ELSEVIER

The Science of the Total Environment 301 (2003) 163–174

the Science of the Total Environment
An International Journal for Scientific Research into the Environment and its Relationship with Man

www.elsevier.com/locate/scitotenv

Contaminants in Svalbard polar bear samples archived since 1967 and possible population level effects

Andrew E. Derocher[a,*], Hans Wolkers[a], Theo Colborn[b], Martin Schlabach[c], Thor S. Larsen[d], Øystein Wiig[e]

[a]*Norwegian Polar Institute, N-9296 Tromso, Norway*
[b]*Wildlife and Contaminants Program, World Wildlife Fund, 1250 24th Street NW, Washington, DC 20037, USA*
[c]*Norwegian Institute for Air Research, P.O. Box 100, N-2027 Kjeller, Norway*
[d]*UNEP/GRID, Longum Park, P.O. Box 706, N-4808 Arendal, Norway*
[e]*Zoological Museum, University of Oslo, P.O. Box 1172 Blindern, N-0318 Oslo, Norway*

Received 28 March 2002; accepted 15 July 2002

Abstract

Blood plasma samples were collected in 1967 from 32 polar bears (*Ursus maritimus*) in eastern Svalbard. These samples were stored frozen until 2001 and then analyzed for 33 polychlorinated biphenyls (PCB), two toxaphene congeners, DDTs, chlordanes (CHL), hexachlorobenzene, hexachlorocyclohexanes (HCHs), and polybrominated flame retardants (biphenyls and diphenyl ethers). The 1967 pollutant levels were compared with values from 1993 to 1994 for adult females and adult males to obtain insights into the historical development of pollution in the Norwegian Arctic. Differences in the OC levels measured between 1967 and 1993–1994 ranged from a decrease (PCB 187 and *p,p*-DDE) to unchanged in both sexes (PCBs 105, 118, 209, and HCH) to an increase in females (PCBs 99, 128, and CHL), to increases in both sexes (PCBs 138, 153, 156, 157, 170, 180, 194, and 206). The maximum change was a nine-fold increase in PCB 157 in adult females. Changes from 1967 to 1993–1994 in contaminant pattern expressed relative to PCB 153 could be explained by a combination of selective metabolism and accumulation of organochlorines in polar bears and temporal changes in the contaminant mixture being transported to the Arctic. Harvest of polar bears in Svalbard ended in 1973 and it appears that most pollutant levels were increasing at the same time that the population was expected to recover from over-harvest. The mean age of adult females in the Svalbard population was similar to other populations where pollution levels are lower but harvest is intense. Females with cubs-of-the-year $\geqslant$16 years old are uncommon in the population for unknown reasons. The impacts of contaminants on the Svalbard polar bear population are inconclusive but there are suggestions of contaminant-related population level effects that could have resulted from reproductive impairment of females, lower survival rates of cubs, or increased mortality of reproductive females.

Keywords: Archive; Arctic; Barents Sea; Organochlorines; Population level effects; Polychlorinated biphenyls; Polar bear; Svalbard; *Ursus maritimus*

*Corresponding author. Department of Biological Sciences, University of Alberta, Edmonton, Alta., Canada T6G 2E9.

0048-9697/03/$ - see front matter
PII: S0048-9697(02)00303-0

1. Introduction

The presence of persistent organic pollutants in Arctic marine ecosystems, and particularly in marine mammals, has been known for over 30 years (Holden, 1970; AMAP, 1998). Trend information is lacking or confounded for many marine mammals due to lack of standardized sampling methods, lack of sex and age class control of samples, dietary and population changes, and different analytical methods (AMAP, 1998; Muir et al., 1999). Polar bears (*Ursus maritimus*) were among the first large mammals in the Arctic known to be contaminated by polychlorinated biphenyls (PCBs) (Bowes and Jonkel, 1975). Studies on geographic variation in PCBs suggested that NE Greenland and Svalbard, Norway were the most polluted polar bear populations in the world (Norstrom et al., 1998; Muir and Norstrom, 2000). More recently, the most polluted polar bears were found in western Russia (Andersen et al., 2001).

As a result of the high pollution levels, the effects of pollution have been intensively studied in the Svalbard population. Studies on the trends of PCBs in polar bears in Svalbard during the 1990s suggest that levels declined or stabilized in the mid-1990s (Henriksen et al., 2001). However, information on pollution levels in polar bears in Svalbard before 1990 are unavailable.

Documentation of historical pollution exposure in the Svalbard polar bears is necessary to understand how the population may have been affected over time. Polar bears in Svalbard are part of a shared population with Russia (Larsen, 1986; Mauritzen et al., 2002). In response to concerns about over-harvest of polar bears and population depletion, hunting was banned in 1956 in Russia and in 1973 in Svalbard (Prestrud and Stirling, 1994). Therefore, the main ongoing anthropogenic impact on polar bears in the Norwegian Arctic was from contaminants. Polar bears have a long-life span but a low reproductive rate and usually do not exceed an annual population growth rate of approximately 2–5% (Taylor et al., 1987) so recovery from over-harvest is expected to be slow. Cessation of the harvest appeared to result in a population increase in the early 1980s but older females were noticeably absent from the population both in the late 1970s and early 1980s (Larsen, 1986; Wiig, 1998). Missing older females and high juvenile mortality were also reported in the Svalbard polar bear population in the 1990s (Wiig, 1998) and these anomalies were suggested to be linked to pollutants (Bernhoft et al., 1997, 2000).

In this paper, we report on levels of organochlorine pollutants in polar bear blood samples collected in 1967 and archived until analysis in 2001 and we compare these results to published values from 1993 to 1994. We also compare the age structure of polar bears in Svalbard to other less polluted areas to assess possible population level effects.

2. Methods

Polar bears were caught by chemical immobilization (Larsen, 1971; Stirling et al., 1989). Blood was collected from the femoral vein and stored in heparinized tubes for studies of genetics and pollution (Larsen et al., 1983; Bernhoft et al., 1997). Plasma was separated from the whole blood by centrifugation or by sedimentation over 24 h. The plasma was stored in glass tubes, thoroughly sealed, and frozen. We believe the samples were exposed to some variation in temperature over the first 26 years of storage in different freezers but the details are uncertain. From 1994 to 2001, the storage was below −20 °C. The amount of plasma available for each bear was approximately 1 ml. Ages of bears were determined by counts of annuli from a premolar tooth (Calvert and Ramsay, 1998). We pooled samples by sex and age to create three groups: subadults (0.5–2.5 years old of both sexes), adult females (4–15 years of age), and adult males (5–18 years of age). The age groups were selected because of the samples available from 1967. However, the age groups are biologically meaningful because bears up to 2.5 years of age are still with their mother and the ages for adult females and adult males correspond approximately with the age of sexual maturity meaning both groups should be reproductively active (Wiig, 1998). Samples were unavailable for bears 3 or 4 years old which are often missed in capture programs. We used ages ($\log_{10}$ transformed) of cap-

tured bears up to 2000 and published studies to examine population structure.

3. Chemical analyses

3.1. Extraction and clean-up

The plasma samples were extracted using a liquid–liquid extraction with ethanol, *n*-hexane and deionised water saturated with ammonium sulphate according to methods developed at Centre de Toxicologie du Quebec, Canada (Romieu et al., 2000). Internal standards (^{13}C-labeled PCBs, hexachlorocyclohexanes (HCHs), DDE and DDT) were added before the first extraction. Two millilitres of plasma, 2 ml of ethanol and 2 ml of deionised water saturated with ammonium sulfate was extracted three times with 6 ml of *n*-hexane in a small glass tube. The organic fraction was collected and isooctane was added as a keeper before evaporation to 0.5 ml. The extract was purified on two florisil columns arranged in tandem (1.5 g florisil 0.5% deactivated topped with 2 g anhydrous Na_2SO_4). The contaminant fraction was eluted with 9 ml dichloromethane: *n*-hexane (1:3) and reduced to a volume of 25–100 μl using a gentle stream of nitrogen.

3.2. Quantification

The separation of the compounds was performed by high-resolution gas chromatography on a Hewlett-Packard HP6980 or HP5890 II with splitless injection of 1 μl aliquot of the sample extract and helium as the carrier gas. A Hewlett-Packard HP5973 MSD low-resolution mass spectrometer running in the electron capture negative ion mode (LRMS-ECNI) was used for detection and quantification of the chlorinated pesticides (with exception of the DDT-group) and the brominated flame retardants. A Micromass AutoSpec (formerly VG Analytical AutoSpec) high-resolution mass spectrometer (resolution $>10\,000$) was used for detection and quantification of the PCB congeners and the DDT group. Quantification was based on the following ^{13}C-labelled analogues: hexachlorobenzene (HCB), PCBs 28, 52, 101, 118, 153 and 180, α-HCH, γ-HCH, *p*,*p*′-DDE and *p*,*p*′-DDT. Compounds without a ^{13}C-labelled analogue in the internal standard mixture were quantified using the closest eluting ^{13}C-labelled compound in the respective GC/MS runs. We quantified 33 PCB congeners but nine congeners (PCBs 18, 31, 37, 47, 60, 66, 114, 122, and 123) were below the detection limit. We also quantified two toxaphene congeners (Parlar 26 and 50), DDTs, chlordanes (CHL), HCB, HCHs, and polybrominated flame retardants (biphenyls and diphenyl ethers) (see Table 2 for details).

3.3. Quality control

Rigorous quality control was based on the recommendations of the Arctic Monitoring and Assessment Program (AMAP, 1998) and on the requirements in the international norm for laboratory accreditation ISO/IEC 17025. The quality of the methods used was verified regularly in international inter-calibrations. The use of isotopically labeled internal standards for quantification and the frequent control of complete method blank values insured high quality analytical results. Blank values were not subtracted. The measurement uncertainties of the analytical methods were estimated to be $\pm 20\%$ for PCBs, ± 20–30% for pesticides, and $\pm 40\%$ for the brominated flame retardants.

3.4. Comparison of pollutant levels between 1967 and 1993–1994

Pollutant levels from 1967 were compared to samples collected in spring 1993–1994 to obtain insight into the degree of change of compounds and accumulation patterns. The pollutant data in the 1993–1994 samples were previously reported (Bernhoft et al., 1997, 2000; Skaare et al., 2001; Haave, 2001; Olsen, 2001; Henriksen et al., 2001) and details on analyses of these samples can be obtained within these publications.

Although the analyses of the 1967 samples were done 34 years after collection, we were certain we could gain some insight on trends because of the high quality of present day technology. The 1967 samples were still sealed and were frozen the entire time. Differences between 1967 and 1993–

1994 pollutant levels and changes in contaminants relative to PCB 153 (ln transformed) were tested using a *t*-test. Differences were considered significant at $P \leqslant 0.05$.

4. Results

Plasma samples from 1967 were available from nine subadult bears (0.5–2.5 years old, three females and six males), nine adult females (4–15 years old), and 14 adult males (5–18 years old). From 1993 to 1994, data were available for 24 adult males and 21 adult females with an age distribution similar to the 1967 samples. Levels of contaminants were low in 1967 (Tables 1 and 2), with brominated flame retardants (polybrominated biphenyl congeners 15, 52, and 153; polybrominated diphenyl ethers congeners 47 and 99) and toxaphenes below the detection limit. Only PCB 187 and *p,p*-DDE were significantly lower in 1993–1994 than in 1967 (Fig. 1). Levels of PCBs 105, 118, 209, and HCH were not significantly different in the two periods for both sexes. The other compounds showed overall higher values in 1993–1994 (from roughly two to nine times increase) that were significant in females only for PCBs 99, 128 and CHL, and for both sexes for all other compounds (Table 3 and Fig. 1). The highest increase took place in female polar bears for PCBs 138, 153, 157, 170, 180 and 194 and CHL. PCBs 99, 138, 153, 170 and 180 dominated in both sampling periods.

For both male and female bears, a decrease in relative presence, expressed as a percentage of PCB 153, between 1967 and 1993–1994 occurred for PCBs 99, 105, 118, 128, 156, 187, 206 and 209 (Fig. 2). Similarly, *p,p*-DDE and HCHs showed a decrease in relative presence in 1993–1994. In contrast, the relative presence of PCBs 170 and 194 was higher in 1993–1994 (Fig. 2). The relative presence of CHL increased substantially between the two sampling periods only in females. Likewise, females showed a higher prevalence of these compounds as compared to males in both sampling periods (Fig. 2).

5. Population structure

The mean age of adult females $\geqslant 4$ years old (reproductively mature) was 10.9 years (S.E. = 0.3, $n = 357$) in the Svalbard area in 1987–2000. We compared the mean age of Svalbard females (1987–2000) to the mean age of both adult females and males harvested and captured from the Svalbard area in 1954–1980 ($\bar{x} = 8.1$ years, S.E. = 0.2, $n = 384$) (Larsen, 1986), adult females in the Beaufort Sea ($\bar{x} = 10.2$ years, S.E. = 0.4, $n = 194$) (Stirling et al., 1988), adult females in the northern Beaufort Sea ($\bar{x} = 14.3$ years, S.E. = 0.9, $n = 46$) (Lunn et al., 1995), and adult females in western Hudson Bay ($\bar{x} = 11.5$ years, S.E. = 0.20, $n = 802$) (Derocher, 1991; Derocher and Stirling, 1995) and found significant variation between the populations (ANOVA, $F_{4,1778} = 34.6$, $P < 0.001$). The mean age of adult females in the Svalbard population was only significantly different from the northern Beaufort Sea and the 1954–1980 Svalbard male and female ages (GT2 test). There was a general increase in the age of adult females over 1987–2000 (linear regression, $F_{1,354} = 26.6$, $P < 0.001$) in the Svalbard population and this can be seen in the greater representation of older females in 1994–2001 (Fig. 3).

In the Svalbard population (1987–2000), the age distribution by age groups for females with cubs-of-the-year were 50.9% (56/110) 5–9 years old, 36.4% (40/110) 10–15 year olds, 11.8% (13/110) 16–21 year olds, and 0.9% (1/110) $\geqslant 22$ years old. In contrast, in western Hudson Bay, the same age groups (1980–1992) had 27.4% (72/263), 32.3% (85/263), 31.9% (84/263), and 8.4% (22/263) of the females with cubs-of-the-year, respectively (Derocher and Stirling, 1995). Comparing the Svalbard to the western Hudson Bay population, the proportion of females with cubs-of-the-year $\geqslant 16$ years old in Svalbard (12.7%) was significantly lower (G-test, $P < 0.001$) than western Hudson Bay (40.3%). Females $\geqslant 16$ years old with cubs-of-the-year comprised 25% (4/16) of the females with cubs-of-the-year in the northern Beaufort Sea (Lunn et al., 1995) and 16% (5/31) in the Beaufort Sea (Stirling et al., 1988) but neither population was

Table 1
Geometric mean and 95% confidence limits of polychlorinated biphenyl (PCB) concentrations (ng/g wet weight) in blood plasma of adult males, adult females, and subadult polar bears in 1967 and 1993–1994

	PCB	28	33	52	74	99	101	105	118	128	138	141	149	153	156	157	167	170	180	183	187	189	194	206	209
Adult males 1967 ($n=24$)	$\bar{x}$	0.8	0.8	0.4	0.2	1.9	0.2	0.2	0.5	0.1	1.8	0.0	0.0	8.2	0.5	0.2	0.0	2.1	5.1	0.2	0.1	0.1	1.0	0.5	0.3
	−95%	0.7	0.6	0.3	0.2	1.5	0.1	0.2	0.4	0.1	1.4	0.0	0.0	6.7	0.4	0.2	0.0	1.6	4.1	0.1	0.1	0.1	0.7	0.4	0.2
	+95%	1.1	1.0	0.6	0.3	2.4	0.3	0.3	0.7	0.1	2.3	0.0	0.0	10	0.6	0.3	0.0	2.7	6.3	0.2	0.2	0.1	1.4	0.6	0.4
Adult males 1993–1994 ($n=24$)	$\bar{x}$	–	–	–	–	2.5	–	0.1	0.3	0.1	4.7	–	–	23.3	0.8	0.9	–	8.7	13.4	–	0.0	–	4.8	0.7	0.3
	−95%	–	–	–	–	1.6	–	0.1	0.2	0.1	3.5	–	–	17.5	0.7	0.7	–	6.9	10.2	–	0.0	–	3.8	0.6	0.2
	+95%	–	–	–	–	3.9	–	0.2	0.4	0.2	6.4	–	–	31.1	1.0	1.1	–	10.9	17.6	–	0.1	–	6.1	0.8	0.5
Adult females 1967 ($n=9$)	$\bar{x}$	0.6	0.6	0.3	0.2	1.5	0.2	0.2	0.5	0.1	1.7	0.0	0.0	5.9	0.3	0.1	0.0	1.2	3.4	0.2	0.1	0.0	0.5	0.4	0.3
	−95%	0.4	0.4	0.2	0.2	1.2	0.1	0.2	0.4	0.1	1.4	0.0	0.0	4.8	0.2	0.1	0.0	1.0	2.7	0.1	0.1	0.0	0.3	0.3	0.2
	+95%	1.0	0.9	0.5	0.3	1.7	0.2	0.2	0.6	0.1	2.0	0.0	0.0	7.1	0.3	0.1	0.0	1.6	4.3	0.2	0.2	0.1	0.7	0.5	0.4
Adult females 1993–1994 ($n=21$)	$\bar{x}$	–	–	–	–	5.0	–	0.1	0.3	0.2	6.6	–	–	30.5	1.0	0.9	–	7.8	13.5	–	0.1	–	4.1	0.7	0.4
	−95%	–	–	–	–	3.9	–	0.1	0.2	0.1	5.5	–	–	24.7	0.8	0.6	–	6.0	10.5	–	0.0	–	3.3	0.6	0.3
	+95%	–	–	–	–	6.4	–	0.2	0.5	0.2	8.0	–	–	37.8	1.2	1.1	–	10.0	17.5	–	0.1	–	5.1	0.8	0.5
Subadults 1967 ($n=9$)	$\bar{x}$	0.7	0.6	0.3	0.2	3.9	0.1	0.2	0.5	0.1	3.2	0.0	0.0	15.4	0.6	0.3	0.0	3.4	7.6	0.3	0.1	0.1	0.8	0.4	0.3
	−95%	0.4	0.4	0.2	0.1	2.9	0.1	0.1	0.4	0.1	2.3	0.0	0.0	11.4	0.5	0.2	0.0	2.5	5.9	0.2	0.1	0.1	0.7	0.4	0.2
	+95%	1.2	1.1	0.5	0.3	5.3	0.1	0.2	0.6	0.2	4.3	0.0	0.0	20.6	0.8	0.4	0.0	4.5	9.9	0.4	0.2	0.1	1.0	0.6	0.3

Table 2
Geometric mean concentrations (ng/g wet weight) and 95% confidence limits of pesticides in blood plasma of adult male, adult female and subadult polar bears captured in 1967

		o,p-DDT	*p,p*-DDT	*o,p*-DDE	*p,p*-DDE	*o,p*-DDD	*p,p*-DDD	ΣDDT	*c*-CHL	*t*-CHL	*c*-nona	*t*-nona	Oxy	Hept	ΣCHL	α-HCH	β-HCH	γ-HCH	ΣHCH
Adult males 1967 ($n=24$)	$\bar{x}$	0.0	0.6	0.0	1.7	0.0	0.4	2.8	0.0	0.0	0.0	0.3	1.3	0.1	1.8	0.8	1.4	0.1	2.5
	− 95%	0.0	0.4	0.0	1.2	0.0	0.2	1.9	0.0	0.0	0.0	0.2	1.0	0.1	1.4	0.6	1.2	0.1	2.1
	+ 95%	0.0	0.8	0.0	2.6	0.0	0.7	4.3	0.0	0.0	0.0	0.5	1.8	0.1	2.4	1.1	1.8	0.2	3.1
Adult females 1967 ($n=9$)	$\bar{x}$	0.0	0.2	0.0	1.4	0.0	0.5	2.2	0.0	0.0	0.0	0.2	1.5	0.1	2.0	0.5	0.6	0.1	1.3
	− 95%	0.0	0.1	0.0	0.9	0.0	0.3	1.6	0.0	0.0	0.0	0.2	1.1	0.1	1.6	0.5	0.5	0.1	1.1
	+ 95%	0.0	0.4	0.0	2.0	0.0	0.8	3.2	0.0	0.0	0.0	0.4	2.0	0.2	2.6	0.7	0.8	0.1	1.5
Subadults 1967 ($n=9$)	$\bar{x}$	0.0	0.1	0.0	1.6	0.0	0.3	2.3	0.0	0.0	0.0	0.4	4.6	0.1	5.2	0.8	2.0	0.1	2.9
	− 95%	0.0	0.1	0.0	1.1	0.0	0.2	1.6	0.0	0.0	0.0	0.3	3.1	0.1	3.6	0.6	1.5	0.1	2.2
	+ 95%	0.0	0.4	0.0	2.4	0.0	0.5	3.2	0.0	0.0	0.0	0.5	6.8	0.2	7.6	1.0	2.7	0.1	3.7

DDT, dichlorodiphenyltrichloro ethane; DDE, dichlorodiphenyldichloro-ethylene; DDD, dichlorodiphenyldichloroethane; ΣDDT, total DDT levels; *c*, cis; *t*, trans; CHL, chlordane; nona, nonachlor; oxy, oxychlordane; hepta, heptachlorepoxide; ΣCHL, total chlordane levels; HCH, hexachlorocyclohexane; ΣHCH, total HCH levels.

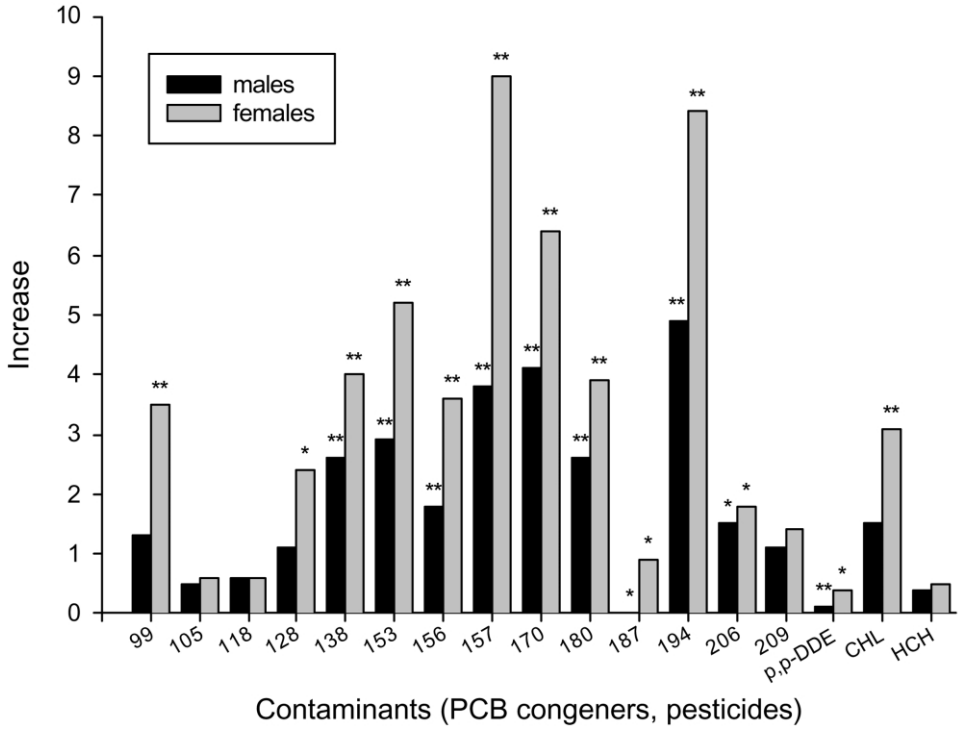

Fig. 1. Relative increase of PCB congeners and pesticides from 1967 to 1993–1994 for adult female and male polar bears sampled at Svalbard, Norway. Significant increases (*$P \leqslant 0.05$, **$P \leqslant 0.001$) are indicated above each bar.

significantly different from Svalbard (G-test, $P > 0.22$).

6. Discussion

The archived samples used in this study were available more by chance than good planning. The 1967 samples were originally collected for a study of protein and genetic variability in polar bears (Larsen et al., 1983). To understand the dynamics of pollution in the Arctic, we believe it is imperative that an appropriate archive system be established where samples can be properly stored for future analyses along with detailed information on their background and handling. Such a system is currently lacking for the Norwegian Arctic.

The lower concentrations for a few compounds in 1993–1994 compared to 1967 could be due to differences in contaminant mixtures reaching the Arctic over time. The increases in most PCBs and CHL, between 1967 and 1993–1994 can be attributed to their increased use in the 1950s and 1960s and their transport into the Arctic (AMAP, 1998). It was not until the early to mid-1970s that both PCBs and DDTs were banned in Europe. The low concentrations of many contaminants in the polar bear samples from 1967 suggest the bulk of contaminants had not yet reached the Arctic. Long-range transport brings these persistent compounds to the arctic environment where they enter the food chain. In 1967, most of the compounds studied were already in use for many years, however, their transport rates to the Arctic were likely increasing. The lag-time to the Arctic depends on many factors, including the distance from the source areas, transport routes (i.e. water currents and atmosphere), and the chemical properties of the pollutant. Similar to our results, PCB levels in eastern Canadian Arctic polar bears increased two-fold and CHL increased fourfold between 1969 and 1984 (Norstrom et al., 1988; Muir and Norstrom, 2000). Our results suggest that most con-

Table 3
Comparison of ΣPCB, *p,p*-DDE, ΣCHL, and ΣHCHs (ng/g wet weight) in adult male and adult female polar bears sampled in 1967 and in 1993–1994 in Svalbard, Norway

	ΣPCB[a]		*p,p*-DDE		ΣCHL[b]		ΣHCH[b]	
	1967	1993–1994	1967	1993–1994	1967	1993–1994	1967	1993–1994
Adult males								
$\bar{x}$	23.3	56.6	1.7	0.2	3.7	5.6	5.2	2.0
−95%	19.0	43.5	1.2	0.1	2.9	3.9	4.8	1.6
+95%	28.6	73.7	2.6	0.3	4.8	8.0	5.6	2.6
Adult females								
$\bar{x}$	17.6	67.7	1.4	0.6	2.0	17.6	1.3	1.9
−95%	13.8	55.1	0.9	0.4	1.6	13.9	1.1	1.3
+95%	22.4	83.1	2.0	0.8	2.6	22.4	1.5	2.8

[a] ΣPCB = IUPAC 99, 105, 118, 128, 138, 153, 156, 157, 170, 180, 187, 194, 206, 209.
[b] See Table 2 legends.

taminants continued to increase in Svalbard into the early 1990s. Subsequently, some contaminants have declined or stabilized in the Arctic in the 1990s (Muir et al., 1999; Muir and Norstrom, 2000; Henriksen et al., 2001).

Generally, the highest increase between 1967 and 1993–1994 took place for the more persistent, higher chlorinated PCBs. This augmentation is a function of both an increase in the environment, i.e. exposure through the food of the polar bears,

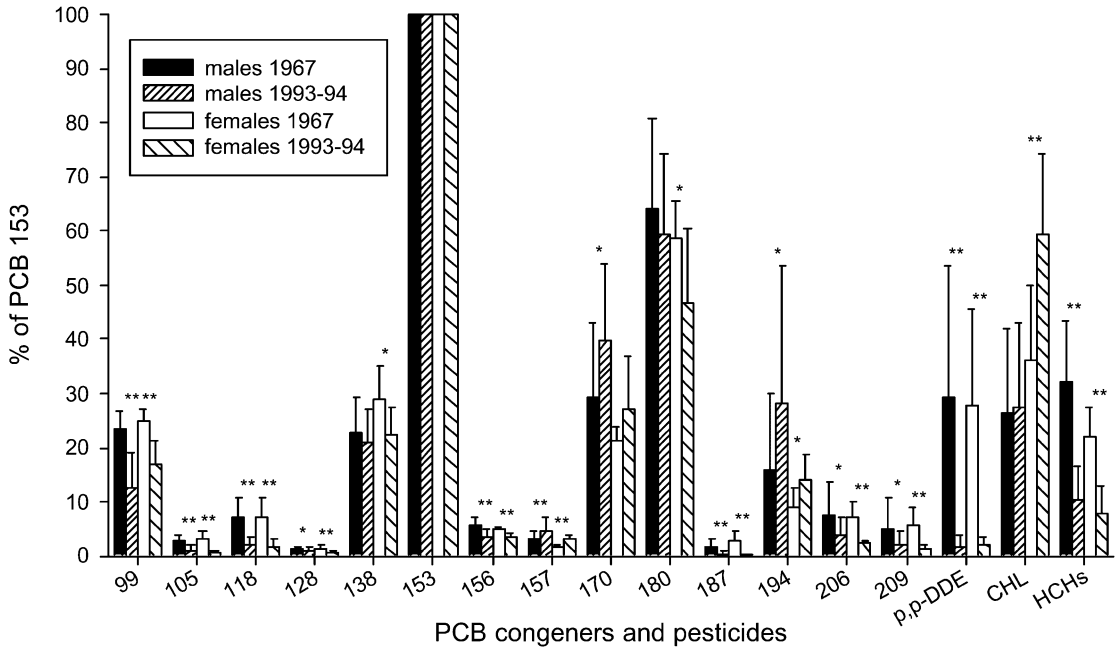

Fig. 2. PCB congeners and pesticides expressed as a percentage of PCB 153 from adult female and male polar bears captured in 1967 and in 1993–1994. Error bars indicate +standard deviation. Results of statistical testing ($*P \leqslant 0.05$, $**P \leqslant 0.001$) of the two sampling periods, by sex, is indicated above each pair of bars.

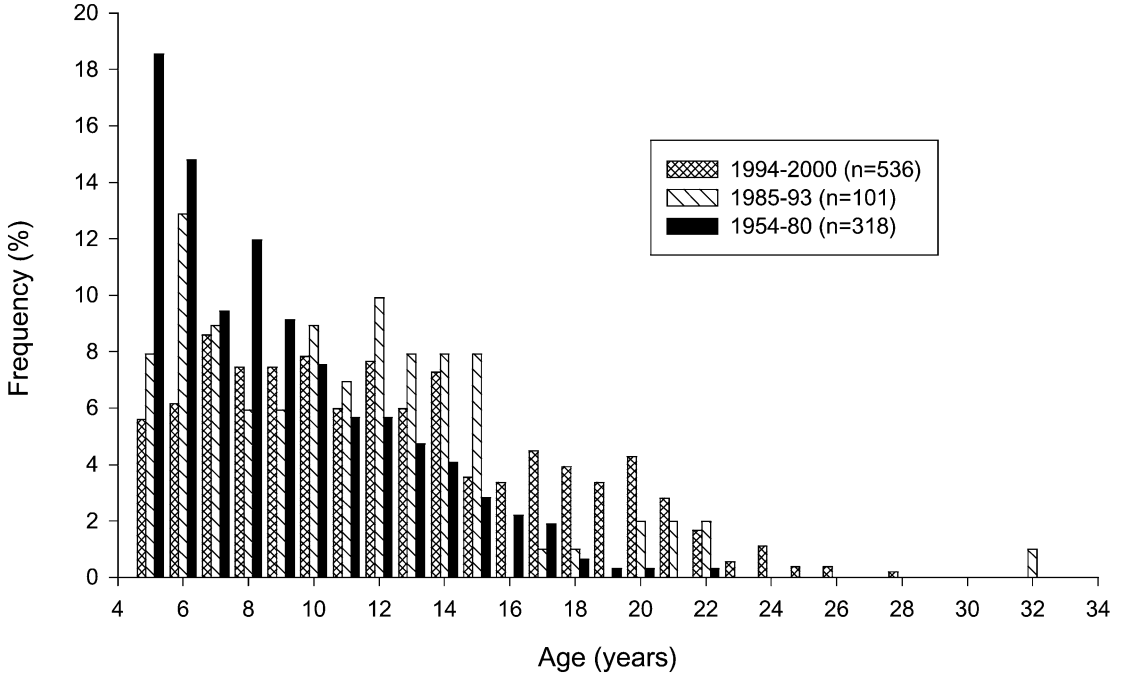

Fig. 3. Age structure of female and male polar bears (5+ years old) from Svalbard in 1954–1980, 1985–1993, and 1994–2000.

and contaminant metabolism by specialized enzymes from the cytochrome P450 enzyme system (Bandiera et al., 1997). The activity of these enzymes increases in response to contaminant exposure and accelerates contaminant metabolism (Goksøyr, 1995; Wolkers et al., 1998). As a result, the pattern of contaminants accumulated in the bears has changed with a relative decrease of the less persistent compounds. Changes between sampling periods of contaminants relative to PCB 153 suggests differential metabolism and accumulation of PCBs and pesticides in polar bears. However, these findings are possibly biased due to differences in contaminant exposure between sampling periods (AMAP, 1998). A substantial reduction in the relative presence of *p*,*p*-DDE confirms earlier findings of the unique ability of polar bears to metabolize this DDT metabolite (Norstrom et al., 1988; Polischuk et al., 2002), however, an earlier reduction in DDT use compared to PCB use may have confounded the results. The relatively large increase of CHL in females as compared to males, in addition to a strong increase in the relative presence in females, suggests a sex-based difference in the metabolism of this compound group. A marked reduction in CHL body burdens after fasting was found in male polar bears (Polischuk et al., 2002) and supports the idea of sex-specific CHL metabolism.

There are suggestions that polar bears in Svalbard have both an impaired immune system (Bernhoft et al., 2000) and endocrine system (Skaare et al., 2001) but to date, there is no clear indication of population level effects from contaminants. Studies suggest that the Svalbard population began recovering from over-harvest after 1973 (Larsen, 1986; Wiig, 1998; Fig. 3). Results suggest two demographic issues of concern relative to population level effects: the relative scarcity of bears $\geqslant$16 years old and the associated scarcity of females of this age group with cubs-of-the-year. Over a period of 28 years, a polar bear population with a 4% annual growth rate should increase approximately three-fold (1.04^{28}) with no harvest and should have permitted sufficient time for older females ($\geqslant$16 years old) to be well represented in

the population. The gradual increase in representation of older bears in the population in the 1990s suggests that the recovery was slower than expected. In general, harvest tends to reduce the mean age of large mammal populations by reducing survival rates. In western Hudson Bay, the mean age of females in the population increased after 1966 when harvest controls were introduced (Derocher et al., 1997). However, very few females $\geqslant$16 years old were found in Svalbard during research conducted in 1988–1993 and the cause of this was unknown (Wiig, 1998). While older females are increasing in the population (Fig. 3) they still appear under-represented compared to other populations. The mean age of Svalbard females was significantly lower than females from the northern Beaufort Sea population where harvest is limited (Lunn et al., 1995) and more similar to two Canadian populations managed for maximum sustainable harvest (Derocher et al., 1998). The reasons for the similarity with the harvest populations are unclear particularly given the lack of harvest in Svalbard. The reasons for the large difference in the proportion of females $\geqslant$16 years of age with cubs-of-the-year between the Svalbard and western Hudson Bay population are also unknown. The lack of differences between Svalbard and the Beaufort Sea and the northern Beaufort Sea populations may be related to small sample sizes in these two populations and further comparisons are warranted as data becomes available. It is unlikely that the Svalbard population is still recovering from over-harvest after 28 and 45 years of protection in Norway and Russia, respectively. However, lack of population estimates precludes assessment of population trend. It is possible that harvest patterns in western Hudson Bay have shifted the reproducing population towards older ages classes but this is unclear. An alternative hypothesize is that the Svalbard population has suffered population level effects from the higher levels of contaminants. Several studies have suggested altered hormone levels, impaired immune system, abnormalities, and lower cub survival in polar bears as a result of contaminants (Skaare et al., 2001; Bernhoft et al., 2000; Wiig et al., 1998; Polischuk et al., 1995, 2002). We believe it is possible that the Svalbard population had contaminant-related interference with reproduction and/or survival rates. For example, if contaminants lowered either the cub production rate or the survival rate of cubs in the 1970s and 1980s, then lowered recruitment into the population would be expressed by the scarcity of older females ($\geqslant$16 years of age) in the late 1980s and 1990s. Alternatively, reproducing females may be at greater risk of contaminant-related mortality when they undergo prolonged fasts and deplete their adipose stores (Watts and Hansen, 1987) and mobilize associated contaminants (Polischuk et al., 2002). However, additional factors must be considered because polar bears in Svalbard are exposed to *Brucella* which can cause reproductive failure in some species (Tryland et al., 2001). If the immune system of polar bears is impaired by contaminants there is also a possibility for increased disease effects.

A serious difficulty in assessing population level effects of contaminants on demography is that large amounts of data are required to obtain sufficient resolution. Further, a meaningful population for comparison is needed and virtually all populations studied, with the exception of Svalbard, undergo some level of harvest and are exposed to varying levels of pollution. While it is tempting to speculate on the significance of the differences between populations, further research is required before conclusions can be drawn. Following individual females to assess reproductive success and monitoring cub survival over time relative to pollution loads are obvious areas to pursue. Our pollutant levels from 1967 make it clear that the contaminant levels were increasing in the polar bears through the 1970s and 1980s when the population should have been recovering from overharvest and there are suggestions of population level effects that warrant further investigation.

Acknowledgments

This study was made possible by funding from World Wildlife Fund, USA. Additional funding was provided by NILU's strategic institute program. Samples from bears captured in 1993–1994

were analyzed by J.U. Skaare and E. Lie, Veterinary Institute, Oslo, Norway with funding provided by the Norwegian Research Council.

References

AMAP. AMAP assessment report: Arctic pollution issues. Arctic Monitoring and Assessment Programme, Oslo, 1998.

Andersen M, Lie E, Derocher AE, Belikov SE, Boltunov AN, Garner GW, Skaare JU, Wiig Ø. Geographic variation in selected PCB congeners in polar bears (*Ursus maritimus*) from Svalbard east to the Chukchi Sea. Polar Biol 2001;24:231–238.

Bandiera SM, Torok SM, Letcher RJ, Norstrom RJ. Immunoquantitation of cytochromes P450 1A and P450 2B and comparison with chlorinated hydrocarbon levels in archived polar bear liver samples. Chemosphere 1997;34:1469–1479.

Bernhoft A, Skaare JU, Wiig Ø, Derocher AE, Larsen HJS. Possible immunotoxic effects of organochlorines in polar bears (*Ursus maritimus*) at Svalbard. J Toxicol Environ Health 2000;59(Part A):561–574.

Bernhoft A, Wiig Ø, Skaare JU. Organochlorines in polar bears (*Ursus maritimus*) at Svalbard. Environ Pollut 1997;95:159–175.

Bowes GW, Jonkel CJ. Presence and distribution of polychlorinated biphenyls (PCB) in arctic and subarctic marine food chains. J Fish Res Board Can 1975;32:2111–2123.

Calvert W, Ramsay MA. Evaluation of age determination of polar bears by counts of cementum growth layer groups. Ursus 1998;10:449–453.

Derocher AE. Population dynamics and ecology of polar bears in western Hudson Bay. Edmonton, Alberta: University of Alberta, 1991.

Derocher AE, Garner GW, Lunn NJ, Wiig Ø. Polar bears. Proceedings of the 12th Working Meeting of the IUCN Polar Bear Specialist Group, 1998, Gland, Switzerland and Cambridge, UK, IUCN. Occasional Paper of the IUCN Species Survival Commission No. 19.

Derocher AE, Stirling I. Temporal variation in reproduction and body mass of polar bears in western Hudson Bay. Can J Zool 1995;73:1657–1665.

Derocher AE, Stirling I, Calvert W. Male-biased harvesting of polar bears in western Hudson Bay. J Wildl Manage 1997;61:1075–1082.

Goksøyr A. Cytochrome P450 in marine mammals: isozyme forms, catalytic functions, and physiological regulations. In: Blix AS, Walløe L, Ultang Ø, editors. Whales, seals, fish and man. Amsterdam: Elsevier, 1995. p. 629–639.

Haave M. Variation in plasma progesterone and estradiol of female polar bears (*Ursus maritimus*) in relation to reproductive status and concentrations of polychlorinated biphenyls (PCBs). Trondheim, Norway: Norwegian University of Science and Technology, 2001.

Henriksen EO, Derocher AE, Gabrielsen GW, Skaare JU, Wiig Ø. Monitoring PCBs in polar bears: lessons learned from recent data. J Environ Monit 2001;3:493–498.

Holden AV. Monitoring organochlorine contamination of the marine environment by the analysis of residues in seals. In: Ruivo M, editor. Marine pollution and sea life FAO. Surrey, UK: Fishing News Books, 1970. p. 266–272.

Larsen T. Capturing, handling, and marking polar bears in Svalbard. J Wildl Manage 1971;35:27–36.

Larsen T. Population biology of the polar bear (*Ursus maritimus*) in the Svalbard area. Norsk Polarinstitutt Skr 1986;184:1–55.

Larsen T, Tegelström H, Kumar Juneja R, Taylor MK. Low protein variability and genetic similarity between populations of the polar bear (*Ursus maritimus*). Polar Res 1983;1:97–105.

Lunn NJ, Stirling I, Andriashek D. Movements and distribution of polar bear in the Northeastern Beaufort Sea and western M'Clure Strait. Edmonton, Alberta: Canadian Wildlife Service, 1995.

Mauritzen M, Derocher AE, Wiig Ø, Belikov SE, Boltunov A, Hansen E, Garner GW. Using satellite telemetry to define spatial population structure in polar bears in the Norwegian and western Russian Arctic. J Appl Ecol 2002;39:79–90.

Muir D, Braune B, DeMarch B, Norstrom R, Wagemann R, Lockhart L, Hargrave B, Bright D, Addison R, Payne J, Reimer K. Spatial and temporal trends and effects of contaminants in the Canadian Arctic marine ecosystem: a review. Sci Total Environ 1999;230:83–144.

Muir DCG, Norstrom RJ. Geographical differences and time trends of persistent organic pollutants in the Arctic. Toxicol Lett 2000;112–113:93–101.

Norstrom RJ, Belikov SE, Born EW, Garner GW, Malone B, Olpinski S, Ramsay MA, Schliebe S, Stirling I, Stishov MS, Taylor MK, Wiig Ø. Chlorinated hydrocarbon contaminants in polar bears from eastern Russia, North America, Greenland and Svalbard. Arch Environ Contam Toxicol 1998;35:354–367.

Norstrom RJ, Simon M, Muir DCG, Schweinsburg RE. Organochlorine contaminants in Arctic marine food chains: identification, geographical distribution and temporal trends in polar bears. Environ Sci Technol 1988;22:1063–1071.

Olsen GH. Space-use strategies affect PCB levels in female polar bears (*Ursus maritimus*) in the Norwegian Arctic. Trondheim, Norway: Norwegian University of Science and Technology, 2001.

Polischuk SC, Letcher RJ, Norstrom RJ, Ramsay MA. Preliminary results of fasting on the kinetics of organochlorines in polar bears (*Ursus maritimus*). Sci Total Environ 1995;160/161:465–472.

Polischuk SC, Norstrom RJ, Ramsay MA. Body burdens and tissue concentrations of organochlorines in polar bears (*Ursus maritimus*) vary during seasonal fasts. Environ Pollut 2002;118:29–39.

Prestrud P, Stirling I. The international polar bear agreement and the current status of polar bear conservation. Aquat Mamm 1994;20:113–124.

Romieu I, Hernandez-Avila M, Lazcano-Ponce E, Weber JP, Dewailly E. Breast cancer, lactation history, and serum organochlorines. Am J Epidemiol 2000;152:363–370.

Skaare JU, Bernhoft A, Wiig Ø, Norum KR, Haug E, Eide DM, Derocher AE. Relationships between plasma levels of organochlorines, retinol and thyroid hormones from polar bears (*Ursus maritimus*) at Svalbard. J Environ Health Toxicol 2001;62:227–241.

Stirling I, Andriashek D, Spencer C, Derocher A. Assessment of the polar bear population in the eastern Beaufort Sea. Final report to the northern oil and gas action program. Canadian Wildlife Service, 1988.

Stirling I, Spencer C, Andriashek D. Immobilization of polar bears (*Ursus maritimus*) with Telazol® in the Canadian Arctic. J Wildl Dis 1989;25:159–168.

Taylor MK, DeMaster DP, Bunnell FL, Schweinsburg RE. Modeling the sustainable harvest of female polar bears. J Wildl Manage 1987;51:811–820.

Tryland M, Derocher AE, Wiig Ø, Godfroid J. *Brucella* sp. antibodies in polar bears from Svalbard and the Barents Sea. J Wildl Manage 2001;37:523–531.

Watts PD, Hansen SE. Cyclic starvation as a reproductive strategy in the polar bear. Symp Zoo Soc Lond 1987;57:305–318.

Wiig Ø. Survival and reproductive rates for polar bears at Svalbard. Ursus 1998;10:25–32.

Wiig Ø, Derocher AE, Cronin MM, Skaare JU. Female pseudohermaphrodite polar bears at Svalbard. J Wildl Dis 1998;34:792–796.

Wolkers J, Witkamp RF, Nijmeijer SM, Burkow IC, De Groene EM, Lydersen C, Dahle S, Monshouwer M. Phase I and phase II enzyme activities in ringed seals (*Phoca hispida*): characterization of hepatic cytochrome P450 by activity patterns, inhibitions studies, mRNA analyses, and western blotting. Aquat Toxicol 1998;44:103–115.