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Review

An assessment of the toxicological significance of anthropogenic contaminants in Canadian arctic wildlife

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Abstract

Anthropogenic contaminants have been a concern in the Canadian arctic for over 30 years due to relatively high concentrations of bioaccumulating and biomagnifying organochlorine contaminants (OCs) and toxic metals found in some arctic biota and humans. However, few studies have addressed the potential effects of these contaminants in Canadian arctic wildlife. Prior to 1997, biological effects data were minimal and insufficient at any level of biological organization. The present review summarizes recent studies on biological effects related to contaminant exposure, and compares new tissue concentration data to threshold effects levels. Weak relationships between cadmium, mercury and selenium burdens and health biomarkers in common eider ducks (*Somateria mollissima borealis*) in Nunavut were found but it was concluded that metals were not influencing the health of these birds. Black guillemots (*Cepphus grylle*) examined near PCB-contaminated Saglek Bay, Labrador, had enlarged livers, elevated EROD and liver lipid levels and reduced retinol (vitamin A) and retinyl palmitate levels, which correlated to PCB levels in the birds. Circulating levels of thyroid

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hormones in polar bears (*Ursus maritimus*) were correlated to PCB and HO-PCB plasma concentrations, but the impact at the population level is unknown. High PCB and organochlorine pesticide concentrations were found to be strongly associated with impaired humoral and cell-mediated immune responses in polar bears, implying an increased infection risk that could impact the population. In beluga whale (*Delphinapterus leucas*), cytochromes *P*450 (phase I) and conjugating (phase II) enzymes have been extensively profiled (immunochemically and catalytically) in liver, demonstrating the importance of contaminants in relation to enzyme induction, metabolism and potential contaminant bioactivation and fate. Concentrations of OCs and metals in arctic terrestrial wildlife, fish and seabirds are generally below effects thresholds, with the possible exception of PCBs in burbot (*Lota lota*) in some Yukon lakes, Greenland shark (*Somniosus microcephalus*), glaucous and great black-backed gulls (*Larus hyperboreus* and *L. marinus*), and TEQs of dioxin-like chemicals in seabird eggs. PCB and DDT concentrations in several arctic marine mammal species exceed effects thresholds, although evidence of stress in these populations is lacking. There is little evidence that contaminants are having widespread effects on the health of Canadian arctic organisms, with the possible exception of polar bears. However, further research and better understanding of organohalogen exposure in arctic biota is needed considering factors such as tissue levels that exceed effects thresholds, exposure to “new” organohalogen contaminants of concern, contaminated regions, and climate change.

Keywords: Arctic; Canadian arctic; Organochlorines; Heavy metals; Biological effects; PCB; DDT; PCDD/F; Mercury; Cadmium; Selenium; Fish; Seabirds; Mammals

Contents

1. Introduction

Anthropogenic contaminants, mainly organochlorine contaminants (OCs) and a few heavy metals such as mercury and cadmium, have been little used or released in the Arctic but are a concern for wildlife and human health because they are transported to the Arctic via long-range atmospheric transport, and to a lesser extent, via ocean currents and rivers (de Wit et al., 2004; Fisk et al., 2003a; Muir et al., 1999a; AMAP, in press). The characteristics of arctic food webs, in particular high lipid levels in biota and marine food webs with many trophic levels, result in higher than expected OC and mercury levels in upper trophic level organisms (de Wit et al., 2004; Fisk et al., 2003a; Borgå et al., 2001). The physiological status of arctic animals has an influence of OC dynamics, but also the nature, potency and complexity on effects in exposed individuals. For example, many arctic animals go through dramatic periods of fat accumulation followed by long periods of fasting, which also dramatically affects the internal dynamics of OCs.

The first Canadian Arctic Contaminants Assessment Report (CACAR) concluded, "*with the possible exception of peregrine falcons, contaminant levels or biochemical indicators of effects have not been linked to effects on* (Canadian) *Arctic animals at the individual or population levels*" (Muir et al., 1997). The conclusion was reached by assessing the limited number of biological effects studies that were available at the time (pre-1997) and to a lesser extent by comparing contaminant levels with threshold effects levels. The report further concluded that a lack of information on biological effects related to contaminant exposure in arctic organisms was a major knowledge gap. Since this report there have been very few studies examining the possible effects of contaminants on arctic biota, but a fairly substantial quantity of data on levels and trends of contaminants in arctic biota has been generated (Fisk et al., 2003a).

Examination of possible biological effects of contaminants in arctic organisms is a difficult task. There are inherent challenges in the study of wildlife in the Arctic due to the difficulty and expense of travel and fieldwork in this hostile and remote environment. Beyond this the linkage of biological effects to contaminants is difficult in any ecosystem. Biological effects can be measured at different levels of biological organization, from the molecular to the ecosystem level. Biomarkers measurable at a molecular level respond early, but are not readily interpreted ecologically, while measures with established ecological relevance, such as population declines or reduced reproductive rates, respond too late to have diagnostic or preventative value (de Wit et al., 2004). Regardless, the substrate- and concentration-dependence with respect to potency and efficacy of biomarkers at the molecular and cellular level can provide valuable information on the potential biological effects and toxicity of anthropogenic contaminants.

There are two basic approaches in assessing the possible biological effects of anthropogenic contaminants in wildlife. The first involves comparison and extrapolation. The possible effects of contaminants on arctic species are assessed by comparing levels of a contaminant of interest to levels known to be detrimental, with this knowledge coming from laboratory studies, semi-field studies or from observations on affected animals in the wild. Based on species differences alone, these types of assessments (e.g., sensitivity) are not necessarily comparable. Furthermore, studies with laboratory animals often incorporate treatments with single organic contaminants or metals, or with technical mixtures, which are not reflective of the actual exposure profile or level under realistic conditions. The treatments are generally at a few high levels of exposure, which elicit dramatic effects, but do not provide much resolution of effects that could occur under realistic exposure conditions. The majority of laboratory effects studies are also performed over short time periods relative to exposure duration in the wild. It is thus difficult to determine if the effects observed at high acute doses under the laboratory setting using model laboratory species are applicable to the possible adverse effects that may be the result of lower and chronic exposures in arctic biota. Wildlife is exposed to weathered mixtures due to the change in composition as a consequence of, e.g., abiotic degradation, metabolism and subsequent alterations via the food web. This is particularly true for arctic animals as the relative proportions of OCs are influenced by long-range transport (de Wit et al., 2004; Fisk et al., 2003a). For example, marine mammals at high trophic levels will be exposed to very different PCB compositions expressed as ΣPCB, than is seen in a PCB technical product, because some congeners are metabolized at different steps in the food web and congeners biomagnify at different rates (Fisk et al., 2001). Several arctic species have delayed implantation (e.g., seals, walrus (*Odobenus rosmarus*) and polar bears (*Ursus maritimus*)), which may make them more sensitive to the reproductive effects of contaminants than tested laboratory animals without delayed implantation (Sandell, 1990). Unfortunately, very little is known about the sensitivity of arctic species to toxic effects of OCs (de Wit et al., 2004) and metals (AMAP, in press).

The second approach studies biological effects by measuring biological responses that are related to contaminant exposure, often called biomarkers. At present, biomarkers are one of the only methods available to test the hypothesis that trace contaminants are acting biologically on the animals. Almost any measurable biological change, from molecular to ecological, can serve as a biomarker; however, the term most often refers to changes at sub-cellular levels (McCarthy and Shugart, 1990; Huggett et al., 1992; Peakall, 1992). Biomarkers typically are measures of normal processes that take on abnormal values as a result of exposure to chemicals of interest. The sensitivity of biomarkers has generally been established using laboratory animals and their applicability to arctic wildlife is little studied (de Wit et al., 2004). High trophic level species in the Arctic, predominantly marine mammals and seabirds, are most at risk from organic and metal contaminant exposure. However, it is logistically not possible to assess the effects of contaminant exposure in a controlled laboratory setting with such organisms. It is also not possible to determine causality of a biomarker assessed in a wild animal, only that a statistical association has been found between a biomarker and the contaminant in question. Most contaminants co-vary and thus it is often not possible to state equivocally that the biomarker response has been caused by a particular contaminant. There may also be other contaminants not analyzed that are just as important in influencing biomarker responses.

This present review summarizes the small number of studies that have examined possible biological effects of OCs and heavy metals on wildlife in the Canadian arctic since the first CACAR report (post-1996). It then compares threshold levels for effects established for OCs and metals that were chosen for the Arctic Monitoring and Assessment Programme (AMAP) assessments of OCs (de Wit et al., 2004) and metals (AMAP, in press) in the circumpolar Arctic, with newly generated OC and metal concentration data to assess the potential for effects of OCs and metals in Canadian arctic wildlife.

2. Studies relating biological function with metal and OC levels in Canadian arctic wildlife

Since the first CACAR report (i.e., post-1996), there have been a limited number of studies assessing the possible biological effects of OCs and metals in Canadian arctic wildlife, and for arctic wildlife in general. In all these new cases, the biological effect relationships to contaminants have been based on correlative assessment of biomarker responses and contaminant concentration in tissues, rather than causal relationships. These studies include: biomarkers that assessed general health were correlated to metal concentrations in eider ducks (*Somateria spp.*) from Southampton Island, Nunavut (Wayland et al., 2002, 2003); comparisons of blood-circulating thyroid hormone and vitamin A levels relative to OC concentrations were examined in polar bears from Resolute Bay (Norstrom, 1999a, 2000, 2001; Sandau, 2000); studies of the relationship between OCs and humoral and cellular immune responses in polar bears from Svalbard (Norway) and a Canadian population from Churchill, (Bernhoft et al., 2000; Skaare et al., 2002; Lie et al., 2004, 2005); examination of OC and PCB dynamics in western Hudson Bay bears over an annual season were related to reproductive effects (Polischuk, 1999; Norstrom, 1999b); an assessment of biochemical markers in black guillemot (*Cepphus grylle*) exposed to PCBs at a contaminated site in coastal Labrador (Kuzyk et al., 2003); and lastly, a recent immunologic and catalytic assessments of phase I cytochrome *P*450 (CYP) and phase II conjugating enzymes carried out in the liver of beluga whale (*Delphinaterus leucas*) from the Arviat region (McKinney et al., 2003, 2004).

2.1. Eider ducks

Populations of eider ducks in Alaska and the Canadian arctic have declined precipitously in the past several decades (Gratto-Trevor et al., 1998; Robertson and Gilchrist, 1998; Suydam et al., 2000), but the causes have not been identified. Contaminants may be one of several risk factors for North American sea duck populations (Canadian Wildlife Service et al., 1998). Trace elements, in particular cadmium and selenium, have been found at elevated concentrations in eider ducks in Arctic and subarctic areas (Norheim, 1987; Nielsen and Dietz, 1989; Henny et al., 1995; Dietz et al., 1996; Trust et al., 2000; Wayland et al., 2001). A paucity of useful information exists concerning the possible toxic effects of trace elements on sea ducks. Most comes from laboratory-based, captive-feeding experiments wherein surrogate species such as the mallard (*Anas platyrhynchos*) were exposed to relatively high levels of single trace elements (Di Giulio and Scanlon, 1985; Heinz et al., 1989; Bennett et al., 2000). Such studies are hard to apply to sea ducks because they may differ in their sensitivities to a contaminant. Sea ducks are also exposed simultaneously to varying levels of multiple trace elements, and natural environmental stressors, of the type that wild animals routinely encounter (e.g. adverse weather), which may act together with contaminants to produce physiological impairment (Forsyth, 2001).

To examine the potential effects of metals on the eider ducks of the Canadian arctic a number of health-related biomarker measurements were taken on common (*Somateria mollissima*) and king (*Somateria spectabilis*) eiders in 1997, 1999 and 2000 collected at the East Bay Migratory Bird Sanctuary on Southampton Island, Nunavut (64°04′ N 81°40′ W) and compared to levels of mercury, selenium and cadmium (Wayland et al., 2002, 2003). The health-related biomarkers included body condition, parasitic infestation, immune function, and stress response. In waterfowl, various measures of body condition have been linked to reproductive effort (Milne, 1976) and success (Blums et al., 1997) and to survival rates (Bergan and Smith, 1993). If exposure to elevated concentrations of certain trace elements is related to reduced body condition, as has been alluded to by Henny et al. (1991), then population dynamics could ultimately be affected. The incidence and severity of disease may also impact wild populations of sea ducks. Poor health and large die-offs of common eiders have been attributed to infestations of *Acanthocephalan* parasites (Persson et al., 1974; Hollmén et al., 1999). Thus, parasitic infestations could impact eider populations. Furthermore, it has been shown experimentally that exposure to certain trace elements can increase the severity of parasitic infestations (Borošková et al., 1995), providing a possible mechanism by which such exposure could impact eider populations. Changes in the immune system, which can affect susceptibility to disease, may provide sensitive, early warning signals of the toxic effects of contaminants, including metals (Lawrence, 1985). The response to acute stress, as evaluated by measurements of corticosteroids, is another biomarker that may be affected by some trace metals (Hontela, 1997). Corticosteroids regulate processes related to energy metabolism, salt gland function and immune function (Hontela, 1997) and therefore are important in the maintenance of homeostasis in animals.

2.1.1. Eider body condition in relation to trace elements

It is now widely accepted that body mass and fat stores are important determinants of fitness and survival in ducks (Milne, 1976; Bergan and Smith, 1993; Blums et al., 1997) and negative relationships between metal concentrations in various tissues of sea ducks and indices of their body condition have been reported (Henny et al., 1991; Ohlendorf et al., 1991; Hoffman et al., 1998; Franson et al., 2000). However, these findings have generally not been supported by experimental studies, wherein captive birds have been fed a range of concentrations of mercury or cadmium with no effect on body condition (Heinz et al., 1989; Heinz and Hoffman, 1998; White and Finley, 1978; Di Giulio and Scanlon, 1985; Bennett et al., 2000). The lack of agreement between the results of experimental feeding studies and field studies may indicate that cadmium and mercury can affect wild birds through indirect mechanisms that are not operational in captive feeding studies. A more probable explanation is that in the wild, normal, seasonal changes in body and organ mass directly affect concentrations of relatively immobile trace elements such as cadmium and mercury. For example, during periods of fasting with consequent loss of body and organ mass, cadmium and mercury may become increasingly concentrated in kidney and liver (Wayland et al., 2005–this issue). In contrast, when birds are gaining weight, as it occurs during the pre-laying period in eiders, cadmium and mercury may become increasingly diluted in the rapidly expanding tissue pool.

Relationships between body condition and metal concentrations in common eiders from the eastern Canadian arctic were reported by Wayland et al. (2002). Body weight was inversely related and abdominal fat mass (AFM) was negatively related to hepatic mercury concentrations in three and two out of four years in which this relationship was examined, respectively (Fig. 1). After adjusting for the effects of mercury, cadmium was negatively related to body weight in females sampled in 2000 and to AFM in females sampled in 1999. However, it is possible that the inverse relationships between body/organ mass and mercury/cadmium concentrations in this study simply reflect the relative degree to which individual birds are gaining or losing weight at the time they are sampled, rather than an effect of the metals on metabolic pathways affecting body and organ mass.

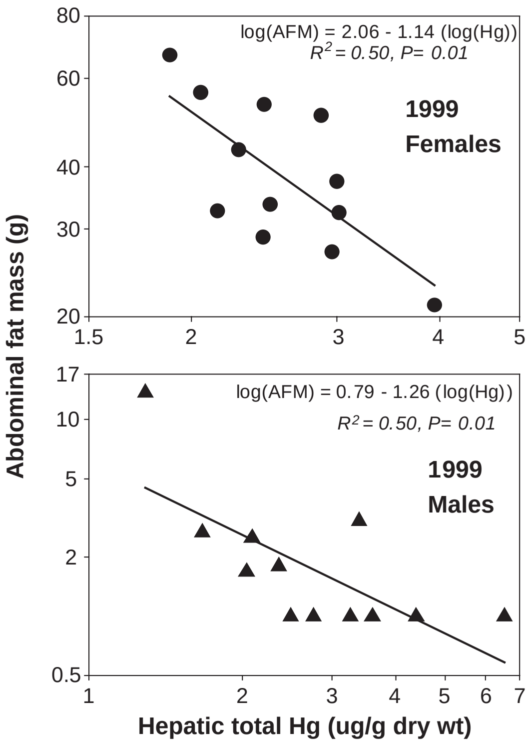

Fig. 1. Abdominal fat mass in relation to hepatic total mercury concentrations in common eiders sampled at the East Bay Migratory Bird Sanctuary, Southampton Island, 1999 (Wayland et al., 2002).

2.1.2. Parasite burdens in eiders in relation to trace elements

Relationships between parasite infestations and contaminants have been observed in wildlife. Notably, the number of gastrointestinal nematodes was positively correlated with PCB concentrations in livers of glaucous gulls (*Larus hyperboreus*) from Svalbard, Norway (Sagerup et al., 2000). Also, dead loons in poor body condition had higher concentrations of mercury and greater numbers of intestinal trematodes than those in good body condition (Daoust et al., 1998). However, Daoust et al. were unable to ascertain whether the high numbers of trematodes could be directly attributed to high mercury levels.

The relationship between counts of gastrointestinal parasites and tissue trace element concentrations was examined in eider ducks collected in 1997 (Wayland et al., 2001). Nematodes and cestodes were found in 18 and 17 of 20 king eiders, respectively, and in 30 of 33 common eiders, while acanthocephalans were found in eight king eiders and 25 common eiders. In both species, after adjusting for differences among sampling locations, residuals of cestode and *Acanthocephalan* numbers were not negatively correlated with those of trace element concentrations. *Acanthocephalan* parasites, but not nematodes, have been associated with die-offs or poor health of eiders ducks (Persson et al., 1974; Hollmén et al., 1999). In Wayland et al. (2001) acanthocephalans were not correlated with trace element concentrations and thus we cannot conclude that exposure to trace elements is likely to increase parasite-induced mortality in either species of eider. However, in common eiders, but not in king eiders, residual total and organic mercury concentrations were positively correlated with residual nematode numbers (Fig. 2). The findings of Wayland et al. (2001) suggest that parasite infections in free-living birds may be affected by contaminant exposure.

2.1.3. Other biomarkers in eiders

In 1999 and 2000, the relationship between trace element exposure and certain aspects of the immune response was examined in common eiders at East Bay on Southampton Island (Wayland et al., 2002). Two immune function assays were conducted: (1) the skin-swelling response to an intradermal injection of phytohemagglutinin-P (PHA-P) and (2) the antibody titer in response to an intraperitoneal injection of sheep red

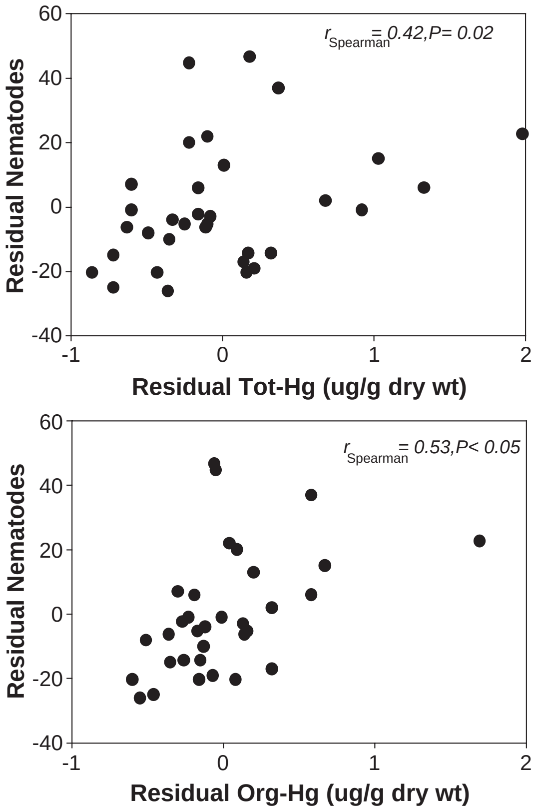

Fig. 2. Gastrointestinal nematode numbers in relation to hepatic total and organic mercury concentrations in common eiders sampled at three arctic locations (Holman on Victoria Island, East Bay Migratory Bird Sanctuary on Southhampton Island and the Belcher Islands) in 1997. To account for differences among sampling locations, residuals of parasite counts and mercury concentrations were determined by subtracting the mean value from individual values at each location (Wayland et al., 2001).

blood cells (SRBC). The former assay elicits a simple and rapid cell-mediated immune response while the latter elicits a humoral immune response. In 1999, selenium concentrations in common eiders were positively related to the PHA-P response but were not significantly related to the antibody titer (Wayland et al., 2002). Beyond this effect, tissue trace element concentrations did not significantly dampen the immune responses of eider ducks in this study based on the tests used. Yet these trace metals are known to be immunotoxic in experimental animals (Descotes, 1992; Pollard and Hultman, 1997). Either the immune function assays that were used were ineffective in demonstrating immunomodulation by mercury and cadmium exposure in these eiders or the concentrations of these trace metals to which the eiders were exposed were below threshold effect levels.

2.2. Thyroid hormones and vitamin levels in polar bears

Thyroid hormones and vitamin A play important roles in the growth and development of organisms (Morriss-Kay and Ward, 1999; Morse et al., 1993; Morse, 1995). In fish, mammals and birds, the thyroid gland produces predominantly T4 (thyroxine) (Kühn, 1990; Sefkow et al., 1996), which is transported in plasma to target tissues by the transport proteins such as transthyretin (TTR), but also thyroxine binding globulin (TBG) and plasma albumins. The relative importance of these thyroid hormone transport proteins is dependent on species and taxonomy (Bentley, 1998). TTR is important in mammals, whereas TBG is absent in all non-mammals. Once delivered, T4 is deiodinated by T4-monodeiodinase to triiodothyronine (T3), which is the active hormone.

Potential mechanisms of thyroid and retinol disruption by endocrine-disrupting organic contaminants include induction of thyroid hormone conjugating and metabolizing enzymes and enzymes that metabolize body stores of vitamin A. The modulation of these enzymes, especially in the liver, can affect circulating levels of thyroid hormone and retinol and their homeostasis at target tissues. One of the more sensitive physiological functions affected by PCB exposure in mammals is retinol (vitamin A) and thyroid hormone homeostasis (Brouwer et al., 1989). In mammals, some OC effects on thyroid hormone status may also be a function of phenolic metabolites, such as hydroxylated (HO) PCBs (Maervoet et al., 2004) and halogenated phenolic compounds (HPCs), which can competitively bind to TTR-retinol-binding protein complex (TTR-RBP) in plasma (Rolland, 2000; Simms and Ross, 2000).

Thyroxine-like HO-PCBs have been identified in the blood of mammals, birds, humans and even fish, and are often among the most significant classes of circulating organohalogen contaminants in blood (Campbell et al., 2003; Letcher et al., 2000; Li et al., 2003a). The polar bear has the highest concentrations of PCBs of any arctic animal examined to date (de Wit et al., 2004) and HO-PCBs have been identified in polar bears from Svalbard, Greenland and Canadian arctic populations (Letcher et al., 2005; Sandala et al., 2004; Sandau, 2000; Sandau et al., 2000). It is therefore important that thyroid hormone and retinol toxicology be studied in this species, especially since cub survival appears to be lower in areas with high PCB contamination (Derocher et al., 2003).

To assess the potential impact of PCBs and HO-PCB metabolites on polar bear thyroid and vitamin homeostasis, the correlative relationships between PCB and HO-PCB concentrations and that of thyroid hormones (T3 and T4) and retinol were determined in plasma of polar bears from Resolute Bay, Canada and Svalbard, Norway (Sandau, 2000). These populations have among the lowest (Resolute Bay) and highest (Svalbard) OC concentrations among polar bears (Norstrom et al., 1998).

In the study by Sandau et al. (2000), biological measures in bears from Resolute Bay were not significantly related to age, which contrasts with a previous study on Svalbard bears, which found relationships with age in males, but not for females (Skaare et al., 2001a). However, sex differences in retinol and thyroid hormone parameters were found bears from these populations (Table 1). Resolute bears had significantly higher total T4, and free T4 (FT4 index), and lower total T3 and free T3 (FT3) index in plasma than Svalbard bears (Sandau, 2000). The free thyroid indices (FT3 and FT4) are determined analytically and provide an estimate of free thyroid concentrations (Sandau, 2000). Changes in free thyroid concentrations may indicate distribution of thyroid-protein binding in plasma.

Differences in thyroid homeostasis as function of PCB and HO-PCB concentrations in plasma was suggested between Resolute and Svalbard polar bears ($N=60$) (Table 1 and Fig. 3), whereas retinol concentrations were not significantly different between bears among regions, but were related to PCB and HO-PCB concentrations (Sandau, 2000). Based on the PCA, retinol concentrations were negatively correlated with persistent PCB congener concentrations and positively correlated with HO-PCB concentration (Fig. 3). If only Resolute bears ($N=25$) were included in the analysis, retinol was more highly correlated with persistent PCB and HO-PCB concentrations. It was suggested that the plasma retinol concentrations are more likely to be affected by the influence of persistent PCBs on retinol metabolism and storage in liver rather than by interference of HO-PCBs with the transport of retinol via retinol binding protein:TTR dimer formation. Based on PCA, total T4 concentrations in plasma were negatively associated with concentrations of a range of persistent PCB congeners and ΣPCBs, but not with any other contaminant group including HO-PCBs. This correlation suggests a common mechanism of action of all PCB congeners in reduction of

Table 1
Thyroid hormone concentrations, free hormone indices and retinol concentrations in polar bear plasma from Resolute Bay in the Canadian Arctic collected in 1997 and in Svalbard collected in 1998 (Sandau, 2000)

	Resolute cubs ($n=3$)	Resolute juveniles ($n=5$)	Resolute male adults ($n=12$)	Resolute female adults ($n=13$)
Age (years)	0–2	3–4	10 ± 4	14 ± 7
T3 (nmol/L)	0.21 ± 0.21	0.08	0.19 ± 0.19	0.15 ± 0.08
FT3 index	4.57 ± 1.74	2.91 ± 0.95	5.19 ± 1.34	2.81 ± 1.59
T4 (nmol/L)	8.00 ± 2.75	6.25 ± 0.41	4.93 ± 4.05	6.62 ± 2.06
FT4 index	1.08 ± 0.09	1.17 ± 0.03	1.13 ± 0.04	1.15 ± 0.04
Retinol (μmol/L)	1.16 ± 0.33	1.17 ± 0.50	0.70 ± 0.20	1.08 ± 0.46
	Svalbard cubs ($n=3$)	Svalbard juveniles ($n=5$)	Svalbard male adults ($n=12$)	Svalbard female adults ($n=13$)
Age (years)	0–2	3–4	14 ± 6	14 ± 4
T3 (nmol/L)	0.23 ± 0.13	0.69 ± 0.67	0.34 ± 0.30	0.59 ± 0.36
FT3 index	6.18 ± 0.15	6.24 ± 0.05	6.31 ± 0.08	6.25 ± 0.09
T4 (nmol/L)	1.15 ± 1.70	0.39 ± 0.25	2.33 ± 1.87	3.88 ± 4.82
FT4 index	0.79 ± 0.11	0.75 ± 0.14	0.93 ± 0.18	0.84 ± 0.18
Retinol (ìmol/L)	1.22 ± 0.26	0.88 ± 0.43	0.79 ± 0.34	0.96 ± 0.29

All values are mean ± SD.

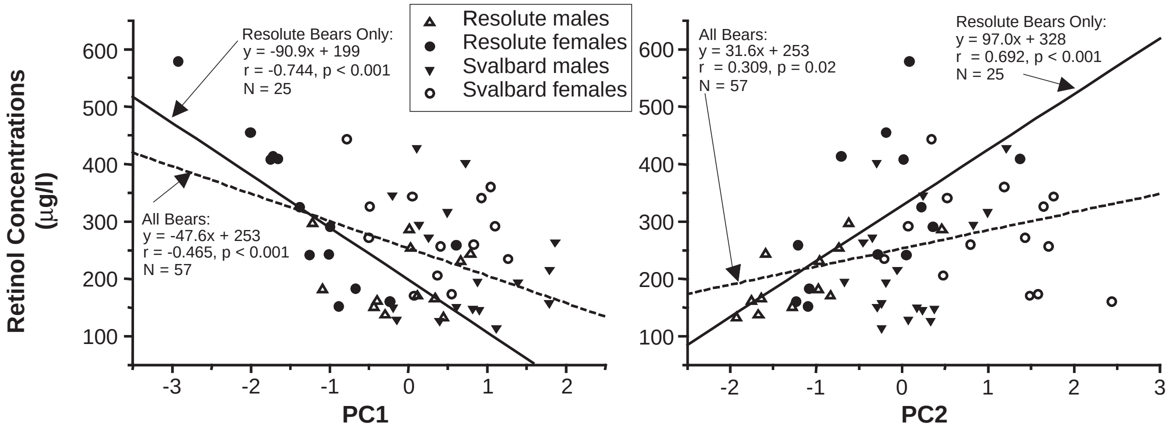

Fig. 3. Correlation between retinol concentrations (μg/l) and the first two principal components from analysis of OC concentrations in polar bear plasma from Resolute Bay in the Canadian arctic in April–May 1997, and from Svalbard in April–May 1998. PC1 represents persistent PCBs and PC2 represents HO-PCBs. From Sandau (2000).

plasma T4 concentrations. PCBs have been shown to reduce total plasma T4 concentrations in rats (Goldey et al., 1995), probably due to increased peripheral T4 metabolism (Morse et al., 1993). The FT4 index was negatively correlated to ΣHO-PCB concentrations, but was likely an artifact of poor sample preservation (Sandau, 2000). Another class of PCB metabolites, the persistent and bioaccumulative methyl sulfone ($MeSO_2$) PCBs are being found as tissue residues in an increasing number of arctic wildlife species, including those from the Canadian arctic (Letcher et al., 2000). Braune et al. (2005) recently summarized the increasing detection of $MeSO_2$-PCB and -*p*,*p*′-DDE metabolite contaminants in the tissues of Canadian arctic biota, almost exclusively in mammals. $MeSO_2$-PCBs have demonstrated endocrine-related activity including effects on estrogen, thyroid and glucocorticoid hormone-dependent processes in laboratory mammals and in in vitro cell-based assays (Letcher et al., 2000, 2002).

The potential hormone-related effects of contaminant exposure in Canadian polar bear are indicated from recent reports on the relationships between endocrine parameter and OC concentrations in Svalbard bears (Skaare et al., 2001b, 2002; Oskam et al., 2003; Haave et al., 2003) Although OC concentrations in Canadian polar bears are lower than those observed in Svalbard they fall within the continuum of the relationships observed between PCB and HO-PCB concentrations and thyroid and retinol parameters, and thus concerns exist for the health of polar bears from circumpolar populations. However, OC exposure and thyroid and retinol hormones relationships are inconsistent in the literature. This is likely due in large part to the physiological complexity of thyroid hormone and vitamin homeostasis, e.g. production of thyroid-stimulating-hormone (TSH), activity (induction and inhibition) of hormone producing and metabolizing enzymes (e.g., iodinases, deiodinases, uridine diphosphoglucuronosyl transferases (UDPGTs), sulfotransferases). This physiological complexity is in turn influenced by ecological factors such as seasonal variation, etc. Considerably more research is warranted and required to study the effects and impacts of contaminant exposure on endocrine function.

2.3. Reproductive effects in polar bears

The ecological significance of effects of contaminants on reproduction is well established. Whereas changes at the molecular, biochemical or physiological level are used as an early warning system of responses to contaminant exposure, it is generally accepted that effects observed at the individual or population level are less reversible, more detrimental, and of greater relevance to ecological health (de Wit et al., 2004). Reproductive and developmental toxicity of PCBs and OC pesticides is well-documented in controlled laboratory studies (see de Wit et al., 2004 for examples relevant to arctic animals) and has also been implicated in declines in wild populations (e.g., Colborn, 1991).

Recent studies have investigated the effect of fasting, gestation and lactation on toxicokinetics of OCs in adipose tissue, plasma and milk samples from seven female polar bears and their cubs near Cape Churchill, Hudson Bay, between 1992 and 1996 (Polischuk, 1999; Norstrom, 1999b). Pregnant females were captured from August 7 to October 7, and the same females with cubs were captured from March 2 to March 17 of the following year before they had moved onto the ice to begin hunting seals. All females had therefore been fasting 5 to 7 months by the time of their second capture. Body composition of females was determined from ^{2}H dilution in blood and body weight. The total body mass of females declined by 43 ± 5%, and total fat mass declined by 42 ± 3%. The proportion of mass lost as fat, ranged between 55 to 66%. Between their first and second capture, mother bears lost on average 24 to 29% of their body burdens of the ΣCHL and ΣPCB, and especially the more slowly rapidly metabolized congeners. The mean period between first and second capture (mostly in their dens) was 188 ± 22 days, and the mean number of days of lactation prior to sampling was approximately 79 ± 4 days. The toxicokinetics of OCs during the first 100 days were likely similar to that during the subsequent summer fast. The polar bear cub weighs only about 0.7 kg at birth (I. Stirling, personal communication, Canadian Wildlife Service, Environment Canada, Edmonton AB) and two newborn cubs represent about 0.4% of the mother's weight. Therefore, OC transfer to the fetus is unlikely to significantly reduce the mother's body burden. Thus, most of the 24 to 29% loss of ΣCHL and ΣPCB from the mother during the 188-day fast must have been transferred to the cub in milk during the 79 day lactation period prior to capture, at which time the cubs weighed 12.7 ± 0.9 kg. However, on average, only about 25 to 50% of this body burden loss could be accounted for by ΣCHL and ΣPCB burdens in the cubs. A crude estimate of lactational transfer based on cub growth rates could only account for half of the body burden loss from the females. These discrepancies are difficult to reconcile and require further study.

In the Polischuk (1999) studies, polar bear mothers that were recaptured in the fall without cubs had higher OC concentrations in their milk when emerging from their dens in spring (Fig. 4). By comparison,

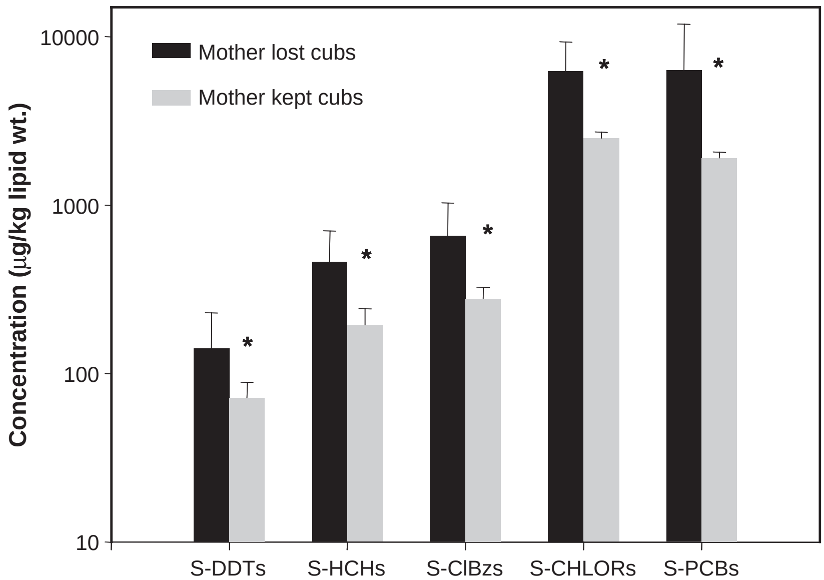

Fig. 4. Concentrations (ng/g lw, mean ± SD) of major OCs in polar bear milk of females with cubs after emerging in March from dens in the Cape Churchill area, Hudson Bay (1992–1996) (Polischuk, 1999; Norstrom, 2000). The data are grouped according to whether the female still had her cubs the following fall, or had lost them. The asterisk (*) indicates that the difference was statistically significant.

mothers recaptured in the fall and still accompanied by cubs had lower OC concentrations in their milk the previous spring. The differences in concentrations were significant for all residue classes. For example, PCBs were approximately 3 times higher (5780 ng/g lw) in females that lost their cubs than in females that kept their cubs (1830 ng/g lw). It is not known how much significance can be attached to this finding in terms of reproductive performance, but it is suggestive, at least, that cub survival may be dependent on the degree of exposure to OCs in milk.

2.4. Immune effects in polar bears

Many OCs disrupt both humoral and cell-mediated immune responses of the specific (acquired) arm of the immune system, as well as causing effects on the non-specific (innate) arm. As a result, the resistance to infectious agents may be reduced. Humoral-mediated immunity involves the body's ability to recognize foreign substances (via helper T-cells) and mount a response by stimulating the production of antibodies (B-cells). Cell-mediated immunity is involved in delayed hypersensitivity reactions (e.g. skin reactions to allergens) and the production of cytotoxic T-cells against tumors and viruses. Natural killer cells are involved in the non-specific (i.e. absence of memory) immune response and provide a first line of defence against virus-infected cells and tumors.

Immunosuppressive effects may be one of the most sensitive and environmentally relevant effects of OCs (Vos and Luster, 1989). For example, immunosuppression has been measured in harbor seals (*Phoca vitulina*) fed Baltic fish in semi-field experiments and was found to correlate with levels of polychlorinated dibenzo-p-dioxins (PCDD), polychlorinated dibenzofurans (PCDFs) and planar PCBs expressed as toxic equivalents (TEQs) (De Swart et al., 1995; Ross et al., 1995, 1996). Two previous studies on polar bears at Svalbard have demonstrated negative correlations between PCBs and antibody-mediated immunity (Bernhoft et al., 2000; Lie et al., 2002).

Based on these findings, a joint Norwegian–Canadian field experiment was undertaken to assess if high levels of OCs are associated with decreased ability to produce antibodies and with changes in lymphocyte proliferation and part of the cell-mediated immunity in free-ranging polar bears from Churchill, Canada and from Svalbard (Lie et al., 2004, 2005). Previous studies have shown that polar bears from Svalbard had higher PCB exposure than polar bears from Canada (Norstrom et al., 1998).

In 1998 and 1999, 26 polar bears from Svalbard (13 males, 13 females) and 30 polar bears from Churchill (all males) were recaptured after having been immunized with inactivated influenza-, reo- and herpes viruses and inactivated tetanus toxoid (*Mannheimia haemolytica*, previous *Pasteurella haemolytica)* to stimulate antibody production (Lie et al., 2004). They were also immunized with hemocyanin from keyhole limpets (KLH), which together with tetanus toxoid, would sensitize lymphocytes that could be tested in in vitro lymphocyte proliferation tests (Lie et al., 2005). Blood was sampled at immunization and at recapture 32–40 days following immunization and analyzed for plasma levels of PCBs (12 congeners), organochlorine pesticides (HCB, oxychlordane, *trans*-nonachlor, HCHs, p,p'-DDE), IgG concentrations and specific antibodies against influenza-, reo- and herpes viruses, tetanus toxoid and *Mannheimia* sp. Blood samples from recapture were also used to study lymphocyte proliferation after in vitro stimulation with specific mitogens: phytohemagglutinin (PHA), poke weed mitogen (PWM), concanavalin A (Con A), lipopolysaccharide (LPS), purified protein derivative of *Myobacterium avium* subsp. *paratuberculosis* (PPD), and with the antigens tetanus toxoid and KLH.

Two approaches were used to study the immunological variables: (1) Study of biological variation, location and sex differences in the immunological variables using the data from Svalbard ($n=26$) and Canada ($n=30$) separately, and (2) study the effects of OCs on the immunological variables using all the bears ($n=56$). Results were analyzed statistically, including using multiple linear and non-linear regression models (Lie et al., 2004, 2005).

Mean ΣPCB concentrations were not significantly different between males and females from Svalbard, or between bears from Canada and Svalbard. Organochlorine pesticide (ΣOCP) levels were significantly higher in female bears from Svalbard compared to Svalbard males, and in the male Canadian bears compared to male Svalbard bears. The finding of no significant difference in ΣPCB levels between locations was unexpected. Even if the sum PCB levels did not differ significantly between locations, concentrations of PCBs 118, 156, 157, 170, 180 and 194 were significantly higher at Svalbard than in Canada. Thus, the proportion of higher chlorinated PCBs was higher at Svalbard compared to Canada.

The change in mean serum IgG levels in polar bears from Canada and Svalbard between the time of immunization and recapture did not differ significantly by location (Fig. 5). The mean serum IgG concentration was significantly higher both at immunization and at recapture in the Canadian polar bears compared with the corresponding level in the Svalbard bears. Lower IgG levels and antibody titers produced following immunization of the Svalbard bears may be a result of impaired immunity produced by various factors in the environment, including environmental contaminants. However, when corrected for body mass and sex, the difference between locations was only significant at immunization. There was no significant effect of age on the serum IgG concentration in male polar bears at recapture.

Immunization led to increased antibody production in all bears. Canadian polar bears had significantly higher antibody titers to influenza-, reo- and herpes viruses after recapture than Svalbard bears (Fig. 5). In the Canadian bears, antibody titers to reo virus at immunization demonstrated that the Canadian population had been exposed to environmental reo-viruses. The antibody titers to reo-virus in the Svalbard bears were low. Thus, at recapture, the titers measured in the Canadian bears were generally secondary responses whereas the titers in the Svalbard bears were generally primary responses to reo-virus. Following immunization with herpes virus the bears both in Canada and at Svalbard responded with very low mean titers of neutralizing antibodies. A high mean titer of antibodies against tetanus toxoid after immunization was found in both populations.

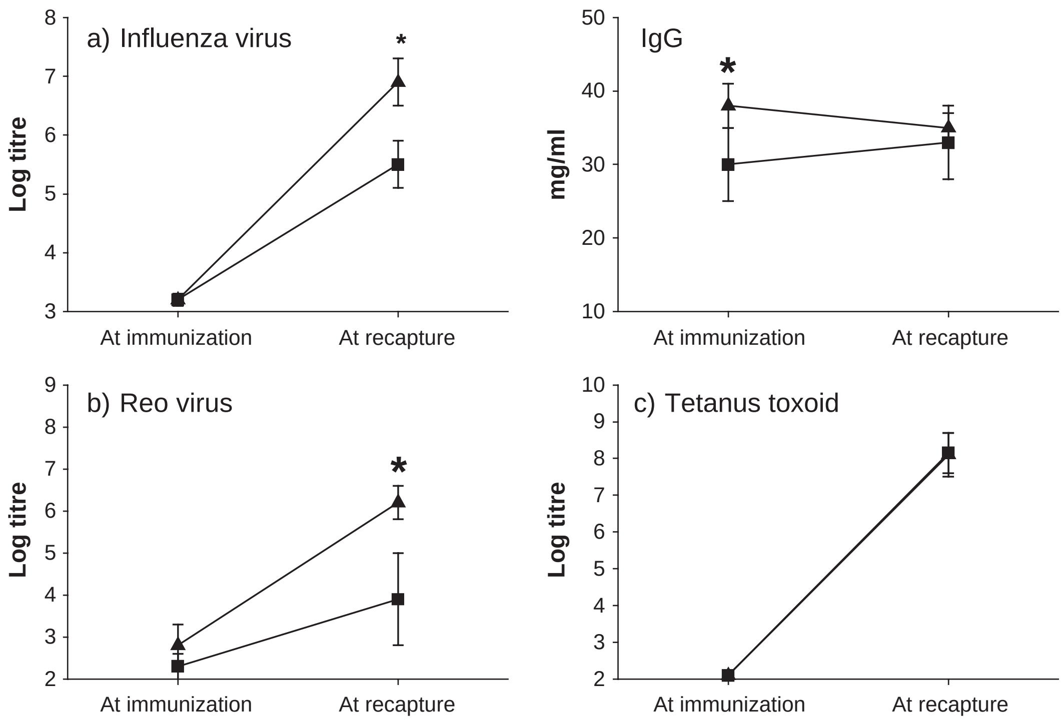

Fig. 5. Serum IgG levels at immunization and recapture in male polar bears (top right panel mean IgG, mg/ml and 95% confidence interval) and antibody responses against a) influenza virus, b) reo virus and c) tetanus toxoid at immunization and recapture in male polar bears (mean $\log_2$ antibody titer and 95 % confidence interval). Data are for male polar bears at Svalbard ($n = 13$) (■) and in Canada ($n = 30$) (▲). An asterisk indicates that there were significant differences between locations ($p \leq 0.05$). Figures from Lie et al. (2004), by permission of J. Toxicol. Environ. Health.

There was no significant difference for tetanus toxoid antibody titers by location. *M. haemolytica* antibody titers were high in both populations both at immunization and at recapture. This indicates that *Mannheimia* spp. bacteria are common pathogens in the environment of the polar bears. However, at recapture the antibody titer to *M. haemolytica* was significantly higher in the Svalbard population compared to Canada. This could reflect increased exposure of the immune system to *Mannheimia* spp. pathogens present in the respiratory system in the Svalbard population leading to stronger stimulation of specific immunity that results in an increased antibody titer.

The results of the statistical analyses show that OCs in combination with specific biological factors (sex, age, condition, body mass) significantly influence serum IgG levels and antibody production after immunization with influenza and reo-viruses and tetanus toxoid (Lie et al., 2004). These combinations contributed between 40% and 60% of the variation in the immunological parameters and the linear combination of ΣPCBs and ΣOCPs contributed up to 7% to the variation, a significant effect of OC exposure. In addition, the results demonstrate in particular, a negative association between ΣPCBs and the serum IgG levels or production of antibodies to influenza and reo-viruses. This study thus demonstrates that high levels of ΣPCBs influence parts of humoral immunity in polar bears, as the ability to produce antibodies following immunization was impaired.

For mitogen-induced lymphocyte proliferation, the mean lymphocyte response to PWM was higher in male polar bears from Canada than in males from Svalbard (Fig. 6a). Mean lymphocyte responses to

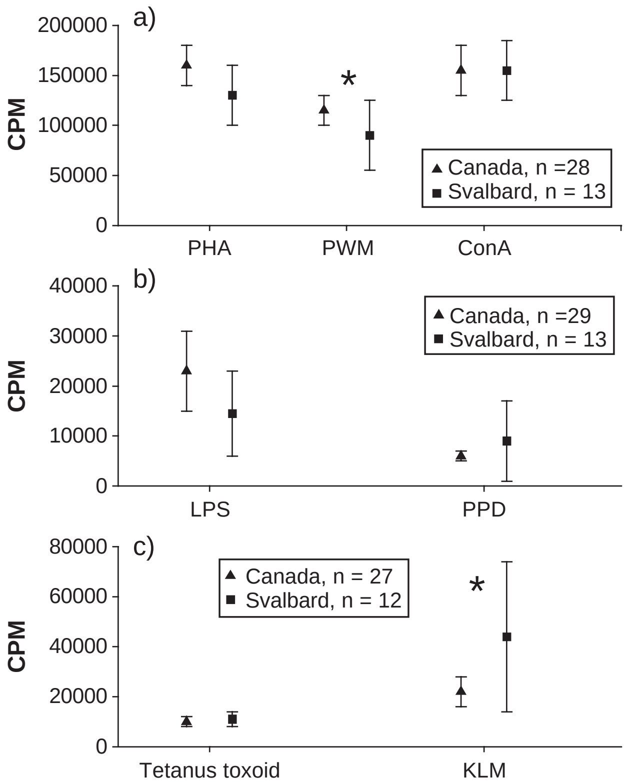

Fig. 6. The lymphocyte proliferation response after stimulation with a) phytohemagglutinin (PHA), poke weed mitogen (PWM), concanavalin A (Con A), b) lipopolysaccharid (LPS) and purified protein derivate of *Mycobacterium avium subsp. Paratuberculosis* (PPD), c) tetanus toxoid and keyhole limpets (KLH) in heparinized blood from male polar bears at Svalbard (■) and in Canada (▲). The lymphocyte response was expressed as the difference of the mean counts per minute (CPM) of stimulated cultures and control cultures. The results are expressed as mean responses and 95% confidence interval (C.I.). An asterisk indicates that there were significant differences ($p<0.05$). Figures from Lie et al. (2005), by permission of J. Toxicol. Environ. Health.

PHA, Con A, LPS and PPD were not significantly different in the males from the two locations (Fig. 6a and b). At Svalbard, there were no differences in any of these lymphocyte responses between males and females. In the males from Canada, lymphocyte response to Con A was positively correlated and PPD negatively correlated to body mass. No effect of age or condition was found on PHA, Con A, LPS or PPD-stimulated lymphocyte proliferation in any of the bears.

For antigen-induced lymphocyte proliferation, the mean lymphocyte response to tetanus toxoid was not significantly different in male bears from Canada and Svalbard, whereas the mean response to KLH was significantly higher in the Svalbard males compared to the Canadian males (Fig. 6c). The mean response to tetanus toxoid was significantly lower in females than males from Svalbard, but no difference was found for KLH response.

For lymphocyte proliferation, the results of the statistical analyses show that OCs in combination with specific biological factors significantly influenced specific lymphocyte proliferation responses (Lie et al., 2005). These combinations contributed 45%–70% of the variation in the immune parameters and the combination of ΣPCBs and ΣOCPs contributed up to 15% of the variations in the lymphocyte responses. Thus, high levels of ΣPCBs, ΣOCPs or the interaction of ΣPCBs and ΣOCPs were found to be associated with decreased ability of different lymphocyte populations to proliferate after stimulation with mitogens and antigens in vitro. The product of ΣPCBs and ΣOCPs was the dominant factor contributing negatively to the variation in responses to KLH, PWM and LPS. ΣPCBs were the dominant contributor to the variation in response to PHA, Con A and PPD. LPS usually stimulates B-cells and PWM stimulates T- and B-cells. PHA and Con A usually stimulate T-cells. Thus, the effects seen indicate impaired B- and T-cell activity associated with high OC exposure. The impairment of lymphocyte proliferation responses and part of the cell-mediated immunity demonstrated in this study, taken together with the finding of impaired antibody production following immunization (Lie et al., 2004), demonstrate the immunotoxic effect of OC exposure in polar bears. These results are very significant and suggest that contaminant exposure may have increased the polar bear's susceptibility to infections.

2.5. Biochemical markers in black guillemot nestlings exposed to PCBs at Saglek, Labrador

Saglek Bay, on Canada's Labrador coast, has been the site of a military radar station since the late 1950s. Prior to remediation in 1997–1999, PCB-contaminated soil at the site was a source of contamination to the surrounding land as well as the inshore marine environment. PCB concentrations are elevated in sediments as well as in the local benthic-based marine food web (Kuzyk et al., 2005–this issue). The Saglek situation is unusual since only PCB concentrations are elevated, while the levels of other OCs and metals are comparatively low. The sediment PCB contamination decreases in concentration by several orders of magnitude with distance from the radar station. PCB concentrations in the benthic-based marine food web, including invertebrates, fish and diving seabirds, are highly correlated and relate directly to the concentrations in sediments (Kuzyk et al., 2005–this issue).

In 1999, an ecological risk assessment was initiated to provide a scientific basis from which to evaluate the need for sediment remediation (Kuzyk et al., 2003). A diving seabird, the black guillemot, was chosen as one focus of the assessment. A suite of liver biomarkers used in Great Lakes' monitoring was examined, including ethoxyresorufin-*O*-deethylase (EROD) activity and concentrations of retinoids (vitamin A), malic enzyme and porphyrins. Liver biomarkers and ΣPCB concentrations were measured in 31 nestlings from three PCB-exposure groups: reference group (range: 15–46 ng/g liver, wet wt.), moderately exposed Islands group (24–150 ng/g), and highly exposed Beach group (170–6200 ng/g) (Fig. 7). Livers of female Beach nestlings were enlarged 36% relative to livers of reference females. In both sexes of Beach nestlings, liver EROD activities were elevated by 79% and liver retinol concentrations were reduced by 47%, compared to the reference group. Retinyl palmitate concentrations were also reduced 50% but only among female Beach nestlings. Island nestlings also exhibited EROD induction (57%) and reductions in retinol and retinyl palmitate concentrations (28% and 58%, respectively). Liver lipid content increased with

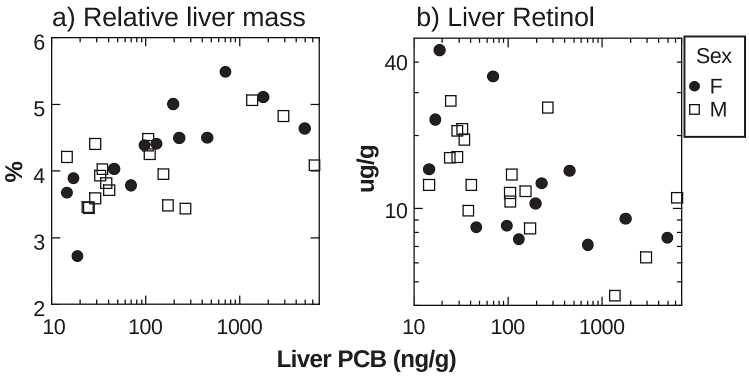

Fig. 7. a. Relationship between nestling liver mass, expressed as percentage of body mass, and liver PCB concentration (wet weight) among male and female guillemot nestlings at Saglek, Labrador (r^2=0.38, df=29, p<0.0001, liver mass=0.487 * PCB concentration+3.135) (Kuzyk et al., 2003). b. Relationship between liver retinol concentration and liver PCB concentration (wet weight) by sex among nestlings at Saglek, Labrador (r^2=0.35, df=29, p<0.0001, retinol concentration=−0.177 * PC concentration+1.483) (Kuzyk et al., 2003).

ΣPCBs in both sexes, and correlated with liver mass in males. Malic enzyme activity and porphyrin concentrations showed minimal associations with ΣPCBs. Although similar associations between liver biomarkers and OC exposure in fish-eating birds are well documented (see e.g., Champoux et al., 2002), exposures elsewhere involve multiple contaminants and there is uncertainty about specific PCB effects.

The findings of this assessment indicate that liver biomarkers respond to relatively low PCB exposures (~73 ng/g ww in liver) in guillemots. These results are very significant and show that local contamination in the Canadian arctic can have an important impact on the physiology of local wildlife. Additional research is warranted at Saglek and other locations where PCBs, or other OCs, may be at levels greater than what is normally observed across the Canadian arctic.

2.6. Biomarkers and OCs in beluga

Biomarkers in the form of induced enzyme measurement, catalytic or immunological, can be indicative of the metabolic capacity of exposed organisms towards xenobiotic compounds and contaminants. The cytochrome *P*450 monooxygenases (CYPs) play a central role in the oxidative biotransformation (phase I) of a wide range of xenobiotic and endogenous compounds (Goksøyr, 1995; Lewis et al., 1998). Products from phase I metabolism are conjugated to larger endogenous molecules via catalytic mediation by phase II enzymes such as glutathione-*S*-tranferases (GSTs) and UDPGTs (Wolkers et al., 1998a). To date there has been limited documentation of the enzyme capacity as indicators of metabolic potential in arctic biota, with the lone exceptions of Canadian polar bears (Letcher et al., 1996; Bandiera et al., 1995) and beluga whales (White et al., 1994, 2000; McKinney et al., 2004), Svalbard ringed seals (*Phoca hispida*) (Wolkers et al., 1998a,b) and Barents Sea harp seals (Wolkers et al., 1999).

Homologues of CYP1A, CYP2B and CYP2E have been determined in the liver of MacKenzie River beluga (White et al., 1994, 2000). Relative to, e.g., the St. Lawrence River Estuary, Canadian arctic beluga whales are generally exposed to lower OC contaminant concentrations. McKinney et al. (2003, 2004) recently reported the immunological and catalytic characterization of CYP1A, CYP2B, CYP3A, CYP2E, epoxide hydrolase (EH) and UDPGT proteins in the liver of beluga whale from the SL and the Arviat area of the Canadian arctic. Tissues from 11 juvenile and adult female and adult male beluga from Arviat (sampled 2002 and 2003) demonstrated EROD activities of 68 ± 34, 73 ± 30 and 175 ± 85 pmol/mg/min, respectively. Faint cross-reaction with anti-CYP2B antibodies and a lack of formation of hydroxytestosterone isomers associated with CYP2B-like enzymes in other mammals suggests that these belugas have low CYP2B expression, which is consistent with MacKenzie River beluga (White et al., 2000). The formation of 6β-hydroxytestosterone and the presence of a cross-reactive CYP3A-type protein in liver (Fig. 8a) suggests that Arviat belugas possess the metabolic capacity to form arene oxides from *ortho*-chlorine substituted PCBs, which may lead to the formation of HO-PCBs. HO-PCBs were recently reported in liver tissue of the same animals (McKinney et al., submitted for publication). Phase II UDPGT activity was higher than any of the phase I activities in Arviat beluga (Fig. 8b), suggesting the potential biological importance of phase II process to the conjugation of susceptible contaminant substrates such as HO-PCBs and other halogenated phenolics. ΣHO-PCB concentrations reported in the same liver samples from beluga whales from Arviat and the St. Lawrence River Estuary, where ∑HO-PCB to ∑PCB concentration ratios were generally <0.2%, suggested beluga whales have a lower metabolic capacity to form and retain HO-PCBs, and/or greater capacity to deplete HO-PCBs by phase II conjugation processes (McKinney et al., in press). Enzymatically viable tissues are not easily retrieved from SL individuals, which was evident from the lower degree of microsomal enzyme antibody cross-reactivity and the lack of measurable catalytic activity relative to the Arviat animals, from which fresh liver samples were obtained and the enzyme activity preserved (Fig. 8a and b). From the immunological studies, McKinney et al. (2003, 2004) concluded that the SL and Arviat belugas share similar phase I and II hepatic enzyme profiles, and therefore the liver of Arviat beluga could be used to model xenobiotic biotransformation in beluga whale in general. Further research is currently ongoing to determine OC, PCB, polybrominated diphenyl ether (PBDE), HO-PCB and HO-PBDE levels in the liver of Arviat beluga, as well as in vitro metabolism/depletion studies on selected PCB and PBDE congeners using liver microsomes (Li et al., 2003b; McKinney et al., in press; Van Hezik et al., 2001).

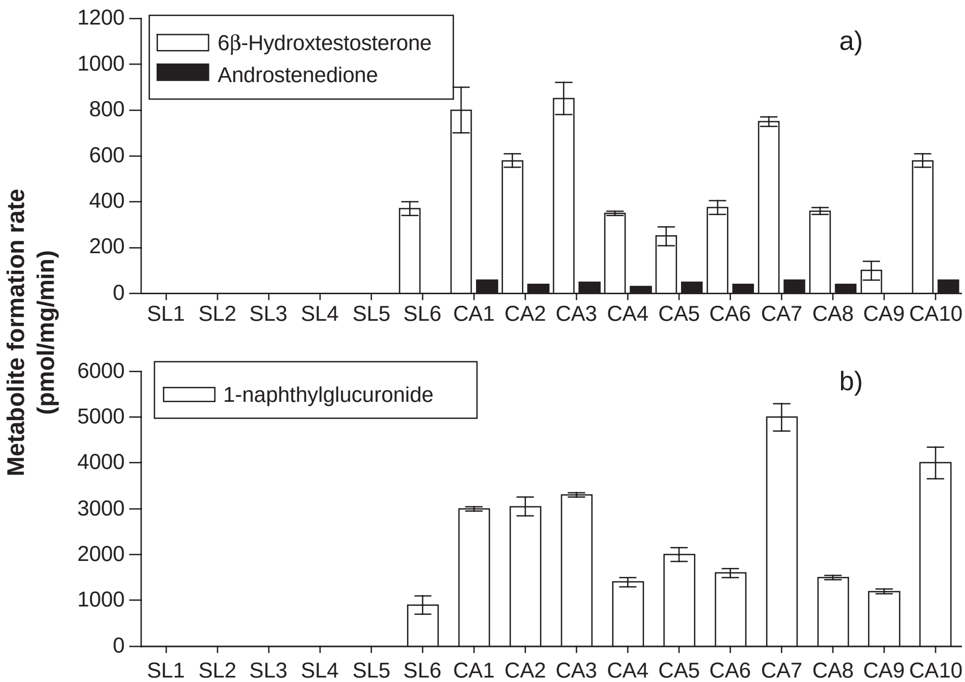

Fig. 8. a. Testosterone metabolite formation in the hepatic microsomes of St. Lawrence River Estuary and Canadian arctic beluga whales (McKinney et al., 2004). Microsomes (1 mg/ml) were incubated with 250 μM testosterone at 37 °C in PBS (100mM, pH 8.0, 3mM $MgCl_2$, 1mM EDTA), and the reaction was initiated by the addition of NADPH regenerating solution. Each bar is the mean of $n = 3$ replicates. Error bars represent the standard deviation. b. Formation of 1-naphthylglucuronide in the hepatic microsomes of St. Lawrence River Estuary and Canadian arctic beluga whales (McKinney et al., 2004). Microsomes (1 mg/ml) were incubated with 250 μM 1-naphthol and 1 mg/mg (protein) Brij-58 at 37°C in PBS (100 mM, pH 8.0, 3 mM $MgCl_2$). The reaction was initiated by the addition of UDP-glucuronic acid (3 mM). Each bar is the mean of $n = 3$ replicates, and error bars indicate the standard deviation.

3. Comparison of contaminant concentrations observed in Canadian arctic biota with threshold levels of biological effects

3.1. OCs concentrations and threshold levels for effects

An extensive review of relevant threshold levels for effects of OCs in arctic biota was carried out for the second AMAP report on persistent organic pollutants (POPs) (de Wit et al., 2004). This report also compared all circumpolar data for arctic organisms to these thresholds, including Canadian data. The amount of data for Canadian species is more limited than what is available for the circumpolar arctic both in terms of the organisms examined and the chemicals measured. Relevant threshold levels available for PCBs, DDT and dioxin-like chemicals for arctic fish, seabirds and marine mammals are provided in Table 2. These chemicals were chosen because they are generally found at the highest concentrations among OCs in arctic biota and the greatest amount of threshold level information has been generated for these chemicals. We have limited the species examined to those with the highest OC concentrations. Levels of OCs in terrestrial animals and invertebrates are generally low and considered to be at levels that are below concern for wildlife health.

3.1.1. Fish

Despite a large amount of research on PCBs and DDT in fish there are few government guideline levels for assessing effects on the fish based on their tissue levels. There are a number of guidelines based on water concentrations, which are not appropriate for this assessment. Environment Canada (2002) has established guideline levels for ΣDDT for the protection of wildlife that consumes fish at 14 ng/g ww, and for ΣPCB, 15 ng/g ww for mammals and 48 ng/g ww for birds. However these guidelines are not for the protection of the fish themselves. Recent work on the effects of PCBs in Arctic char (*Salvelinus alpinus*) provides a lowest observable effects level (LOEL) for EROD induction at 1000 ng/g ww in liver (Jørgensen et al., 1999). This is similar to levels observed to cause EROD induction in Great Lakes fish. Luxon et al. (1987) observed EROD induction in Lake Ontario lake trout (*Salvelinus namaycush*) that had ΣPCB concentrations of 3000 ng/g but not in Lake Huron lake trout which had concentrations of 2000 ng/g. A recent study of EROD activity in shorthorn sculpin (*Myoxocephalus scorpius*) at a PCB-contaminated site in northern Labrador found PCB-related induction above a threshold of 50 ng/g ww (whole fish excluding liver) and 25-fold induction at concentrations of 800 ng/g ww (Kuzyk et al., 2005–this issue). Further laboratory studies of Arctic char have found that single doses of PCB as low as 1000 ng/g body weight affect disease resistance and stress responses in starved specimens (Jørgensen et al., 2002). These results are consistent with Niimi's (1996) conclusion that PCBs cause cellular and biochemical changes in fish at tissue concentrations in the 1000 ng/g range, which was based on a review of a large number of field and laboratory studies.

Fig. 9 shows that, with the exception of burbot (*Lota lota*) from Lake Laberge and Greenland sharks (*Somniosus microcephalus*), and shorthorn sculpin from locally contaminated areas, levels of ΣPCB in arctic fish are well below the 1000 ng/g LOEL for EROD induction. Although the levels of PCBs in the burbot, sculpin and Greenland shark cross this threshold it does not indicate that the health of these fish is compromised by PCBs. Although EROD induction may be correlated to toxic effects (Safe, 1994) there are many biological variables associated with induction of EROD (Stegeman, 1979; Addison and Willis, 1982). In the case of the Greenland shark, there is very little information on either the potential for EROD induction in that species or the levels and potential effects of PCBs and other contaminants on other sharks (Fisk et al., 2002a). Both burbot and Greenland shark have high lipid livers, which may reduce the effects of contaminants by sequestering contaminants and reducing concentrations at sites of toxicological action (Geyer et al., 1993).

Levels of PCBs and ΣDDT in arctic fish do, in general, exceed the Environment Canada guidelines for the protection of wildlife that consume the fish. This does not suggest that PCBs and ΣDDT are influencing the health of the fish but that the concentrations in fish tissues may pose a risk to other wildlife. Also, these guidelines are conservative and are generally lower than levels established by the US EPA. Levels of potential prey items of these arctic fish, generally pelagic zooplankton and smaller fish,

Table 2
Threshold levels of biological effects for ΣPCB, ΣDDT and TEQs for dioxin-like chemicals for fish, seabirds and marine mammals

	Lowest			Lowest		
	NOEL	Effect	Reference	LOEL	Effect	Reference
Fish						
ΣPCB				1000 ng/g ww in liver	EROD induction in Arctic char	Jørgensen et al., 1999
Seabirds						
ΣPCB	2300 ng/g ww in eggs	Hatching success in Forster's terns (*Sterna forsteri*)	Bosveld and van den Berg, 1994	3500 ng/g ww in eggs	Egg mortality in double-crested cormorants (*Phalacrocorax auritus*)	Giesy et al., 1994; Barron et al., 1995
ΣDDT				3000 ng/g ww	Reproduction in bald eagles (*Haliaeetus leucocephalus*)	Wiemeyer et al., 1984
TEQ	4.6 pg TEQ/g ww in eggs	Reproduction in cormorants	Giesy et al., 1994	20 pg TEQ/g	Reproductive effects in wood ducks (*Aix sponsa*)	Giesy et al., 1994
	10 pg TEQ/g ww in eggs	Reproduction in herring gulls	Giesy et al., 1994	200 pg TEQ/g ww in eggs	Reproduction in bald eagles	Elliott et al., 1996
	100 pg TEQ/g ww in eggs	Cytochrome *P*450 1A induction in bald eagles	Elliott et al., 1996			
Marine Mammals						
ΣPCB	1000 ng/g lw in blood serum	Visual memory in rhesus monkeys	Ahlborg et al., 1992	500–1000 ng/g lw in blood serum	Short time memory in rhesus monkeys	Ahlborg et al., 1992
	4000 ng/g lw in liver	Vitamin A reduction in otters	Murk et al., 1998	11,000 ng/g lw in liver	Vitamin A reduction in otters	Murk et al., 1998
TEQ	2000 pg TEQ/g lw in liver	Vitamin A reduction in otter	Murk et al., 1998	210 pg TEQ/g lw in blubber	Immunosuppression in harbor seals	Ross et al., 1995

These values were compiled for the second AMAP assessment of POPs in arctic biota (de Wit et al., 2004). NOEL: no-observable-effects level; LOEL: lowest-observable-effects level.

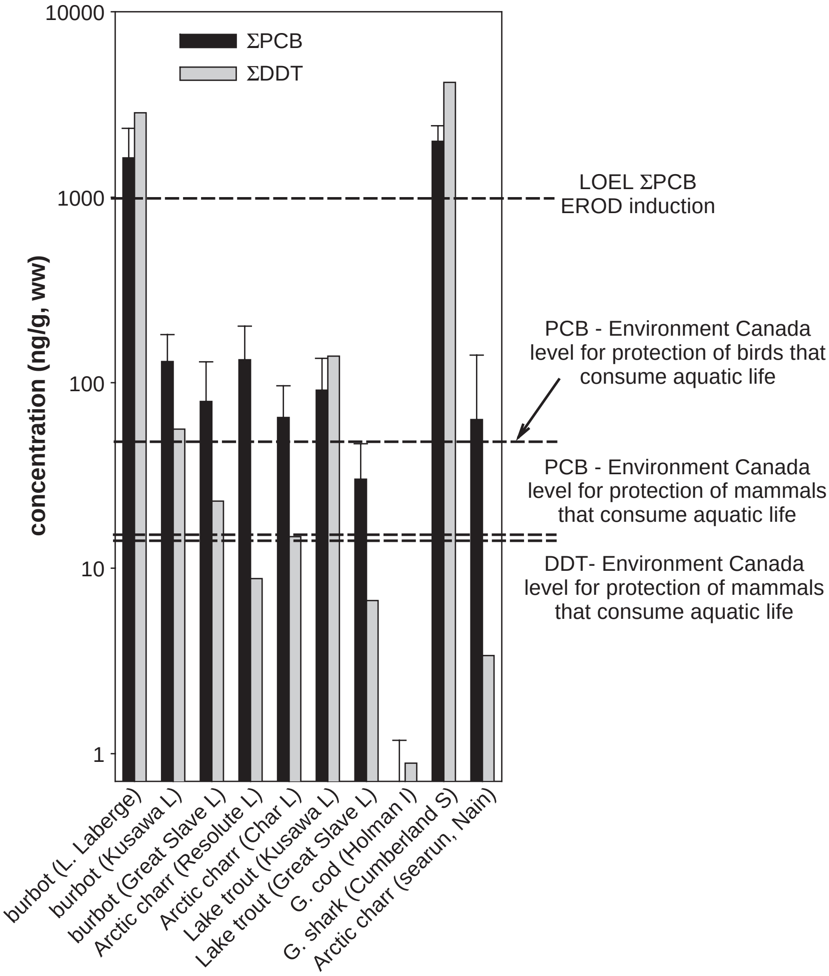

Fig. 9. Mean ΣPCB and ΣDDT concentrations in freshwater and marine fish compared with threshold effects levels (LOEL) for ΣPCB EROD induction in Arctic char (Jørgensen et al., 1999) and Environment Canada guidelines for the protection of wildlife which consume aquatic biota (Environment Canada, 2002). Concentration data references: burbot (Evans and Muir, 2001; Stern et al., 2000; Palmer and Roach, 2001), Arctic char (Muir et al., 2001a), lake trout (Evans and Muir, 2001; Stern et al., 2000; Palmer and Roach, 2001); Greenland cod (Hoekstra et al., 2003b), Greenland shark (Fisk et al., 2002a) and sea run char (Muir et al., 2000). Due to numerous limitations in the threshold data, in quantification of PCBs and problems with extrapolating such data across tissues and species, this comparison should be used with caution.

are well below these guideline levels. The exception may be the Greenland shark that at least occasionally feeds on marine mammals (Fisk et al., 2002a).

There is a fairly substantial database for the effects of dioxin-like chemicals (PCDDs, PCDFs and co-planar PCBs) in fish and in particular for EROD induction. However there have been few new data produced for these chemicals since the first CACAR report (pre-1997). That report found that EROD levels did not vary across the Canadian arctic in freshwater fish, even in lakes, such as Lake Laberge, that were more contaminated (Muir et al., 1997). These EROD levels were lower than levels measured in Lake Ontario lake trout. Since EROD induction is a conservative threshold for the effects of dioxin-like chemicals it would seem unlikely that these chemicals are having a significant influence on the health of Canadian arctic fish. The exception is the EROD induction observed in shorthorn sculpin from a PCB-contaminated site in northern Labrador.

There are some effects data for toxaphene exposure in fish. Mayer et al. (1977) found decreases in hydroxyproline levels in exposed offspring of fathead minnows (*Pimephales promelas*) and channel catfish (*Ictalurus punctatus*). Toxaphene concentrations of 3400 ng/g ww in channel catfish fry tissues were associated with decreased growth, and 600 ng/g led to altered bone development (Stickel and Hickey, 1977), identifying offspring as being more sensitive than adults. Mayer et al. (1975) exposed brook trout (*Salvelinus fontinalis*) to toxaphene in water and found higher mortality during spawning. A 50% mortality was associated with toxaphene concentrations of 870 ng/g ww in muscle and 2400 ng/g ww in whole body. Toxaphene levels in the muscle tissue of some burbot from the east arm of Great Slave Lake, Northwest Territories and Lake Laberge and Kusawa Lake, Yukon, Canada, exceed levels associated with effects on bone development in channel catfish (600 ng/g ww) but not other effects levels. Based on mean toxaphene levels, no anadromous or marine fish from the Canadian arctic exceed effects levels.

In light of the fact that levels of PCBs, DDTs, PCDD/Fs and toxaphene in most Canadian arctic fish are well below the most conservative levels for possible biological effects, the potential for widespread OC-related effects in arctic fish of Canada is low. There are a small number of cases where OCs may be causing some biological effects, in particular Greenland shark, but the threat should not be considered major. There are also a number of lakes, e.g., Lake Laberge, where levels of OCs have been and are higher than threshold levels for possible effects but population level effects have not been observed.

3.1.2. Seabirds

A fair amount of research has been carried out on the effects of OCs in birds. This is in part due to the strong effect of DDT on egg-shell thinning and subsequent reduction of birds of prey populations as well as the reproductive effects of PCBs and dioxin-like compounds seen in many bird species. Because of this there are established no-observable-effects levels (NOELs) and LOELs for seabirds for ΣDDT, ΣPCB and for dioxin-like compound concentrations in eggs. A recent comparison of OC levels in arctic seabird eggs with a variety of threshold effects levels found that only PCB levels in the eggs of glaucous gulls were in the range of the LOEL established for hatching success in white-leghorn chicken (*Gallus domesticus*) (approximately 1000 ng/g ww) (Muir et al., 1999b). The chicken is considered a very sensitive species to the effects of OCs, and therefore may not be a good surrogate for assessing the effects of OCs in seabirds. NOELs and LOELs determined for several wild species of piscivorous birds are shown in Fig. 10, and these may be more relevant for comparisons.

As part of a continuing temporal trend study of contaminants in seabird eggs, thick-billed murre (*Uria lomvia*), northern fulmar (*Fulmarus glacialis*) and black-legged kittiwake (*Rissa tridactyla*) eggs were collected on Prince Leopold Island in Lancaster Sound in 1998 and analyzed for OCs (Braune et al., 2001). ΣPCB and ΣDDT were the predominant OCs in all three species but concentrations were well below threshold levels for biological effects for wild birds (Fig. 10). However, seabirds in contaminated locations and higher trophic level species have ΣPCB concentrations that exceed threshold effects levels. The most highly exposed black guillemots from Saglek Bay (Kuzyk et al., 2005–this issue) had geometric mean ΣPCB levels of 32,900 ng/g ww in eggs (range: 29,500–36,600 ng/g ww), which were an order of magnitude greater than the NOEL for reproductive effects and the LOEL for egg mortality. The eggs of great black-backed gulls (*Larus marinus*) from sites in northern Labrador had mean ΣPCB levels (1090 ng/g and 740 ng/g ww) (Kuzyk et al., 2005–this issue) that were in the range of the LOEL for chickens but below the thresholds for wild birds (Fig. 10). These results suggest that there is a potential risk of adverse contaminant-related effects in gulls and in seabirds inhabiting areas of local contamination in northern Canada.

Recently, TCDD toxic equivalent (TEQs) concentrations were determined in the liver and eggs of seabird species collected in 1975 and 1993 (Braune et al., 2001; Braune and Simon, 2003). TEQs were based on concentrations of PCDDs, PCDFs and non-*ortho* PCBs. As can be seen in Fig. 11, TEQ concentrations in kittiwake, fulmar and thick-billed murre eggs, which is the most relevant tissue to compare with, were above the NOEL levels established for relevant seabird species including herring gulls (*Larus argentatus*) and cormorants (*Phalacrocorax auritus*) and the LOEL for reproductive effects in

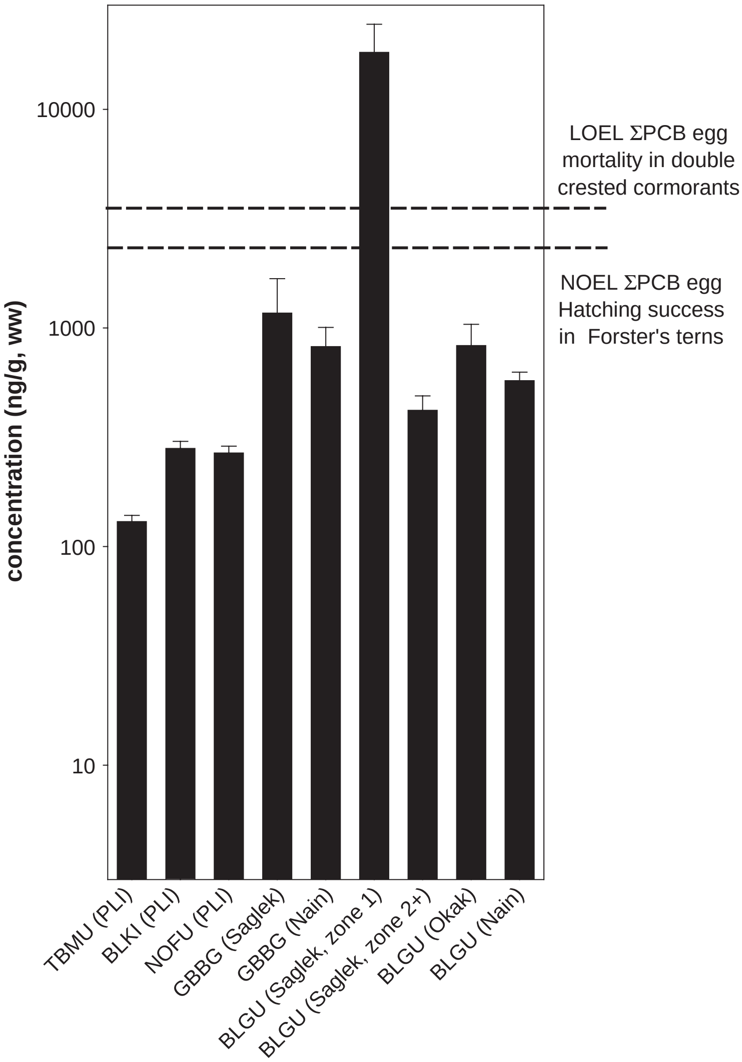

Fig. 10. Mean ΣPCB concentration in arctic seabird eggs from the Canadian arctic. LOEL for PCBs is from Giesy et al. (1994) and Barron et al. (1995), and the NOEL is from Bosveld and van den Berg (1994). Due to numerous limitations in the threshold data, in quantification of PCBs and problems with extrapolating such data across species, this comparison should be used with caution. Prince Leopold Island (PLI) data is from 1998 (Braune et al., 2001) and Saglek (zone 1 = contaminated zone; zone 2+ = outside contaminated zone), Okak and Nain data are from 2002 (Kuzyk et al., 2003, 2005–this issue). TBMU = thick-billed murre, BLKI = black-legged kittiwakes, NOFU = northern fulmar, GBBG = great black backed gull, BLGU = black guillemot.

wood duck, a sensitive species. For the highly exposed black guillemots from Saglek Bay, PCDD/F and non-*ortho* PCB TEQs in liver were 25 pg TEQ/g ww (Kuzyk et al., 2003), which is similar to the levels found in thick-billed murres (eggs and liver) from Lancaster Sound. These levels exceed the LOEL for reproductive effects in wood duck. TEQ values in seabirds were much higher than levels observed in arctic marine mammals (Fisk et al., 2003a). Further investigation of dioxin-like chemicals in arctic seabirds is thus warranted.

A second method for assessing the possible effects of OCs in seabirds is to examine levels of OCs in their diet. Environment Canada (2002) has chosen 14 ng/g ww for ΣDDT, 48 ng/g ww ΣPCB, and 4.75 pg TEQ/g ww for PCDD/F and 2.4 pg TEQ/g ww for dioxin-

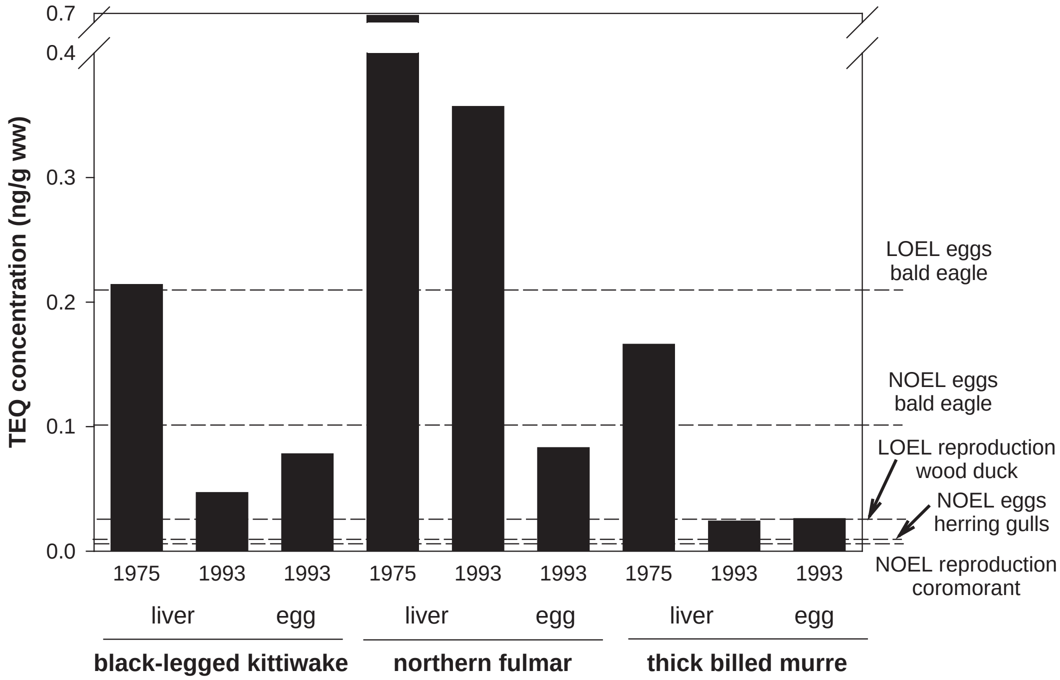

Fig. 11. Mean TEQ concentrations in the eggs and livers of seabirds from Lancaster Sound collected in 1975 and 1993 (Braune et al., 2001; Braune and Simon, 2003) compared with threshold effects levels (for references see Table 2).

like PCBs as the tissue residue guidelines in aquatic organisms that are safe for the protection of organisms/birds that consume these organisms. The US EPA has established less conservative levels for ΣDDT and ΣPCB that are several times higher. Many of the seabirds, including dovekie (*Alle alle*), black guillemots, thick-billed murres and black-legged kittiwakes consume zooplankton and small marine fish. Levels of PCBs and ΣDDT are well below the dietary guideline levels in arctic cod (*Boreogadus saida*) but occasionally exceed the guideline levels in zooplankton and other marine invertebrates. For ΣPCB, the only anadromous/marine fish species that might be preyed on and that exceeds Canadian guidelines for avian predators is Arctic char (muscle) from Nain, Canada (Fig. 9) or shorthorn or four-horn sculpin (*Myoxocephalus quadricornis*) near communities or other local contaminant sources. Few data are available for levels of dioxin-like compounds in anadromous and marine fish so the dietary intake of TEQs cannot be assessed.

Glaucous and great black-backed gulls prey on eggs, chicks, and even adult seabirds as well as fish. For ΣPCBs and ΣDDT, levels in liver and eggs of many seabird species are above the tissue residue guidelines. PCDD/F and non-*ortho* PCB levels given as TEQs exceed Canadian (avian) guideline values in kittiwake, fulmar and murre eggs and liver and black guillemot liver (highly exposed Saglek Bay group). Thus, seabirds that prey on seabird eggs, chicks and adults may have dietary intakes of TEQs, ΣDDT and ΣPCB high enough to lead to effects. Glaucous, great black-backed, and ivory gulls (*Pagophila eburnea*) and northern fulmar are also known to scavenge dead marine mammals and tissues, particularly blubber, and OC concentrations in these are also above the tissue residue guidelines.

The potential for OCs such as ΣDDT and PCBs to cause significant biological effects in seabirds seems to be low for many species in the Canadian arctic but there appears to be some risk of effects for gulls, and for seabirds inhabiting areas with a significant local source, as in Saglek Bay (Section 2.5). Additional work is warranted on dioxin-like chemicals in seabirds.

3.1.3. Terrestrial mammals

Although levels of OCs in arctic terrestrial mammals are generally low, there are a few predator species that have concentrations high enough to warrant some concern. A number of studies are available for establishing threshold effects levels for PCBs in mammals. These include assessments based on subtle neurobehavioral effects in offspring of rhesus monkeys (*Macaca mulatta*) treated with PCBs and children of human mothers eating PCB-contaminated fish, which have resulted in an estimated LOAEL for effects on short-term memory of 500–1000 ng/g lw, and a NOEL for effects on visual memory of 1000 ng/g lw in offspring or cord blood serum (Ahlborg et al., 1992). The mean ΣPCB level in wolverine (*Gulo gulo*) liver from Kugluktuk, NU, was 2000 ng/g lw (Hoekstra et al., 2003a). This concentration exceeds the NOEL and LOEL for subtle neurobehavioral effects and may indicate some risk of these effects in wolverine unless there are significant species differences in sensitivity. Mean concentrations of ΣPCB in wolves (*Canis lupis*) (180 ng/g lw in liver) from the Canadian Yukon (Gamberg and Braune, 1999) are below those expected to result in effects. An assessment of OC concentrations in the diet, assuming a primary diet of caribou (*Rangifer tarandus*), indicates that the levels in caribou are much lower than those expected to cause biological effects.

3.1.4. Marine mammals

PCBs are the predominant OC found in marine mammals and similar to seabirds, a number of studies are available for establishing threshold effects levels for PCBs in marine mammals besides the assessments based on subtle neurobehavioral effects given in Section 3.1.3 (Table 2). Many of the mammalian thresholds for ΣPCB are based on studies in mink (*Mustela vison*) and otter (*Lutra* sp.). Mink, in particular, have been used as a surrogate for seals in many PCB studies, since mink also have delayed implantation. Other than those given in Table 2, available thresholds include the NOEL for otter reproduction (7500 ng/g lw) (Roos et al., 2001) and the NOEL for mink kit survival (9000 ng/g lw) (Kihlström et al., 1992). Mink and otter are extremely sensitive to the effects of PCBs and dioxin-like substances, and thresholds based on effects in these species may overestimate the risk if arctic species are less sensitive. However, there are very few data from which to assess the sensitivity of arctic species relative to these other species. The exception is an in vitro study that found that beluga had similar aryl hydrocarbon receptor characteristics as other mammals that are considered sensitive to the toxic effects of planar halogenated aromatic hydrocarbons (Jensen and Hahn, 2001). As well, semi-field studies carried out to examine the effects of PCBs on captive harbor seals, a non-arctic species, found reduced immune function and disruption of vitamin A physiology in seals fed herring from the relatively contaminated Baltic Sea for 2.5 years. Total PCB concentrations of 16,500 ng/g lw, corresponding to total PCDD/F and non- and mono-*ortho* PCB TEQ levels of approximately 210 pg/g lw, had accumulated in the blubber of these seals, suggesting a threshold for these effects below these concentrations (Ross et al., 1995).

A number of arctic marine mammals have levels of ΣPCBs in blubber or fat that are near or above some of the thresholds. ΣPCB levels in walrus from Canada are below all thresholds for effects (Fig. 12). The mean ΣPCB levels in Canadian ringed seals from most sites exceed the LOAEL for effects on short-term and NOEL for visual memory in rhesus monkeys. Highest ΣPCB levels were found for a ringed seal from Saglek Bay, Labrador (9400 ng/g lw) (ESG, 1998) and these levels also exceed the NOEL for vitamin A and reproductive effects in otters and the NOEL for kit survival in mink. The mean ΣPCB levels in beluga from arctic Canada range from 2500–8000 ng/g lw. These ΣPCB levels exceed the NOAEL and LOAEL for subtle neurobehavioral effects in offspring of rhesus monkeys, the NOEL and LOEL for vitamin A reduction in otter and the NOEL for otter reproduction. Narwhal (*Monodon monoceros*) from all sites have ΣPCB concentrations which exceed the NOAEL and LOAEL for subtle neurobehavioral effects in offspring of rhesus monkeys and at a few sites concentrations exceed the NOEL for vitamin A reduction in otter.

Species that feed on marine mammal tissues also have exposure levels that exceed thresholds for effects. Mean concentrations of ΣPCB in Arctic fox (*Alopex lagopus*) collected in 1999–2000 from Holman Island, Northwest Territories in Canada were 860 ng/g lw in muscle (range: 76–8050 ng/g lw) and 1350 ng/g lw in liver (range: 110-14600 ng/g lw). The mean

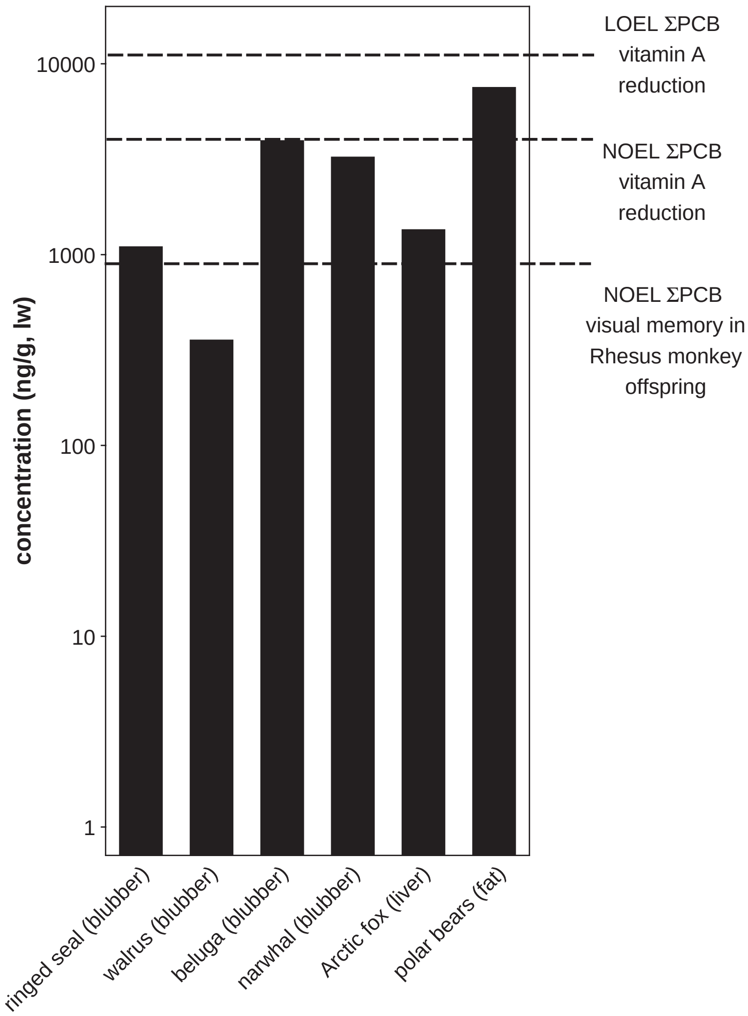

Fig. 12. Mean ΣPCB concentrations in marine mammals compared with threshold effects levels (for references see Table 2). Concentration data: ringed seal (Hoekstra et al., 2002a; Muir et al., 1999a,b, 2001b; ESG, 2002; Fisk et al., 2002b), walrus (Muir et al., 2000), beluga (Metcalfe et al., 1999; Sang et al., 2000; Stern, 1999, 2001; Hobbs et al., 2002; Stern and Addison, 1999; Muir et al., 1999c), narwhal (Stern, 2001), Arctic fox (Hoekstra et al., 2002b), polar bear (Muir and Norstrom, 2000).

ΣPCB levels for Arctic fox exceed the NOAEL and LOAEL for subtle neurobehavioral effects in offspring of rhesus monkeys. However, based on the range, some individuals also have ΣPCB levels that exceed the NOELs for vitamin A reduction in otter, effects on otter reproduction and mink kit survival, the LOELs for vitamin A reduction in otter, and decreased kit production and kit body weight gain in mink (12,000 ng/g lw)(Brunström and Halldin, 2000). ΣPCB concentrations from a recent temporal-trend study of Canadian polar bear (1989 to 1991) from five sites across the Canadian arctic ranged from 720 to 20,200 ng/g lw (Muir and Norstrom, 2000). The geometric mean ΣPCB levels at all sites exceed the NOEL and LOEL for subtle neurobehavioral effects in offspring of rhesus monkeys, at three sites (northern Baffin Bay, northern Hudson Bay, Davis Strait) they exceed the NOEL for vitamin A reduction in otter, and at Davis Strait, they are just at the NOEL for effects on otter reproduction. The ranges indicate that there are individuals that also have ΣPCB levels that exceed the NOEL for mink kit survival, the LOELs for vitamin A reduction in otter, and decreased kit production and kit body weight gain in mink and the threshold for immunosuppression and vitamin A depression in harbor seals.

Although concentrations of ΣPCBs exceed effects thresholds for many marine mammal species and their predators, the available data suggest that TEQ concentrations are generally below the thresholds. For ringed seals from Pangnirtung, non- and mono-*ortho* PCB concentrations in blubber given as TEQs were 0.51 to 0.85 pg/g lw and for ringed seals from Holman Island, TEQs based on PCDD/Fs, non- and mono-*ortho* PCBs ranged from 4.8 to 97 pg/g lw. These TEQ levels are all below the threshold for immunosuppression in harbor seals. TEQ concentrations based on non- and mono-*ortho* PCBs in beluga from Kimmirut were 0.3–2.4 pg/g lw and based on non-*ortho* PCBs in beluga from Cumberland Sound (Baffin Island) were 6.1 pg/g lw. The TEQ levels in beluga are well below the threshold associated with immunosuppression in harbor seals. Levels of 2,3,7,8-TeCDD and 1,2,3,7,8-PeCDD in Canadian polar bears from four sites sampled in 1990 range from 1.2 to 3.8 pg TEQ/g lw in adipose tissue. These are well below the threshold for immunosuppression in harbor seals. This may, however, be an underestimation since mono-*ortho* PCBs usually contribute most to the total TEQs compared to non-*ortho* PCBs and PCDD/Fs (Letcher et al., 1996) and these were not included in the study.

As with the seabirds, an assessment of the possible effects of OCs in marine mammals can be made by examining levels of OCs in their diet. Environment Canada (2002) has chosen 14 and 15 ng/g ww for ΣDDT and ΣPCB, 0.71 pg TEQ/g ww for PCDD/F and 0.79 pg TEQ/g ww for dioxin-like PCBs, and 6.3 ng/g ww for toxaphene as the tissue residue guidelines in aquatic organisms that are protective of organisms/mammals that consume these organisms. Seals and walrus generally consume zooplankton, benthic invertebrates and small fish (e.g., Arctic cod), although this varies with the species of pinniped. Marine invertebrates and fish have a range of OC concentrations, some below the guideline levels (e.g., levels in Arctic cod) and some above (e.g., levels in the euphausiid *Pandalus* in Baffin Bay (ΣPCBs = 29.9 ng/g) (Fisk et al., 2003b)). Mean ΣPCB levels in blue mussels from some sites in northern Quebec, Canada and mean levels in some benthic invertebrates (e.g., the echinoderm *Gorgonocephalus arcticus*) (Fisk et al., 2003b) also exceed the guideline levels. Walrus that consume ringed seals represent somewhat of an exception, since ringed seal blubber has ΣDDT and ΣPCB concentrations that are orders of magnitude above the Canadian guidelines. These walrus have also been shown to have higher OC levels (Muir et al., 1995).

Beluga and narwhal consume zooplankton and small pelagic fish, which have levels of OCs below the Environment Canada guidelines, but they also consume benthic fish, of which some species exceed these guidelines. Polar bears are clearly feeding on dietary items, i.e., ringed seal blubber, that have PCB and DDT concentrations well above the Environment Canada guidelines for these contaminants. Toxaphene levels in ringed seal blubber, where measured, exceed Canadian guidelines for protecting wildlife consumers. Data for dioxin-like compounds are very limited but TEQs in ringed seal blubber from Canada also exceed Canadian (mammalian) guidelines. Besides feeding on terrestrial mammals and birds, Arctic foxes may also eat marine birds and eggs, seal pups and placentas, as well as scavenge on seals, mostly from polar bear kills, depending on what is available at the particular site they are located. For PCBs and DDT, levels in liver and eggs of most seabird species are above the Canadian tissue residue guidelines for mammalian predators. PCDD/F and non-*ortho* PCB levels given as TEQs exceed Canadian (mammalian) guideline values in kittiwake, fulmar and murre eggs and liver and black guillemot liver (highly exposed Saglek Bay group). Thus, Arctic fox that prey on seabird eggs, chicks and adults may have dietary intakes of TEQs, ΣDDT and ΣPCB high enough to lead to effects. The dietary assessment for Arctic fox that feed on seals is the same as for polar bears. As well, Arctic fox that feed on terrestrial prey are likely exposed to OCs at a similar rate as wolves, for which there is little or no concern regarding biological effects.

Overall, comparisons of OC concentrations in arctic marine mammals and their predators to thresholds for effects suggest that some species are at risk from their current levels of exposure to OCs, especially PCBs. However, there are a number of limitations in the comparisons since quantification of PCBs may not be the same in the various studies, there is uncertainty about the biological significance of some effects (e.g., vitamin A depletion), and extrapolating data across species and tissues is problematic. Very few effects thresholds have been developed using arctic marine mammals or closely related species and species differences in sensitivity to contaminants are generally not known. In light of recent biological effects work that suggests that PCB may be influencing the immune (Bernhoft et al., 2000; Lie et al., 2002, 2004, 2005; Skaare et al., 2001c, 2002) and thyroid hormone systems (Skaare et al., 2001a,b) of polar bears, the greatest concern for possible biological effects of OCs in the Canadian arctic is for polar bears. arctic seals, beluga whales, and narwhals also have PCB concentrations that exceed thresholds for effects in other mammals. There is currently minimal data on biological effects in these organisms in the Canadian arctic and further study of potential risks is warranted. Arctic foxes that consume marine mammal tissues may also have OC concentrations that exceed the thresholds but only a few individuals in a population are likely to have such high levels of exposure. With the exception of dietary assessments, TEQ concentrations do not appear to be as significant as PCBs for any of the species considered.

3.2. Metals concentrations and threshold levels of effects

Threshold levels for effects of heavy metals on arctic wildlife were established for the AMAP assessment of metals in the Arctic (Dietz et al., 1998). These threshold levels are to be used in the second AMAP assessment of metals in Arctic (AMAP, in press) and are summarized in Table 3. This report has restricted the metals examined to mercury, cadmium and selenium based on the relative amount of data available for the Canadian arctic and the level of concern for these metals.

3.2.1. Terrestrial animals

Concern regarding the levels of heavy metals in terrestrial animals, particularly the caribou, has been raised (Muir et al., 1997). Unlike OCs, metals are at levels in terrestrial mammals that are similar to or greater than those seen in marine mammals. There are limited new data for metals in the terrestrial arctic, only cadmium and mercury data in caribou. Mean concentrations of mercury and cadmium in the kidney of five caribou herds are below the threshold levels for biological effects established for arctic organisms in the AMAP heavy metals report (AMAP, in press) (Fig. 13). Cadmium concentrations in two of the caribou herds (Beverly and Tayherd) approached the threshold effects level established for cadmium and warrant continued monitoring. However, past assessments of cadmium in kidney and liver of caribou concluded that levels were comparable to findings in other big game species of Canada and should be considered natural given the general elevation of concentrations on a circumpolar basis (Elkin, 1997).

3.2.2. Fish

Few guidelines for effects of metals in fish have been established for the protection of the fish. Guide-

Table 3
Threshold levels of biological effects for cadmium, mercury and selenium for birds and mammals as compiled by Deitz et al. (1998) and AMAP (in press)

Metal	Group	Tissue	Concentration ug/g ww	Effect	Reference
Cadmium	Birds	Liver	>40	Cadmium poisoning	Furness, 1996
		Kidney	>100	Cadmium poisoning	
	Mammals	Liver	>20–200	Potential renal dysfunction	Law, 1996
		Kidney	>50–400	Potential renal dysfunction	
Mercury	Birds	Liver	>30	Lethal level in free ranging birds	Thompson, 1996
	Marine mammals	Liver	>60	Liver damage	Law, 1996
	Terrestrial mammals	Liver	>25	Laboratory succumbed animals due to mercury intoxication	Thompson, 1996
Selenium	Birds	Liver	>9	Deformed embryos	Heinz, 1996
	Terrestrial mammals	Liver	>7	Hepatic lesions	WHO, 1987

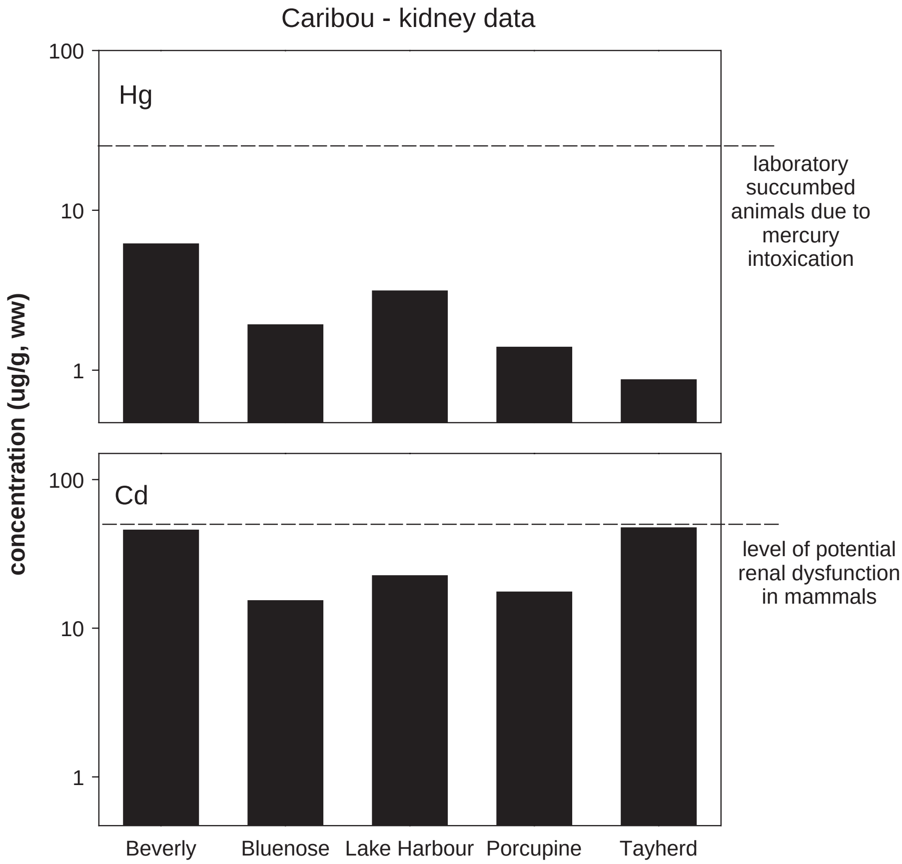

Fig. 13. Mercury and cadmium concentrations in caribou (Fisk et al., 2003a) compared with threshold effects levels (from Thompson, 1996; Law, 1996). Due to numerous limitations in the threshold data and problems with extrapolating such data across tissues and species, this comparison should be used with caution.

line values are available for fish consumers, both human and wildlife. Mercury levels in freshwater fish surpass the guidelines for human subsistence or commercial sale in many lakes and for many fish-eating species in the Canadian arctic.

3.2.3. Seabirds

A fairly large database on metals in arctic seabirds has been generated and is available for assessment of potential biological effects (Fisk et al., 2003a). Using the guidelines established by AMAP (Table 3) levels of cadmium and mercury in the liver of 8 seabird species are below threshold levels for biological effects (Fig. 14). Levels of selenium in the livers are near threshold levels for causing deformities in embryos but this should be considered a very conservative threshold level. These results are consistent with the second AMAP assessment of metals in the Arctic, which concluded that there was little evidence of biological effects of heavy metals in seabirds (AMAP, in press).

The risks of elevated mercury concentrations in freshwater fish have not been assessed for fish-eating wildlife in the Arctic, such as river otter (*Lutra canadensis*), mink, red-throated (*Gavia stellata*) and arctic loons (*Gavia arctica*).

3.2.4. Marine mammals

Metal data has been generated since 1996 for a number of marine mammal species, and for ringed seals and beluga from a number of locations (Fisk et

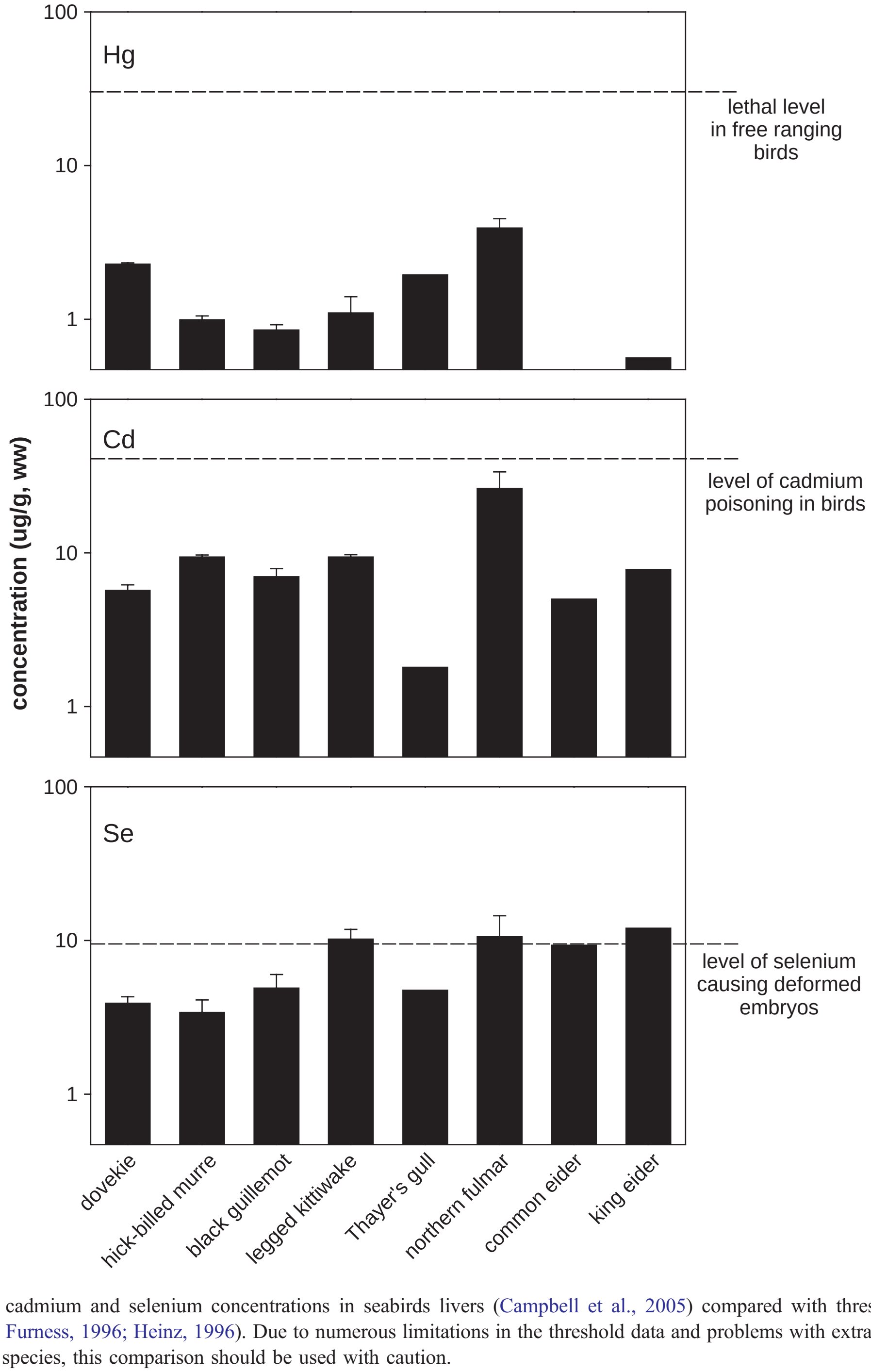

Fig. 14. Mercury, cadmium and selenium concentrations in seabirds livers (Campbell et al., 2005) compared with threshold effect levels (Thompson, 1996; Furness, 1996; Heinz, 1996). Due to numerous limitations in the threshold data and problems with extrapolating such data across tissues and species, this comparison should be used with caution.

al., 2003a). Levels of mercury and cadmium in ringed seals and walrus are below threshold levels established by the AMAP report on metals in the Arctic (Fig. 15). Levels of selenium in ringed seals from Ungava Bay are at the threshold effects level but this threshold is conservative.

4. Summary and conclusions on potential biological effects of contaminants on Canadian arctic biota

The lack of information on the biological effects of OCs and metals in arctic biota was identified as a major data gap in the first CACAR assessment (Muir et al., 1997). Since that time (post-1996) there have been few biological effects related studies on Canadian arctic biota, most likely because of a low level of concern and the difficulty and high expense of carrying out appropriate studies in the Arctic.

Although there is currently little or no evidence to suggest that OCs or metals are having a widespread

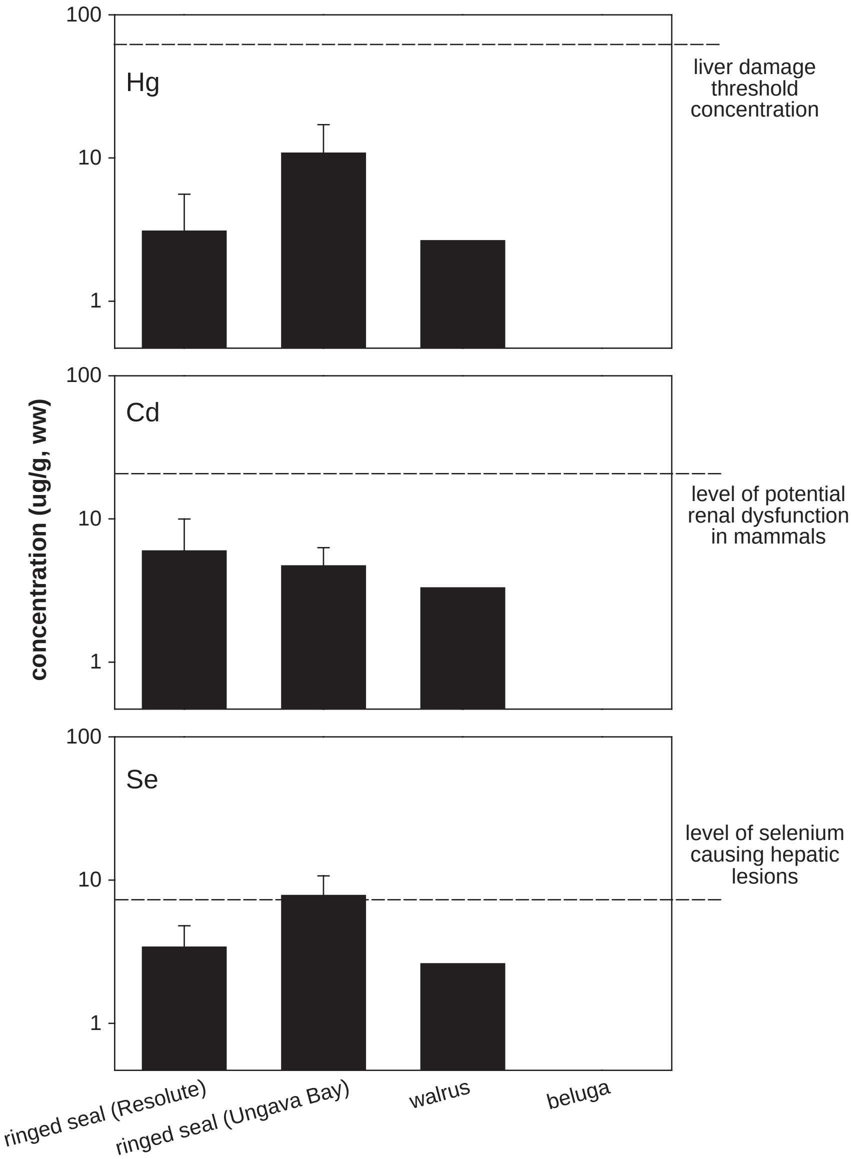

Fig. 15. Mercury, cadmium and selenium concentrations in marine mammals (Fisk et al., 2003a) compared with threshold effects levels (Law, 1996; WHO, 1987). Due to numerous limitations in the threshold data and problems with extrapolating such data across tissues and species, this comparison should be used with caution.

negative effect on the reproduction or survival of Canadian arctic wildlife, very few studies have investigated effects at this level of organization and it is difficult to speculate on the basis of the minimal data available. Correlative relationships between circulating levels of thyroid hormones and PCB and HO-PCB in polar bears suggests potential endocrine- and/or hormone-related impacts on at least one Canadian polar bear population (Resolute Bay area), but again these impacts are difficult to assess and it is not possible to draw population-level conclusions from the available data. High PCB and organochlorine pesticide concentrations were found to be strongly associated with impaired humoral and cell-mediated immune responses in polar bears from Svalbard and from Churchill, implying an increased infection risk that could impact the population. Based on comparisons to effects thresholds, polar bears are at risk of possible biological effects from their current levels of contaminant (particularly PCB) exposure. Preliminary findings of higher levels of contaminants in mothers that lost their cubs during their first season (compared to mothers that did not) should be further investigated along with other components of polar bear health.

Other marine mammals and species that consume marine mammal tissues would also appear to be at risk, based on comparisons between current levels and effects thresholds. However, there is significant uncertainty in these comparisons and there are virtually no data from the Canadian arctic to better inform an assessment. Further study of potential effects in highly exposed arctic marine mammals is warranted.

Weak relationships between metal concentrations and health biomarkers in eider ducks have been found but this may be due to seasonal changes in the ducks. The levels of OCs and metals in most seabirds are typically below effects thresholds. Seabirds with higher levels of OCs, such as the glaucous gulls, have levels that exceed thresholds for very sensitive species (like chickens) and may therefore be susceptible to some effects from contaminants. Where local contamination has occurred, physiological effects have been observed in arctic seabirds. A recent study of physiological effects in guillemot nestlings with elevated PCBs in the Saglek Bay region demonstrated that elevated levels of PCBs can cause effects. This study found that PCB levels only 1 to 2 orders of magnitude higher than typical levels caused enzyme induction and changes in the vitamin economy of the birds.

With current OC exposure levels exceeding effects thresholds in several arctic species and preliminary evidence of effects (e.g. among polar bears), there is sufficient reason to continue monitoring the health of the most exposed arctic wildlife populations and investigating possible biological effects of anthropogenic contaminants. Impending climate warming, with possible effects on contaminant dynamics, and the discovery of a range of new contaminants, several of which are increasing in the Arctic, further justify these efforts. Recent studies on Canadian polar bears have also suggested that changing ice conditions can affect the recruitment and population characteristics and size of polar bears (Stirling et al., 1999). Forecasted changes in climate and ice conditions in the Canadian arctic may result in increased significance of stress related to anthropogenic contaminants exposure in polar bears and potentially other species, as well.

Although few examples were found of current metals concentrations exceeding threshold effect levels, there is evidence that warmer temperatures can result in higher tissue levels of metals in fish and beluga. Monitoring of arctic wildlife health, in conjunction with continued monitoring of trace metal concentrations is recommended.

5. Gaps and recommendation for assessing biological effects of contaminants in the Canadian arctic

There have been few studies on the biological effects of contaminants in arctic organisms from Canada or the world. Some biochemical effects have been found in more contaminated species but the importance of this on population dynamics is unknown. There is often a lack of basic biological information for arctic organisms that hinders efforts to assess potential changes caused by contaminants, or any anthropogenic stresses. With potential changes in contaminant dynamics due to climate changes the levels of contaminants in arctic organisms may also change.

Most effects thresholds established for contaminants have been generated using non-arctic animals. Arctic organisms often have unique biological strategies and systems to deal with the harsh arctic climate. Therefore, comparison of threshold effects levels to current levels in arctic organisms may be confounded by other stressors, for example long periods of fasting. Effects studies and risk assessment should focus on thresholds for arctic species so that proper comparisons can be made. More knowledge about starvation effects is needed in birds and mammals as well.

Physiological changes have been observed in seabirds with higher levels of PCBs due to local contamination. The impact of these changes and other sensitive effects on population dynamics should be studied. The health of wildlife near other sources of local contaminants should also be assessed.

Biological effects of new chemicals, as well as persistent, toxic metabolites such as HO-PCBs, other HPCs and $MeSO_2$-PCBs and -DDEs require additional study. For example, clearly the correlative evidence in polar bear, and from cause–effect studies at the cellular level in vitro (human and fish), suggest the potential for the modulation of hormone-dependent processes, for both classes of these primary and metabolite contaminants. However, whether there is a health impact at the population or ecosystem level, or what the relative importance is of these contaminant stressors compared to the complexity of multiple environmental stresses, is currently unknown and unstudied. Interpretation of correlative hormone studies is hampered by lack of information on what other variables may affect these in wild populations. This makes drawing conclusions from some biomarker studies tenuous.

There are indications of compromised immune systems and modulation of hormone-dependent systems in polar bears. Effects of high PCB and organochlorine pesticide levels and reduced immune response were found in polar bears from both Canada and Svalbard. It may be that bears with elevated PCB levels are more susceptible to viral infections similar to the situation with seals in the North Sea in the 1980s (Heidejorgensen et al., 1992). Furthermore, we are currently unaware of basic endocrine-related effects, or any other effects for that matter, on other arctic species and populations. In addition, the influence of seasonal bioenergetic and other physiological changes on observed effects is also not known at this time for organohalogens in general. The contaminant stress profile becomes more complex when one considers "new" contaminants such as PBDEs and perfluorinated acids (PFAs). For example, can the effects of chemicals like the PBDEs be discerned in the presence of much higher levels of PCBs? Do other chemicals, particularly perfluorinated acids (PFAs), with unique properties have different effects that can be measured? The complexity of the overall contaminant exposure profile and the overlapping and probable unique interactions of different contaminant classes with biochemical processes are factors to consider as well.

Biological effects research in Canadian arctic species other than polar bears is needed. Other than studies on the effects of metals in eider ducks, where metal levels were low and minor effects were seen, and on the effects of PCBs on marine organisms near a contaminated site, there have been no recent studies directly assessing the biological effects of contaminant on Canadian arctic wildlife. Biomarkers for toxaphene exposure need to be developed and biological effects monitoring undertaken in those species with high levels.

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