

Chapter 6

Biological Effects

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 preventive value.

In the previous AMAP assessment, there was relatively little knowledge of the biological effects of POPs in Arctic species, but concern was raised that the concentrations in some species put them at risk for effects on reproduction, the immune system, and on subtle neurobehavioral functions. Newer studies of biomarkers linked to POP exposures are beginning to show evidence that chemical contaminants may be present in sufficient quantities to have biologically significant effects on some species.

Two approaches have generally been taken in identifying and estimating the risk for possible effects. The first involves comparison and extrapolation. Researchers assess the risk for possible effects by comparing levels of POPs in Arctic species of interest to known detrimental levels, with this knowledge coming from laboratory studies, semi-field studies or from observations on affected animals in the wild. Extrapolation is routinely used in toxicology and is based on the conserved nature of many of the endpoints being measured across taxa. These include commonalities in endocrinology, immunology, and to a lesser extent, anatomy and reproductive biology. The extrapolation is based on similarities in mechanisms of action (e.g., mediation through the Ah-receptor, hormone receptors, CYP1A induction, OH-PCB disruption of vitamin A/thyroid, etc.). The difficulties in extrapolation relate generally to differences in sensitivity, where the same types of effects are seen but at different doses, or to differences in structure and function that are fairly obvious among species, especially those that are more distantly related to each other (i.e. fish vs. birds vs. mammals) (Kim and Hahn, 2002).

Comparison and extrapolation have some inherent weaknesses, however. Laboratory animals are most often exposed to single POPs or technical products at high doses for short periods of time, and it is difficult to extrapolate the toxic effects seen at high acute doses to possible adverse effects at lower but chronic exposures. Wild animals are generally exposed to lower concentrations of OCs than laboratory rodents in experimental studies, but they are exposed to mixtures of POPs and other stressors, and they are exposed over their entire lifetime.

They are also exposed to weathered mixtures due to the change in composition of many POP mixtures caused by abiotic degradation, metabolism and subsequent filtering up through the food web. For example, marine mammals at high trophic levels will be exposed

to very different PCB compositions, expressed as Σ PCBs, than is seen in a PCB technical product. As an illustration, a weathered mixture of PCB in contaminated fish was found to be more toxic to mink than a comparable Σ PCB concentration of a technical product (Aroclor 1254) (Giesy and Kannan, 1998).

Differences in species sensitivities to the effects of POPs make it difficult to know which of the tested species best represents those in the Arctic. For example, several Arctic species have delayed implantation (mink, otter, other mustelids, seal, walrus, and polar bear), which may make them more sensitive to the reproductive effects of POPs than tested laboratory animals without delayed implantation (Sandell, 1990). Arctic species also differ from laboratory animals because of their fat dynamics, differences in life styles and life strategies, and differences in toxicokinetics. For example, there are wide differences in metabolic capacity in different Arctic species compared to laboratory animals (see Section 3.2.2). Very little is known about the sensitivity of Arctic species, particularly marine mammals, to the effects POPs.

The second approach studies biological effects by examining subtle indicators of biological responses (biomarkers) to contaminants. Examination of the animals for responses known to be associated with the contaminants found in their tissues is perhaps the only way to make a convincing case either for or against the hypothesis that trace contaminants are acting biologically on the animals. Almost any 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. Most of the biomarkers studied have established sensitivities (in laboratory animals) to some of the same contaminants measured in Arctic marine mammals and fish, notably several PCB congeners, PCDD/Fs, and PAHs.

The MFO cytochrome P450 system, a ubiquitous enzyme system common to mammals, birds, fish, and microorganisms, has probably been one of the most widely used biomarkers to date, with numerous laboratory and field cases of responses established (Payne *et al.*, 1987; Rattner *et al.*, 1989; Goksøyr and Förlin, 1992; Haasch *et al.*, 1993; Beyer *et al.*, 1996; Hylland *et al.*, 1996). The preferred field study design has been the comparison of an exposed group of individuals with similar groups not exposed to the same source. Another design, less commonly encountered, is the comparison of individuals within a group to search for linkages between biomarker values and exposure as indicated by chemical residues. This is a valuable approach for those contaminants stable enough to remain identifiable as residues.

There are also limitations with these types of studies, however. It is not possible to determine causality, only that a statistical association has been found between a biomarker and the contaminant in question. Most POPs covary, and thus, it is not possible to state unequivocally that the biomarker response has been caused by a particular contaminant. There may be other contaminants not analyzed that are just as important, or the response may be the result of synergistic, additive or antagonistic effects of contaminant mixtures. Biological variables such as age, sex, body condition, presence of disease or other stresses also may act as confounders, as they can cause similar biological effects as those seen from POPs. This means that for most reported biological effects in wildlife, the evidence for a causal link with a specific chemical contaminant is weak or non-existent. This is mainly due to the complexity of contaminant mixtures, the lack of chemical exposure data, lack of data on the sensitivity of the species concerned, and knowledge of mechanisms of action. Understanding the linkages between contaminants and health effects (e.g., on reproduction or immunosuppression) is most likely to come from studies in laboratory animals. Crucial in establishing causal evidence for chemical-induced wildlife effects are semi-field or laboratory studies using the wildlife species of concern. Semi-field studies represent a useful approach to bridge the gap between the controlled conditions of the laboratory experiments and the uncontrolled exposure conditions in the field. Based on these different types of studies, a weight-of-evidence argument can be established. For example, based on the combination of field, semi-field, and laboratory studies, it was concluded that dioxin-like PCBs that accumulated through the marine food chain aggravated the severity and extent of the 1988 morbillivirus-related epizootic in harbour seals in northwestern Europe (Ross *et al.*, 1996; Vos *et al.*, 2003).

In the following assessment, results from studies of biomarkers in Arctic biota are presented first if these have been performed. Where possible, levels of specific POPs in Arctic biota are also compared to no-observed-adverse-effect-levels (NOAEL) or no-observed-effect-levels (NOEL), and lowest-observed-adverse-effect-levels (LOAEL) or lowest-observed-effect-levels (LOEL) known to cause subtle effects in sensitive species. The purpose of these comparisons is to assess the likelihood that some Arctic species may be at risk for the effects of some POPs, and to identify these species, the types of effects they may be at risk for, and the contaminants that might be associated with these effects.

There are some problems in making such comparisons, particularly for mammals, and some caution is advised. Where possible, effects thresholds derived from wild species have been used, but the problems with comparing different species has already been discussed above. In the case of mammals, the Σ PCB thresholds are based on concentrations found in different types of tissues (blood, blubber, muscle) with very different lipid contents.

Some scientists consider it better to compare thresholds based on the same tissue on a wet weight basis in both the threshold species and the Arctic species,

whereas others prefer comparisons on a lipid weight basis. Wet weight comparisons are not always possible because different tissues with different lipid contents have been analyzed. Therefore, the values in this report have been normalized to lipid content to enable comparison of POP concentrations in different tissues. This conversion assumes that POP concentrations are evenly distributed in the lipid stores of all organs in an organism and are not fluctuating and, therefore, that the lipid-normalized concentrations in all organs will be the same. This is also known to not always be the case. In seals, however, lipid-normalized levels in blubber, liver and blood in the same individual were within a factor of 2 (Boon *et al.*, 1994) and in other marine mammals, they were usually within a factor of 2 and at most, a factor of 5 (Aguilar, 1985; Boon *et al.*, 1987; Boon *et al.*, 1992; Kannan *et al.*, 1993; Bernhoft and Skaare, 1994; Kannan *et al.*, 1994; Jenssen *et al.*, 1996; Nakata *et al.*, 1998b).

Lipid conversion also adds more uncertainty into the values since there are errors introduced from the lipid determination. In some cases, literature threshold values have been given on a lipid weight basis with no information available for converting back to wet weight. The comparisons for birds are somewhat less problematic, as thresholds have been determined in eggs and are expressed on a wet weight basis, and most data available for Arctic birds are for eggs on a wet weight basis as well.

Another weakness in this approach is that methods for quantifying Σ PCBs used in the thresholds studies and in the analyses of Arctic species are not identical, and use varying numbers of congeners or are based on quantification using a technical product (total PCB). Similarly, TEQs may have been calculated using PCDD/F, nPCB and/or mono-ortho PCB concentrations combined with toxic equivalency factors (TEF) from different schemes. Previously, TEF schemes were based on mammalian models, but fish- and bird-specific TEFs are now available (van den Berg *et al.*, 1998). Thus, the terms Σ PCBs and TEQ may not be completely comparable, and this may lead to under- or overestimations of exposure and risk. Mammalian TEFs are derived from studies in laboratory rodents and may not be appropriate for risk assessment in marine mammals. Similarly, bird TEFs are based on studies in domestic chickens, which are more sensitive to dioxin-like substances than fish-eating bird species (Sanderson *et al.*, 1998).

Many of the mammalian thresholds for Σ PCBs are based on studies in mink and otter. Mink, in particular, have been used as a surrogate for seals in many PCB studies, since mink also have delayed implantation. Mink and otter are extremely sensitive to the effects of PCBs and dioxin-like substances, and thresholds based on effects may overestimate the risk if Arctic species are less sensitive.

In a few cases, a threshold value is available only for laboratory species such as rats, mice, rabbits or dogs (e.g., toxaphene, PFOS, TBT and DBT). Because of the presence of these 'new' substances in the Arctic, these thresholds have been used to indicate whether any species is at potential risk for these substances, rather than not making any statement at all. For some

Table 6.1. Selected criteria, action levels or guidelines for critical pollutants in the Great Lakes. Parts of the table are modified from De Vault *et al.* (1995). All values for fish/aquatic organisms are based on wet weight, and for sediment on dry weight.

Contaminant	US FDA ¹	IJC ²	GLI ⁴	OMEE ⁵	IJC ³	TRG ⁶	USEPA ⁷	ERL ⁸	ERM ⁸	EQG ⁶	EQG ⁶
	Fish	Water	Water	Water	Fish tissue	Fish tissue	Fish tissue	Sedi-ment	Sedi-ment	Water	Sedi-ment
2,3,7,8 TCDD/ other PCDD/Fs as TEQs	25 pg/g		0.0096 pg/L			0.71 pg/g ^{b,c} 4.75 pg/g ^{b,d}	0.5 pg/g			0.02 pg/L	0.091 pg/g
DDT	5 µg/g	0.003 µg/L	0.00087 µg/L	0.003 µg/L	1.0 µg/g ^a	0.014 µg/g ^b	0.039 µg/g	1.6 ng/g	46 ng/g		
Total PCBs			17 pg/L	0.001 µg/L	0.1 µg/g ^a	0.015 µg/g ^{b,c} (0.79 pg/g TEQ) ^{b,c} 0.048 µg/g ^{b,d} (2.4 pg/g TEQ) ^{b,d}	0.16 µg/g	23 ng/g	180 ng/g		
Mirex	2 µg/g	< detection		0.001 µg/L							
Toxaphene	5 µg/g	0.008 µg/L		0.008 µg/L		0.0063 µg/g ^b					
Aldrin/dieldrin	0.3 µg/g	0.001 µg/L		0.001 µg/L	0.3 µg/g ^a						

^a Whole fish.

^b Aquatic organism.

^c TEQ refers to dioxin toxic equivalents using toxic equivalency factors (TEFs) for mammals (van den Berg *et al.*, 1998).

^d TEQ refers to dioxin toxic equivalents using toxic equivalency factors (TEFs) for birds (van den Berg *et al.*, 1998).

¹ USFDA (U.S. Food and Drug Administration) Action Levels in edible portions of fish for regulation of interstate commerce.

^{2,3} International Joint Commission Annex 1 – objectives for protection of aquatic life and wildlife.

⁴ USEPA (U.S. Environmental Protection Agency) Great Lakes Water Quality Guidance proposed criteria for protection of wildlife (USEPA, 1995).

⁵ OMEE (Ontario Ministry of Environment and Energy) guideline for the protection of sediment quality (OMEE, 1993).

⁶ Canadian Tissue Residue Guidelines for the protection of wildlife consumers of aquatic biota (Environment Canada, 2002). Total PCB values are calculated assuming that average PCB TEQs are equal to 5×10^{-5} of ΣPCB concentrations, which is the ratio in Arctic char (D. Muir, unpublished data).

⁷ USEPA guideline values for assessment of hazards to fish-eating wildlife (USEPA, 1995).

⁸ ERL = Effects range low and ERM = effects range median for sediments (Long *et al.*, 1995).

substances such as PBDEs, no threshold data are currently available, and the concentrations in Arctic biota can not be assessed for possible effects.

An attempt has also been made to compare POP levels in the diet of selected Arctic biota to known dietary no-observed-adverse-effect-concentrations (NOAEC) or no-observed-effect-concentrations (NOEC), and lowest-observed-adverse-effect-concentrations (LOAEC) or lowest-observed-effect-concentrations (LOEC), or to environmental quality criteria/guideline values for protecting aquatic biota/fish-eating wildlife that have been developed in various countries (Table 6·1). There are considerable limitations in this latter approach, as there is a general lack of knowledge on the diets of many Arctic organisms. It is also assumed that predators eat only one type of food, but even where food preferences are known, there may not be analytical data for these particular food items.

Thresholds for effects in birds, mammals, and fish

Birds

In several review articles (Bosveld and van den Berg, 1994; Giesy *et al.*, 1994b; Barron *et al.*, 1995), the no-effect and low-effect levels for bird eggs and adults as well as dietary intakes associated with no or low effects have been compiled from the literature for ΣPCBs and dioxin-like compounds. For eggs of fish-eating and predatory birds (bald eagle, herring gull, Caspian tern, double-crested cormorant (*Phalacrocorax auritus*), common tern, Forster's tern (*Sterna forsteri*), great blue heron, black-crowned night heron (*Nycticorax nycticorax*)), the following ranges were found. For reproductive success, the NOEL range for ΣPCBs was 1.3-11 µg/g ww in eggs. The LOEL range for various endpoints of reproductive success (hatching success, egg mortality, deformities, and parental attentiveness) ranged from 3.5

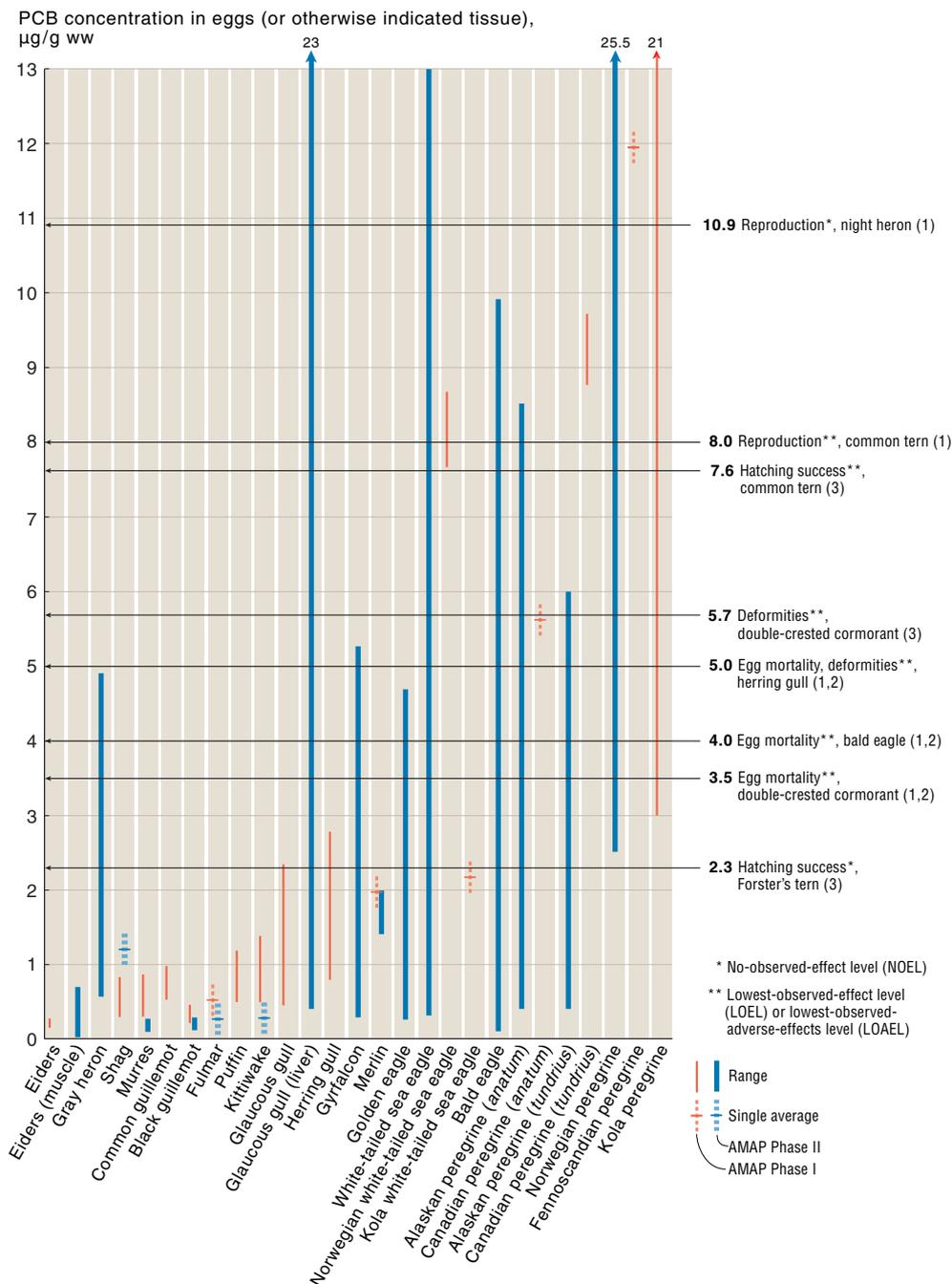


Figure 6-1. Concentrations of ΣPCB in Arctic bird eggs (or otherwise indicated tissue) compared to thresholds for avian effects (1. Barron *et al.*, 1995; 2. Giesy *et al.*, 1994b; 3. Bosveld and van den Berg, 1994). Due to numerous limitations in the thresholds data, in quantification methods for PCBs and problems with extrapolating such data across species, this comparison should be used with caution. See pages 163-165 for more details. AMAP Phase I data are from de March *et al.*, 1998.

to 22 μg $\Sigma\text{PCBs/g ww}$ in eggs. For adults, ΣPCB concentrations in brain tissue higher than 300 $\mu\text{g/g ww}$ were associated with mortality. Relevant thresholds are presented in Figure 6-1.

PCBs with TCDD-like activity (nPCBs, mono-ortho PCBs) have been shown to adversely affect patterns of survival, reproduction, growth, and metabolism, with CB126 being among the most toxic of all PCB congeners to birds (Eisler and Belisle, 1996). For dioxin-like substances, the NOAEL range for reproductive effects was 1.5-200 pg TEQ/g ww in eggs, and the LOAEL range for various reproductive endpoints (deformities, hatching success, and mortality) ranged from 10 to 2200 pg TEQ/g ww in eggs. Elliott *et al.* (1996) have suggested a

NOEL of 100 pg/g and a LOEL of 210 pg/g TEQ on a whole egg (wet weight basis) for induction of cytochrome P450 (CYP) 1A in bald eagle chicks. Bosveld *et al.* (2000) have estimated the LOEL for induction of CYP1A in the common tern at approximately 25 000 pg TEQ/g lw in liver. These and other relevant thresholds are presented in Figure 6-2.

It has been concluded that peregrine falcon eggs with DDE residues of 15 to 20 $\mu\text{g/g ww}$ would experience reproductive failure (20% eggshell thinning) (Peakall *et al.*, 1990). The adverse effects threshold for dieldrin in peregrine falcon eggs is 1-4 $\mu\text{g/g ww}$, and for heptachlor epoxide, the adverse effects threshold is 1.5 $\mu\text{g/g ww}$ (Peakall *et al.*, 1990). For HCB, results from Vos *et al.*

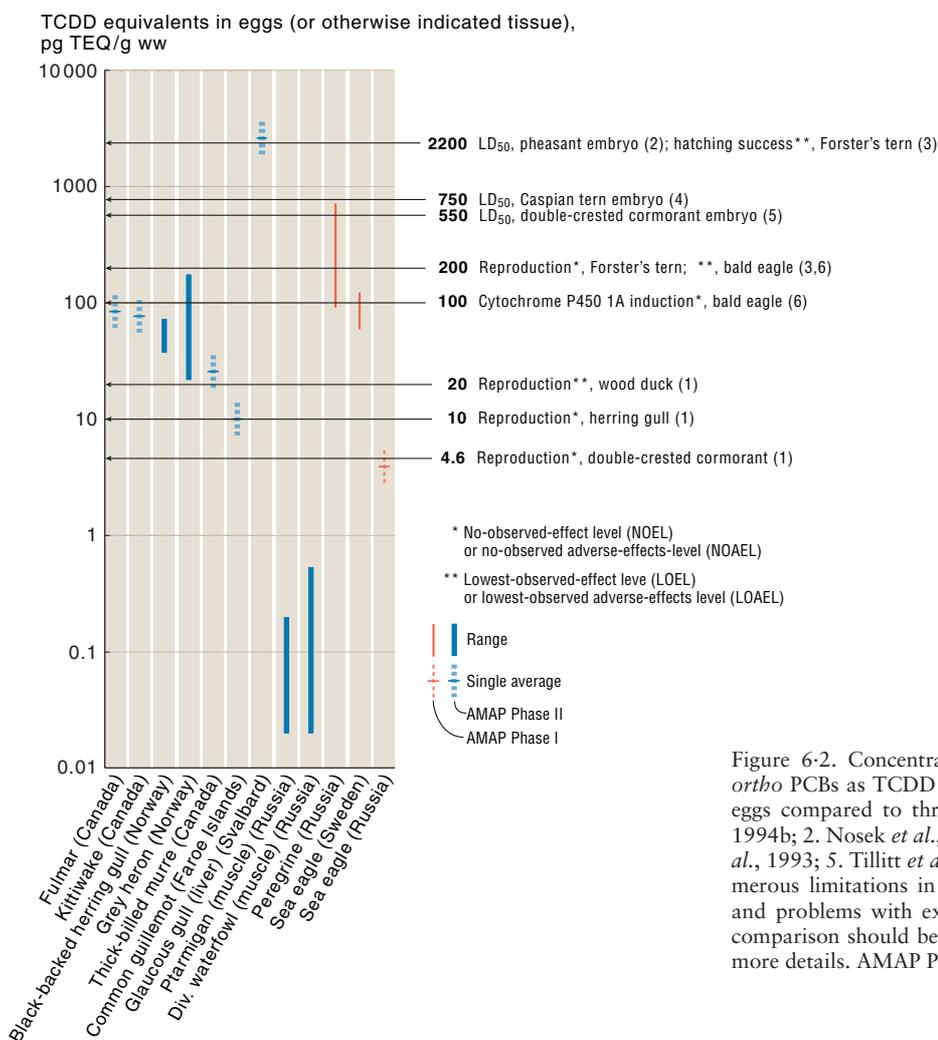


Figure 6-2. Concentrations of PCDD/Fs and non- and mono-ortho PCBs as TCDD equivalents (pg TEQ/g ww) in Arctic bird eggs compared to thresholds for avian effects (1. Giesy *et al.*, 1994b; 2. Nosek *et al.*, 1992; 3. Kubiak *et al.*, 1989; 4. Ludwig *et al.*, 1993; 5. Tillitt *et al.*, 1992; 6. Elliott *et al.*, 1996). Due to numerous limitations in the thresholds data, in calculating TEQs and problems with extrapolating such data across species, this comparison should be used with caution. See pages 163-165 for more details. AMAP Phase I data are from de March *et al.*, 1998.

(1972) indicate a NOEL in kestrels (*Falco tinnunculus*) of 40-49 µg/g ww in liver. Lindane concentrations of 100-200 µg/g ww in eggs of quail and chickens are associated with decreased egg production (Whitehead *et al.*, 1972a; 1972b; 1974). For bald eagle eggs, DDE concentrations below 3 µg/g ww are associated with near normal productivity, concentrations above 5.1 µg/g ww are associated with marked productivity declines, and concentrations above 15 µg/g ww are associated with complete reproductive failure (Wiemeyer *et al.*, 1984).

Mammals

Platonow and Karstad (1973) reported that 1230 ng/g ww of Aroclor 1254 in mink liver tissue was associated with impaired reproductive success. Reduced growth and survival of mink kits were observed in female mink with 2000 ng/g ww Aroclor 1254 in liver tissue (Wren *et al.*, 1987a; 1987b). Olsson and Sandegren (1991a; 1991b) proposed an EC₅₀ of 50 000 ng total PCB/g lw, and Kihlström *et al.* (1992) proposed an EC₅₀ of 65 000 ng/g lw and a NOEL of 9000 ng/g lw for litter size in mink, based on muscle concentrations.

In a reassessment of all reproductive studies of PCBs on mink, the EC₅₀ in adult females for litter size was calculated to be 40 000-60 000 ng total PCB/g lw (approximately 1200 ng total PCB/g ww in muscle) and 2400 ng/g ww in muscle for kit survival (Leonards *et al.*, 1995). Brunström and Halldin (2000) found a LOAEL

of 12 000 ng PCB/g lw (muscle) in dams for decreased kit production and reduced kit body weight gain. The NOEL and LOEL for vitamin A reduction in otter were 4000 and 11 000 ng/g lw in liver for ΣPCB₇ (170 and 460 ng/g ww, respectively) (Murk *et al.*, 1998). Captive harbour seals from the Dutch Wadden Sea, exposed to PCBs via different fish diets, had reduced reproductive success at ΣPCB levels of 25000 ng/g lw in blood (16 ng/g ww) (Reijnders, 1986; Boon *et al.*, 1987).

Similarly, the EC₅₀ for dioxin-like compounds was calculated to be 160 pg TEQ/g ww (5300-8000 pg/g lw) in mink muscle for litter size and 200 pg TEQ/g ww (6600-10 000 pg/g lw) for kit survival (Leonards *et al.*, 1995). In experiments, Heaton (1992) found the LOAEL for mink kit survival to be 490 pg TEQ/g ww in liver. In free-living otter, the NOEL for vitamin A reduction was found to be 2000 pg TEQ/g lw in liver or blood (84 pg TEQ/g ww in liver) based on non- and mono-ortho PCBs (Murk *et al.*, 1998). The EC₉₀ level for vitamin A reduction was 5000 pg TEQ/g lw in liver and blood (210 pg TEQ/g ww in liver).

Experiments found reduced immune function and disruption of vitamin A physiology in captive harbour 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 TEQ levels of approximately 210 pg/g lw, had accumulated in the blubber of these seals, suggesting a threshold for these effects below

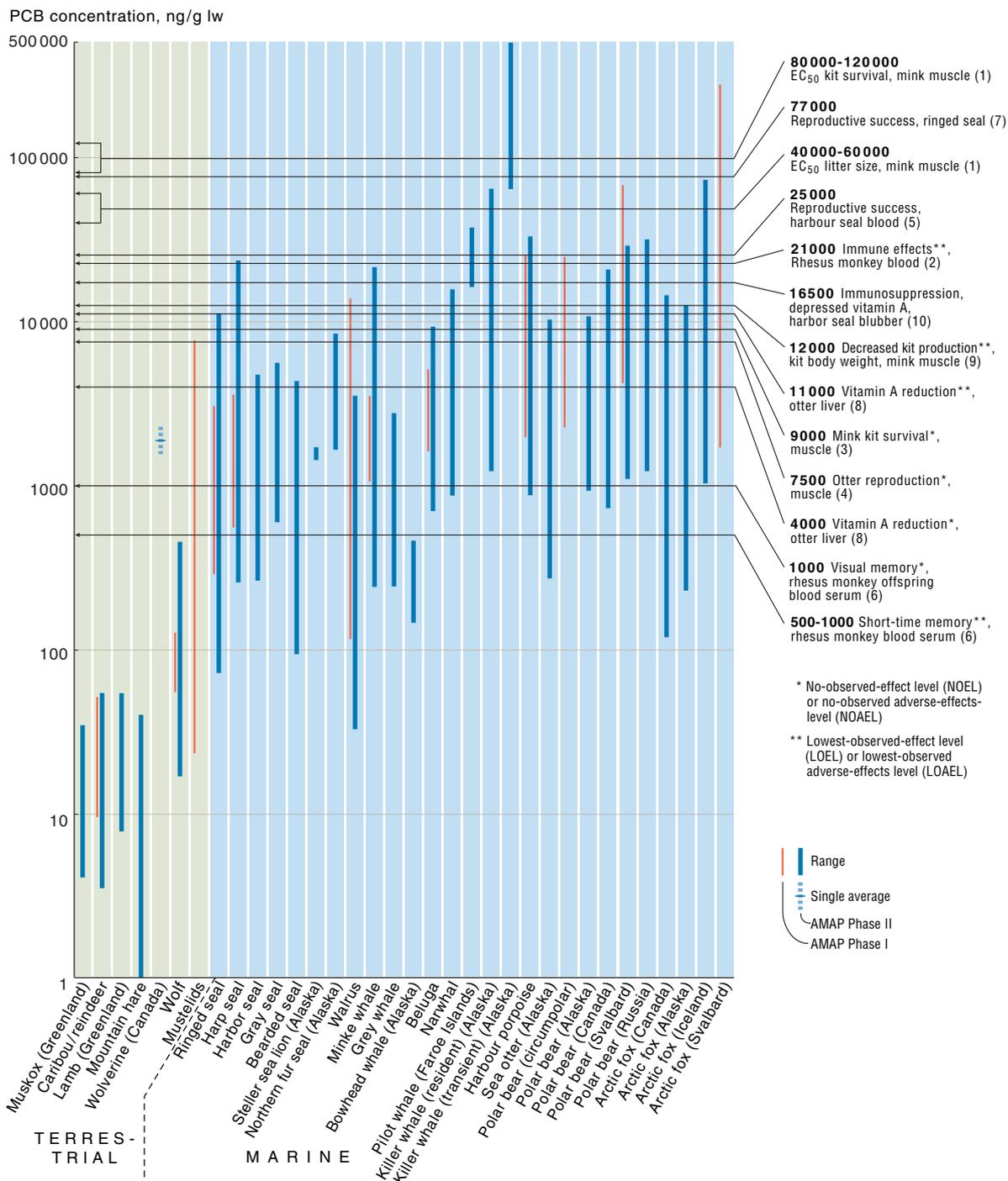


Figure 6-3. Concentrations of ΣPCBs in Arctic animals compared to thresholds for mammalian effects (1. Leonards *et al.*, 1995; 2. Tryphonas, 1994; 3. Kihlström *et al.*, 1992; 4. Roos *et al.*, 2001; 5. Boon *et al.*, 1987; 6. Ahlborg *et al.*, 1992; 7. Helle *et al.*, 1976; 8. Murk *et al.*, 1998; 9. Brunström and Halldin, 2000; 10. de Swart *et al.*, 1994, 1996). Due to numerous limitations in the thresholds data, in quantification of PCBs and problems with extrapolating such data across tissues and species, this comparison should be used with caution. See pages 163-165 for more details. AMAP Phase I data are from de March *et al.*, 1998.

these concentrations. Of the TEQs, PCDD/F accounted for 18 pg TEQ, nPCB for 51 pg TEQ, and mono- and di-ortho PCB for 140 pg TEQ (Ross *et al.*, 1995).

Kannan *et al.* (2000) have reviewed semi-field and field studies on seals, mink, and otter to establish mean threshold tissue concentrations of PCBs and TEQs linked to effects on hepatic vitamin A stores, thyroid hormone concentrations and the immune system. The purpose of this study was to derive threshold concentrations that could be used for estimating risk in marine mammals. For PCBs, the threshold concentrations in liver or blood for these effects were found to be 6600-

11 000 ng/g lw, and the authors suggest the geometric mean of 8700 ng/g lw as a threshold value. For TEQs, the threshold concentrations in liver and blood for these effects were 160-1400 pg/g lw (geometric mean of 520 pg/g lw).

Assessments based on subtle neurobehavioral effects in offspring of rhesus monkeys treated with PCBs and human mothers eating PCB-contaminated fish, have resulted in an estimated LOAEL for effects on short-term memory of 500-1000 ng/g lw, and a NOAEL for effects on visual memory of 1000 ng/g lw in offspring or cord blood serum (Ahlborg *et al.*, 1992). The LOAEL for im-

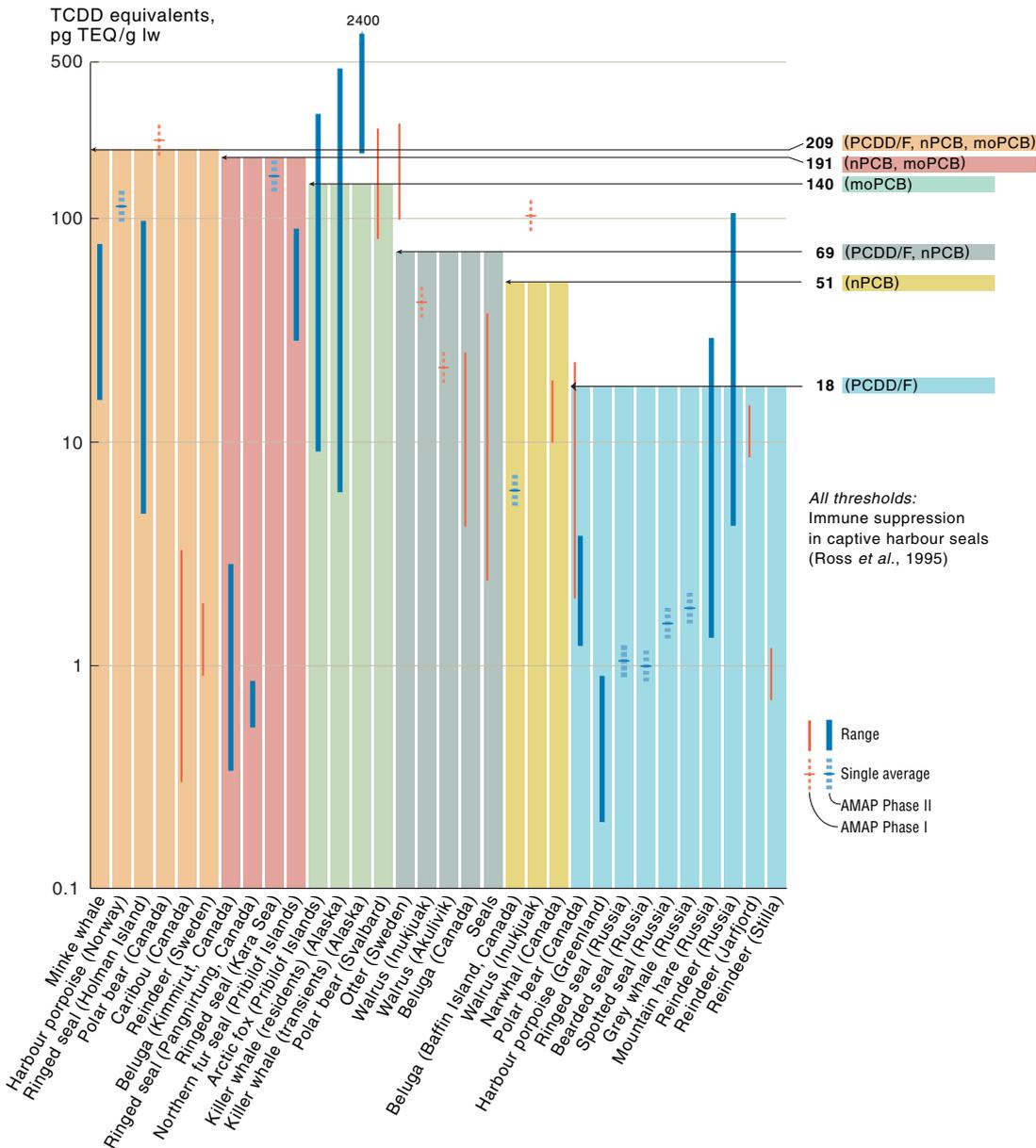


Figure 6-4. Body concentrations of PCDD/Fs and non- and mono-ortho PCBs as TCDD equivalents (pg/g lw) in Arctic mammals compared to thresholds for immunosuppression in harbour seal for the same combination of substances analyzed (Ross *et al.*, 1995). Due to numerous limitations in the thresholds data, in calculating TEQs and problems with extrapolating such data across tissues and species, this comparison should be used with caution. See pages 163-165 for more details. AMAP Phase I data are from de March *et al.*, 1998.

munosuppression is 21 000 ng ΣPCBs/g lw in rhesus monkeys (Tryphonas, 1994). ΣPCB concentrations of 2000-5000 ng/g ww in non-human primate brain tissue were associated with decreased dopamine concentrations (Seegal *et al.*, 1990). Relevant thresholds for ΣPCBs are presented in Figure 6-3 and for TEQs and effects on immunosuppression, in Figure 6-4.

Several effects thresholds have been determined for the effects of TBT or its metabolite, DBT. For *in vitro* inhibition of hepatic cytochrome P450 in Dall's porpoise and Steller sea lion liver, the threshold was found to be 100 μM TBT (29000 ng/g ww) (Kim *et al.*, 1998b). For *in vivo* hepatotoxicity in mice, a threshold concentration of 2600 ng DBT/g ww in liver was found (Ueno *et al.*, 1994). For *in vitro* immunotoxicity to rat thymocytes (Snoei *et al.*, 1986) and rabbit polymorphonuclear neutrophils (PMNs) (Elferink *et al.*, 1986), the threshold concentration was found to be 1.0 μM TBT (290 ng/g ww).

Fish

The LOEL for EROD induction by ΣPCBs in Arctic char is 1000 ng/g ww in liver (Jørgensen *et al.*, 1999). 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, 2002). Mayer *et al.* (1977) found that adult fat-head minnows (*Pimephales promelas*) and channel catfish exposed to toxaphene had no decreases in hydroxyproline levels, but levels in exposed offspring were significantly decreased. 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 are to toxic effects. Mayer *et al.* (1975) exposed brook trout to toxaphene in water and found higher mortality during spawning. A 50% mortality was associated with tox-

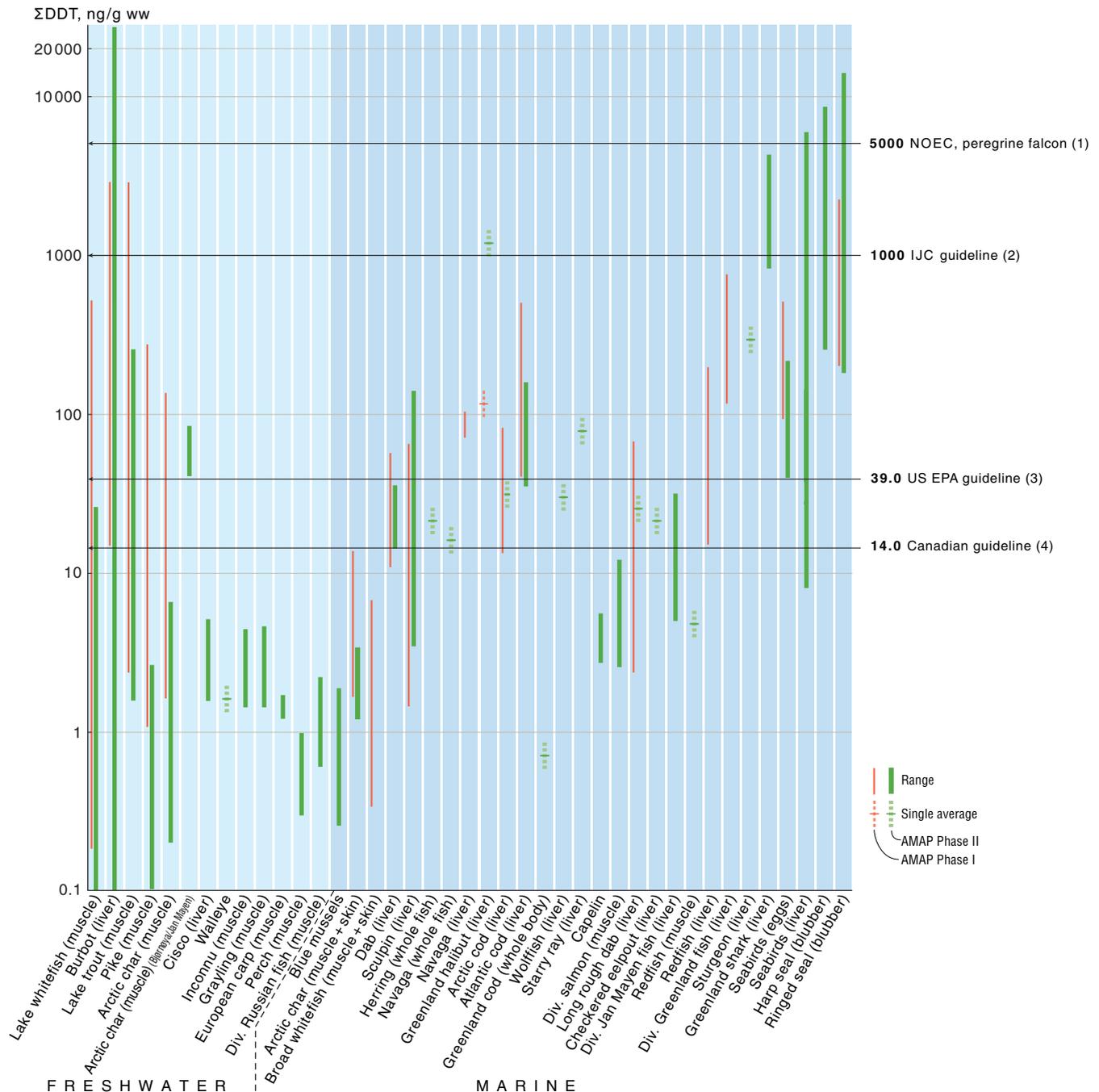


Figure 6-6. ΣDDTs in Arctic animals as food items compared to NOAECs and environmental quality guidelines for protecting fish-eating/aquatic wildlife (1. Baril *et al.*, 1990; 2. USEPA, 1995; 3. De Vault *et al.*, 1995; 4. Environment Canada, 2002). Due to problems with extrapolating data across tissues and species, this comparison should be used with caution. See pages 163-165 for more details. AMAP Phase I data are from de March *et al.*, 1998.

mono-*ortho* PCB TEQs was 11 pg TEQ/g lw in fish, and the LOEC was 29 pg TEQ/g lw in fish (Murdock *et al.*, 1998). For heptachlor epoxide, the dietary NOAEC for adult mink was 50 000 ng/g ww food (Aulerich *et al.*, 1990), and the LOAEC for kit growth was 6250 ng/g ww (Crum *et al.*, 1993). No information on the sensitivity of mink to toxaphene is available; however, a NOAEC of 4000 ng toxaphene/g ww food for thyroid effects has been estimated from studies in rats and dogs (Chu *et al.*, 1986b).

Relevant thresholds and guidelines for intakes of ΣPCBs are given in Figure 6-5, and for ΣDDTs in Figure 6-6. The environmental quality guidelines/objectives have been taken from Table 6-1. These have been developed by various organizations (International Joint Com-

mission, U.S. Environmental Protection Agency, Environment Canada) for the protection of aquatic life and wildlife that consume aquatic biota. The guidelines have been derived using contaminant concentrations in prey such as fish, known thresholds for effects in sensitive fish-eating wildlife species, and research results on bioaccumulation and biomagnification rates of particular substances. These data have then been used to back-calculate fish tissue concentrations that should be without effects in wildlife, and these have been designated as environmental quality guidelines/objectives.

In the discussion that follows, only new information (from 1996 or later) on biological effects studies, or where new concentration data warrant a discussion of the potential for biological effects, will be addressed.

6.1. Terrestrial environment

6.1.1. Terrestrial herbivores

6.1.1.1. Arctic hare/mountain hare

No biological effects studies have been conducted on Arctic or mountain hares. Concentrations of Σ PCBs, Σ DDTs, and other POPs are available from West Greenland, several Russian sites and the Faroe Islands (Annex Table 5). The mean Σ PCB levels in different tissues (muscle, kidney, liver) range from 0.7 to 41 ng/g lw. These concentrations do not exceed any effects thresholds for Σ PCBs (Figure 6-3). TEQ values based on PCDD/F concentrations are also available for Russian mountain hares, and these varied from 1.4 to 30 pg TEQ/g lw (0.03 to 0.6 pg/g ww) in muscle (Annex Table 16). The highest TEQs were found in samples from hares from the Kola Peninsula (Lovozero) and are higher than levels associated with immunosuppressive effects in harbour seal (Figure 6-4).

6.1.1.2. Caribou and reindeer

In the previous AMAP assessment (de March *et al.*, 1998), the concentrations of α -HCH, HCB, Σ PCBs, chlor-danes, Σ DDTs, PCDD/F, and dieldrin in caribou and reindeer from across the Yukon, the Northwest Territories, Norway, Sweden, and Russia were very low. The Russian data were very limited, and concentrations from two consecutive years varied considerably. The POP levels were several orders of magnitude lower than those expected to lead to subtle biological effects. No effects studies have been carried out since the previous assessment.

Levels and intake assessment

A much better data set for Σ PCBs, Σ DDTs and other POPs is available from several sites across Russia (Annex Table 5). The mean Σ PCB levels in different tissues (muscle, kidney, liver) range from 0.2 to 2.4 ng/g ww. Based on an estimated lipid content of 4.5% in liver, Σ PCB concentrations are estimated to be 5.6-53 ng/g lw in the Russian reindeer. Σ PCB levels in reindeer from Finland were even lower (3.5 ng/g lw). These concentrations do not exceed any effects thresholds for Σ PCBs (Figure 6-3). TEQ values based on PCDD/F concentrations are also available for Russian reindeer, and these varied from 0.75 to 20 pg TEQ/g lw (0.05 to 0.98 pg/g ww) in muscle and 4.2 to 105 pg TEQ/g lw (0.24 to 6.5 pg/g ww) in liver (Annex Table 16). The highest TEQs were found in muscle and liver samples from reindeer from the Kola Peninsula (Lovozero) and are higher than levels associated with immunosuppressive effects in harbour seal (Figure 6-4). TEQs in liver from reindeer from Pechora and Taymir Dudinka were also above this threshold, but TEQs in muscle were not.

6.1.1.3. Muskox

No biological effects studies have been conducted on muskox. Concentrations of Σ PCBs, Σ DDTs and other POPs are available from West Greenland (Annex Table 5). The mean Σ PCB levels in different tissues (muscle, kidney, liver, fat) range from 4.1 to 34 ng/g lw. These concentrations do not exceed any effects thresholds for Σ PCBs (Figure 6-3).

6.1.1.4. Lamb

Concentrations of Σ PCBs, Σ DDTs and other POPs are available from West Greenland and Faroe Island lamb (Annex Table 5). The mean Σ PCB levels in different tissues (muscle, kidney, liver) range from 7.9 to 38 ng/g lw and are similar to the levels found in muskox. These concentrations do not exceed any effects thresholds for Σ PCBs (Figure 6-3).

6.1.2. Terrestrial birds

6.1.2.1. Ptarmigan/willow grouse

No biological effects studies have been conducted on ptarmigan or willow grouse. Concentrations of Σ PCBs, Σ DDTs and other POPs are available from West Greenland and several Russian sites (Annex Table 5). Σ PCB concentrations ranged between 0.6 and 18.3 ng/g ww (24-270 ng/g lw). The levels are difficult to assess since the tissues analyzed are liver and muscle, but if the lipid levels in these tissues are assumed to be similar to that of eggs, then the liver and muscle concentrations do not exceed any thresholds for Σ PCBs (Figure 6-1). TEQ values based on PCDD/F concentrations for ptarmigan/willow grouse from several Russian sites varied from 0.02 to 0.2 pg TEQ/g ww (1.2-3.1 pg/g lw) in muscle (Annex Table 16). These are below concentrations expected to result in reproductive effects, assuming similar lipid levels in muscle compared to eggs (Figure 6-2).

6.1.3. Waterfowl

In the previous AMAP assessment (de March *et al.*, 1998), several waterfowl groups and species (molluscivores, piscivores, semipalmated plover, pintail, and oldsquaw) from Canada had levels of Σ PCBs above NOELs and LOELs for reproductive effects. No effects studies have been carried out since the previous assessment.

Levels and intake assessment

Σ PCB and TEQ concentrations based on PCDD/Fs are available for several species of waterfowl from several Russian sites (Annex Tables 11 and 16). For Σ PCBs, results are available for liver and muscle from oldsquaw (*Clangula hyemalis*), goldeneye (*Bucephala clangula*), pintail (*Anas acuta*), wigeon (*Anas penelope*), green-winged teal (*Anas crecca*), greater white-footed goose (*Anas albifrons*), greater scaup (*Aythya marila*), scoters (*Melanitta* spp.), and bean goose (*Anser fabalis*). Σ PCB concentrations in liver range from 0.001 to 0.042 μ g/g ww, and for muscle from 0.001 to 0.017 μ g/g ww. The levels are difficult to assess since the tissues analyzed are liver and muscle, but if the lipid levels in these tissues are assumed to be similar to that of eggs, then the liver and muscle concentrations do not exceed any thresholds for Σ PCBs (Figure 6-1). For TEQs, results are available for oldsquaw, goldeneye, and pintail. TEQs range from 0.02 to 0.52 pg TEQ/g ww (0.5 to 6.4 pg/g lw) in muscle. These are below concentrations expected to result in reproductive effects, assuming similar lipid levels in muscle compared to eggs (Figure 6-2).

6.1.4. Birds of prey

6.1.4.1. Peregrine falcon

In the previous AMAP assessment, it was concluded that Arctic populations of Canadian peregrine falcons were still at risk for reproductive effects from Σ DDTs and Σ PCBs in their eggs and in their food. For Fennoscandian peregrines, only high levels of Σ PCBs in eggs exceeded NOELs and LOELs for reproductive effects in other bird species. Peregrine falcons from the Kola Peninsula, Russia, had Σ PCB levels in eggs that exceeded most NOEL and LOEL levels for reproductive effects in other bird species. TEQs based on PCDD/F and dioxin-like PCB concentrations exceeded most NOAELs and LOAELs for reproductive effects in other bird species and several LD₅₀ values (the dose that causes the death of 50% of a group of test animals). No new effects or levels studies have been carried out in these populations since the previous assessment.

Reproductive effects

Peregrine falcon populations in North America and Europe declined drastically in most parts of the species' range in the 1960s and 1970s, primarily due to the effects of OC pesticides, especially DDT and dieldrin. Subspecies found in Alaska are the Arctic peregrine, which nests in northern tundra, the American peregrine, which nests in the forested interior and the Peale's peregrine, which nests along the southern coast from the Aleutian Islands to southeast Alaska.

Persistent OC contaminants were measured in American and Arctic peregrine falcon eggs from Alaska from 1979 to 1995 (Ambrose *et al.*, 2000). Dieldrin, *p,p'*-DDE, heptachlor epoxide, oxychlorodane, and total Arochlor PCBs were consistently measured and detected, and were tested multivariately for relationships with time and productivity. Eggshell thickness was significantly negatively correlated with *p,p'*-DDE concentrations, and mean eggshell thicknesses in 1991-1995 were 12.0 and 10.6% thinner in American and Arctic subspecies, respectively, than pre-DDT era peregrine falcon eggs from Alaska (Anderson and Hickey, 1972).

Significant multivariate analyses indicated that some POPs were associated with decreased reproduction. Differences were greatest early in the study, but over the entire time span, dieldrin, oxychlorodane and total PCBs were significantly greater in eggs from unsuccessful nests (no young at expected age of 1-3 wk) compared to successful nests (≥ 1 young) for the American subspecies. There were no significant differences in *p,p'*-DDE or heptachlor epoxide, and no significant differences for any of the five contaminants in the Arctic subspecies. Eggs from unsuccessful nests also had higher mercury concentrations, which may also have affected reproduction (AMAP, 2004).

Levels and intake assessment

Concentrations of Σ PCBs in Alaskan peregrine eggs collected in 1991-1995 ranged between 0.4 and 8.5 $\mu\text{g/g}$ ww (geometric mean of 1.6 $\mu\text{g/g}$ ww) for the American subspecies and between 0.6 and 6.0 $\mu\text{g/g}$ ww (geometric mean of 1.3 $\mu\text{g/g}$ ww) for the Arctic subspecies (Ambrose *et al.*, 2000) (Annex Table 5). These mean Σ PCB levels are below thresholds for effects, but the maximum

Σ PCB levels exceed most NOELs and LOELs found for hatching success, egg mortality and deformities in a number of wild bird species (Figure 6-1).

The *p,p'*-DDE concentrations in the American subspecies ranged between 0.48 and 14.1 $\mu\text{g/g}$ ww (geometric mean of 3.4 $\mu\text{g/g}$ ww), and in the Arctic subspecies between 1.2 and 13.3 $\mu\text{g/g}$ ww (geometric mean of 3.0 $\mu\text{g/g}$ ww), indicating that some individuals have concentrations just below the critical threshold for reproductive failure (15-20 $\mu\text{g/g}$ ww).

Geometric mean dieldrin concentrations were below the adverse effects threshold of 1-4 $\mu\text{g/g}$ ww for both subspecies and in 1991-1995, no individual eggs exceeded these levels. Geometric mean heptachlor epoxide concentrations never exceeded the adverse effects threshold of 1.5 $\mu\text{g/g}$ ww.

Σ PCB concentrations in Norwegian peregrine falcon eggs collected in 1991-1997 ranged from 2.3 to 25.5 $\mu\text{g/g}$ ww (mean of 10.6 $\mu\text{g/g}$ ww) (Annex Table 5). These concentrations exceed most thresholds for reproductive effects in other bird species (Figure 6-1).

6.1.4.2. Merlin

In the previous AMAP assessment, Fennoscandian merlin were suffering from 10% shell thinning, but the population was recovering. Merlin seemed to be less sensitive to the effects of DDT, since Σ DDT levels exceeded those expected to cause reproductive failure in peregrine falcons (20% eggshell thinning results in crushed shells). Σ PCB levels were in the range of some NOELs and LOELs for effects in white leghorn chicken, an extremely sensitive species. No effects studies have been carried out since the previous assessment.

Levels and intake assessment

Σ PCB concentrations in Norwegian merlin eggs collected from 1991 to 1997 ranged between 1.4 and 2.0 $\mu\text{g/g}$ ww (Annex Table 5). These concentrations are below thresholds for reproductive effects in other bird species (Figure 6-1).

6.1.4.3. White-tailed sea eagle

In the previous AMAP assessment, effects studies showed improvement in reproduction and population numbers for white-tailed sea eagles in Sweden and Norway, with only a small amount of eggshell thinning (2-5%). However, levels of TEQs in Swedish eagles, measured as PCDD/F and several dioxin-like PCBs, exceeded most NOAELs and LOAELs for reproductive effects in other bird species and approached the LD₅₀ for white leghorn chicken embryo. Σ PCB levels in the northerly Norwegian eagles exceeded or overlapped most NOELs and LOELs for subtle reproductive effects in other bird species. Σ PCB and TEQ levels in one Russian white-tailed sea eagle egg from the Kola Peninsula exceeded some thresholds for effects in other bird species. The intake assessment for Norwegian sea eagles, which prey mainly on deep-water marine fish species, concluded that dietary intakes of Σ DDTs and Σ PCBs may be high enough to lead to effects. No effects studies have been carried out since the previous assessment.

Levels and intake assessment

ΣPCB concentrations in Norwegian white-tailed sea eagle eggs collected in 1991-1997 ranged from 3.1 to 13.0 µg/g ww (mean of 9.0 µg/g ww) (Annex Table 5). These concentrations exceed most thresholds for reproductive effects in other bird species (Figure 6·1).

6.1.4.4. Bald eagle*Reproductive effects*

The relationship between OC concentrations, diet, and productivity was studied in nesting bald eagles from Adak, Tanaga, Amchitka, and Kiska Islands in the Aleutian archipelago (Anthony *et al.*, 1999). Productivity on Kiska Island (the most westerly island) was depressed (lower breeding success and low clutch sizes), averaging 0.67 young per occupied site. This was associated with higher DDE and OC pesticide concentrations. Mercury concentrations were also higher. Productivity on the other three islands was comparable to healthy populations in other parts of the U.S. (0.88-1.24 young per occupied site), and DDE concentrations were lower. Eggshell thickness was not significantly different among the islands and was not correlated to DDE concentrations, but was significantly negatively correlated to PCB concentrations. PCB concentrations were not correlated to DDE concentrations, but instead, reflected previous military activity.

Levels and intake assessment

Mean ΣPCB concentrations in bald eagle eggs from the above four islands in the Aleutian chain were between 0.7 and 2.1 µg/g ww (range 0.1-9.9 µg/g ww) (Annex Table 11). The mean concentrations are just below the NOEL for hatching success in Forster's tern (Figure 6·1). However, the maximum ΣPCB concentrations at some sites exceed most of the thresholds for reproductive effects in other bird species including the LOAEL for egg mortality in bald eagles. The mean *p,p'*-DDE concentrations were between 0.7 and 2.75 µg/g ww (range 0.3 to 4.1 µg/g ww) with the highest concentrations on Kiska Island (range 1.5-4.1 µg/g ww). These concentrations are within the range known to cause reproductive impairment in bald eagles (Wiemeyer *et al.*, 1984).

6.1.4.5. Golden eagle*Levels and intake assessment*

ΣPCB concentrations in Norwegian golden eagle eggs collected in 1991-1997 ranged from 0.26 to 4.7 µg/g ww (mean of 1.4 µg/g ww) (Annex Table 5). These concentrations exceed a few thresholds for reproductive effects in other bird species (Figure 6·1).

6.1.4.6. Gyrfalcon

In the previous AMAP assessment, DDE and ΣPCB concentrations in Canadian gyrfalcon eggs were several orders of magnitude lower than those considered to cause reproductive effects. Based on dietary LOECs, OC concentrations in gyrfalcon prey from Canada were also below those expected to cause effects. No effects studies have been carried out since the previous assessment.

Levels and intake assessment

ΣPCB concentrations in Norwegian gyrfalcon eggs collected in 1991-1997 ranged from 0.29 to 5.3 µg/g ww (mean of 2.0 µg/g ww) (Annex Table 5). These concentrations exceed a few thresholds for reproductive effects in other bird species (Figure 6·1).

6.1.5. Carnivores**6.1.5.1. Mink**

In the previous AMAP assessment, Canadian mink from Quebec (Grand Baleine) and Fort Providence, NWT, had ΣPCB levels that exceeded the NOAEL and LOAEL for effects on subtle neurobehavioral effects in offspring of rhesus monkeys, and the Quebec mink levels were close to the NOEL for mink kit survival. The intake assessment for Canadian mink indicated that dietary intakes of dioxin-like compounds and ΣPCBs from some species of fish at some lakes may be high enough to lead to effects. No effects studies have been carried out since the previous assessment. No new information on current POP levels is available.

6.1.5.2. Otter

In the previous assessment, otter populations in northern Sweden were beginning to recover in the 1990s after having existed only as isolated groups (de March *et al.*, 1998). The ΣPCB levels were considered to be at the NOEL for otter reproduction. These levels exceeded the NOAEL and LOAEL for effects on subtle neurobehavioral effects in offspring of rhesus monkeys, however. The intake assessment for Swedish otter indicated that dietary intakes of dioxin-like compounds from some species of fish at some lakes may be high enough to lead to effects. No effects studies have been carried out since the previous assessment. No new information on current POP levels is available.

6.1.5.3. Wolverine

No biological effects studies have been conducted on wolverines. Concentrations of ΣPCBs, ΣDDTs, and other POPs are available from the Northwest Territories, Canada (Annex Table 5). The mean ΣPCB levels in liver are 1960 ng/g lw. These concentrations exceed the NOAEL and LOAEL for subtle neurobehavioral effects if wolverines are as sensitive as offspring of rhesus monkeys and humans (Figure 6·3).

6.1.5.4. Wolf

In the previous AMAP assessment, concentrations of ΣPCB levels in wolves from the Canadian Arctic were several orders of magnitude lower than those expected to result in effects on reproduction. The levels in caribou/reindeer were also much lower than the dietary concentrations expected to cause such effects. No biological effects studies have been conducted on wolf since the previous assessment.

Levels and intake assessment

Mean concentrations of ΣPCBs in wolves (liver) from the Canadian Yukon, and median concentrations from northwest Russia were between 17 ng/g lw and 450 ng/g

lw (Gamberg and Braune, 1999; Shore *et al.*, 2001). These concentrations are below those expected to result in effects (Figure 6-3). The dietary assessment is the same as for the previous AMAP assessment.

6.2. Freshwater environment

6.2.1. Fish

In the previous AMAP assessment, freshwater fish did not exceed effect levels for toxaphene, but fish from some lakes had levels close to the threshold for bone development effects and mortality during spawning, indicating that some individuals in Canada may exceed these levels. Besides Lake Laberge, possibly affected locations included several lakes in the Yukon (Bennett, Tagish, and Marsh Lakes), Atlin Lake in northern British Columbia, and Great Slave Lake in the Northwest Territories.

Cytochrome P450 activities

Arctic char from two lakes on Bjørnøya, Norway, have been studied for OC contamination and biomagnification: Ellasjøen, which is strongly affected by seabird guano from nearby colonies and has high Σ PCB levels and Øyangen, an oligotrophic lake unaffected by the bird colonies and with lower Σ PCB levels. Arctic char have also been studied for liver enzyme activity including testosterone hydroxylation enzymes. When comparing char from both lakes, correlations were seen between CYP enzyme activities and high Σ PCB levels. Levels of testosterone hydroxylation enzymes were similar at both lakes and not correlated to Σ PCB levels (Skotvold *et al.*, 1999).

Levels and intake assessment

Toxaphene levels in some burbot from the East Arm of Great Slave Lake, NWT and Lake Laberge and Kusawa Lake, Yukon (Canada), exceed levels associated with effects on bone development in channel catfish (600 ng/g ww) (Annex Table 7). Toxaphene levels in freshwater fish from all other sites measured in the Arctic are below threshold levels for effects. Σ PCB concentrations in burbot liver from Fairbanks and Yukon Flats (Alaska), Lake Laberge (Canada) and in some Arctic char from Bjørnøya (Ellasjøen) are close to or exceed the LOEL for induction of EROD in Arctic char (Annex Table 7). Σ PCB levels in other Arctic freshwater fish do not exceed this threshold.

6.3. Marine environment

6.3.1. Invertebrates

In the previous AMAP assessment, imposex, the induction of male sex characteristics in females, was found in females of the common whelk in Kongsfjorden, Svalbard, dogwhelks in Norway and Iceland, and a marine snail (*Nucella lima*) in Alaska exposed in harbors with significant boat mooring.

Reproduction

There have been only modest studies on TBT and its effects on gastropods recently in Arctic or subarctic waters. In Norway, TBT has been measured, and imposex has been evaluated in dogwhelks in 41 populations sam-

pled in 1993-1995 along the Norwegian coast (Følsvik *et al.*, 1999). Some degree of imposex occurred in almost all populations of dogwhelks studied, except in four from northern Norway. The concentration of organotin compounds in the gastropods from the unaffected populations was below the detection limit (7 ng Sn/g dw). The concentration of TBT in dogwhelks from affected populations was in the range of 48-1096 ng Sn/g dw. A positive relationship between the concentration of TBT in dogwhelks and the degree of imposex was found.

In Iceland, the status of imposex was evaluated in the dogwhelk in 1998 and compared to the levels of imposex evaluated in 1992/1993 (Svavarsson, 2000). The level of imposex has decreased considerably since 1992/1993, two years after implementation of restrictions on the use of TBT-based anti-fouling paint. VDSI and RPSI levels (see definition in Table 6-2) have declined considerably, both near large and small harbors. The impact area of a large harbor complex has decreased considerably, while lesser changes were seen in the impact areas near smaller harbors. This study shows that, at least in Icelandic waters, the situation has improved considerably.

Imposex has been observed in the subtidal common whelk near Svalbard and in Icelandic waters (Brick and Bolte, 1994; Svavarsson *et al.*, 2001). The levels observed in Breiðafjörður, southwestern Iceland, were low, and only 26.4% of the females observed had imposex, and the penises were small (0.7 mm \pm 0.6 mm SD) (Svavarsson *et al.*, 2001). Near harbors (Reykjavík and Straumsvík harbors), the frequency was higher, and the female penises were considerably longer (mean 3.8 and 6.8 mm, respectively).

Studies of the occurrence of imposex in dogwhelks in the Faroe Islands were done in 1996 and in 2001 (FEA, 2002). The results do not indicate any major changes in the five-year period from the first to the second sampling period (Table 6-2). The occurrence of imposex in the Faroe Islands is still widespread, but there are sites where the phenomenon is hardly seen.

Table 6-2. Imposex in dogwhelks, *Nucella lapillus*, in the Faroe Islands.

Station	VDSI ^a	VDSI	RPSI ^b	RPSI
	Feb.-Mar. 1996	March 2001	Feb.-Mar. 1996	March 2001
<i>Tórshavn</i>				
Argir	4.4	4.0	32	48
Kirkjubour	0.1	0.6	0	0
Vestmanna	4.3	4.1	13	17
<i>Trongisvágsfjörður</i>				
Kolatoftr	4.4		31	
Hvítanæs (April '01)		4.2		36
<i>Klaksvík</i>				
Kunoy	4.1	4.0	13	10
Skálafjörður	4.0		53	
<i>Nólsoy</i>				
Víkin ^c		4.1		20
Kirkjutangi	4.3	1.3 ^d	8	0 ^d

^a VDSI = Vas deferens sequence index. The VDSI scale ranges from 0 (normal) to 6 (last stages of development of vas deferens and penis in females leading to sterility).

^b RPSI = Relative penis size index (female mean penis length³/male mean penis length³)·100.

^c Too few females in 1996 sample for analyses.

^d Imposex in all stages were found, a sample with unusually large individual variation.

Source: FEA (2002).

In Greenland, TBT concentrations in blue mussels and the occurrence of imposex in several whelk species was studied in and around five harbors (Nuuk, Qaqortoq, Manitoq, Qeqertarsuak, and Uummannaq), and at an uninhabited reference site (Strand and Asmund, 2003). Some degree of imposex was found in several neogastropod species (*Buccinum* spp.) in all the harbors but not outside the harbors or at the reference site. Effects of contaminants on blue mussels have been evaluated at sites in southwestern Icelandic waters by studying Scope for Growth (SFG) (Widdows, 1998). This method allows evaluation of combined effects of contaminants on the physiology of the mussel. The studies show that SFG is low near Reykjavik harbor amongst mussels with high levels of TBT and PAHs, while among mussels living near small harbors (low TBT levels; intermediate PAH levels), the SFG is fairly high (Halldórsson, 2002).

6.3.2. Fish

In the previous AMAP assessment, only Greenland halibut (turbot) from the Canadian Arctic had toxaphene levels close to the effect levels for bone development and mortality during spawning. No effects studies have been carried out since the previous assessment.

Levels and intake assessment

Mean Σ PCB levels in marine and anadromous fish are below the LOEL for EROD induction in Arctic char except for Greenland shark (liver) from Cumberland Sound and for Greenland halibut (liver) from West Greenland (Annex Table 10). Based on mean toxaphene levels, no anadromous or marine fish exceed effects levels.

6.3.3. Seabirds

In the previous AMAP assessment, Σ PCB levels in seabirds from the Canadian and Norwegian Arctic approached or exceeded reproductive NOELs and LOELs at the low range of the scale compared to peregrine falcons. Mean Σ PCB levels in eider eggs were below those expected to result in reproductive effects, levels in shag (a species of cormorant) and fulmar were at or exceeded the LOAEL for embryo deformities and the lower NOEL for hatching success in white leghorn chicken, an extremely sensitive species. In addition, mean Σ PCB levels in puffin, murre, common guillemot, black guillemot, and kittiwake exceeded the upper NOEL for hatching success in white leghorn chicken.

Mean Σ PCB levels in glaucous gull and herring gull eggs were somewhat higher and approached or exceeded the LOEL for hatching success in white leghorn chicken and the NOEL for hatching success found for Forster's tern. None of the Arctic seabirds studied appear to have Σ PCB levels high enough to be associated with thresholds for egg mortality. Mean Σ DDT, HCH, dieldrin, and HCB levels in all seabirds studied were several orders of magnitude below those expected to cause effects.

Based on dietary LOECs for Σ PCBs, HCB, DDE, and dieldrin in peregrine falcon prey, levels in anadromous and marine fish were several orders of magnitude below those expected to cause effects in falcons. Mean dieldrin levels in Arctic anadromous and marine fish from Cana-

dian, Norwegian, Greenlandic, Icelandic, and Russian waters did not exceed any of the guidelines for protecting fish-eating wildlife. However, Σ PCB and Σ DDT levels in a range of fish species from many Arctic sites exceeded some, and in a few cases for Σ PCBs, all of the environmental quality guidelines, implying that dietary intakes of Σ DDTs and Σ PCBs in fish-eating seabirds might be high enough to lead to effects if seabirds prey on these fish species. A few fish species in Canada and Norway had PCDD/F or nPCB levels that exceeded the guidelines.

Based on POP levels found in eggs from guillemots, puffins, cormorants, and fulmars from the Canadian and Norwegian Arctic, dieldrin, HCB, Σ DDT, and Σ PCB levels were below dietary LOECs for reproductive effects in peregrine falcon. Mean dieldrin levels in these seabird species' eggs also did not exceed any environmental guidelines for protecting aquatic wildlife. However, Σ DDT levels in these species from these sites exceeded Canadian Environmental Quality Guidelines for protection of animals that consume aquatic biota and U.S. EPA guideline values for assessment of hazards to fish-eating wildlife. Σ PCB levels exceeded all guideline levels, implying that dietary intakes of Σ DDTs and Σ PCBs in some seabirds that prey on seabird eggs and chicks might be high enough to lead to effects.

6.3.3.1. Eiders

In the previous AMAP assessment, common and Steller's eiders (*Polysticta stelleri*) from Canada, Russia, and Norway had Σ PCB levels below effects thresholds. A recent study in spectacled eiders (*Somateria fischeri*) from western Alaska found no biological effects that could be attributed to the low concentrations of POPs present (Trust *et al.*, 2000).

Levels and intake assessment

Mean Σ PCB levels in common eider muscle from Greenland, Iceland, and western Russia (Chukotka), and King eider (*Somateria spectabilis*) from Greenland, are below those expected to result in reproductive effects, assuming similar lipid levels in muscle compared to eggs (Figure 6-1).

Based on dietary LOECs for Σ PCBs (5000 ng/g ww), HCB (100 000 ng/g ww), DDE (5000 ng/g ww), and dieldrin (100 ng/g ww) in peregrine falcon prey, levels in blue mussels and anadromous and marine fish are several orders of magnitude below those expected to cause effects in animals that eat them (Annex Table 10). Mean dieldrin levels in blue mussels and Arctic anadromous and marine fish from Canadian, Norwegian, and Greenlandic waters (Annex Table 10) do not exceed any of the guidelines for protecting aquatic biota-eating wildlife (Table 6.1). Σ PCB and Σ DDT levels in a range of fish species from many Arctic sites exceed some, and in a few cases for Σ PCBs, all of the environmental quality guidelines for protecting aquatic biota consumers/fish-eating wildlife given in Table 6.1 (Figures 6.5 and 6.6). Toxaphene levels in several fish species also exceed the Canadian Tissue Residue Guideline for protecting wildlife consumers of aquatic biota (Table 6.1). This implies that dietary intakes of Σ DDTs, Σ PCBs, and toxaphene in some fish-eating seabirds may be high enough to lead to

effects if seabirds prey on these species. No data are available for levels of dioxin-like compounds in anadromous and marine fish so the dietary intake of TEQs cannot be assessed.

For ΣDDTs, the fish species that exceed both Canadian and U.S. EPA guidelines are spotted wolffish (liver) from southwest Greenland; Atlantic cod (liver) from Greenland, Iceland, and the Faroe Islands; short-horn sculpin (liver) from northeast and southwest Greenland and the Faroe Islands; starry ray (liver) from Greenland; Greenland shark (liver) from Canada; sturgeon (liver) from the Kara Sea; and, Greenland halibut (liver) from Greenland. The fish species that exceeded the International Joint Commission objectives for protection of aquatic life and wildlife of 1000 ng/g ww (Table 6.1) were Greenland shark (liver) from Canada and Greenland halibut (liver) from West Greenland (Figure 6.6).

For ΣPCBs, the species that exceed Canadian guidelines for avian predators are polar cod (liver) from Jan Mayen (Norway); Atlantic cod (liver) from Greenland and Iceland; shorthorn sculpin (liver) from Greenland and the Faroe Islands; starry ray (liver) from Greenland; Greenland shark (liver) from Canada; sturgeon (liver) from the Kara Sea; Greenland halibut (liver) from Greenland; Arctic char (muscle) from Nain, Canada, and a number of other marine species (liver) from around Jan Mayen (coalfish, long rough dab, checkered eelpout, and Atlantic poacher). The fish species that exceed International Joint Commission objectives are Atlantic cod (liver) from Greenland; shorthorn sculpin (liver) from southwest Greenland and the Faroe Islands; sturgeon (liver) from the Kara Sea; Greenland shark (liver) from Canada; Greenland halibut (liver) from Greenland; and, long rough dab (liver) and checkered eelpout from Jan Mayen. Only liver from sturgeon from the Kara Sea, Greenland halibut and Atlantic cod from Greenland, Greenland shark from Canada, and long rough dab and checkered eelpout from Jan Mayen exceed the U.S. EPA guideline values for assessment of hazards to fish-eating wildlife (Figure 6.5).

For toxaphene, the species that exceed the Canadian Tissue Residue Guideline are spotted wolffish (liver) from southwest Greenland; polar cod (liver) from Barrow, Alaska; Atlantic cod (liver) from southwest Greenland and Iceland; herring (whole fish) from the White Sea, Russia; shorthorn sculpin (liver) from southwest Greenland and the Faroe Islands; dab (liver) from Iceland; starry ray (liver) from southwest Greenland and Arctic char (fillet) from Barrow (Annex Table 10).

6.3.3.2. Grey heron, shag

Reproductive effects

A significant negative correlation was found between wet weight ΣPCB concentrations in yolk and egg volume, yolk mass, and hatchling mass in one-day-old shag hatchlings from the central Norwegian coast (Murvoll *et al.*, 1999).

Retinol effects

Grey heron hatchlings from two rookeries in Norway, one at Frøya on the central Norwegian coast and one at Finnjordøy on the north coast, were found to have sim-

ilar ΣPCB and TEQ (based on eight mono-*ortho* CBs) levels. No effects of either PCBs or TEQs on plasma retinol levels were found at either site (Jenssen *et al.*, 2001).

A borderline significant positive correlation was found between lipid weight ΣPCB concentrations in yolk and plasma retinol levels in one-day-old shag hatchlings from the central Norwegian coast (Murvoll *et al.*, 1999).

Levels and intake assessment

Mean ΣPCB concentrations in Norwegian shag hatchlings were 1.2 µg/g ww and for grey heron hatchling yolk sacs, 2.1-2.5 µg/g ww (range 0.57-4.9 µg/g ww) (Annex Table 11). Grey heron levels exceed the NOEL for hatching success in Forster's tern and several LOELs for egg mortality and embryo deformities in other bird species (Figure 6.1). For grey heron, the mean TEQ concentrations based on eight mono-*ortho* PCBs were 50-79 pg/g ww (range 21-179 pg TEQ/g ww) (1020-1170 pg/g lw), exceeding some NOAELs and LOAELs for reproductive effects in other bird species (Figure 6.2) (Annex Table 16).

The assessment for dietary intake of dieldrin, HCB, PCB, DDT, and toxaphene is the same as given for eiders (Section 6.3.3.1).

6.3.3.3. Alcids

Cytochrome P450 activities/retinol effects

Three groups of black guillemot nestlings in Saglek Bay, Canada, with low, medium, and high PCB exposures, were studied for several liver biomarkers (Kuzyk *et al.*, 2003). This bay has PCB-contaminated marine sediments due to a former military site. There was a significant dose-dependent increase in liver size, EROD activity up to a threshold of 100 ng/g ww liver, and reduced liver retinol and retinyl palmitate (females only) with increasing liver PCB concentrations. Many effects were more pronounced in female nestlings than in males.

Levels and intake assessment

In many cases, tissues other than eggs (muscle, liver and fat) have been analyzed in alcids (little auk, thick-billed murres, black guillemots and common guillemots) at various sites. This makes comparisons difficult because concentrations in other tissues may not be reflective of those found in the eggs (Braune and Norstrom, 1989; Drouillard and Norstrom, 2001). If only ΣPCB concentrations in eggs are used, values range from 0.087 to 0.27 µg/g ww for thick-billed murres from the Pribilof Islands, Alaska (Bering Sea), Lancaster Sound, Canada and Bjørnøya, and 0.11 to 0.29 µg/g ww in black guillemots from Qeqertarsuaq and Ittoqqortoormiit, Greenland and Bjørnøya (Annex Table 11). The ΣPCB concentrations based on eggs are below thresholds for effects in other fish-eating birds (Figure 6.1).

ΣPCB concentrations in liver of black guillemots from Baffin Bay and two sites at Saglek Bay (Canada), Qeqertarsuaq and Ittoqqortoormiit (Greenland), the Faroe Islands, and Svalbard are similar to or lower than levels seen in eggs from Greenland and Bjørnøya (eggs have lipid contents of approximately 10%, liver has a lipid content of 4-6%). However, ΣPCB concentrations in black guillemot livers from Jan Mayen (1.2 µg/g ww)

and at the more contaminated site at Saglek Bay (0.17–6.5 µg/g ww) are higher, and indicate that these populations probably have elevated levels in eggs as well. The levels at Jan Mayen are probably below thresholds for effects, but at Saglek Bay, they indicate that some individuals probably have levels that may be above thresholds for effects (Figure 6-1). ΣPCB concentrations in liver of thick-billed murres from Baffin Bay (Canada), Nuuk (Greenland), and Svalbard are lower than in eggs from Alaska and Lancaster Sound (Canada), indicating that eggs from these populations probably have levels below thresholds for effects.

TEQ concentrations based on PCDDs, PCDFs and non-*ortho* PCBs were 24 pg/g ww (719 pg/g lw) in liver and 26 pg/g ww in eggs for Canadian thick-billed murres collected in 1993 (Braune and Simon, 2002), 25 pg/g ww (640 pg/g lw) in black guillemot liver from Saglek Bay collected in 1999 and 9.8 pg/g ww (66 pg/g lw) for common guillemot eggs collected from the Faroe Islands in 2000 (Annex Table 16). The value for Canadian thick-billed murre eggs exceeds the LOAEL for reproductive effects in wood duck, a sensitive species, but for common guillemots, the levels are below thresholds (Figure 6-2). Lipid weight TEQ levels in liver in these species are well below the LOEL for induction of CYP1A in common terns (25000 pg TEQ/g lw).

The assessment for dietary intake of dieldrin, HCB, PCB, DDT, and toxaphene is the same as given for eiders (Section 6.3.3.1).

6.3.3.4. Gulls

At Bjørnøya, Bourne and Bogan (1972) were the first to observe aberrant behaviors in glaucous gulls with high PCB levels. In the Svalbard archipelago, sick and dying glaucous gulls were found to have high PCB levels (Daelemans, 1994; Gabrielsen *et al.*, 1995).

Reproductive effects

Recently, a study was carried out to examine whether behavior at the nesting stage was negatively influenced by PCBs. Individual patterns of incubation and nest-site attentiveness were studied in relation to PCB concentrations in the blood of twenty-seven glaucous gulls in two breeding areas. PCB concentrations in the blood ranged from 52 ng/g ww to 1079 ng/g ww. There were significant differences between the two breeding areas, and females had significantly lower concentrations than males (Bustnes *et al.*, 2001).

Plasma PCB concentrations were positively related to the proportion of time the birds were absent from the nest, both overall and when not incubating, and to the number of absences, when controlling for possible confounding variables (area, sex, body condition, and others). Increased absence from the nest site in individual glaucous gulls with high blood concentrations of OCs suggests that they need more time to gather food because of either endocrine disruption or neurological disorders. This probably led to increased energetic costs during incubation and reduced reproductive output (Bustnes *et al.*, 2001).

To examine the effects of OCs on individual fitness, blood concentrations of various compounds were compared to reproductive parameters and adult survival in

individuals with different levels, controlling for a set of potential confounding variables (body condition, breeding areas, laying dates and others) (Bustnes *et al.*, 2003). Females with high circulating levels of OCs, including HCB, oxychlorane, DDE, and PCBs, were more likely to have non-viable eggs in the nest than females with low PCB levels. Moreover, the body condition at hatching was poorer for the first chick in the clutch in females with high concentrations of all persistent OCs. For the second chick in the clutch, only HCB, β-HCH, and CB28 showed such a negative relationship. Apart from a negative association between concentration of some OCs and laying date, no other reproductive parameters, such as clutch size, egg size, incubation time, nest predation or early chick survival, showed any association with OCs. Hence, reproduction seems moderately affected by OCs. Adult survival was significantly negatively related to four different OCs: HCB, DDE, CB153, and, in particular, oxychlorane. In long-lived birds, such as glaucous gulls, adult survival probability is the key parameter to which the population growth rate is most sensitive, suggesting that OCs may have considerable effect on growth in glaucous gull populations. However, the effects on specific populations will depend on the proportion of the population exposed to high intake of contaminants via the diet.

It was also demonstrated that glaucous gulls with high levels of various persistent OCs had asymmetry in wing feathers (i.e. feathers on the left and right wing were of different length). Feather asymmetry is a well-known indication of developmental stress, suggesting that OCs are an extra stress for the birds during molt. In this study, it was shown that the effect of HCB was much stronger than for the other OCs, such as PCBs and DDE (Bustnes *et al.*, 2002).

Cytochrome P450 activities

Only a weak association was found between ΣPCB (nine congeners) concentrations and CYP activity measured as EROD activity in Bjørnøya glaucous gulls (Henriksen *et al.*, 2000). No correlations were seen between OC levels and testosterone hydroxylation activity.

Immunosuppression

Numbers of intestinal macroparasites were compared to hepatic concentrations of several POPs in 40 glaucous gulls from Bjørnøya (Sagerup *et al.*, 2000). After controlling for nutritional condition, no single parasite species was significantly correlated with PCB or pesticide concentrations. However, the intensity of all nematodes grouped together was positively correlated with concentrations of *p,p'*-DDT, mirex, and ΣPCBs (nine congeners). This indicates that high POP concentrations may affect immune function in the glaucous gull.

Thyroid and retinol effects

No significant correlations were seen between ΣPCB concentrations and hepatic retinol and retinyl palmitate concentrations in forty glaucous gulls from Bjørnøya (Henriksen *et al.*, 2000). A significant negative correlation was seen between HCB, *p,p'*-DDE, and ΣPCB blood concentrations and plasma T4 levels in male glaucous gulls from Bjørnøya (Verreault *et al.*, 2002).

Levels and intake assessment

PCB concentrations in the brains of glaucous gulls found dead on Svalbard ranged from 0.9 to 29.5 µg/g ww (Gabrielsen *et al.*, 1995) and in living glaucous gulls, from 0.5 to 9.5 µg/g ww (Henriksen *et al.*, 1998b).

Tissues other than eggs (muscle, liver, and fat) have been analyzed in glaucous gulls at various sites (Annex Table 11). As stated previously, this makes comparisons difficult. Mean ΣPCB concentrations in liver range from 0.45 µg/g ww to 22.7 µg/g ww and indicate that several populations probably have elevated levels in eggs as well, some of which may be above thresholds for effects (Figure 6-1). The populations with high ΣPCB concentrations in liver that might be associated with egg levels above thresholds for effects are from Bjørnøya, Svalbard, Novaya Zemlya, Jan Mayen, and Franz Josef Land.

TEQ concentrations were calculated to be 2500 pg/g ww (approximately 60 000 pg/g lw) based on the nPCB and mono-*ortho* PCB concentrations in Svalbard glaucous gull liver (Daelemans *et al.*, 1992) (Annex Table 16). These TEQ levels exceed all NOAELs and LOAELs for reproductive effects and LD₅₀s in a range of other bird species (Figure 6-2). Lipid weight TEQ levels in liver also exceed the LOEL for induction of CYP1A in common terns.

For ivory gull from north Baffin Bay, liver and fat tissues were analyzed (Annex Table 11). ΣPCB concentrations in liver are 0.3 µg/g ww, and indicate that levels in eggs are also low and probably below thresholds for effects. For great black-backed gulls from Jan Mayen, liver was analyzed. ΣPCB concentrations in liver are 9.6 µg/g ww, and indicate that levels in eggs may also be high and probably exceed some thresholds for effects.

TEQ concentrations based on PCDD/Fs, nPCBs and mono-*ortho* PCBs were determined in herring gulls and great black-backed gulls from four sites in northern Norway (Gabrielsen, 2002). These concentrations ranged from 37 to 72 pg TEQ/g ww. These TEQ levels exceed the LOAEL for reproductive effects in wood duck, a sensitive species (Figure 6-2).

Levels of DBT (n.d.-51 ng/g ww) and MBT (n.d.-14 ng/g ww) in glaucous gull livers from Bjørnøya (Berge *et al.*, 2002) are well below the thresholds associated with hepatic and immune effects. TBT levels were below the detection limits.

Ivory, herring, and great black-backed gulls feed primarily on fish but also scavenge. Glaucous gulls prey on eggs, chicks, and even adult seabirds as well as fish. The assessment for dietary intake of dieldrin, HCB, PCB, DDT, and toxaphene from fish is the same as given for eiders (Section 6.3.3.1). Based on contaminant levels found in eggs from thick-billed murres, black guillemots, black-legged kittiwakes, and fulmars from the Alaskan, Canadian, Greenlandic, and Norwegian Arctic (Annex Table 11), dieldrin, ΣDDT, and ΣPCB levels are below dietary LOECs for reproductive effects found in peregrine falcons. Mean dieldrin levels in these seabird species' eggs also do not exceed any environmental guidelines for protecting aquatic wildlife. However, ΣDDT levels in eggs and liver from these species from many of these sites, as well as kittiwakes from Russian sites in the Barents Sea, exceed Canadian and U.S. EPA guideline levels for protecting wildlife

that consume aquatic biota (Figure 6-6). Most ΣPCB levels exceed all guideline levels (Figure 6-5) and where measured, toxaphene levels exceed Canadian guidelines. PCDD/F and nPCB levels given as TEQs exceed Canadian (avian) and U.S. EPA guideline values (Table 6.1) in kittiwake, fulmar and murre eggs and liver, black guillemot liver and common guillemot eggs (Annex Table 16). Thus, seabirds that prey on seabird eggs, chicks and adults may have dietary intakes of TEQs, ΣDDTs, ΣPCBs and toxaphene high enough to lead to effects.

6.3.3.5. Black-legged kittiwakes

On a Swedish Arctic research expedition in 1996 in the Barents Sea, several juvenile kittiwakes with crossed bills and clump feet were observed (Kylin, 1997a; 1997b).

Levels and intake assessment

In many cases, tissues other than eggs (muscle, liver and fat) have been analyzed in black-legged kittiwakes at various sites. As stated previously, this makes comparisons difficult. If only values for ΣPCB concentrations in eggs are used, the mean is 0.28 µg/g ww at Lancaster Sound, Canada (Annex Table 11). The ΣPCB concentrations based on eggs do not exceed any thresholds for effects in fish-eating birds (Figure 6-1). ΣPCB concentrations in liver range from 0.12 to 1.4 µg/g ww in kittiwakes from sites in Baffin Bay, Nuuk (Greenland) Bjørnøya, Jan Mayen, Svalbard, Franz Josef Land, Novaya Zemlya, and along the Russian Barents Sea coast. This indicates that several populations probably have elevated levels in eggs as well, but these are probably below thresholds for effects.

TEQ concentrations based on PCDD/Fs and nPCBs were 47 pg/g ww (1117 pg TEQ/g lw) in liver and 78 pg/g ww in eggs for Canadian black-legged kittiwakes collected in 1993 (Annex Table 16) (Braune and Simon, 2002). The value for eggs exceeds the LOAEL for reproductive effects in wood duck, a sensitive species (Figure 6-2). Lipid weight levels in liver are well below the LOEL for induction of CYP1A in common terns.

The assessment for dietary intake of dieldrin, HCB, PCB, DDT, and toxaphene is the same as given for eiders (Section 6.3.3.1).

6.3.3.6. Fulmar

No effects studies have been carried out on fulmars in the Arctic.

Levels and intake assessment

Besides eggs, other tissues (liver, fat) have been analyzed in northern fulmars at various sites (Annex Table 11). As stated previously, this makes comparisons difficult. If only values for ΣPCB concentrations in eggs are used, the mean is 0.27 µg/g ww at Lancaster Sound, Canada (Annex Table 11). The mean ΣPCB concentrations in eggs from Lancaster Sound are below any effects thresholds (Figure 6-1). ΣPCB concentrations in liver range from 0.16 µg/g ww to 0.69 µg/g ww in fulmars from northern Baffin Bay and Jan Mayen. These ΣPCB concentrations indicate that levels in eggs are probably low and below thresholds for effects.

TEQ concentrations based on PCDD/Fs and non-ortho PCBs were 357 pg/g ww (8192 pg TEQ/g lw) in liver and 83 pg/g ww in eggs for Canadian Arctic fulmars collected in 1993 (Annex Table 16) (Braune and Simon, 2002). The value for eggs exceeds the LOAEL for reproductive effects in wood duck, a sensitive species (Figure 6-2). Lipid weight levels in liver are well below the LOEL for induction of CYP1A in common terns.

The assessment for dietary intake of dieldrin, HCB, PCB, DDT, and toxaphene is the same as given for eiders (Section 6.3.3.1).

6.3.3.7. Great skua

No effects studies have been carried out on great skuas in the Arctic.

Levels and intake assessment

Only liver has been analyzed in great skuas from Jan Mayen. The mean Σ PCB concentration in liver was 15.9 μ g/g ww. This indicates that great skuas at Jan Mayen probably have elevated levels in eggs as well, and these are probably above thresholds for effects (Figure 6-1). The dietary assessment is the same as for glaucous gulls (Section 6.3.3.4).

6.3.4. Pinnipeds

In the previous AMAP assessment, increased CYP1A enzyme activities were correlated to Σ PCB and dieldrin levels in Canadian ringed seals and to Σ PCB levels in West Ice (Jan Mayen area) hooded seals. Σ PCB levels in Arctic harp, ringed, harbour and grey seals from all sites studied exceeded the NOAEL and LOAEL for subtle neurobehavioral effects if they were as sensitive as the offspring of rhesus monkeys and humans, but were below the NOEL for otter reproduction and mink kit survival (Figure 6-3). PCDD/F and nPCB levels expressed as TEQs in ringed and harp seals from Svalbard, the Greenland Sea, and several sites in Canada were somewhat lower than levels associated with immunosuppressive effects in harbour seal (Figure 6-4).

Concentrations of dieldrin, Σ DDTs, and chlordanes in marine crustaceans and fish were several orders of magnitude below those expected to result in effects on seal reproduction, and toxaphene levels in fish were below those associated with thyroid effects. Σ PCB levels were also below effect levels in crustaceans, but levels exceeded the dietary NOAEC for reproduction in several fish species. However, Σ PCB and Σ DDT levels in a range of fish species from many Arctic sites exceeded some, and in a few cases for Σ PCBs, all of the environmental quality guidelines for protecting fish-eating wildlife. This indicates that dietary intakes of Σ DDTs and Σ PCBs in fish-eating marine mammals may be high enough to lead to effects.

6.3.4.1. Seals and sea lions

Reproductive effects

Breeding rookeries for more than 72% of the world's population of northern fur seals are located on the two largest Pribilof Islands, St. Paul and St. George, Alaska in the Bering Sea (Loughlin *et al.*, 1994). The current

Pribilof stock abundance is less than half of historical levels and is listed as depleted under the *Marine Mammal Protection Act* (Loughlin *et al.*, 1994; York *et al.*, 1997). The St. George subpopulation underwent an unexplained decline of 4-6% per year for more than a decade prior to the mid-1990s study (York *et al.*, 1997). Long-term monitoring of population trends suggest that the decline was due, at least in part, to increased post-weaning mortality at sea (Trites and Larkin, 1989; Trites, 1992). Cause(s) of the increased mortality is unknown, but is thought to be associated with shifts in the abundance and composition of their primary prey species.

Steller sea lion populations have been declining in western Alaska including the Aleutian Islands, and the western stocks are considered endangered.

Cytochrome P450 effects

Studies have been done on liver enzyme induction in ringed and harp seals around Svalbard, correlated to levels of OCs such as PCB and toxaphene. Elevated hepatic EROD activities were found in subadult harp seals sampled from the northwest Barents Sea, east of Svalbard (Wolkers *et al.*, 2000). No correlation was found between these activities and the PCB concentrations in the seals. A highly positive correlation was, however, found between toxaphene levels and testosterone 6- β hydroxylation activities (CYP3A). A positive relationship was found between CYP enzyme activity and Σ PCB levels in ringed seals from Svalbard (Wolkers *et al.*, 1998b). Ringed seals from Svalbard and grey seals from Sable Island (Canadian Arctic) had lower EROD and PROD activities than grey and ringed seals from the Baltic Sea (Nyman *et al.*, 2000).

Thyroid and retinol effects

Retinol levels in northern fur seal neonates were negatively correlated to two recalcitrant PCB congeners in whole blood: CB138 and CB153/87 (rank correlation coefficients of -0.403 and -0.452, respectively, $p < 0.05$) (Beckmen, 1999). Serum retinol levels in neonates were also negatively correlated to the TEQs in the perinatal milk (correlation coefficient of -0.475, $p = 0.029$, $n = 21$). Total T4 was negatively correlated with CB101/99/149/196, CB118, CB138, and TEQs (Spearman rank correlation coefficients = -0.313, -0.519, -0.457, -0.354, respectively, $p < 0.05$). The negative correlation of retinol and thyroid hormones with PCBs in northern fur seal blood suggests that blood levels of these may be reduced because of PCB exposure in young pups. Thus, PCB exposure in young pups has the potential to affect immune function, and therefore, health, both through direct immunosuppression and indirectly, by lowering circulating levels of retinol and thyroxine.

Immune system effects

The native Aleut populations of the villages on St. George and St. Paul are dependent on an annual subsistence harvest of subadult (2-4 years of age) male fur seals both culturally and as a major source of protein. Aleut concerns over the unexplained decline in the population of northern fur seals prompted a study to evaluate the potential effects of OC contaminant exposure on immune function in a cohort of free-ranging pups.

In 1996, 50 perinatal pups were captured for blood sample collection (during the ten-day perinatal period and referred to as neonates) and forty-three were re-sampled ('pups') 29 to 51 days later. Groups of pups were compared based on the relative age of the dam to study immunologic effects of exposure via milk to high (young dam, presumably primiparous) or low (old dam, multiparous) doses of OCs (Beckmen *et al.*, 1999). There were no significant differences in the mass or length (when adjusted for sex) of pups during the perinatal period between pups born to young and old dams. Likewise, growth rates during the study were similar, and the survival rates to the middle of the nursing period were not significantly different although the survival rates of pups of young dams were higher (88% vs. 84%, $p = 0.858$). The only significant difference was a slightly higher body condition index score in pups of old dams ($p = 0.047$). Mean blood OC levels were higher in neonates than at recapture, and neonates of young dams had higher mean blood OC levels than neonates of older dams (Beckmen, 1999; Beckmen *et al.*, 1999).

In the same pups, humoral immune function was assessed by antibody responses to tetanus toxoid vaccination and total immunoglobulin levels. Cellular immune function was assessed using mitogen-induced lymphocyte proliferation assays. Additional indicators of health status included complete blood cell counts and haptoglobin levels. A higher proportion of pups born to old dams developed a two-fold or greater increase in tetanus antibodies compared to the pups of young dams. Forty-one percent (9 of 22) of the pups of old dams responded with the expected 2-fold or greater increase, whereas only 5% of the pups of young dams responded to the antigen (1 of 21). The difference in the proportion of pups responding by the dam's age was significant ($Z = 2.443$, $p = 0.015$). When hemograms of neonates of young and old dams were compared, the mean hematocrit, total plasma protein and absolute eosinophil counts were significantly greater in the neonates of old dams. In neonates, elevated immature (band) neutrophil counts (indicative of inflammation) were correlated to higher TEQs ($r = 0.347$, $p = 0.016$, $n = 48$). In recaptured pups, the total leukocyte counts were positively correlated to higher total PCB concentrations in blood ($r = 0.359$, $p = 0.023$, $n = 41$).

Levels of serum haptoglobin were used as a further measure of subclinical inflammation. There were no differences between neonatal and recapture levels for the pups of old dams, but haptoglobin levels increased significantly from neonatal to recapture for pups of young dams (Mann-Whitney Rank Sum test, $p < 0.001$). No direct correlations of serum haptoglobin concentration with blood OC concentrations were detected. Neonates of young dams had significantly ($p < 0.001$) lower mean immunoglobulin levels than neonates of old dams, 3.14 ± 1.2 $\mu\text{g/ml}$ versus 4.6 ± 1.1 $\mu\text{g/ml}$, respectively. Lymphoproliferative responses, when combined with the results of the previous year's cohort of neonatal fur seal pups, were negatively correlated with blood OC levels, indicating a possible suppression of cellular immune function (Beckman, 1999).

A recent study examining a cohort of 12, known-age, female pups born to young dams and sampled at ap-

proximately monthly intervals from birth to weaning, further substantiated the results of the original study. The pups were given both primary and secondary vaccinations with tetanus toxoid and examined for primary and secondary (memory) antibody response. When converted on a molecular weight basis, the effect of PCB blood levels at the time the initial and booster vaccinations with tetanus toxoid were given had a highly significant ($p = 0.008$ and 0.026 , respectively) negative effect on the increase in antibody titer. All congeners examined, except for *p,p'*-DDE, were significant individually. In these first-born pups, using set correlation analysis, it was determined that the total immunoglobulin concentrations were related to both TEQs (negative) and the age (in days) of the pup ($p < 0.0001$, $p = 0.0512$, respectively). Additionally, total immunoglobulins were related to ΣPCB concentrations and the age of the pup. Perinatal TEQs had a significant negative correlation to the total immunoglobulin levels at that time and at the next recapture. TEQs at the first recapture were also significantly negatively correlated to total immunoglobulins at that time and nearly significantly negatively correlated subsequently ($r = -0.65$ and -0.633 ; $p = 0.0501$ and 0.0583 , respectively). Set correlation analysis confirms this effect is correlated to TEQs by taking into consideration the effect of age on immunoglobulin levels. The strong correlation with PCB exposure expressed as TEQs suggests that the effect on humoral immunity may be mediated through, or by a similar mechanism as, the Ah-receptor. ΣPCBs , TEQs, and *p,p'*-DDE were all negatively correlated (Spearman Rank Order Correlation) to lymphoproliferative assays. However, when a set correlation analysis was applied, age explained most of the effect. Thus, PCB exposure appears to have a negative impact on humoral immunity (immunoglobulin levels and tetanus antibody responses), but the effects on cellular immune function were not substantiated in this cohort (Beckmen, 2002).

Blood and blubber samples obtained from 24 free-ranging Steller sea lion pups and juveniles during the course of live-capture studies in the ranges of both eastern and western stocks, have recently been subjected to similar OC contaminant analysis and immune function assays as the northern fur seals. In this species as well, there were significant negative correlations between lymphocyte proliferative responses to both T cell and mixed B and T cell mitogens and increased blubber TEQs (concanavalin A: $r = -0.440$; pokeweed mitogen: $r = -0.448$, $p < 0.05$). In a subset of eight pups in the eastern, endangered stock, the total immunoglobulin serum concentrations were strongly negatively correlated with blubber concentrations of ΣPCBs , TCDD TEQs and *p,p'*-DDE ($r = -0.933$, $p = 0.005$; $r = -0.881$, $p < 0.001$; $r = -0.893$, $p < 0.001$, respectively) (Beckmen, 2002).

Levels and intake assessment

The mean ΣPCB levels in Arctic ringed, harbour, grey, spotted, and bearded seals, Steller sea lions (Gulf of Alaska and southeast Alaskan populations) and northern fur seals range from 72 to 8400 ng/g lw in blubber (Annex Table 12). These levels exceed the NOAEL and LOAEL for subtle neurobehavioral effects in offspring

of rhesus monkeys and humans, the NOEL for vitamin A reduction in otter, and in some species, are above the NOEL for otter reproduction, but below that for mink kit survival (Figure 6-3). Highest Σ PCB levels were found for ringed seal from Saglek Bay, Labrador (9400 ng/g lw) and from the Kara Sea (1900-11000 ng/g lw). Both exceed the NOEL for mink kit survival, and the levels in the Kara Sea seals are just at the LOEL for vitamin A reduction in otter. These levels are considerably lower than those associated with poor reproductive success in harbour and ringed seal (Figure 6-3). No biopsy data are available for Steller sea lions from endangered populations in the Aleutian Islands. The Σ PCB levels are much higher in scat (up to 7000 ng/g lw) from these populations when compared to those from the other two areas, indicating higher body burdens, but no assessment of these endangered populations can be made. The Σ PCB levels in harp seal from Svalbard range from 247 to 20 400 ng/g lw in blubber. These levels exceed the thresholds for subtle neurobehavioral effects, the NOEL and LOEL for vitamin A reduction in otter, the NOEL for mink kit survival, the LOAEL for decreased kit production and kit body weight gain in mink (Figure 6-3). As well, these levels exceed the threshold associated with immunosuppression in harbour seals and are just below the threshold for immune effects in rhesus monkeys (Figure 6-3).

Levels of nPCBs and mono-*ortho* PCBs given as TEQs range from 27 to 90 pg/g lw in blubber and blood from various age groups of Alaskan northern fur seals (Annex Table 16). Highest TEQs are found in pups. The TEQ levels for pups are below the combined nPCB and mono-*ortho* TEQs (190 pg/g lw) associated with immunosuppressive effects in harbour seals (Figure 6-4). For ringed seals from Pangnirtung, nPCB and mono-*ortho* PCB concentrations in blubber given as TEQs were 0.51 to 0.85 pg/g lw. For ringed seals from Holman Island, TEQs based on PCDD/Fs, nPCBs and mono-*ortho* PCBs ranged from 4.8 to 97 pg/g lw and for ringed seals from the Kara Sea, nPCB and mono-*ortho* PCB concentrations in blubber given as TEQs were 160 pg/g lw. For ringed seals, bearded seals and spotted seals from eastern Russia (Chukotka), TEQs based on PCDD/Fs were 1.0-1.5 pg/g lw. These TEQ levels are all below the threshold for immunosuppression in harbour seals (Figure 6-4).

PFOS levels in grey, ringed, and northern fur seal from the Arctic as well as Steller sea lions from the Alaskan coast are lower than in marine mammals from more southerly latitudes, ranging from <3 to 120 ng/g ww in liver or plasma (Giesy and Kannan, 2001; Kannan *et al.*, 2001a). These levels are well below the NOAEL (15 000 ng/g ww in liver) and LOAEL (58 000 ng/g ww in liver) for second generation effects in rats.

Levels of TBT (1.9-5.6 ng/g ww) and DBT (n.d.-20 ng/g ww) in Alaskan Steller sea lions, in Svalbard ringed seals (DBT-3.1 ng/g ww, MBT-1.5 ng/g ww, TBT not detected), and Canadian ringed seals (below detection) are well below the thresholds associated with hepatic and immune effects in laboratory rodents.

Based on the dietary NOAECs and LOAECs given for mink, mean levels of dieldrin, Σ DDTs (Figure 6-6), and chlordanes in marine fish are several orders of magnitude below those expected to result in effects on ma-

rine mammal reproduction. Based on the NOAEC for rats and dogs, toxaphene levels in fish are below those associated with thyroid effects. Assuming that marine mammals are as sensitive as mink, mean levels of Σ PCBs in tissues from several fish species exceed the dietary NOAEC for reproduction of 72 ng/g ww (Figure 6-5). These include shorthorn sculpin (liver) from several sites on Greenland and the Faroe Islands; Greenland halibut (liver) and Atlantic cod (liver) from Greenland and Iceland; starry ray (liver) from Greenland; sturgeon (liver) from the Kara Sea; Greenland shark (liver) from Davis Strait and Cumberland Sound; polar cod (liver) from Jan Mayen; and, several species (liver) from around Jan Mayen (long rough dab, Atlantic poacher, checkered eelpout) (Annex Table 10).

Mean dieldrin levels in Arctic anadromous and marine fish from Canadian, Norwegian, and Greenlandic waters (Annex Table 10) do not exceed any of the guidelines for protecting fish-eating wildlife (Table 6.1). However, Σ PCB and Σ DDT levels in a range of fish species from many Arctic sites do exceed some, and in a few cases for Σ PCBs, all of the environmental quality guidelines for protecting fish-eating wildlife given in Table 6.1 (Figures 6.5 and 6.6). Toxaphene levels in several fish species also exceed the Canadian Tissue Residue Guideline for protecting wildlife consumers of aquatic biota. This implies that dietary intakes of Σ DDTs, Σ PCBs, and toxaphene in some fish-eating mammals may be high enough to lead to effects if mammals prey on these fish species. No data are available for levels of dioxin-like compounds in anadromous and marine fish so the dietary intake of TEQs cannot be assessed.

For Σ DDTs, the fish species that exceed both Canadian and U.S. EPA guidelines are spotted wolffish (liver) from southwest Greenland; Atlantic cod (liver) from Greenland, Iceland and the Faroe Islands; shorthorn sculpin (liver) from northeast and southwest Greenland and the Faroe Islands; starry ray (liver) from Greenland; Greenland shark (liver) from Canada; sturgeon (liver) from the Kara Sea; and, Greenland halibut (liver) from Greenland. The fish species that exceeded the International Joint Commission objectives for protection of aquatic life and wildlife of 1000 ng/g ww were Greenland shark (liver) from Canada and Greenland halibut (liver) from West Greenland (Figure 6-6).

For Σ PCBs, the species that exceed Canadian guidelines for mammalian predators are spotted wolffish (liver) from southwest Greenland; polar cod from Jan Mayen (Norway); Atlantic cod (liver) from Greenland, the Faroe Islands, Iceland and Svalbard; shorthorn sculpin (liver) from Greenland and the Faroe Islands; dab (liver) from Iceland; starry ray (liver) from Greenland; Greenland shark (liver) from Canada; sturgeon (liver) from the Kara Sea; Greenland halibut (liver and muscle) from Greenland; Arctic char (muscle) from several sites in Canada and liver from some Russian sites; Atlantic salmon (*Salmo salar*) (muscle) from West Greenland; and, a number of other marine species (liver) from around Jan Mayen (coalfish, long rough dab, checkered eelpout, daubed shanny, Atlantic poacher, grey gurnard) and the White Sea, Russia (*Gadus sp.*, herring, navaga (*Eliginius navaga*), sculpin). The fish species that exceed International Joint Commission objectives are Atlantic cod (liver) from Greenland; short-

horn sculpin (liver) from southwest Greenland and the Faroe Islands; sturgeon (liver) from the Kara Sea; Greenland shark (liver) from Canada; Greenland halibut (liver) from Greenland; and, long rough dab (liver) and checkered eelpout from Jan Mayen. Only liver from sturgeon from the Kara Sea, Greenland halibut, and Atlantic cod from Greenland, Greenland shark from Canada as well as long rough dab and checkered eelpout from Jan Mayen exceed the U.S. EPA guideline values for assessment of hazards to fish-eating wildlife (Figure 6-5).

For toxaphene, the species that exceed the Canadian Tissue Residue Guideline are spotted wolffish (liver) from southwest Greenland; polar cod (liver) from Barrow, Alaska; Atlantic cod (liver) from southwest Greenland and Iceland; herring (whole fish) from the White Sea, Russia; shorthorn sculpin (liver) from southwest Greenland, and the Faroe Islands; dab (liver) from Iceland; starry ray (liver) from southwest Greenland and Arctic char (fillet) from Barrow (Annex Table 10).

6.3.4.2. Walrus

In the previous AMAP assessment, Σ PCB levels for the NOAEL and LOAEL for subtle neurobehavioral effects in offspring of rhesus monkeys and humans were exceeded in walrus from eastern Baffin Island, eastern Hudson Bay, northeastern Hudson Bay, and Svalbard. Σ PCB levels in walrus from eastern Hudson Bay (Inukjuak) and Svalbard also exceeded the NOEL for otter reproduction and mink kit survival (Figure 6-3).

Walrus from Inukjuak in eastern Hudson Bay had PCDD/F and/or nPCB TEQ levels that exceeded TEQ levels associated with immunosuppression in harbour seals (Figure 6-4).

For walrus feeding on marine invertebrates, no risks for reproductive effects were indicated based on levels of dieldrin, Σ DDTs, chlordanes, Σ PCBs, and PCDD/Fs, or based on environmental guidelines.

Some walrus prey on ringed seal, and Σ PCBs, Σ DDTs, and TEQs in ringed seal blubber exceeded dietary NOAECs and a range of guidelines for protecting aquatic wildlife, indicating exposure to these substances at levels that could be expected to lead to effects.

No effects studies have been carried out since the previous assessment.

Levels and intake assessment

Mean Σ PCB levels in walrus from Alaska (Bering Sea), Canada, and northeast and northwest Greenland ranged from 33 ng/g lw to 3412 ng/g lw (Annex Table 12). Σ PCB levels in walrus from Alaska, Canada, and northwest Greenland (Avanersuaq) are below all thresholds for effects (Figure 6-3). Σ PCB levels in northeast Greenland walrus from Ittoqqortoormiit exceed the NOAEL and LOAEL for subtle neurobehavioral effects in offspring of rhesus monkeys and humans, and are close to the NOEL for vitamin A reduction in otter (Figure 6-3).

Based on the dietary NOAECs and LOAECs given for mink, mean levels of dieldrin, Σ DDTs, and chlordanes in marine invertebrates are below those expected to result in effects on walrus reproduction. Σ PCB levels in the diet are also below effect levels in invertebrates.

Levels of Σ DDTs, Σ PCBs, and PCDD/Fs in invertebrates are also below the various environmental guidelines (Table 6.1). Mean dieldrin levels in blue mussels from Canadian and Greenlandic waters (Annex Table 10) do not exceed any of the guidelines for protecting fish-eating wildlife (Table 6.1). Toxaphene levels in blue mussels from one Greenland site and from several sites around Iceland are below the Canadian guideline. However, Σ PCB levels in blue mussels from some sites in northern Quebec, Canada, exceed the Canadian Tissue Residue Guideline for protecting wildlife consumers of aquatic biota (Figure 6-5).

6.3.5. Cetaceans

In the previous AMAP assessment, increased CYP1A enzyme activities were correlated to mono-*ortho* and nPCB concentrations in beluga from the Canadian Arctic. Minke whale, beluga, and narwhal had Σ PCB levels which exceeded the NOAEL and LOAEL for subtle neurobehavioral effects if they are as sensitive as offspring of rhesus monkeys and humans, but were below the NOEL for otter reproduction and mink kit survival (Figure 6-3). For harbour porpoise from the southern Barents Sea, Σ PCB levels also exceeded the NOEL for otter reproduction, mink kit survival, and the levels associated with immunosuppression, and approached the levels associated with poor reproductive success in harbour seal (Figure 6-3). Canadian beluga and narwhal had TEQ levels that were considerably lower than those associated with immunosuppressive effects (Figure 6-4).

Concentrations of dieldrin, Σ DDTs, and chlordanes in cetacean food items, such as marine crustaceans and fish, were several orders of magnitude below those expected to result in effects on reproduction, and dietary toxaphene levels in fish were below those associated with thyroid effects. Σ PCB levels were also below dietary effect levels in crustaceans, but levels exceeded the dietary NOAEC for reproduction in several fish species. However, Σ PCB and Σ DDT levels in a range of fish species from many Arctic sites exceeded some, and in a few cases for Σ PCBs, all of the environmental quality guidelines for protecting fish-eating wildlife, indicating that dietary intakes of Σ DDTs and Σ PCBs in fish-eating mammals could be high enough to lead to effects.

No biological effects studies have been carried out on cetaceans since the previous assessment.

6.3.5.1. Mysticetes

6.3.5.1.1. Minke whales

Levels and intake assessment

Σ PCB levels in minke whales ranged from 230 ng/g lw to 20760 ng/g lw. Mean Σ PCB levels in all minke whales exceed the NOAEL and LOAEL for subtle neurobehavioral effects if they are as sensitive as the offspring of rhesus monkeys and humans. Some minke whales from several sites (Jan Mayen, North Sea, Svalbard, northwest Kola Peninsula, northern Norway/northwest Russia) have Σ PCB levels that exceed the NOEL and LOEL for vitamin A reduction in otter. Some minke whales from northern Norway/northwest Russia also exceed

the NOEL for otter reproduction and mink kit survival, the threshold for decreased kit production and kit body weight gain in mink, the threshold associated with immunosuppression and vitamin A disruption in harbour seals, but are just below the LOAEL for immune effects in rhesus monkeys (Figure 6-3).

TEQ concentrations based on PCDD/Fs, nPCB, and mono-*ortho* PCBs were 15-74 pg TEQ/g lw in Svalbard (Spitsbergen) minke whales (Annex Table 16). These concentrations are below the threshold associated with immunosuppression in harbour seals (Figure 6-4).

The dietary intake assessment for Σ PCBs, Σ DDTs, dieldrin, chlordanes, and toxaphene is the same as for seals and sea lions (Section 6.3.4.1).

6.3.5.1.2. Grey whales

Levels and intake assessment

Σ PCB levels in Bering Sea and Chukotka grey whales ranged from 230 to 2700 ng/g lw, with means of 460-1400 ng/g lw. The levels in the Bering Sea whales exceed the NOAEL and LOAEL for subtle neurobehavioral effects if they are as sensitive as the offspring of rhesus monkeys and humans, but are below the NOELs for otter reproduction and mink kit survival (Figure 6-3). The TEQ concentration based on PCDD/F levels in the Chukotka whales was 1.8 pg/g lw, which is below thresholds for effects (Figure 6-4).

6.3.5.1.3. Bowhead whales

Levels and intake assessment

Mean Σ PCB levels in Alaskan bowhead whales are below thresholds for effects (Figure 6-3).

6.3.5.2. Odontocetes

6.3.5.2.1. Beluga

Levels and intake assessment

The mean Σ PCB levels in beluga from Alaska, Canada, Greenland, and Svalbard range from 700 to 9000 ng/g lw with the highest concentrations at Point Lay (Alaska), Hendriksen Island (western Canada) and Kimmirut (eastern Canada) (Annex Table 13). These Σ PCB levels exceed the NOAEL and LOAEL for subtle neurobehavioral effects in offspring of rhesus monkeys and humans, the NOEL and LOEL for vitamin A reduction in otter, and the NOEL for otter reproduction and mink kit survival (Figure 6-3). The Σ PCB levels in beluga brain from Alaska are below those associated with decreased dopamine concentrations in non-human primate brain tissue.

TEQ concentrations based on nPCBs and mono-*ortho* PCBs in beluga from Kimmirut were 0.3-2.4 pg/g lw (Annex Table 16). If PCNs are included, the TEQs were 0.33-2.9 pg/g lw. TEQ concentrations based on nPCBs 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 harbour seals (Figure 6-4).

Levels of TBT and DBT in Canadian beluga were below detection limits. The dietary intake assessment for Σ PCBs, Σ DDTs, dieldrin, chlordanes, and toxaphene is the same as for seals and sea lions (Section 6.3.4.1).

6.3.5.2.2. Killer whales

Levels and intake assessment

The mean Σ PCB levels in killer whales from Alaska (Prince William Sound) are 14 400 ng/g lw in residents (range: 1100-65 000 ng/g lw) and 240 000 ng/g lw in transients (range: 66 000-550 000 ng/g lw) (Annex Table 13). The mean Σ PCB levels in resident killer whales exceed the NOAEL and LOAEL for subtle neurobehavioral effects in offspring of rhesus monkeys and humans, the NOEL and LOEL for vitamin A reduction in otter, the NOEL for otter reproduction and mink kit survival, and the threshold for decreased kit production and kit body weight gain in mink. Based on the maximum levels, Σ PCB concentrations exceed the threshold associated with immunosuppression and vitamin A disruption in harbour seals, the LOAEL for immune effects in rhesus monkeys, the threshold for poor reproductive success in harbour seal, and the EC₅₀ for decreased litter size in mink (Figure 6-3). For the transient killer whales, mean and maximum Σ PCB levels are above all thresholds (Figure 6-3).

Mean TEQ concentrations based on mono-*ortho* PCBs were 100 pg/g lw (range: 5.9-470 pg/g lw) for residents and 860 pg/g lw (range: 190-2400 pg/g lw) for transients (Annex Table 16). The mean TEQ for residents is below the threshold associated with immunosuppression in harbour seals, but individuals with maximum values exceed this threshold (Figure 6-4). All TEQs for transients exceed this threshold.

The resident killer whales feed primarily on fish. Therefore, the dietary intake assessment for Σ PCBs, Σ DDTs, dieldrin, chlordanes, and toxaphene for resident killer whales is the same as for seals and sea lions (Section 6.3.4.1). The transient killer whales feed on other marine mammals. Based on mean POP levels in ringed, bearded and northern fur seals as well as Steller sea lion blubber from various sites in Alaska (Annex Table 12), dieldrin, chlordanes (heptachlor epoxide), and Σ DDT levels are below the dietary NOAECs for reproductive effects in mink. Assuming that marine mammal-eating transient killer whales are as sensitive as mink, mean levels of Σ PCBs in these seal species exceed the dietary NOAEC for reproduction of 72 ng/g ww. As prey items, mean Σ DDT levels in all Alaskan seal species exceed Canadian and U.S. EPA guidelines for protecting aquatic wildlife, and for northern fur seal, International Joint Commission objectives for protection of aquatic life and wildlife are also exceeded (Figure 6-6). For Σ PCBs, all Alaskan seal species exceed all environmental guidelines for protecting aquatic wildlife (Figure 6-5) (Table 6.1). For nPCB TEQs, northern fur seal blubber exceeds both the Canadian mammalian tissue residue guideline and the U.S. EPA guideline for protecting wildlife. No data for toxaphene are available in Alaskan seal species and the dietary intake cannot be assessed.

6.3.5.2.3. Long-finned pilot whales

Levels and intake assessment

The mean Σ PCB concentrations in long-finned pilot whales from the Faroe Islands ranged from 16 000 ng/g lw to 38 000 ng/g lw (Annex Table 13). The mean Σ PCB

concentrations exceed the NOAEL and LOAEL for subtle neurobehavioral effects in offspring of rhesus monkeys and humans, the NOEL and LOEL for vitamin A reduction in otter, the NOEL for otter reproduction and mink kit survival, and the threshold for decreased kit production and kit body weight gain in mink (Figure 6-3). Based on the maximum levels, Σ PCB concentrations exceed the threshold associated with immunosuppression and vitamin A disruption in harbour seals, the LOAEL for immune effects in rhesus monkeys, and the threshold for poor reproductive success in harbour seal (Figure 6-3). Low concentrations of DBT and TBT were found in a few adults and one fetus, but levels were well below thresholds for effects.

The dietary intake assessment for Σ PCBs, Σ DDTs, dieldrin, chlordanes, and toxaphene is the same as for seals and sea lions (Section 6.3.4.1).

6.3.5.2.4. Narwhal

Levels and intake assessment

Many measurements of Σ PCB concentrations in narwhal blubber are based on only ten congeners (Σ PCB₁₀). The mean Σ PCB₁₀ concentrations in narwhal range from 260 to 4750 ng/g lw (Annex Table 13). In general, the Σ PCB₁₀ levels are approximately 30% of Σ PCB levels in beluga, and in one set of narwhal data where both values were calculated. Assuming this is the case for the other narwhal data, then the range of Σ PCB concentrations in narwhal from Canada, Greenland, and Svalbard is estimated to be 900 to 16 000 ng/g lw. These Σ PCB levels exceed the NOAEL and LOAEL for subtle neurobehavioral effects in offspring of rhesus monkeys and humans, the NOEL and LOEL for vitamin A reduction in otter, the NOEL for otter reproduction and mink kit survival, and the threshold for decreased kit production and kit body weight gain in mink, but are just below the threshold associated with immunosuppression and vitamin A disruption in harbour seals (Figure 6-3).

Narwhal eat squid, Arctic cod, shrimp, and Greenland halibut. Based on POP concentrations in Arctic marine fish, the dietary intake assessment for Σ PCBs, Σ DDTs, dieldrin, chlordanes, and toxaphene is the same as for seals and sea lions (Section 6.3.4.1).

6.3.5.2.5. Harbour porpoise

Levels and intake assessment

Mean Σ PCB levels in harbour porpoise from northern Norway are 15 000 ng/g lw (range: 7200-33 000 ng/g lw) and from southwest Greenland, 1300 ng/g lw (range: 880-1530 ng/g lw) (Annex Table 13). The mean Σ PCB levels from both sites exceed the NOAEL and LOAEL for subtle neurobehavioral effects if harbour porpoise are as sensitive as offspring of rhesus monkeys and humans. The mean Σ PCB levels for the porpoises from Norway also exceed the NOEL and LOEL for vitamin A reduction in otter, the NOEL for otter reproduction and mink kit survival, and the threshold for decreased kit production and kit body weight gain in mink, but are below the threshold associated with immunosuppression and decreased vitamin A in harbour seals (Figure 6-3). The maximum Σ PCB levels, however, exceed the threshold associated with immunosuppression and vitamin A

disruption in harbour seals, the LOAEL for immune effects in rhesus monkeys, and the threshold for poor reproductive success in harbour seals. The dietary intake assessment for Σ PCBs, Σ DDTs, dieldrin, chlordanes, and toxaphene is the same as for seals and sea lions (Section 6.3.4.1).

Levels of PCDD/Fs given as TEQs in Greenland harbour porpoise range from 0.2 to 0.9 pg/g lw, and the mean TEQ concentration for PCDD/Fs, nPCBs, and mono-*ortho* PCBs in the Norwegian harbour porpoises was 111 pg/g lw (Annex Table 16). These TEQ levels do not exceed the threshold associated with immunosuppression in harbour seals (Figure 6-4).

Levels of TBT (55-151 ng/g ww), DBT (127-490 ng/g ww), and MBT (11-58 ng/g ww) in Norwegian harbour porpoise from the coast of the Barents Sea are below the thresholds associated with hepatic and immune effects in laboratory rodents.

6.3.6. Polar bear

In the previous AMAP assessment, high cub mortality was found in Svalbard polar bears. A significant negative correlation was found between retinol and Σ PCB concentrations in Svalbard polar bears. Hepatic CYP1A1 and 1A2 content in Canadian male polar bears were found to be correlated with levels of mono-*ortho*- and nPCBs, and CYP2B content was correlated with concentrations of total chlordane (mainly oxychlordane and nonachlor), and total *ortho*-substituted PCBs. A spatial study of POP concentrations in fat was carried out which included polar bear populations from 16 areas across the Arctic; from Canada, Greenland, and Svalbard. All Σ PCB levels exceeded the NOAEL and LOAEL levels found for subtle neurobehavioral effects in offspring, if polar bear are as sensitive as offspring of rhesus monkeys and humans. The Σ PCB levels also exceeded the NOEL for kit survival in mink in four of the areas: Svalbard, East Greenland, M'Clure Strait, and eastern Hudson Bay. Σ PCB levels were close to the kit survival NOEL in several other areas. Σ PCB levels in polar bear from three areas were at or above the LOAEL for immunosuppression in rhesus monkeys: Svalbard, East Greenland, and M'Clure Strait.

Highest Σ PCB levels were found in the population at Svalbard. The Σ PCB levels for different groups of Svalbard polar bears exceeded the NOAEL and LOAEL for offspring neurobehavioral effects as well as the NOEL for mink kit survival. Some individuals in all groups over three years of age exceeded the LOAEL for immunosuppression. Some individuals also exceeded the levels known to be correlated with poor reproductive success in harbour seals, as well as those correlated with poor reproductive success in ringed seal and the EC₅₀ for reduced litter size in mink.

Some polar bear from the Canadian and Svalbard populations had TEQ levels based on PCDD/F and/or nPCB levels that exceeded levels associated with immunosuppressive effects in harbour seal.

Concentrations of Σ PCBs, Σ DDT, and TEQs in ringed seal blubber exceeded dietary NOAEC levels and a range of guidelines for protecting aquatic wildlife, indicating exposure to these substances at levels that could be expected to lead to effects.

The elevated levels of PCBs in adipose tissue of polar bears from the Svalbard area (ranging from 4790 to 80 300 ng/g) (Bernhoft *et al.*, 1997) reported in the previous POPs assessment, have prompted further biological effects studies linked to contaminant analyses.

Reproductive and developmental effects

Some adult female polar bears at Svalbard were equipped with satellite transmitters programmed to send information every six days for two to three years. Information on the location of the transmitter as well as sensor data on internal transmitter temperature and short- and long-term bear activity were recorded. Female polar bears normally have a three-year reproductive cycle (Ramsay and Stirling, 1988). Only pregnant bears den over winter. Reproductive rates were estimated from satellite data (Wiig, 1995; 1998).

From these studies of adult female polar bears at Svalbard, the reproductive rate was found to be approximately 0.75. This is similar to corresponding values found in other polar bear populations (Wiig, 1998). No difference in the PCB levels between females available for mating that became pregnant, and those that did not become pregnant, was found. The sample sizes for this comparison were small, however (Bernhoft *et al.*, 1997). Relatively low cub survival was found at Svalbard, and there are indications that the reproduction cycle was less than three years (Wiig *et al.*, 1992; Wiig, 1998). Thus, epizootological studies suggest that reproduction and cub survival in polar bears at Svalbard may be impaired. The high intake of PCBs at a crucial period could adversely influence the early development of cubs and lead to higher mortality. However, other factors such as population density may play a role, and a causal link with PCBs cannot be established.

POP concentrations were determined in adipose tissue, plasma, and milk samples from seven female polar bears and their cubs near Cape Churchill, Hudson Bay, between 1992-1996 (Norstrom, 1999b; Polischuk, 1999). 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, after emerging from dens but before they had moved onto the ice to begin hunting seals. Mothers that were recaptured a third time in the following autumn without cubs were found to have had high OC concentrations in their milk when they emerged from the den in the previous spring (Figure 4-55). By comparison, mothers recaptured in autumn and still accompanied by cubs, had low OC concentrations in their milk the previous spring. The differences in concentrations were significant ($p < 0.05$) for all residue classes. For example, PCB concentrations were approximately three 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 degree of exposure to OCs in milk (Norstrom, 1999b).

The age structure of polar bears on Svalbard was compared to that of several other populations from less contaminated areas (Derocher *et al.*, 2003) as a low frequency of older female polar bears on Svalbard has been

reported previously (Wiig, 1998). The proportion of females with cubs-of-the-year that were older than 16 years old was found to be significantly lower on Svalbard (12.7%) when compared to western Hudson Bay (40.3%).

A significant negative relationship was found between Σ PCBs and testosterone, as well as total pesticides and testosterone in polar bear plasma, after correcting for possible confounders such as age and condition, indicating that PCBs and/or pesticides may decrease circulating testosterone levels in male polar bears (Oskam *et al.*, 2001) (Figure 6-7). Testosterone is the major androgenic steroid hormone playing a crucial role in male sexual development.

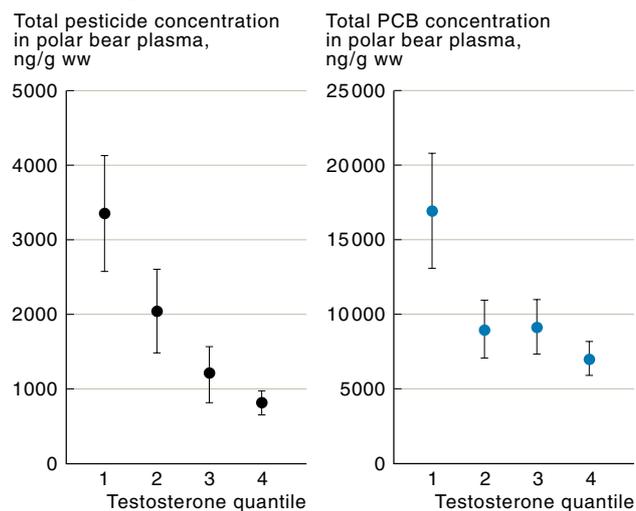


Figure 6-7. Total pesticide and PCB concentrations (mean and 95% C.I.) by testosterone quantile (Oskam *et al.*, 2001).

In 1999, 52 experienced polar bear hunters living in East Greenland were interviewed about their observations of aberrant bears including pathological changes/anomalies in internal organs (Dietz *et al.*, 2001; Sandell *et al.*, 2001). Information on approximately 1110 bears that had been shot between 1945 and 1999 was obtained. Thirteen anomalous polar bears were reported. The anomalies included supernumerary nipples or claws, unilateral collapse of lung, abnormal and missing claws, partial melanism, missing limbs, and a malformed newborn.

A study of contaminant-induced changes in polar bear skulls from East Greenland was initiated in 1999. The results are still very preliminary as both the POP analyses and the age determination of the bears are still in progress. Approximately 180 skulls (1892-1987) from the National Zoological Museum in Copenhagen were compared with 100 recent (1999-2001) skulls to detect possible macroscopical pathological changes, asymmetrical changes as well as osteopenia (osteoporosis and osteomalacia). Both directional and fluctuating asymmetry were found in several bones of the skulls. Surprisingly, the amount of fluctuating asymmetry seemed to have decreased from the period of 1892-1960 to the period 1960-2001. The results are only preliminary, however, and awaiting the inclusion of age data in the analysis.

The frequency of paradontitis (loose teeth) among subadult polar bears was higher, but not statistically significant, in the period of 1960-2001 than in the period

1892-1960. A clear sex difference was observed in the bone mineral density measured as calcium-phosphate content. There was also a tendency for the males from the period of 1960-2001 to have lower bone mineral density as compared to males from the period of 1900-1960.

Several female polar bears from Svalbard have been found to be pseudohermaphrodites, and high PCB levels have been hypothesized as one possible causative factor (Wiig *et al.*, 1998). The link to POP exposure; however, is not strong because of the occurrence of the same syndrome in black bears (Norstrom, 2002). One pseudohermaphroditic female was also found among more than 100 sampled polar bears in 1999 on East Greenland (Sonne-Hansen *et al.*, 2002) and the occurrence of this one case is lower than the estimated frequency found at Svalbard of approximately 3% (Derocher, 2002).

Cytochrome P450 activities

CYP1A1 has been determined in white blood cells of polar bears and the results ($n=13$) show a significant positive correlation between Σ PCBs (the sum of PCBs 99, 118, 153, 156, 180, and 194) and CYP1A1 area in western blots ($p=0.026$). The strongest correlation was found with CB156 ($p=0.011$). These results are promising with regard to the potential for CYP1A1 in blood cells as a biomarker for PCB exposure in polar bears (Skaare *et al.*, 2000).

Thyroid and retinol effects

Normal regulation of vitamin A and thyroid hormones is important for a wide range of biological functions, such as growth, cell differentiation, reproduction, behavior, and the immune system. In Svalbard polar bears, retinol and thyroid hormones (T3 and T4), and concentrations of several OC contaminants have been determined in blood plasma of 71 individuals collected from 1991 to 1994. The determination of multivariate associations between retinol, thyroid hormones, and the ratio of total and free thyroid hormones, respectively, and the concentrations of various OC components (PCBs, DDE, HCB and HCHs) revealed significant OC associations for retinol and the ratio of total T4:free T4, after correcting for age and sex (Skaare *et al.*, 2001a). Significant negative correlations were found between retinol and Σ PCBs (Figure 6:8), as well as retinol and HCB and HCHs. Significant negative correlations were also found between total T4:free T4 and Σ PCBs (Figure 6:9) as well as HCB.

Concentrations of the thyroid hormones, T3 and T4, and retinol were determined in plasma of polar bears from Resolute Bay in Canada and Svalbard, which have among the lowest and highest POP concentrations, respectively, in polar bears (Norstrom, 2000). Free T3 (FT3) and free T4 (FT4) indices were also determined. Resolute bears had significantly higher total T4 and FT4 index, and lower total T3 and FT3 index than Svalbard bears ($p<0.001$). Retinol concentrations were not significantly different between regions. None of the biological measures were significantly related to age, even when separated into region and sex categories. This is in contrast to the above study, which found that total T4, FT4, total T3, and FT3 are associated with age in male polar bears (Skaare *et al.*, 2001a). Total T3 was higher

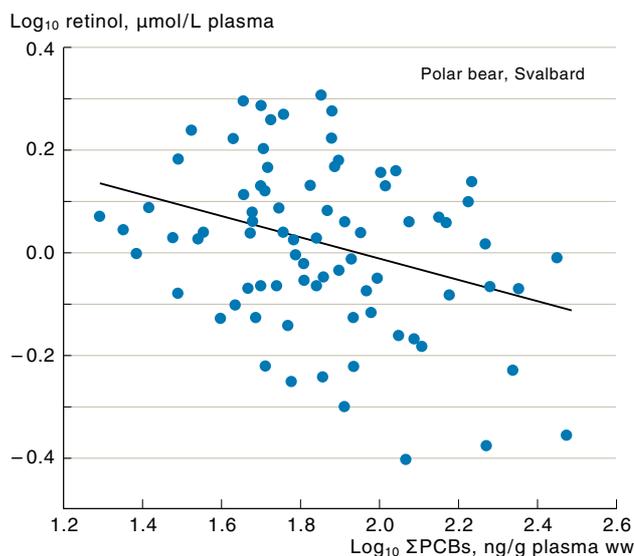


Figure 6-8. The association between retinol (corrected for age and sex) and Σ PCBs in plasma of 79 polar bears at Svalbard by regression analysis. The regression line is shown (\log residual retinol = $0.40 - 0.21 \log \Sigma$ PCBs). The Pearson correlation coefficient is $r = -0.33$ ($p = 0.003$) (Skaare *et al.*, 2001a).

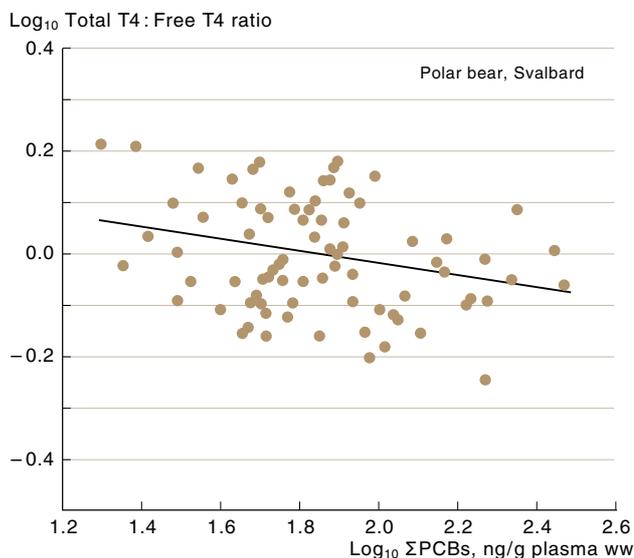


Figure 6-9. The association between the ratio of total T4 and free T4 (TT4:FT4) (corrected for age and sex) and Σ PCBs in plasma of 78 polar bears at Svalbard by regression analysis. The regression line is shown (\log residual total T4:free T4 = $0.21 - 0.12 \log \Sigma$ PCBs). The Pearson correlation coefficient is $r = -0.28$ ($p = 0.013$) (Skaare *et al.*, 2001a).

in females from the Svalbard population, and FT3 index was lowest in Resolute females.

Correlations among thyroid hormone concentrations, FT3 and FT4 indices, retinol concentrations and the complete suite of POP concentrations in adult polar bear plasma ($n=60$) from Resolute Bay, Canada, and Svalbard were examined by principal component analysis. Because hydroxy compounds may affect circulating levels of free and bound T4 due to competitive binding of hydroxy metabolites and T4 to TTR, all chemical concentrations were converted from mass to molar concentrations prior to statistical analysis.

Using the entire data set ($n=60$), retinol concentrations were negatively correlated ($r=-0.465$, $p<0.001$) with persistent PCBs (PC1) and positively correlated

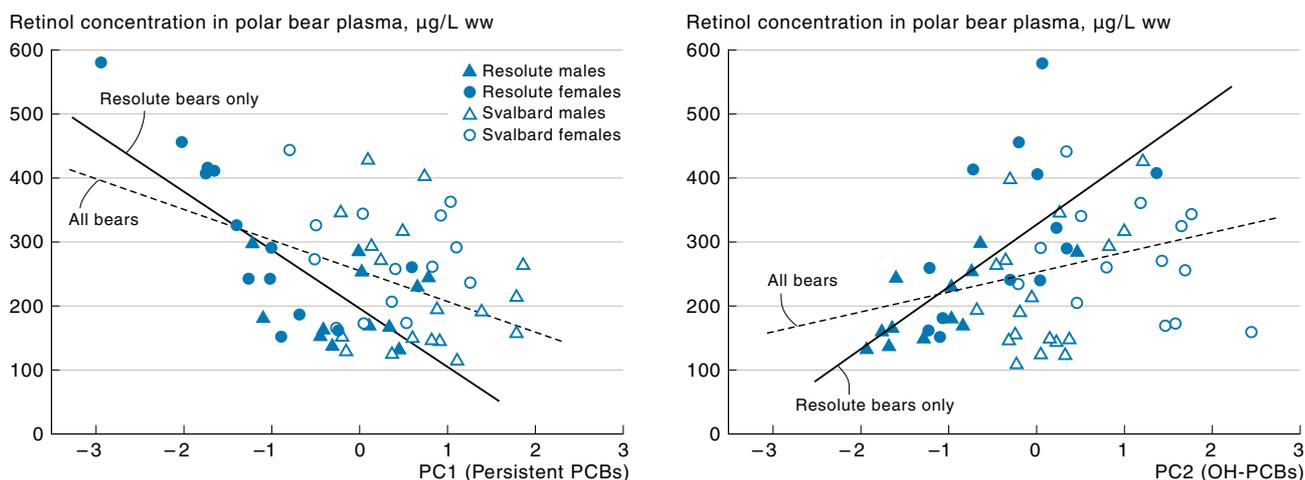


Figure 6-10. Correlation between retinol concentrations ($\mu\text{g/L}$) and the first two principal components from analysis of POP concentrations in polar bear plasma from Resolute Bay in the Canadian Arctic in April-May 1997, and from Svalbard in April-May 1998.

($r = 0.309$, $p = 0.02$) with OH-PCBs (PC2) (Figure 6-10). If only Resolute bears ($n = 25$) were included in the analysis, retinol was more highly correlated with PC1 ($r = -0.744$, $p < 0.001$) and PC2 ($r = 0.692$, $p < 0.001$). These results suggest that plasma retinol concentrations are more likely to be affected by the influence of persistent PCBs on retinol metabolism and storage in liver than by the interference of OH-PCBs with the transport of retinol via RBP:TTR dimer formation.

Total T4 plasma concentrations were negatively associated with both persistent PCBs (PC1), $r = -0.337$, $p = 0.01$, $n = 56$) and non-persistent PCBs (PC3) ($r = -0.293$, $p = 0.03$, $n = 56$), but not with any other contaminant group, including OH-PCBs. Since both PC1 and PC3 are PCB-related, this correlation suggests a common mechanism of action of all PCB congeners in reduction of plasma T4 concentrations. Total T4 concentrations were also negatively correlated with concentrations of ΣPCBs ($r = -0.29$, $p = 0.04$, $n = 56$). Thirty-three female polar bears without cubs and with two-year old cubs were sampled at Svalbard and in the Barents Sea region during the period 1995-1998, and blood samples were analyzed for a range of POPs, thyroid hormones, and progesterone (Braathen *et al.*, 2000). Significant differences were seen in progesterone levels between the females from the two areas, with higher progesterone levels in the Barents Sea bears. However, no relationship could be shown between progesterone levels and the ΣPCB concentrations. For thyroid hormones, significant differences were found for the ratio of total T3: free T3 with higher ratios in the Svalbard bears, but no relationship could be seen between these and ΣPCB levels.

Immune effects

Production of antibodies plays an important role in protection against infections. Antibodies are divided into different immunoglobulin classes, where IgG is the major one in blood. IgG and OC concentrations were determined in blood sampled from 56 free-living polar bears of different ages and both sexes between 1991 and 1994 (Bernhoft *et al.*, 2000). Total IgG concentration increased with age and was significantly higher in males than in females. A significant decrease of IgG with increased ΣPCB level was found ($r = -0.29$, $p = 0.03$). IgG was standardized for sex and age, since both sex and age

may influence the levels. Three individual PCB congeners showed significant inverse correlations to IgG: CBs 99, 194 and 206 (Figure 6-11). In addition, HCB was also inversely correlated to IgG. OCs were found to account for 11% of the variation of IgG levels.

In a study of mothers and cubs on Svalbard sampled in 1995-1998, a negative correlation was found between IgG levels in cub plasma and plasma ΣPCB concentrations (Lie *et al.*, 2002). The results of these studies demonstrate a possible contaminant-associated suppression of antibody-mediated immunity in polar bears at Svalbard.

The effects of high PCB exposure on the immune system have been further studied by comparing the immune system functions in polar bears with high (Svalbard) and low (Canada) PCB exposure (Skaare *et al.*, 2001b; 2002; Larsen *et al.*, 2002c). This field study comprised a vaccination model with recapture after immunization. Thirty bears at Svalbard and thirty in Canada were immunized with different herpes-, reo-, and influenza viruses and tetanus toxoid to stimulate the production of protective antibodies such as virus-neutralizing antibodies, virus hemagglutination inhibition antibodies and toxin-neutralizing antibodies. The immunization also included keyhole limpet hemocyanin that, together with tetanus toxoid, would stimulate cell-mediated immune response. Blood was sampled at immunization, and four to six

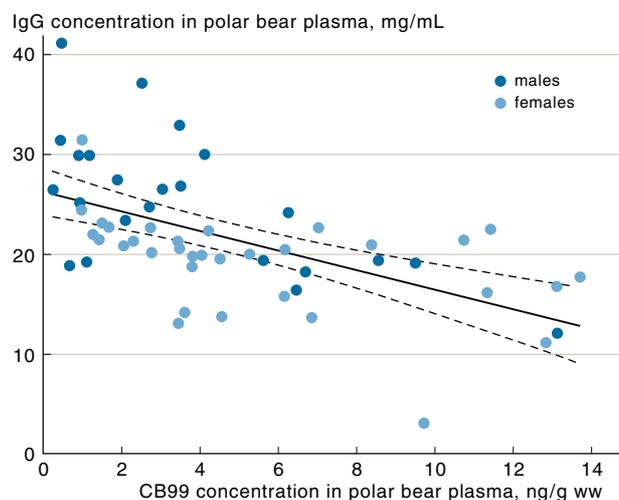


Figure 6-11. Linear regression of IgG level and CB99 in blood plasma of polar bears of both sexes at Svalbard (Bernhoft *et al.*, 2000).

weeks later, for detection of antibodies. In addition, *in vitro* lymphocyte stimulation was performed with mitogens and specific antigens (phytohemagglutinin, concanavalin A, pokeweed mitogen, *Mycobacterium* antigen, and lipopolysaccharide) (Larsen *et al.*, 2002a; 2002b). The resistance factor was measured directly as neutralization of virus infections in cell cultures, inhibition of virus hemagglutination, and toxin neutralization. Therefore, the effect of PCBs on infection resistance was measured without dependence on infection experiments or by conducting registration of disease outbreaks.

Preliminary results from this study demonstrated significantly higher PCB plasma levels in polar bears at Svalbard as compared to Canada. In the Canadian bears, significantly higher antibody titers against influenza, reo- and herpes viruses were found following immunization than at Svalbard. Furthermore, significantly lower antibody titers against influenza virus following immunization, were associated with higher PCB levels. Significantly lower lymphocyte responses were found to pokeweed mitogen, lipopolysaccharide from *E. coli* and *Mycobacterium* antigen with high PCB exposure levels, and significant negative correlations between PCBs and IgG were also found.

From the present preliminary results on effects of PCBs on the immune system of polar bears, it is reasonable to assume that PCBs are associated with decreased resistance to infections. This is supported by the finding that polar bears with high plasma PCB have higher incidence of *Pasteurella* bacteria, one of the most common microbes in the environment (Larsen, 2002).

Levels and intake assessment

Σ PCB concentrations (22-40 congeners) in polar bear fat collected between 1996 and 1997 at several sites around Alaska (Beaufort Sea, Bering Sea, Chukchi Sea) ranged between 910 and 11 000 ng/g lw with means in the range of 2600-9640 ng/g lw (Annex Table 14). Mean Σ PCB concentrations in polar bear fat collected in 1999-2000 from East Greenland ranged from 4800 to 7700 ng/g lw. In polar bear fat from one individual from the pack ice near Iceland and collected in 1993, Σ PCB concentration (7 congeners) was 7860 ng/g ww (approximately 10 000 ng/g lw) (Annex Table 14). Σ PCB concentrations (1990) from a recent temporal-trend study of Canadian polar bear from several sites ranged from 720 to 20 200 ng/g lw with geometric means in the range of 2600-7500 ng/g lw (Annex Table 14). Russian polar bears collected from four areas (Franz Josef Land, Kara Sea, Siberian Sea, Chukchi Sea) have plasma Σ PCB concentrations (5 congeners) ranging from 1100 to 31 000 ng/g lw, and Svalbard polar bears have plasma Σ PCB concentrations ranging from 2100-14 000 ng/g lw in the same study (Andersen *et al.*, 2001b). For Svalbard polar bears of different ages collected in 1995-1998, plasma Σ PCB levels range from 1060 to 29 000 ng/g lw, with highest concentrations found in cubs-of-the-year and yearlings (Annex Table 14).

Most Σ PCB levels in these polar bears exceed the NOAELs and LOAELs found for subtle neurobehavioral effects in offspring if polar bears are as sensitive as offspring of rhesus monkeys and humans, and also, the NOEL for vitamin A reduction in otter (Figure 6-3). The Σ PCB levels in polar bears from several sites in Alaska and Canada, on the Iceland pack ice, at Svalbard and

three sites in Russia (Franz Josef Land, Kara Sea, Siberian Sea) overlap the NOEL for kit survival in mink, and several of these also exceed the LOEL for vitamin A reduction in otter (Figure 6-3). The threshold associated with immunosuppression and vitamin A disruption in harbour seal blubber is 16 500 ng Σ PCB/g lw, and the LOAEL for immunosuppression is 21 000 ng Σ PCBs/g lw (blood) in rhesus monkeys. Σ PCB levels in polar bears from three areas are at or above these two thresholds: Franz Josef Land, Kara Sea, and Svalbard (cubs and yearlings). Since Svalbard polar bears with high Σ PCB levels have been shown to exhibit signs of immunosuppression, it is probable that such effects are also occurring in Russian polar bears from these sites. Σ PCB levels in some polar bears from Davis Strait, Canada are above the threshold for immunosuppression in harbour seals and close to the LOAEL for immunosuppression in rhesus monkeys. Temporal-trend studies for Σ PCBs in Canadian and Svalbard polar bears indicate that there have been no declines during the 1990s. This implies that the assessment of contaminant levels compared to thresholds for other polar bear populations, given in the previous AMAP assessment, is still valid.

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 (Annex Table 16). These are well below the threshold for immunosuppression in harbour seals (Figure 6-4). This may, however, be an underestimation since mono-ortho PCBs usually contribute most to the total TEQs compared to nPCBs and PCDD/Fs (Letcher *et al.*, 1996) and these were not included in the study.

PFOS levels in polar bears range from 26 to 52 ng/g ml in plasma in Beaufort Sea individuals and 180-680 ng/g ww in liver from bears from Barrow, Alaska (Kannan *et al.*, 2001a; Giesy and Kannan, 2001). These levels are well below the NOAEL (15000 ng/g ww in liver) and LOAEL (58000 ng/g ww in liver) for second-generation effects in rats.

Based on mean POP levels in ringed seal blubber from various sites in the Arctic (Annex Table 12), dieldrin, chlordanes (heptachlor epoxide), and Σ DDT levels are below the dietary NOAECs found for reproductive effects in mink. Some harp seals from east of Svalbard have chlordane levels that exceed the LOAEC found for kit growth in mink. Based on the NOAEC (4000 ng/g ww) for rats and dogs, toxaphene levels in ringed and harp seals are below those associated with thyroid effects. Assuming that polar bears are as sensitive as mink, mean levels of Σ PCBs in ringed and harp seal blubber from all sites exceed the dietary NOAEC for reproduction of 72 ng/g ww. Mean Σ DDT levels in ringed and harp seals exceed Canadian and U.S. EPA guidelines for protecting aquatic wildlife, and in some cases (i.e. Svalbard, Russia), International Joint Commission objectives are also exceeded (Table 6.1) (Figure 6-6). For Σ PCB levels, ringed and harp seal blubber from all sites exceed all environmental guidelines for protecting aquatic wildlife (Figure 6-5). Toxaphene levels in ringed and harp 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 and Russia exceed Canadian (mammalian) and U.S. EPA guidelines (Annex Table 16).

6.3.7. Arctic fox

In the previous AMAP assessment, mean Σ PCB levels in Svalbard Arctic fox from 1993 and 1994 were found to exceed the NOAEL and LOAEL for subtle neurobehavioral effects in offspring of rhesus monkeys and humans, the NOEL for kit survival in mink, levels known to cause poor reproductive success in harbour seals, and the EC_{50} for reduced litter size in mink. Some individuals also exceeded the levels associated with poor reproductive success in ringed seals and the EC_{50} for kit survival in mink (80 000-120 000 ng/g lw). The mean Σ PCB levels in Arctic fox were also above the LOAEL for immunosuppression in rhesus monkeys. No effects studies have been carried out since the previous assessment.

Levels and intake assessment

Mean concentrations of Σ PCBs in Arctic fox collected in 1999-2000 from Holman Island, NWT, were 860 ng/g lw in muscle (range: 76-8050 ng/g lw) and 1350 ng/g lw in liver (range: 110-14 600 ng/g lw). Mean Σ PCB concentrations in Arctic fox from Barrow, Alaska and from inland Iceland were 1000-1600 ng/g lw in liver. Mean concentrations of Σ PCBs in Arctic fox fat collected in 1996-1997 in the Pribilof Islands, Alaska, were somewhat higher, between 2100 ng/g lw and 5000 ng/g lw (range: 240-12 000 ng/g lw). Highest mean Σ PCB concentrations were seen in Arctic fox from the coast of Iceland at 72 500 ng/g lw (Annex Table 14). The Σ PCB levels for Arctic fox from all sites exceed the NOAEL and LOAEL for subtle neurobehavioral effects in offspring of rhesus monkeys and humans. The Σ PCB levels in Canadian, Pribilof Islands, and coastal Iceland foxes also exceed the NOELs for vitamin A reduction in otter, effects on otter reproduction and mink kit survival, the LOAELs for vitamin A reduction in otter, and decreased kit production and kit body weight gain in mink (Figure 6-3). The Σ PCB levels in the coastal Iceland foxes also exceed the threshold for immunosuppression and vitamin A disruption in harbour seals, the LOAEL for immune effects in rhesus monkeys, the threshold for poor reproduction in harbour seals, and the EC_{50} for litter size in mink. Levels of mono-*ortho* PCBs given as TEQs in Pribilof Islands Arctic foxes ranged between 9 and 290 pg/g lw (means: 48 and 151 ng/g lw for females and males, respectively) and these exceed the mono-*ortho* TEQs associated with immunosuppressive effects in harbour seals (Figure 6-4).

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. Based on contaminant levels found in eggs and liver from thick-billed murre, black guillemots, black-legged kittiwakes, glaucous gulls, herring gulls, great black-backed gulls, and fulmars from the Alaskan, Canadian, Greenlandic, Norwegian, and western Russian Arctic (Annex Table 11), dieldrin, Σ DDT, and heptachlor levels are below the dietary NOECs and LOECs for reproductive effects in mink. Σ PCB levels are above the dietary NOEC for reproductive effects and TEQs, based on PCDD/Fs, nPCBs, and/or mono-*ortho* PCBs are above the NOECs and LOAECs for effects on reproduction and vitamin A in mink. Dietary toxaphene levels are below those asso-

ciated with thyroid effects. Mean dieldrin levels in these seabird species' eggs also do not exceed any environmental guidelines for protecting aquatic wildlife. However, Σ DDT levels in eggs and liver from many of these species from many of these sites exceed Canadian and U.S. EPA guideline levels for protecting wildlife that consume aquatic biota (Figure 6-6). Most Σ PCB levels exceed all guideline levels (Figure 6-5) and where measured, toxaphene levels exceed Canadian guidelines. PCDD/F and nPCB levels given as TEQs exceed Canadian (mammalian) and U.S. EPA guideline values (Table 6.1) in kittiwake, fulmar, murre, black guillemot, common guillemot, glaucous gull, herring gull, and great black-backed gull eggs and/or liver (Annex Table 16). Thus, Arctic fox that prey on seabird eggs, chicks, and adults may have dietary intakes of TEQs, Σ DDTs, Σ PCBs, and toxaphene high enough to lead to effects. The dietary assessment for Arctic fox that feed on seals is the same as for polar bears (Section 6.3.6).

6.3.8. Sea otter

No biological effects studies have been carried out in sea otter.

Levels and intake assessment

Mean concentrations of Σ PCBs in sea otter from southeast Alaska were 270 ng/g lw and for sea otter from the Aleutian Islands, 10 300 ng/g lw (Annex Table 14). The Σ PCB levels in sea otter from southeast Alaska do not exceed any threshold levels. The Σ PCB levels in sea otter from the Aleutian Islands exceed the NOAEL and LOAEL for subtle neurobehavioral effects in offspring of rhesus monkeys and humans, the NOELs for vitamin A reduction in otter, and effects on otter reproduction and mink kit survival (Figure 6-3).

6.4. Summary and conclusions – biological effects

It is very difficult to link contaminant levels or biochemical indicators of effects to effects on Arctic animals at the individual or population level. Such assessments are also complicated by the fact that the thresholds for effects of many contaminants are not well known, and very little is known about effects of contaminant mixtures. It is also possible that there may be other causes of the effects seen that are unrelated to POP exposure.

As far as OCs are concerned, Arctic marine mammals are often regarded as controls for much more contaminated members of their populations or related species in temperate regions. Biological effects studies on Arctic animals do, however, show some subtle responses, and there are now stronger links between these effects and current levels of some POP contaminants. Based on the results of the biological effects studies that have been carried out, the following conclusions can be drawn.

6.4.1. Observed effects

6.4.1.1. Field studies

Reproduction

- Alaskan peregrine falcons (*tundrius* and *anatum* subspecies) still suffer from eggshell thinning of 10.6%

and 12%, respectively, when compared to eggs from the pre-DDT era.

- Bald eagles showed reduced productivity on Adak Island in the Aleutians, which was associated with the higher DDE concentrations found there than at three other Aleutian islands. Eggshell thickness was significantly negatively correlated with PCB concentrations on the four islands.
- Positive correlations were found between TBT concentrations and degree of imposex in dogwhelks from the Norwegian coast, and imposex is also found along the Faroe Islands and Greenlandic coasts. Imposex has also been seen in dogwhelk from Icelandic harbors but has decreased after restrictions on the use of TBT as an anti-fouling paint on boat hulls were implemented in 1990.
- In glaucous gulls from Bjørnøya, Norway, increased absence from nests was correlated with high OC levels. Female gulls with high OC levels were more likely to have non-viable eggs and chick body condition was poorer. Adult survival was significantly negatively correlated to OC levels. Glaucous gulls with high OC levels have wing feather asymmetry, an indication of developmental stress.
- Egg volume, yolk mass, and hatchling mass in one-day-old shag were negatively correlated to PCB concentrations.
- A significant negative correlation was found between testosterone levels and PCB levels, as well as total pesticide levels in polar bear plasma, indicating that these POPs may decrease circulating testosterone levels. Females from southwest Hudson Bay that had lost their cubs between emerging from the den and the following summer/autumn had significantly higher OC concentrations when emerging from the den than females that kept their cubs.

Other observations have been made where POPs are suspected of playing a role, but where there is as yet no certain link between the effects seen and POP concentrations. These are the following.

- Black-legged kittiwakes with crossed bills and clump feet have been observed during an expedition in the Barents Sea.
- Steller sea lion populations have been declining in western Alaska including the Aleutian Islands and the western stock is considered endangered. These populations excrete higher concentrations of POPs than less-affected populations with lower POP concentrations.
- Reproductive rates in Svalbard polar bears are similar to those of other polar bear populations, but cub survival is lower. Female pseudohermaphroditic polar bears have been found on Svalbard and Greenland. The proportion of females with cubs-of-the-year that were older than 16 years was significantly lower on Svalbard than in western Hudson Bay.

Cytochrome P450 activity

- CYP enzyme activities are correlated to PCB levels in Arctic char from two freshwater lakes (one with high POP levels, one with low POP levels) on Bjørnøya.
- A weak association between PCB levels and EROD activity was seen in glaucous gulls at Bjørnøya.
- A positive correlation was found in black guillemot from Saglek Bay, Canada, between PCB concentrations

and EROD activity up to a threshold of 100 ng/g ww in liver. After this, EROD activity leveled off.

- High correlations have been found between testosterone 6- β hydroxylation (CYP3A) and toxaphene levels in harp seals around Svalbard. Positive correlations have been found between EROD levels and Σ PCBs in Svalbard ringed seals.
- A significant correlation was found between CYP1A1 in polar bear white blood cells and PCB levels.

Thyroid and retinol effects

- A borderline significant correlation was found between PCB concentrations in yolk and plasma retinol levels in one-day-old shag from Norway.
- A significant negative correlation was found between liver PCB concentrations and liver retinol and retinyl palmitate levels in black guillemots at Saglek Bay.
- A significant negative correlation was seen between HCB, *p,p'*-DDE, and Σ PCB blood concentrations and plasma T4 levels in male glaucous gulls from Bjørnøya.
- Retinol levels and total T4 levels in northern fur seal pups were found to be negatively correlated to PCB congeners and to TEQs.
- In Svalbard polar bears, significant negative correlations were found between retinol and PCB levels, as well as retinol and HCB and HCHs. Significant negative correlations were also found between the ratio of total T4:free T4 and PCB as well as HCB. In a comparison of polar bears from Svalbard (high PCB levels) and Resolute, Canada (low PCB levels), a significant negative correlation was found between total T4 levels and Σ PCBs but not with OH-PCBs. Retinol levels were negatively correlated with persistent PCBs and positively correlated with OH-PCBs, suggesting that PCB affects retinol metabolism and storage in the liver.

Immune effects

- In glaucous gulls from Bjørnøya, nematode density was positively correlated with concentrations of *p,p'*-DDT, mirex, and Σ PCBs.
- In northern fur seal and Steller sea lion pups, various measures of normal immune function were negatively correlated to PCB levels, indicating that high PCB exposure may be causing immunodysfunction.
- In Svalbard polar bears, a significant decrease in antibodies (IgG) with increased PCB levels was found. In mothers and cubs, a similar negative correlation was found for IgG levels in cubs and plasma PCB levels as well as with a number of specific congeners (CBs 99, 137, 153, 157, 170, 180, and 194). In a vaccination study using two polar bear populations, one with higher PCB levels (Svalbard) and one with lower PCB levels (Canada), polar bears with high PCB levels were found to exhibit immunosuppression expressed as reduced IgG production and lowered lymphocyte responses. This may indicate decreased resistance to infections.

6.4.1.2. Laboratory studies using Arctic species

Cytochrome P450 activity

- Increased EROD activity was found in Arctic char fed a single oral dose of Aroclor 1260 and then starved, compared to PCB-exposed and fed char, non-exposed and fed char, and non-exposed and starved char.

Immune effects

- Increased cortisol levels were found in Arctic char fed a single oral dose of Aroclor 1260 and then given food, compared to PCB-exposed and starved char, non-exposed and fed char, and non-exposed and starved char. In another study, basal cortisol levels were suppressed by PCBs in starved fish but were elevated in fed fish after handling, indicating that stress responses are compromised by PCBs, and the effect of fasting makes char sensitive to the effects of PCBs. Fasting and PCB exposure were also studied in relation to disease susceptibility. Disease susceptibility was highest in the fed char with no difference due to PCB exposure, while disease susceptibility increased with PCB exposure in the starved group. The results indicate that PCBs reduce immunocompetence in starved Arctic char, but that starved fish are more disease resistant than fed fish.
- Glaucous gull chicks with high dietary PCB exposure show an impaired ability to produce antibodies when challenged with an antigen.
- Two juvenile harp seals treated with increasing doses of selected PCB congeners for 40 days and then fasted for 30 days had increased serum cortisol and aldosterone levels as well as tumor necrosis factor alpha compared to the controls.

Mutagenic effects

- Higher frequencies of chromosome aberrations and DNA adducts were found in glaucous gull chicks fed a POPs contaminated diet.

6.4.2. Assessment of current levels in biota

Current concentrations of some POPs in several Arctic species are at or above the known thresholds associated with effects that have been seen in other species studied either in the laboratory or in the field.

Canadian wolverines have Σ PCB levels that exceed those associated with subtle neurobehavioral effects in offspring of rhesus monkeys and humans. Mountain hare and reindeer from the Kola Peninsula (Lovozero) have PCDD/F concentrations, expressed as TEQs, which exceed levels associated with immunosuppression in harbour seals. TEQs in reindeer from Pechora and Taymir Dudinka also exceed this threshold.

Eggshell thinning in Alaskan peregrine falcon eggs has improved, but eggshells are still thinner than pre-DDT era eggs, indicating that present DDE levels are still causing effects. The *p,p'*-DDE concentrations in some Alaskan peregrine falcon eggs are just below the critical threshold for reproductive failure. Σ PCB levels in Alaskan peregrines exceed some NOELs and LOELs for reproductive endpoints in a wide range of wild bird species. Σ PCB concentrations in Norwegian peregrine falcons and white-tailed sea eagle exceed most or all thresholds for reproductive effects in other bird species. Norwegian gyrfalcon and golden eagles have lower Σ PCB concentrations, but these still exceed some thresholds for reproductive effects. Bald eagles from one of four studied Aleutian Islands have *p,p'*-DDE concentrations in the range known to cause reproductive impairment. Maximum Σ PCB concentrations in some Aleutian bald eagles exceed most of the thresholds for reproductive effects in other bird species including the LOAEL for egg mortality in bald eagles.

Σ PCB levels in Alaskan and Canadian burbot liver from some sites and in some Arctic char from Bjørnøya are close to or exceed the LOEL for induction of liver enzymes found in Arctic char. Toxaphene levels in some Canadian burbot exceed levels associated with effects on bone development found in channel catfish. In marine and anadromous fish, Σ PCB levels in Greenland shark from Cumberland Sound and Greenland halibut from West Greenland exceed the LOEL for induction of liver enzymes found in Arctic char.

Piscivorous seabirds such as grey herons, alcids, and kittiwakes have lower Σ PCB levels than predatory seabirds, and only grey herons had Σ PCB levels that exceeded some reproductive thresholds for effects. Black guillemots from the most contaminated site at Saglek Bay, Canada and predatory seabirds, such as glaucous and great black-backed gulls and great skuas, have Σ PCB levels in liver that indicate that egg levels might exceed some threshold levels for reproductive effects. For grey heron, mono-*ortho* PCB levels, expressed as TEQs, exceeded some reproductive thresholds in other bird species. TEQs based on PCDD/F and nPCB levels for Canadian thick-billed murrets, black-legged kittiwakes, and northern fulmars, and TEQs based on PCDD/Fs, nPCBs and mono-*ortho* PCBs for herring gull and great black-backed gull eggs from northern Norway exceed the LOAEL for reproductive effects in wood duck, a sensitive species. Glaucous gulls from Svalbard have nPCB and mono-*ortho* PCB levels expressed as TEQs that exceed all thresholds for reproductive effects in other bird species and the threshold for induction of liver enzymes found in common terns.

Ringed seals from Alaska, Canada, and Greenland, harbour seals from Alaska, Steller sea lions from Alaskan eastern stocks, harbour porpoises and walrus from Greenland, and grey whales from the Bering Sea, have Σ PCB levels that are low but that still exceed levels associated with subtle neurobehavioral effects in offspring of rhesus monkeys and humans. Higher Σ PCB levels are found in ringed seal from the Barents Sea/Kara Sea area and Saglek Bay, Labrador, harbour, grey, bearded and northern fur seals and some Steller sea lions. These Σ PCB levels also exceed the NOEL associated with decreased vitamin A in otter. Svalbard, Kara Sea, and Saglek Bay ringed seals and northern fur seals also exceed the threshold for decreased reproduction in otter, and sea otter from the Aleutian Islands. Some ringed seal (Kara Sea) and some beluga exceed the NOEL for mink kit survival. Harbour porpoises from Norway, long-finned pilot whales, narwhal, killer whales (residents and transients), harp seals, and some minke whales have even higher Σ PCB levels that are associated with decreased vitamin A in otter, and decreased otter and mink reproduction. Harp seals and some minke whales exceed Σ PCB levels associated with immunosuppression and vitamin A disruption in harbour seals but are below the threshold for immunosuppression in rhesus monkeys. In addition, some harbour porpoises from northern Norway, some resident killer whales and all transient killer whales from Alaska, and some long-finned pilot whales from the Faroe Islands have Σ PCB levels that exceed thresholds for immunosuppression in rhesus monkeys and poor reproductive success in harbour seal. Some resident and many transient killer whales exceed the EC₅₀

for litter size in mink, and many transient killer whales also exceed the threshold for poor reproductive success in ringed seal and the EC₅₀ for kit survival in mink.

Levels of mono-*ortho* PCBs, given as TEQs, in Arctic fox from the Pribilof Islands, and in resident and transient killer whales, exceed the mono-*ortho* TEQs associated with immunosuppressive effects in harbour seals.

Current ΣPCB levels in polar bear from Alaska, Canada, Greenland, the Iceland ice pack, Svalbard, and Russia indicate that these populations have levels that exceed those associated with subtle neurobehavioral effects in offspring of rhesus monkeys and humans, decreased vitamin A in otter, and NOELs for otter and mink reproduction. ΣPCB levels are highest in the Svalbard and Russian populations, and these levels exceed those associated with immunosuppression in harbour seals and rhesus monkeys and poor reproductive success in harbour seals. The results from biological marker studies, particularly immune system effects in Svalbard polar bears and observations of decreased cub survival, support this assessment.

Arctic fox from Barrow, Alaska, and inland Iceland have ΣPCB levels associated with subtle neurobehavioral effects in offspring of rhesus monkeys and humans. Arctic fox from the Pribilof Islands, Alaska, Canada, and coastal Iceland have ΣPCB levels associated with subtle neurobehavioral effects in offspring of rhesus monkeys and humans, decreased vitamin A in otter and reproductive effects in otter and mink. ΣPCB levels in Arctic fox from coastal Iceland also exceed those associated with immunosuppression in harbour seals and rhesus monkeys, poor reproductive success in harbour seals and the EC₅₀ for litter size in mink. Levels of mono-*ortho* PCBs, given as TEQs, in Pribilof Islands Arctic foxes exceed the mono-*ortho* TEQs associated with immunosuppressive effects in harbour seals.

Consequently, based on known thresholds for effects, several Arctic species appear to be at risk for, primarily, reproductive and/or immunosuppressive effects from current levels of ΣDDTs, ΣPCBs, and/or dioxin-like substances. Those at greatest risk include peregrine falcons, bald eagles, white-tailed sea eagles, glaucous and great black-backed gulls, great skuas, some alcids, harbour porpoises, seals, northern fur seals, Steller sea lions, belugas, long-finned pilot whales, narwhal, minke whales, killer whales, sea otters, polar bears, and Arctic fox. If the risk for subtle neurobehavioral effects from ΣPCBs in exposed offspring of mammals is included, then some reindeer and mountain hares, wolverines, walrus, and grey whales are also potentially at risk. Burbot from some sites have toxaphene levels high enough to affect bone development in fry. Burbot, land-locked char (Bjørnøya), Greenland shark, and Greenland halibut have ΣPCB levels associated with increased liver enzyme production in Arctic char. Some invertebrates are at risk for the reproductive effects of TBT, particularly those that are exposed in harbors.

An assessment of risks from dietary intake has been attempted based on results from laboratory feeding experiments, POP levels in prey items, and information on different species' food preferences. Reproduction in piscivorous marine mammals, such as minke whale, beluga, narwhal, long-finned pilot whale, harbour porpoise, and seals, may be affected by dietary levels of ΣPCBs in ma-

rine fish. Killer whales, walrus, polar bear, and Arctic fox that prey on seals have dietary intakes of ΣPCBs that may cause reproductive effects. When environmental quality guidelines for protecting aquatic wildlife are used, dietary intakes of ΣDDTs, ΣPCBs, and dioxin-like substances are problematic for many marine and freshwater piscivorous species and, in some cases, for molluscivores. The same is true for predatory seabirds feeding on seabird eggs and chicks, and for killer whale, walrus, polar bear, and Arctic fox that consume marine mammals.

6.4.3. Conclusions

Assessment of contaminant levels in the previous AMAP assessment suggested that several species were at risk for neurobehavioral, reproductive, and immune system effects, particularly polar bears, predatory birds, seabirds, and seals. This has been borne out in biological effects studies done on polar bears, northern fur seals, glaucous gulls and possibly also Steller sea lions. The implications of these findings are that there are other populations of these species, and other highly contaminated species, that are being affected by current levels of some POPs.

Effects that are biologically significant are those that affect resistance to infection, reproduction, and behavior. Anything that affects these negatively reduces the margin of safety for the affected species, putting them at higher risk. Biomarkers for POP effects measure changes at the cellular or individual level, and are warning signals. The results from biomarker studies in the Arctic have shown that there are associations between several biomarkers and concentrations of some POPs. Results from field experiments and laboratory studies give added weight to the possible link between some POPs and specific effects. Therefore, based on the present evidence, it is believed that effects of biological significance are occurring in some Arctic species related to POPs exposure. These effects of biological significance are:

- polar bears are at higher risk for infections due to immune effects of POPs;
- glaucous gulls with high POP levels are at higher risk of immune, behavioral and reproductive effects, and effects on adult survival;
- northern fur seals are at higher risk for infections due to immune dysfunction correlated to POP exposure;
- peregrine falcons continue to exhibit egg shell thinning and reproductive effects of POPs;
- Arctic char exhibit immune effects of PCB; and
- dogwhelks exhibit the reproductive effects of TBT.

Knowledge gaps, such as understanding the influence of confounding factors, indicate that other biomarkers studied in Arctic biota (thyroid hormones, vitamin A and cytochrome P450 activity) should be considered indicators of increased exposure. It is not yet possible to conclude that any documented changes in these biomarkers imply increased risk. Studies on the effects of other stressors, such as long periods of fasting, on PCB effects in Arctic char show that PCB exposure increases susceptibility.

The lack of experimental dosage/response data continues to limit the ability to interpret residues in Arctic animals.