 2010; Simon et al., 2011; Bytingsvik et al., 2013) and branched nonylphenol (Simon et al., 2013) bind to transthyrethin in competition with thyroid hormones, and that these compounds may fully saturate the available transthyrethin. A study on East Greenland polar bears confirmed negative relationships between individual PCB congeners and their OH-PCB metabolites and T4 in both plasma and muscle (Gabrielsen et al., 2015). Furthermore, in general, PCBs, OH-PCBs and PBDEs were positively correlated to D1 and D2 activities, whereas concentrations of OCPs were negatively associated (Gabrielsen et al., 2015). In East Greenland polar bears, some OHCs were especially important in explaining variation in circulating thyroid hormone levels. BDE99, BDE100, and BDE153, CB52 and CB118, cis- and trans-nonachlor, and tri- and pentachlorobenzene, showed both negative and positive relationships with thyroid hormones (Villanger et al., 2011a). Furthermore, thyroid hormone levels in adult male polar bears seemed less influenced by OHC exposure than in females. Experimental studies on Greenland sledge dogs showed that free and total T3 and T4 were lower in exposed versus control females over 10 months of age, and TT3 was lower through 3 to 12 months of age in exposed pups (Kirkegaard et al., 2011). A positive association between dieldrin and TT3 was also reported. An experimental study on juvenile domesticated Arctic foxes showed no impacts of administered POP exposure for thyroid hormone and thyroid-stimulating hormone (Hallanger et al., 2012). Thus, the thyroid hormone system appears to be affected by OHCs and their metabolites in several Arctic species. However, these effects appear to show sex and age differences.

Since the last AMAP report (Letcher et al., 2010), a number of new studies have focused on the impact of OHC exposure on cortisol. Up to 2009, only a single study investigating 251 Svalbard polar bears had been published (Oskam et al., 2004). The authors concluded that high concentrations of OHCs in polar bears may alter plasma cortisol concentrations. Such disturbances during critical stages of fetal life may induce abnormal neurobehavioral function in adult life (Oskam et al., 2004; Ropstad et al., 2006). Since 2009, six publications have become available, focusing mainly on hair cortisol (Bechshøft et al., 2011, 2012a,b, 2013, 2015) with just one investigating plasma cortisol (Chow et al., 2011). Bechshøft et al. (2012b) concluded that hair cortisol concentrations in 23 East Greenland polar bears were correlated with adipose burdens of OHCs, although showing both up- and down-regulation of cortisol levels depending on the specific compound. The down-regulating OHCs included BDE99 and BDE153, CB170/CB190, CB180, and CB201, while those up-regulating cortisol levels were CB66/CB95, α-HCH, heptachlor epoxide, dieldrin, BDE47, and dichlorodiphenyldichloroethane (*p,p'*-DDD). Bechshøft et al. (2012a) did not show any obvious OHC impact on interannual hair cortisol levels during both a pre- (1892–1927) and post-industrial period (1988–2009) or any trends over the study period. In contrast, Bechshøft et al. (2015) found cortisol in male Western Hudson polar bear guard hair to be positively correlated to Hg in males when controlling for age and fitness. No such relationship was found for females.

With respect to reproductive steroid hormones, Svalbard polar bears had concentrations of OH-PCB that were correlated to reduced plasma concentrations of pregnelolone and androstenedione in female polar bears (Gustavson et al., 2015). The authors suggested that CYP17 may be a potential target of OH-PCBs. In an experimental study on Greenland sledge dogs, an over-compensation of levels of reproductive hormones in OHC-exposed female dogs was reported, possibly due to disruption of negative feedback systems (Sonne et al., 2014a). In addition, plasma testosterone concentrations in OHC-exposed juvenile male domesticated Arctic foxes were lower than in control males (Hallanger et al., 2012).

3.1.3 Reproduction and genotoxicity

Previous assessments have outlined the effects of various OHCs on reproductive pathology and potential influences on reproductive performance (see Letcher et al., 2010). Since 2010, there have been only three new studies that examined the effects of POPs on reproduction in polar bear (Dietz et al., 2015; Pavlova et al., 2016a,b). Pavlova et al. (2016a) documented that PCBs in addition to reproduction failure are associated with endocrine disruption, cancer and immunosuppression. Pavlova et al. (2016a,b) used individual-based models to explore whether and how PCB-associated reproductive failure could affect the dynamics of a hypothetical polar bear population exposed to PCBs to the same degree as the East Greenland subpopulation. Two alternative types of reproductive failure in relation to maternal ΣPCB concentrations were considered: increased abortion rate and increased cub mortality as well as allee effects (correlation between population size and mean individual fitness) in Svalbard bears. An allee effect occurs when decreasing population density negatively influences some aspects of individuals' fitness or population growth. It was found that the quantitative impact of PCB-induced reproductive failure on population growth rate depended largely on the type of reproductive failure involved. Comparing the model predictions of the age-dependent trend of ΣPCBs in females with actual field measurements from East Greenland indicated that it was unlikely that PCB exposure caused a high incidence of abortions in the specific subpopulation. However, the study could not exclude that PCB exposure might contribute to higher cub mortality. These results highlight the need for further research on the possible influence of PCBs on polar bear reproduction regarding their physiological pathway. This includes determining the exact cause of reproductive failure, i.e. *in utero* exposure versus lactational exposure of offspring, the timing of offspring death, and establishing the most relevant reference metrics for the dose-response relationship. With respect to the allee effect it was concluded that the low representation of 10–14 year-old males among breeding males documented in Svalbard in the mid-1990s could have resulted from PCB contamination.

Dietz et al. (2015) modelled the risk of reproductive impairment and genotoxicity using risk quotients (RQ = BR / CBR = Body Residue / Critical Body Residue) in OHC-exposed polar bears harvested over the period 1999–2008 in eleven circumpolar subpopulations from Alaska to Svalbard. This RQ evaluation was based on the CBR concept and a physiologically-based pharmacokinetic (PBPK) modelling approach using OHC concentrations measured in polar bear adipose or liver tissue. The total additive RQ from all OHCs was above the toxic effect threshold (i.e., RQ = 1) in all polar bear subpopulations for both endpoints. Risk was lowest in Alaska and highest in East Greenland. For reproductive effects, PCBs were the main contributor, contributing 87–94% and the sum of methylsulfonyl-PCBs (ΣMeSO$_2$-PCBs), metabolites of PCBs, were the second highest effect contributor (3–5%). For carcinogenic effects, PCBs were likewise the main contributor (71–90%) while perfluorooctane sulfonic acid (PFOS) was the second highest contributor (3–19%).

Gilmore et al. (pers. comm.) quantified DNA strand breaks in lymphocytes from Svalbard polar bears. Whole blood from 13 males and 34 females was sampled, and lymphocytes were isolated and subject to the comet assay. Baseline strand breaks (tail intensity) ranged from 2% to 18% for males and 7% to 19% for females, with medians of 11% and 13% respectively. Plasma samples were analyzed for PCBs, HCB, β-HCH, *p,p'*-DDE, *p,p'*-DDT, OH-PCBs, oxychlordane, *trans*-nonachlor, PBDEs, and hydroxylated polybrominated diphenyl ethers (OH-PBDEs), concentrations of which were within ranges reported earlier. Principal Component Analysis showed a significant negative relationship between DNA strand breaks and a range of PCBs (CB137, CB138, CB156, CB180, CB183), OH-PCBs (OH-CB-130, OH-CB-146, OH-CB-187), and BDE47. Age, gender and body condition did not appear to affect the level of strand breaks. The authors noted some possible explanations for this observation: (i) increased POP exposure may have led to induced DNA repair and/or antioxidant defense mechanisms, thereby decreasing the number of strand breaks, and (ii) exposure to POPs may have increased the turnover of lymphocytes, thereby changing the relative proportion to cells with lower accumulated damage.

Polar bear studies over 35 years in East Greenland are among the most thorough time trend and effect investigations on contaminants in the Arctic.

The isolation of a complex contaminant mixture from polar bear adipose and killer whale blubber samples with high PCB loadings was recently used for *in vitro* exposure experiments with live lymphocytes from a range of Arctic mammals (Desforges et al., 2017a). The study included lymphocyte samples from three polar bears and although results were not entirely consistent, there were indications of the same inverse dose-response as observed in the field (i.e., cells exposed to higher contaminant mixture concentrations had reduced DNA damage). Increased cell proliferation did not appear to play an important role, so results from this study would suggest induction of other mechanisms of cellular protection.

3.1.4 Immunology

The ultimate function of the immune system is to protect the host against infectious diseases as well as aberrant macromolecules such as cancerous cells (Abbas et al., 2012). Resistance to infection requires the concerted effort from the complex network of tissues, cells and molecules that comprise the immune system. Mammalian immunity consists of innate and adaptive responses, which are two separate but interconnected functional arms of the immune system (see Figure 3.1). Immune responses are also divided into cellular (cell mediated) and humoral immunity. Modulation of both arms of the immune system in marine mammals has been associated with exposure to POPs and metals (Desforges et al., 2016).

The previous AMAP report included several studies of immunotoxicity associated with POPs in polar bears from East Greenland and Svalbard as well as in sledge dogs exposed in a controlled feeding experiment (Letcher et al., 2010 and references therein). These studies found significant effects of POPs on humoral and cellular immunity for these Arctic species suggesting that contaminants may be impairing the ability of animals to respond to infectious pathogens. Since 2010, ten new articles have been published on POPs and Hg and their immune system effects in Arctic marine and terrestrial animals.

In a controlled feeding experiment on West Greenland sledge dogs, seven females and their four pups were fed minke whale blubber rich in POPs while a control group of eight females and their five pups were fed uncontaminated pork fat (Sonne, 2010). Exposed pups, but not adults, had reduced and delayed Immunoglobulin G (IgG) antibody production with circulating levels of IgG in all pups correlating to blood concentrations of ΣPCBs, ΣPBDEs and HCB. The specific antibody response to influenza virus, tetanus toxoid and diphtheria toxoid were also measured in the dogs, although reduced antibody production against influenza virus in exposed pups was the only adaptive/humoral parameter reportedly affected (Sonne et al., 2006a). Hepatic mRNA expression of interleukin-1β (IL-1β), an important

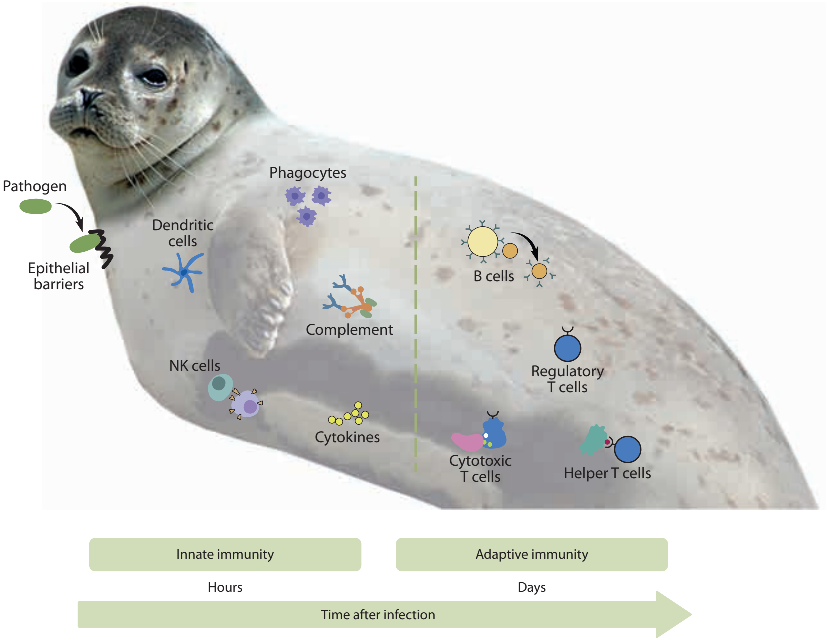

Figure 3.1 Cells and molecules of the mammalian innate and adaptive immune system (modified from Desforges et al., 2016).

pro-inflammatory cytokine, in ringed seals from Svalbard and the Baltic Sea were positively correlated with hepatic ΣPOPs (Routti et al., 2010a). Similarly, hepatic IL-1 mRNA expression was positively correlated with blubber ΣPCBs in 41 ringed seals collected by aboriginal hunters in northern Labrador, Canada (Brown et al., 2014). An effect threshold for this immune endpoint in Labrador seals was established at 1.37 µg/g lw.

In vitro T-lymphocyte proliferation after exposure to four PCB congeners (CB138, CB153, CB169, CB180) and two perfluoroalkyl substances (PFASs), i.e. PFOS and PFOA (perfluorooctanoic acid), was assessed in leukocytes collected from lymph nodes of 20 free-ranging ringed seals sampled in East Greenland (Levin et al., 2016). Nonplanar PCB congeners CB138, CB153 and CB180 reduced T-cell proliferation with EC50 (half maximal effective concentration) values of 13.3, 20.7 and 20.8 µg/g, respectively, while the coplanar CB169 had no effect (up to 20 µg/g). Neither PFOS nor PFOA modulated lymphocyte proliferation at concentrations up to 0.3 µg/g ww. The authors pointed out that although the EC50 values are typically above observed adipose PCB concentrations in Greenland ringed seals, the threshold effect levels of approximately 1–2 µg/g are commonly exceeded.

In another study, peripheral blood leukocytes were isolated from four captive beluga and used for Hg exposures in *in vitro* assays of lymphocyte proliferation, intracellular thiol production and metallothionein production (Frouin et al., 2012). Reduced T-lymphocyte proliferation was found at 1 µM Hg and 0.33 µM methylmercury (MeHg) and reduced production of intracellular thiol occurred at 10 µM Hg and 0.33 µM MeHg. Metallothionein was induced by 0.33 µM MeHg, but not by Hg (up to 10 µM). Selenium offered protection in lymphocyte proliferation assays against Hg toxicity only at the highest exposure levels. The authors highlighted that current Hg levels in Arctic beluga fall within the range of exposures that elicited *in vitro* immune suppression, thus potentially contributing to impaired resistance to infections.

In an extensive study of immunotoxicity in marine mammals using blubber-derived complex contaminant 'cocktails' from polar bears and killer whales, Desforges et al. (2017a,b) reported significant *in vitro* effects on lymphocyte proliferation, natural killer cell activity, and phagocytosis in lymphocytes from cetaceans, seals and polar bears. This *in vitro* study found that the polar bear cocktail was more toxic/potent than the killer whale cocktail and related this difference to the contaminant composition making up each mixture of compounds, with the polar bear lymphocytes having a more pronounced ability to metabolize POPs to potentially more immune active metabolites. For lymphocyte proliferation, the overall marine mammal EC50 values were 0.94 and 6.06 µg/g for the polar bear and killer whale cocktails, respectively. Ultimately, this study reported that *in vitro* immune effects occurred at lower concentrations using the realistic contaminant mixtures relative to previous studies utilizing single compound exposures.

Finally, in a review and meta-analysis of all immunotoxicity studies of marine mammals to date, including several Arctic species, exposure to environmental pollutants in field studies, captive-feeding studies and *in vitro* laboratory studies was associated with alterations of the two arms of the immune system, the innate and adaptive, and included cellular and humoral effects (Desforges et al., 2016). Despite differences in study design and animal life history, the review concluded that pollutants, especially PCBs and Hg, systematically suppressed marine mammal immune function. Immune endpoints evaluated in marine mammals fell into several major categories: immune tissue histopathology, hematology/circulating immune cell populations, functional immune assays (lymphocyte proliferation, phagocytosis, respiratory burst, and natural killer cell activity), immunoglobulin production, and cytokine gene expression. Sufficient data were available to calculate effect threshold levels for lymphocyte proliferation and phagocytosis; threshold effect levels for lymphocyte proliferation in polar bears, cetaceans and pinnipeds fell within the range <0.01–10 µg/g lw for PCBs and <0.01–2.4 µg/g ww for metals (Hg, MeHg and cadmium), and 0.6–1.4 (PCBs) and 0.08–1.9 µg/g ww (Hg) in phagocytosis assays. Given the weight of evidence for contaminant-mediated immune suppression, Desforges et al. (2016) concluded that exposure to immunotoxic contaminants may have significant population level consequences as a contributory factor to increasing anthropogenic stress and infectious disease outbreaks.

3.1.5 Skeletal system

Measures and analyses of the skeletal system have gained recognition as valid health endpoints as this organ system can reflect the negative impacts of nutritional and endocrine status. The skeletal system is mainly composed of proteins, calcium, phosphorus and magnesium, and is linked to the maintenance of a dynamic micronutrient homeostatic reservoir (Sarazin et al., 2000; Ganong, 2005). Bone composition and mineral density have been shown to reflect environmental stress, such as exposure to contaminants (Andrews, 1989; Lind et al., 1999, 2000; Jämsä et al., 2001; Lundberg et al., 2006) and starvation (Talbott et al., 2001). Disruptions of the hypothalamic-pituitary endocrine axis may lead to changes in parathyroid hormone, cortisol, steroid and thyroid hormone concentrations that increase bone resorption and to decreased bone formation (Selye, 1973; Colborn et al., 1993; Feldman, 1995; Damstra et al., 2002; Ganong, 2005; Tung and Iqbal, 2007). In addition, starvation and other energetic-driven stressors may also result in loss of bone density and cause compositional changes (Polischuk et al., 2002; Mustonen et al., 2006). Maintaining a healthy skeletal system is therefore important for Arctic wildlife both at the individual and population level.

The previous AMAP report focused mainly on polar bear studies, while controlled studies of captive Arctic fox and sledge dogs were reported the results did not really support the polar bear findings. In more recent studies, genetic stock differences were found for fluctuating asymmetry and morphology between East Greenland and Svalbard polar bears, while pollutant effects could not be confirmed (Sonne, 2010; Pertoldi et al., 2012). The results did suggest that POP exposure probably reduced bone density in skulls and bacula (penile bone) of East Greenland polar bear subadult and adult males, while female skulls did not seem to be affected at all (Sonne, 2010).

Since the last AMAP POP and Hg effects reports, two new articles have been published on skeletal system effects from contaminant exposure in the Arctic. Sonne et al. (2013b) investigated the size and bone density of polar bear skulls sampled in East Greenland over the period 1892–2010, and found that bone mineral density (BMD) had decreased significantly in males over the study period (Figure 3.2). Dieldrin had a significant negative association with BMD while DDT, dieldrin and PBDEs were positively correlated with condylobasal length. The reasons for these correlations were not clear, however, given the timing and relationships found, the decrease in skull size and BMD over the past 120 years was likely to have been influenced by a combination of POP exposure and climate change, with selective hunting and genetic changes as possible additional factors.

A second study on the skeletal system, linked bacula BMD with concentrations of POPs that had been long-range transported to Canadian and East Greenland polar bears (Sonne et al., 2015a). Size and BMD of penile bones were measured in 279 individuals born between 1990 and 2000 from eight polar bear subpopulations, including seven stocks from the Canadian Eastern Arctic and one in East Greenland. There was a clear latitudinal and longitudinal gradient with Western Hudson bears having the highest BMD and northern East Greenland polar bears having the lowest. The BMD results showed the opposite geographical pattern to POP concentrations, which was supported by an RQ estimation showing that PCBs may disrupt normal reproduction and development in the bears with highest POP levels. Reductions in BMD may in general reflect reduced health, but a lower penile BMD theoretically leading to fractures could lead to increased risk of species extinction because of mating and subsequent fertilization failure.

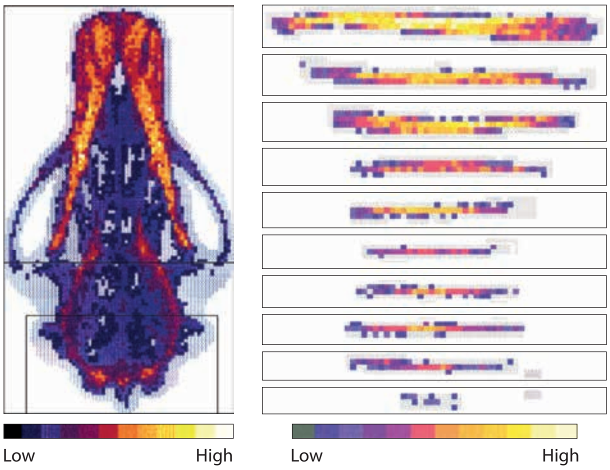

Figure 3.2 DXA scanning images of (left) a 12-year-old female East Greenland polar bear skull sampled in 1972 and (right) polar bear penis bone / baculum from ten Canadian polar bears aged 2–28 years sampled in the period 1997–2003 (modified from Sonne et al., 2004, 2006b).

Daugaard-Petersen et al. (2018) investigated skull size (condylobasal length) and BMD in polar bears from East Greenland and Svalbard sampled during the period 1892–2015 in East Greenland and 1964–2004 at Svalbard. Adult males from East Greenland showed a continuous decrease in BMD from 1892 to 2015 indicating that adult male skulls collected in the early pre-pollution period had the highest BMD. A similar decrease in BMD over time was not found for the East Greenland adult females. No temporal change was found for BMD or skull size in Svalbard polar bears nor was there any significant difference in BMD between the Svalbard and East Greenland subpopulations. When correcting for age and sex, BMD in East Greenland polar bears increased with increasing concentrations of ΣPCB, ΣHCH, HCB and ΣPBDE while skull size increased with ΣHCH concentrations in the period 1999–2014. The authors suggested that environmental changes over time, including exposure to POPs, may affect bone density and polar bear size.

As with any other species, a healthy and functional skeletal system is important for polar bears. Any impact on for instance baculum or mandibular functioning and pathology will influence their wellbeing, and ability to reproduce and hunt, and thereby general fitness. Ultimately, these effects may have an impact on the bears' immune status and ability to reproduce. It is therefore important to continue monitoring programs on the skeletal status and bone density of polar bears and other important key species in the Arctic.

3.1.6 Histopathology

Detecting adverse health effects in Arctic animals is not an easy task in such a remote region. Access to plasma and serum samples is often limited due to lack of centrifuges and electricity, which results in limited screening of important

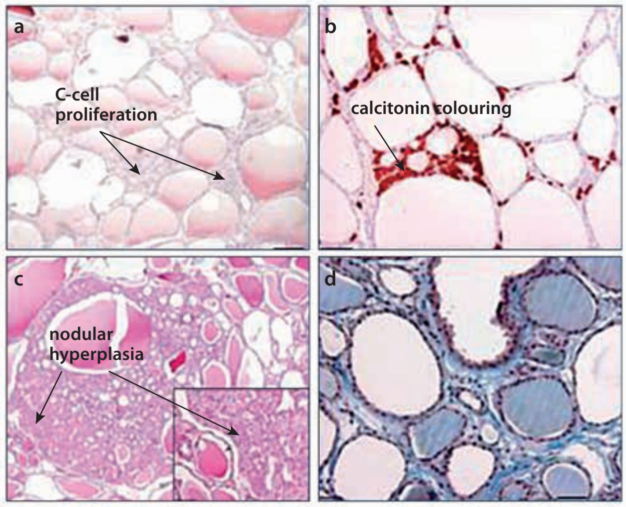

Figure 3.3 Thyroid lesions in East Greenland polar bears sampled during 1999–2009. (a) C-cell proliferations, (b) calcitonin colouring, (c) nodular hyperplasia, (d) interstitial fibrosis (modified from Sonne et al., 2011).

health parameters, including diseases. On the other hand, histological screening of various tissues is less cumbersome. Tissues are immediately stored in formaldehyde with alcohol to prevent freezing artifacts, and tissue alterations and lesions may be observed on later examination, and as needed, interpreted in relation to a broad suite of different biological and chemical parameters (Sonne, 2010). Such analyses may provide an indication of the specific organ status, the value of which is important for Arctic wildlife at both the individual and population level.

The liver is a target organ as many chemical contaminants concentrate there. The liver is also susceptible to toxic damage because it is the main organ to metabolize accumulated chemical contaminants. In this metabolic chain reaction, metabolites are produced that may be even more cell toxic than the initial parent contaminant (Letcher et al., 2010). This means that despite the large metabolic and detoxifying capacity of the liver, lesions are often seen in this organ due to oxidative stress and metabolic disorders (Al-Saleh et al., 2003; Klaassen et al., 2007). The actual effects of chemical contaminant exposure are hard to distinguish since they may elicit or mimic other normal reactions to stress from external/environmental factors. Effects are divided into direct cellular toxicity and indirect toxicity mediated by a disruption in various endocrine and immunological feedback systems (Letcher et al., 2010; Sonne, 2010; Dietz et al., 2013c). The kidney is also an important organ for multiple vital functions including clearance of metabolic waste products, water and electrolyte homeostasis, thereby blood pressure maintenance, vitamin D production and calcium homeostasis (Ganong, 2005). Chronic kidney damage may therefore disrupt bone density and clearing of metabolic waste products from blood. Such impacts may be detected via specific analytical modalities (Maxie, 1993; Confer and Panciera, 1995; Klaassen et al., 2007).

Since the previous AMAP assessments, seven articles have been published on histopathology in relation to contaminant exposure in Arctic marine and terrestrial mammals. The summarized results suggest that liver and renal alterations in polar bears, in addition to the chemical contaminant exposure, are likely to be due to a combination of age and recurrent infections.

Sonne et al. (2012a) were the first to present information on histopathological effects over a longer time period. Liver and kidney samples from 122 East Greenland polar bears were examined over the 12-year period 1999–2010 where the prevalence of various liver and kidney lesions were reported. Polar bear adipose PCB concentrations (5674–17 591 ng/g lw) were studied for associations with seven observed liver and kidney lesions and changes. Of these, fat accumulation in the liver, kidney cell infiltrations, glomerular sclerosis and tubular hyperplasia decreased, while detectable liver Ito cells, kidney glomerular capillary wall thickening and interstitial fibrosis increased over time. Several of these observed tissue alterations were either positively or negatively correlated with adipose concentrations of Hg and POPs.

Thyroid glands were examined in a smaller number of individuals (n=20) around the same period (Sonne et al., 2011). The analyses showed 12 polar bears to have normal thyroid tissue, while the remaining eight showed C-cell proliferations, nodular hyperplasia or interstitial fibrosis (Figure 3.3). There were however, no significant differences in POP concentrations between individuals with and without lesions. The observed lesions in polar bears were similar to those found in POP-exposed laboratory animals and other highly contaminated wildlife, and were likely to be the combined result of POP exposure, normal metabolism- and thermoregulation, autoimmunity or infections. The authors could not exclude the possibility that lesions were coincidental idiopathic alterations due to hypothalamic-pituitary-thyroid axis interactions.

The histopathology of livers and kidneys has also been investigated in Faroese pilot whales (n=14), Greenland narwhal (n=19) and ringed seals (n=40) (Sonne et al., 2010a, 2013a, 2018). Chemical analyses showed PCB (13 947–34 543 ng/g lw) and Hg (54–351 µg/g ww) concentrations in pilot whale blubber and liver to be high compared to other marine mammals. Histopathological examinations also showed high prevalence (>35%) of both renal and liver lesions, which is similar to those previously found in contaminated Arctic marine mammals (Sonne et al., 2010a). Studies of narwhal revealed lower liver Hg concentrations (0.4–32 µg/g ww), but nevertheless similar histological changes in this tissue as found in other histological contaminant studies of Arctic mammals (Sonne et al., 2013a, 2018). A single study on ringed seals reported hepatic Hg levels in Thule (0.28–23.3 µg/g ww) and Godhavn (0.45–8.0 µg/g ww). The study showed that Hg had a significant correlation to the occurrence of hepatic hemosiderinosis in ringed seals. The histological changes found were similar to those found in comparable studies on other Arctic mammals. The authors suggested that the histopathological changes were likely to be the combined result of age and contaminant exposure, with contaminant exposure acting as an aggravating co-factor in the development of tissue lesions. However, larger studies are required to determine the influence of natural versus anthropogenic factors on liver and kidney histopathology. Likewise, efforts should be made to resolve the issue of extrapolating histopathological changes to particular health effects, as well as effects at the population level.

Only a few studies have been conducted on Arctic terrestrial mammals. In a study on sledge dogs fed marine food items, a physiologically-based pharmacokinetic (PBPK) model was developed to estimate the risk of POP exposure on various health endpoints (Sonne et al., 2015b), allowing the estimation of risk quotients (RQs) for POP exposure and potential effects in Arctic top predators, such as polar bears. The PBPK model was based on a sledge dog cohort of females and showed that POPs induced liver histopathological effects. In 75% of the dogs, dieldrin was detected in concentrations sufficient to induce pathological changes in liver tissue followed by ΣCHLs > ΣPCBs > ΣDDTs > ΣPBDEs > HCB > ΣHCHs. The authors concluded that RQs based on adipose tissue concentrations were the best reflector of health effects, and suggested that this metric should be used in future risk assessments of POP exposure in Arctic top predators. Altogether, these studies on terrestrial and marine mammals point towards POPs and Hg as co-factors in the development of liver, kidney and thyroid lesions, and possibly hormonal disruptions.

Larter et al. (2016) reported Hg to be higher in mountain caribou (*Rangifer tarandus caribou*) than moose (*Alces alces*), although levels were much lower than for cadmium (<1.0 µg/g ww). Minor histological changes in the proximal tubules and glomerulus of the kidney were found in each species, although such changes were rare and did not indicate damage consistent with metal accumulation. The authors concluded that while elevated cadmium levels did not appear to cause pathological damage in these terrestrial mammals, there remained significant implications for northern populations who consume country foods.

3.1.7 Neurotoxicology

The neurotoxicological potential of MeHg is firmly established with well-characterized human poisoning events in Japan, Iraq, and Brazil (Mergler et al., 2007). Similarly, for OHCs there is an established and growing literature base showing that, for example, exposure to PCBs and PBDEs is associated with adverse neurological outcomes (Mariussen and Nonnum, 2006). High-trophic level, fish-eating wildlife species can biomagnify contaminants such as PCBs and PBDEs, which have been shown to elicit exposure-associated neurological effects (Basu and Head, 2010; Basu, 2012, 2015). Although many studies show Arctic marine mammals can bioaccumulate substantial levels of MeHg and OHCs, little is known about their uptake into brain tissue. This makes it difficult to establish potential health effects.

Recent studies have emerged that characterize Hg levels in brain tissues of Arctic marine mammals. In one of the first studies on potential neurological effects of Hg, total Hg and MeHg levels were measured in the brain stem region of 82 polar bears collected by subsistence hunters in Greenland (Basu et al., 2009). Concentrations of total Hg were found to be <1 µg/g dw. In this same brain region, MeHg comprised 82.8% of the total Hg load. Similar total Hg levels were reported for the brain stem of Canadian polar bears (Krey et al., 2012), as well as the finding that nearly all Hg in these tissues occurred in the methylated form. In follow-up studies, Hg levels were characterized in ten brain regions (pituitary, occipital cortex, frontal cortex, cerebellum, brainstem, thalamus, hypothalamus, temporal cortex, hippocampus, basal ganglia) from polar bear, narwhal, ringed seal, pilot whale, and harbor porpoise (*Phocoena phocoena*) (Basu, pers. comm.). In all cases, mean total Hg levels were highest in the pituitary and lowest in the brain stem. A study by Krey et al. (2015) comparing total Hg levels in the cerebellum and frontal cortex found no differences for ringed seal, polar bear, and beluga. In the aforementioned study, mean MeHg levels as a percentage of total Hg ranged from 51% to 96%. Notably, MeHg in the pituitary was less than 50% of total Hg. In the cerebellum and frontal cortex of ringed seal and polar bear, MeHg level as a percentage of total Hg was over 90% but in beluga was only about 12% (Krey et al., 2012). These studies point to interesting differences across species. Brain Hg levels are consistently lowest in ringed seal and polar bear, and across all brain regions studied in these species. Brain Hg levels in pilot whale, harbor porpoise, and narwhal are much higher and range at upwards of 80 µg/g dw, levels associated with clinical neurotoxicity in mammals (Basu, 2012, 2015).

Much less is known about the uptake of OHCs into the brain tissues of Arctic marine mammals. Gebbink et al. (2008) measured a large suite of chlorinated and brominated contaminants and their by-products in brain tissue of East Greenland polar bears. They found that compared to liver, blood and adipose, the lowest OHC concentrations consistently occurred in the brain, often below method detection limits. The authors estimated that the OHC burden in the brain accounted for approximately 0.002% of the total amounts in the four body compartments studied. This suggests that the blood-brain barrier may effectively limit neural exposure to certain OHCs in polar bears.

In the first study of its kind, Greaves et al. (2012) investigated the comparative accumulation of perfluoroalkyl acids (PFAAs) in eight brain regions of polar bears collected in 2006 from East Greenland. On a wet-weight basis, blood–brain barrier transport of PFAAs occurred for all brain regions, although inner regions of the brain closer to incoming blood flow (pons/medulla, thalamus, and hypothalamus) contained consistently higher PFAA concentrations compared to outer brain regions (cerebellum, striatum, and frontal, occipital, and temporal cortices). For pons/medulla, thalamus, and hypothalamus, the most concentrated PFAAs were perfluorooctane sulfonate (PFOS) and perfluorotridecanoic acid. Longer-chain perfluorinated carboxylic acids (PFCAs; C_{10}–C_{15}) were significantly positively correlated with lipid content for all brain regions. The study demonstrated that both PFCAs and perfluoroalkyl sulfonates (PFSAs) cross the blood–brain barrier in polar bears and that wet-weight concentrations are brain region-specific. In a follow-up study on PFASs in several brain regions of polar bears from East Greenland collected in 2011–2012 (Pedersen et al., 2015, 2016), the most abundant PFAS was again PFOS and accounted for approximately 90% of ΣPFSAs. The highest concentrations of PFASs were measured in brainstem, cerebellum and hippocampus.

There have been a few new studies on Arctic marine mammals that have attempted to relate contaminant exposure with neurological effects using neurochemical biomarkers. *In vitro* studies on brain tissue extracted from polar bears (Krey et al., 2014) and ringed seals (Basu et al., 2006a) have shown Hg to inhibit binding of neurochemical receptors and activity of neurochemical enzymes of physiological importance. Extrapolating these *in vitro* findings to the whole organism has proved more challenging. For example, both Krey et al. (2014) and Basu et al. (2009) did not observe any Hg-associated changes in brain cholinergic receptors or enzymes despite *in vitro* evidence showing potential effects as well as previous *in vivo* results on mink (*Mustela vison*) (Basu et al., 2006b), eagles (*Haliaeetus leucocephalus*), and loons (*Gavia immer*) (Scheuhammer et al., 2007) pointing to exposure-related changes. In another study, a negative association was found between brain Hg levels and N-methyl-D-aspartate (NMDA) receptors in the brain stem of polar bears (Basu et al., 2009), as previously seen in *in vivo*- and naturally-exposed mink (Basu et al., 2007) and in wild eagles and loons (Scheuhammer et al., 2007). In a study of beluga using both biochemical and molecular approaches, Ostertag et al. (2014) found a number of Hg-associated changes in components of the GABAergic and glutamatergic systems. Fewer studies have examined OHC-associated effects on Arctic marine mammal neurochemistry. In a study of brain stem tissues of polar bears, several chlorinated and brominated organic chemicals were not observed to associate with neurochemical biomarkers (Basu et al., 2009). In a study of multiple polar bear brain regions sampled in East Greenland in 2011 and 2012, correlations were found between PFSA/PFCA concentrations in polar bears and the neurochemical biomarkers from cholinergic, glutaminergic, GABAergic and dopaminergic systems, for example monoamine oxidase and GABA-A receptors (Pedersen et al., 2015). These correlations may be an indication of early adverse changes in neuro-behavior and animal health. The authors recommended that further studies be performed to investigate the neurotoxic effects of PFASs (especially PFCAs) on Arctic mammals, including the polar bear. In a recent study, Pedersen et al. (2016) investigated whether PFAS could induce alterations in brain steroid concentrations in the same East Greenland polar bears. This study showed significant positive correlations across all brain regions between 17α-hydroxypregnenolone (OH-PRE) and all investigated PFASs; PFOS, ΣPFSAs, perfluoro-*n*-undecanoic acid (PFUnDA), perfluoro-*n*-dodecanoic acid (PFDoDA), perfluoro-*n*-tridecanoic acid (PFTrDA) and ΣPFCAs. Positive correlations were found between ΣPFCAs and several steroid hormones in the occipital lobe. Positive correlations between PFCAs and steroids were especially evident for pregnonelone, progesterone, OH-PRE, DHEA, androstenedione and testosterone. The results from the present study generally indicate that an increase in PFAS concentrations seems to coincide with an increase in steroid hormones in East Greenland polar bears. However, it is not possible to determine whether alterations in brain steroid concentrations arise from interference with *de novo* steroid synthesis or via disruption of peripheral steroidogenic tissues mainly in gonads and feedback mechanisms.

3.1.8 Bioenergetics

There has only been one new study reported on contaminants and effects on bioenergetics in Arctic mammals since 2010. Tartu et al. (2017a) reported on female polar bears from Svalbard and analysis for biomarkers for energy metabolism, and included abundance profiles for nine lipid-related genes, fatty acid synthesis and elongation indices in adipose tissue, and concentrations of lipid-related variables in plasma (cholesterol, high-density lipoprotein, triglycerides). The results showed that several OHCs affect lipid biosynthesis and catabolism in female polar bears. More pronounced effects were noted when combined with reduced sea-ice extent and thickness, suggesting that climate-driven sea-ice decline and OHCs have synergistic negative effects on polar bears. For polar bears, Jenssen et al. (2015) emphasized that one of the main effects of climate warming is limited access to seals – their main prey – due to loss of their sea-ice habitat, which can be viewed as a perturbation of the bears' bioenergetic homeostasis. This will result in prolonged fasting periods that are likely to result in emaciation and condition-related negative effects on survival and reproductive success. Prolonged fasting, and especially emaciation, will result in increased POP concentrations in polar bear tissues, with an increased likelihood of POP levels exceeding threshold levels for effects on health, and thus potentially affecting reproductive success and survival.

3.1.9 Clinical chemistry

No new studies appear to have been reported on the effect of OHC and Hg exposure on clinical chemistry in Arctic marine and freshwater mammals since the previous effects assessments (Letcher et al., 2010; Dietz et al., 2013a).

3.2 Marine and terrestrial birds

3.2.1 Vitamins and oxidative stress

At the time of the previous AMAP effects report (Letcher et al., 2010) there had been reports for a few species and populations of marine birds in the Arctic linking tissue contaminant burdens to circulating and tissue vitamin levels as well as oxidative stress. For example, for seabird eggs from Svalbard and the Russian Arctic, Miljeteig et al. (2009) reported positive correlations between retinol and contaminant levels in ivory gull (*Pagophila eburnea*) eggs, while concentrations of the antioxidant vitamin E were decreasing with increasing contaminant levels. Levels of vitamin A, retinyl palmitate and vitamin E were measured in plasma and liver of black-legged kittiwake (*Rissa tridactyla*) hatchlings from Kongsfjorden (Svalbard) and Runde, northern Norway, but no significant association between concentrations of OHCs (organohalogen compounds and BFRs) in egg yolk sacs and vitamin levels or morphological variables were found (Murvoll et al., 2006). Newly hatched common eider (*Somateria mollissima*) chicks from Kongsfjorden, showed significant positive relationships between egg yolk sac concentrations of PCBs, HCH and oxychlordane, and liver vitamin E levels (Murvoll et al., 2007).

Since 2009, there have been a few new published reports on contaminants in relation to vitamins in ivory gulls and northern fulmars (*Fulmarus glacialis*) from the Arctic. Using a meta-analysis approach, the variations in retinoids were investigated in northern fulmars that breed in three populations differentially exposed to organochlorines: Nunavut (Canadian Arctic), Svalbard (Norwegian Arctic) and the Faroe Islands (Verreault et al., 2013). Organochlorine levels were found to be substantially higher in the liver of birds breeding in the Faroe Islands than in birds breeding in Svalbard and Nunavut. Furthermore, organochlorine concentrations (PCBs; polychlorinated dibenzo-*p*-dioxins, PCDDs; polychlorinated dibenzofurans, PCDFs; HCB; *p,p'*-DDE; oxychlordane) in the liver were positively correlated with hepatic retinyl palmitate levels. These retinyl palmitate levels were significantly associated

Thick-billed murre breeding colony, Prince Leopold Island, Canada.

with decreasing plasma retinol levels as well as somewhat unchanged liver retinol levels. It was concluded that these three geographically-distant sites of breeding fulmars show that organochlorine exposure (mainly PCBs and dioxins/furans) may be associated with modulation of the thyroid and retinoid homeostasis. Effects of confounding environmental factors (such as temperature and nutritional status) on physiological variables could not be ruled out, and thus cause-effect linkages between retinoid system perturbation and organochlorine exposure were not shown.

In a follow up study by Miljeteig et al. (2009) on ivory gull eggs from Svalbard and the Russian Arctic, associations were reported between high levels of contaminants (OCPs, PCBs, BFRs, PFASs, Hg) and three response variables: eggshell thickness, retinol (vitamin A) and α-tocopherol (vitamin E) (Miljeteig et al., 2012). Negative associations were found between levels of OCPs, PCBs and BFRs and eggshell thickness and α-tocopherol, but not with retinol. There were no associations between PFASs and Hg and the three response variables. Furthermore, eggshell thickness was 7–17% thinner compared to archived ivory gull eggs (from before 1930). In general, thinning of more than 16–20% has been associated with a decline in bird populations, suggesting that contaminant-induced eggshell thinning may constitute a serious threat to ivory gull populations globally.

Braune et al. (2011) examined relationships between hepatic concentrations of OHCs and vitamin A concentrations in liver as well as retinol levels in blood plasma of northern fulmars at two breeding colonies in the Canadian High Arctic. Biomarker levels or responses did not differ significantly between males and females at either colony, nor were there any significant differences between colonies. No significant relationships were found between hepatic retinoid concentrations and any of the dioxin-like compounds or their toxic equivalents (TEQs) although significant positive correlations were found with plasma retinol. Helgason et al. (2010) investigated whether liver retinoid concentrations were explained by liver and blood levels of OHCs in free-ranging breeding northern fulmars from Bjørnøya in the Norwegian Arctic. Results showed no strong relationships between OHC concentrations and liver vitamin A levels in the breeding birds. It was suggested that the OHC levels were too low to affect liver levels of retinal and retinyl palmitate.

Since 2009 there have been a few new published reports on contaminants in relation to oxidative stress in birds from the Arctic. OHCs may have adverse effects on the health of birds, especially marine avian top predators that accumulate high OHC loads. Contaminants may influence the antioxidant enzyme activity (oxidative stress). Moreover, physical conditions and oxidative stress during development may reduce telomere lengths, one of the main mechanisms explaining cell senescence. Sletten et al. (2016) examined the potential effects of environmental contaminants on physiological biomarkers of health in white-tailed eagles (*Haliaeetus albicilla*) (samples collected in late June and early July in 2011 and 2012) in the Norwegian subarctic. OHCs with different physicochemical properties were related to superoxide dismutase enzyme (SOD) activity in blood plasma, and telomere length (measured in red blood cells) in individual 7- to 8-week old nestlings. Different organochlorines and PFASs were measured in blood plasma of nestlings, demonstrating higher concentrations of PFASs and notably PFOS as compared to legacy organochlorines. There were significant, negative relationships between OHC loadings and SOD activity suggesting that some legacy OHCs challenge the antioxidant capacity in nestlings of white-tailed eagles.

Bourgeon et al. (2012) reported on the contribution of legacy OHCs and PBDEs to individual variations in oxidative stress occurring in three breeding colonies of a top predator seabird, great skua (*Stercorarius skua*), distributed from temperate regions to the High Arctic: Shetland Islands, Iceland and Bjørnøya. Plasma concentrations of organochlorines in great skua from Bjørnøya were among the highest in North Atlantic seabirds. A latitudinal gradient in OHC levels was observed with all compounds being significantly higher in Bjørnøya than in Iceland and the Shetland Islands. Skuas breeding at the least contaminated site (Shetland Islands) experienced 50% higher oxidative stress compared to the two other colonies. However, the authors' results failed to identify consistent within-colony relationships between biomarkers of health. It was suggested that other ecological factors such as food availability were constraining the physiological indicators more than anthropogenic contaminants.

Wayland et al. (2010) investigated glaucous gulls (*Larus hyperboreus*) from the Canadian Arctic in terms of relationships between organochlorines, PCBs, Hg and Se, and measures of oxidative stress (glutathione [GSH] metabolism and lipid peroxidation). Contaminant levels were low and associations between contaminant exposure and oxidative stress were weak. Nevertheless, levels of thiols declined as Hg and organochlorine/PCB levels rose (glutathione peroxidase activity rose with increasing hepatic Se concentrations) and at one of the two study sites levels of lipid peroxidation were elevated with increasing levels of hepatic Hg. It was suggested that there was a possibility of a deleterious effect of exposure to contaminants on gull physiology even at low exposures.

3.2.2 Endocrinology

Exposure to endocrine-disrupting chemicals may alter the ability of Arctic animals to adapt to ongoing environmental change (Jenssen, 2006). In polar regions including the Arctic, where the productive summer season is short, proper endocrine regulation of the timing of breeding, molting, and migration is especially important. Since the last AMAP effects report there have been a few new studies on contaminants and endocrine effects in Arctic birds (Letcher et al., 2010).

Glucocorticoids are involved in a range of physiological processes including reproduction, behavior, adaptation to stress, and immunology (Wingfield and Sapolsky, 2003). More specifically, the release of the hypothalamic–pituitary–adrenal axis-mediated corticosterone (CORT) during stressful events triggers physiological and behavioral adjustments that redirect energy investment away from reproduction towards self-preservation and survival (Wingfield and Sapolsky, 2003). Release of the anterior pituitary hormone prolactin stimulates parental behavior, such as brood provisioning and egg incubation (Buntin, 1996), and has been shown to decrease during acute stress responses (Chastel et al., 2005). In male glaucous gulls, baseline prolactin levels and the rate of decrease in prolactin after a restraint protocol seem to vary negatively with plasma OHC concentrations (OCPs, PCBs, BFRs, and associated metabolic products), and were found to be significant for PBDEs only (Verreault et al., 2008).

In the post-2010 period, Nordstad et al. (2012) found ΣPCB concentrations to be positively associated with baseline plasma CORT concentrations during the pre-laying period of black-legged kittiwakes at Svalbard. Yet, the plasma CORT-PCB relationship in black-legged kittiwakes seems sensitive to a number of factors, including environmental conditions, sex and breeding status. Several studies have investigated CORT-PCB relationships in plasma of kittiwakes during different years, on both sexes and at different breeding periods (Nordstad et al., 2012; Tartu et al., 2014a, 2015a,b). The overall pattern is an increase in CORT with increasing ΣPCB concentrations, although the authors did not find a relationship between CORT and ΣOCPs. The underlying mechanism associating plasma CORT to plasma ΣPCB levels could result from the detoxification process loop. High levels of PCBs can induce the expression of cytochrome P450 enzymes, which can catalyze the transformation of steroids leading to the formation of, among others, glucocorticoids. A precursor of costicosterone for example would be 11-deoxycosticosterone. Thus, glucocorticoid levels can be increased via POP exposure. For example, the activation of receptors for adrenocorticotropic hormones were associated with PCB concentrations in exposed Arctic seabirds (Tartu et al., 2015a).

The association between CORT and ΣPCB is likely to be species-specific (Tartu et al., 2015b). The authors did not observe any effect of PCBs on baseline or stress-induced CORT levels in common eiders. However, they did see a lowered stress-induced release of plasma CORT in glaucous gulls with higher blood PCB levels, meaning that the use of CORT as a bioindicator may be constrained by ecological factors. This was in fact emphasized by Bourgeon et al. (2012) studying great skua in the High Arctic. They showed feather CORT levels to be highest in the least POP-contaminated colony, and inter-individual associations between feather CORT and plasma POP levels to be non-existant. Finally, Tartu et al. (2014a) studied the effects of unrestricted OHCs, i.e. PFASs, in adult chick-rearing black-legged kittiwakes. In contrast to what was found for restricted OHCs (Nordstad et al., 2012; Tartu et al., 2014a) baseline plasma CORT concentrations were negatively associated with some PFASs during the chick-rearing period. In contrast to PCBs (Nordstad et al., 2012), PFASs may disrupt the hypothalamic–pituitary–adrenal axis resulting in lower ability to secrete sufficient baseline concentrations of CORT. Nonetheless, the absence of a significant relationship between PFAS levels and stress-induced plasma CORT concentrations may indicate this association to reflect hormone displacement due to high protein affinity of PFASs.

Thyroid hormones regulate metabolic processes, thermogenesis, and the growth and differentiation of tissues, including the regulation of neuronal proliferation, cell migration, and differentiation of the developing animal (Zoeller et al., 2002). In birds, thyroid hormones are also central in regulating molting and replacement of feathers (Kuenzel, 2003; Leeson and Walsh, 2004). Northern fulmars in the Canadian High Arctic did not show any significant association between plasma total thyroxine (TT4) and hepatic organochlorine or PFAS concentrations (Braune et al., 2011). For total triiodothyronine (TT3), there was a significant negative association with hepatic dichlorodiphenyldichloroethane (DDD) only, largely supporting observations on northern fulmar from Bjørnøya in the Norwegian Arctic, for which no associations between plasma total and free T3 and T4 or hepatic organochlorine levels were found (Helgason et al., 2010). The lack of significant associations may be concentration dependent, as breeding adult glaucous gulls at Bjørnøya showed free and total plasma T4 levels (Verreault et al., 2004), as well as the T4:T3 ratio to be negatively impacted by organochlorines (Verreault et al., 2004, 2007). Many chemicals, including PCBs, alter the ratio of T4:T3 by direct action on the thyroid gland by interference with transport proteins, such as transthyretin or albumin, or by inhibition of the peripheral conversion of T4 to T3 by deiodinases and increase in glucuronidation and excretion (Rattner et al., 1984). Therefore, the ratio of T4:T3 is indeed a sensitive indicator of contaminant exposure (Peakall, 1992). However, the positively associated ratio of TT4:TT3 to hepatic concentrations of DDD in Canadian northern fulmar (Braune et al., 2011) contrasts with observed negative associations in Norwegian glaucous gull (Verreault et al., 2004, 2006). Another interesting biomarker is the TT4:FT4 and TT3:FT3 ratios as they were reported to be negatively impacted by organochlorines and PFASs (Nøst et al., 2012). Interestingly, the latter study found black-legged kittiwake and northern fulmar chicks at Svalbard to show positive associations between free and total plasma T4 and T3 and plasma PFASs and organochlorines, respectively. Although a positive association to hepatic PFCAs was also found in Canadian northern fulmar (Braune et al., 2011), similar to the above-mentioned CORT (Tartu et al., 2014a,b), this dynamic is often believed to be the result of a common affinity for protein binding, which is particularly high for T3 in birds (McNabb, 2007), rather than PFASs being disruptive to thyroid function which would be reflected by a negative relationship.

Glaucous gull at Bjørnøya is a well-studied species, and positive associations have been found between POPs, more-specifically methoxy-PBDEs, PCBs, chlordanes, HCHs, mirex, organochlorines, PBDEs and hexabromocyclododecane (HBCD) isomers, and levels of testosterone and 17β-estradiol in yolk, as well as between PBDEs, PCBs, DDTs and mirex, and 17β-estradiol (Verboven et al., 2010). Although plasma testosterone levels did not associate to blood levels of these compounds in this colony (Verreault et al., 2006), plasma progesterone levels were positively associated.

3.2.3 Reproduction and genotoxicity

Studying the effects of POPs and Hg on reproduction is particularly important in the suite of biological effects as negative impacts on reproduction can alter a population's reproductive rate in several ways. In Arctic birds, this can include the most common direct measures of reproduction including propensity to breed, breeding onset (i.e., timing of breeding), breeding productivity (i.e., clutch size, egg development), and breeding success (i.e., nest success, chicks fledged). POPs and Hg can also affect other facets of reproduction, such as sex bias of clutches and lack of return to the breeding colony in the following year that can have further effects.

Although the number of studies on Arctic avifauna is limited, there is a large body of work examining the impacts of POPs and Hg on avian reproduction which suggests that exposure to these contaminants can lead to aberrant reproductive behavior, reduced clutch size, increased rates of embryonic deformity and mortality, and reduced hatchability (Thompson, 1996; Wolfe et al., 1998; Scheuhammer et al., 2011, 2015).

Most studies of the biological effects of Hg and POPs on reproduction in Arctic birds have focused on a few species that are relatively easy to study during the breeding season. This includes species with high levels of POPs and Hg (e.g., glaucous gull) and species commonly found to have lower levels of these contaminants (e.g., common eider). There are species for which reported levels are again much higher than for those most commonly studied species (e.g., ivory gull), but biological effect studies on such species are difficult if not impossible in many regions due to their ecology (including limited access to breeding sites and low numbers).

In general, Arctic avian studies on POPs and Hg have focused mostly on marine birds. There are some data available for insectivorous passerines, but concentrations in those terrestrial avian species that have been studied are low compared to aquatic high-trophic level species (Scheuhammer et al., 2011).

Breeding propensity refers to whether an individual undertakes a breeding attempt in a given year. As many Arctic bird species are relatively long-lived an individual in poor condition may forego breeding, and so propensity can be indicative of an individual's ability to reach breeding condition (Goutte et al., 2011). Tartu et al. (2013) found that black-legged kittiwakes from Svalbard that had higher blood concentrations of total Hg were more likely to avoid breeding (and have abnormal reproductive hormonal responses). Recent work on Hudson Bay, Canada showed that the propensity for eider duck to breed was unrelated to blood Hg concentration in eider duck on arrival at the breeding colony (Provencher et al., 2017).

Onset of breeding refers to when an individual initiates breeding, which can differ by species, but in general signals that an individual has reproduced that year. Onset can be highly variable, and in Arctic birds where the breeding season is limited by weather and resources there is evidence that those who initiate breeding earlier are often more successful (Descamps et al., 2011). Tartu et al. (2013) found that Hg concentrations had no relationship with lay date in black-legged kittiwakes from Svalbard. Similarly, there was no relationship found between blood total Hg concentrations and lay date in common eider from northern Hudson Bay, Canada (Provencher et al., 2016).

Effects can also be determined from the development of eggs and chicks. In a laboratory experiment Braune et al. (2012) found that increased MeHg concentrations lead to decreased survival of eggs of both thick-billed murres (*Uria lomvia*) and Arctic terns (*Sterna paradisaea*). Other Arctic bird species that have been studied are Canada goose (*Branta canadensis*) and sandhill crane (*Grus canadensis*). Canada geese were found to have a low sensitivity to MeHg in eggs, with an LC50 of 1 µg/g MeHg or above (Heinz et al., 2009). Sandhill cranes were found to have a medium sensitivity, with an LC50 of 0.25–1 µg/g MeHg (Heinz et al., 2009). The two raptor species included in this study (American kestrel *Falco sparverius* and osprey *Pandion haliaetus*) showed a high sensitivity to MeHg, with an LC50 when injected of less than 0.25 µg/g MeHg (Heinz et al., 2009). This suggests that Arctic birds of prey may also show sensitivity in egg survival when exposed to high levels of Hg. Tartu et al. (2013) found no relationship between total Hg and clutch size or breeding success in black-legged kittiwakes from Svalbard.

Erikstad et al. (2011) found the body mass of male chicks to be negatively correlated with maternal levels of organochlorines, but found no relationship for female chicks. This study also showed that both condition and maternal organochlorine concentrations can influence the sex ratio in clutches. For females with low organochlorine levels, mothers with high body condition hatched more male chicks, while those with poor body condition hatched more female chicks. In females with high organochlorine levels, there was an overall skew in clutches towards male offspring, which was most apparent in birds with poor body condition. Provencher et al. (2016) found a significant positive relationship between Arctic-breeding female common eider arrival condition and blood Hg concentrations. This finding is counter to previous findings in eider and other bird species where higher Hg burdens are associated with poorer condition. This may be related to recent feeding bouts leading to an increase in condition, while long-term Hg accumulation leads to negative effects over the longer term (Provencher et al., 2016). Studies examining survival rates (resighting at the colony in subsequent years) and contaminants, show variable results. Previous work on king eiders (*Somateria spectabilis*) in north central Canada (Wayland et al., 2008) established a link between individual blood contaminant concentrations and survival, while other studies on the return of Arctic-breeding birds to colonies showed no link between Hg and POPs with return rates (Bustnes et al., 2006). Similarly, no relationship was found in eider between blood Hg concentrations and return rates (Provencher et al., 2017). These mixed results on the effects of POPs and Hg on return rates to colonies suggest survival is influenced by several variables.

Although indirect, recent studies by Blevin et al. (2014) have shown that higher POP levels can have a negative association with several components of integument coloration: saturation of eye-ring, gapes and tongue, suggesting that POPs could disrupt coloration of labile integuments in female kittiwakes. Many studies suggest that coloration is an indicator of condition, which suggests that POPs may also be altering breeding condition.

Work has shown that contaminants in association with other stressors may have cumulative effects on reproduction (Bustnes et al., 2006). Although similar experiments in common eiders in the Canadian Arctic have shown that Hg exposure had no significant effect on reproduction, parasites can influence breeding, suggesting that cumulative effects are likely to be more of a concern in species with higher exposure rates (Provencher et al., 2017).

A common theme throughout the studies reviewed above is that age and gender may also play a role in how contaminants affect reproduction (see Costantini et al., 2014). It is also important to note that due to fasting during incubation studies, female eider had higher levels of lipid-soluble organochlorines (Bustnes et al., 2010). As a result, reproduction in females may inherently increase the mobilization of some organochlorines, and increase the potential effects of these POPs in females during this period. Increased POP concentrations during reproduction are also of concern because they often coincide with poor body condition and weakened immune systems.

Since the last AMAP effects reports (Letcher et al., 2010; Dietz et al., 2013a), only one study has reported on genotoxicity in Arctic birds. Lipophilic POPs are released from the fat reserves of birds during fasting, causing increased blood concentrations. Thus, POPs represent a potential anthropogenic stressor during bird fasting periods. Fenstad et al. (2014) reported on PCB and DNA double strand-breaks in female common eiders during the eider incubation period in the High Arctic. In 2008 and 2009, nine POPs and DNA-FTM were sought in blood samples at day 5 of the incubation period, and then again in the same individuals at day 20. During the incubation period, eider body mass decreased by 21–24%, whereas POP levels increased by 148–639%. The DNA-FTM increased by 61–67% (being proportional to the increase in double strand-breaks). At day 5, but not day 20, DNA-FTM was positively correlated with most POPs sought. The increase in DNA-FTM was positively correlated with the decrease in body mass during incubation. The authors suggested that fasting stress (body mass loss) decreases DNA integrity and that stress caused by fasting on body mass loss appeared to override the additional stress caused by concurrent increase in the levels of the POPs sought in the eiders. The study concluded that blood levels of POPs in the Svalbard eiders were relatively low, and that additive and/or synergistic genotoxic effects of fasting stress and POP exposure may occur in populations with higher POP levels.

Haarr et al. (2018) measured DNA damage in lymphocytes from six Arctic seabird species from Kongsfjorden, Svalbard, using the comet assay. The species studied were common eider (n=12), black-legged kittiwake (n=15), black guillemot (*Cepphus grylle*, n=10), Arctic skua (*Stercorarius parasiticus*, n=10), glaucous gull (n=14), and great skua (n=4). Fresh lymphocytes were isolated from whole blood sampled during the incubation/early chick-rearing period. The sensitivity of lymphocytes from each individual to oxidative stress was assessed *in vitro* through exposure to hydrogen peroxide, and the ability to repair induced damage was assessed by allowing exposed lymphocytes to recover for one hour. Baseline DNA damage was low and not significantly different for lymphocytes of five species: average tail intensity was 1.7% (common eider), 1.7% (black-legged kittiwake), 0.4% (black guillemot), 3.9% (Arctic skua), and 1.5% (glaucous gull). Great skua had a significantly higher baseline than the other species at 8.6%, but only four individuals were sampled for this species. Contaminant concentrations in the different species were within the ranges found in previous studies and as expected due to feeding ecology and biomagnification: the lowest concentrations were found in common eider (ΣOHCs 3.4 ng/g ww), with higher concentrations in black guillemot (35 ng/g ww), kittiwake (33 ng/g ww) and Arctic skua (36 ng/g ww) and highest concentrations in top predators: glaucous gull (255 ng/g ww) and great skua (515 ng/g ww).

Haarr et al. (2018) found no significant relationship between DNA strand breaks and contaminant concentrations in lymphocytes for any of the species or POPs studied, which suggests that avian lymphocytes are robust cells, apparently unaffected by contaminant exposure. Another explanation may be that the contaminant concentrations in Kongsfjorden are too low for genotoxic effects to occur. Exposure to oxidative stress caused increased DNA damage in lymphocytes from all species. Average tail intensity was 39% (common eider), 47% (black-legged kittiwake), 26% (black guillemot), 47% (Arctic skua), 40% (glaucous gull) and 27% (great skua). The differences were not significant and there was no indication of reduced DNA damage after one hour of recovery in buffer to allow repair. On the contrary, kittiwake lymphocytes showed a significant increase in DNA damage one hour after exposure to H_2O_2. El-Bibany et al. (2014) showed that DNA damage could continue to increase for up to six hours after exposure to H_2O_2 and that complete recovery was not evident until 24 hours after exposure.

3.2.4 Immunology

There are two main components of immunity that are traditionally studied: cell-mediated immunity and humoral immunity. Cell-mediated immunity refers to immune responses that do not involve antibodies; this includes activation of phagocytes, lymphocytes and cytokines when exposed to an antigen. These processes usually involve cells or proteins that destroy or attack pathogens within the body. Humoral immunity is the antibody-mediated immune system, which consists of macromolecules that are found in extracellular fluids. POPs and Hg have both been shown to affect cell-mediated and humoral immunity functions (Corsini et al., 2014). However, there are few studies that have tested this directly in Arctic marine and terrestrial birds.

In an examination of cell-mediated immunity function effects, feeding experiments with captive adult and nestling American kestrels fed environmentally relevant levels of MeHg exhibited immunosuppression under both the low and high MeHg doses (Fallacara et al., 2011a,b). Three treatment groups of adults were given MeHg for 13 weeks: control (0), low (0.6 µg/g dw) and high (3.9 µg/g dw). Adult kestrels in the low and high exposure group had lower responses to phytohemagglutinin skin-swelling assays, and a depletion of T cell-dependent splenic lymphoid tissue (Fallacara et al., 2011a). The high exposure group also exhibited an increase in absolute heterophil counts, elevated heterophil to lymphocyte ratios, and higher total white blood cell counts in adults (Fallacara et al., 2011a). The authors suggested that their findings demonstrate that adult kestrels are more sensitive to immunotoxic effects of environmentally relevant levels of MeHg than reproductive effects.

American kestrel nestlings were also given environmentally relevant levels of dietary MeHg for up to 18 days post-hatch (Fallacara et al., 2011b): control (0), low (0.6 µg/g dw) and high (3.9 µg/g dw). Nestlings in the low and high dose groups also showed suppressed cell-mediated immunity to the phytohemagglutinin skin-swelling assay at day 11. Nestlings in the high dose group also showed lymphoid depletion and a lack of macrophage activity in the thymus. In the same study MeHg doses did not alter B cell-dependent histological variables in American kestrels (Fallacara et al., 2011b).

For humoral immunity function effects, more work has been done recently on humoral immunity function in relation to OHCs and Hg in Arctic marine birds, although the results show mixed findings on whether POPs and Hg affect humoral immunity function. In captive adult American kestrels the secondary response to the sheep red blood cell hemagglutination assay showed no difference between the MeHg exposure groups, although the low exposure group showed suppressed primary immune responses to the sheep red blood cell hemagglutination assay (Fallacara et al., 2011a). In captive nestling American kestrels MeHg did not have any detectable effect on antibody-mediated immune function when exposed to the sheep red blood cell hemagglutinin assay (Fallacara et al., 2011b).

Work has also been undertaken in natural settings. A study examining immunity metrics in both sexes of great skua nesting in the Shetland Islands, Iceland and Bjørnøya found no significant relationship between immunoglobulin Y (IgY) and organochlorines or PBDEs (Bourgeon et al., 2012). Provencher et al. (2017) also found no relationship between levels of IgY and blood Hg concentration in common eider in northern Canada. Similarly, no relationship was found between concentrations of POPs – HCB; oxychlordane; *p,p*'-DDE; CB28, CB52, CB99, CB101, CB118, CB138, CB153, CB170 and CB180 (ΣPCB_9); and BDE47, BDE99, BDE100, BDE153 and BDE154 ($\Sigma PBDE_5$) – and levels of IgY or the primary response of IgY to tetanus toxoid in black-legged kittiwakes from Hornøya, Norway (Sagerup et al., 2014). Although, in the same study Sagerup et al. (2014) did find significant correlations between the IgY response to tetanus toxoid and several POPs for Atlantic puffins (*Fratercula arctica*). The relationships had variable directions and the explanatory values were quite low. Male puffins had a significant negative correlation between circulating levels of IgY and CB170 and CB180, while IgY levels in females were positively correlated with HCB, oxychlordane, CB28, CB99, CB170 and CB180 (Sagerup et al., 2014). The authors suggested that the opposing correlation directions for males and females may indicate gender-specific differences in immune responses. Of particular note, is that studies of POPs and Hg in Arctic birds often show large interannual differences (Bustnes et al., 2004), as well as significant differences between the sexes (Sagerup et al., 2014). This demonstrates that although POPs and Hg may affect some immune responses, any given response may be heterogeneous in nature and vary with season, time and sex.

3.2.5 Skeletal system

Since the last AMAP effects reports (Letcher et al., 2010; Dietz et al., 2013a), no new studies have reported on the effects of OHC and Hg exposure on the skeletal system in Arctic marine and freshwater birds.

3.2.6 Histopathology

Previous AMAP reports could not summarize histological studies on Arctic bird species because none existed at the time (Letcher et al., 2010; Dietz et al., 2013a). In the post-2010 period there have been two new histopathological studies on glaucous gull, one focused on thyroid glands (Sonne et al., 2010b) and one on multiple organs (Sonne et al., 2013c). In the latter, the histology of liver, kidney and thyroid tissues was studied in 10 gulls from Svalbard in 2011, showing hepatic PCB concentrations ranging from 150 to 2820 ng/g ww. All specimens showed histological changes, including kidney lesions (70%) and thyroid gland changes (50%). POP concentrations were non-significantly higher in individuals with various lesions and histopathological changes, which were all similar to those found in laboratory studies and wild polar bears. In their study on thyroid gland lesions, Sonne et al. (2010b) examined 10 adult gulls from Bjørnøya in the Norwegian Arctic during their incubation period, and reported blood plasma PCB concentrations ranging from 186 to 1027 ng/g ww. High densities of small follicles and follicular epithelial cell proliferations (70%) were found, as well as focal thyroiditis and nodular hyperplasia (20%). Such changes may affect hypothalamic-pituitary-thyroid axis functioning, thus possibly affecting energy metabolism in adults, and cell differentiation, growth and development of hatched chicks. The authors suggested that POP exposure may be a co-factor in the development of organ alterations in glaucous gulls, but highlighted the need for further investigation into the natural variance of tissue pathology.

3.2.7 Neurology and behavior

Since 2009, there have been a few new reports on contaminants in relation to neurology and behavior in Arctic birds. Life-history theory predicts that long-lived organisms should reduce parental effort under inclement environmental conditions in order to favor long-term survival. Seabirds are long-lived and feed higher in the marine food web and thus are often exposed to higher levels of environmental contaminants such as Hg, which can cause disrupted parental behavior in birds. Parental behavior is governed by two key hormones in birds: corticosterone (CORT, a glucocorticoid hormone) and prolactin (a pituitary hormone involved in parental care). Any disruption of these hormones may alter the ability of an individual to adjust parental behavior to environmental conditions. Tartu et al. (2016) recently reported on the relationships between blood Hg concentrations, plasma prolactin and reproductive performance in Arctic black-legged kittiwakes. There was a negative relationship between plasma initial prolactin and blood Hg concentrations in males. Hg concentration was negatively related to breeding success in chick-rearing males. To study the effect of a chronic increase in CORT levels on the Hg-prolactin relationship, the stress of the birds was increased experimentally with CORT pellet implantation. Adding CORT did not steepen the Hg-prolactin relationship. Hatching success was significantly lower in CORT-implanted males than in controls, and breeding success was not reduced in CORT-implanted male kittiwakes with high levels of blood Hg. It was suggested that Hg may impair reproductive performance through a disruption in prolactin secretion.

Hargreaves et al. (2010) addressed a hypothesis to explain the global decline in shorebirds, which is of particular concern in the Arctic. Elemental contaminants were studied in three biparental shorebird species nesting in Nunavut, Canada: ruddy turnstones (*Arenaria interpres*), black-bellied plovers (*Pluvialis squatarola*) and semipalmated plovers (*Charadrius semipalmatus*). Blood, feathers and eggs were analyzed for Hg as well as for arsenic, beryllium, cadmium, cobalt, chromium, copper, iron, manganese, molybdenum, nickel, lead, antimony, selenium, thallium, vanadium and zinc. Non-essential elements were found at lower concentrations than essential elements, with the exception of Hg. Maximum Hg levels in blood approached those associated with toxicological effects in other bird species, but other elements were well below known toxicological thresholds. Reproductive success was negatively correlated with paternal Hg and maternal lead, although these effects were generally weak and varied among tissues. Element levels were positively correlated within pairs for blood-Hg (turnstones) and feather-nickel and chromium (semipalmated plovers); concentrations in eggs and maternal blood were never correlated. The authors concluded that there was little evidence supporting that exposure to the majority of these elements is leading to declines in these species.

3.2.8 Bioenergetics

Most Arctic work on bioenergetics was carried out prior to the last OHC assessment by Letcher et al. (2010). In this review based on previous findings it was suggested that circulating thyroid hormone homeostasis may be perturbed in highly OHC-exposed glaucous gulls (males) breeding on Bjørnøya (Verreault et al., 2004, 2007). In males and females combined, negative associations were found between basal metabolic rate (BMR) and plasma concentrations of ΣPCBs, ΣDDTs and particularly ΣCHLs, thus suggesting potentially altered functions of the basal metabolism. However, levels of free and total T4 and T3 were not significantly associated with the variation in BMR or concentrations of any of the OHCs determined. A companion study by Verboven et al. (2009) using the same colonies on Bjørnøya showed that the nest temperature of glaucous gull males and females was negatively correlated with the concentrations of certain OHCs, BFRs (PBDEs and HBCD) and OH-PCB metabolites in plasma of the incubating parent. To test the parental control of incubation conditions in relation to OHC exposure, the energetic cost of incubation was augmented by artificially increasing clutch size from two to four eggs using dummy eggs. Clutch enlargement in glaucous gulls was followed by a decrease in nest temperature, although this decrease was not associated with plasma OHC concentrations.

3.2.9 Clinical chemistry

Analyzing blood plasma clinical-chemical parameters (BCCPs) gives a holistic evaluation of the biochemical, metabolic, and endocrine status of the vertebrate organism. BCCPs have been proven to serve as a minimally-invasive strategy to quantify adverse health effects of OHCs in free-ranging birds (Sonne et al., 2010b, 2012b, 2013d). BCCPs are usually used as biomarkers of health and function of different organ systems in mammals and birds (Sonne, 2010; Sonne et al., 2012b). Other factors such as feeding behavior and food items, time since last feeding, infectious diseases, dehydration and age are other factors that influence BCCPs (Thrall et al., 2004; Candido et al., 2011; Harr, 2012; Sonne et al., 2012b; López et al., 2015; Peng et al., 2015).

Since the last AMAP reports (Letcher et al., 2010; Dietz et al., 2013a), three new studies on BCCPs in Arctic avian wildlife, including top avian predators such as great skua, northern goshawk (*Accipiter gentilis*), white-tailed eagle and golden eagle (*Aquila chrysaetos*) have focused on OHCs in the marine environment of the North Atlantic (Sonne et al., 2010c, 2012b, 2013d). Concentrations of most BCCPs were positively or negatively associated with those for OHCs. While renal and liver functioning may be affected by different compounds, all three studies showed consistently increased alanine transferase, albumin and total protein as a result of OHC exposure, which was shown to explain 10.6% of the total observed BCCP variation. In one study, Sonne et al. (2013d) compared BCCPs in birds from three geographically distinct North Atlantic great skua colonies. Birds from these sites bioaccumulate different OHC concentrations and this made it possible to compare great skua BCCP regulation under different exposure scenarios. PCBs, DDTs, chlordanes, HCB, HCHs, mirex and PBDEs, and 19 BCCPs were sought in 114 adult great skuas sampled during summer 2009 in North Atlantic colonies at Bjørnøya, Iceland and the Shetland Islands. Specimens from Bjørnøya had the highest blood plasma concentrations of all contaminant groups followed by Iceland and the Shetland Island birds, respectively. Most of the 19 BCCPs followed the pattern of colony differences found for the OHCs with Bjørnøya having the greatest contaminant concentrations of BCCPs. However, seven BCCPs, the three liver enzymes (alkaline phosphatase, alanine transaminase, gamma glutamyl transferase), as well as bile acids, cholesterol, sodium and potassium concentrations did not differ among colonies. Analyses of combined colony data showed that the blood plasma concentration of alanine transaminase and gamma glutamyl transferase increased with increasing concentrations of PBDEs and HCHs, HCB and chlordanes, respectively. Based on these results the authors suggested that liver and renal functions could be negatively affected by different OHCs. But it is not clear whether the colony-specific BCCP concentrations and their relationship to OHC exposure reflect health effects that could have an overall impact on the population via reduced survival and reproduction.

3.3 Marine and freshwater fish

Fish can accumulate high concentrations of OHCs and Hg and are an important vector of these contaminants to humans and wildlife. Compared to mammals and birds, little is known about contaminant impacts directly on fish, including the large confounding factor of climate change / Arctic warming. Since the last AMAP effects reports on Hg and OHCs (Letcher et al., 2010; Dietz et al., 2013a), for Arctic marine and freshwater

Warming conditions may affect Hg bioaccumulation in cold-adapted freshwater species like Arctic char.

fish there have been no new studies reported on the effects of contaminant exposure on endocrinology, immunology, neurology and behavior, or clinical chemistry. However, there have been some recent reports on Arctic fish effects on bioenergetics, vitamins and oxidative stress, reproduction, genotoxicity, the skeletal system and histology. The previous sections on birds and mammals were sub-divided based on types of effects. Given the sparse amount of recent data on fish, this section is species and/or Arctic region based. The focus is on biological effects of PCBs and Hg on Arctic char (*Salvelinus alpinus*) from lakes of Bjørnøya (Svalbard, Norway) and Cornwallis Island (Nunavut, Canada), respectively. The section concludes with (sparse) information on Greenland shark (*Somniosus microcepalus*), sculpins, and other fish taxa.

3.3.1 Arctic char

Arctic char are widely distributed in the Arctic, and can be an important food source for indigenous peoples. In Canada, the federally-funded Northern Contaminants Program has supported extensive efforts to measure contaminants in Arctic char. Anadromous (sea-run) Arctic char are relatively low in contaminants and are promoted as a nutritious food source by public health authorities. In contrast, landlocked Arctic char (restricted to lakes and connecting streams) are relatively high in contaminants, especially Hg (Swanson et al., 2011). In an overview of the peer-reviewed literature, Drevnick (2012) reported that 30% (12 of 40) of landlocked populations sampled in northern Canada and Greenland exceed Hg concentrations known to cause toxicity (in fish) and thus are at risk from effects of Hg. For any given population, Hg concentrations increase with age and size, and in some lakes a large increase in Hg concentrations occurs with a switch, in the biggest char, from invertivory to piscivory. Among Arctic lakes, concentrations of Hg in Arctic char show a positive correlation with watershed-to-lake area ratio (Gantner et al., 2010a), indicating that the relative size of the 'collection basin' for wet and dry atmospheric Hg deposition determines the level of contamination. The Northern Contaminants Program uses annual sampling of landlocked Arctic char from lakes on Cornwallis Island and Ellesmere Island for tracking temporal changes in inputs of atmospheric contaminants. As expected, concentrations of legacy OHCs are declining but replacements (e.g., bis(2,4,6-tribromophenoxy) ethane) for banned flame retardant chemicals are increasing (Muir et al., 2014). Concentrations of Hg in landlocked Arctic char show a significant downward trend since 2005. Interestingly though, peaks are evident throughout the record in association with warm summer temperatures, which is also seen in a lake in southwest Greenland (Rigét et al., 2010), and Pacific air masses. Mercury has an extremely complex biogeochemical cycle with many processes affected by climate and it appears that temporal trends in Hg concentration in char may be recording changes in inputs of Hg (to lakes) and in climate.

3.3.1.1 Biological effects of Hg and other metals on Arctic char from the Canadian Arctic

From a climate change and bioenergetics perspective for fish, Stern et al. (2012) summarized the literature on how climate change influences Arctic Hg levels. This included the finding that long-term warming of freshwaters is likely to alter fish growth rates (Reist et al., 2006a) and hence affect their bioaccumulation, resulting in lower Hg concentrations in fish with higher growth rates (Simoneau et al., 2005). Also, that cold-adapted species such as Arctic char and lake trout grow less efficiently in warmer waters (Reist et al., 2006a), which could result in higher Hg concentrations. Temperature-induced metabolic stress in fish may also enhance Hg bioaccumulation (Reist et al., 2006b). A multi-year study of Arctic char in a High Arctic lake revealed that fish were under greater metabolic stress and had severe glycogen depletion near the end of an abnormally warm summer compared to two colder years (Reist et al., 2006b). However, Stern et al. (2012) concluded that further research is required to determine the effect of temperature stress on Hg bioaccumulation in cold-adapted fish species.

In order to determine the potential toxic effects of Hg, landlocked Arctic char (n=114) were sampled in 2011 and 2012 from four lakes in the Canadian High Arctic that span a gradient of Hg contamination (Drevnick, 2012, 2013; Barst et al., 2016). Individual fish from these lakes (Small, 9-Mile, North, Amituk), located on Cornwallis Island, were collected in conjunction with routine sampling for contaminant monitoring (Muir et al., 2012, 2013). Total Hg and stable isotopes ($\delta^{15/14}N$, $\delta^{13/12}C$) were measured in muscle. Bulk liver tissues were analyzed for total Hg, Hg speciation, and Se. To investigate how Hg and Se were distributed at the subcellular level, char liver tissues from Small Lake (low Hg) and Amituk Lake (high Hg) were subject to a trace element partitioning procedure based on differential centrifugation. This yielded six operationally defined fractions which were then analyzed for total Hg and Se. Fractions were assigned to one of two groups, including a potentially sensitive compartment (mitochondria + heat-denatured proteins including enzymes + microsomes and lysosomes) and a detoxified compartment (peptides and heat-stable proteins including metallothionein + granule-like concretions) (Wallace et al., 2003; Giguère et al., 2006; Rosabal et al., 2012, 2015). Liver tissues were preserved for histological analysis. Reproductive status was assessed using the gonadosomatic index and (for females only) egg quality (size) and quantity. General (body) condition was determined according to relative weight (LeCren, 1951), with the denominator (w') determined from a regression of length versus weight with data from all four lakes for char sampled (as part of routine sampling) from 2009 to 2014.

Mean total Hg concentrations in muscle tissue were greatest in char from Amituk Lake, intermediate in char from North Lake and 9-Mile Lake, and lowest in char from Small Lake. $\delta^{15/14}N$ values followed a similar trend. Total Hg concentrations in liver were two to five times greater than in muscle, with a maximum of 6.5 µg/g ww for an individual from Amituk Lake. For all livers, the fraction of MeHg ranged from 51% to 90% of total Hg, and increased exponentially with $\delta^{15/14}N$ values, which is consistent with studies demonstrating that food web structure plays an important role in determining the level of Hg contamination in landlocked Arctic char (Muir et al., 2005; Gantner et al., 2010a,b; Lescord et al., 2015).

In the same study, Se concentrations in the liver samples ranged from 0.9 to 6.5 µg/g ww. Concentrations of hepatic Se were significantly higher in char from Amituk Lake than in char from the remaining study lakes ($p<0.0001$) and were positively correlated with total Hg. The molar concentration of Se was greater than total Hg (mean molar ratio Hg:Se = 0.15) in all char livers, although this value increased with increasing Hg concentration to a maximum of 0.7 in a fish from Amituk Lake. Hg concentrations were significantly higher in isolated fractions of Amituk char than in char from Small Lake, which was consistent with the results of bulk liver analysis. Despite the higher Hg concentrations in subcellular fractions prepared from Amituk livers, Hg was distributed similarly among char liver fractions from both lakes. The sensitive compartments contributed 73% and 61% of the contributions of total Hg in Small and Amituk livers. This suggests that at low (Small Lake) and high (Amituk Lake) concentrations, Hg is not effectively detoxified in the livers of these fish. The remaining Hg was found in the detoxified compartments, which comprised 10% and 19% of the total Hg contributions of Small and Amituk char, respectively. Almost all of the Hg in the detoxified compartments was associated with the heat-stable protein fraction, presumably containing metallothionein, whereas very little was found in the granule-like fraction. In the potentially sensitive compartments, Hg accumulated mainly in the mitochondria and heat-denatured protein fraction containing enzymes. Selenium was distributed similarly among fractions, suggesting an interaction between the two elements.

Histological investigation revealed hepatic fibrosis in the perisinusoidal region, predominately in the livers of Amituk Lake char. A significantly greater number of individuals from Amituk Lake (83%) exhibited this abnormality than individuals from the other study lakes. Evidence of fibrosis was also found in individuals from 9-Mile (29%) and North Lake (27%). None of the livers from Small Lake char presented this abnormality. Arctic char are commonly hosts to cestode parasites, which have been shown to increase in number in piscivorous char (Frandsen et al., 1989). These parasites may cross the intestinal wall, and enter the liver and other organs, resulting in fibrosis and visceral adhesions (Frandsen et al., 1989; Hammar, 2000). Although Hg may have resulted in the observed fibrosis the role of parasites cannot be excluded as these may be particularly abundant in the char sampled from Amituk Lake. Future work should aim to decouple the effects of parasites and non-essential metals such as Hg.

For both females and males, gonad size (i.e., gonadosomatic index) was a function of whether gonads were developed and body size. The four study lakes are very unproductive, meaning that Arctic char have restricted energy budgets. Since energy is allocated first for the maintenance of basic biological functions, then for growth and reproduction, char do not spawn every year. Even after sampling for two consecutive summers, only a few individuals were sexually mature. There was no relationship between whether ovaries or testes were developed (i.e., maturation index) and total Hg. In terms of body size, as char grow bigger, they have proportionally bigger gonads, but also more Hg. If the 'effect' of body size on gonadosomatic index is removed (by ANCOVA), the results indicate that total Hg is not related to gonadosomatic index. For females with developed ovaries, egg number and size (diameter) increased with body size. Correcting for body size, by calculating relative fecundity (number of eggs per 100 g of body weight) and relative egg diameter (egg diameter / fork length), yields data that show possible effects of Hg. Total Hg concentration in muscle is negatively related to relative fecundity and positively related to relative egg diameter. Venne and Magnan (1989) predicted that, for a female char with a limited energy budget, "the first response to an additional stress would involve a decrease of the number of eggs and an increase of their diameter without affecting the gonadosomatic index." Higher probability of larval survival (owing to an increase in egg size) is balanced by a reduction in the number of eggs. The results thus indicate that Hg may be exerting effects on reproduction indirectly, that is, energy used by the liver for tissue repair (see above) may result in females altering their egg production.

Condition factor of char decreased with age, probably due to the energy requirements of maintenance, growth, and reproduction, but at large sizes char switched to piscivory, causing increases in growth (size at age) and condition (plumpness). The latter demonstrates the benefits of cannibalism (only Arctic char are present in these lakes). The increase in condition factor with piscivory was not observed in char with the highest total Hg concentrations (from Amituk Lake), which suggests that benefits of piscivory may be counteracted by the added stress of Hg in these fish.

In addition to Hg and OHCs, other metals have also been monitored in Arctic char in these lakes for two decades. In the early years of the study there was a predictable relationship (see Köck et al., 1996) between climate and metal concentrations in liver and kidney. During the El Niño year of 1998, Köck and co-workers (see Wrona et al., 2005: their box 8.12) documented high water temperatures, elevated cadmium and zinc levels in char livers, and induction of detoxification responses (glutathione system) that depleted energy (glycogen) reserves and vitamin C. Landlocked Arctic char in the High Arctic live in challenging environments, and contaminants (OHCs, Hg, and other metals) and climate change introduce added stress.

3.3.1.2 Biological effects of PCBs on Arctic char from Bjørnøya lakes

Lake Ellasjøen and other (reference) lakes on Bjørnøya, the southernmost island of the Svalbard archipelago in the Barents Sea, have been used to document the biotransport of OHCs from the marine environment to land (which includes lakes that Arctic char inhabit), via seabirds. Bjørnøya is a site of global significance for bird conservation (BirdLife International, 2018) with nesting colonies of threatened bird species that number in the tens of thousands. In the pre-2010 period, Evenset et al. (2004, 2007) used nitrogen isotopes ($\delta^{15}N$) to trace the input of seabird guano into Ellasjøen and determined that guano accounts for the majority (>80%) of OHC inputs to the lake. Wiseman et al. (2011) and Neerland (2016), have studied the impacts of this novel contaminant pathway to lakes on the health of fish. Wiseman et al. (2011) examined the expression of selected protein markers of exposure and effects between landlocked Arctic char from highly-contaminated Lake Ellasjøen and reference Lake Øyangen at Bjørnøya. Fieldwork was conducted in 2002 and samples were collected from a limited study group of three immature male char from each lake. The mean concentration of ΣPCBs in liver of char in Ellasjøen (281 ng/g ww) was about 25-times greater than in Øyangen (11.3 ng/g ww). Likewise, liver expression of cytochrome P4501A (CYP1A), a protein induced in phase-I metabolism of ligands of the aryl hydrocarbon receptor (including PCBs), was about 50-times greater in char from Ellasjøen than from Øyangen. CYP1A is considered a marker of exposure (gene/protein expression), and the molecular response (detoxification or activation of PCBs) and biological consequences are unknown (for these char). Expression of heat-shock protein 70 (hsp70) (brain, but not liver) and the glucocorticoid receptor (liver) were also greater in char from Ellasjøen than from Øyangen. No difference was reported in (body) condition of char between the two lakes.

Neerland (2016) recently reported on DNA lesions, measured as DNA double strand-breaks in landlocked Arctic char at Bjørnøya (n=39 individuals, 18 from exposed Lake Ellasjøen and 21 from control Lake Laksvatn). Concentrations of ΣOCs were about 40-times greater in char from Ellasjøen compared to char from Laksvatn, and char from Lake Ellasjøen had a significantly higher frequency of DNA double strand-breaks compared to char from Lake Laksvatn. Furthermore, the difference was significantly positively correlated with the levels of organochlorines in the fish.

3.3.2 Greenland shark

High levels of OHCs and Hg have been detected in Greenland shark from the Canadian Arctic (Baffin Island) and Iceland (Fisk et al., 2002; Strid et al., 2007). Little is known about Greenland shark ecology but it has been suggested that they could reach 100 years of age (Hansen, 1963) and that their diet consists of everything from invertebrates to fish and marine mammals (Compagno, 1984; Fisk et al., 2002; Yano et al., 2007; Lucas and Natanson, 2010). Their migratory range appears to be relatively limited, as reported for the Greenland sharks tagged at Svalbard that appear to remain in the waters around Svalbard (Fisk et al., 2002). A recent study on Greenland shark sampled outside Svalbard indicates that marine mammals are a common part of their diet (Leclerc et al., 2012). Both their great age and their consumption of high trophic level species can explain the high levels of OHCs and Hg documented in the sharks.


Nick Caloyianis

Greenland sharks can attain high concentrations of OHCs and Hg due to their great age and their consumption of high trophic level species.

Female and male Greenland sharks (n=43) were caught by longlines in the Kongsfjorden of Svalbard in 2008 and 2009 (Molde et al., 2013). Plasma samples were collected for contaminant and vitamin analysis to examine the potential impact of OHCs on vitamin A (retinol) and its precursor retinyl palmitate, and vitamin E (α-tocopherol) homeostasis. DDTs were the dominant contaminant group in the plasma (ΣDDTs: 8069±8793 ng/g lw) followed by PCBs (ΣPCBs: 5766±3716 ng/g lw) and chlordanes (ΣCHLs: 1551±1152 ng/g lw). PBDEs were detected only in a few individuals. Multivariate correlation analyses (principal component analysis) indicate negative associations between POPs and plasma levels of vitamin A and retinyl palmitate, while a positive association was found between POPs and plasma levels of vitamin E. Follow-up analyses of the vitamin A model using multivariate regression analysis indicate that the strongest negative associations between OHCs and vitamin A were between vitamin A and the dioxin-like PCBs, CB118 and CB156/CB171, as well as the non-dioxin-like PCBs, CB99 and CB128. In these models, covariates such as lipid percentage, hepatosomatic index and the animal's condition factor were taken into account. As for vitamin A, PCBs, as well as chlordanes, were also the dominating compounds found to correlate with retinyl palmitate, while PCBs exclusively correlated with vitamin E. Vitamin A, retinol palmitate and vitamin E were not associated with gut content mass. Although the reported associations do not necessarily reflect a cause-effect relationship, the high levels of OHCs and the relationships with vitamin levels are of concern due to the vital role of vitamins in development, embryogenesis, reproduction and immune function.

The Canadian Arctic Contaminants Assessment Report on Hg in Canada's north (Chetelat and Braune, 2012) identified Greenland shark as the single marine fish species (in Canada's Arctic) with Hg concentrations in excess of values known from laboratory and field studies to cause "changes in biochemical processes, damage to cells and tissues, and reduced reproduction in fish" (Sandheinrich and Wiener, 2011). As such, the authors stated that studies are warranted to investigate toxic effects of Hg on Greenland sharks and other large predatory marine fishes. No studies have been conducted to date.

3.3.3 Sculpin

Generally high site fidelity of sculpin make this superfamily of fishes (Cottoidea) an attractive indicator of ecosystem health, especially in relation to point sources of pollution. Kuzyk et al. (2005) used shorthorn sculpin (*Myoxocephalus scorpius*) to understand exposure to and ecological risks in benthic fishes from PCBs released from a military installation to Saglek Bay, Labrador, Canada. Thirty-five sculpin were collected from four zones of contamination and a reference site and analyzed for PCBs and biological endpoints, including condition factor, lipid content, hepatosomatic index, and ethoxyresorufin-*O*-deethylase (EROD) activity. ΣPCB concentrations in whole carcasses (excluding liver) were in the range 5.1–6920 ng/g ww. EROD activity, a measure of CYP1A induction, was positively correlated to ΣPCB concentration, and was 25-times greater in the most contaminated sculpin compared to those from the reference site. Other biological endpoints were not related to ΣPCB concentrations or EROD activity.

Sculpin have also begun to be included in Arctic assessments of environmental contaminant effects on organ histology. A single study has investigated common sculpins (n=33) collected at five sites of different distances from the former lead-zinc mine in Maarmorilik, West Greenland (Sonne et al., 2014c). Eleven different elements were sought together with the histopathological investigations. Chronic lesions were observed in liver and gill tissue of common sculpins and individuals with hepatic cell infiltrates had the highest concentrations of most elements. For 20% of the sculpins, Hg concentrations were above the LOED (lowest observed effect dose) for toxic thresholds on reproduction and subclinical endpoints. The frequency of liver necrosis and vacuolated hepatocytes increased with increasing Hg concentrations and were significantly highest at the three most contaminated sites. Similar relationships were found for gill lesions. The authors concluded that lesions in sculpin liver and gill tissues were likely to be affected by exposure to toxic elements. Therefore, sculpins may be suitable monitoring species in future Arctic studies of mining activity and long-range transport of Hg.

3.3.4 Other fish taxa

Duffy et al. (1999) reported no relationship between Hg exposure and expression heat shock proteins (Hsp60, Hsp70) in gills and skeletal muscle for northern pike (*Esox lucius*), burbot (*Lota lota*), whitefish (*Coregonus nelsoni*), grayling (*Thymallus arcticus*), and sheefish (*Stenodus lencichthys*) from the western North American Arctic including in Alaska. Marine fish species (Greenland halibut *Reinhardtius hippoglossoides*, daubed shanny *Leptoclinus maculatus*, sea tadpole *Careproctus reinhardti*, Atlantic spiny lumpsucker *Eumicrotremus spinosus*, polar cod *Boreogadus saida*, Arctic rockling *Gaidropsarus argentatus*, doubleline eelpout *Lycodes eudipleurostictus*, black seasnail *Paraliparis bathybius*, polar sculpin *Cottunculus microps*, Atlantic cod *Gadus morhua* and capelin *Mallotus villosus*) collected from reference locations (Faroe Islands, Jan Mayen, Svalbard and the Barents Sea) had concentrations of DNA adducts below or close to the method detection limit (Aas et al., 2003). A similar result was reported by Hylland et al. (2017) for dab (*Limanda limanda*) and haddock (*Melanogrammus aeglefinus*) sampled off Iceland. The latter study also included DNA strand breaks, which were lower in dab collected in Iceland than anywhere in the North Sea. Studies with two of the species (dab, Atlantic cod) in the North Sea or under laboratory conditions have however shown that fish residing in areas with high oil production will accumulate increased concentrations of adducts and/or higher levels of DNA strand breaks (Hylland et al., 2006, 2017; Balk et al., 2011; Holth and Tollefsen, 2012).

4. Challenges and new approaches to assess biological effects

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4.1 Contaminant mixtures and multiple stressors

Understanding and predicting the biological effects of complex contaminant mixtures within a multi-stressor framework is one of the great challenges of Arctic ecotoxicology. There are obstacles related to different experimental, modelling and predictive environmental risk assessment approaches. Ongoing environmental risk assessment guidelines and manual developments in Europe aim to incorporate the combined effects of chemical contaminants, the use of different experimental approaches for providing combined effect data, the involvement of biomarkers to characterize modes and mechanisms of action and toxicity pathways, and efforts to identify relevant risk scenarios related to combined effects (Beyer et al., 2014).

For more than a decade, political and scientific communities have requested information that ties together the cumulative effects of multiple stressors (e.g., immune- and endocrine disrupting pollution, infectious disease, climate change, etc.). Since the last AMAP effects report (Letcher et al., 2010), several studies have been published on this, and various hypotheses have been put forward. Polar bears (*Ursus maritimus*) have recently received a considerable amount of focus as a wildlife species impacted by climate change due to the projections of sea-ice loss, and decline in access to their main prey, the ringed seal (*Pusa hispida*) (Durner et al., 2009; Molnár et al., 2011; Stirling and Derocher, 2012). As a long-lived k-selected top predator species (i.e., large body size, long-living, low population density, few offspring, opportunistic) polar bears may be at greater risk of severe population declines through exposure to endocrine-disrupting chemicals.

4.2 Temporal trends in Hg and POP exposure

The recent AMAP temporal trend report (AMAP, 2016) provides updates on temporal trends for persistent organic pollutants (POPs) in Arctic air and biota. One of its major findings is that of the more than 1100 statistical runs considered on biota time series, only ~12% are of adequate power, that is, cover a time span to detect a 5% annual change with a statistical power of 80% for the current number of years. This highlights the need to maintain monitoring programs to achieve sufficiently powerful time series for the detection of temporal trends. The greatest annual declines (typically 5–10% per year) were observed for legacy organochlorine pesticides, including toxaphenes, hexachlorocyclohexanes (HCHs) and DDTs. Chlordanes and industrial chemicals such as polychlorinated biphenyls (PCBs) and hexachlorobenzene (HCB) have also temporally declined, but at slightly lower rates. In many cases, more recently banned POPs, including polybrominated diphenyl ethers (PBDEs), perfluorooctane sulfonic acid (PFOS) and hexabromocyclododecane (HBCD) still exhibit increasing temporal trends, although fewer time-series datasets are currently available for these chemicals. This highlights that with temporal trends generally not being optimally resolved for POPs, this is a limitation in making connections to changes in biological effects as suggested by temporal changes in biomarker endpoint measurements.

The POPs annexed at the onset of the Stockholm Convention, including DDTs, HCHs, PCBs and chlordanes, mainly show statistically significant declining temporal trends in Arctic air and biota, confirming the continued effectiveness of their ban over 20 to 30 years ago in most developed countries. Interestingly, the downward trend in many of the time-series began decades before the Stockholm Convention entered into force. This is likely to reflect the impact of control measures introduced at the national level in the 1980s and 1990s, in Arctic countries and neighboring regions. However, the rate of decrease of these POPs in Arctic air and biota is now slowing, which suggests that their concentrations are approaching steady state in the environment and also that secondary sources are now likely to be having a more dominant influence. More recently annexed POPs, including PBDEs, HBCD and PFOS, show a more mixed temporal pattern. In several cases concentrations increase until the mid-1990s to 2000 and thereafter decrease.

There are some sites in the Arctic where increasing concentrations of highly regulated POPs, including *trans*-nonachlor, PCBs (ΣPCB_{10} and CB153), ΣDDTs and HCB, have been observed and this is likely to be attributable to local sources. Examples include time series of HCB and PCB congeners (CB52 and CB101) in air, and HCB (in air and biota) in the eastern Greenland-Iceland-Svalbard region. The rising levels in air have been attributed to enhanced re-emission from oceans and land surfaces as well as melting ice and snow due to climate warming. A consistent increase in levels of PCBs and other POPs in polar bears from eastern Greenland in recent years, relative to levels in the mid-2000s, seems to be related to a change in diet as a consequence of sea ice loss (McKinney et al., 2013). Atmospheric levels of PBDEs have been declining at the European air monitoring stations (Zeppelin, Stórhöfði and Pallas) since 2000, but a similar trend has not been observed at Alert, Canada. The lack of decline at Alert may be related to the influence of a nearby military site and to generally much higher use of PBDEs in North America. Regulatory measures for PBDEs introduced in the late 1990s and 2000s have been effective at curbing increasing trends in Arctic biota post-2000. Another brominated flame retardant (BFR) compound, α-HBCD, has seen declines in atmospheric levels at Svalbard since 2006, while biota trends have been increasing since the 1980s. In contrast, PFOS levels over that same period peaked around the mid-2000s in biota and declined thereafter. The declines are thought to reflect the voluntary phase-out of PFOS and PFOS-related products in 2000 by the US company 3M. However, particle phase PFOS and perfluorooctanoic acid (PFOA) are not showing a decline in Arctic air at Zeppelin, and other perfluorinated carboxylic acids (PFCAs) continue to show rising trends in biota.

Temporal trends in contaminant levels are often interpreted as a response to changes in emission levels, however, previous Arctic assessments have revealed that interpreting trends in this over-simplistic manner is not appropriate. Changes in food web structure and in species' feeding habits can strongly affect levels in biota and humans, exemplified by the increasing trend of ΣPCB_{10} since 2000 in polar bears from eastern Greenland. Trends can also reflect changes in environmental processes, several of which can be associated with climate change and natural variability. A detailed examination of trends in individual datasets is necessary for reliable interpretation. Nonetheless, consistency in results from a large number of trend studies, over a wide geographical area, and involving different matrices, may provide an indication that global controls on emissions or global processes are responsible for at least some of the observed results. There is also a degree of consistency in the trend results obtained from air and biota monitoring, which lends support to the argument that other media should be included under the Stockholm Convention Global Monitoring Programme to further evaluate its effectiveness. Modelling results from studies associated with the EU-ArcRisk project (www.arcrisk.eu) indicate that most legacy POP decreases associated with gradual degradation in the environment will continue to exceed any possible increases due to enhanced remobilization resulting from climate change. Notwithstanding this, however, stockpiles of banned pesticides still exist in some countries and these represent a potential source of future contamination (AMAP, 2016). Results of the ArcRisk studies highlight the need to better characterize primary and secondary sources of POPs and to improve models by including indirect effects, such as carbon cycling, catchment hydrology, land use, and vegetation cover (Pacyna et al., 2015).

For mercury (Hg), the most recent time trend assessment was conducted by Rigét et al. (2011). A statistically robust method was applied to 83 time series of Hg in Arctic biota spanning marine, freshwater and terrestrial ecosystems with the aim to generate a meta-analysis of temporal trend data collected over the past two to three decades, mostly under the auspices of AMAP. There was a clear west-to-east gradient in the occurrence of recent increases in Hg, with the Canadian and Greenland region of the Arctic reporting a greater number of datasets showing significant increases in Hg in biota compared to the North Atlantic. Most of the increasing Hg levels were observed in marine species, especially marine mammals. A total of 16 of 83 time series (19%) could be classified as 'adequate', defined as the number of actual monitoring years in a time series being equal to or greater than the number of years of sampling required to detect a 5% annual change in Hg concentration, with significance ($p<0.05$) and 80% statistical power. This was a slight improvement over the 10% adequacy of Hg time series in the previous AMAP assessment. If an additional five years of data were to be added to the current set of time series, it is predicted that 53% of time series would become adequate.

Dietz et al. (2009) reported on time trends of Hg dating back to pre-industrial times in humans, some Arctic marine mammals and birds of prey, to determine the anthropogenic contribution to present-day Hg concentrations and the historical timing of any changes. Trends in hard tissues, such as teeth, hair and feathers, consistently showed an order-of-magnitude increase in Hg in Arctic marine animals that began in the mid- to late-19th century and accelerated into the 20th century. The median anthropogenic contribution to present-day Hg concentrations was 92.4% (range: 74.2–94.4%). Confidence in the data extracted by Dietz et al. (2009) was increased by accompanying data in some studies on chemical dietary proxies ($\delta^{13}C$ and $\delta^{15}N$), allowing for normalization to changes in trophic position and feeding location over time. Including these dietary proxies proved to be crucial for interpreting temporal trends, baseline concentrations, and the anthropogenic contribution of Hg. In a later study, Hg concentrations in polar bear fur collected until 2008 in the North Water Polynya region, Greenland, were reportedly as high as 27-fold above baseline levels from 1300 AD, suggesting that 96% of the current polar bear Hg body burden may be derived from present-day anthropogenic sources (Dietz et al., 2013a). Article 19 of the Minamata Convention on Mercury states that biotic media (e.g., fish, marine mammals, birds) can be monitored, and so these types of temporal trend studies from the Arctic may help inform the effectiveness of the Convention (Article 22).

4.3 Exposure in relation to effect thresholds

4.3.1 Methodology

Using an *in silico* approach to model the additive effects of contaminants in polar bears, Dietz et al. (2015) reported on the use of risk quotients (RQ = BR/CBR = Body Residue/Critical Body Residue). This work evaluated the risk of immunotoxicity (together with reproduction and carcinogenic effects), in 11 circumpolar subpopulations of polar bear from Alaska to Svalbard. RQ calculations were estimated using the rather low critical daily doses for PCBs (0.469 μg/g) determined from laboratory studies on rats and physiologically-based pharmacokinetic (PBPK) modelling to establish CBR levels for polar bears. All subpopulations had RQs above the effect threshold, with East Greenland and Alaskan bears showing the highest and lowest risk, respectively. ΣPCBs alone (not including

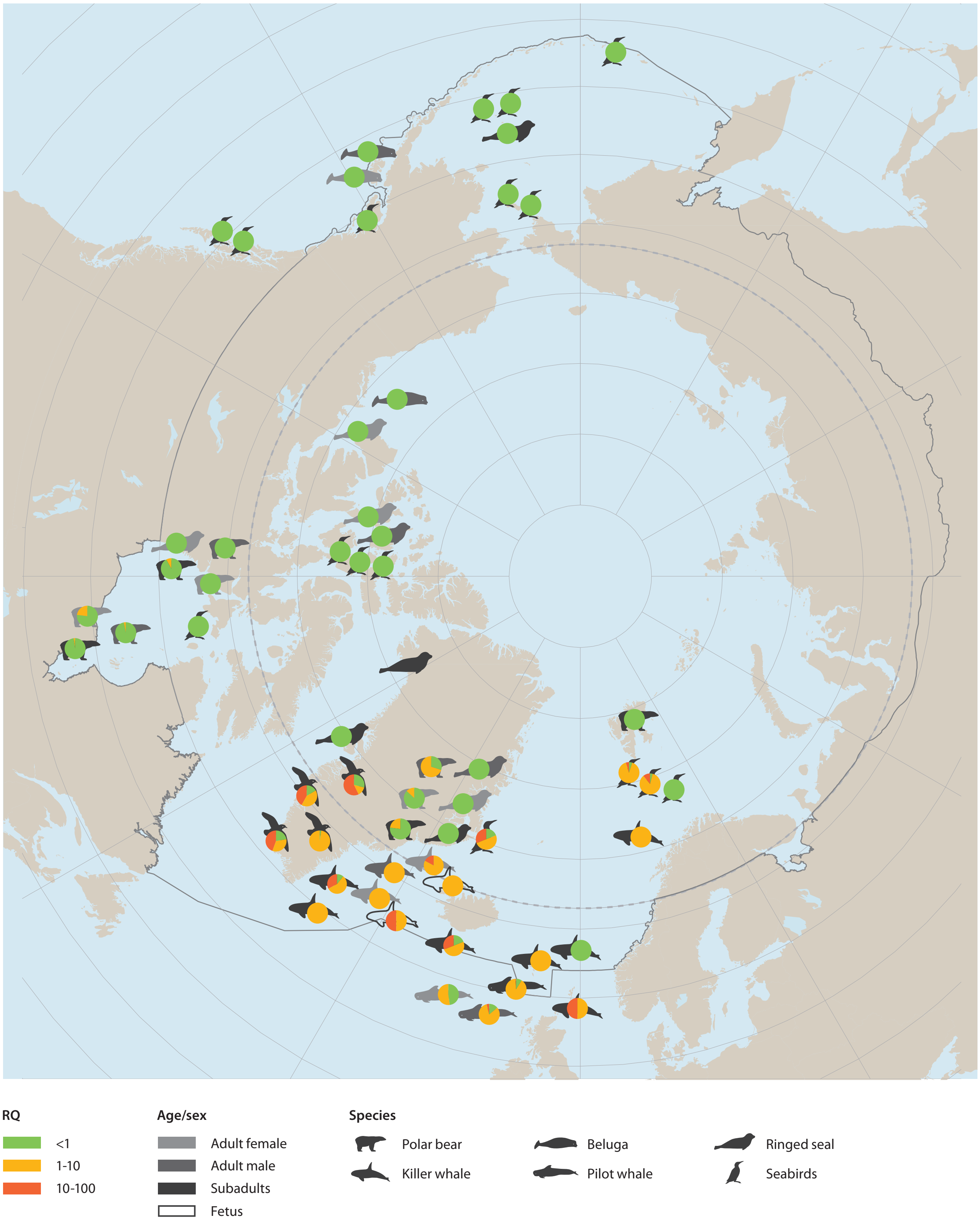

Figure 4.1 Risk quotients for PCB-mediated effects on the immune and hormone systems based on post-2000 sampling of Arctic key species and their ΣPCB loads using a conservatively determined critical body residue of 10 µg/g lw PCBs. See Annex 2 for the detailed information upon which this summary graphic is based.

ΣMeSO$_2$-PCBs) accounted for the majority of the risk (95% range: 88–98%). Due to the overall importance of ΣPCBs, and as most studies report on varying numbers of organohalogen compounds (OHCs), in this study it was decided to calculate the RQs for a number of trans-Arctic species using the RQ from ΣPCBs alone. As for the critical daily doses, a broad range is available in the literature and the relative sensitivity between species, between regions or even over time has never been resolved. It was therefore decided to use a more conservative CBR of 10 µg/g lw of PCBs for both effects on the immune system and effects on the hormone system, and hence the type of effects could be depicted in the same graphic (see Figure 4.1; Annex 2).

Table 4.1 Estimated risk for health effects in wildlife owing to total mercury exposure. See Annex 3 for the detailed information underlying this summary table.

	Risk category, µg/g ww				
	No risk	Low risk	Moderate risk	High risk	Severe risk
Marine mammals	<16	16 – 64	64 – 83	83 – 126	>126
Terrestrial mammals	<4.2	4.2 – 20.5	20.5 – 26.4	26.4 – 44.1	>44.1
Birds (blood)	<0.2	0.2 – 1.0	1 – 3	3 – 4	>4
Birds (liver)	<1.4	1.4 – 7.3	7.3 – 22.7	22.7 – 30.5	>30.5
Birds (egg)	<0.11	0.11 – 0.47	0.47 – 1.30	1.3 – 1.7	>1.7

This threshold concentration represents the upper range of what was reported as the immune threshold by Desforges et al. (2016) for polar bears, cetaceans and pinnipeds. It is thus likely that the RQs are even higher if the actual critical daily doses are proven to be lower in future studies.

Using an *in silico* approach similar to the assessment of PCB exposure, it proved possible to estimate risk for health effects in wildlife as a result of total Hg exposure (see Table 4.1 and Annex 3). These health effects include effects on reproduction and adverse effects on condition, behavior, and productivity. For all species, five risk categories are presently evaluated: no risk, low risk, moderate risk, high risk, and severe risk. Entries are retained that represent the development level (juvenile versus adult) of a species per region, but both levels are merged when one is represented by fewer than five individuals. Data on newborns were limited, and because of the overall insufficiency of the sample size, they are not included in the assessment.

Table 4.1 presents the estimated risk for health effects in wildlife in relation to total Hg exposure. For marine mammals, hepatic total Hg threshold values were used as identified for harp seal (*Pagophilus groenlandicus*) (Ronald et al. 1977). For terrestrial mammals, hepatic total Hg threshold values were used as determined for mink (*Mustela vison*) (Wobeser et al., 1976; Wren et al., 1987). For birds, the assessment methodology employed by Ackerman et al. (2016) was adapted; this methodology also allowed for the reconstruction of risk categories based on liver concentrations as well as egg concentrations.

4.3.2 Marine mammals

4.3.2.1 Risk for PCB-mediated health effects

Toothed whales

Available data and calculated RQs for POPs are summarized in Figure 4.1 and Annex 2. Killer whales (*Orcinus orca*) are clearly the most exposed species and all five North Atlantic subpopulations studied had animals at risk of PCB-mediated effects on the immune system and hormone system. Between 50% and 100% of the East Greenland, Iceland and Shetland subpopulations fell within the RQ range 1–10, and some (age/sex) categories within three of the five studied North Atlantic subpopulations had 31% to 50% in the RQ range 10–100, indicating a high risk of PCB-mediated effects. However, there were also categories in three of the five subpopulations where 11–100% of animals were not critically affected by PCB exposure (RQ<1). Pacific killer whales from the coastal waters of central and northern British Columbia, Canada (data not shown due to their distribution outside the AMAP region) also had high RQs (Buckman et al. 2011), particularly transient killer whales that predate on other marine mammals. All adult transient males fell within the RQ range 10–100 indicating high risk of PCB-mediated effects. The proportion of adult transient females in the RQ range 10–100 was lower (83%) than for adult males (100%). Transient juveniles had lower RQs than adults (all RQs within the range 1–10). Killer whales that are northern residents with a diet based on fish were found to have RQs 12–23 times lower than those predating on other marine mammals, with none falling within the very high risk category (RQ range 10–100). Between 57% and 91% of animals had RQs of 1 to 10, and 9% to 43% were not affected by PCB exposure (RQ<1) (Buckman et al. 2011). These findings on the potential for PCB-mediated effects on the hormone system agree with those of Jepson et al. (2016) who concluded that killer whale subpopulations at lower latitudes (the UK, Ireland, Canary Islands and Gibraltar) are no longer reproducing and so are likely to become extinct. As well as linking contaminant loads to effect thresholds, Reeves and Notarbartolo di Sciara (2007), Guinet et al. (2007), Beck et al. (2013), Hammond et al. (2013) and Jepson et al. (2016) concluded that only very small killer whale populations now occur in industrialized regions of Europe. Their findings include observations of low numbers, major and long-term population declines, significant contraction of range, lack of observations, low or no reproduction, and strandings.

A population of long-finned pilot whale (*Globicephala melas*) from the Faroe Islands also had individuals at risk of PCB-mediated effects. Although the majority of adult males (82%) and immature pilot whales (90%) had RQs of between 1 and 10, the RQs for a small percentage of individuals did fall within the highest risk category (10–100); 4% and 1%, respectively. The pattern for adult females was quite different as no individuals fell within the highest risk category (RQs 10–100) and almost half the population (48%) showed no risk of PCB-mediated effects (RQ<1), with the remaing 52% falling within the intermediate risk catgory (1–10). This is due to maternal transfer of PCBs to the offspring. The finding that toothed whales other than killer whales can exceed the PCB toxicity thresholds was also reported by Jepson et al. (2016) for striped dolphin (*Stenella coeruleoalba*) and bottlenose dolphin (*Tursiops truncatus*) but not for harbor porpoise (*Phocoena phocoena*).

The RQs for beluga (*Delphinapterus leucas*) were all <1 which indicates that this species is not facing major PCB-mediated effects on its immune or hormone system.

Polar bears

Although polar bears are on average feeding at a higher marine trophic level than killer whales, their PCB body residues and hence RQs are lower because killer whales are less able to degrade OHCs such as PCBs. In contrast to killer whales, none of the polar bear subpopulations reported here had RQs in the highest PCB-mediated risk category (10–100) using the conservative CBR of 10 µg/g lw instead of 0.469 µg/g lw as used by Dietz et al. (2015, 2018). Hence the majority of bears had RQs of <1 indicating that their immune and hormonal systems are probably not being affected by PCB exposure. The exception was adult males from East Greenland for which 70% of animals showed some risk of PCB exposure (having RQs within the range 1–10). Overall, the RQs reported here are lower than those for 11 polar bear management areas as reported by Dietz et al. (2015) as well for bears in East Greenland reported between 1983 and 2013 due to the use of the ~20-fold higher CBR of 10 µg/g lw PCBs compared to the 0.469 µg/g lw PCBs from rats (*Rattus rattus*).

Seals

All ringed seals and northern fur seals (*Callorhinus ursinus*) had RQs of <1 and thus that their immune and hormonal systems are probably not being affected by PCB exposure.

Other species

Sonne et al. (2015b, 2016) used a similar approach to evaluate RQs calculated from oral daily doses of PCBs and other OHCs as well as adipose body residues in sledge dogs (*Canis familiaris*) from the controlled feeding study described previously. For both RQ evaluations, the exposed dogs but not the control dogs exceeded immune effect thresholds, corroborating results of *in vivo* immunotoxicity in the sledge dog cohort studies.

Population effect studies

A recent study by Desforges et al. (2018) used a state-of-the-art individual-based model framework combining PCB effects on calf survival and disease mortality to determine population effect predictions of PCBs on killer whale populations around the world. Despite their near global ban over 30 years ago, PCBs remain among the dominant environmental pollutants in wildlife and are present at tissue levels that raise concern in many marine species (Letcher et al., 2010; Jepson et al., 2016). Killer whales are one of the most highly PCB-contaminated species on Earth, raising questions regarding consequences for individuals and at the population level. Desforges et al. (2018) showed that PCB-mediated effects on reproduction and immunity can have severe consequences for the long-term population viability of 10 of the 20 killer whale populations with measured PCB tissue burdens. The majority of the Arctic killer whale population were not predicted to decline (northern Norway, Faroe Islands and Iceland) but the East Greenland killer whales eating marine mammals showed a decline (Figure 4.2). The study reported that killer whales near industrialized regions at lower latitudes and those feeding at higher trophic levels regardless of location were at high risk of PCB population collapse over a 100-year simulation period based on current blubber PCB levels. Their results highlight the vulnerability of killer whale populations to the persistent threat of PCBs, with many of these populations already listed under

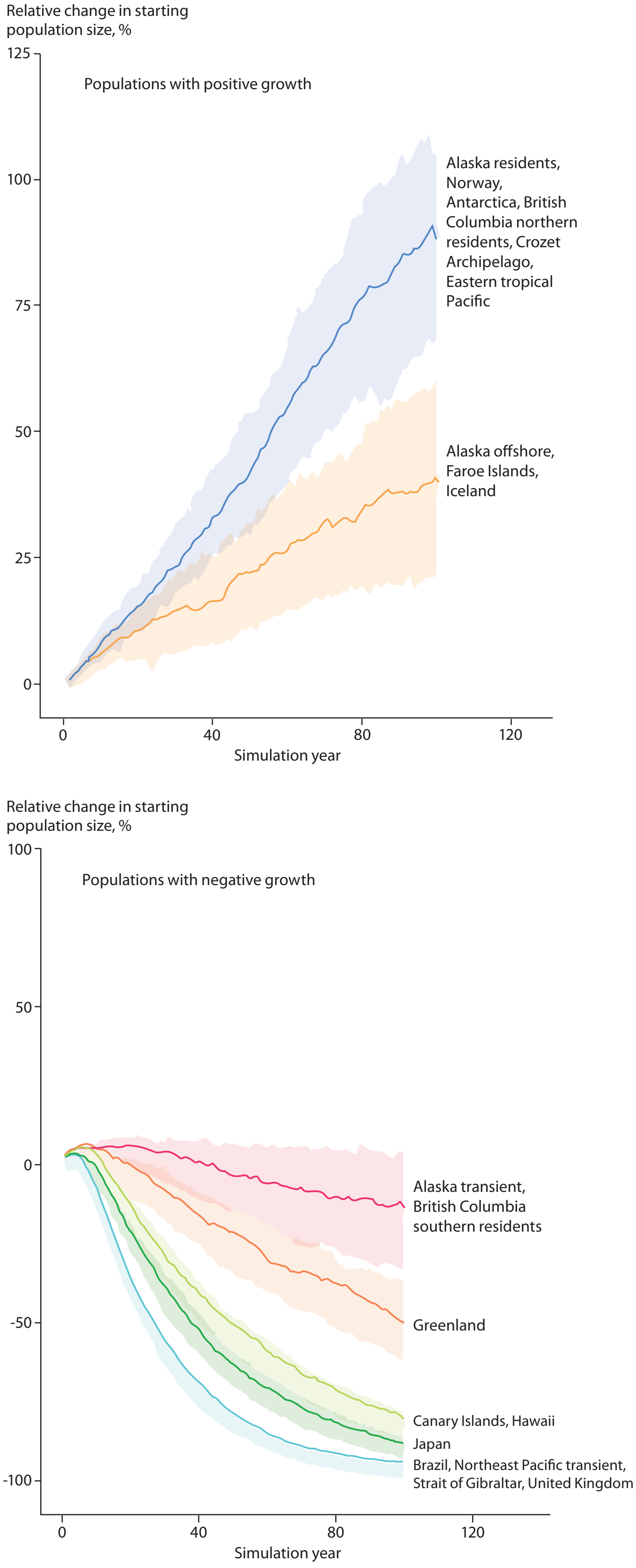

Figure 4.2 Combined reproductive and immune effects of PCBs on population size of killer whales simulated for a 100-year period. Bold line and shading represent the median and interquartile range represented as a percentage of starting population size. Panels are divided by populations with positive growth and negative growth. Modified from Desforges et al. (2018).

various national threatened species programs despite the ban on PCBs more than 30 years ago. Given that PCB levels have stopped declining in many marine mammal populations, the current status of global PCB remediation efforts are obviously not sufficient to protect most wild killer whale populations.

4.3.2.2 Risk for Hg-mediated health effects

Toothed whales

In general, most marine mammal species are at low risk from total Hg exposure (Figures 4.3 and 4.4; Annex 3). However, results for some high trophic-level species, such as polar bear, certain toothed whales including pilot whale, narwhal (*Monodon monoceros*) and beluga, as well as hooded seal (*Cystophora cristata*), fell outside this broad pattern.

As much as 60% of juvenile pilot whales from the Faroe Islands are at severe risk for total Hg-mediated health effects. This is critical for these animals due to their developmental life stage. For adult pilot whales, 33% of individuals were classified as being at severe risk and an additional 20% at high risk. This is also of substantial concern. As reported in Section 4.3.2.1, the PCB loads of Faroese pilot whales are also very high, and cumulative effects can be expected. Risk for total Hg-mediated health effects in adult killer whales is also high; with 33% of individuals at severe risk and 33% at high risk. This contrasts with the situation for fetus killer whales which were all within the no risk group owing to the effective placental barrier for MeHg and total Hg. Juveniles

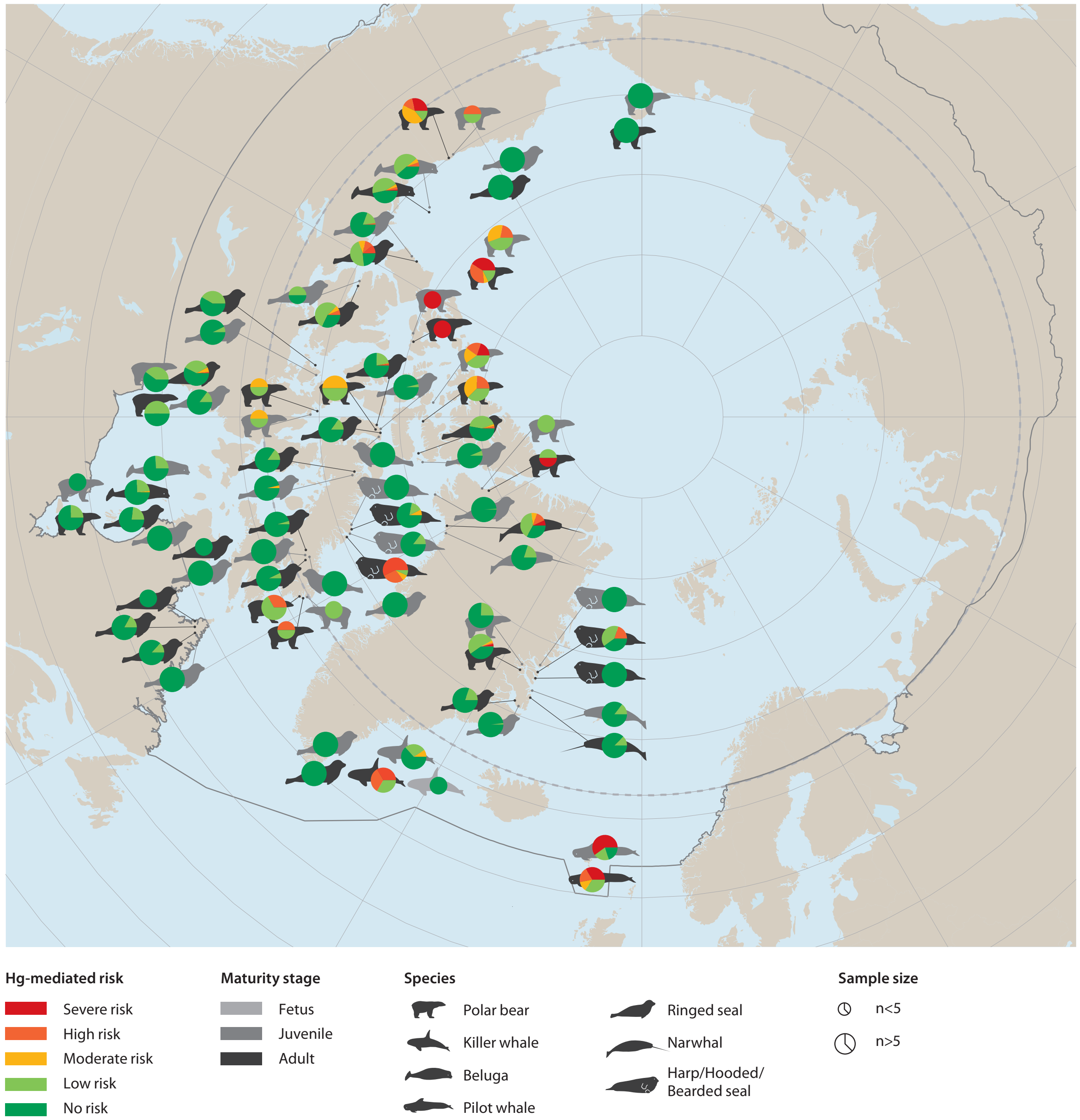

Figure 4.3 Geographical overview of the proportion of individuals of specific Arctic marine mammal populations that are at risk of Hg-mediated health effects; based on post-2000 monitoring data grouped according to maturity where possible. The five risk categories are defined using effect threshold categories observed for harp seals (Ronald et al., 1977). See Annex 3 for the detailed information upon which this summary graphic is based.

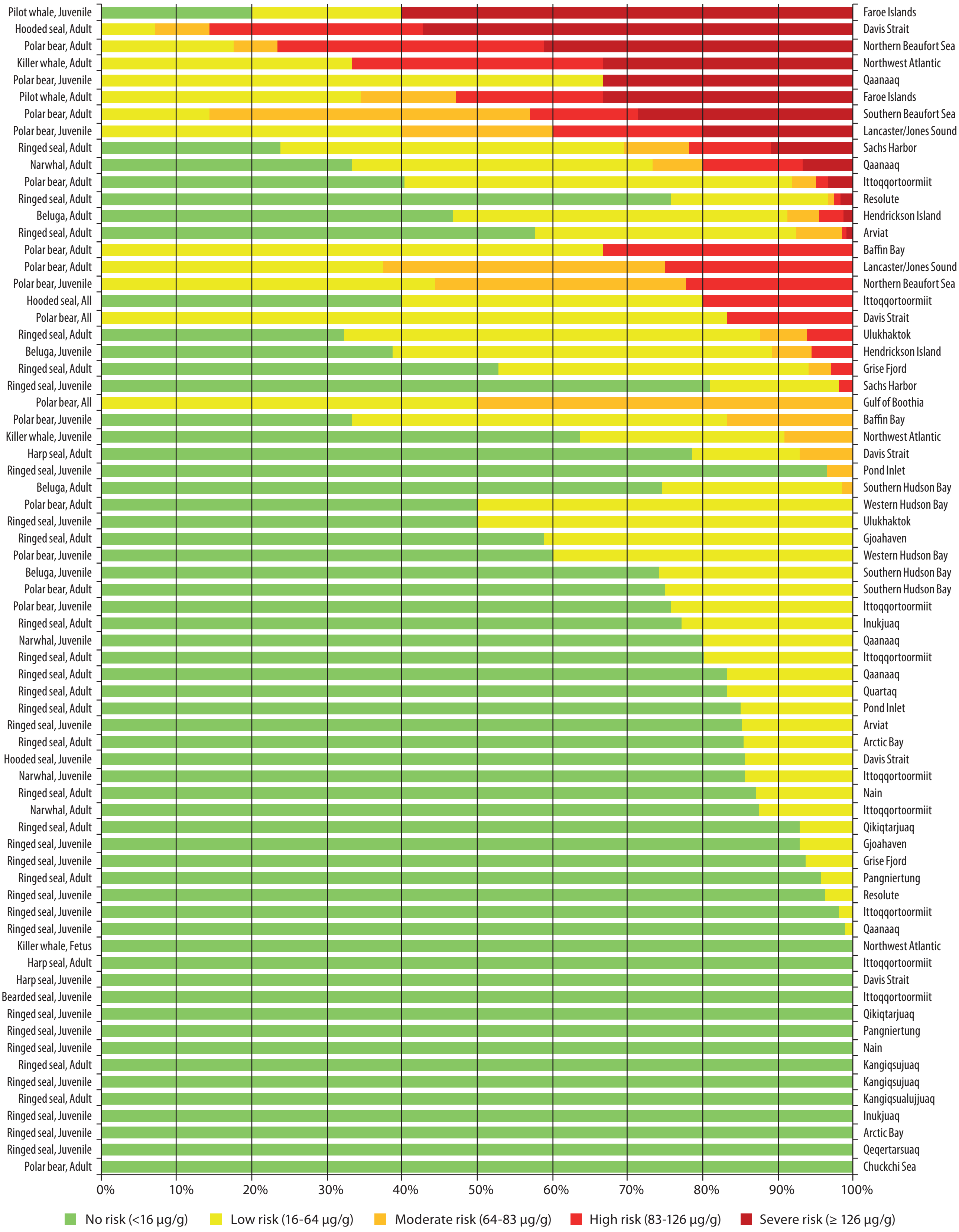

Figure 4.4 Ranked overview (from highest to lowest risk) of the proportion of individuals, where possible grouped according to maturity, of specific Arctic marine mammal populations that are at risk of total Hg-mediated health effects. Following 2000–2015 hepatic concentrations, five risk categories are reported based upon effect threshold categories observed for harp seals (Ronald et al., 1977).

were all within the three lowest risk categories: moderate risk (9%), low risk (27%) and no risk (64%). However, it is unclear whether toothed whales can cope with higher total Hg loads than harp seals, which have been used to categorize the risk thresholds. Adult narwhal from the Qaanaaq region were also shown to be at risk, with 7% of individuals from this population at severe risk and 13% at high risk, which was not the case for narwhal on the Greenland East Coast around Ittoqqortoormiit for which there were no individuals in either risk category. For adult beluga from Hendrickson Island 1% of individuals are considered to be at severe risk and 3% at high risk, for juveniles from this population the proportion at high risk is slightly higher (5%) but none are considered at severe risk.

Seals

Adult hooded seals from Davis Strait in the Canadian eastern Arctic had the second highest number of individuals with total Hg exposure of severe risk (57%) and high (29%) risk. Around Ittoqqortoormiit, on the Greenland east coast, the exposure of hooded seal (all age classes) seemed to be lower with only 20% of the population at high risk and none at severe risk. Adult harp seals from the Davis Strait had even lower exposure with 7% of individuals at moderate risk and none at high or severe risk, while the juvenile harp and hooded seals were at low or no risk. Three of the 33 ringed seal groups had individuals in the severe risk category: adult seals from Sachs Harbor (11%), Resolute (2%) and Arviat (1%). Six of the 33 ringed seal groups had individuals in the high risk category: adult seals from Sachs Harbor (11%), Resolute (1%), Arviat (1%), Ulukhaktok (6%) and Grise Fjord (3%), as well as juveniles from Sachs Harbor (2%). All of these populations were in the northeastern to central Canadian high north in exactly the same area that had the highest Hg concentrations in polar bears. These areas have previously been identified as high Hg risk areas (e.g., Dietz et al., 1998, 2013c). Bearded seals (*Erignathus barbatus*) seemed to be at no risk from total Hg exposure as the majority of individuals were exposed to levels that are thought not to be associated with any risk.

Polar bears

Adult polar bears from the northern Beaufort Sea were the third highest total Hg exposure group with 41% of the population at severe risk, and 35% at high risk of Hg-mediated health effects. The other polar bear populations at severe risk are juvenile bears from Qaanaaq (33%), adults from the southern Beaufort Sea (29%), juveniles from Lancaster/Jones Sound (20%) and adults from Ittoqqortoormiit (3%). All these populations have a substantial percentage (2–35%) at high risk, for example adults from Northern Beaufort Sea (35%), Baffin Bay (33%) and Lancaster Sound/Jones Sound (25%), and juveniles from the northern Beaufort Sea (22%). In addition, some polar bear populations had data from a few individuals that show alarmingly high exposure of severe risk; such as adult bears ranging up to 414 µg/g ww from the northern Beaufort Sea. Other polar bear groups with low sample size (n<8), such as juveniles from the Gulf of Boothia and adults from the southern Beaufort Sea, showed exposure associated with high risk of effects.

The molar ratio of Hg:Se has been found to be 1:1 in tissues of marine mammals with high Hg exposure, suggesting that Hg and Se can be bound in complexes (Koeman et al., 1973, 1975; Dietz et al., 2000). So if the molar ratio of Hg:Se is greater than 1 this may indicate a Se deficiency for binding to and thus detoxifying tissue Hg, or that the essential nutrient Se has been scavenged. The molar ratio of Hg:Se was reported by Routti et al. (2011) for polar bears harvested during the period 2005–2008 from ten circumpolar subpopulations. The molar ratio of Hg:Se was found to be ~1.5 for bears from the Gulf of Boothia and East Greenland, but also above one in the southern and northern Beaufort Sea, southern Hudson Bay, and Lancaster/Jones Sound subpopulations. For Hudson Bay polar bears, annual monitoring from 2006 to 2015 of total Hg in liver shows total Hg concentrations have remained essentially unchanged, ranging between 10 and 15 µg/g ww over this period (Letcher and Dyck, 2016). Thus, Hudson Bay polar bears can be considered at low risk. Nevertheless, the findings for the Canadian Arctic and East Greenland polar bears highlight the severity of the total Hg exposure issue in polar bears.

4.3.3 Terrestrial mammals

4.3.3.1 Risk for PCB-mediated health effects

There do not appear to be any recent data on PCBs in terrestrial mammals.

4.3.3.2 Risk for Hg-mediated health effects

The majority of Hg loads in terrestrial mammals fall within the two lowest risk categories for total Hg-mediated health effects (no risk and low risk, see Figure 4.5). For sheep on the Faroe Islands, 15% are at moderate risk, which could be attributed to agricultural fertilization by fish remains or eutrophication by bird droppings (from the extensive seabird colonies on the islands). Caribou/reindeer (*Rangifer tarandus*), and even Hudson Bay and Svalbard Arctic fox (*Vulpes lagopus*)

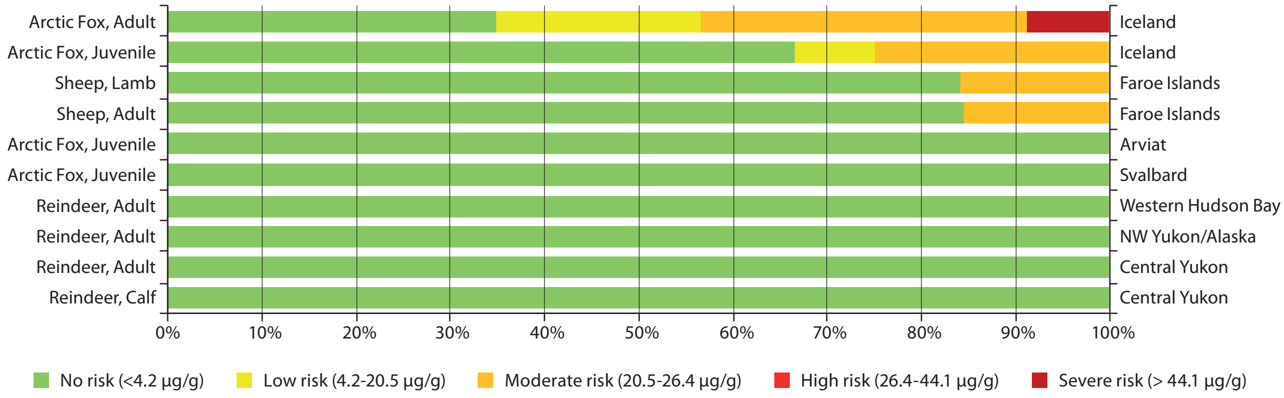

Figure 4.5 Ranked overview (from highest to lowest risk) of the proportion of individuals, where possible grouped according to maturity, of specific Arctic terrestrial mammal populations that are at risk of total Hg-mediated health effects. Following 2000–2015 hepatic concentrations, five risk categories are reported based upon effect threshold categories observed for mink (Wobeser et al., 1976; Wren et al., 1987).

(a scavenger expected to have elevated body residues), seem not to be at risk. Icelandic Arctic fox, however, had low to moderate risk for 8–35% of its population (juvenile and adult), with 9% of the adult population being at severe risk.

4.3.4 Marine birds

4.3.4.1 Risk for PCB-mediated health effects

For the majority of seabirds, PCB data were only available for eggs and blood, resulting in less accurate RQs. Concentrations of PCBs in seabird eggs from Alaska, Canada, East Greenland and Norway (Bjørnøya) all translated into RQs of <1; indicating little risk of PCB-mediated effects on the immune or hormone system. In contrast, based on PCBs in blood, glaucous gulls (*Larus hyperboreus*) from Bjørnøya had a much higher risk of PCB-mediated effects with most birds having RQs that fell within the range 1–10 (90% of females, 85% of males). Relatively few birds had RQs within the highest risk group (10–100) (5% of females, 11% of males) or lowest risk group (<1) (5% of females, 4% of males).

4.3.4.2 Risk for Hg-mediated health effects

In contrast to PCB loads, which were highest in the eastern Arctic, the highest total Hg concentrations were found in Alaska, with intermediate concentrations in Canada and the lowest concentrations in the North Atlantic region (Figures 4.6 and 4.7; Annex 3). Concentrations in three species of adult birds from western North America showed populations of northern fulmar (*Fulmarus glacialis*), double-crested cormorant (*Phalacrocorax auritus*) and pigeon guillemot (*Cepphus columba*) to be at severe risk (4–15%) or high risk (3–26%). In addition, eggs from ivory gull (*Pagophila eburnea*) at Cape Kluyw (14%), thick-billed murre (*Uria lomvia*) at Prince Leopold Island (4%) and black guillemot (*Cepphus grylle*) from the Faroe Islands (1%) were found to be at high risk. The remaining species were not found to be at risk or were at low risk (Figure 4.6; Annex 3).

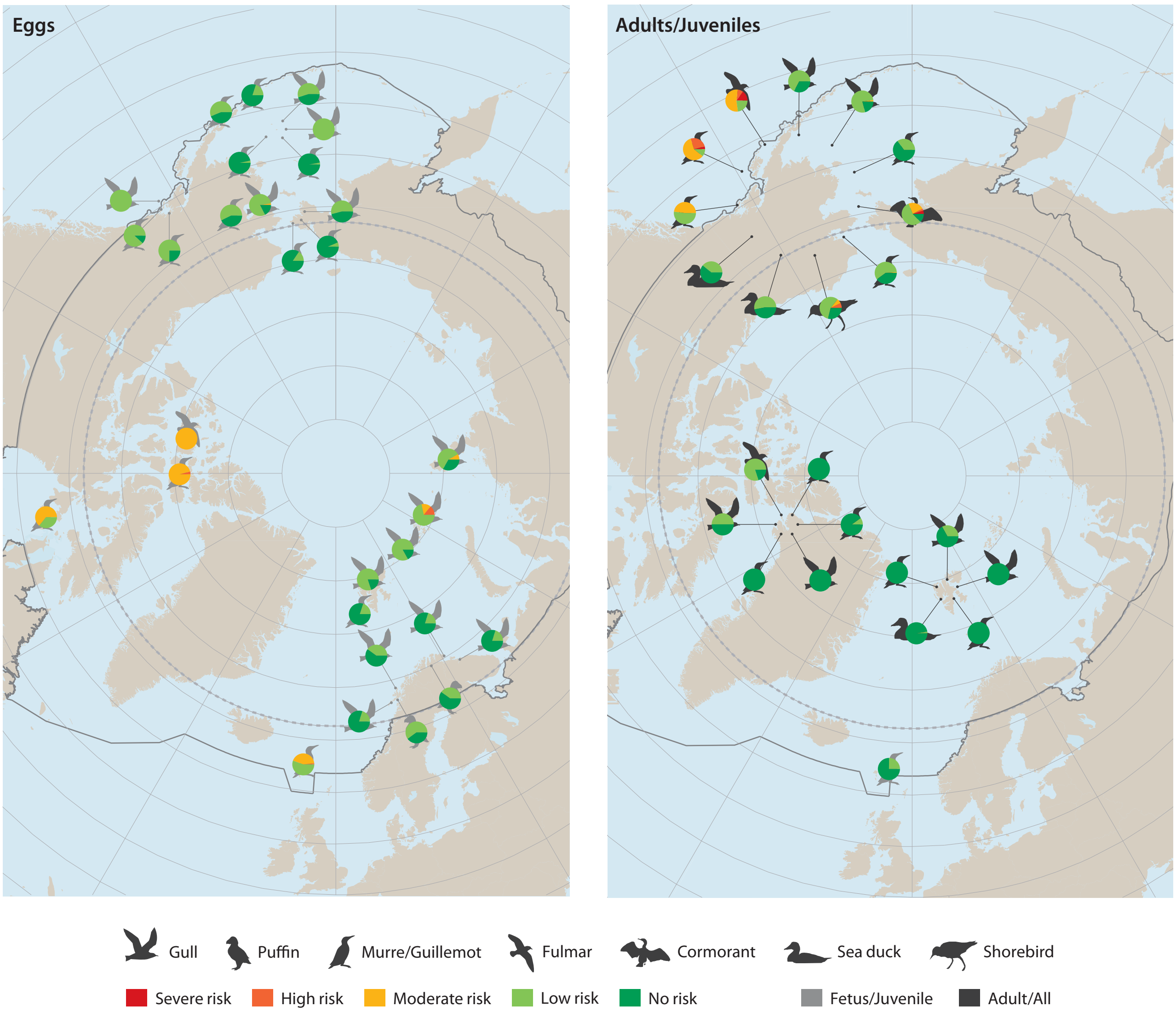

Figure 4.6 Geographical overview of the proportion of individuals of specific Arctic marine bird populations that are at risk of total-Hg mediated health effects; based on post-2000 monitoring data grouped according to maturity where possible. Results based on bird egg monitoring data and results from monitoring of adult/juvenile birds. The five risk categories are defined using effect threshold categories for North American bird species (Ackerman et al., 2016). See Annex 3 for details of datasets, sources and effects category thresholds.

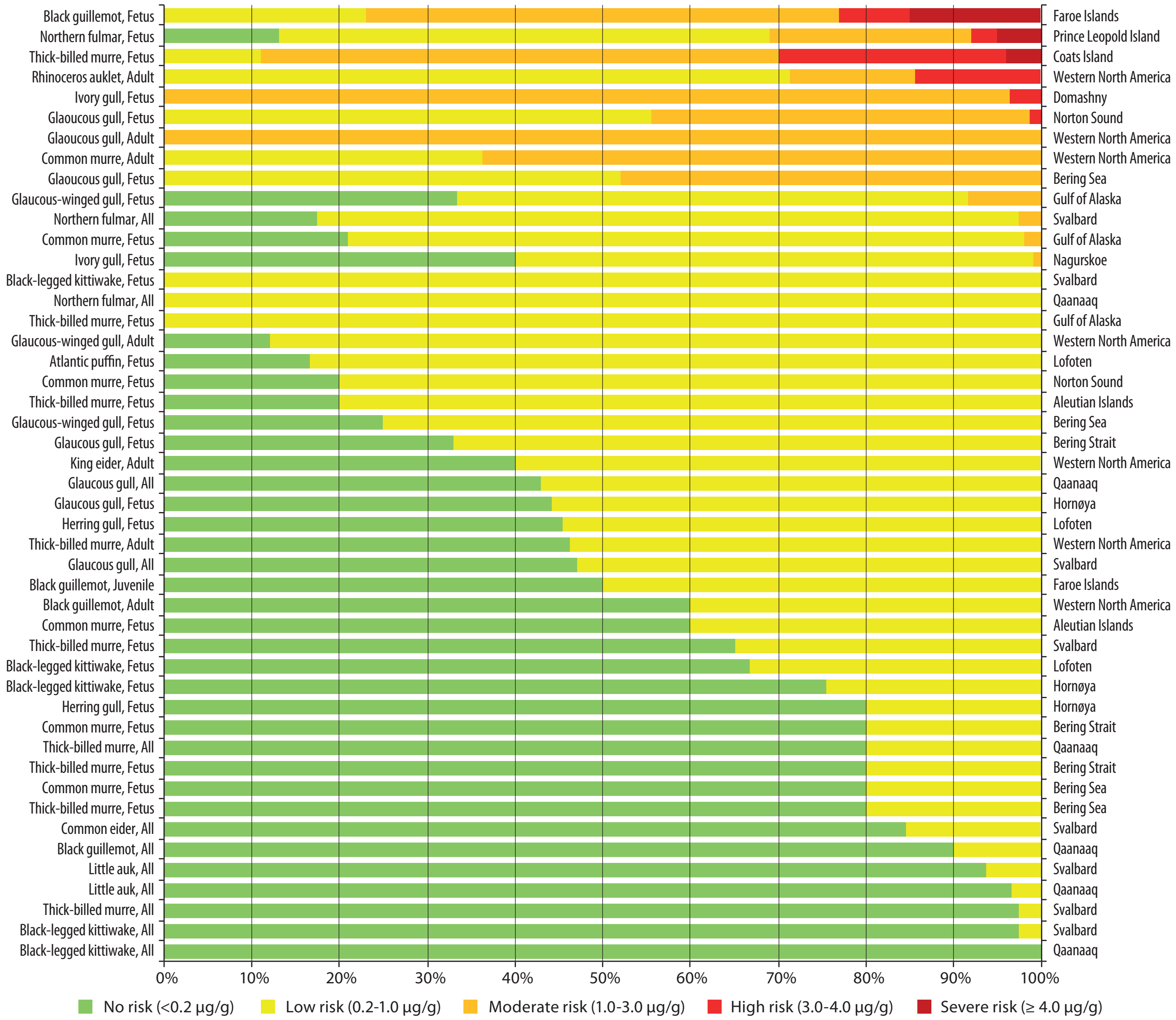

Figure 4.7 Ranked overview (from highest to lowest risk) of the proportion of individuals, where possible grouped according to maturity, of specific Arctic marine bird populations that are at risk of total Hg-mediated health effects. Following 2000-2015 hepatic concentrations, five risk categories are reported based upon effect threshold categories observed for North American bird species (Ackerman et al., 2016).

4.3.5 Terrestrial birds

4.3.5.1 Risk for PCB-mediated health effects

Loadings for PCBs in birds of prey were determined in liver. White-tailed eagles (*Haliaeetus albicilla*) from West Greenland were used where adipose correction factors allowed for harmonization as a result of comparisons of multiple tissues. The three birds of prey studied on Greenland (white-tailed eagle, gyrfalcon *Falco rusticolus* and peregrine falcon *F. peregrinus*) showed high risk of PCB-mediated effects. The proportion of animals within the highest risk category (RQ range 10–100) varied from 38% to 57%, with slightly lower proportions (14% to 38%) in the RQ range 1–10. Only 15% to 29% of individuals in each population showed no risk of PCB exposure (RQ<1). In contrast, snowy owl (*Bubo scandiacus*) had PCB concentrations that were 10–20 times lower and so had much lower RQs. At 83%, the proportion of birds that were not affected by PCB exposure (RQ<1) was also much higher.

4.3.5.2 Risk for Hg-mediated health effects

The northern shoveler (*Anas clypeata*) from western North America was the terrestrial bird species with the highest risk for total Hg-mediated health effects, with 14% of individuals at severe risk, 5% at high risk, 29% at moderate risk, 35% at low risk, and only 17% at no risk (Figures 4.8 and 4.9; Annex 3). Other terrestrial bird species at severe risk are adult common loon (*Gavia immer*; 5%), peregrine falcon (4%), Pacific loon (*G. pacifica*; 3%), red-throated loon (*G. stellate*; 3%), bald eagle (*Haliaeetus leucocephalus*; 2%), green-winged teal (*A. carolinensis*; 2%), and yellow-billed loon (*G. adamsii*; 1%). Adult greater scaup (*Aythya marila*, 2%) from western North America was found to be at high risk. Moderate risk was observed for white-tailed eagle from Swedish Lapland (17%), adult red-necked grebe (*Podiceps grisegena*; 14%), juvenile gyrfalcon from Nuuk on the west coast of Greenland (8%) and adult osprey (*Pandion haliaetus*) from western North America (2%). In the moderate risk group almost all groups were represented and the five highest were adult common loon (35%), peregrine falcon (29%), northern shoveler (29%), bald eagle (26%) and greater scaup (21%), all from western North America (Ackerman et al., 2016).

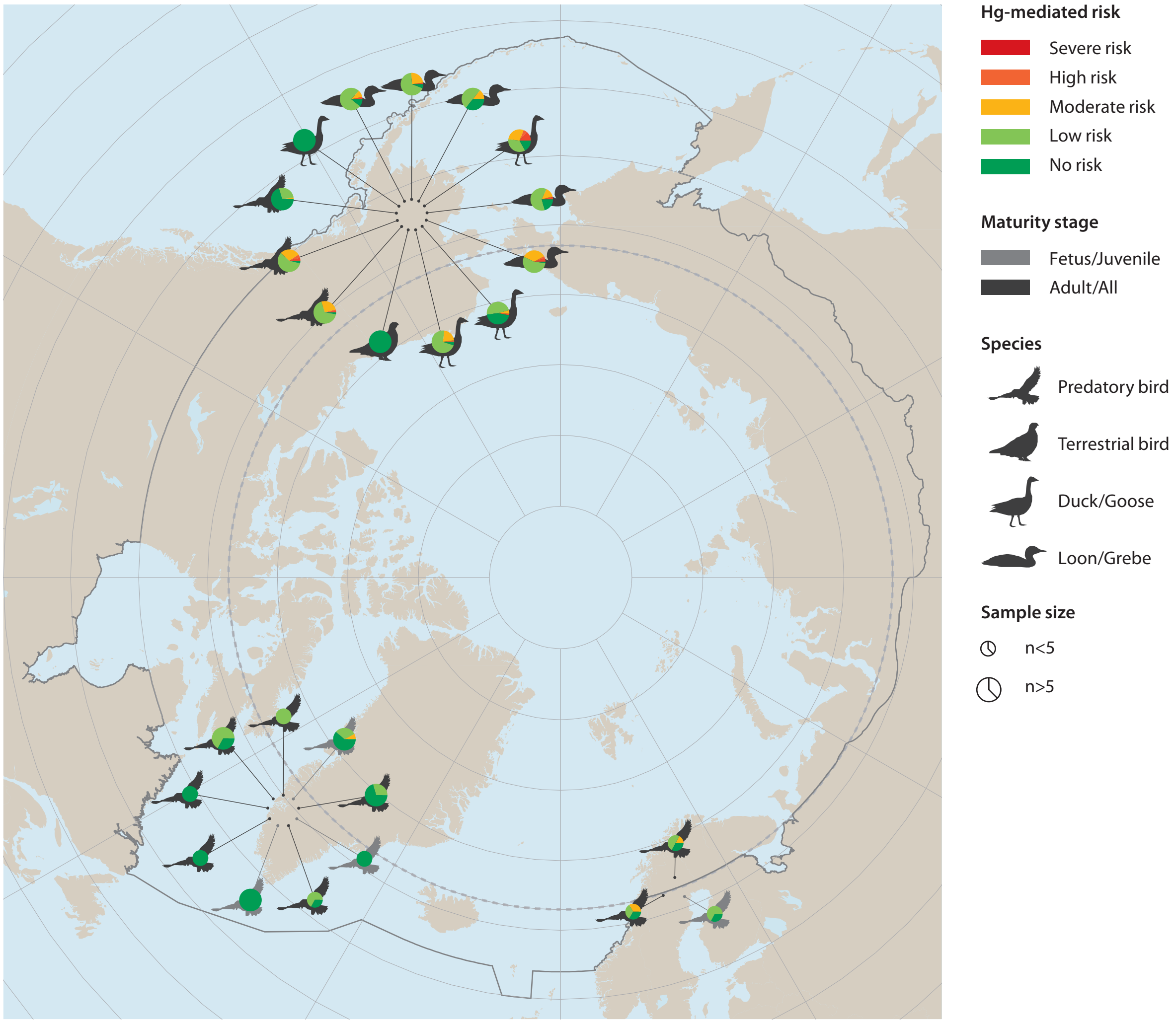

Figure 4.8 Geographical overview of the proportion of individuals of specific Arctic terrestrial and freshwater bird populations that are at risk of total-Hg mediated health effects; based on post-2000 monitoring data grouped according to maturity when possible. The five risk categories are defined using effect threshold categories for North American bird species (Ackerman et al., 2016). See Annex 3 for details of datasets, sources and effects category thresholds.

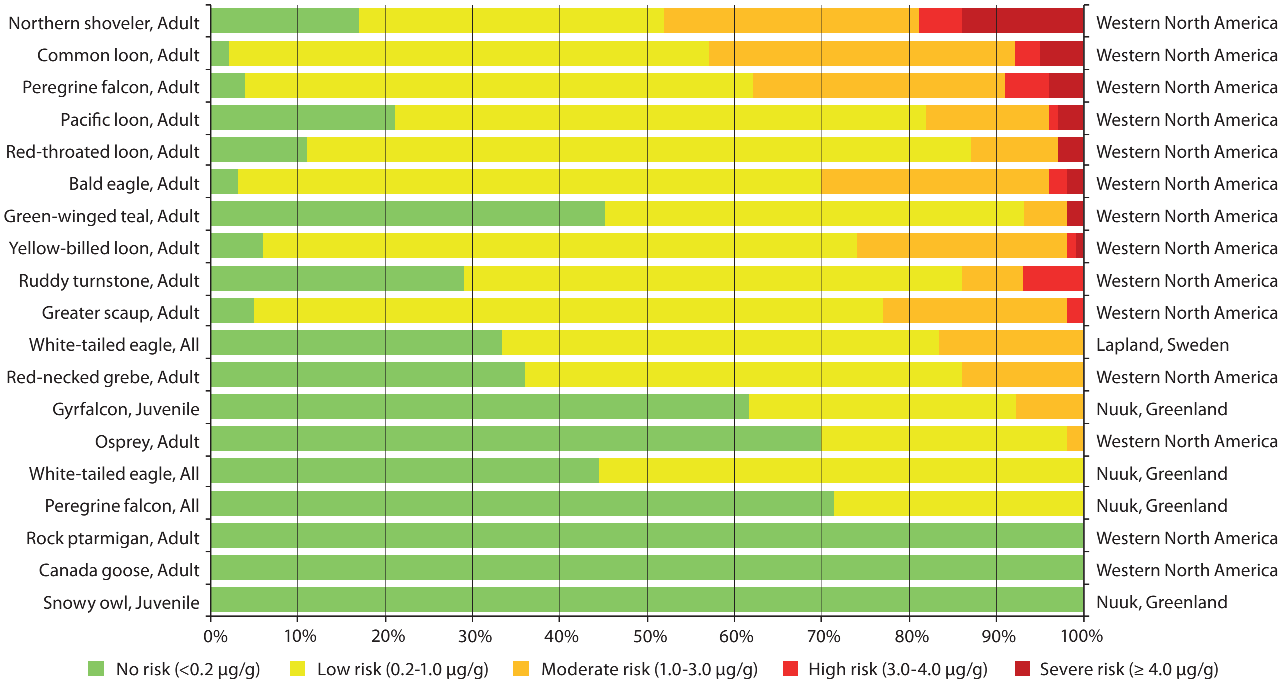

Figure 4.9 Ranked overview (from highest to lowest risk) of the proportion of individuals, when possible grouped according to maturity, of specific Arctic terrestrial bird populations that are at risk of total Hg-mediated health effects. Following 2000–2015 hepatic concentrations, five risk categories are reported based upon effect threshold categories observed for North American bird species (Ackerman et al., 2016).

4.4 What is a ‘normal’ physiological range?

The normal range, or reference intervals, for important physiological endpoints in Arctic terrestrial and marine species are not yet known. Examples of endpoints include serum biochemistry, hematology, hormone and cytokine values, functional immune assays, and body condition. Statistically-derived reference intervals provide a baseline against which individual test results can be compared, thus allowing an evaluation of individual health relative to a ‘normal’ healthy population. Reference values are typically reported as upper and lower bounds of a reference interval comprising 95% of a healthy population. To date and in regards to Arctic species, reference intervals have been established for serum chemistry values for free-ranging beluga, harp seal, and ringed seal (Nordoy and Thoresen, 2002; Tryland et al., 2006a,b), as well as for managed-care beluga (Norman et al., 2013).

There are several challenges in developing reference intervals for free-ranging Arctic species. The main challenge is obtaining an adequate sample size to establish a reference interval for a particular biomarker. Procedures for establishing reference intervals based on reference sample size and distribution have been recommended by the American Society for Veterinary Clinical Pathology for wildlife species, with sample sizes ranging from 20 to 120 (Friedrichs et al., 2012). Other considerations include the tissue/sample to collect (e.g., blood/serum/plasma, urine), time to collection (e.g., fresh/live capture versus stranded versus fresh dead), tissue storage (e.g., room temperature, -20°C, -80°C, liquid nitrogen), central tissue bank/repository, and permits to ship samples internationally (i.e., endangered/protected species) to a laboratory with the equipment to analyze the sample. Most important are the validation and usefulness of the biomarkers and reference intervals as tools to help make the link between contaminant exposure and adverse health outcomes in individuals and populations. Reference intervals for free-ranging and managed-care animals may need to be established, as these two types of population are exposed to different stressors, which may influence reference intervals for different endpoints.

Sonne et al. (2008) used reference intervals to investigate the impacts of OHCs on the health of Arctic top predators using domestic Greenland sledge dogs as a model for East Greenland polar bears. Two urine and 29 different blood plasma parameters were measured and compared to established reference interval values for canines. Overall, the use of reference intervals allowed for the detection of differences in kidney and liver chemistry values between exposed and control dogs. The authors suggested that their data shows that a number of clinical blood parameters are potential biomarkers of OHC exposure in wildlife.

4.5 Case studies of multiple stressors in the Arctic

4.5.1 Climate change

A literature review by McKinney et al. (2015) (see also Chapter 2) gave initial insights into the ecological aspects of climate change-contaminant interactions. Other studies have suggested ecological change without actually collecting such corroborating data. In remote and aquatic environments it can be very difficult to directly assess ecosystem change, particularly for species and their trophic relationships (Bowen and Iverson, 2013). This may be improved by using ecological tracers, among which are bulk tissue stable isotope ratios and fatty acid profiles. The associated analyses are inexpensive relative to contaminant analyses and can often be achieved with little or no additional sample collection or requirements. Ecological tracer data paired with contaminant exposure data can increase understanding of the ecological changes altering contaminant pathways within species, populations or food webs. An example of this approach is provided by McKinney et al. (2013) who investigated how climate change may have affected the long-term (1984–2011) feeding ecology and contaminant exposure of East Greenland polar bears. Dietary analysis using quantitative fatty acid concentrations revealed that changes in the distribution and availability of ice-dependent seals had resulted in higher OHC concentrations in the polar bears due to a prey-shift from ringed seals to hooded and harp seals. Developing and refining additional tissue-derived tracers of habitat use and trophic interactions (e.g., compound-specific isotope analysis; McKinney et al., 2013) together with other ecological information (e.g., lipid content as a body condition metric; McKinney et al., 2014) may be useful for improving understanding of changing contaminant levels and pathways, and their associated health effects.

As an iconic Arctic species, polar bears have been the focus of many studies on the effects of climate change on animal health. Based on recent temporal studies on plasma biomarkers for starvation in eastern Beaufort Sea polar bears, it has been suggested that a greater proportion of bears now fast. Increased fasting is likely to be due to broad-scale changes in Arctic sea ice composition owing to climate change and the consequent impact this has on prey availability (Cherry et al., 2009). Body condition indices of Hudson Bay polar bears also support evidence of increased starvation due to reduced access to ice-associated seals (Molnár et al., 2011). Furthermore, qualitative and/or quantitative changes in diet and reduced body condition (i.e., adipose stores) of polar bears can affect their tissue concentrations of lipophilic pollutants. A recent study by Tartu et al. (2017b) examined Svalbard polar bears and considered how variations in adipose stores are associated with both breeding status and spatial changes in sea-ice conditions, and how diet influences concentrations and biotransformation of POPs. Relative to diet, body condition was found to be negatively related to sea-ice extent at both temporal and spatial scales, and was the most important predictor for concentrations of POPs in plasma and fat. Fatter females were also more efficient at biotransforming PCBs than leaner females. The authors concluded that declining sea ice indirectly leads to increased concentrations of lipophilic pollutants in (Svalbard) polar bears mediated through reduced feeding opportunities and declining body condition rather than changes in dietary composition. The size of the polar bear subpopulations of Hudson Bay has decreased by approximately 22% and it has been proposed that this could be due to reduced body condition index and lowered sub-adult survival (Stirling et al., 1999; Obbard et al., 2006; Regehr et al., 2007). Because of the southern distribution of the Hudson Bay subpopulation, these polar bears may serve as a proxy for future climate change impacts on their circumpolar health and survival. For example, the consequence of energetic stress combined with the physiological effects caused by the bioaccumulation of chemical contaminants could prove significant for the distinct East Greenland subpopulation since that carries some of the highest loads of endocrine-disrupting chemicals in the Arctic (Sonne, 2010).

Jenssen et al. (2015) reviewed available literature on the combined effects of climatic change and exposure to POPs in polar bears and concluded that one of the greatest threats was reduced access to seals due to loss of sea ice, resulting in prolonged fasting periods which would in turn affect body condition (emaciation), survival and reproductive success. Catabolic processing of peripheral adipose tissue, in order to compensate for the lower energy intake, is likely to increase blood concentrations of toxic and bioavailable contaminants and metabolites that will exacerbate harmful impacts on various organ systems. Poorer body condition is also likely to induce metabolic changes with negative impacts on hormones, blood chemistry homeostasis, survival rates and fecundity (Sonne, 2010). For instance, Chow et al. (2011) found that fasting bears reduce amino acid- and protein catabolism by limiting cortisol bioavailability through the elevation of serum cortisol-binding-globulin. Nevertheless, studies focusing on climate change, POP interactions and their effects on wildlife and humans, still remain to be conducted on a larger scale, to generate results that would further clarify these relationships.

Decreasing their consumption of seals may lower polar bear OHC exposure, but may also have indirect consequences such as decreasing crucial vitamin intake. Lower seal consumption could also lead to more bear-bear and bear-human interactions and time on land, which makes them more susceptible to wounding, hunting by humans and drowning (Monnett and Gleason, 2006). Ultimately, these factors increase bear mortality and reduce their total circumpolar number, although the survival of each subpopulation will depend on the specific sea-ice reductions in each particular habitat. For example, the most pronounced polar bear habitat loss since 2000 has occurred around East Greenland and Svalbard, and in the same period polar bear body size has decreased in both subpopulations probably due to a combination of energetic stress and neuroendocrine disruption (Sonne, 2010). Ecotourism (e.g., at Svalbard), oil exploration/exploitation, anthropogenic underwater noise, increased and changing shipping routes including the possibility of traverse through the North-West and North-East passages, and an influx of new species are likely to be additional co-factors in the combined stressors to polar bears.

4.5.2 Infectious diseases and zoonoses

Novel and shifting of wildlife disease patterns in the Arctic are of increasing interest (Shope, 1991; Burek et al., 2008). As well as novel distributions of various disease pathogens, this interest covers significant changes in prevalence, increasing pathogen-survivability and changes in morbidity/mortality. A novel distribution is largely driven by climate change and specifically Arctic warming, which has seen a northward influx to the Arctic of new wildlife and insect species as vectors for disease, as well as accelerated pathogen lifecycles. Kutz (2009) reported that the lifecycle of the muskoxen (*Ovibos moschatus*) lungworm (*Umingmakstrongylus pallikuukensis*) had shifted from a two-year cycle to a one-year cycle, and that its distribution had recently shifted several hundred kilometers northward to populations previously regarded as devoid of the parasite (Kutz et al., 2013; Solomon, 2014). Similarly, the nematode *Setaria tundrae* has migrated northward in Finland infecting previously unaffected reindeer following consecutive warm summers (Ball et al., 2001). Another example is the moose tick (*Dermacentor albipictus*) which has been spreading northward in the Northwest Territories, Canada. Ticks are arthropods, and arthropods generally hold great potential as disease vectors for both animals and humans (Sonenshine and Mather, 1994; Kutz, 2009). Because arthropods are sensitive to temperature (Elias, 1994), higher temperatures are likely to increase the abundance of disease-carrying arthropod vectors. As in the Northwest Territories of Canada, ticks have also been observed to be moving northward in Sweden (Lindgren and Gustafson, 2001). Human cases of tick-borne encephalitis increased 50-fold between 1980 and 2009 in the northern Russian province of Arkhangelsk Oblast (Solomon, 2014). Extreme weather events linked to climate change have also been tied to outbreaks of zoonotic disease (Ahern et al., 2005). This is thought to be mediated through rodent vectors being expelled from their burrows by flooding, sewage contaminated waterways, and prime conditions for waterborne disease pathogens such as *Cryptosporidium*, *Vibrio cholera*, *Escherichia coli*, *Salmonella* and *Giardia* as well as for mosquito breeding grounds that are left behind after floods (Epstein, 1999; Palmieri et al., 2012).

Fish-borne zoonoses such as *Anisakis simplex*, *Diphyllobothrium latum*, *Contracaecum osculatum* and *Pseudoterranove decipiens*, carried by typical fish for human consumption, are also expected to gain better lifecycle conditions in warmer waters. Climate associated changes include accelerated embryonation and hatching of eggs in seawater as well as accelerated development of larval stages, increased numbers of crustacean intermediate hosts and a northward movement of paratenic and final hosts (Rokicki, 2009; Scholz et al., 2009). Many of these pathogens have already been found in high prevalence in Greenland cod (*Gadus ogac*) and Atlantic cod (*G. morhua*) as well as Greenland halibut (*Reinhardtius hippoglossoides*; Mouritsen et al., 2010; Karpiej et al., 2013).

Few studies have examined the incidence of zoonotic transmitted diseases in humans in the Arctic. The majority of such studies have been performed in Alaska and Canada, with just a small number of studies in Greenland and Russia. This is despite a significant proportion of the indigenous Arctic residents living in close contact with wildlife. Indeed, traditional food preparation often avoids heat treatment. This process can be extremely effective against zoonotic diseases as shown by the case of *Toxoplasma gondii* (which causes life-long parasitic cysts in the human brain): 80% of Inuit consuming dried marine mammal meat tested seropositive for the disease, while prevalence was only 10% in Cree who prefer cooked meat (Messier et al., 2008).

Andersen-Ranberg (pers. comm.) undertook a comprehensive review of zoonotic diseases present in key Arctic wildlife species. This showed that at least 26 zoonotic pathogens occur in a wide range of species that are typically hunted throughout the Arctic. It is clear that zoonoses are a very real threat to Arctic residents, and this highlights the need for an overview of the most serious zoonotic diseases, especially for those Arctic regions where information is particularly sparse. The need for an overview is strengthened by the context of changing disease patterns, as well as ongoing bioaccumulation and exposure to toxic chemical contaminants, some of which have been proven to be immune-suppressive. This would render both animal and human hosts more vulnerable to diseases, especially hosts that are already immunologically vulnerable to novel diseases (Dietz et al., 1989; Ross et al., 1995; Sweet and Zelikoff, 2001; Sonne et al., 2006a; Corsini et al., 2014; Bogomolni et al., 2016).

Antibiotic resistant bacteria have also been found in the Arctic. This signals that antimicrobial resistance is now spreading worldwide (Sjölund et al., 2008) and highlights the need for more knowledge-based diagnostics and treatment concerning pathogens actually infecting people in the Arctic. Weber et al. (2013) proposed that species in the Arctic have adapted to life in harsh climatic conditions that limit pathogen density and this has resulted in conditioned immune defenses to lower pathogen loads. Their proposal was based on the results of a study on major histocompatibility complex (MHC) genes, essential for immune function, which showed low variance in the MHC binding site in Canadian polar bears. This would theoretically mean that because the variance of the MHC binding sites is low, it can detect fewer infecting pathogens than if the variance was high. If this is true, it puts polar bears and perhaps other Arctic resident wildlife species under further constraints in terms of the effects of multiple stressors.

4.6 Novel tools and approaches

4.6.1 Physiologically-based pharmacokinetic modelling

The weight of evidence provided by correlative associations of OHC exposure with various physiological and biochemical endpoints may identify certain contaminant ‘hot spots’, including East Greenland and Svalbard. However, comprehensive reviews of OHC exposure and effects in Arctic wildlife show there are virtually no data demonstrating a direct OHC-mediated cause-effect (Letcher et al., 2010; Sonne, 2010). To address these concerns, Sonne et al. (2009) employed a risk quotient (RQ) approach to investigate the causal linkage between OHC body burdens and reproductive toxicity in an East Greenland polar bear population. Dietz et al. (2015) employed this approach to evaluate linkages between OHC exposure and reproductive, immunotoxic and carcinogenic effects in polar bear subpopulations across the Arctic, using the most recent spatial trend data for OHCs (McKinney et al., 2011) and PFOS (AMAP, 2016). To achieve this, RQ calculations were performed based on the ratio of adipose OHC concentrations to estimated critical body residues (CBRs). CBRs were calculated from critical daily doses determined by laboratory rat studies using physiologically-based pharmacokinetic (PBPK) modelling (Nielsen et al., 2006; US EPA, 2008). The present assessment conducted a broader geographic comparison with a 20-fold higher CBR including a large number of top predators and using only blubber ΣPCBs concentrations as this was shown to be the major driver of health effects (Dietz et al., 2015). Even with these conservative estimates several species, especially killer whales, and subpopulations were found to be at risk for OHC exposure effects (see Section 3.1.4; Figure 4.1).

According to Sonne et al. (2016), a biomonitoring equivalent is the conversion of an external guideline value such as tolerable daily intake to an internal dose against which biomonitoring tissue data can be directly compared (Hays et al., 2007, 2008). The use of biomonitoring equivalents for interpreting POP concentrations in Arctic top predators has previously been conducted for polar bears (Sonne et al., 2009; Dietz et al., 2015; 2018). These studies have been conducted based on estimates of oral intake of prey tissues and risk assessment of reproductive effects of internal doses obtained from PBPK modelling. Such studies are useful since information about POP exposure and health effects in Arctic top predators is difficult, expensive and time consuming to obtain in order to have adequate sample sizes (Sonne, 2010). It is therefore necessary to convert oral POP exposure to internal organ-tissue doses using PBPK modelling that describes absorption, accumulation and excretion from the body. Moreover, the use of RQs based on single value effect threshold criteria is being widely questioned in the risk assessment community.

In the Arctic, PBPK modelling of POPs in polar bears and other Arctic top predators has been based on contaminant concentrations in prey and food, respectively (Redding et al., 2008; Sonne et al., 2009, 2014a; Verner et al., 2009; Dietz et al., 2015). In these cases, information about internal tissue concentrations is limited and this therefore warrants the development of PBPK models that convert POP exposure to internal concentrations in susceptible organ-systems. In order to evaluate the already established generic PBPK model described by Cahill et al. (2003) for Arctic ecosystems, the distribution, bioaccumulation and excretion of POPs in a controlled setup has been investigated. For that purpose, studies used four Greenland female sledge dogs that were fed on POP-contaminated minke whale (*Balaenoptera acuterostrata*) blubber for 500 to 635 days (Sonne et al., 2006a) from which data on both oral POP exposure and subsequent accumulated concentrations were available. In the Arctic, previous studies have successfully used PBPK modelling as a tool to evaluate specific health endpoints (e.g., Dietz et al., 2015; Sonne et al., 2016). This approach is used because actual oral exposure is often hard to estimate while tissue residues and blood concentrations are measured within the AMAP monitoring programs. The PBPK model estimates CBRs typically in adipose tissue based on literature values of critical daily doses as described by Cahill et al. (2003). The CBR values are then used to calculate RQs (e.g., for reproductive or immunotoxic effects) as RQ = BR/CBR with values ≥1 representing a risk for a particular effect in a particular species (e.g., polar bears) (Hays et al., 2007, 2008; Dietz et al., 2015).

4.6.2 Population- and biology-based modelling

Most toxicological studies, including those reviewed in this effects assessment, apply a traditional risk assessment approach whereby biological/physiological effects are measured at the individual level, be it on the molecular, cellular or organ scale.

In contrast, environmental protection goals within regulatory and conservation bodies are most interested in impacts at the population level. Owing to obvious logistical, financial and ethical restrictions, it is effectively impossible to conduct exposure experiments on whole populations. Thus, effects on individuals must be used as proxies and extrapolated to the population scale. Despite the importance of this exercise, few studies have attempted to evaluate the effects of POPs and Hg on wildlife populations, especially in the Arctic; this is likely to be related to knowledge gaps in wildlife population metrics and mathematical modelling challenges.

Ecological modelling, particularly individual-based models, has increased in popularity for extrapolating effects at the individual level to the population level (Martin et al., 2013). These individual-based models can incorporate physiologically relevant parameters on each individual within a population, which is particularly important for ecotoxicology where exposure and risk of effects changes throughout life-history (e.g., reproductive transfer, growth, bioenergetics, diet). Pavlova et al. (2016a,b) used an individual-based model to evaluate the population-level consequences of PCB exposure and effects on female reproduction in East-Greenland polar bears. This study was a crucial step forward in extrapolating individual reproductive effects from laboratory experiments on mink as a surrogate species, to population responses in a polar bear population shown to be vulnerable to effects on various biomarkers of health (see this and previous AMAP assessments). Although a low risk for reduced population growth was found in the original mink-based toxicity model, the authors showed that slightly more potent dose-response curves for polar bear cub mortality (EC50 <18 ppm lw) would result in significant population declines (Pavlova et al., 2016a). Modeling approaches such as this are essential for testing hypotheses and predicting population responses to various exposure/toxicity scenarios. Furthermore, PBPK modelling can be instrumental in the development of bioaccumulation models that feed into larger population-effect assessments. Bioaccumulation models have already been developed for many marine mammals, including beluga, pilot whale, harbor porpoise and ringed seal (Hickie et al., 2000, 2005; Weijs et al., 2012, 2014). These models are currently used to compare body burdens with established toxicity thresholds, providing useful measures of the proportions of populations at risk of adverse effects. However, future models should incorporate PBPK bioaccumulation into more elaborate individual-based models or other population-level models of toxic effects.

Traditional approaches to risk assessment and combined stressors are typically descriptive in nature, and while these can explain patterns in collected data and allow statistical analyses of interactions, they provide no information on the underlying processes causing the effects. Furthermore, descriptive methods cannot easily be extrapolated to conditions outside those tested, meaning they cannot inform about new stressors, time-dependent effects, new combinations of stressors, or effects on different endpoints (Jager et al., 2010). It is not feasible to test every exposure scenario given the infinite combination of possible contaminant mixtures combined with other natural and anthropogenic stressors. Biology-based, or process-based, approaches are therefore necessary to describe the physiological mechanisms that underpin the effects of stressors in animals. Bioenergetics models can be useful in this context as these provide a framework to explain animal metabolic organization and mechanistically link stressors to physiological modes of action (Kooijman and Bedaux, 1996). Mechanistic models, such as dynamic energy budget models, therefore explain the impact of stressors through the alteration of energy-costly metabolic processes which are fundamentally linked in an organism via changes in energy allocation. In this way, multiple stressors can be compared on a similar basis (i.e., energy) and predictions can be extrapolated to combined effects.

Regardless of modelling methods used, further knowledge is required on species-specific dose responses for various health endpoints that can be related to population impacts, such as reproduction, fertility and mortality. More challenging but just as interesting, is the need to develop innovative models that can measure impacts of morbidity. The majority of biomarkers described in this assessment indeed influence morbidity, more so than direct mortality/reproduction. The consequences of endocrine or immune dysfunction are difficult to model at the population level because these parameters have not yet been directly linked to demographic parameters which are used in traditional population-matrix models. Here process-based energy budgets can provide exciting opportunities to quantify these often ignored morbidity effects of pollutants by means of energetic trade-offs between important physiological systems. Although most dynamic energy budget models currently focus on lower trophic organisms due to the ease of their study, models do exist for polar bears and North Atlantic right whales (*Eubalaena glacialis*).

The polar bear model has been used to estimate the impact of climate change on bear survival and reproduction as well as measuring the Allee effect (Molnár et al., 2010, 2011, 2014). The right whale dynamic energy budget model incorporates PBPK modeling and bioenergetics to predict individual growth, reproduction, bioaccumulation, and transfer of energy and contaminants from mothers to their young (Klanjscek et al., 2007). Although these are not population-effect models for contaminant exposure, they illustrate how morbidity can affect physiological functions, and indirectly influence population growth. A recent systematic literature review by Bechshøft et al. (2018) investigated the ecological and physiological variables that have been integrated in early POP studies on polar bears and more recent toxicology-based studies, where a total of 207 research papers were published between 1970 and 2016. The frequency with which ecological and physiological variables were integrated into toxicological papers varied. Among the highlights were that age and (or) sex was the only ecological variable(s) considered in 51% of papers. A total of 37% of the papers investigated physiological effects in relation to contaminant concentrations. A total of 98% dealt with contaminant exposure at the individual level, thus leaving population level effects largely unstudied. Body condition was included in 26% of research papers, whereas variables related to polar bear diet were included in less than 9%. From these findings, it was suggested that future polar bear toxicology studies need to increase sample sizes, include more ecological variables, increase studies on family groups, and increase the applicability of studies to management and conservation by examining pollution effects on reproduction and survival.

4.6.3 New ‘omics’ based techniques

The past five years or so have seen advances in approaches and techniques for assessing toxicology and biological effects and different levels of biological organization. As reviewed by Bahamonde et al. (2016), a major area of advancement has been the ‘omics’ revolution, which has generated unprecedented amounts of data on molecular changes that occur in a cell, tissue, or whole organism. For ecotoxicology, these approaches have been used to annotate adverse outcome pathways for environmental contaminants; to characterize chemical modes of action; to investigate mechanisms of organismal adaptation; and to identify candidate biomarkers of exposure and effect. There are numerous examples of studies applying ‘omics’ technologies to study environmental contaminants, and these have most often included gene transcription analysis and more recently next-generation sequencing (Jefferies et al., 2012; Wiseman et al., 2013).

Furthermore, proteomic and metabolomic studies are becoming more prevalent, as they can address post-transcriptional changes associated with contaminant exposure. A new and promising field of research termed exposomics has the daunting task of combining ‘omics’ approaches in order to study cumulative environmental exposures and associated biological responses throughout life (Dennis and Jones, 2016). This study of the exposome is based on the understanding that health and disease are influenced not only by genetics but also by environmental factors, such as chemical exposure.

There only appears to be one published study that has reported on the use of ‘omics’ approaches in Arctic wildlife and fish and in relation to POP or OHC exposure. In a first report for any Arctic species or population, Morris et al. (2018) recently reported on the measurement and profiling of low molecular weight endogenous metabolites (metabolomics) that reflect the dynamic response of biological systems to stress factors. A targeted, quantitative metabolomics platform (219 metabolites including amino acids, biogenic amines, acylcarnitines, phosphatidylcholines, sphingomyelins, hexoses and fatty acids), was evaluated in muscle and liver of polar bears from representative southern and western Hudson Bay (Canada) subpopulations (SHB and WHB, respectively). Multivariate statistics were applied to the data to establish whether bears were discriminated by sex and/or location (subpopulation). Five metabolites discriminated the hepatic profiles of SHB males and females (hexose, arginine, glutamine, one phosphatidylcholine, one sphingomyelin), while the 15 discriminatory metabolites contrasting the livers of males from SHB and WHB were primarily phosphatidylcholines, along with leucine. Metabolite profiles in the muscle of male and female bears could not be differentiated; however muscle samples of SHB and WHB males were again discriminated primarily by membrane lipids. It was concluded that the unique metabolite profiles discriminating the sexes, and particularly those discriminating the subpopulations, may have utility for future assessments of the effects of stressors (including contaminant exposure) on the physiological state of Hudson Bay polar bears and those from other subpopulations. In a follow-up study, Morris et al. (accepted 2018) used these metabolomics profiles and in combination with POPs and total Hg data, and applied multivariate statistics to assess differential influences on the hepatic metabolome of male polar bears from the SHB and WHB. Per- and polyfluoroalkyl substances (PFASs), PBDEs, *p,p'*-DDE and some highly chlorinated *ortho*-PCB congeners were greater in the SHB bears, and inversely correlated with the discriminating metabolites in the liver. Significant elevation of symmetric dimethylarginine (SDMA) in WHB bears, where concentrations of legacy organochlorine pesticides were greater, suggested differences in renal function. The arachidonic acid, glycerophospholipid and amino acid pathways were identified as impacted. Elevated levels of arachidonic acid in the SHB could be related to differences in chronic exposure to hepatotoxic contaminants, since arachidonic acid accompanies the inflammation response. Overall, the results suggested linkages between elements of the hepatic metabolome and differential contaminant exposure in Hudson Bay polar bears.

Goals of ‘omics’ technologies in field-based ecotoxicology are to develop monitoring tools for regulatory purposes. Data from various ‘omics’ approaches as well as from other levels of biological integration, are increasingly integrated and part of Adverse Outcome Pathways (AOP) frameworks. AOP frameworks inherently attempt to provide some estimated extrapolation to individual or ideally population level outcomes (Kramer et al., 2011). In AOP frameworks, molecular initiating events precede adverse cell and tissue responses, and are therefore more sensitive than higher level apical endpoints (such as gonadosomatic index, or fecundity and egg production for birds), in theory allowing for earlier detection of potential adverse effects. Although controlled laboratory experimental conditions determine ‘omics’ responses under various environmental stressors, identifying these responses in natural populations is essential for understanding the cumulative effects of multiple environmental stressors (both chemical and natural) under field conditions, and the relevance of the changes in intact ecosystems.

The recent emphasis on determining ‘omics’ responses in wild populations generates new perspectives on the impacts of contaminants in complex environments, but also detracts from the ability to link cause and effect as a result of these same uncontrolled and poorly defined environmental conditions. Fent and Sumpter (2011) and Hook (2010) recently reviewed the advantages and limitations of omics in ecotoxicology. There was a consensus that although high-throughput analyses offer unsurpassed data generation, there are significant challenges related to reproducibility, variability, and interpretation, especially in non-model species for which few molecular datasets exist. There are necessary steps to move omics from proof-of-concept to a more functional and useful tool that can then be applied to regulatory or traditional environmental monitoring approaches. Thus, researchers are already aware that studies addressing basic modifying factors in the field (i.e., impact of variables such as season, sex, species, and nutritional status on normal biological variation) are required before further discussions regarding the use of ‘omics’ technologies in environmental regulation and monitoring can proceed.

5. Future perspectives

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5.1 Indigenous knowledge holder engagement

Studies on contaminants and human exposure in the Arctic are greatly enhanced by the involvement of indigenous peoples. Within AMAP's work, indigenous knowledge holders and scientific activities are particularly connected through hunting / sample collection, but also more generally through wildlife, environmental and human studies, observation programs, and management – all critical core initiatives of Arctic research. Examples include: (i) the measurement and monitoring of long-term trends and variability in contaminant concentrations in Arctic biota for focal ecosystems; (ii) research on the influence of environmental change on exposure, levels and trends of contaminants in the Arctic environment; (iii) the assessment of human health risks using information on levels and trends of contaminants in harvested traditional country foods; and (iv) the combination of scientific and indigenous knowledge that supports territorial, national and international management initiatives and priorities.

Monitoring and research in focal ecosystems are complementary and contribute to syntheses and integrative studies that further refine our understanding of contaminant cycling in Arctic ecosystems, especially for interactions with other ongoing environmental changes, such as climate change. This is important because the Arctic acts as an early-warning system for other ecosystems and communities that may be adversely affected by future environmental change. Monitoring and research programs need to reflect the recognition that indigenous knowledge holds its own methodology, evaluation, and validation processes. For this reason, it is important to develop processes that include and integrate indigenous knowledge holders throughout the entirety of the programs.

Research scientists and northern communities must cooperate at all levels and with mutual understanding and respect. An example is sample collection, which is governed by annual community harvesting and occurs in accordance with set harvesting quotas by indigenous hunters. Sampling often takes place without the presence of scientific researchers – most are not locally situated and costs would be prohibitive if scientists were to participate in all sampling efforts. Indigenous hunter contributions are therefore vital for optimally successful scientific research initiatives. Here, indigenous knowledge holders use their knowledge to aid in gathering the information identified as needed by scientists. However, indigenous knowledge holders are able to offer so much more than the identification of places to sample and sample collection: they can aid in the optimal understanding of ecological and physiological changes and their connections with humans. Ideally, indigenous knowledge holders would be involved in decisions concerning what information is needed, how it is collected, and how it is used. In some regions a successful and safe field trip relies heavily on indigenous knowledge and the experience of indigenous knowledge holders when investigating wildlife and fish, and long-range transported contaminants and their effects, especially in relation to the ongoing influence of climate change. The contributions of indigenous knowledge holders is already established for national and international sampling programs that have been running for decades and provide the mainstay of research programs such as AMAP activities in Greenland and the Northern Contaminants Program in Canada. Many Arctic communities are participating in effects and monitoring research, including in the Nunatsiavut (Labrador), Nunavik (Quebec), Nunavut and the Inuvialuit Settlement Region of the Northwest Territories (Arctic Council Indigenous Peoples Secretariat, 2017). An indigenous knowledge mapping platform (IK-MAP) has already provided proof-of-concept for sharing the results of community-driven research in near-real time among Inuit communities and stakeholders. These incorporate a wide variety of culturally relevant tools that allow Inuit to interpret results using their own knowledge system (Heath et al., 2015; Arctic Eider Society, 2016). With respect to Greenland, at least five larger towns and villages have consistently participated in effects and monitoring research on contaminants including Thule, Godhavn, Nuuk, Tasiilaq/Kulusuk and Scoresby Sound (Glahder, 1995; Dietz et al., 2001; Sandell et al., 2001; Born et al., 2011).

Indigenous knowledge is based on very complex knowledge systems and includes information on sea ice and climate conditions, animal distribution and behavior, diet and body condition, and many other ecosystem and human society related variables. The combination of both indigenous knowledge and scientific perspectives substantially benefit the interpretation of research results. It is therefore critical that these relationships continue and further evolve to explore how ongoing environmental change, contaminants and disease exposure synergize to impact fish, wildlife, and human health. This requires efforts to continue the inclusion and involvement of indigenous knowledge holders and communities in the necessary field work. Importantly, more effort is needed to actively engage communities in leading research / monitoring programs and helping to shape research agendas that are relevant to healthy communities.

Frameworks to enable a more holistic understanding of the Arctic already exist or are in development. Organizational frameworks such as One Health (which attempts to combine environment, wildlife and human health) may help to bridge the gaps across these human, socio-cultural, animal, and ecosystem domains in the Arctic (Aguirre et al., 2016). An important indigenous framework is a food-security approach as described by the Inuit Circumpolar Council (ICC-Alaska, 2015). The food security framework demonstrates an indigenous knowledge-based holistic understanding of the ecosystem in which humans are part of the ecosystems. Under a food security lens, focus is placed on the relationship between components and understanding the interconnectivity of multiples pieces, such as the connections between biotic, abiotic, social and cultural components (ICC-Alaska, 2015).

5.2 Indigenous concerns and perspectives

Wildlife at the top of Arctic food chains, for example the polar bear (*Ursus maritimus*) and caribou or reindeer (*Rangifer tarandus*) are of great importance to indigenous communities across the Arctic. Therefore, indigenous peoples are very concerned about the potential cumulative and synergistic effects that persistent organic pollutants (POPs) and mercury (Hg) may have with other environmental stressors, and in particular climate change, on wildlife in a given Arctic region. For example, many Inuit are raising concerns about noise and light pollution, which are caused by an increase in Arctic activities, offshore exploration, and shipping activities related to research, tourism, transport and the extraction industry. Hunters have observed very agitated behavior in beluga (*Delphinapterus leucas*) that indicates an effect of stress and are concerned about the ways in which the beluga may be affected (Behe, 2013), such as impacts on well-being and reproductive activity. Coupled with the biological effects of POPs and Hg, low levels of contaminants in the presence of light and noise pollution may lead to a significant decrease in reproduction.

Indigenous knowledge holders from across the circumpolar Arctic have noted concerns with regard to biological effects of POPs and Hg that they observe in their environment and that may result from contamination from both local activities and long-range transported substances. There are several observations by Arctic indigenous peoples relating to biological effects from POPs and Hg but to be fully addressed they require concerted research efforts. These observations include discoloration of blubber, liver and meat of walrus (*Odobenus rosmarus*), seal and fish, change in morphology (e.g., distorted growth of scales) and texture of fish meat and liver, and increased secretion of mucus (e.g., in eels). Moreover, there is concern that people are unable to eat the livers of some species of freshwater fish and freshwater associated mammals for nutrients because they have seen livers that are deformed, inflamed, swollen, or shrunken. There are frequent reports that country foods taste and smell different. In walrus, hunters have observed 'black', hard hearts, and in both walrus and seals patchy skin/fur. In some bird species, indigenous harvesters have noticed smaller clutches, eggshell thinning in some areas, and chicks not surviving incubation. Finally, there is the widespread observation that animals in general are physically more lean.

Many of the observed effects in wildlife may relate to diseases that could be affected by a deprived immune response from exposure to high contaminant loads. Indigenous peoples have also expressed concern about the sources of POPs and Hg that may lead to biological effects, and thus would like to see research prioritize questions around the sources and biological effects of POPs and Hg in relation to the following. (1) There are concerns about pollution sourced from the exhaust, ballast water and sewage routinely discharged from ships, which are probable sources of POPs and Hg to the receiving marine environment. (2) There is a need for short- and long-term information about the health effects and impacts on the environment and animals around and within the extraction industry areas. The long-term consequences and risks of pollution caused by mining activities have been underestimated by mining companies, such as at the Blaiken mine in Sweden, south of Sorsele. Now the environment, its inhabitants and its water systems are experiencing constant pollution at this Swedish site. Saami living near the mining sites are seeing different effects on the reindeer and in the environment in those impacted areas. (3) Ingestion of plastics debris and other particulate pollution is increasingly being reported for marine species, including seabirds and marine mammals. Most of the work on ingested debris in Arctic waters has focused to date on marine bird species. Although there are only a few reports of ingested debris and plastics by Inuit hunters, community members are becoming increasingly aware of these phenomena through collaboration with research programs, Hunters and Trappers Organizations and in Canada through the Nunavut Arctic College's Environmental Technology program in Iqaluit. When plastic ingestion is reported to northern communities, most of the questions concern potential effects on the health of seabirds, including questions about the potential for contaminants ingested with the plastics to be transferred to eggs and meat. Some hunters have shared stories of friends or family who have found plastic pollution such as plastic shopping bags, inside the stomachs of marine mammals, but such observations are rare events. (4) Use of flame retardants is a big concern and particularly from combatting wildfires. (5) POPs and Hg contamination derived from inadequate waste management includes pollutants from batteries, locally generated waste, breakdowns and leaks from industrial equipment, and potential leakage from sanitation sites.

Future research is urgently needed to address concerns voiced by indigenous knowledge holders and peoples, and to investigate, through a co-produced knowledge-based approach, the details of observed effects as well as necessary mitigation actions, and the implications for policy development and resource management.

5.3 Use of archived samples

Although ecological change will influence contaminant pathways, exposure remains a function of physical, chemical and biological factors, as well as source emissions (Macdonald et al., 2003). However, isolating the key factors impacting contaminant time trends is difficult. This is especially true for legacy POPs for which secondary emissions and other environmental drivers play an increasingly important role (UNEP/AMAP, 2011). Nevertheless, time series data are used to assess the effectiveness of international restrictions on emissions (de Wit et al., 2006). Thus, failure to consider other ecosystem changes that may contribute to increasing or decreasing exposure could lead to erroneous interpretations (Hebert and Weseloh, 2006). There has been some success in examining the impacts of changing physical and chemical factors (Ma et al., 2011) and ecological factors (McKinney et al., 2015). As summarized in the present assessment, altered contaminant exposures linked to global climate change have been documented in a limited number of polar bear and ringed seal populations. These studies are the result of successful long-term contaminant monitoring projects and such monitoring should remain a priority, especially as the trends in sea-ice loss are projected to continue (Perovich and Richter-Menge, 2009). Long-term climate data are also available, generally for at least as long or longer periods of time. In contrast, ecological change data are far more difficult to obtain (Wassmann et al., 2011). Developing new and refining existing ecological tracers and biomarkers that can be measured in archived tissues with paired contaminant measurements will be an important part of these ongoing studies.

Archiving tissue samples for contaminant and other types of analysis is a complicated issue. How to sample, how much to sample and how to store samples will always depend on what kind of analyses are planned. The examples listed here constitute a representative sampling of the most commonly used procedures. For metals, samples are often stored in metal-free polyethylene plastic bags in a -20°C freezer. For POPs, samples are often wrapped in aluminum foil and kept in airproof glass containers. If the samples are to be analyzed for fatty acids, then storage in airproof glass containers at -80°C is preferred. Tissue for histopathological investigations is usually stored in formaldehyde and if shipped in a freezer, the formaldehyde must be mixed with alcohol to avoid freezing damage of the tissue. Blood sampling requires vacutainer tubes and usually requires centrifugation and additional treatment, which may be a problem under field conditions. Here, new techniques such as the use of dried blood spots could have potential, as was recently shown for Hg (Basu et al., 2017). In other cases, storage in liquid nitrogen is required to keep cells alive for subsequent laboratory analysis, which can be a challenge during shipment and for longer term storage.

In the previous AMAP effects report (Letcher et al., 2010) it was recommended that there should be more harmonization of sample collection and of the measurements made (e.g., biomarkers) to assess changes in effects/biological endpoints. A major point was the potential for using sample archives more effectively as sources for materials to assess metabolites, nutrients and biomarkers. Also, that scientists should indicate tissue types and storage conditions for use in assessing contaminant impacts (e.g., histology, serum chemistries), and not just archiving in specimen banks safeguarded for future chemical analyses. However, there have been no harmonization efforts since the last effects assessment. Such efforts would almost certainly require an international project with focus on these issues. One of the challenges in long-term projects of this type is that the spin-off will not be evident for some time, which makes funding for such initiatives difficult to obtain. Similar problems arise for the scientists involved in such projects because their interest in participation depends on results gained and the subsequent publication of these results.

5.4 Linking wildlife, human, and ecosystem health

Rapid population growth and enormous urban and coastal development in many of the world's coastal regions have resulted in a major concern that anthropogenic pollution may reduce the biodiversity and productivity of marine ecosystems, and so reduce and deplete human marine food resources. Natural environments are also important for recreation, and thus for human health and welfare, and there is now increased awareness that nature has its own intrinsic value. Pollution reduces the aesthetic value and perhaps also the intrinsic value of the marine environment, whether the pollution is visual (such as oil pollution and plastic debris) or invisible (such as chemical contaminants). Another reason for concern about marine pollution relates to the direct effects of pollution on human health. Because many pollutants accumulate in marine organisms, humans are exposed to pollutants when they consume marine food from polluted areas. Several studies have shown that human populations consuming large amounts of marine food have high body burdens of POPs, such as polychlorinated dibenzo-*p*-dioxins (PCDDs) and dibenzofurans (PCDFs), polychlorinated biphenyls (PCBs) and some heavy metals including Hg. There has been a particular focus on indigenous people who consume large amounts of marine food, including the blubber of marine mammals (Dewailly et al., 1999; Donaldson et al., 2010). Effects related to contaminants are not evident immediately after intake but will usually appear after years of exposure. Thus, a cause-effect relationship is often hard to establish for contaminants, in contrast to diseases from pathological sources such as zoonotics. As previously mentioned, such diseases may be linked to contaminants through depression of the immune system. Exposure of marine organisms to increasing concentrations of human bacteria, antibiotics and other pharmaceuticals and their metabolites may also pose a threat to coastal ecosystems.

There is also concern about chemicals with endocrine-disrupting properties. One such example is tributyltin, an ingredient of antifouling paints applied to marine shipping. This has been shown to cause imposex in gastropods, to affect coastal and estuarine mollusk populations, and to reduce species diversity in estuarine benthic and epibenthic invertebrate communities (Matthiessen and Law, 2002). Until now, most studies on the effects of marine pollution have focused on biochemical and physiological effects. In the future, studies should address the effects of pollution on behavioral traits that can potentially alter biodiversity and ecosystem functioning. For humans, recent reports have documented relationships between Hg, PCDDs, PCDFs and PCBs, as well as correlative relationships with several reproductive, cognitive, and neurological biomarkers. Breastfeeding and the quality of the home environment are examples of confounding factors that may modify or counteract harmful effects of POPs (Walkowiak et al., 2001; Jacobson and Jacobson, 2002). Many marine mammals are highly dependent on well-developed cognitive abilities and normal behavior to survive. Selection against cognitive and neurological dysfunction or retardation is likely to be far more significant in wildlife than in humans. This indicates a need for wildlife studies that focus on the effects of marine pollution on cognitive abilities and related neurological effects, and on how such effects could impact on biodiversity and ecosystem functioning.

The harmful effects of POPs on human and environmental health have been recognized for some time. In 2004, an international ban (Stockholm Convention on POPs) was implemented on the 12 most noxious POPs, the so-called 'dirty dozen' (Kaiser and Enserink, 2000). The number of POPs listed under the Stockholm Convention has continued to grow ever since as there is a plethora of compounds exhibiting POP-like characteristics leaching into the environment, all potentially ending up in the Arctic environment. In terms of Hg, the Minamata Convention entered into force worldwide in 2017. Every year thousands of new synthetic chemicals are produced, and there have been reports that some of these are chemicals of emerging concern for humans and marine biota (see AMAP 2017a,b).

6. Synthesis and knowledge gaps

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6.1 New information since the previous AMAP assessment

The previous AMAP effects report on persistent organic pollutants (POPs) provided an in-depth review of how organohalogen compounds (OHCs) affect health in various species of Arctic wildlife and fish for which published data were available up until 2009 (Letcher et al., 2010). The most recent AMAP mercury (Hg) effects report provided similar insights specific to Hg exposure (Dietz et al., 2013a), and included information on Hg transport pathways, geographic trends, linkage to climate change and human health (AMAP, 2011). Since 2010, new information has become available for both OHCs and Hg.

In terms of POPs, effects studies on wildlife and fish have largely focused on legacy OHCs and in particular on summed levels for different classes of chemical (polychlorinated biphenyls, ΣPCBs; dichlorodiphenyltrichloroethanes, ΣDDTs; chlordanes, ΣCHLs; hexachlorocyclohexanes, ΣHCHs) rather than individual compounds or congeners. More recently, polybrominated diphenyl ethers (ΣPBDEs) and poly- and perfluoroalkyl substances (ΣPFASs) have received increasing attention, although information on the latter is almost completely focused on perfluoroalkyl acids (PFAAs) and specifically on the highly bioaccumulative perfluorooctane sulfonic acid (PFOS).

Common among all Arctic wildlife and fish studies since 2010 is that a number of different groups of biomarker and/or effect endpoints have been consistently measured in wildlife: hormones (steroid and thyroid), vitamins (especially vitamins A and E), immune system activation (antibodies or cell proliferation) and liver enzyme activity (CYP-450), in correlative relation to contaminant levels in tissues or body compartments (blood or plasma). Histopathology has also been studied for some species. As summarized in Table 3.1, the increase or decrease in levels and expression of these biomarker endpoints in association with levels of PCB and/or Hg varies across phyla, species, populations and regions.

CYP-450 enzyme activity is reported to have increased with OHC concentrations across all studies since 2010. In contrast, concentrations of hormones and vitamins generally showed negative associations with OHCs although this varied depending on species and tissue/body compartment. The immune system was also negatively correlated with OHC levels across species, and this included both the humoral and cellular systems. Novel biomarkers of reactive oxygen species and genotoxicity have recently been employed in a few species of mammals and birds. Histopathological changes have been shown to occur in liver, kidney and thyroid glands and seem to be affected negatively by OHC and Hg concentrations; this is also the case for bone mineral density. Interpretation of these effects on the population level, however, remains a challenge. New studies on polar bear brains have shown steroid hormones to be positively correlated with PFAS levels, while neurochemical endpoints were negatively correlated with levels of the same PFAS compounds. Polar bear (*Ursus maritimus*), pilot whale (*Globicephala melas*), beluga (*Delphinapterus leucas*) and glaucous gull (*Larus hyperboreus*) are the four species analyzed for a broader range of biomarkers. Across these species, thyroid hormones generally decreased while vitamins A and E increased in liver and decreased in blood. Concentrations of steroid hormones did not show a clear association with OHCs, although there was a tendency for testosterone to be negatively correlated with PCB levels. Of the recently applied biomarkers, genotoxicity has been shown for polar bears and for a few birds and fish species, and mainly driven by PCB exposures.

Specific effects associated with (combined) contaminant exposure have been reported in several Arctic wildlife species and populations. Based on PCB concentrations being the dominant effect contributor, a conservative PCB critical body residue (CBR) of 10 µg/g lw has been used to calculate risk quotients (RQs) for a number of species in the Arctic region and bordering sea areas, and where sufficient published data were available. Killer whales (*Orcinus orca*) were found to be the marine mammal species most at risk, with a large proportion of individuals in all five study populations having an RQ above 1, and thus implying risk of PCB-mediated health effects. This is also the case for transient Pacific killer whales just south of the Arctic region. These findings are in agreement with recently published information for killer whale populations at lower latitudes (from the UK, Ireland, and further south at the Canary Islands and Gibraltar), where it has been indicated that killer whales are no longer reproducing at levels necessary to support viable populations, with the prospect of severe consequences in the future for these populations. Long-finned pilot whales from the Faroe Islands also had large group proportions with RQs above 1. The finding that toothed whales other than killer whales can exceed the PCB toxicity thresholds has also been documented in lower latitude species, such as striped dolphin (*Stenella coeruleoalba*) and bottlenose dolphin (*Tursiops truncatus*). In contrast, beluga and ringed seals (*Pusa hispida*) from all circumpolar populations studied all seem to have RQs below 1, indicating that the immune and hormonal systems of these Arctic species are unlikely to be detrimentally affected by PCB exposure. Several populations of birds of prey, however, do seem to be at risk of PCB-mediated biological health effects: White-tailed eagle (*Haliaeetus albicilla*), gyrfalcon (*Falco rusticolus*) and peregrine falcon (*F. peregrinus*) all had a large proportion of individuals with RQs above 1.

In general, based on the most recently published information, most marine mammal species are at no or low risk for health effects mediated by Hg exposure. However, for some species at high marine trophic levels, such as polar bear, pilot whale, narwhal (*Monodon monoceros*) and beluga, as well as hooded seal (*Cystophora cristata*), a proportion of the population is at high or severe risk for health effects mediated by Hg exposure. Polar bears from the central Canadian High Arctic and Alaska, as well as pilot whales from the Faroe Islands, carried the hepatic total Hg burdens of greatest concern.

Bird Hg concentrations were above toxicity benchmarks in many areas of the marine environment. This was particularly the case for seabirds, including northern fulmar (*Fulmarus glacialis*), double-crested cormorant (*Phalacrocorax auritus*) and pigeon guillemot (*Cepphus columba*). Freshwater and terrestrial species in the Alaskan environment including northern shoveler (*Anas clypeata*), common loon (*Gavia immer*), peregrine falcon, red-throated loon (*G. stellate*), Pacific loon (*G. pacifica*), bald eagle (*Haliaeetus leucocephalus*), green-winged teal (*A. carolinensis*) and yellow-billed loon (*G. adamsii*) also exhibited Hg concentrations above effect thresholds. In the North Atlantic, only a small proportion of unhatched chicks of black guillemot (*Cepphus grylle*) were at high risk of Hg exposure. As expected, terrestrial mammals, with the exception of Arctic foxes (*Vulpes lagopus*) on Iceland, which showed low to moderate risk, did not show risk for Hg intoxication, based on the limited recent Hg data available. There is a lack of Hg data for many Arctic species and regions, including the Russian Arctic. In addition, in some studies, the number of individuals of a given species obtained was not sufficient to allow conclusions to be drawn regarding the proportions of the population falling within a given health risk category.

6.2 Knowledge gaps and recommendations

New information from effects studies on Arctic species has resulted in important new insights into the risks for biological effects on Arctic wildlife from exposure to environmental contaminants. However, major knowledge gaps remain with regard to the biological and toxicological effects of OHCs, Hg and, in particular, contaminants of emerging Arctic concern that are now being reported in Arctic biota. These knowledge gaps include concentration thresholds for biologically relevant health effects and toxic threshold endpoints used to develop RQs. Integrating wildlife and human health assessments has the potential to improve knowledge regarding the combined effects of contaminant exposure and natural stressors (e.g., infectious and zoonotic diseases), and how multiple stressors are directly and indirectly influenced in a changing Arctic, especially in connection with climate change. There is a need for new methods and approaches, including the expansion of *in vitro* experimental approaches, and studies on the individual and combined effects of single compounds and chemical mixtures. Increased use of *in silico* modeling is needed to better predict biological effects at the population and ecosystem level. Specific knowledge gaps and information deficiencies are as follows.

Improving future monitoring programs

- Certain Arctic regions, such as Russia, Fennoscandia and Alaska, are not adequately represented in the monitoring of wildlife and fish exposure and effects, despite relevant studies in the Russian Arctic in particular having been called for in all previous AMAP assessments since 1998.
- Greater attention needs to be given to access to data on hunting quotas and takes for relevant species.
- There are a number of invasive and migratory wildlife and fish species, including killer whale and capelin (*Mallotus villosus*), which are not represented or are under-represented in Arctic monitoring and research addressing biotic changes due to the changing Arctic climate.
- Further pan-Arctic harmonization is required in relation to target species, sampling frequency and season, and methods applied for the measurement of contaminants and associated biomarkers and biological endpoints that are applicable to effects assessment. In doing so, there is a need for increased communication and collaboration with local and indigenous people.

Improving effect assessments

- Effects need to be assessed within a changing Arctic and multi-stressor framework.
- From a policy perspective, projections regarding future OHC- and Hg-elicited effects on wildlife and fish populations should be improved using growing data compilations on OHC and Hg exposure levels and trends, and in relation to observed climate change-related variables (e.g., sea ice loss). This information should be incorporated within, for example, scenario assessments performed by AMAP and those summarized by the Intergovernmental Panel on Climate Change.
- There is an ongoing need to establish and refine concentration thresholds for biologically relevant health effects in (Arctic) wildlife and fish. Such studies are often expensive so international collaboration is encouraged.
- Physical-chemical and industrial production data are required for newer chemicals of emerging Arctic concern.

Multidisciplinary studies

- Effects need to be assessed in relation to spatial and temporal variation in dietary pathways of exposure. There is also a need to assess the combined effects of contaminant exposure and natural stressors.
- The 'One Health' concept should be explored in future assessments integrating information from wildlife and human health studies.
- There is a need for better identification of cumulative and interactive effects thresholds of contaminant exposure.

Specific recommendations concerning possible improvements and future directions for this new area of global climate change-linked ecotoxicology, include (i) routine analysis of ancillary ecological metrics together with OHC and Hg studies, (ii) simultaneous consideration of the multiple mechanisms by which global climate change and contaminant interactions can occur, (iii) targeted research on species known to be sensitive to both global climate change and contaminants, and (iv) studies linking these changes to changes in major impact parameters such as immune and reproductive function and development, particularly at the population level. Moving forward, environmental chemists studying contaminant levels in biotic and abiotic media, eco(toxico)logists and indigenous peoples of the Arctic should work together to consider the combined impacts of these changes on contaminant exposures in Arctic marine and terrestrial biota.

References

Personal communications

Aars, J., Norwegian Polar Institute, Tromsø, Norway

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Treu, G., Leibniz Institute for Zoo and Wildlife Research, Berlin, Germany

General references

Aas, E., B. Liewenborg, B.E. Grosvik, L. Camus, G. Jonsson, J.F. Borseth and L. Balk, 2003. DNA adduct levels in fish from pristine areas are not detectable or low when analyzed using the nuclease P1 version of the P-32-postlabelling technique. Biomarkers, 8:445-460.

Abbas, A.K., A.H. Lichtman and S. Pillai, 2012. Basic Immunology: Functions and Disorders on the Immune System. Fourth edition. Elsevier.

Ackerman, J.T., C.A. Eagles-Smith, M.P. Herzog, C.A. Hartman, S.H. Peterson, D.C. Evers, A.K. Jackson, J.E. Elliott, S.S. Vander Pol and C.E. Bryan, 2016. Avian mercury exposure and toxicological risk across western North America: A synthesis. Science of the Total Environment, 568:749-769.

Aguirre, A., V. Beasley, T. Augspurger, W.H. Benson, J. Whaley and N. Basu, 2016. One Health-Transdisciplinary opportunities for SETAC leadership in integrating and improving the health of people, animals, and the environment. Environmental Toxicology and Chemistry, 35:2383-2391.

Ahern, M., R. Kovats, P. Wilkinson, R. Few and F. Matthies, 2005. Global health impacts of floods: epidemiologic evidence. Epidemiologic Reviews, 27:36-46.

Al-Saleh, I., F. Khogali, M. Al-Amodi, I. El-Dousch, N. Shinwari and R. Al-Baradei, 2003. Histopathological effects of mercury in skin-lightening cream. Journal of Environmental Pathology, Toxicology and Oncology, 22:287299 doi:10.1615/JEnvPathToxOncol.v22.i4.30.

AMAP, 1998. Arctic Pollution Issues. Arctic Monitoring and Assessment Programme (AMAP), Oslo, Norway.

AMAP, 2004. AMAP Assessment 2002: Persistent Organic Pollutants in the Arctic. Arctic Monitoring and Assessment Programme (AMAP), Oslo, Norway.

AMAP, 2011. AMAP Assessment 2011: Mercury in the Arctic. Arctic Monitoring and Assessment Programme (AMAP), Oslo, Norway.

AMAP, 2016. AMAP Assessment 2015: Temporal Trends in Persistent Organic Pollutants in the Arctic. Arctic Monitoring and Assessment Programme (AMAP), Oslo, Norway.

AMAP, 2017a. Chemicals of Emerging Arctic Concern. Summary for Policy-makers. Arctic Monitoring and Assessment Programme (AMAP), Oslo, Norway. 16pp.

AMAP, 2017b. AMAP Assessment 2016: Chemicals of Emerging Arctic Concern. Arctic Monitoring and Assessment Programme (AMAP), Oslo, Norway. xvi+353pp.

Andersen, M.S., E. Fuglie, M. Konig, I. Lipasti, A.O. Pedersen, A. Polder, N.G. Yoccoz and H. Routti, 2015. Levels and temporal trends of persistent organic pollutants (POPs) in arctic foxes (*Vulpes lagopus*) from Svalbard in relation to dietary habits and food availability. Science of the Total Environment, 511:112-122.

Andrews, J.E., 1989. Polychlorinated biphenyl (Aroclor 1254) induced changes in femur morphology calcium metabolism and nephrotoxicity. Toxicology, 57:83-96.

Arctic Council Indigenous Peoples Secretariat, 2017. Persistent toxics. Accessed at http://www.arcticpeoples.org/persistent-toxics on 21 March 2017.

Arctic Eider Society, 2016. 2015-2016 Annual Report. 38pp. https://arcticeider.com/en/about

Armitage, J.M., C.L. Quinn and F. Wania, 2011. Global climate change and contaminants: An overview of opportunities and priorities for modelling the potential implications for long-term human exposure to organic compounds in the Arctic. Journal of Environmental Monitoring, 13:1532-1546.

Atwood, T.C., C. Duncan, K.A. Patyk, P. Nol, J. Rhyan, M. McCollum, M.A. McKinney, A.M. Ramey, C.K. Cerqueira-Cezar, O.C.H. Kwok, J.P. Dubey and S. Hennager, 2017. Environmental and behavioral changes may influence the exposure of an Arctic apex predator to pathogens and contaminants. Scientific Reports, 7:Article 13193.

Bahamonde, P.A., M.E. McMaster, M.R. Servos, C.J. Martynuik and K.R. Munkittrick, 2016. Characterizing transcriptional networks in male rainbow darter (*Etheostoma caeruleum*) that regulate testis development over a complete reproductive cycle. Plos ONE, 11:e0164722.

Balk, L., K. Hylland, T. Hansson, M.H.G. Berntssen, J. Beyer, G. Jonsson, A. Melbye, M. Grung, B.E. Torstensen, J.F. Borseth, H. Skarphedinsdottir and J. Klungsoyr, 2011. Biomarkers in natural fish populations indicate adverse biological effects of offshore oil production. Plos ONE, 6:e19735.

Ball, M.C., M.W. Lankester and S.P. Mahoney, 2001. Factors affecting the distribution and transmission of *Elaphostrongylus rangiferi* (Protostrongylidae) in caribou (*Rangifer tarandus caribou*) of Newfoundland, Canada. Canadian Journal of Zoology, 79:1265-1277.

Barst, B.D., M. Rosabal, P.G.C. Campbell, D.G.C. Muir, X. Wang, G. Köck and P.E. Drevnick, 2016. Sub-cellular distribution of trace elements and liver histology of landlocked Arctic char (*Salvelinus alpinus*) sampled along a mercury contamination gradient. Environmental Pollution, 212:574-583.

Basu, N., 2012. Piscivorous mammalian wildlife as sentinels of methylmercury exposure and neurotoxicity in humans. Current Topics in Neurotoxicity, 2:357-370.

Basu, N., 2015. Applications and implications of neurochemical biomarkers in environmental toxicology. Environmental Toxicology and Chemistry, 34:22-29.

Basu, N. and J. Head, 2010. Mammalian wildlife as complementary models in environmental neurotoxicology. Neurotoxicology and Teratology, 32:114-119.

Basu, N., M. Kwan and H.M. Chan, 2006a. Mercury but not organochlorines inhibits muscarinic cholinergic receptor binding in the cerebrum of ringed seals (*Phoca hispida*). Journal of Toxicology and Environmental Health A, 69:1133-1143.

Basu, N., A.M. Scheuhammer, K. Rouvinen-Watt, N. Grochowina, K. Klenavic, R.D. Evans and H.M. Chan, 2006b. Methylmercury impairs components of the cholinergic system in captive mink (*Mustela vison*). Toxicological Sciences, 91:202-209.

Basu, N., A.M. Scheuhammer, K. Rouvinen-Watt, N.M. Grochowina, R.D. Evans, M. O'Brien and H.M. Chan, 2007. Decreased N-methyl-D-aspartic acid (NMDA) receptor levels are associated with mercury exposure in wild and captive mink. NeuroToxicology, 28:587-593.

Basu, N., A.M. Scheuhammer, C. Sonne, R. Dietz and R.J. Letcher, 2009. Is mercury in the environment of neurotoxic concern to polar bears? Environmental Toxicology and Chemistry, 28:133-140.

Basu, N., J. Eng, M. Perkins, A. Santa-Rios, G. Martincevic, K. Carlson and R.L. Neitzel, 2017. Development and application of a novel method to characterize methylmercury exposure in newborns using dried blood spots. Environmental Research, 159:276-282.

Bechshøft, T.Ø., C. Sonne, R. Dietz, E.W. Born, M.A. Novak, E. Henchey and J.R. Meyer, 2011. Cortisol levels in hair of East Greenland polar bears (*Ursus maritimus*). Science of the Total Environment, 409:831-834.

Bechshøft, T.Ø., F.F. Rigét, C. Sonne, R.J. Letcher, D.C.G. Muir, M.A. Novak, E. Henchey, J.S. Meyer, I. Eulaers, V. Jaspers, A. Covaci and R. Dietz, 2012a. Measuring environmental stress in East Greenland polar bears, 1892-2009: what does hair cortisol tell us? Environment International, 45:15-21.

Bechshøft, T.Ø., C. Sonne, R. Dietz, E.W. Born, D.C.G. Muir, R.J. Letcher, M.A. Novak, E. Henchey, J.S. Meyer, B.M. Jenssen and G.D. Villanger, 2012b. Associations between complex OHC mixtures and thyroid and cortisol hormone levels in East Greenland polar bears. Environmental Research, 116:26-35.

Bechshøft, T.Ø., C. Sonne, F.F. Rigét, R.J. Letcher, M.A. Novak, E. Henchey, J.S. Meyer, I. Eulaers, V. Jaspers, A. Covaci and R. Dietz, 2013. Polar bear stress hormone cortisol fluctuates with the North Atlantic Oscillation climate index. Polar Biology, 36:1525-1529.

Bechshøft, T.Ø., A.E. Derocher, E. Richardson, P. Mislan, N.J. Lunn, C. Sonne, R. Dietz, D. Janz and V. St. Louis, 2015. Mercury and cortisol in hair from Western Hudson Bay polar bears. Ecotoxicology, 24:1315-1321.

Bechshøft, T.Ø., C. Sonne, J. Jakobsen, F.F. Rigét, E.W. Born, R.J. Letcher, B.M. Jenssen and R. Dietz, 2016. Vitamins A and E in liver, kidney, and whole blood of East Greenland polar bears sampled 1994-2008: Reference values and temporal trends. Polar Biology, 39:743-754.

Bechshøft, T.Ø., A.E. Derocher, M. Viengkone, H. Routti, J. Aars, R.J. Letcher, R. Dietz, C. Sonne, B.M. Jenssen, E. Richardson and N.J. Lunn, 2018. On the integration of ecological and physiological variables in polar bear toxicology research: a systematic review. Environmental Reviews, 26:1-12.

Beck, S., A.D. Foote, S. Kötter, O. Harries, L. Mandleberg, P.T. Stevick, P. Whooley and J.W. Durban, 2013. Using opportunistic photo-identifications to detect a population decline of killer whales (*Orcinus orca*) in British and Irish waters. Journal of the Marine Biological Association of the UK, 94:1327-1333.

Behe, C., 2013. Inuit Circumpolar Council-Alaska North Slope Regional Food Security Workshop: How to Assess Food Security from an Inuit Perspective. Building a Conceptual Framework on How to Asses Food Security in the Alaskan Arctic. 2013. http://iccalaska.org/wp-icc/wp-content/uploads/2016/03/North-Slope-Regional-Food-Security-Workshop_012914.pdf Accessed 05/03/2016.

Beyer, J., K. Petersen, y. Song, A. Ruus, M. Grung, T. Bakke and K.E. Tollefsen, 2014. Environmental risk assessment of combined effects in aquatic ecotoxicology: a discussion paper. Marine Environmental Research, 96:81-91.

BirdLife International, 2018. Data Zone - Bjørnøya (Bear Island). datazone.birdlife.org/site/factsheet/bjørnøya-(bear-island)-iba-svalbard-and-jan-mayen-islands-(to-norway)

Blevin, P., S. Tartu, F. Angelier, S. Leclaire, J.-O. Bustnes, B. Moe, D. Herzke, G.W. Gabrielsen and O. Chastel, 2014. Integument colouration in relation to persistent organic pollutants and body condition in Arctic breeding black-legged kittiwakes (*Rissa tridactyla*). Science of the Total Environment, 470:248-254.

Blomhoff, R., 1994. Transport and metabolism of vitamin A. Nutrition Reviews, 52:S13-23.

Bogomolni, A., S. Frasca Jr, M. Levin, K. Matassa, O. Nielsen, G. Waring and S. De Guise, 2016. In vitro exposure of harbor seal immune cells to Aroclor 1260 alters phocine distemper virus replication. Archives of Environmental Contamination and Toxicology, 70:121-132.

Born, E.W., A. Heilmann, L. Kielsen Holm and K.L. Laidre, 2011. Polar bears in Northwest Greenland: An interview survey about the catch and the climate. Museum Tusculanum Press. 232 pp.

Bourgeon, S., E.H.K. Leat, E. Magnusdottir, A.T. Fisk, R.W. Furness, H. Strøm, S.A. Hanssen, A. Petersen, K. Olafsdottir, K. Borgå, G.W. Gabrielsen ajnd J.O. Bustnes, 2012. Individual variation in biomarkers of health: Influence of persistent organic pollutants in great skuas (*Stercorarius skua*) breeding at different geographical locations. Environmental Research, 118:31-39.

Bowen, W.D. and S.J. Iverson, 2013. Methods of estimating marine mammal diets: a review of validation experiments and sources of bias and uncertainty. Marine Mammal Science, 29:719-754.

Braune, B., 2015. Temporal trends of contaminants in Arctic seabird eggs. In: Synopsis of Research Conducted Under the 2014-2015 Northern Contaminants Program. Aboriginal Affairs and Northern Development Canada, pp. 229-238.

Braune, B.M., P.M. Outridge, A.T. Fisk, D.C.G. 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, 2005. Persistent organic pollutants and mercury in marine biota of the Canadian Arctic: An overview of spatial and temporal trends. Science of the Total Environment, 351-352:4-56.

Braune, B.M., S. Trudeau, D.A. Jeffrey and M.L. Mallory, 2011. Biomarker responses associated with halogenated organic contaminants in northern fulmars (*Fulmarus glacialis*) breeding in the Canadian Arctic. Environmental Pollution, 159:2891-2898.

Braune, B.M., A.M. Scheuhammer, D. Crump, S. Jones, E. Porter and D. Bond, 2012. Toxicity of methylmercury injected into eggs of thick-billed murres and arctic terns. Ecotoxicology, 21:2143-2152.

Braune, B.M., A.J. Gaston, K.A. Hobson, H.G. Gilchrist and M.L. Mallory, 2014. Changes in food web structure alter trends of mercury uptake at two seabird colonies in the Canadian Arctic. Environmental Science and Technology, 48:13246-13252.

Braune, B.M., A.J. Gaston, K.A. Hobson, H. Grant Gilchrist and M.L. Mallory, 2015. Changes in trophic position affect rates of contaminant decline at two seabird colonies in the Canadian Arctic. Ecotoxicology and Environmental Safety, 115:7-13.

Braune, B.M., A.J. Gaston and M.L. Mallory, 2016. Temporal trends of mercury in eggs of five sympatrically breeding seabird species in the Canadian Arctic. Environmental Pollution 214:124-131.

Brown, T.M., P.S. Ross, K.J. Reimer, N. Veldhoen, N.J. Danger, A.T. Fisk and C.C. Helbing, 2014. PCB related effects thresholds as derived through gene transcript profiles in locally contaminated ringed seals (*Pusa hispida*). Environmental Science and Technology, 48:12952-12961.

Buckman, A.H., N. Veldhoen, G. Ellis, J.K.B. Ford, C.C. Helbing and P.S. Ross, 2011. PCB-Associated changes in mRNA expression in killer whales (*Orcinus orca*) from the NE Pacific Ocean. Environmental Science and Technology, 45:10194-10202.

Buntin, J.D., 1996. Neural and hormonal control of parental behaviour in birds. In: Rosenblatt, J.S. and C.T. Snowdon (eds.), Advances in the Study of Behavior. Volume 25. Academic Press.

Burek, K.A., F.M.D. Gulland and T.M. O'Hara, 2008. Effects of climate change on Arctic marine mammal health. Ecological Applications, 18:S126-S134.

Bustnes, J.-O., S.A. Hanssen, I. Folstad, K.E. Erikstad, D. Hasselquist and J.-U. Skaare, 2004. Immune function and organochlorine pollutants in arctic breeding glaucous gull. Archives of Environmental Contamination and Toxicology, 47:530-541.

Bustnes, J.-O., K.E. Erikstad, S.A. Hanssen, T. Tveraa, I. Folstad and J.U. Skaare, 2006. Anti-parasite treatment removes negative effects of environmental pollutants on reproduction in an Arctic seabird. Proceedings of the Royal Society B, 273:3117-3122.

Bustnes, J.-O., G.W. Gabrielsen and J. Verreault, 2010. Climate variability and temporal trends of persistent organic pollutants in the Arctic: A study of glaucous gulls. Environmental Science and Technology, 44:3155-3161.

Bustnes, J.-O., B. Moe, S.A. Hanssen, D. Herzke, A.A. Fenstad, T. Nordstad, K. Borga and G.W. Gabrielsen, 2012. Temporal dynamics of circulating persistent organic pollutants in a fasting seabird under different environmental conditions. Environmental Science and Technology, 46:10287-10294.

Bytingsvik, J., E. Simon, P.E.G. Leonards, M. Lamoree, E. Lie, J. Aars, A.E. Derocher, Ø Wiig, B.M. Jensen and T. Hamers, 2013. Transthyretin-binding activity of contaminants in blood from polar bear (*Ursus maritimus*) cubs. Environmental Science and Technology, 47:4778-4786.

Cahill, T.M., I. Cousins and D. Mackay, 2003. Development and application of generalized physiologically-based pharmacokinetic model for multiple environmental contaminants. Environmental Toxicology and Chemistry, 22:26-34.

Candido, M.V., L.C.C. Silva, J. Moura, T.D.M.M. Bona, F. Montiani-Ferreira and E. Santin, 2011. Comparison of clinical parameters in captive Cracidae fed traditional and extruded diets. Journal of Zoo and Wildlife Medicine, 42:437-443.

Chastel, O., A. Lacroix, H. Weimerskirch and G.W. Garielsen, 2005. Modulation of prolactin but not corticosterone responses to stress in relation to parental effort in a long-lived bird. Hormones and Behaviour, 47:459-466.

Cherry, S.G., A.E. Derocher, I. Stirling and E.S. Richardson, 2009. Fasting physiology of polar bears in relation to environmental change and breeding behavior in the Beaufort Sea. Polar Biology, 32:383-391.

Chetelat, J. and B. Braune (Eds.), 2012. Canadian Arctic Contaminants Assessment Report III: mercury in Canada's North. Northern Contaminants Program. Aboriginal Affairs and Northern Development Canada.

Chow, B.A., J. Hamilton, M.R. Cattet, G. Stenhouse, M.E. Obbard and M.M. Vijayan, 2011. Serum corticosteroid binding globulin expression is modulated by fasting in polar bears (*Ursus maritimus*). Comparative Biochemistry and Physiology A, 158:111-115.

Colborn, T., F.S.V. Saal and A.M. Soto, 1993. Developmental effects of endocrine-disrupting chemicals in wildlife and humans. Environmental Health Perspectives, 101:378-384.

Compagno, L.J.V., 1984. Sharks of the World; Hexanchiformes to Lamniformes. Fishery synopsis No. 125, Volume 4, Part 1. Food and Agriculture Organization of the United Nations, Rome, Italy.

Confer, A.W. and R.J. Panciera, 1995. Thomsons Special Veterinary Pathology: The urinary system. Mosby Year Book, 1995, St. Louis, USA.

Corsini, E., R.W. Luebke, D.R. Germolec and J.C. DeWitt, 2014. Perfluorinated compounds: emerging POPs with potential immunotoxicity. Toxicology Letters, 230:263-270.

Costantini, D., A. Meillere, A. Carravieri, V. Lecomte, G. Sorci, B. Fiavre, H. Weimerskirch, P. Bustamante, P. Labadie, H. Budzinski and O. Chastel, 2014. Oxidative stree in relation to reproduction, contaminants, gender and age in a long-lived seabird. Oecologia, 175:1107-1116.

Damstra, T., S. Barlow, A. Bergman, R. Kavlock and G.V.D. Kraak, 2002. Global Assessment of the State-of-the-Science of Endocrine Disruptors. World Health Organization, Geneva, Switzerland.

Daugaard-Petersen, T., R. Langebæk, F.F. Rigét, R.J. Letcher, L. Hyldstrup, J.-E. Bech Jensen, T. Bechshoft, Ø. Wiig, B.M. Jenssen, C. Pertoldi, E.D. Lorenzen, R. Dietz and C. Sonne, 2018. Persistent organic pollutants, skull size and bone density of polar bears (*Ursus maritimus*) from East Greenland 1892–2015 and Svalbard 1964–2004. Environmental Research, 162:74-80.

de Wit, C.A., M. Alaee and C.D.G. Muir, 2006. Levels and trends of brominated flame retardants in the Arctic. Chemosphere, 64:209-233.

de Wit, C.A., D. Herzke and K. Vorkamp, 2010. Brominated flame retardants in the Arctic environment – Trends and new candidates. Science of the Total Environment, 408:2885-2918.

Debier, C. and Y. Larondelle, 2005. Vitamins A and E: metabolism, roles and transfer to offspring. British Journal of Nutrition, 93:153-174.

Dennis, K.K. and D.P. Jones, 2016. The Exposome: A new frontier for education. The American Biology Teacher, 78:542-548.

Descamps, S., J. Bety, O.P. Love and H.G. Gilchrist, 2011. Individual optimization of reproduction in a long-lived migratory bird: a test of the condition-dependent model of laying date and clutch size. Functional Ecology, 25:671-681.

Desforges, J.-P., P.S. Ross, N. Dangerfield, V.P. Palace, M. Whiticar and L.L. Loseto, 2013. Vitamin A and E profiles as biomarkers of PCB exposure in beluga whales (*Delphinapterus leucas*) from the western Canadian Arctic. Aquatic Toxicology, 142:317-328.

Desforges, J.-P., C. Sonne, M. Levin, U. Siebert, S. De Guise and R. Dietz, 2016. Immunotoxic effects of environmental pollutants in marine mammals. Environment International, 86:126-139.

Desforges, J.-P., M. Levin, L. Jasperse, S. De Guise, I. Eulaers, R.J. Letcher, M. Acquarone, E. Nordøy, L. Folkow, T. Hammer Jensen, C. Grøndahl, M. Bertelsen, J. St. Leger, J. Almunia, C. Sonne and R. Dietz, 2017a. Effects of polar bear and killer whale derived contaminant cocktails on marine mammal immunity. Environmental Science and Technology, 51:11431-11439.

Desforges, J.-P., I. Eulaers, L. Periard, C. Sonne, R. Dietz and R.J. Letcher, 2017b. A rapid analytical method to quantify complex organohalogen contaminant mixtures in large samples of high lipid mammalian tissues. Chemosphere, 176:243-248.

Desforges, J.-P., A. Hall, B. McConnell, I. Eulaers, C. Sonne, A. Rosing Asvid, G. Vikingson, F. Samarra, P. Jepson and R. Dietz, 2018. Projecting the global fate of killer whales from PCB pollution. Science, 361:1373-1376.

Dewailly, É., G. Mulvad, H.S. Pedersen, P. Ayotte, A. Demers, J.P. Weber and J.C. Hansen, 1999. Concentration of organochlorines in human brain, liver, and adipose tissue autopsy samples from Greenland. Environmental Health Perspectives, 107:823-828.

Dietz, R., C.T. Hansen, P. Have and M.P. Heide-Jørgensen, 1989. Clue to seal epizootic? Nature, 338:627.

Dietz, R., P. Paludan-Müller, C.T. Agger and C. Overgaard Nielsen, 1998. Cadmium, mercury, zinc and selenium in ringed seals (*Phoca hispida*) from Greenland and Svalbard. NAMMCO Scientific contributions, 1:242-272.

Dietz, R., F.F. Rigét and E.W. Born, 2000. An assessment of selenium to mercury in Greenland marine animals. Science of the Total Environment, 245:15-24.

Dietz, R., C. Sonne-Hansen, E.W. Born, H.T. Sandell and B. Sandell, 2001. Forekomst af "afvigende" isbjørne i Østgrønland. Faglig rapport fra DMU, No. 359. 50 pp.

Dietz, R., P.M. Outridge and K.A. Hobson, 2009. Anthropogenic contribution to mercury levels in present-day Arctic animals – A review. Science of the Total Environment, 407:6120-6131.

Dietz, R., E.W. Born, F.F. Rigét, A. Aubail, C. Sonne, R.C. Drimmei and N. Basu, 2011. Temporal trends and future predictions of mercury concentrations in northwest Greenland polar bear (*Ursus maritimus*) hair. Environmental Science and Technology, 45:1458-1465.

Dietz, R., F.F. Rigét, C. Sonne, E.W. Born, T.Ø. Bechshøft, M. McKinney, D.C.G. Muir and R.J. Letcher, 2013a. Three decades (1983-2010) of contaminant trends in East Greenland polar bears (*Ursus maritimus*). Part 1: Legacy organochlorine contaminants. Environment International, 59:485-493.

Dietz, R., F.F. Rigét, C. Sonne, E.W. Born, T.Ø. Bechshøft, M. McKinney, D.C.G. Muir and R.J. Letcher, 2013b. Three decades (1983-2010) of contaminant trends in East Greenland polar bears (*Ursus maritimus*). Part 2: Brominated flame retardants. Environment International, 59:494-500.

Dietz, R., C. Sonne, N. Basu, B. Braune, T. O'Hara, R.J. Letcher, T. Scheuhammer, M. Andersen, C. Andreasen, D. Andriashek, G. Asmund, A. Aubail, H. Baagee, E.W. Born, H.M. Chan, A.E. Derocher, P. Grandjean, K. Knott, M. Kirkegaard, A. Krey, N. Lunn, F. Messier, M. Obbard, M.T. Olsen, S. Ostertag, E. Peacock, A. Renzoni, F.F. Rigét and J.U. Skaare, 2013c. What are the toxicological effects of mercury in Arctic biota? Science of the Total Environment, 443:775-790.

Dietz, R., K. Gustavson, C. Sonne, J.P. Desforges, F.F. Rigét, V. Pavlova, M.A. McKinney and R.J. Letcher, 2015. Physiologically-based pharmacokinetic modelling of immune, reproductive and carcinogenic effects from contaminant exposure in polar bears (*Ursus maritimus*) across the Arctic. Environmental Research, 140:45-55.

Dietz, R., J.-P. Desforges, K. Gustavson, F.F. Rigét, E.W. Born, R.J. Letcher and C. Sonne, 2018. Immunologic, reproductive, and carcinogenic risk assessment in East Greenland polar bears (*Ursus maritimus*) during the period 1983-2013. Environmental International, 118:169-178.

Donaldson, S.G., J. Van Oostdam, C. Tikhonov, M. Feeley, B. Armstrong, P. Ayotte, O. Boucher, W. Bowers, L. Chan, F. Dallaire and R. Dallaire, 2010. Environmental contaminants and human health in the Canadian Arctic. Science of the Total Environment, 408:5165-5234.

Drevnick, P., 2012. Investigation of mercury toxicity in landlocked char in High Arctic lakes. In: Synopsis of Research Conducted under the 2011-2012 Northern Contaminants Program. Aboriginal Affairs and Northern Development Canada, pp. 327-337.

Drevnick, P., 2013. Investigation of mercury toxicity in landlocked char in High Arctic lakes. In: Synopsis of Research Conducted under the 2012-2013 Northern Contaminants Program. Aboriginal Affairs and Northern Development Canada, pp. 305-316.

Duffy, L.K., E. Scofield, T. Rodgers, M. Patton and R.T. Bowyer, 1999. Comparative baseline levels of mercury, Hsp 70 and Hsp 60 in subsistence fish from the Yukon-Kuskokwim delta region of Alaska. Comparative Biochemistry and Physiology C, 124:181-186.

Durner, G.M., D.C. Douglas, R.M. Nielson, S.C. Amstrup, T.L. McDonald, I. Stirling, M. Mauritzen, E.W. Born, Ø. Wiig, E. DeWeaver and M.C. Serreze, 2009. Predicting 21st-century polar bear habitat distribution from global climate models. Ecological Monographs, 79:25-58.

El-Bibany, A.H., A.G. Bodnar and H.C. Reinardy, 2014. Comparative DNA damage and repair in echinoderm coelomocytes exposed to genotoxicants. PLoS ONE, 9:107815.

Elias, S.A., 1994. Quaternary Insects and their Environments. Smithsonian Institution Press.

Epstein, P.R., 1999. Climate and health. Science, 285:347.

Erikstad, K.E., T. Moum, J.-O. Bustnes and T.K. Reiertsen, 2011. High levels of organochlorines may affect hatching sex ratio and hatchling body mass in arctic glaucous gulls. Functional Ecology, 25:289-296.

Evenset, A., G.N. Christensen, T. Skotvold, E. Fjeld, M. Schlabach, E. Wartena and D. Gregor, 2004. A comparison of organic contaminants in two high Arctic lake ecosystems, Bjørnøya (Bear Island), Norway. Science of the Total Environment, 318:125-141.

Evenset, A., J. Carroll, G.N. Christensen, R. Kallenborn, D. Gregor and G.W. Gabrielsen, 2007. Seabird guano is an efficient conveyer of persistent organic pollutants (POPs) to Arctic lake ecosystems. Environmental Science and Technology, 41:1173-1179.

Evers, D.C., S.E. Keane, N. Basu and D. Buck, 2016. Evaluating the effectiveness of the Minamata Convention on Mercury: Principles and recommendations for next steps. Science of the Total Environment, 569-570:888-903.

Fallacara, D.M., R.S. Halbrook and J.B. French, 2011a. Toxic effects of dietary methylmercury on immune function and hematology in American kestrels (*Falco sparverius*). Environmental Toxicology and Chemistry, 30:1320-1327.

Fallacara, D.M., R.S. Halbrook and J.B. French, 2011b. Toxic effects of dietary methylmercury on immune system development in nestling American kestrels (*Falco sparverius*). Environmental Toxicology and Chemistry, 30:1328-1337.

Feldman, E.C., 1995. Hyperadrenocorticism. In: Ettinger, S.J. and E.C. Feldman (Eds.), Textbook of Veterinary Internal Medicine. Volume II. Saunders, Philadelphia, USA.

Fenstad, A., B. Jenssen, B. Mow, S. Hanssen, C. Bingham, D. Herzke, J.-O. Bustnes and A. Krokje, 2014. DNA double-strand breaks in relation to persistent organic pollutants in fasting common eiders (*Somateria molissima*). Mutagenesis, 29:539-539.

Fent, K. and J.P. Sumpter, 2011. Progress and promises in toxicogenomics in aquatic toxicology: Is technical innovation driving scientific innovation? Aquatic Toxicology, 105:25-39.

Fisk, A.T., S.A. Tittlemier, J.L. Pranschke and R.J. Norstrom, 2002. Using anthropogenic contaminants and stable isotopes to assess the feeding ecology of Greenland sharks. Ecology, 83:2162-2172.

Fisk, A.T., C.A. de Wit, M. Wayland, Z.Z. Kyzyk, N. Burgess, R. Robert, B.M. Braune, R.J. Norstrom, S.P. Blum, C. Sandau, E. Lie, H.J.S. Larsen, J.U. Skaare and D.C.G. Muir, 2005. An assessment of the toxicological significance of anthropogenic contaminants in Canadian Arctic wildlife. Science of the Total Environment, 351:57-93.

Flato, G., J. Marotzke, B. Abiodun, P. Braconnot, S.C. Chou, W. Collins, P. Cox, F. Driouech, S. Emori, V. Eyring, C. Forest, P. Gleckler, E. Guilyardi, C.Jakob, V. Kattsov, C. Reason and M. Rummukainen, 2013. Evaluation of climate models. In: Climate Change 2013: The Physical Science Basis.Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. pp. 741-866. Cambridge University Press.

Frandsen, F., H.J. Malmquist and S.S. Snorrason, 1989. Ecological parasitology of polymorphic Arctic charr, *Salvelinus alpinus* (L.), in Thingvallavatn, Iceland. Journal of Fish Biology, 34:281-297.

Friedrichs, K.R., K.E. Harr, K.P. Freeman, B. Szladovits, R.M. Walton, K.F. Barnhart and J. Blanco-Chavez, 2012. ASVCP reference interval guidelines: determination of de novo reference intervals in veterinary species and other related topics. Veterinary Clinical Pathology, 41:441-453.

Frouin, H., L.L. Loseto, G.A. Stern, M. Haulena and P.S. Ross, 2012. Mercury toxicity in beluga whale lymphocytes: Limited effects of selenium protection. Aquatic Toxicology, 109:185-193.

Gabrielsen, K.M., G.D. Villanger, E. Lie, M. Karimi, C. Lydersen, K.M. Kovacs and B.M. Jenssen, 2011. Levels and patterns of hydroxylated polychlorinated biphenyls (OH-PCBs) and their associations with thyroid hormones in hooded seal (*Cystophora cristata*) mother-pup pairs. Aquatic Toxicology, 105:482-491.

Gabrielsen, K.M., J.S. Krokstada, G.D. Villanger, D.A.D. Blair, M.-J. Obregond, C. Sonne, R. Dietz, R.J. Letcher and B.M. Jenssen, 2015. Thyroid hormones and deiodinase activity in plasma and tissues in relation to high levels of organohalogen contaminants in East Greenland polar bears (*Ursus maritimus*). Environmental Research, 136:413-423.

Gaden, A. and G.A. Stern, 2010. Temporal trends in beluga, narwhal and walrus mercury levels: Links to climate change. In: Ferguson, S.H., L.L. Loseto and M.L. Mallory (eds.), Little Less Arctic: Top Predators in the Worlds Largest Northern Inland Sea, Hudson Bay. pp. 197-216. Springer-Verlag.

Gaden, A., S.H. Ferguson, L. Harwood, H. Melling and G.A. Stern, 2009. Mercury trends in ringed seals (*Phoca hispida*) from the Western Canadian Arctic since 1973: Associations with length of ice-free season. Environmental Science and Technology, 43:3646-3651.

Gaden, A., S.H. Ferguson, L. Harwood, H. Melling, J. Alikamik and G.A. Stern, 2012. Western Canadian Arctic ringed seal organic contaminant trends in relation to sea ice break-up. Environmental Science and Technology, 46:4427-4433.

Ganong, W.F., 2005. Review of Medical Physiology. 22nd edition, Appleton and Lange, USA.

Gantner, N., M. Power, D. Iqaluk, M. Meili, H. Borg, M. Sundbom, K.R. Solomon, G. Lawson and D.C. Muir, 2010a. Mercury concentrations in landlocked Arctic char (*Salvelinus alpinus*) from the Canadian Arctic. Part I: Insights from trophic relationships in 18 lakes. Environmental Toxicology and Chemistry, 29:621-632.

Gantner, N., D.C. Muir, M. Power, D. Iqaluk, J.D. Reist, J.A. Babaluk, M. Meili, H. Borg, J. Hammar, W.B. Michaud, B. Dempson and K.R. Solomon, 2010b. Mercury concentrations in landlocked arctic char (*Salvelinus alpinus*) from the Canadian Arctic. Part II: Influence of lake biotic and abiotic characteristics on geographic trends in 27 populations. Environmental Toxicology and Chemistry, 29:633-643.

Gebbink, W.A., C. Sonne, R. Dietz, M. Kirkegaard, F.F. Riget, E.W. Born, D.C.G. Muir and R.J. Letcher, 2008. Tissue-specific congener composition of organohalogen and metabolite contaminants in East Greenland polar bears (*Ursus maritimus*). Environmental Pollution, 152:621-629.

Giguère, A., P.G.C. Campbell, L. Hare and P. Couture, 2006. Sub-cellular partitioning of cadmium, copper, nickel and zinc in indigenous yellow perch (*Perca flavescens*) sampled along a polymetallic gradient. Aquatic Toxicology, 77:178-189.

Glahder, C., 1995. Hunting in Kangerlussuaq, East Greenland 1951-1991: An assessment of local knowledge Man & Society, No. 19. 79 pp.

Goutte, A., M. Kriloff, H. Weimerskirch and O. Chastel, 2011. Why do some adult birds skip breeding? A hormonal investigation in a long-lived bird. Biology Letters, 7:790-792.

Grandjean, P. and P.J. Landrigan, 2006. Review: Developmental neurotoxicity of industrial chemicals. Lancet, 16:2167-2178.

Greaves, A.K., R.J. Letcher, C. Sonne and R. Dietz, 2012. Brain region distribution and patterns of bioaccumulative perfluoroalkyl carboxylates and sulfonates in East Greenland polar bears (*Ursus maritimus*). Environmental Toxicology and Chemistry, 32:713-722.

Guinet, C., P. Domenici, R. de Stephanis, L. Barrett-Lennard, J.K.B. Ford and P. Verborgh, 2007. Killer whale predation on bluefin tuna: exploring the hypothesis of the endurance-exhaustion technique. Marine Ecology-Progress Series, 347:111-119.

Gustavson, L., T.M. Ciesielski, J. Bytingsvik, B. Styrishave, M. Hansen, E. Lie, J. Aars and B.M. Jenssen, 2015. Hydroxylated polychlorinated biphenyls decrease circulating steroids in female polar bears (*Ursus maritimus*). Environmental Research, 138:191-201.

Gutleb, A.C., P. Cenijn, M. van Velzen, E. Lie, E. Ropstad, J.U. Skaare, T. Malmber, Å. Bergman, G.W. Gabrielsen and J. Legler, 2010. In vitro assay shows that PCB metabolites completely saturate thyroid hormone transport capacity in blood of wild polar bears (*Ursus maritimus*). Environmental Science and Technology, 44:3149-3154.

Haarr, A., K. Hylland, N. Eckbo, G.W. Gabrielsen, J.O. Bustnes, D. Herzke, P. Blevin, O. Chastel, S.A. Hanssen, B. Moe, K. Sagerup and K. Borgå, 2018. DNA damage in breeding Arctic seabirds: Baseline, sensitivity to oxidative stress and association to organohalogen contaminants. Environmental Toxicology and Chemistry, 37:1084-1091.

Hallanger, I.G., N.A. Warner, A. Ruus, A. Evenset, G. Christensen, D. Herzke, G.W. Gabrielsen and K. Borgå, 2011. Seasonality in contaminant accumulation in Arctic marine pelagic food webs using trophic magnification factor as a measure of bioaccumulation. Environmental Toxicology and Chemistry, 30:1026-1035.

Hallanger, I.G., E.H. Jorgensen, E. Fuglei, O. Ahlstrom, D.C.G. Muir and B.M. Jenssen, 2012. Dietary contaminant exposure affects plasma testosterone, but not thyroid hormones, vitamin A, and vitamin E, in male juvenile foxes (*Vulpes lagopus*). Journal of Toxicology and Environmental Health A, 75:1298-1313.

Hammar, J., 2000. Cannibals and parasites: conflicting regulators of bimodality in high latitude Arctic char, *Salvelinus alpinus*. Oikos, 88:33-47.

Hammond, P.S., K. Macleod, P. Berggren, D.L. Borchers, L. Burt, A. Cañadas, G. Desportes, G.P. Donovan, A. Gilles, D. Gillespie, J. Gordon, L. Hiby, I. Kuklik, R. Leaper, K. Lehnert, M. Leopold, P. Lovell, N. Øien, C.G.M. Paxton, V. Ridoux, E. Rogan, F. Samarra, M. Scheidat, M. Sequeira, U. Siebert, H. Skov, R. Swift, M.L. Tasker, J. Teilmann, O. Van Canneyt and J.A. Vázquez, 2013. Cetacean abundance and distribution in European Atlantic shelf waters to inform conservation and management. Biological Conservation, 164:107-122.

Hansen, P.M., 1963. Tagging experiments with the Greenland shark (*Somniosus microcephalus*, Bloch and Schneider) in subarea 1. International Commission Northwest Atlantic Fisheries Special Publication, 4:172-175.

Hargreaves, A.L., D.P. Whiteside and G, Gilchrist, 2010. Concentrations of 17 elements, including mercury, and their relationship to fitness measures in arctic shorebirds and their eggs. Science of the Total Environment, 408:3153-3161.

Harr, K.E., 2012. Clinical chemistry of companion avian species: A review. Veterinary Clinical Pathology, 31:140-151.

Hays, S.M., R.A. Becker, H.W. Leung, L.L. Aylward and D.W. Pyatt, 2007. Biomonitoring equivalents: a screening approach for interpreting biomonitoring results from a public health risk perspective. Regulatory Toxicology and Pharmacology, 47:96-109.

Hays, S.M, L.L. Aylward, J.S. LaKind, J.M. Bartels, H.A. Barton, P.J. Boogaard, C. Brunk, S. DiZio, M. Dourson, D.A. Goldstein, J. Lipscomb, M.E. Kilpatrick, D. Krewskim, K. Krishnan, M Nordberg, M. Okino, Y.-M. Tan, C. Viau and J.W. Yager, 2008. Guidelines for the derivation of biomonitoring equivalents: report from the biomonitoring equivalents expert workshop. Regulatory Toxicology and Pharmacology, 51:S4-15.

Heath, J., G. Ljubicic and L. Arragutaina, 2015. A Social Media Network and Interactive Knowledge-Mapping Platform for the North: Developing the tools needed to meaningfully engage communities in research and environmental stewardship. ArcticNet Annual Scientific Meeting, 2015, Vancouver, British Columbia, Canada.

Hebert, C.E. and D.C. Weseloh, 2006. Adjusting for temporal change in trophic position results in reduced rates of contaminant decline. Environmental Science and Technology, 40:5624-5628.

Heinz, G., D.J. Hoffman, J. Klimstra, K. Stebbins, S. Kondrad and C. Erwin, 2009. Species differences in the sensitivity of avian embryos to methylmercury. Archives of Environmental Contamination and Toxicology, 56:129-138.

Helgason, L.B., R. Barrett, E. Lie, A. Polder, J.U. Skaare and G.W. Gabrielsen, 2008. Levels and temporal trends (1983-2003) of persistent organic pollutants (POPs) and mercury (Hg) in seabird eggs from Northern Norway. Environmental Pollution, 155:190-198.

Helgason, L.B., J. Verreault, B.M. Braune, K. Borgå, R. Primicerio, B.M. Jenssen and G.W. Gabrielsen, 2010. Relationship between persistent halogenated organic contaminants and TCDD-toxic equivalents on EROD activity and retinoid and thyroid hormone status in northern fulmars. Science of the Total Environment, 408:6117-6123.

Helgason, L.B., K. Sagerup and G.W. Gabrielsen, 2011. Organohalogen pollutants in seabird eggs from northern Norway and Svalbard. In: Loganathan, B. and P.K.S. Lam (Eds.), Global Contamination Trends of Persistent Organic Chemicals. CRC Press, pp. 547-569.

Helgason, L.B., H. Wolkers, E. Fuglei, O. Ahlstrom, D.C.G. Muir and E.H. Jørgensen, 2013. Seasonal emaciation causes tissue redistribution and an increased potential for toxicity of lipophilic pollutants in farmed Arctic fox (*Vulpes lagopus*). Environmental Toxicology and Chemistry, 32:1784-1792.

Hickie, B.E., M.C. Kingsley, P.V. Hodson, D.C. Muir, P. Béland and D. Mackay, 2000. A modelling-based perspective on the past, present, and future polychlorinated biphenyl contamination of the St. Lawrence beluga whale (*Delphinapterus leucas*) population. Canadian Journal of Fisheries and Aquatic Sciences, 57:101-112.

Hickie. B.E., D.C.G. Muir, R.F. Addison and P.F. Hoekstra, 2005. Development and application of bioaccumulation models to assess persistent organic pollutant temporal trends in arctic ringed seal (*Phoca hispida*) populations. Science of the Total Environment, 351-352:413-426.

Hirakawa, S., D. Imaeda, K. Nakayama, M. Udaka, E.-Y. Kim and T. Kunisue, 2011. Integrative assessment of potential effects of dioxins and related compounds in wild Baikal seals (*Pusa sibirica*): Application of microarray and biochemical analyses. Aquatic Toxicology, 105:89-99.

Hoekstra, P.F., B.M. Braune, B. Elkind, F.A.J. Armstrong and D.C.G. Muir, 2003. Concentrations of selected essential and non-essential elements in arctic fox (*Alopex lagopus*) and wolverines (*Gulo gulo*) from the Canadian Arctic. Science of the Total Environment, 309:81-92.

Holth, T.F. and K.E. Tollefsen, 2012. Acetylcholine esterase inhibitros in effluents from oil production platforms in the North Sea. Aquatic Toxicology, 112:92-98.

Hook, S.E., 2010. Promise and progress in environmental genomics: A status report on the applications of gene expression-based microarray studies in ecologically relevant fish species. Journal of Fish Biology, 77:1999-2022.

Hoydal, K. and M. Dam, 2005. AMAP Faroe Islands Heavy metals and POPs Core programme 2004. Food, Veterinary and Environmental Agency, Report no. 2005:2, Torshavn, Faroe Islands.

Hoydal, K. and M. Dam, 2009. AMAP Faroe Islands Heavy metals and POPs programme 2005-2008. Food, Veterinary and Environmental Agency, Report no. US 2009:1, Torshavn, Faroe Islands.

Hoydal, K.S., R.L. Letcher, D.A.D. Blair, M. Dam, C. Lockyer and B.M. Jenssen, 2015. Legacy and emerging organic pollutants in liver and plasma sample pairs of long-finned pilot whales (*Globicephala melas*) harvested from waters surrounding the Faroe Islands. Science of the Total Environment, 520:270-285.

Hoydal, K.S., T. Ciesielski, A. Borrell, A. Wasik, R.J. Letcher, M. Dam and B.M. Jenssen, 2016. Relationships between concentrations of selected OHCs and thyroid hormones and vitamins A, E and D in Faroese pilot whales. Environmental Research, 148:386-400.

Hylland, K., O.O. Aspholm, J.A. Knutsen and A. Ruus, 2006. Biomarkers in fish from dioxin-contaminated fjords. Biomarkers, 11:97-117.

Hylland, K., B.B. Shei, G. Brunborg, T. Lang, M.J. Gubbins, J. le Goff and T. Burgeot, 2017. DNA damage in dab (*Limanda limanda*) and haddock (*Melanogrammus aeglefinus*) from European seas. Marine Environmental Research, 124:54-60.

ICC-Alaska, 2015. Alaskan Inuit Food Security Conceptual Framework: How to assess the Arctic from an Inuit perspective. Technical Report. Inuit Circumpolar Council (ICC) Alaska, Anchorage.

Imaeda, D., K. Nomiyama, T. Kunisue, H. Iwata, O. Tsydenova, M. Amano, E.A. Petrov, V.B. Batoev and Y. Tajima, 2014. Blood levels of polychlorinated biphenyls and their hydroxylated metabolites in Baikal seals (*Pusa sibirica*): Emphasis on interspecies comparison, gender difference and association with blood thyroid hormone levels. Chemosphere, 114:1-8.

Jacobson, J.L. and S.W. Jacobson, 2002. Breast-feeding and gender as moderators of teratogenic effects on cognitive development. Neurotoxicology and Teratology, 24:349-358.

Jæger, I., H. Hop and G.W. Gabrielsen, 2009. Biomagnification of mercury in selected species from an Arctic marine food web in Svalbard. Science of the Total Environment, 407:4744-4751.

Jager, T., T. Vandenbrouck, J. Baas, W.M. De Coen and S.A.L.M. Kooijman, 2010. A biology-based approach for mixture toxicity of multiple endpoints over the life cycle. Ecotoxicology, 19:351-361.

Jämsä, T., M. Viluksela, J.T. Tuomisto, J. Tuomisto and J. Tuukkanen, 2001. Effects of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin on bone in two rats strains with different aryl hydrocarbon receptor structures. Journal of Bone and Mineral Research, 16:1812-1820.

Jefferies, M.K.S., A.C. Mehinto, B.J. Carter, N.D. Denslow and A.S. Lolok, 2012. Taking microarrays to the field: Differential hepatic gene expression of caged fathead minnows from Nebraska watersheds. Environmental Science and Technology, 46:1877-1885.

Jenssen, B.M., 2006. Endocrine-disrupting chemicals and climate change: A worst-case combination for Arctic marine mammals and seabirds? Environmental Health Perspectives, 114:76-80.

Jenssen, B.M., G.D. Villanger, K.M. Gabrielsen, J. Bytingsvik, T.Ø. Bechshøft, T.M. Ciesielski, C. Sonne and R. Dietz, 2015. Anthropogenic flank attack on polar bears: Interacting consequences of climate warming and pollutant exposure. Frontiers in Ecology and Evolution, 3:1-7.

Jepson, P.D., R. Deaville, J.L. Barber, À. Aguilar, A. Borrell, S. Murphy, J. Barry, A. Brownlow, J. Barnett, S. Berrow and A.A. Cunningham, 2016. PCB pollution continues to impact populations of orcas and other dolphins in European waters. Nature, 6:18573.

Kainz, M.J. and A.T. Fisk, 2009. Integrating lipids and contaminants in aquatic ecology and ecotoxicology. In: Arts, M.T., M.T. Brett and M.J. Kainz (Eds.), Lipid in Aquatic Ecosystems. Springer.

Kaiser, J. and M. Enserink, 2000. Treaty takes a POP at the dirty dozen. Science, 290:2053.

Kanerva, M., H. Routti, Y. Tamuz, M. Nyman and M. Nikinmaa, 2012. Antioxidative defense and oxidative stress in ringed seals (*Pusa hispida*) from differently polluted areas. Aquatic Toxicology, 114:67-72.

Karpiej, K., J. Dzido, J. Rokicki and A. Kijewska, 2013. Anisakid nematodes of Greenland halibut *Reinhardtius hippoglossoides* from the Barents Sea. The Journal of Parasitology, 99:650-654.

Kirkegaard, M., C. Sonne, J. Jakobsen, B.M. Jenssen, R.J. Letcher and R. Dietz, 2010. Organohalogens in a whale-blubber-supplemented diet affects hepatic retinol and renal tocopherol concentrations in Greenland sled dogs (*Canis familiaris*). Journal of Toxicology and Environmental Health A, 73:773-786.

Kirkegaard, M., C. Sonne, R. Dietz, R.J. Letcher, A.L. Jensen, B.M. Jenssen and P. Grandjean, 2011. Alterations in thyroid hormone status in Greenland sledge dogs exposed to naturally OHC contaminated whale blubber. Ecotoxicology and Environmental Safety, 74:157-163.

Klaassen, C.D., M.O. Amdur and J. Doull, 2007. The Basic Science of Poisons. 6th Edition. McGraw-Hill.

Klanjscek, T., R.M. Nisbet, H. Caswell and M.G. Neubert, 2007. A model for energetics and bioaccumulation in marine mammals with applications to the right whale. Ecological Applications, 17:2233-2250.

Klecha, A.J., M.L. Barreiro Arcos, L. Frick, A.M. Genaro and G. Cremaschi, 2008. Immune-endocrine interactions in autoimmune thyroid diseases. Neuroimmunomodulation, 15:68-75.

Knott, K.K., P. Schenk, S. Beyerlein, D. Boyd, G.M. Ylitalo and T.M. O'Hara, 2011. Blood-based biomarkers of selenium and thyroid status indicate possible adverse biological effects of mercury and polychlorinated biphenyls in Southern Beaufort Sea polar bears. Environmental Research, 111:1124-1136.

Köck, G., M. Triendl and R. Hofer, 1996. Seasonal patterns of metal accumulation in Arctic char (*Salvelinus alpinus*) from an oligotrophic Alpine lake related to temperature. Canadian Journal of Fisheries and Aquatic Sciences, 53:780-786.

Koeman, J.H., W.H.M. Peeters, C.H.M. Koudstaal-Hol, P.S. Tijoe and J.J.M. de Goeij, 1973. Mercury-selenium correlations in marine mammals. Nature, 245:385-386.

Koeman, J.H., W.S.M. Ven, J.J.M. Goeij, P.S. Tijoe and J.L. Haften, 1975. Mercury and selenium in marine mammals and birds. Science of the Total Environment, 3:279-287.

Kooijman, S.A.L.M. and J.J.M. Bedaux, 1996. The Analysis of Aquatic Toxicity Data. VU University Press, Amsterdam, Netherlands.

Kovacs, K.M., C. Lydersen, J.E. Overland and S.E. Moore, 2011. Impacts of changing sea-ice conditions on Arctic marine mammals. Marine Biodiversity, 41:181-194.

Kramer, V.J., M.A. Etterson, M. Hecker, C.A. Murphy, G. Roesijadi, D.J. Spade, J.A. Spromberg, M. Wang and G.T. Ankley, 2011. Adverse outcome pathways and ecological risk assessment: Bridging to population-level effects. Environmental Toxicology and Chemistry, 30:64-76.

Krey, A., M. Kwan and H.M. Chan, 2012. Mercury speciation in brain tissue of polar bears (*Ursus maritimus*) from the Canadian Arctic. Environmental Research, 114:24-30.

Krey, A., M. Kwan and H.M. Chan, 2014. In vivo and in vitro changes in neurochemical parameters related to mercury concentrations from specific brain regions of polar bears (*Ursus maritimus*). Environmental Toxicology and Chemistry, 33:2463-2471.

Krey, A., S.K. Ostertag and H.M. Chan, 2015. Assessment of neurotoxic effects of mercury in beluga whales (*Delphinapterus leucas*), ringed seals (*Pusa hispida*), and polar bears (*Ursus maritimus*) from the Canadian Arctic. Science of the Total Environment, 15:237-247.

Kuenzel, W.J., 2003. Neurobiology of molt in avian species. Poultry Science, 82:981-991.

Kutz, S., 2009. The Arctic as a model for anticipating, preventing, and mitigating climate change impacts on host-parasite interactions. Veterinary Parasitology, 163:217-228.

Kutz, S.J., S. Checkley, G. Verocai, M. Dumond, E. Hoberg, E. Peacock, R. Wu, J. Orsel, K. Seegers and A.L. Warren, 2013. Invasion, establishment, and range expansion of two parasitic nematodes in the Canadian Arctic. Global Change Biology, 19:3254-3262.

Kuzyk, Z.A., P.V. Hodson, S.M. Solomon and K.J. Reimer, 2005. Biological responses to PCB exposure in shorthorn sculpin from Saglek Bay, Labrador. Science of the Total Environment, 351-352:285-300.

Laidre, K.L., I. Stirling, L.F. Lowry, Ø. Wiig, M.P. Heide-Jørgensen and S.H. Ferguson, 2008. Quantifying the sensitivity of Arctic marine mammals to climate-induced habitat change. Ecological Applications, 18:S97-S125.

Laidre, K.L., G.A. Stern, K.M. Kovacs, L. Lowry, S.E. Moore, E.V. Regehr, S.H. Ferguson, Ø. Wiig, P. Boveng, R.P. Angliss, E.W. Born, D. Litovka, L. Quakenbush, C. Lydersen, D. Vongraven and F. Ugarte, 2015. Arctic marine mammal population status, sea ice habitat loss, and conservation recommendations for the 21st century. Conservation Biology, 29:724-737.

Larter, N.C., C.R. Macdonald, B.T. Elkin, X. Wang, N.J. Harms, M. Gamberg and D.C. Muir, 2016. Cadmium and other elements in tissues from four ungulate species from the Mackenzie Mountain region of the Northwest Territories, Canada. Ecotoxicology and Environmental Safety, 132:9-17.

Leclerc, L.-M., C. Lydersen, T. Haug, L. Bachmann, A.T. Fisk and K.M. Kovacs, 2012. A missing piece in the Arctic food web puzzle? Stomach contents of Greenland sharks sampled in Svalbard, Norway. Polar Biology, 35:1197-1208.

LeCren, E.D., 1951. The length-weight relationship and seasonal cycle in gonad weight and condition in the perch (*Perca fluviatilis*). Journal of Animal Ecology, 20:201-219.

Leeson, S. and T. Walsh, 2004. Feathering in commercial poultry. II: Factors influencing feather growth and feather loss. World's Poultry Science Journal, 60:52-63.

Lescord, G.L., K.A. Kidd, J.L. Kirk, N.J. O'Driscoll, X. Wang and D.C.G. Muir, 2015. Factors affecting biotic mercury concentrations and biomagnification through lake food webs in the Canadian High Arctic. Science of the Total Environment, 509:195-205.

Letcher, R.J. and M. Dyck, 2016. Temporal and spatial trends of contaminants in Canadian polar bears. In: Smith, S.L., J. Stow and J. Edwards (Eds.), Synopsis of research conducted under the 2015-2016 Northern Contaminants Program. Indigenous and Northern Affairs Canada, pp. 211-226.

Letcher, R.J., J.O. Bustnes, R. Dietz, B.M. Jenssen, E.H. Jørgensen, C. Sonne, J. Verreault, M.M. Vijayan and G.W. Gabrielsen, 2010. Exposure and effects assessment of persistent organohalogen contaminants in Arctic wildlife and fish. Science of the Total Environment, 408:2995-3043.

Letcher, R.J., A.D. Morris, M. Dyck, E. Sverko, E. Reiner, D.A.D. Blair, S.G. Chu and L. Shen, 2018. Legacy and new halogenated persistent organic pollutants in polar bears from a contamination hotspot in the Arctic, Hudson Bay Canada. Science of the Total Environment, 610-611:121-136.

Levin, M., E. Gebhard, L. Jasperse, J.-P. Desforges, R. Dietz, C. Sonne, I. Eulaers, A. Covaci, R. Bossi and S. De Guise, 2016. Immunomodulatory effects of exposure to polychlorinated biphenyls and perfluoroalkyl acids in East Greenland ringed seals (*Pusa hispida*). Environmental Research, 151:244-250.

Lind, P.M., E.F. Eriksen, L. Sahlin, M. Edlund and J. Örberg, 1999. Effects of the antiestrogenic environmental pollutants 3,3',4,4',5-pentachlorobiphenyl (PCB-126) in rat bone and uterus: diverging effects in ovarectomized and intact animals. Toxicology and Applied Pharmacology, 154:236-244.

Lind, P.M., S. Larsson, H. Oxlund, H. Håkansson, K. Nyberg, T. Eklund and J. Örberg, 2000. Change of bone tissue composition and impaired bone strength in rats exposed to PCB-126. Toxicology, 150:41-51.

Lindgren, E. and R. Gustafson, 2001. Tick-borne encephalitis in Sweden and climate change. The Lancet, 358:16-18.

López, L.M., M. Flores-Ibarra, I. Banuelos-Vargas, M.A. Galaviz and C.D. True, 2015. Effect of fishmeal replacement by soy protein concentrate with taurine supplementation on growth performance, hematological and biochemical status, and liver histology of totoaba juveniles (*Totoaba macdonaldi*). Fish Physiology and Biochemistry, 41:921-936.

Loseto, L.L., G.A. Stern and R.W. Macdonald, 2015. Distant drivers or local signals: Where do mercury trends in western Arctic belugas originate? Science of the Total Environment, 509-510:226-236.

Lucas, Z.N. and L.J. Natanson, 2010. Two shark species involved in predation on seals at Sable Island, Nova Scotia, Canada. Nova Scotian Institute of Sciences, 45:64-88.

Lundberg, R., J.L. Lyche, E. Ropstad, M. Aleksandersen, M. Ronn, J.U. Skaare, S. Larsson, J. Orberg and P.M. Lind, 2006. Perinatal exposure to PCB 153, but not PCB 126, alters bone tissue composition in female goat offspring. Toxicology, 228:33-40.

Ma, J., H. Hung, C. Tian and R. Kallenborn, 2011. Revolatilization of persistent organic pollutants in the Arctic induced by climate change. Nature Climate Change, 1:255.

Macdonald, R.W., T. Harner, J. Fyfe, H. Loeng and T. Weingartner, 2003. AMAP Assessment 2002: The influence of global change on contaminant pathways to, within, and from the Arctic. Arctic Monitoring and Assessment Programme (AMAP), Oslo, Norway.

Macdonald, R.W., T. Harner and J. Fyfe, 2005. Recent climate change in the Arctic and its impact on contaminant pathways and interpretation of temporal trend data. Science of the Total Environment, 342:5-86.

Mariussen, E. and F. Fonnum, 2006. Neurochemical targets and behavioral effects of organohalogen compounds: an update. Critical Reviews in Toxicology, 36:253-289.

Martin, B.T., T. Jager, R.M. Nisbet, T.G. Preuss, M. Hammers-Wirtz and V. Grimm, 2013. Extrapolating ecotoxicological effects from individuals to populations: a generic approach based on Dynamic Energy Budget theory and individual-based modeling. Ecotoxicology, 22:574-583.

Matthiessen, P. and R.J. Law, 2002. Contaminants and their effects on estuarine and coastal organisms in the United Kingdom in the late twentieth century. Environmental Pollution, 120:739-757.

Maxie, M.G., 1993. Pathology of Domestic Animals. Academic Press.

McDowell, R.L., 2000. Vitamins in Animal and Human Nutrition. 2nd Edition. Iowa University Press.

McKinney, M.A., E. Peacock and R.J. Letcher, 2009. Sea ice-associated diet change increases the levels of chlorinated and brominated contaminants in polar bears. Environmental Science and Technology, 43:4334-4339.

McKinney, M.A., R.J. Letcher, J. Aars, E.W. Born, M. Branigan, R. Dietz, T.J. Evans, G.W. Gabrielsen, E. Peacock and C. Sonne, 2011. Flame retardants and legacy contaminants in polar bears from Alaska, Canada, East Greenland and Svalbard, 2005-2008. Environment International, 37:365-374.

McKinney, M.A., B.C. McMeans, G.T. Tomy, B. Rosenberg, S.H. Ferguson, A. Morris, D.C.G. Muir and A.T. Fisk, 2012. Trophic transfer of contaminants in a changing Arctic marine food web: Cumberland Sound, Nunavut, Canada. Environmental Science and Technology, 46:9914-9922.

McKinney, M.A., S.J. Iverson, A.T. Fisk, C. Sonne, F.F. Rigét, R.J. Letcher, M.T. Arts, E.W. Born, A. Rosing-Asvid and R. Dietz, 2013. Global change effects on the long-term feeding ecology and contaminant exposures of East Greenland polar bears. Global Change Biology, 19:2360-2372.

McKinney, M.A., T. Atwood, R. Dietz, C. Sonne, S.J. Iverson and E. Peacock, 2014. Validation of adipose lipid content as a body condition index for polar bears. Ecology and Evolution, 4:516-527.

McKinney, M.A., S. Pedro, R. Dietz, C. Sonne, A.T. Fisk, D. Roy, B.M. Jenssen and R.J. Letcher, 2015. A review of ecological impacts of global climate change on persistent organic pollutant and mercury pathways and exposures in Arctic marine ecosystems. Current Zoology, 61:617-628.

McNabb, F.M.A., 2007. The hypothalamic-pituitary-thyroid (HPT) axis in birds and its role in bird development and reproduction. Critical Reviews in Toxicology, 37:163-193.

Mergler, D., H.A. Anderson, L.H. Chan, K.R. Mahaffey, M. Murray, M. Sakamoto and A.H. Stern, 2007. Methylmercury exposure and health effects in humans: a worldwide concern. Ambio, 36:3-11.

Messier, V., 2008. Seroprevalence of *Toxoplasma gondii* among Nunavik Inuit (Canada). Zoonoses Public Health, 56:188-197.

Miljeteig, C., H. Strom, M.V. Gavrilo, A. Volkov, B.M. Jenssen and G.W. Gabrielsen, 2009. High levels of contaminants in ivory gull *Pagophila eburnea* eggs from the Russian and Norwegian Arctic. Environmental Science and Technology, 43:5521-5528.

Miljeteig, C., G.W. Gabrielsen, H. Strom, M.V. Gavrilo, E. Lie and B.M. Jenssen, 2012. Eggshell thinning and decreased concentrations of vitamin E are associated with contaminants in eggs of ivory gulls. Science of the Total Environment, 431:92-99.

Molde, K., T.M. Ciesielski, A.T. Fisk, C. Lydersen, K.M. Kovacs and E.G. Sormo, 2013. Associations between vitamins A and E and legacy POP levels in highly contaminated Greenland sharks (*Somniosus microcephalus*). Science of the Total Environment, 442:445-454.

Molnár, P.K., A.E. Derocher, G.W. Thiemann and M. Lewis, 2010. Predicting survival, reproduction and abundance of polar bears under climate change. Biological Conservation, 143:1612-1622.

Molnár, P.K., A.E. Derocher, T. Klanjscek and M.A. Lewis, 2011. Predicting climate change impacts on polar bear litter size. Nature Communications, 2:186.

Molnár, P.K., M. Lewis and A.E. Derocher, 2014. Estimating allee dynamics before they can be observed: Polar bears as a case study. PLoS One, 9:e85410.

Monnett, C. and J.S. Gleason, 2006. Observations of mortality associated with extended open-water swimming by polar bears in the Alaskan Beaufort Sea. Polar Biology, 29:681-687.

Morris, A.D., R.J. Letcher, M. Dyck, B. Chandramouli, A.T. Fisk and J. Cosgrove, 2018. Multivariate statistical analysis of metabolite profiles in tissues of polar bears (*Ursus maritimus*) from the Southern and Western Hudson Bay subpopulations. Polar Biology, 41:433-449.

Morris, A.D., R.J. Letcher, M. Dyck, B. Chandramouli and J. Cosgrove, 2018. Concentrations of legacy and new contaminants are related to metabolomic profiles in Hudson Bay polar bears. Environmental Research. In press.

Mouritsen, K., R. Hedeholm, H. Schack, L. Møller, M. Storr-Paulsen, J. Dzido and J. Rokicki, 2010. Occurrence of anisakid nematodes in Atlantic cod (*Gadus morhua*) and Greenland cod (*Gadus ogac*), West Greenland. Acta Parasitologica, 55:81-89.

Muir, D., X. Wang, D. Bright, L. Lockhart and G. Köck, 2005. Spatial and temporal trends of mercury and other metals in landlocked char from lakes in the Canadian Arctic archipelago. Science of the Total Environment, 351-352:464-478.

Muir, D., G. Köck and X. Wang, 2012. Temporal trends of persistent organic pollutants and mercury in landlocked char in High Arctic lakes. In: Synopsis of Research Conducted under the 2011-2012 Northern Contaminants Program. Aboriginal Affairs and Northern Development Canada, pp. 207-215.

Muir, D., G. Köck and X. Wang, 2013. Temporal trends of persistent organic pollutants and mercury in landlocked char in High Arctic lakes. In: Synopsis of Research Conducted under the 2012-2013 Northern Contaminants Program. Aboriginal Affairs and Northern Development Canada, pp. 211-221.

Muir, D., G. Köck and X. Wang, 2014. Temporal trends of persistent organic pollutants and mercury in landlocked char in High Arctic lakes. In: Synopsis of Research Conducted under the 2013-2014 Northern Contaminants Program. Aboriginal Affairs and Northern Development Canada, pp. 251-260.

Murvoll, K.M., J.-U. Skaare, B. Moe, E. Anderssen and B.M. Jenssen, 2006. Spatial trends and associated biological responses of organochlorines and brominated flame retardants in hatchlings of North Atlantic kittiwakes (*Rissa tridactyla*). Environmental Toxicology and Chemistry, 25:1648-1656.

Murvoll, K.M., J.-U. Skaare, H. Jensen and B.M. Jenssen, 2007. Associations between persistent organic pollutants and vitamin status in Brunnich's guillemot and common eider hatchlings. Science of the Total Environment, 381:134-145.

Mustonen, A.M., T. Pyykönen, M. Puukka, J. Asikainen, S. Hänninen, J. Mononen and P. Nieminen, 2006. Physiological adaptations to fasting in an actively wintering canid, the Arctic blue fox (*Vulpes lagopus*). Journal of Experimental Zoology A, 305:32-46.

Neerland, E.D., 2016. DNA double-strand breaks in arctic char, *Salvelinus alpinus*, from Bjørnøya. MSc. Thesis. Norwegian University of Science and Technology, Trondheim, Norway.

Nielsen, E., J.C. Larsen and O. Ladefoged, 2006. Risk Assessment of Contaminant Intake from Traditional Greenland Food Items. Danish Institute for Food and Veterinary Research.

Nielsen, S.T., R. Mortensen, K. Hoydal, S.V. Erenbjerg and M. Dam, 2014. AMAP Faroe Islands Heavy Metals and POPs Core Programme 2009-2012. Food, Veterinary and Environmental Agency, Torshavn, Faroe Islands.

Noel, M., L.L. Loseto, C.C. Helbing, N. Veldhoen, N.J. Dangerfield and P.S. Ross, 2014. PCBs are associated with altered gene transcript profiles in Arctic beluga whales (*Delphinapterus leucas*). Environmental Science and Technology, 48:2942-2951.

Nomiyama, K., S. Hirakawa, A. Eguchi, C. Kanbara, D. Imaeda, J. Yoo, T. Kunisue, E.-Y. Kim, H. Iwato and S. Tanabe, 2014. Toxicological assessment of polychlorinated biphenyls and their metabolites in the liver of Baikal seal (*Pusa sibirica*). Environmental Science and Technology, 48:13530-13539.

Nordoy, E.S. and S.I. Thoresen, 2002. Reference values for serum biochemical parameters in free-ranging harp seals. Veterinary Clinical Pathology, 31:98-105.

Nordstad, T., M. Børge, J.O. Bustnes, C. Bech, O. Chastel, A. Goutte, K. Sagerup, C. Trouvé, D. Herzke and G.W. Gabrielsen, 2012. Relationships between POPs and baseline corticosterone levels in black-legged kittiwakes (*Rissa tridactyla*) across their breeding cycle. Environmental Pollution, 164:219-226.

Norman, S.A., L.A. Beckett, W.A. Miller, J. St Leger and R.C. Hobbs, 2013. Variation in hematologic and serum biochemical values of belugas (*Delphinapterus leucas*) under managed care. Journal of Zoo and Wildlife Medicine, 44:376-388.

Nøst, T.H., L.B. Helgason, M. Harju, E.S. Heimstad, G.W. Gabrielsen and B.M. Jenssen, 2012. Halogenated organic contaminants and their correlations with circulating thyroid hormones in developing Arctic seabirds. Science of the Total Environment, 414:248-256.

Noyes, P.D., M.K. McElwee, H.D. Miller, B.W. Clark, L.A. Van Tiem, K.C. Walcott, K.N. Erwin and E.D. Levin, 2009. The toxicology of climate change: Environmental contaminants in a warming world. Environment International, 35:971-986.

Nyman, M., M. Bergknut, M.L. Fant, H. Raunio, M. Jestoi, C. Bengs, A. Murk, J. Koistinen, C. Bäckman, O. Pelkonen, M. Tysklind, T. Hirvi and E. Helle, 2003. Contaminant exposure and effects in Baltic ringed and grey seals as assessed by biomarkers. Marine Environmental Research, 55:73-99.

Obbard, M.E., M.R. Cattet, T. Moody, L.R. Walton, D. Potter, J. Inglis and C. Chenier, 2006. Temporal trends in the body condition of Southern Hudson Bay polar bears. Climate Change Research Information Note, No. 3.

Oskam, I., E. Ropstad, E. Lie, A. Derocher, Ø. Wiig, E. Dahl, S. Larsen and J.U. Skaare, 2004. Organochlorines affect the steroid hormone cortisol in free-ranging polar bears (*Ursus maritimus*) at Svalbard, Norway. Journal of Toxicology and Environmental Health A, 67:959-977.

Ostertag, S.K., A.C. Shaw, N. Basu and H.M. Chan, 2014. Molecular and neurochemical biomarkers in Arctic beluga whales (*Delphinapterus leucas*) were correlated to brain mercury and selenium concentrations. Environmental Science and Technology, 48:11551-11559.

Outridge, P.M., H. Sanei, G.A. Stern, M. Goodsite, P.B. Hamilton, J. Carrie, F. Goodarzi and R.W. Macdonald, 2011. Comment on climate change and mercury accumulation in Canadian high and subarctic lakes. Environmental Science and Technology, 45:6703-6704.

Øverjordet, I.B., G.W. Gabrielsen, T. Berg, A. Ruus, A. Evenset, K. Borgå, G. Christensen, S. Lierhagen and B.M. Jenssen, 2015. Effect of diet, location and sampling year on bioaccumulation of mercury, selenium and cadmium in pelagic feeding seabirds in Svalbard. Chemosphere, 122:14-22.

Pacyna, J.M., I.T. Cousins, C. Halsall, A. Rautio, J. Pawlak, E.G. Pacyna, K. Sundseth, S. Wilson and J. Munthe, 2015. Impacts on human health in the Arctic owing to climate-induced changes in contaminant cycling – The EU ArcRisk project policy outcome. Environmental Science and Policy, 50:200-213.

Palmieri, J.R., C.F. Skinner and S. Elswaifi, 2012. Giardiasis revisited: An underappreciated reemerging zoonotic disease. The Journal of the American Osteopathic Association, 112:649-651.

Parmesan, C., C.M. Duarte, E. Poloczanska, A.J. Richardson and M.C. Singer, 2011. Overstretching attribution. Nature Climate Change, 1:2-4.

Pavlova, V., V. Grimm, R. Dietz, C. Sonne, K. Vorkamp, F.F. Rigét, R.J. Letcher, K. Gustavson and J. Nabe-Nielsen, 2016a. Modelling population level consequences of polychlorinated biphenyl exposure in East Greenland polar bears. Archives of Environmental Contamination and Toxicology, 70:143-154.

Pavlova, V., J. Nabe-Nielsen, R. Dietz, C. Sonne and V. Grimm, 2016b. Allee effect in polar bears: a potential consequence of polychlorinated biphenyl contamination. Proceedings of the Royal Society B, 283: Article 20161883.

Peakall, D.B., 1992. Animal Biomarkers as Pollution Indicators. Chapman and Hall.

Pedersen, F. and G.I,. Petersen, 1995. Variability of species sensitivity to complex mixtures. Water Science and Technology, 33:109-119.

Pedersen, K.E., N. Basu, R.J. Letcher, C. Sonne, R. Dietz and B. Styrishave, 2015. Brain region-specific perfluoroalkylated sulfonate (PFSA) and carboxylic acid (PFCA) accumulation and neurochemical biomarker responses in East Greenland polar bears (*Ursus maritimus*). Environmental Research, 138:22-31.

Pedersen, K.E., J.J. Weisser, R.J. Letcher, N. Basu, C. Sonne, R. Dietz and B. Styrishave, 2016. Per- and polyfluoroalkyl substances (PFASs) – New endocrine disruptors in polar bears (*Ursus maritimus*)? Environment International, 96:180-189.

Pedro, S., C. Boba, R. Dietz, C. Sonne, A. Rosing-Asvid, M. Hansen, A. Provatas and M.A. McKinney, 2017. Blubber-depth distribution and bioaccumulation of PCBs and organochlorine pesticides in Arctic-invading killer whales. Science of the Total Environment, 601-602:237-246.

Peng, Y., J.P. Wu, L. Tao, L. Mo, X.B. Zheng, B. Tang, X.J. Luo and B.X. Mai, 2015. Accumulation of Dechlorane Plus flame retardant in terrestrial passerines from a nature reserve in South China: The influences of biological and chemical variables. Science of the Total Environment, 514:77-82.

Perovich, D.K. and J.A. Richter-Menge, 2009. Loss of sea ice in the Arctic. Annual Review of Marine Science, 1:417-441.

Pertoldi, C., C. Sonne, Ø. Wiig, H.J. Baagøe, V. Loeschcke and T.Ø. Bechshøft, 2012. East Greenland and Barents Sea polar bears (*Ursus maritimus*): Adaptive variation between two populations using skull morphometrics as an indicator of gene flow. Hereditas, 149:99-107.

Pistone, K., I. Eisenman and V. Ramanathan, 2014. Observational determination of albedo decrease caused by vanishing Arctic sea ice. Proceedings of the National Academy of Sciences of the United States of America, 111:3322-3326.

Pithan, F. and T. Mauritsen, 2014. Arctic amplification dominated by temperature feedbacks in contemporary climate models. Nature Geoscience, 7:181-184.

Polischuk, S.C., R.J. Norstrom and M.A. Ramsay, 2002. Body burdens and tissue concentrations of organochlorines in polar bears (*Ursus maritimus*) vary during seasonal fasts. Environmental Pollution, 118:29-39.

Post, E., M.C. Forchhammer, M.S. Bret-Harte, T.V. Callaghan, T.R. Christensen, B. Elberling, A.D. Fox, O. Gil, D.S. Hik, T.T. Høye, R.A. Ims, E. Jeppesen, D.R. Klein, J. Madsen, A.D. McGuire, S. Rysgaard, D.E. Schindler, I. Stirling, M.P. Tamstorf, N.J.C. Tyler, R. van der Wal, J. Welker, P.A. Wookey, N.M. Schmidt and P. Aastrup, 2009. Ecological dynamics across the Arctic associated with recent climate change. Science, 325:1355-1358.

Post, E., 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, 2013. Ecological consequences of sea-ice decline. Science, 341:519-524.

Provencher, J.F., M.R. Forbes, H.L. Hennin, O.P. Love, B.M. Braune, M.L. Mallory and H.G. Gilchrist, 2016. Implications of mercury and lead concentrations on breeding physiology and phenology in an Arctic bird. Environmental Pollution, 218:1014-1022.

Provencher, J.F., M.R. Forbes, M.L. Mallory, S. Wilson and H.G. Gilchrist, 2017. Anti-parasite treatment, but not mercury burdens, influence nesting propensity dependent on arrival time or body condition in a marine bird. Science of the Total Environment, 575:849-857.

Rattner, B.A., V.P. Eroschenko, G.A. Fox, D.M. Fry and J. Gorsline, 1984. Avian endocrine response to environmental pollutants. Journal of Experimental Zoology, 232:683-689.

Redding, L.E., M.D. Sohn, T.E. McKone, J.W. Chen, S.L. Wang, D.P. Hsieh and R.S. Yang, 2008. Population physiologically based pharmacokinetic modelling for the human lactational transfer of PCB-153 with consideration of worldwide human biomonitoring results. Environmental Health Perspectives, 116:1629-1635.

Reeves, R. and G. Notarbartolo di Sciara (eds.), 2006. The Status and Distribution of Cetaceans in the Black Sea and Mediterranean Sea. IUCN Centre for Mediterranean Cooperation, Spain.

Regehr, E.V., N.J. Lunn, S.C. Amstrup and I. Stirling, 2007. Effects of earlier sea ice breakup on survival and population size of polar bears in western Hudson Bay. Journal of Wildlife Management, 71:2673-2683.

Regoli, F. and M.E. Giuliani, 2014. Oxidative pathways of chemical toxicity and oxidative stress biomarkers in marine organisms. Marine Environmental Research, 93:106-117.

Reist, J.D., F.J. Wrona, T.D. Prowse, M. Power, J.B. Dempson, R.J. Beamish, J.R. King, T.J. Carmichael and C.D. Sawatzky, 2006a. General effects of climate change on Arctic fishes and fish populations. Ambio, 35:370-380.

Reist, J.D., F.J. Wrona, T.D. Prowse, M. Power, J.B. Dempson, J.R. King and R.J. Beamish, 2006b. An overview of effects of climate change on selected Arctic freshwater and anadromous fishes. Ambio, 35:381-387.

Rigét, F., K. Vorkamp and D. Muir, 2010. Temporal trends of contaminants in Arctic char (*Salvelinus alpinus*) from a small lake, southwest Greenland during a warming climate. Journal of Monitoring, 12:2252-2258.

Rigét, F.F., B. Braune, A. Bignert, S. Wilson, J. Aars, M. Andersen, G. Asmund, A. Aubail, M. Dam, R. Dietz, M. Evans, T. Evans, M. Gamberg, N. Gantner, N. Green, H. Gunnlaugsdóttir, K. Kannan, R.J. Letcher, D. Muir, K. Ólafsdóttir, A. Renzoni, P. Roach, C. Sonne, G. Stern and Ø. Wiig, 2011. Temporal trends of Hg in Arctic biota, an update. Science of the Total Environment, 409:3520-3526.

Rigét, F.F., R. Dietz and K.A. Hobson, 2012. Temporal trends of mercury in Greenland ringed seal populations in a warming climate. Journal of Environmental Monitoring, 14:3249-3256.

Rigét, F.F., K. Vorkamp, K.A. Hobson, D.C.G. Muir and R. Dietz, 2013. Temporal trends of selected POPs and the potential influence of climate variability in a Greenland ringed seal population. Environmental Science-Processes and Impacts, 15:1706-1716.

Rogstad, T.W., C. Sonne, G.D. Villanger, Ø. Ahlstrøm, E. Fuglei, D.C.G. Muir, E. Jørgensen and B.M. Jenssen, 2017. Concentrations of vitamin A, E, thyroid hormones and testosterone in blood plasma and tissues from emaciated adult male Arctic foxes (*Vulpes lagopus*) dietary exposed to persistent organic pollutants (POPs). Environmental Research, 154:284-290.

Rokicki, J., 2009. Effects of climatic changes on anisakid nematodes in polar regions. Polar Science, 3:197-201.

Ronald, K., S.V. Tessaro, J.F. Uthe, H.C. Freeman and R. Frank, 1977. Methylmercury poisoning in the harp seal (*Pagophilus groenlandicus*). Science of the Total Environment, 8:1-11.

Ropstad, E., I.C. Oskam, J.L. Lyche, H.J. Larsen, E. Lie and M. Haave, 2006. Endocrine disruption induced by organochlorines (OCs): field studies and experimental models. Journal of Toxicology and Environmental Health A, 69:53-76.

Rosabal, M., L. Hare, P.G. Campbell, 2012. Subcellular metal partitioning in larvae of the insect *Chaoborus* collected along an environmental metal exposure gradient (Cd, Cu, Ni and Zn). Aquatic Toxicology, 120:67-78.

Rosabal, M., F. Pierron, P. Couture, M. Baudrimont, L. Hare and P.G. Campbel, 2015. Subcellular partitioning of non-essential trace metals (Ag, As, Cd, Ni, Pb, Tl) in livers of American (*Anguilla rostrata*) and European (*Anguilla anguilla*) yellow eels. Aquatic Toxicology, 160:128-141.

Ross, P., R.L. De Swart, P.J. Reijnders, H. Van Loveren, J. Vos and A.D. Osterhaus, 1995. Contaminant-related suppression of delayed-type hypersensitivity and antibody responses in harbor seals fed herring from the Baltic Sea. Environmental Health Perspectives, 103:162-167.

Routti, H., M. Nyman, B.M. Jenssen, C. Backman, J. Koistinen and G.W. Gabrielsen, 2008a. Bone-related effects of contaminants in seals may be associated with vitamin D and thyroid hormones. Environmental Toxicology and Chemistry, 27:873-880.

Routti, H., R.J. Letcher, A. Arukwe, B. van Bavel, N.G. Yoccoz, S.G. Chu and G.W. Gabrielsen, 2008b. Biotransformation and metabolites of PCBs in relation to Phase I and II xenobiotic-metabolizing enzyme activities in ringed seals (*Phoca hispida*) from Svalbard and the Baltic Sea. Environmental Science and Technology, 42:8952–8958.

Routti, H., A. Arukwe, B.M. Jenssen, R.J. Letcher, M. Nyman, C. Bäckman and G.W. Gabrielsen, 2010a. Comparative endocrine disruptive effects of contaminants in ringed seals (*Phoca hispida*) from Svalbard and the Baltic Sea. Comparative Biochemistry and Physiology C, 152:306-312.

Routti, H., B.M. Jenssen, C. Lydersen, C. Backman, A. Arukwe, M. Nyman, K.M. Kovacs and G.W. Gabrielsen, 2010b. Hormone, vitamin and contaminant status during the moulting/fasting period in ringed seals (*Pusa* [*Phoca*] *hispida*) from Svalbard. Comparative Biochemistry and Physiology A, 155:70-76.

Routti, H., R.J. Letcher, E.W. Born, M. Branigan, R. Dietz, T.J. Evans, A.T. Fisk, E. Peacock and C. Sonne, 2011. Spatial and temporal trends of selected trace elements in liver tissue from polar bears (*Ursus maritimus*) from Alaska, Canada and Greenland. Journal of Environmental Monitoring, 13:2260-2267.

Routti, H., R.J. Letcher, E.W. Born, M. Branigan, R. Dietz, T.J. Evans, M.A. McKinney, E. Peacock and C. Sonne, 2012. Influence of carbon and lipid sources on variation of mercury and other trace elements in polar bears (*Ursus maritimus*). Environmental Toxicology and Chemistry, 31:2739-2747.

Sagerup, K., K. Åsbakk, A. Polder, J.U. Skaåre, G.W. Gabrielsen and R.T. Barrett, 2014. Relationships between persistent organic pollutants and circulating immunoglobulin-y in black-legged kittiwakes and Atlantic puffins. Journal of Toxicology and Environmental Health A, 77:481-494.

Sandell, H.T., B. Sandell, E.W. Born, R. Dietz and S. Sonne-Hansen, 2001. Isbjørne i Østgrønland: En interviewundersøgelse om forekomst og fangst. Grønlands Naturinstitut Teknisk Rapport, No. 40.

Sandheinrich, M.B. and J.G. Wiener, 2011. Methylmercury in freshwater fish: recent advances in assessing toxicity of environmentally relevant exposures. In: Beyer, W.N. and J.P. Meador (Eds.), Environmental Contaminants in Biota: Interpreting Tissue Concentrations. 2nd Edition. Taylor and Francis, pp. 169-190.

Sarazin, M., C. Alexandre and T. Thomas, 2000. Influence on bone metabolism of dietary trace elements, protein, fat, carbohydrates and vitamins. Joint Bone Spine, 67:408-418.

Saunes, H., 2011. Do mercury, selenium, cadmium and zinc cause oxidative stress in common eiders (*Somateria mollissima*) from Svalbard? MSc Thesis, Norwegian University of Science and Technology.

Scheuhammer, A.M., M.W. Meyer, M.B. Sandheinrich and M.W. Murray, 2007. Effects of environmental methylmercury on the health of wild birds, mammals, and fish. Ambio, 36:12-18.

Scheuhammer, A.M., N. Basu, D.C. Evers, G.H. Heinz, M.B. Sandheinrich and M.S. Bank, 2011. Ecotoxicology of mercury in fish and wildlife: Recent advances. In: Bank, M. (Ed.), Mercury in the Environment: Pattern and Process. University of California Press, pp. 223-238.

Scheuhammer, A.M., B. Braune, H.M. Chan, H. Frouin, A. Krey, R.J. Letcher, L.L. Loseto, M. Noël, S. Ostertag, P. Ross and M. Wayland, 2015. Recent progress on our understanding of the biological effects of mercury in fish and wildlife in the Canadian Arctic. Science of the Total Environment, 509-510:91-103.

Scholz, T., 2009. Update on the human broad tapeworm (genus *Diphyllobothrium*), including clinical relevance. Clinical Microbiology Reviews, 22:146-160.

Schwacke, L.H., E.S. Zolman, B.C. Balmer, S. De Guise, R.C. George and J. Hoguet, 2012. Hypothyroidism and immune suppression associated with polychlorinated biphenyl exposure in bottlenose dolphins (*Tursiops truncatus*). Proceedings of the Royal Society B, 279:48-57.

Screen, J.A. and I. Simmonds, 2010. The central role of diminishing sea ice in recent Arctic temperature amplification. Nature, 464:1334-1337.

Selye, H., 1973. The evolution of the stress concept. American Scientist, 61:692-699.

Serreze, M.C. and R.G. Barry, 2011. Processes and impacts of Arctic amplification: A research synthesis. Global Planetary Change, 77:85-96.

Shope, R., 1991. Global climate change and infectious diseases. Environmental Health Perspectives, 96:171-174.

Siebert, U., B. Pozniak, K.A. Hansen, G. Nordstrom, J. Teilmann, N. van Elk, A. Vossen and R. Dietz, 2011. Investigations of thyroid and stress hormones in free-ranging and captive harbor porpoises (*Phocoena phocoena*): A pilot study. Aquatic Mammals, 37:443-453.

Simon, E., J. Bytingsvik, W. Jonker, P.E.G. Leonards, J. de Boer and B.M. Jenssen, 2011. Blood plasma sample preparation method for the assessment of thyroid hormone-disrupting potency in effect-directed analysis. Environmental Science and Technology, 45:7936-7944.

Simon, E., M. van Velzen, S.H. Brandsma, E. Lie, K. Loken and J. de Boer, 2013. Effect-directed analysis to explore the polar bear exposome: Identification of thyroid hormone disrupting compounds in plasma. Environmental Science and Technology, 47:8902-8912.

Simoneau, M., M. Lucotte, S. Garceau and D. Laliberté, 2005. Fish growth rates modulate mercury concentrations in walleye (*Sander vitreus*) from eastern Canadian lakes. Environmental Research, 98:73-82.

Sjölund, M., J. Bonnedahl, J. Hernandez, S. Bengtsson, G. Cederbrant, J. Pinhassi, G. Kahlmeter and B. Olsen, 2008. Dissemination of multidrug-resistant bacteria into the Arctic. Emerging Infectious Diseases, 14:70-72.

Sletten, S., S. Bourgeon, B.-J. Badsen, D. Herzke, F. Criscuolo, S. Massemin, A. Zahn, T.V. Johnsen and J.-O. Bustnes, 2016. Organohalogenated contaminants in white-tailed eagle (*Haliaeetus albicilla*) nestlings: An assessment of relationships to immunoglobulin levels, telomeres and oxidative stress. Science of the Total Environment, 539:337-349.

Solomon, C., 2014. Sickness spreads across the Arctic. Scientific American, 311:58-63.

Sonenshine, D.E. and T.N. Mather, 1994. Ecological Dynamics of Tick-borne Zoonoses. Oxford University Press.

Sonne, C., 2010. Health effects from long-range transported contaminants in Arctic top predators: An integrated review based on studies of polar bears and relevant model species. Environment International, 36:461-491.

Sonne, C., R. Dietz, H.J.S. Larsen, K.E. Loft, M. Kirkegaard, R.J. Letcher, S. Shahmiri and P. Møller, 2006a. Impairment of cellular immunity in West Greenland sledge dogs (*Canis familiaris*) dietary exposed to polluted minke whale (*Balaenoptera acutorostrata*) blubber. Environmental Science and Technology, 40:2056-2062.

Sonne, C., P.S. Leifsson, R. Dietz, E.W. Born, R.J. Letcher, L. Hyldstrup, F.F. Riget, M. Kirkegaard and D.C.G Muir, 2006b. Xenoendocrine pollutants may reduce size of sexual organs in East Greenland polar bears (*Ursus maritimus*). Environmental Science and Technology, 40:5668-5674.

Sonne, C., R. Dietz, M. Kirkegaard, R.J. Letcher, S. Shahmiri, S. Andersen, P. Moller, A.K. Olsen and A.L. Jensen, 2008. Effects of organohalogen pollutants on haematological and urine clinical-chemical parameters in Greenland sledge dogs (*Canis familiaris*). Ecotoxicology and Environmental Safety, 69:381-390.

Sonne, C., K. Gustavson, F.F. Rigét, R. Dietz, M. Birkved, R.J. Letcher, D.C.G. Muir, R. Bossi, K. Vorkamp, E.W. Born and G. Petersen, 2009. Reproductive performance in East Greenland polar bears (*Ursus maritimus*) may be affected by organohalogen contaminants as shown by critical body residue modelling and risk quotients estimation. Chemosphere, 77:1558-1568.

Sonne, C., M. Dam, P.L. Leifsson and R. Dietz, 2010a. Liver and renal histopathology of North Atlantic long-finned pilot whales (*Globicephala melas*) with high concentrations of heavy metals and organochlorines. Toxicology and Environmental Chemistry, 92:969-985.

Sonne, C., J. Verreault, G.W. Gabrielsen, R.J. Letcher, P.L. Leifsson and T. Iburg, 2010b. Screening of thyroid gland histology in organohalogen-contaminated glaucous gulls (*Larus hyperboreus*) from the Norwegian Arctic. Toxicology and Environmental Chemistry, 92:1705-1713.

Sonne, C., J.O. Bustnes, D. Herzke, V.L.B. Jaspers, A. Covaci, D.J. Halley, T. Moum, I. Eulaers, M. Eens, R.A. Ims and S.A. Hanssen, 2010c. Relationships between organohalogen contaminants and blood plasma clinical–chemical parameters in chicks of three raptor species from Northern Norway. Ecotoxicology and Environmental Safety, 73:7-17.

Sonne, C., P.S. Leifsson, T. Iburg, R. Dietz, E.W. Born, R.J. Letcher and M. Kirkegaard, 2011. Thyroid gland lesions in organohalogen contaminated East Greenland polar bears (*Ursus maritimus*). Toxicology and Environmental Chemistry, 93:789-805.

Sonne, C., R. Dietz, T.Ø. Bechshøft, M.A. McKinney, P.S. Leifsson, E.W. Born, F.F. Rigét, R.J. Letcher and D.C.G. Muir, 2012a. Monitoring liver and kidney morphology in East Greenland polar bears (*Ursus maritimus*) during 1999-2010. Environment International, 48:143-149.

Sonne, C., J.O. Bustnes, D. Herzke, V.L.B. Jaspers, A. Covaci, I. Eulaers, D.J. Halley, T. Moum, M. Ballesteros, M. Eens and R.A. Ims, 2012b. Blood plasma clinical–chemical parameters as biomarker endpoints for organohalogen contaminant exposure in Norwegian raptor nestlings. Ecotoxicology and Environmental Safety, 80:76-83.

Sonne, C., P.S. Leifsson and R. Dietz, 2013a. Liver and renal lesions in mercury contaminated narwhals (*Monodon monoceros*) from North West Greenland. Toxicology and Environmental Chemistry, 95:515-528.

Sonne, C., T.Ø. Bechshøft, F.F. Rigét, H.J. Baagøe, A. Hedayat, M. Andersen, J.E. Bech-Jensen, L. Hyldstrup, R.J. Letcher and R. Dietz, 2013b. Size and density of East Greenland polar bear (*Ursus maritimus*) skulls as bio-indicators of environmental changes. Ecological Indicators, 34:290-295.

Sonne, C., S.A.B. Mæhre, K. Sagerup, M. Harju, E.S. Heimstad, P.S. Leifsson, R. Dietz and G.W. Gabrielsen, 2013c. A screening of liver, kidney and thyroid gland morphology in organohalogen-contaminated glaucous gulls (*Larus hyperboreus*) from Svalbard. Toxicology and Environmental Chemistry, 95:172-186.

Sonne, C., F.F. Rigét, E.H. Leat, S. Bourgeon, K. Borgå, H. Strøm, S.A. Hanssen, G.W. Gabrielsen, A. Petersen, K. Olafsdottir and E. Magnusdottir, 2013d. Organohalogen contaminants and blood plasma clinical–chemical parameters in three colonies of North Atlantic great skua (*Stercorarius skua*). Ecotoxicology and Environmental Safety, 92:245-251.

Sonne, C., M. Kirkegaard, J. Jacobsen, B.M. Jenssen, R.J. Letcher and R. Dietz, 2014a. Altered 25-hydroxyvitamin D3 in liver tissue from Greenland sledge dogs (*Canis familiaris*) dietary exposed to organohalogen polluted minke whale (*Balaenoptera acuterostrata*) blubber. Ecotoxicology and Environmental Safety, 104:403-408.

Sonne, C., R. Dietz, F.F. Rigét, R.J. Letcher, K. Munk Pedersen and B. Styrishave, 2014b. Steroid hormones in blood plasma from Greenland sledge dogs (*Canis familiaris*) dietary exposed to organohalogen polluted minke whale (*Balaenoptera acutorostrata*) blubber. Toxicology and Environmental Chemistry, 96:273-286.

Sonne, C., L. Bach, J. Sondergaard, F.F. Riget, R. Dietz, A. Mosbech, P.S. Leifsson and K. Gustavson, 2014c. Evaluation of the use of common sculpin (*Myoxocephalus scorpius*) organ histology as bioindicator for element exposure in the fjord of the mining area Maarmorilik, West Greenland. Environmental Research, 133:304-311.

Sonne, C., M. Dyck, F.F. Rigét, J.E. Bech-Jensen, L. Hyldstrup, R.J. Letcher, K. Gustavson, M.T.P. Gilbert and R. Dietz, 2015a. Penile density and globally used chemicals in Canadian and Greenland polar bears. Environmental Research, 137:287-291.

Sonne, C., K. Gustavson, R. Dietz and R.J. Letcher, 2015b. Physiologically-based pharmacokinetic modelling and verification of distribution, elimination and bioaccumulation of persistent organic pollutants in Greenland sledge dogs (*Canis familiaris*). Environmental Research, 142:380-386.

Sonne, C., K. Gustavson, R. Dietz and R.J. Letcher, 2016. Risk evaluation of the Arctic environmental POP exposure based on critical body residue and critical daily dose using captive Greenland sledge dogs (*Canis familiaris*) as surrogate species. Environment International, 88:221-227.

Sonne, C, P.S. Leifsson, J. Søndergaard and R. Dietz, 2018. Hepatic and renal histology and mercury concentrations of North West and North East Greenland narwhals (*Monodon monoceros*). Journal of Toxicology and Environmental Health A, 81:202-211.

Stern, G.A., R.W. Macdonald, P.M. Outridge, S. Wilson, J. Chetelat, A. Cole, H. Hintelmann, L.L. Loseto, A. Steffen and F. Wang, 2012. How does climate change influence arctic mercury? Science of the Total Environment, 414:22-42.

Stirling, I. and A.E. Derocher, 2012. Effects of climate warming on polar bears: a review of the evidence. Global Change Biology, 18:2694-2706.

Stirling, I., N.J. Lunn and J. Iacozza, 1999. Long-term trends in the population ecology of polar bears in western Hudson Bay in relation to climatic change. Arctic, 1:294-306.

Strid, A., H. Jorundsdottir, O. Papke, J. Svavarsson and A. Bergman, 2007. Dioxins and PCBs in Greenland shark (*Somniosus microcephalus*) from the North-East Atlantic. Marine Pollution Bulletin, 54:1514-1522.

Swanson, H., N. Gantner, K.A. Kidd, D.C.G. Muir and J.D. Reist, 2011. Comparison of mercury concentrations in landlocked, resident, and sea-run fish (*Salvelinus* spp.) from Nunavut, Canada. Environmental Toxicology and Chemistry, 30:1459-1467.

Sweet, L.I. and J.T. Zelikoff, 2001. Toxicology and immunotoxicology of mercury: a comparative review in fish and humans. Journal of Toxicology and Environmental Health B, 4:161-205.

Talbott, S.M., M. Cifuentes, M.G. Dunn and S.A. Shapses, 2001. Energy restriction reduces bone density and biomechanical properties in aged female rats. Journal of Nutrition, 131:2382-2387.

Tartu, S., A. Goutte, P. Bustamante, F. Angelier, B. Moe, C. Clement-Chastel, C. Bech, G.W. Gabrielsen, J.-O. Bustnes and O. Chastel, 2013. To breed or not to breed: endocrine response to mercury contamination by an Arctic seabird. Biology Letters, 9:20130317.

Tartu, S., G.W. Gabrielsen, P. Blévin, H. Ellis, J.O. Bustnes, D. Herzke and O. Chastel, 2014a. Endocrine and fitness correlates of long-chain perfluorinated carboxylates exposure in Arctic breeding black-legged kittiwakes. Environmental Science and Technology, 48:13504-13510.

Tartu, S., F. Angelier, D. Herzke, B. Moe, C. Bech, G.W. Gabrielsen, J.O. Bustnes and O. Chastel, 2014b. The stress of being contaminated? Adrenocortical function and reproduction in relation to persistent organic pollutants in female black-legged kittiwakes. Science of the Total Environment, 476-477:553-560.

Tartu, S., F. Angelier, J.O. Bustnes, B. Moe, S.A. Hanssen, D. Herzke, G.W. Gabrielsen, N. Verboven, J. Verreault, P. Labadie, H. Budzinski, J.C. Wingfield and O. Chastel, 2015a. Polychlorinated biphenyl exposure and corticosterone levels in seven polar seabird species. Environmental Pollution, 197:173-180.

Tartu, S., F. Angelier, J.O. Bustnes, B. Moe, S.A. Hanssen, D. Herzke, G.W. Gabrielsen, N. Verboven, J. Verreault, P. Labadie and H. Budzinski, 2015b. Polychlorinated biphenyl exposure and corticosterone levels in seven polar seabird species. Environmental Pollution, 197:173-180.

Tartu, S., P. Bustamante, F. Angelier, A.Z. Lendvai, B. Moe, P. Blevin, C. Bech, G.W. Gabrielsen, J.-O. Bustnes and O. Chastel, 2016. Mercury exposure, stress and prolactin secretion in an Arctic seabird: An experimental study. Functional Ecology, 30:596-604.

Tartu, S., R. Lille-Langoy, T.R. Storseth, S. Bourgeon, A. Brunsvik, J. Aars, A. Goksoyr, B.M. Jenssen, A. Polder, G.W. Thiemann, V. Torget and H. Routti, 2017a. Multiple-stressor effects in an apex predator: combined influence of pollutants and sea ice decline on lipid metabolism in polar bears. Scientific Reports, 7:Article 16487.

Tartu, S., S. Bourgeon, J. Aars, M. Andersen, A. Polder, G.W. Thiemann, J.M. Welker and H. Routti, 2017b. Sea ice-associated decline in body condition leads to increased concentrations of lipophilic pollutants in polar bears (*Ursus maritimus*) from Svalbard, Norway. Science of the Total Environment, 576:409-419.

Thompson, D.R., 1996. Mercury in birds and terrestrial mammals. In: Beyer, W.N., G. Heinz, A.W. Redmon-Norwood (Eds.), Environmental Contaminants in Wildlife: Interpreting Tissue Concentrations. CRC Press, pp. 341-356.

Thrall, M.A., D.C. Baker, T.W. Campbell, D. DeNicola, M.J. Fettman, E.D. Lassen, A. Rebar and G. Weiser, 2004. Veterinary Hematology and Clinical Chemistry. First Edition. Lippincott Williams & Wilkins.

Tryland, M., B.A. Krafft, C. Lydersen, K.M. Kovacs and S.I. Thoresen, 2006a. Serum chemistry values for free-ranging ringed seals (*Pusa hispida*) in Svalbard. Veterinary Clinical Pathology, 35:405-412.

Tryland, M., S.I. Thoresen, K.M. Kovacs and C. Lydersen, 2006b. Serum chemistry of free-ranging white whales (*Delphinapterus leucas*) in Svalbard. Veterinary Clinical Pathology, 35:199-203.

Tung, S. and J. Iqbal, 2007. Evolution, aging, and osteoporosis. Annals of the New York Academy of Sciences, 1116:499-506.

UNEP/AMAP, 2011. Climate change and POPs: Predicting the impacts. Report of the UNEP/AMAP Expert Group. Secretariat of the Stockholm Convention, 2011, Geneva, Switzerland.

US EPA, 2008. Toxicological Review of 2,2',4,4',5-Pentabromodiphenyl ether (BDE-99) (CAS No. 60348-60-9) In Support of Summary Information on the Integrated Risk Information System (IRIS), U.S. Environmental Protection Agency Washington, DC. https://cfpub.epa.gov/ncea/iris/iris_documents/documents/toxreviews/1008tr.pdf

van den Berghe, M., L. Weijs, S. Habran, K. Das, C. Bugli and S. Pillet, 2013. Effects of polychlorobiphenyls, polybromodiphenylethers, organochlorine pesticides and their metabolites on vitamin A status in lactating grey seals. Environmental Research, 120:18-26.

Vazquez-Medina, J.P., T. Zenteno-Savin, R. Elsner and R.M. Ortiz, 2012. Coping with physiological oxidative stress: a review of antioxidant strategies in seals. Journal of Comparative Physiology B, 182:741-750.

Venne, H. and P. Magnan, 1989. Life history tactics in landlocked arctic charr (*Salvelinus alpinus*): a working hypothesis. Physiology and Ecology Japan, 1:239-248.

Verboven, N., J. Verreault, R.J. Letcher, G.W. Gabrielsen and N. Evans, 2009. Nest temperature and incubation behavior of arctic-breeding glaucous gulls exposed to persistent organic pollutants. Animal Behaviour, 77:411-418.

Verboven, N., J. Verreault, R.J. Letcher, G.W. Gabrielsen and N.P. Evans, 2010. Adrenocortical function of Arctic-breeding glaucous gulls in relation to persistent organic pollutants. General and Comparative Endocrinology, 166:25-32.

Verner, M.A., P. Ayotte, G. Muckle, M. Charbonneau and S. Haddad, 2009. A physiologically based pharmacokinetic model for the assessment of infant exposure to persistent organic pollutants in epidemiologic studies. Environmental Health Perspectives, 117:481-487.

Verreault, J., J.U. Skaare, B.M. Jenssen and G.W. Gabrielsen, 2004. Effects of organochlorine contaminants on thyroid hormone levels in Arctic breeding glaucous gulls, *Larus hyperboreus*. Environmental Health Perspectives, 112:532-537.

Verreault, J., R.J. Letcher, E. Ropstad, E. Dahl and G.W. Gabrielsen, 2006. Organohalogen contaminants and reproductive hormones in incubating glaucous gulls (*Larus hyperboreus*) from the Norwegian arctic. Environmental Toxicology and Chemistry, 25:2990-2996.

Verreault, J., C. Bech, R.J. Letcher, E. Ropstad, E. Dahl and G.W. Gabrielsen, 2007. Organohalogen contamination in breeding glaucous gulls from the Norwegian Arctic: Associations with basal metabolism and circulating thyroid hormones. Environmental Pollution, 145:138-145.

Verreault, J., N. Verboven, G.W. Gabrielsen, R.J. Letcher and O. Chastel, 2008. Changes in prolactin in a highly organohalogen contaminated Arctic top predator seabird, the glaucous gull. General and Comparative Endocrinology, 156:569-576.

Verreault, J., L.B. Helgason, G.W. Gabrielsen, M. Dam and B.M. Braune, 2013. Contrasting retinoid and thyroid hormone status in differentially-contaminated northern fulmar colonies from the Canadian Arctic, Svalbard and the Faroe Islands. Environment International, 52:29-40.

Villanger, G.D., B.M. Jenssen, R.R. Fjeldberg, R.J. Letcher, D.C.G. Muir, M. Kirkegaard, C. Sonne and R. Dietz, 2011a. Organohalogen contaminants in relation to circulating thyroid hormones levels in polar bears from East Greenland. Environment International, 37:694-708.

Villanger, G.D., C. Lydersen, K.M. Kovacs, E. Lie, J.U. Skaare and B.M. Jenssen, 2011b. Disruptive effects of persistent organohalogen contaminants on thyroid function in white whales (*Delphinapterus leucas*) from Svalbard. Science of the Total Environment, 409:2511-2524.

Villanger, G.D., K.M. Gabrielsen, K.M. Kovacs, C. Lydersen, E. Lie and M. Karimi, 2013. Effects of complex organohalogen contaminant mixtures on thyroid homeostasis in hooded seal (*Cystophora cristata*) mother-pup pairs. Chemosphere, 92:828-842.

Walkowiak, J., J.A. Wiener, A. Fastabend, B. Heinzow, U. Krämer, E. Schmidt, H.J. Steingrüber, S. Wundram and G. Winneke, 2001. Environmental exposure to polychlorinated biphenyls and quality of the home environment: effects on psychodevelopment in early childhood. Lancet, 358:1602-1607.

Wallace, W.G., B.-G. Lee and S.N. Luoma, 2003. Subcellular compartmentalization of Cd and Zn in two bivalves. I. Significance of metal-sensitive fractions (MSF) and biologically detoxified metal (BDM). Marine Ecology Progress Series, 249:183-197.

Wang, D., W.L. Shelver, S. Atkinson, J.-A.E. Mellish and Q. Li, 2010. Tissue distribution of polychlorinated biphenyls and organochlorine pesticides and potential toxicity to Alaskan northern fur seals assessed using PCBs congener specific mode of action schemes. Archives of Environmental Contamination and Toxicology, 58:478-488.

Wassmann, P., C.M. Duarte, S. Agusti and M.K. Sejr, 2011. Footprints of climate change in the Arctic marine ecosystem. Global Change Biology, 17:1235-1249.

Wayland, M., K.L. Drake, R.T. Alisauskas, D.K. Kellett, J. Traylor, C. Swoboda and K. Mehl, 2008. Survival rates and blood metal concentrations in two species of free-ranging North American sea ducks. Environmental Toxicology and Chemistry, 27:698-704.

Wayland, M., D.J. Hoffman, M.L. Mallory, R.T. Alisauskas and K.R. Stebbins, 2010. Evidence of weak contaminant-related oxidative stress in glaucous gulls (*Larsus hyperboreus*) from the Canadian Arctic. Journal of Toxicology and Environmental Health A, 73:1058-1073.

Weber, D., P.J. Van Coeverden De Groot, E. Peacock, M. Schrenzel, D. Perez, S. Thomas, J. Shelton, C. Else, L. Darby and L. Acosta, 2013. Low MHC variation in the polar bear: implications in the face of Arctic warming? Animal Conservations, 16:671-683.

Weijs, L., A. Covaci, R.S.H. Yang, K. Das and R. Blust, 2012. Computational toxicology: Physiologically based pharmacokinetic models (PBPK) for lifetime exposure and bioaccumulation of polybrominated diphenyl ethers (PBDEs) in marine mammals. Environmental Pollution, 163:134-141.

Weijs, L., A.C. Roach, R.S.H. Yang, R. McDougall, M. Lyons, C. Housand, D. Tibax, T. Manning, J. Chapman, K. Edge, A. Covaci and R. Blust, 2014. Lifetime PCB 153 bioaccumulation and pharmacokinetics in pilot whales: Bayesian population PBPK modeling and Markov chain Monte Carlo simulations. Chemosphere, 94:91-96.

Wingfield, J.C. and R.M. Sapolsky, 2003. Reproduction and resistance to stress: When and how. Journal of Neuroendocrinology, 15:711-724.

Wiseman, S., E.H. Jorgensen, A.G. Maule and M.M. Vijayan, 2011. Contaminant loading in remote Arctic lakes affects cellular stress-related proteins expression in feral charr. Polar Biology, 34:933-937.

Wiseman, S.B., Y.H. He, M.G.E. Din, J.W. Martin, P.D. Jones, M. Hecker and J.P. Giesy, 2013. Transcriptional responses of male fathead minnows exposed to oil sands process-affected water. Comparative Biochemistry and Physiology C, 157:227-235.

Wobeser, G., N.O. Nielsen and B. Schiefer, 1976. Mercury and mink II. Experimental methyl mercury intoxication. The Canadian Journal of Comparative Medicine, 40:34-45.

Wolfe, M.F., S. Schwarzbach and R.A. Sulaiman, 1998. Effects of mercury on wildlife: a comprehensive review. Environmental Toxicology and Chemistry, 17:146-160.

Wolkers, H., P.J. Corkeron, S.M. van Parijs, T. Simila and B. van Bavel, 2007. Accumulation and transfer of contaminants in killer whales (*Orcinus orca*) from Norway: indications for contaminant metabolism. Environmental Toxicology and Chemistry, 26:1582-1590.

Wren, C.D., D.B. Hunter, J.F. Leatherland and P.M. Stokes, 1987. The effects of polychlorinated biphenyls and methylmercury, singly and in combination, on mink. I: Uptake and toxic responses. Archives of Environmental Contamination and Toxicology, 16:441-447.

Wrona, F.J., T.D. Prose, J.D. Reist, R. Beamish, J.J. Gibson, J. Hobbie, E. Jeppsen, J. King, G. Koeck, A. Korhola, L. Levesque, R. Macdonald, M. Power, V. Skvortsov and W. Voncent, 2005. Freshwater ecosystems and fisheries. In: Arctic Climate Impact Assessment. Cambridge University Press, pp. 353-452.

Yano, K., J.D. Stevens and L.D.V. Compagno, 2007. Distribution, reproduction and feeding of the Greenland shark *Somniosus* (*Somniosus*) *microcephalus*, with notes on two other sleeper sharks, *Somniosus* (*Somniosus*) *pacificus* and *Somniosus* (*Somniosus*) *antarcticus*. Journal of Fish Biology, 70:374-390.

Zoeller, R.T., A.L.S. Dowling, C.T.A. Herzig, E.A. Iannacone, K.J. Gauger and R. Bansal, 2002. Thyroid hormone, brain development, and the environment. Environmental Health Perspectives, 110:355-361.

Annex 1

Summary of studies on wildlife examining global climate change-ecological-POP/Hg linkages (adapted from McKinney et al., 2015).

Study species	Year	Location	Climate metric
PCBs			
Terrestrial mammals			
Arctic fox	1997–2013	Svalbard	Sea-ice extent
Marine mammals			
Ringed seal	1993–2008	Eastern Amundsen Gulf	Sea ice break-up date
	1993–2008	Eastern Amundsen Gulf	Sea ice break-up date
	1994–2010	West Greenland	Arctic Oscillation, Ocean temperature, Salinity, Sea-ice cover
Polar bear	1991–2007	Western Hudson Bay	Sea ice break-up date
	1984–2011	East Greenland	North Atlantic Oscillation
Marine birds			
Glaucous gull	1997–2006	Bjørnøya	Arctic Oscillation
Common eider	2005–2009	Northern Norway and Svalbard	Air temperature
Thick-billed murre	1975–2013	Canadian High Arctic	
	1993–2013	Northern Hudson Bay	Sea-ice conditions
Hg			
Marine mammals			
Ringed seal	1994–2010	Central West, North West, and East Greenland	Arctic Oscillation, Ocean temperature, Salinity, Sea-ice cover
	1973–2007	Eastern Amundsen Gulf	Ice-free season length
Atlantic walrus	1982–2008	Foxe Basin	North Atlantic Oscillation
Beluga	1981–2012	Beaufort Sea	Arctic Oscillation, Pacific Decadal Oscillation, Sea-ice minimum
	1984–2008	Hudson Bay	North Atlantic Oscillation
Narwhal	1993–2001	Foxe Basin	North Atlantic Oscillation
Marine birds			
Black-legged kittiwake	2008–2009	Svalbard	Sea-ice cover
Little auk	2008–2009	Svalbard	None identified
Thick-billed murre	1975–2013	Canadian High Arctic	None identified
	1993–2013	Northern Hudson Bay	Sea-ice conditions

Ecological change/variation	PCB/Hg		Reference
	Influence	Variation/change	
Diet (marine vs terrestrial)	ΣPCB	↔	Andersen et al. 2015
Prey availability or type	CB31, CB52, CB101, CB118, CB138, CB153, CB180	↑ in years of earlier break-up	Gaden et al. 2012
Prey availability or type	CB28, CB105, CB156	↔	Gaden et al. 2012
Prey availability or type	CB153 CB52, CB153	↑ in years of ↓ ice ↑ with ↑ AO or ↑ salinity (related to abiotic inputs)	Rigét et al. 2013
Diet (subarctic vs Arctic seals)	ΣPCB	↑ instead of ↓ trend	McKinney et al. 2009
Diet (subarctic vs Arctic seals)	ΣPCB, CB170/CB190, CB180, CB153	Not significantly slower rate of ↓ trend	McKinney et al. 2013
Possibly foraging region, diet or condition	ΣPCB	↑ in colder years (↓ AO), but ↑ if warmer the previous year (↑ AO), (possibly related to ↑ transport)	Bustnes et al. 2010
Body mass loss / lipid mobilization during fasting	CB153	↑ in circulating levels in colder years and in colder region (Svalbard)	Bustnes et al. 2012
	ΣPCB_{69}		Braune et al. 2015
Diet (subarctic vs Arctic fish)	ΣPCB_{69}	Faster rate of ↓ trend	Braune et al. 2015
Prey availability or type	Total Hg	↑ in years of ↓ ice and/or ↑ AO (also possibly related to abiotic inputs)	Rigét et al. 2012
Diet (Arctic cod amount and age classes)	Total Hg	↑ in both long and short ice-free seasons	Gaden et al. 2009
None reported	Total Hg	↔	Gaden and Stern 2010
Unclear (possibly food web structure)	Total Hg	Variable, parallels PDO with 8-year time lag	Loseto et al. 2015
Foraging region or diet	Total Hg	↓ parallels $\delta^{13}C$ ↓ in females	Gaden and Stern 2010
None reported	Total Hg	↔	Gaden and Stern 2010
Diet (subarctic vs Arctic fish)	Total Hg	↓ in years of ↓ ice	Øverjordet et al. 2015
None reported	Total Hg	↔	
Diet (fish vs invertebrates)	Total Hg	Faster rate of ↑ trend	Braune et al. 2014
Diet (subarctic vs Arctic fish)	Total Hg	No trend instead of ↑ trend	Braune et al. 2014

Annex 2

Proportion of individuals, where possible grouped according to maturity (juvenile versus adult), that are at risk of PCB-mediated effects on the immune and hormonal system based on post-2000 sampling of Arctic key species and their ΣPCB loads using the intermediate conservative determined critical body residue of 10 µg/g lw PCBs.

Species	Region	Years	Age/sex	Tissue	*n*	PCB concentration, ng/g lw
						Median (range)
Toothed whales						
Killer whale	East Greenland (Tasiilaaq)	2012–2013	Adult female	Blubber	6	56.51 (24.88–120.90)
		2012–2013	Adult male	Blubber	1	140.91
		2012–2013	Sub-adult	Blubber	5	114.79 (70.84–276.86)
		2012–2013	Fetus	Blubber	3	27.97 (13.74–34.92)
		2013	Adult male	Blubber	1	65.1
		2012–2014	Adult female	Blubber	6	48.60 (19.60–65.50)*
		2012–2014	Sub-adult	Blubber	9	102.0 (9.01–356.0)*
		2012–2014	Fetus	Blubber	2	11.40 (9.47–13.4)
	Iceland	2003–2013		Blubber	16	32.75 (6.59–43.62)
	Faroe Islands	2008		Blubber	2	14.99 (1.32–16.73)
		2008	Adult and Sub-adult female	Blubber	2	5.16 (2.75–7.57)
	Shetland	2013		Blubber	4	99.40 (60.00–380.25)
	Northern Norway (Tysfjord)	2002		Blubber	8	26.94 (16.61–43.69)
Pilot whale	Faroe Islands	2001–2012	Immature	Blubber	177	21.40 (3.84–142.02)
		2001–2007	Adult male	Blubber	28	25.42 (8.99–15.45)
		2001–2011	Adult female	Blubber	66	10.54 (2.56–50.09)
Beluga	Cooks Inlet	2001–2003	Female	Blubber	4	0.72 (0.59–1.11)
		2002–2005	Male	Blubber	2	1.80 (1.52–2.08)
	Hendrickson Island	2001–2012	Male	Blubber	128	1.62 (0.45–4.49)
Bears						
Polar bear	East Greenland	2001–2013	Juvenile	Adipose	61	8.28 (2.61–58.7)
		2001–2013	Adult male	Adipose	47	13.24 (3.56–35.53)
		2001–2013	Adult female	Adipose	25	5.82 (2.85–14.77)
	Southern Hudson Bay	2008–2014	Subadult	Adipose	41	3.71 (1.14–13.27)
		2007–2014	Adult male	Adipose	58	4.69 (1.93–18.52)
		2007–2014	Adult female	Adipose	13	3.86 (1.69–10.68)
	Western Hudson Bay	2001–2013	Subadult	Adipose	26	2.60 (1.08–15.03)
		2003–2013	Adult male	Adipose	36	4.25 (1.10–9.25)
		2001–2012	Adult female	Adipose	11	4.93 (2.03–8.26)
	Svalbard	2002–2011	Adult female	Plasma	67	1.42 (0.66–5.30)
Seals						
Ringed seal	Ittoqqortoormiit	2001–2014	Juvenile	Blubber	120	0.54 (0.21–1.64)
		2001–2014	Adult male	Blubber	51	0.84 (0.44–3.99)
		2001–2014	Adult female	Blubber	33	0.72 (0.19–4.25)
	Qeqertarsuaq	2001–2014	Juvenile	Blubber	123	0.18 (0.08–0.76)
	Arviat	2003–2012	Female	Blubber	71	0.13 (0.04–0.88)
	Resolute	2004–2012	Female	Blubber	49	0.16 (0.03–1.16)
		2007–2012	Male	Blubber	14	0.17 (0.08–0.45)
	Sachs Harbour	2001–2012	Female	Blubber	53	0.24 (0.02–0.83)
Fur seal	Saint Paul Island	2006–2007	3-y old male	Blubber	20	1.16 (0.44–1.77)

PCBs, mg/g lw				RQs, %			Reference and comments
<10	10–100	100–1000	>1000	<1	1–10	10–100	
0	5	1	0	0	83	17	This study
0	1	0	0	0	100	0	This study
0	5	0	0	0	100	0	This study
0	3	0	0	0	100	0	This study
0	1	0	0	0	100	0	Pedro et al. 2017
0	6	0	0	0	100	0	Pedro et al. 2017 (* arithmetic mean)
1	5	3	0	11	56	33	Pedro et al. 2017 (* arithmetic mean)
1	1	0	0	50	50	0	Pedro et al. 2017
3	8	5	0	19	50	31	This study
0	2	0	0	0	100	0	This study
2	0	0	0	100	0	0	Pedro et al. 2017
0	2	2	0	0	50	50	This study
0	8	0	0	0	100	0	Wolkers et al. 2007
16	159	2	0	9	90	1	Dam, pers. comm.
4	23	1	0	14	82	4	Dam, pers. comm.
32	34	0	0	48	52	0	Dam, pers. comm.
4	0	0	0	100	0	0	Loseto and co-workers, pers. comm.
2	0	0	0	100	0	0	Kucklick and co-workers, pers. comm.
128	0	0	0	100	0	0	Stern and Loseto, pers. comm.
47	14	0	0	77	23	0	Dietz and co-workers, pers. comm.
14	33	0	0	30	70	0	
20	3	0	0	80	12	0	
40	1	0	0	98	2	0	Letcher and co-workers, pers. comm.
56	2	0	0	97	3	0	
10	3	0	0	77	23	0	
24	2	0	0	92	8	0	
36	0	0	0	100	0	0	
11	0	0	0	100	0	0	
67	0	0	0	100	0	0	Gabrielsen and Aars, pers. comm.
120	0	0	0	100	0	0	Rigét and co-workers, pers. comm.
51	0	0	0	100	0	0	
33	0	0	0	100	0	0	
123	0	0	0	100	0	0	
71	0	0	0	100	0	0	Muir and co-workers, pers. comm. (ten PCB congeners)
49	0	0	0	100	0	0	
14	0	0	0	100	0	0	
53	0	0	0	100	0	0	
20	0	0	0	100	0	0	Kucklick and co-workers, pers. comm.

Species	Region	Years	Age/sex	Tissue	*n*	PCB concentration, ng/g lw
						Median (range)
Birds of prey						
White-tailed eagle	Southwest Greenland	2002–2013		Liver	9	52.40 (4.49–490.33)
Gyrfalcon	West Greenland	2001–2012		Liver	13	55.10 (2.21–24228.57)
Perigrine falcon	West Greenland	2001–2012		Liver	7	101.66 (3.08–790.48)
Snowy owl	Greenland	2001–2008		Liver	6	4.67 (0.32–12.24)
Seabirds						
Thick-billed murre	Prince Leopold Is.	2003–2012		Egg	45	0.40 (0.19–0.81)
	Coats Island	2003–2011		Egg	40	0.53 (0.334–0.88)
	St. Lazaria Is.	2001–2010		Egg	26	0.70 (0.17–1.91)
	St. George Is.	2002–2010		Egg	17	0.26 (0.09–0.57)
	St. Lawrence Is.	2002–2008		Egg	11	0.38 (0.22–0.55)
	Buldir	2010		Egg	5	0.36 (0.19–0.44)
	Bjørnøya	2002–2007		Egg	19	1.29 (0.79–2.89)
Northern fulmar	Prince Leopold Island	2003–2012		Egg	45	1.35 (0.68–2.40)
Black-legged kittiwake	Prince Leopold Island	2003–2008		Egg	9	1.59 (0.65–2.70)
Black guillemot	Ittoqqortoormiit	2001–2014		Egg	89	1.74 (0.65–3.69)
Common murre	St. Lazaria I.	2001–2010		Egg	38	0.69 (0.20–2.14)
	St. George I.	2009–2010		Egg	10	0.21 (0.11–0.26)
	St. Lawrence I.	2008		Egg	5	0.30 (0.28–0.35)
	Bluff	2005–2008		Egg	10	0.33 (0.14–0.66)
Glaucous gull	Bjørnøya	2001–2006	Female	Blood	84	26.00 (6.45–237.97)
		2001–2006	Male	Blood	75	36.19 (7.53–420.49)

PCBs, mg/g lw				RQs, %			Reference and comments
<10	10–100	100–1000	>1000	<1	1–10	10–100	
2	3	4	0	22	33	44	Dietz and co-workers, pers. comm.
2	5	5	0	15	38	38	Dietz and co-workers, pers. comm.
2	1	4	0	29	14	57	Dietz and co-workers, pers. comm.
5	1	0	0	83	17	0	Dietz and co-workers, pers. comm.
45	0	0	0	100	0	0	Braune and co-workers, pers. comm. (pools of three eggs; ten congeners)
40	0	0	0	100	0	0	
26	0	0	0	100	0	0	Vander Pol and co-workers, pers. comm.
17	0	0	0	100	0	0	
11	0	0	0	100	0	0	
5	0	0	0	100	0	0	
19	0	0	0	100	0	0	Gabrielsen and co-workers, pers. comm. (six PCB congeners)
45	0	0	0	100	0	0	Braune and co-workers, pers. comm. (pools of three eggs; ten congeners)
9	0	0	0	100	0	0	Braune and co-workers, pers. comm. (pools of three eggs; ten congeners)
89	0	0	0	100	0	0	Rigét and co-workers, pers. comm.
38	0	0	0	100	0	0	Vander Pol and co-workers, pers. comm.
10	0	0	0	100	0	0	
5	0	0	0	100	0	0	
10	0	0	0	100	0	0	
4	76	4	0	5	90	5	Gabrielsen and co-workers, pers. comm. (six PCB congeners)
3	64	8	0	4	85	11	

Annex 3

Proportion of individuals, where possible grouped according to maturity (juvenile versus adult), of specific Arctic marine and terrestrial mammal and bird populations that are at risk of total Hg mediated health effects. Following 2000–2015 hepatic, egg or blood equivalent concentrations, five risk categories are reported based upon effect threshold categories observed for harp seals (Ronald et al., 1977), mink (Wobeser et al., 1976; Wren et al., 1987), and North American bird species (Ackerman et al., 2016).

Marine mammals

Species	Region	Maturity	Matrix	Years	n	Hg concentration
						Liver, µg/g
						Median (Min–Max)
Polar bear *Ursus maritimus*	Baffin Bay	Juvemile	Liver, ww	2007–2008	6	46.79 (4.55–67.7)
		Adult	Liver, ww	2007–2008	6	52.15 (34.07–106.8)
	Chukchi Sea	Juvemile	Liver, ww	2005–2007	5	6.09 (4.20–15.6)
		Adult	Liver, ww	2005–2007	7	5.10 (3.93–12.7)
	Davis Strait	All	Liver, ww	2008	6	28.85 (16.90–93.6)
	Gulf of Boothia	All	Liver, ww	2007	6	60.77 (38.16–79.1)
	Lancaster/Jones Sound	Juvenile	Liver, ww	2007–2008	5	67.93 (39.05–149.8)
		Adult	Liver, ww	2007–2008	8	66.38 (24.67–95.0)
	Northern Beaufort Sea	Juvenile	Liver, ww	2007	9	68.69 (22.08–115.1)
		Adult	Liver, ww	2007	17	122.13 (34.96–414.3)
	Southern Beaufort Sea	Adult	Liver, ww	2007	7	78.83 (63.06–322.9)
	Southern Hudson Bay	Adult	Liver, ww	2007–2008	12	12.90 (6.80–30.7)
	Western Hudson Bay	Juvenile	Liver, ww	2007	5	11.93 (6.43–30.6)
		Adult	Liver, ww	2007	6	17.70 (4.75–35.6)
	Qaanaaq	Juvenile	Liver, ww	2000–2013	6	45.34 (21.32–351.2)
	Ittoqqortoormiit	Juvenile	Liver, ww	2000–2015	62	12.50 (1.42–39.3)
		Adult	Liver, ww	2000–2015	62	18.30 (6.53–186.8)
Ringed seal *Pusa hispida*	Qeqertarsuaq	Juvenile	Liver, ww	2000–2015	160	0.93 (0.18–8.2)
	Ittoqqortoormiit	Juvenile	Liver, ww	2000–2015	104	4.69 (0.57–21.1)
		Adult	Liver, ww	2000–2015	71	9.31 (1.02–37.2)
	Qaanaaq	Juvenile	Liver, ww	2004–2014	104	2.60 (0.20–23.2)
		Adult	Liver, ww	2004–2014	12	3.46 (2.13–24.1)
	Arctic Bay	Juvenile	Liver, ww	2000–2009	18	2.03 (0.24–8.4)
		Adult	Liver, ww	2000–2009	41	7.16 (1.68–58.0)
	Arviat	Juvenile	Liver, ww	2003–2015	82	2.97 (0.066–54.1)
		Adult	Liver, ww	2003–2015	132	13.26 (0.068–254.9)
	Gjoahaven	Juvenile	Liver, ww	2004–2009	14	0.85 (0.176–25.7)
		Adult	Liver, ww	2004–2009	17	8.45 (0.201–40.0)
	Grise Fjord	Juvenile	Liver, ww	2003–2008	16	0.95 (0.351–19.8)
		Adult	Liver, ww	2000–2008	34	15.74 (0.24–87.0)
	Inukjuaq	Juvenile	Liver, ww	2002–2007	15	1.05 (0.308–4.5)
		Adult	Liver, ww	2002–2007	22	4.73 (0.931–60.0)
	Kangiqsualujjuaq	Adult	Liver, ww	2002	4	4.82 (1.516–7.8)
	Kangiqsujuaq	Juvenile	Liver, ww	2002	5	0.92 (0.367–2.0)
		Adult	Liver, ww	2002	4	1.68 (0.616–4.1)
	Nain	Juvenile	Liver, ww	2005–2015	25	1.35 (0.231–6.4)
		Adult	Liver, ww	2005–2015	31	4.66 (0.553–40.6)
	Pangniertung	Juvenile	Liver, ww	2002–2011	24	2.32 (0.156–15.8)
		Adult	Liver, ww	2002–2011	24	2.41 (0.265–31.5)
	Pond Inlet	Juvenile	Liver, ww	2000–2009	29	1.44 (0.314–69.8)
		Adult	Liver, ww	2004–2009	20	5.36 (0.434–34.3)

Risk category					Reference
<16	16–64	64–83	83–126	≥126	
No effect	Low risk	Moderate risk	High risk	Severe risk	
33	50	17	0	0	Routti et al. 2012
0	67	0	33	0	
100	0	0	0	0	
100	0	0	0	0	
0	83	0	17	0	
0	50	50	0	0	
0	40	20	20	20	
0	38	38	25	0	
0	44	33	22	0	
0	18	6	35	41	
0	14	43	14	29	
75	25	0	0	0	
60	40	0	0	0	
50	50	0	0	0	
0	66.7	0	0	33.3	Dietz and co-workers, pers. comm.
76	24	0	0	0	Dietz and co-workers, pers. comm.
40	52	3	2	3	
100	0	0	0	0	Rigét and co-workers, pers. comm.
98	2	0	0	0	
80	20	0	0	0	
99	1	0	0	0	
83	17	0	0	0	
100	0	0	0	0	Muir and Houde, pers. comm.
85	15	0	0	0	
85.4	14.6	0	0	0	
57.6	34.8	6.1	0.8	0.8	
92.9	7.1	0	0	0	
58.8	41.2	0	0	0	
93.8	6.3	0	0	0	
52.9	41.2	2.9	2.9	0	
100.0	0	0	0	0	
77.3	22.7	0	0	0	
100	0	0	0	0	
100	0	0	0	0	
100	0	0	0	0	
100	0	0	0	0	
87.1	12.9	0	0	0	
100	0	0	0	0	
95.8	4.2	0	0	0	
96.6	0	3.4	0	0	
85	15	0	0	0	

Marine mammals cont.

Species	Region	Maturity	Matrix	Years	n	Hg concentration
						Liver, µg/g
						Median (Min–Max)
	Qikiqtarjuaq	Juvenile	Liver, ww	2005	6	3.98 (3.022–5.2)
		Adult	Liver, ww	2005	14	8.89 (3.894–45.4)
	Quartaq	Adult	Liver, ww	2002	6	5.35 (3.274–26.7)
	Resolute	Juvenile	Liver, ww	2000–2015	81	3.72 (0.14–23.4)
		Adult	Liver, ww	2000–2015	124	8.47 (0.469–228.0)
	Sachs Harbor	Juvenile	Liver, ww	2001–2015	53	1.53 (0.342–117.7)
		Adult	Liver, ww	2001–2015	92	41.16 (0.446–320.3)
	Ulukhaktok	Juvenile	Liver, ww	2001–2010	4	14.17 (3.479–18.7)
		Adult	Liver, ww	2001–2010	65	24.20 (3.374–93.6)
Bearded seal *Erignathus barbatus*	Ittoqqortoormiit	Juvenile	Liver, ww	2015	7	1.82 (1.17–8.4)
Harp seal *Pagophilus groenlandicus*	Davis Strait	Juvenile	Liver, ww	2005–2006	6	1.88 (0.59–4.2)
		Adult	Liver, ww	2005–2009	14	11.10 (1.81–77.7)
	Ittoqqortoormiit	adult	Liver, ww	2015	6	0.78 (0.14–8.9)
Hooded seal *Cystophora cristata*	Davis Strait	Juvenile	Liver, ww	2002–2008	7	3.01 (0.64–16.6)
		Adult	Liver, ww	2000–2015	14	154.78 (61.49–358.2)
	Ittoqqortoormiit	All	Liver, ww	2015	5	23.06 (14.13–100.6)
Beluga *Delphinapterus leucas*	Hendrickson Island	Juvenile	Liver, ww	2002–2013	93	20.08 (0.36–111.2)
		Adult	Liver, ww	2001–2015	244	17.69 (0.30–143.7)
	Southern Hudson Bay	Juvenile	Liver, ww	2002–2015	62	7.40 (0.14–60.8)
		Adult	Liver, ww	2002–2015	71	10.14 (0.63–75.9)
Narwhal *Monodon monoceros*	Qaanaaq	Juvenile	Liver, ww	2010–2015	5	1.38 (0.65–20.7)
		Adult	Liver, ww	2010–2015	15	30.14 (0.16–132.1)
	Ittoqqortoormiit	Juvenile	Liver, ww	2015	14	3.29 (0.62–32.5)
		Adult	Liver, ww	2015	8	11.48 (0.42–18.1)
Pilot whale *Globicephala* spp.	Faroe Islands	Juvenile	Liver, ww	2001–2007	5	139.00 (11.50–351.0)
		Adult	Liver, ww	2001–2010	87	90.10 (17.50–574.0)
Killer whale *Orcinus orca*	E Greenland, Iceland and Faroe islands	Foetus	Liver, ww	2012–2013	3	0.18 (0.11–0.47)
		Juvenile	Liver, ww	1996-2013	11	2.54 (0.53–22.81)
		Adult	Liver, ww	1998-2013	6	112.05 (26.71–199.78)

Terrestrial mammals

Species	Region	Maturity	Matrix	Years	n	Hg concentration
						Liver, µg/g
						Median (Min–Max)
Arctic fox *Vulpes lagopus*	Arviat	Juvenile	Liver, ww	2001	50	0.12 (0.03–4.7)
	Iceland	Juvenile	Liver, ww	2011–2012	12	2.81 (0.44–11.4)
		Adult	Liver, ww	2011–2012	23	5.82 (0.28–46.2)
	Svalbard	Juvenile	Liver, ww	2000–2014	82	0.12 (0.01–1.1)
Reindeer/caribou *Rangifer tarandus*	Central Yukon	All	Liver, ww	2004	5	0.06 (0.05–0.2)
	NW Yukon/Alaska	Adult	Liver, ww	2006–2007	21	0.24 (0.12–0.5)
	Western Hudson Bay	Adult	Liver, ww	2007	5	0.75 (0.26–1.0)
Sheep *Ovis aries*	Faroe Islands	Adult	Liver, ww	2001–2011	13	0.01 (0.01–29.9)

Risk category					Reference
<16	16–64	64–83	83–126	≥126	
No effect	Low risk	Moderate risk	High risk	Severe risk	
100	0	0	0	0	
92.9	7.1	0	0	0	
83.3	16.7	0	0	0	
96.3	3.7	0	0	0	
75.8	21.0	0.8	0.8	1.6	
81.1	17.0	0	1.9	0	
23.9	45.7	8.7	10.9	10.9	
50	50	0	0	0	
32.3	55.4	6.2	6.2	0	
100	0	0	0	0	Dietz and co-workers, pers. comm.
100	0	0	0	0	Dietz and co-workers, pers. comm.
79	14	7	0	0	
100	0	0	0	0	
86	14	0	0	0	Dietz and co-workers, pers. comm.
0	7	7	29	57	
40	40	0	20	0	
39	51	5	5	0	Loseto et al. 2015
47	45	4	3	1	
74	26	0	0	0	
75	24	1	0	0	
80	20	0	0	0	Dietz and co-workers, pers. comm.
33	40	7	13	7	
86	14	0	0	0	
88	13	0	0	0	
20	20	0	0	60	Hoydal and Dam 2005, 2009
0	34	13	20	33	Hoydal and Dam 2005, 2009; Nielsen et al. 2014
100	0	0	0	0	Dietz and co-workers, pers. comm.
63.6	27.3	9.1	0	0	
0	33	0	33	33	

Risk category					Reference
<4.2	4.2–7.3	7.3–22.7	22.7–30.5	≥30.5	
No effect	Low risk	Moderate risk	High risk	Severe risk	
98	2	0	0	0	Hoekstra et al. 2003
67	8	25	0	0	Treu and co-workers, pers. comm.
35	22	35	0	9	
100	0	0	0	0	Hallanger and co-workers, pers. comm.
100	0	0	0	0	Gamberg and co-workers, pers. comm.
100	0	0	0	0	
100	0	0	0	0	
85	0	15	0	0	Hoydal and Dam 2005, 2009; Nielsen et al. 2014

Marine birds

Species	Region	Maturity	Matrix	Years	n	Hg concentration Egg, µg/g Liver, µg/g Blood equivalent, µg/g Median (Min–Max)
Northern fulmar *Fulmarus glacialis*	Prince Leopold Is.	Foetus	Egg, ww	2003–2015	57	0.90 (0.54–1.3)
	Western North America	Adult	Blood eq, ww	2000–2015	13	2.07 (0.79–4.2)
	Qaanaaq	All	Liver, ww	2015	10	2.61 (0.25–5.3)
	Svalbard	All	Liver, ww	2005–2006	10	2.54 (1.64–3.4)
Glaucous-winged gull *Larus glaucescens*	Bering Sea	Foetus	Egg, ww	2005–2009	11	0.12 (0.06–0.2)
	Gulf of Alaska	Foetus	Egg, ww	2005	6	0.13 (0.01–0.4)
	Western North America	Adult	Blood eq, ww	2000–2015	30	0.23 (0.14–1.0)
Glaucous gull *Larus hyperboreus*	Bering Sea	Foetus	Egg, ww	2005–2009	8	0.15 (0.11–0.2)
	Bering Strait	Foetus	Egg, ww	2005–2008	13	0.13 9 (0.01–0.4)
	Norton Sound	Foetus	Egg, ww	2005–2009	40	0.13 (0.01–0.6)
	Western North America	Adult	Blood eq, ww	2000–2015	61	0.28 (0.13–0.5)
	Qaanaaq	All	Liver, ww	2015	10	1.87 (1.06–3.0)
	Svalbard	All	Liver, ww	2005–2006	9	1.13 (0.40–2.0)
Herring gull *Larus argentatus*	Hornøya	Foetus	Egg, ww	2003	5	0.07 (0.06–0.2)
	Lofoten	Foetus	Egg, ww	2003	5	0.07 (0.04–0.2)
Ivory gull *Pagophila eburnea*	Nagurskoe	Foetus	Egg, ww	2006	6	0.24 (0.08–0.2)
	Cape Kluyv	Foetus	Egg, ww	2006	7	0.20 (0.16–0.5)
	Domashny	Foetus	Egg, ww	2006	12	0.11 (0.03–0.3)
Black-legged kittiwake *Rissa tridactyla*	Qaanaaq	All	Liver, ww	2015	10	0.35 (0.20–0.6)
	Svalbard	All	Liver, ww	2005–2006	10	0.97 (0.53–1.4)
	Svalbard	Foetus	Egg, ww	2003	5	0.14 (0.07–0.2)
	Hornøya	Foetus	Egg, ww	2003	5	0.07 (0.06–0.1)
	Lofoten	Foetus	Egg, ww	2003	5	0.09 (0.07–0.1)
Thick-billed murre *Uria lomvia*	Aleutian Islands	Foetus	Egg, ww	2000–2010	34	0.11 (0.01–0.4)
	Bering Sea	Foetus	Egg, ww	2000–2010	39	0.04 (0.01–0.2)
	Bering Strait	Foetus	Egg, ww	2002–2008	32	0.13 (0.01–0.4)
	Gulf of Alaska	Foetus	Egg, ww	2001–2010	36	0.14 (0.02–0.6)
	Prince Leopold Is.	Foetus	Egg, ww	2003–2014	55	0.85 (0.47–1.5)
	Coats Is.	Foetus	Egg, ww	2003–2015	55	0.50 (0.32–0.8)
	Western North America	Adult	Blood eq, ww	2000–2015	141	0.12 (0.02–0.6)
	Qaanaaq	All	Liver, ww	2015	10	1.06 (0.66–1.5)
	Svalbard	All	Liver, ww	2005–2006	10	0.34 (0.26–0.6)
	Svalbard	Foetus	Egg, ww	2003	5	0.03 (0.02–0.2)
Common murre *Uria aalge*	Aleutian Islands	Foetus	Egg, ww	2005	5	0.04 (0.02–0.1)
	Bering Sea	Foetus	Egg, ww	2004–2010	30	0.03 (0.01–0.2)
	Bering Strait	Foetus	Egg, ww	2002–2008	13	0.06 (0.02–0.1)
	Gulf of Alaska	Foetus	Egg, ww	2001–2010	83	0.13 (0.01–0.4)
	Norton Sound	Foetus	Egg, ww	2002–2013	35	0.13 (0.05–0.6)
	Western North America	Adult	Blood eq, ww	2000–2015	239	0.24 (0.03–0.7)
Little auk *Alle alle*	Qaanaaq	All	Liver, ww	2015	10	0.25 (0.16–0.4)
	Svalbard	All	Liver, ww	2005–2006	11	0.22 (0.02–0.5)

Risk category					Reference
<0.11	0.11–0.47	0.47–1.30	1.30–1.70	≥1.70	
<1.4	1.4–7.3	7.3–22.7	22.7–30.5	≥30.5	
<0.2	0.2–1.0	1.0–3.0	3.0–4.0	≥4.0	
No effect	Low risk	Moderate risk	High risk	Severe risk	
0	0	100	0	0	Braune 2015; Braune et al. 2016
0	23	54	8	15	Ackermann et al. 2016
20	80	0	0	0	Dietz and co-workers, pers. comm.
0	100	0	0	0	Jæger et al. 2009
45	55	0	0	0	Ackermann et al. 2016
0	100	0	0	0	
33	67	0	0	0	
0	100	0	0	0	Ackermann et al. 2016
46	54	0	0	0	
18	80	3	0	0	
21	77	2	0	0	
50	50	0	0	0	Dietz and co-workers, pers. comm.
67	33	0	0	0	Jæger et al. 2009
80	20	0	0	0	Helgason et al. 2008
60	40	0	0	0	
17	83	0	0	0	Miljeteig et al. 2009
0	71	14	14	0	
33	58	8	0	0	
100	0	0	0	0	Dietz and co-workers, pers. comm.
100	0	0	0	0	Jæger et al. 2009
20	80	0	0	0	Helgason et al. 2011
80	20	0	0	0	Helgason et al. 2008
80	20	0	0	0	
44	56	0	0	0	Ackermann et al. 2016
97	3	0	0	0	
94	6	0	0	0	
25	75	0	0	0	
0	0	96	4	0	Braune et al. 2016
0	36	64	0	0	Braune et al. 2014, 2016; Braune 2015
65	35	0	0	0	Ackermann et al. 2016
90	10	0	0	0	Dietz and co-workers, pers. comm.
100	0	0	0	0	Jæger et al. 2009
80	20	0	0	0	Helgason et al. 2011
80	20	0	0	0	Ackermann et al. 2016
97	3	0	0	0	
85	15	0	0	0	
12	88	0	0	0	
43	57	0	0	0	
40	59	1	0	0	
100	0	0	0	0	Dietz and co-workers, pers. comm.
100	0	0	0	0	Jæger et al. 2009

Marine birds cont.

Species	Region	Maturity	Matrix	Years	n	Hg concentration
						Egg, µg/g
						Liver, µg/g
						Blood equivalent, µg/g
						Median (Min–Max)
Black guillemot *Cepphus grylle*	Faroe Islands	Foetus	Egg, ww	2000–2012	142	0.43 (0.14–1.3)
		Juvenile	liver (ww)	2002–2011	65	0.90 (0.49–2.5)
	Qaanaaq	All	Liver, ww	2015	10	0.64 (0.43–1.3)
Pigeon guillemot *Cepphus columba*	Western North America	Adult	Blood eq, ww	2000–2015	27	1.99 (0.90–3.8)
Rhinoceros auklet *Cerorhinca monocerata*	Western North America	Adult	Blood eq, ww	2000–2015	21	0.94 (0.46–2.0)
Double-crested cormorant *Phalacrocorax auritus*	Western North America	Adult	Blood eq, ww	2000–2015	310	0.66 (0.12–4.0)
Ruddy turnstone *Arenaria interpres*	Western North America	Adult	Blood eq, ww	2000–2015	14	0.23 (0.15–1.9)
Atlantic puffin *Fratercula arctica*	Hornøya	Foetus	Egg, ww	2003	5	0.10 (0.09–0.1)
	Lofoten	Foetus	Egg, ww	2003	5	0.11 (0.08–0.1)
Common eider Somateria mollissima	Western North America	Adult	Blood eq, ww	2000–2015	102	0.17 (0.07–0.5)
	Svalbard	All	Liver, ww	2008–2009	40	0.49 (0.23–1.4)
King eider *Somateria spectabilis*	Western North America	Adult	Blood eq, ww	2000–2015	143	0.21 (0.08–0.5)

Terrestrial birds

Species	Region	Maturity	Matrix	Years	n	Hg concentration
						Liver, µg/g
						Blood equivalent, µg/g
						Median (Min–Max)
Bald eagle *Haliaeetus leucocephalus*	Western North America	Adult	Blood eq, ww	2000–2015	365	0.73 (0.25–2.4)
White-tailed eagle *Haliaeetus albicilla*	Nuuk	All	Liver, ww	2002–2013	9	1.98 (0.83–4.5)
	Sweden, Lapland	All	Liver, ww	2000–2004	6	1.53 (0.74–10.4)
Gyrfalcon *Falco rusticolus*	Nuuk	Juvenile	Liver, ww	2000–2012	13	1.22 (0.35–7.9)
Peregrine falcon *Falco peregrinus*	Western North America	Adult	Blood eq, ww	2000–2015	55	0.81 (0.28–3.5)
	Nuuk	All	Liver, ww	2001–2012	7	0.87 (0.11–2.8)
Snowy owl *Bubo scandiacus*	Nuuk	Juvenile	Liver, ww	2000–2008	6	0.07 (0.04–0.5)
Osprey *Pandion haliaetus*	Western North America	Adult	Blood eq, ww	2000–2015	241	0.13 (0.04–0.6)
Canada goose Branta canadensis	Western North America	Adult	Blood eq, ww	2000–2015	159	0.01 (0.00–0.1)
Greater scaup *Aythya marila*	Western North America	Adult	Blood eq, ww	2000–2015	64	0.68 (0.20–1.4)
Green-winged teal *Anas carolinensis*	Western North America	Adult	Blood eq, ww	2000–2015	96	0.22 (0.05–1.1)
Northern shoveler *Anas clypeata*	Western North America	Adult	Blood eq, ww	2000–2015	109	0.92 (0.12–5.8)
Rock ptarmigan *Lagopus muta*	Western North America	Adult	Blood eq, ww	2000–2015	12	0.03 (0.01–0.1)

Risk category					Reference
<0.11	0.11–0.47	0.47–1.30	1.30–1.70	≥1.70	
<1.4	1.4–7.3	7.3–22.7	22.7–30.5	≥30.5	
<0.2	0.2–1.0	1.0–3.0	3.0–4.0	≥4.0	
No effect	Low risk	Moderate risk	High risk	Severe risk	
0	56	43	1	0	Hoydal and Dam 2005, 2009; Nielsen et al. 2014
75	25	0	0	0	
100	0	0	0	0	Dietz and co-workers, pers. comm.
0	11	59	26	4	Ackermann et al. 2016
0	52	48	0	0	Ackermann et al. 2016
13	56	23	3	5	Ackermann et al. 2016
29	57	7	7	0	Ackermann et al. 2016
60	40	0	0	0	Helgason et al. 2008
40	60	0	0	0	
80	20	0	0	0	Ackermann et al. 2016
98	3	0	0	0	Saunes 2011
47	53	0	0	0	Ackermann et al. 2016

Risk category					Reference
<1.4	1.4–7.3	7.3–22.7	22.7–30.5	≥30.5	
< 0.2	0.2–1.0	1.0–3.0	3.0–4.0	≥4.0	
No effect	Low risk	Moderate risk	High risk	Severe risk	
3	67	26	2	2	Ackermann et al. 2016
44	56	0	0	0	Dietz et al. unpubl.
33	50	17	0	0	Helander et al. unpubl.
62	31	8	0	0	Dietz et al. unpubl.
4	58	29	5	4	Ackermann et al. 2016
71	29	0	0	0	Dietz et al. unpubl.
100	0	0	0	0	Dietz et al. unpubl.
70	28	2	0	0	Ackermann et al. 2016
100	0	0	0	0	Ackermann et al. 2016
5	72	21	2	0	Ackermann et al. 2016
45	48	5	0	2	Ackermann et al. 2016
17	35	29	5	14	Ackermann et al. 2016
100	0	0	0	0	Ackermann et al. 2016

Terrestrial birds cont.

Common loon *Gavia immer*	Western North America	Adult	Blood eq, ww	2000–2015	682	0.88 (0.25–3.9)
Pacific loon *Gavia pacifica*	Western North America	Adult	Blood eq, ww	2000–2015	68	0.42 (0.09–2.2)
Red-throated loon *Gavia stellata*	Western North America	Adult	Blood eq, ww	2000–2015	112	0.43 (0.18–1.8)
Yellow-billed loon *Gavia adamsii*	Western North America	Adult	Blood eq, ww	2000–2015	177	0.60 (0.20–2.4)
Red-necked grebe *Podiceps grisegena*	Western North America	Adult	Blood eq, ww	2000–2015	28	0.23 (0.14–1.6)

2	55	35	3	5	Ackermann et al. 2016
21	61	14	1	3	Ackermann et al. 2016
11	76	10	0	3	Ackermann et al. 2016
6	68	24	1	1	Ackermann et al. 2016
36	50	14	0	0	Ackermann et al. 2016

Acronyms and Abbreviations

ΣPCB	Sum of polychlorinated biphenyls
AMAP	Arctic Monitoring and Assessment Programme
BCCP	Blood clinical-chemical parameter
BFR	Brominated flame retardant
BMD	Bone mineral density
BMR	Basal metabolic rate
BR	Body residue
CBR	Critical body residue
CEC	Chemical of emerging concern
CHL	Chlordane
CORT	Corticosterone
dw	Dry weight
HBCD	Hexabromocyclododecane
HCB	Hexachlorobenzene
HCH	Hexachlorocyclohexane
Hg	Mercury
IgG	Immunoglobulin G
lw	Lipid weight
MeHg	Methylmercury
OCP	Organochlorine pesticide
OHC	Organohalogen compound
OH-PBDE	Hydroxylated polybrominated diphenyl ether
PBDE	Polybrominated diphenyl ether
PBPK	Physiologically-based pharmacokinetic (model)
PCB	Polychlorinated biphenyl
PFAA	Perfluoroalkyl acid
PFAS	Per- and polyfluoroalkyl substance
PFCA	Perfluoroalkyl carboxylic acid
PFOA	Perfluorooctanoic acid
PFOS	Perfluorooctane sulfonate
PFSA	Perfluoroalkyl sulfonate
POP	Persistent organic pollutant
RQ	Risk quotient
SOD	Superoxide dismutase enzyme
TEQ	Toxic equivalent
TRα	Thyroid receptor alpha
TRβ	Thyroid receptor beta
T3	Triiodothyronine
T4	Thyroxine
TT3	Total triiodothyronine
TT4	Total thyroxine
ww	Wet weight

Species names

Fish	
Arctic char	*Salvelinus alpinus*
Arctic cod	*Arctogadus glacialis*
Capelin	*Mallotus villosus*
Greenland shark	*Somniosus microcepalus*
Lake trout	*Salvelinus namaycush*
Shorthorn sculpin	*Myoxocephalus scorpius*
Birds	
American kestrel	*Falco sparverius*
Arctic skua	*Stercorarius parasiticus*
Arctic tern	*Sterna paradisaea*
Atlantic puffin	*Fratercula arctica*
Black guillemot	*Cepphus grylle*
Black-bellied plover	*Pluvialis squatarola*
Black-legged kittiwake	*Rissa tridactyla*
Canada goose	*Branta canadensis*
Common eider	*Somateria mollissima*
Double-crested cormorant	*Phalacrocorax auritus*
Glaucous gull	*Larus hyperboreus*
Golden eagle	*Aquila chrysaetos*
Great skua	*Stercorarius skua*
Gyrfalcon	*Falco rusticolus*
Ivory gull	*Pagophila eburnea*
King eider	*Somateria spectabilis*
Little auk	*Alle alle*
Northern fulmar	*Fulmarus glacialis*
Northern goshawk	*Accipiter gentilis*
Osprey	*Pandion haliaetus*
Peregrine falcon	*Falco peregrinus*
Pigeon guillemot	*Cepphus columba*
Ruddy turnstone	*Arenaria interpres*
Semipalmated plover	*Charadrius semipalmatus*
Snowy owl	*Bubo scandiacus*
Thick-billed murre	*Uria lomvia*
White-tailed eagle	*Haliaeetus albicilla*

Marine mammals	
Baikal seal	*Pusa sibirica*
Bearded seal	*Erignathus barbatus*
Beluga	*Delphinapterus leucas*
Bottlenose dolphin	*Tursiops truncatus*
Grey seal	*Halichoerus grypus*
Harbor porpoise	*Phocoena phocoena*
Harp seal	*Pagophilus groenlandicus*
Hooded seal	*Cystophora cristata*
Killer whale	*Orcinus orca*
Minke whale	*Balaenoptera acuterostrata*
Narwhal	*Monodon monoceros*
Northern fur seal	*Callorhinus ursinus*
Pacific walrus	*Odobenus rosmarus divergens*
Pilot whale	*Globicephala melas*
Polar bear	*Ursus maritimus*
Ringed seal	*Pusa hispida*
Striped dolphin	*Stenella coeruleoalba*
Terrestrial mammals	
Arctic fox	*Vulpes lagopus*
Caribou	*Rangifer tarandus*
Greenland sledge dog	*Canis familiaris*
Mink	*Mustela vison*
Moose	*Alces alces*

Arctic Monitoring and Assessment Programme

The Arctic Monitoring and Assessment Programme (AMAP) was established in June 1991 by the eight Arctic countries (Canada, Kingdom of Denmark, Finland, Iceland, Norway, Russia, Sweden and the United States) to implement parts of the Arctic Environmental Protection Strategy (AEPS). AMAP is now one of six working groups of the Arctic Council, members of which include the eight Arctic countries, the six Arctic Council Permanent Participants (indigenous peoples' organizations), together with observing countries and organizations.

AMAP's objective is to provide 'reliable and sufficient information on the status of, and threats to, the Arctic environment, and to provide scientific advice on actions to be taken in order to support Arctic governments in their efforts to take remedial and preventive actions to reduce adverse effects of contaminants and climate change'.

AMAP produces, at regular intervals, assessment reports that address a range of Arctic pollution and climate change issues, including effects on health of Arctic human populations. These are presented to Arctic Council Ministers in 'State of the Arctic Environment' reports that form a basis for necessary steps to be taken to protect the Arctic and its inhabitants.

This report has been subject to a formal and comprehensive peer review process. The results and any views expressed in this series are the responsibility of those scientists and experts engaged in the preparation of the reports.

The AMAP Secretariat is located in Tromsø, Norway. For further information regarding AMAP or ordering of reports, please contact the AMAP Secretariat (The Fram Centre, Box 6606 Langnes, N-9296 Tromsø, Norway) or visit the AMAP website at www.amap.no.

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