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Concentrations and patterns of organochlorine contaminants in white whales (*Delphinapterus leucas*) from Svalbard, Norway

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Abstract

Blubber was collected from live-captured, adult male white whales (*Delphinapterus leucas*) from Svalbard, Norway, and analysed for levels and patterns of organochlorine (OC) contaminants. The OC compounds analysed were HCB, dieldrin, ΣHCH (α-HCH, β-HCH and γ-HCH), ΣChl (heptachlor epoxide, oxychlordane, *cis*-chlordane, *trans*-nonachlor, and *cis*-nonachlor), ΣDDT (*pp′*-DDT, *pp′*-DDE and *pp′*-DDD) and ΣPCB (27 PCB congeners). The major OC compounds detected in the blubber were ΣPCB (5103 ± 1874 ng/g l.w.) and ΣDDT (5108 ± 1089 ng/g l.w.), which made up 70% of the ΣOC. These compounds were followed in prevalence by ΣChl (2872 ± 1177 ng/g l.w.), which contributed 20% of the ΣOC burden. ΣHCH, HCB and dieldrin were present, but at low concentrations. This OC pattern is typical of top predators in Arctic marine food chains. OC levels in white whales from Svalbard are lower than white whales from the St Lawrence River in Canada and are generally similar to values reported previously for other Arctic white whale stocks. Some geographic patterns in relative prevalence of various OC compounds appear to be quite consistent among various marine mammal species in the Arctic. PCB and DDT concentrations in Svalbard's white whales are below the levels that are thought to have negative effects on reproduction or the immune system.

Keywords: Arctic; Pollution; Beluga; Marine mammals; Upper-trophic level feeding; Biomagnification

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1. Introduction

Despite its relative isolation, the Arctic Ocean has become a recipient and sink for many pollutants, including organochlorine (OC) contaminants that originate mainly from agricultural and industrial areas in Europe, Asia, North America and northern Africa (Ottar, 1981). Atmospheric transport, and subsequent condensation and fallout from cold Arctic air masses, is thought to be the most important input-pathway of these semivolatile compounds to Arctic environments (Ottar, 1981; Wania and Mackay, 1993; Oehme et al., 1996). OC pollutants can also be transported to Arctic regions via ocean currents, river outflows, continental run-off and ice-drift (Ottar, 1981; Oehme, 1991; Barrie et al., 1992; Macdonald and Bewers, 1996). Due to their specific chemical and physical properties, OC compounds are resistant to physical and biological degradation. Their low water solubility and high lipid solubility leads to their bioconcentration in lipid-rich tissues of organisms and their biomagnification between increasing trophic levels within ecosystems (Thomann, 1989; Macdonald and Bewers, 1996).

OC concentrations are generally high in top predators such as pinnipeds, polar bears (*Ursus maritimus*) and cetaceans in Arctic communities owing to the high trophic positions and large lipid reserves in these animals (Tanabe et al., 1984; Norstrom and Muir, 1994; Skaare, 1995; De March et al., 1998). Some of the most conspicuous effects of OCs on wildlife have been reported in white whales. High concentrations of OCs in white whales from the St Lawrence Estuary are believed to be a factor in the unusually high incidences of tumours, cancers, lesions, immune dysfunction, reduced reproductive rates, abnormal sexual development, and high mortality in this population (Martineau et al., 1987, 1994; Beláand et al., 1993; De Guise et al., 1995). Studies of other white whale populations in the Canadian Arctic, Alaska and West-Greenland have revealed that body burdens of OCs in these areas are much lower than those reported from the St Lawrence Estuary (Muir et al., 1990; Stern et al., 1994; Becker et al., 1997; Krahn et al., 1999). However, there is some indication of a general west–east increase in the body burden of OCs in Arctic top predators within the North Atlantic. OC concentrations in ringed seals (*Phoca hispida*) tend to be higher at Svalbard and in adjacent waters compared with values from the North-American Arctic (Luckas et al., 1990; Muir et al., 1992a; Wolkers et al., 1998). Likewise, OC burdens in polar bears (*Ursus maritimus*) from Svalbard are higher than in their Canadian or Alaskan counterparts (Norheim et al., 1992; Kleivane et al., 1994; Bernhoft et al., 1997; Norstrom et al., 1998). Additionally, recent reports of pseudohermaphroditic bears in Svalbard are suggested to be due to endocrine disruption from environmental pollutants (Wiig et al. 1998). Fish and flesh-eating seabirds in Svalbard have been reported to have very high OC burdens in some of their tissues (Gabrielsen et al., 1995; Barrett et al., 1996).

The reports of high OC contaminant burdens in upper trophic level animals from Svalbard has lead to concern for the local white whale population and a need for information about the body burdens of OCs from this species in this area. This population was hunted heavily from the mid-1800s until it was protected in 1960 (Gjertz and Wiig, 1994) and its current status is unknown (see Lydersen and Wiig, 1995). The genetic relationship of the white whale stock in Svalbard to that of neighbouring populations in the Russian or Greenlandic Arctic is unknown; this research question is currently under investigation. This study reports the levels and patterns of OC pollutants in blubber biopsies taken from live-captured white whales from Svalbard.

2. Methods

2.1. Study area and sampling

Ten adult male white whales were live-captured between July 7th and August 12th 1995–1997 in

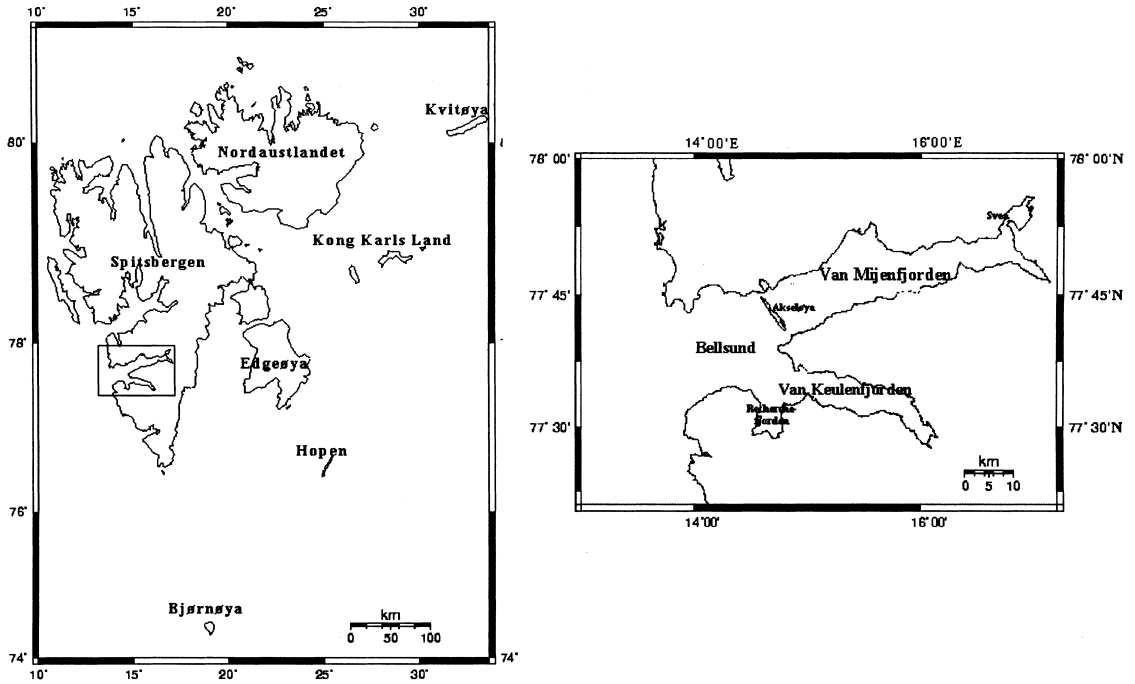

Fig. 1. Map of Svalbard, Norway (left), and the study area on the West Coast of Spitsbergen (box and right).

Van Mijenfjorden (77°45′N, 15°00′E) and Van Keulenfjorden (77°35′N, 15°00′E), on the west coast of Spitsbergen, Svalbard, Norway (Fig. 1). The whales were captured using a nylon net (70 × 29 m, mesh size = 20 cm) set from the beach. The animals were herded into the net opening using two zodiac boats. After entanglement the whales were drawn into shallow water near the shore while sampling and measuring were conducted. Standard length (ASM, 1961) was measured to the nearest 5 cm and sex was determined by examining the genitals. Based on body size and skin colour, all animals sampled during this study were classified as adults (see Brodie, 1971; Sergeant, 1973). Several (four to 10) blubber biopsies were sampled from an area just anterior of the mid-dorsal ridge of each whale. The four animals captured in 1995 and 1996 were sampled using a gauge biopsy punch (diameter = 8 mm, Medizin Gmblt, Köln, Germany). The six animals captured in 1997 were sampled using a custom-made, hollow, stainless steel rod (diameter = 6 mm, length = 150 mm). All biopsies included the epidermis, dermis and some hypodermic blubber tissue. The metal rods produced a deeper, longer core of blubber compared with the commercially produced punches, that represented most of the blubber layer. All samples were wrapped in aluminium foil, and kept frozen at −20°C until analysis. Sampling procedures took approximately 15 min for an individual and all animal-handling procedures were conducted according to Norwegian National Animal Care Guidelines, and under permits from the Governor of Svalbard.

2.2. Chemical analysis of chlorinated pesticides and PCB

Chemical analyses of chlorinated pesticides and PCBs were performed at the Laboratory for Environmental Toxicology at the Norwegian College of Veterinary Medicine, Oslo, Norway. The methods used are described in Brevik (1978), with modifications described in Bernhoft and Skaare (1994). As the quantities of tissue retrieved from the biopsy punches were small, methodological adjustments were developed in this study to ensure that homogenization and extraction were

performed with minimal loss of sample tissue and maximal lipid and OC output (see below).

2.3. Extraction and clean-up

The skin was removed from the blubber biopsies. Several biopsies from each whale were pooled, macerated and then frozen at −20°C, before being homogenized (Ike Ultra Turrax™, Janke & Kunkel, Gmbh & Co., KG, Germany). A solution with three internal standards, PCB-29, PCB-112 and PCB-207 (Promochem GmbH, Wesel, Germany; Ultrascientific, North Kingstown, RI, USA), was added to the homogenized samples. The samples were further homogenized with an ultrasonic homogenizer (4710 Series, Cole Parmer Instruments Co., Chicago, IL, USA) and extracted using cyclohexane and acetone. This procedure was repeated three times. Between the second and the third extraction, samples were extracted over-night at room temperature. The extractable lipid content in each sample was calculated from a 2-ml aliquot of extract that was evaporated on a 50°C sand bath. Lipids were removed from the extracts by acidic breakdown or alkaline hydrolysis of the fat before GC analyses were run. The two different clean-up procedures were used because some OC components analysed in this study are broken down in acidic solution while others break down in association with alkaline compounds. Acidic clean-up (ultra-clean concentrated sulfuric acid, H_2SO_4, purity 98.08%, Scanpure, Chemscan AS, Elverum, Norway) was applied to analyse PCBs, DDTs, HCHs, chlordanes, HCB and Mirex. Acid-cleaned extracts were concentrated or diluted, depending on their lipid content in order to detect and quantify the components within the linear range of the detector. The liquid phase was evaporated in three steps using N_2 gas. For each step the tubes were flushed with 1 ml cyclohexane to redissolve the components that adhered to the walls during each step.

Alkaline clean-up procedures were used for analyses of dieldrin, aldrin, endrin, heptachlor and heptachlor epoxide. The methods followed Jensen et al. (1972), with the following modifications. Potassium hydroxide (KOH) (Merck, Darmstadt, Germany) and ethanol (96%) were added to the extracts and gently mixed. Then the samples were placed in a 50°C water bath for 30 min and subsequently cooled in ice water for 10 min. A phosphoric-reagent (6.85 ml *ortho*-phosphoric acid, *ortho*-H_3PO_4, and 11.6 g NaCl mixed in a 1000-ml volumetric flask) was added and the contents were gently mixed. The samples were then left over-night in a −20°C freezer. The samples were then thawed for 30 min at room temperature and subsequently centrifuged for 10 min at 3000 rev./min. The supernatants were dried (600°C for 6 h) using sodium sulfate (Na_2SO_4) (Merck, Darmstadt, Germany).

2.4. GC analyses

All sample extracts were analysed on two high-resolution gas chromatographs (HRGC, 5300 Mega Series, Carlo Erba, Milano, Italy). Both GCs were equipped with a Fisons AS 800 autosampler and Ni^{63} electron capture detector (Carlo Erba, Milano, Italy). One GC had a relatively non-polar SPB-5 column (poly-5% diphenyl/95% dimethylsiloxane film) and the other had a more polar SPB-1701 column (poly-14%-cyanopropanol/86% dimethylsiloxane film). Both columns (Supelco Inc. Bellefonte, PA, USA) were 60 m long, had internal diameters of 0.25 mm, were coated with 0.25-μm-thick film, and had a 1-m-long deactivated pre-column.

One microlitre of the cleaned extracts was injected into the split/splitless injector at 270°C using an autosampler. The splitless time was 2 min and the split ratio was 1:20. The septum flush was 2 ml/min and the detector temperature was 310°C. The carrier gas (2 ml/min) was H_2 (purity 5.0, Hydro Gas, Rjukan, Norway), which had a linear velocity of 40 cm/s. The makeup gas (30 ml/min) was 5% methane in Argon (Hydro Gas, Rjukan, Norway). The temperature program was: 90°C (2 min hold); increase 25°C/min to 180°C (2 min hold); 1.5°C/min increase to 220°C (2 min hold); and 3°C/min increase to 275°C (15 min hold). The total run time for each sample was 70 min. The OC compounds were identified and quantified by comparing retention times and peak heights in the sample chromatogram to chro-

matograms of standard OC solutions analysed on the GC prior to the sample extracts. The standard solutions consisted of the following OC components: hexachlorobenzene (HCB); Mirex; dieldrin; endrin; aldrin; three hexachlorocyclohexane isomers (α-HCH, β-HCH and γ-HCH); eight chlordane compounds; and metabolites (heptachlor, heptachlor epoxide, oxychlordane, *cis*-chlordane, *trans*-chlordane, *trans*-nonachlor, and *cis*-nonachlor); five DDT components and metabolites (*pp′*-DDE, *pp′*-DDT, *op′*-DDT, *pp′*-DDD and *op′*-DDD); and 34 PCB congeners (28, 31, 47, 52, 56, 66, 74, 87, 99, 101, 105, 114, 110, 118, 128, 136, 137, 138, 141, 149, 151, 153, 156, 157, 170, 180, 183, 187, 189, 194, 196, 199, 206 and 209). Most of these components were quantified on the SPB-5 column. However, PCB-52, *cis*-chlor-dane, *trans*-nonachlor, PCB-110, -105, -137, *cis*-nonachlor, heptachlor, aldrin, heptachlor epoxide, dieldrin and endrin were quantified on the SPB-1701 column because of co-elution with other components or interference with ghost peaks (unknown components) on the SPB-5 column. Mirex, *trans*-chlordane, *op′*-DDD, *op′*-DDT, PCB-56 and PCB-209 could not be quantified on either of the columns.

Because lipid content is believed to be the major determinant of wet weight (w.w.) OC concentrations in biological samples (Thomann, 1989; Bignert et al., 1994; Hebert and Keenleyside, 1995), variation in lipid content in the blubber samples in this study made it necessary to normalize OC concentrations to the lipid weight (l.w.) of the samples. Only lipid based OC concentrations are used further in this study.

2.5. Analytical quality

Quality assurance during analyses in this study, included three-point linear calibration curves of the analysed OC standard solutions used to detect and quantify OC levels in the sample extracts. The detection limits of the extracts were defined as 3 × the average instrument signal (back ground noise) in the chromatograms. In this study, the detection limit ranged from 0.160 ng/g w.w. to 28.358 ng/ w.w., depending on the OC compound, concentrations in the standard solution, height of the standard peak, sample weight and the dilution factor of the sample. The quantification limits were set as three times the detection limit. The following OC compounds were below the detection limit in all samples: aldrin; endrin; heptachlor; PCB-28, PCB-31, PCB-114, PCB-189, PCB-199 and were, therefore, defined as not detected. The PCB congeners PCB-137, -141, -157, -194, -196 and -206 were below the detection limit in only some of the samples. They were given a value of half the detection limit. Every 3 months the laboratory makes an extended six-point calibration curve which defines the linear range of the detector from the detection limit to the most concentrated standards times two. All quantification in the present study was done within this range.

The internal standards PCB-29, -112 and -207 were used to detect and correct for changes in OC concentrations during the chemical preparation and injection of the extracts into the GC and during the GC run. PCB-29 covered α-HCH to PCB-47, PCB-112 covered PCB-74 to -183, and PCB-207 covered PCB-156 to -206 (listed according to their retention times on the SPB-5 column). Following every batch of 12–24 samples analysed on the GC, one blank sample consisting of solvents and another sample with non-contaminated tissue were analysed. Recovery from samples of seal blubber spiked with OC standard solutions were also analysed following each sample-series. OC standards were run every tenth sample during the GC analysis to detect any deviations (drift) in the response of the system. The blanks, recoveries and drift were found to be within acceptable ranges. Also, parallel samples from two white whales were within accepted values (10% of mean) when normalized for lipid content. Two seal blubber control samples were analysed with every batch, which provided a measure of the repeatability and reproducibility of results over time. The concentrations of PCB-118, PCB-153 and *pp′*-DDE in these control samples were compared with previous control samples (mean of 1996) and were within the acceptable ± two standard deviations.

2.6. Statistics

Statistical analyses were performed using the statistical package SPSS (Version 7.0 for Windows 95, Inc., IL, USA). Due to the small sample size in this study ($n = 10$) and because OC concentrations deviated from a normal distribution, non-parametric Spearman rank correlation (r_s) was used to test the relationship between animal length (as an indicator of age) and OC concentrations and ratios of major OC classes. Significance level was set to $P < 0.05$ and two-tailed P-values are presented. Numerical values are presented as arithmetic means ± standard deviation (S.D.) in the text, and in addition as medians with maximum and minimum values in Table 1. Although, median values are probably the best measure of central tendency for data that are not normally distributed (Norman and Streiner, 1994), mean ± S.D. were also calculated and are presented in order to permit comparisons with other studies.

3. Results

The OC compounds detected in white whale males from Svalbard included: HCB; dieldrin; ΣHCH (α-HCH, β-HCH and γ-HCH); ΣChl (chlordane compounds: heptachlor epoxide, oxychlordane, *cis*-chlordane, *trans*-nonachlor and *cis*-nonachlor); ΣDDT (*pp′*-DDT, *pp′*-DDE and *pp′*-DDD); and ΣPCB (27 PCB congeners: 47, 52, 66, 74, 87, 99, 101, 105, 110, 118, 128, 136, 137, 138, 141, 149, 151, 153, 156, 157, 170, 180, 183, 187, 194, 196 and 206). The concentrations and group sums for these compounds are presented in Table 1.

ΣDDT and ΣPCB were the two predominant OC groups, contributing approximately equally to ΣOC (35.2 ± 5.7% and 34.0 ± 2.8%, respectively, see Table 1). ΣChl was also a significant contributor to the overall contaminant load, being 19.1 ± 1.8% of ΣOC. Dieldrin, HCB and ΣHCH were found in low concentrations (7.5 ± 1.1%, 3.2 ± 0.8% and 0.9 ± 0.4% to ΣOC, respectively). The single compound that dominated ΣOC was *pp′*-DDE, which accounted for 27.2 ± 5.2%, followed in prevalence by *trans*-nonachlor (9.8 ± 1.3% of ΣOC) and dieldrin (approx. 7.5% of ΣOC).

Table 1
Concentrations of different organochlorine contaminants in blubber biopsies male white whales from Svalbard ($N = 10$)

OC compounds (ng/g l.w.)	Median	Min.–max.	Mean ± S.D.
HCB	430	324–1423	507 ± 327
Dieldrin	1005	746–2657	1145 ± 554
α-HCH	26	16–110	38 ± 31
β-HCH	59	33–210	72 ± 52
γ-HCH	29	17–190	46 ± 52
ΣHCH[a]	111	68–510	155 ± 133
Heptachlor epoxide	209	152–633	246 ± 140
Oxychlordane	641	491–2037	760 ± 456
cis-Chlordane	120	36–880	190 ± 247
trans-Nonachlor	1369	1198–1860	1408 ± 189
cis-Nonachlor	238	171–733	269 ± 167
ΣChl[b]	2566	2099–6143	2872 ± 1177
pp′-DDT	775	455–1200	733 ± 222
pp′-DDD	413	293–887	460 ± 164
pp′-DDE	4046	2431–5149	3915 ± 807
ΣDDT[c]	5083	3272–6770	5108 ± 1089
ΣPCB[d]	4680	3198–10075	5103 ± 1874
ΣOC[e]	13 939	9787–27578	14 890 ± 4504
Lipid content	20.17	0.75–95.76	27.87 ± 30.36

[a] ΣHCH includes α-HCH, β-HCH and γ-HCH.
[b] ΣChl includes heptachlor epoxide, oxychlordane, *cis*-nonachlor, *trans*-nonachlor and *cis*-chlordane.
[c] ΣDDT includes *pp′*-DDT, *pp′*-DDE and *pp′*-DDD.
[d] ΣPCB includes PCB congener numbers 47, 52, 66, 74, 87, 99, 101,105, 110, 118, 128, 136, 137, 138, 141, 149, 151, 153, 156, 157, 170, 180, 183, 187, 194, 196 and 206 (IUPAC numbering according to Ballscmither and Zell, 1980).
[e] ΣOC contains HCB, dieldrin, ΣHCH, ΣChl, ΣDDT and ΣPCB.

Body length and concentrations of HCB, dieldrin, ΣHCH, ΣChl, ΣDDT or ΣPCB were not correlated (Fig. 2; $|r_s| < 0.5$, $n = 10$, $P > 0.10$), but individual variations in OC concentrations were apparent despite the fact that the whales in this study were consistent with respect to sex and very similar with respect to size. One whale (Fig. 2) had two to three times higher concentrations of HCB, dieldrin, ΣHCH, ΣChl and ΣPCB compared with the other nine whales. However, this individual had a ΣDDT concentration that was similar to the other animals in the sample. The OC ratios were quite similar among all 10 whales (Fig. 3).

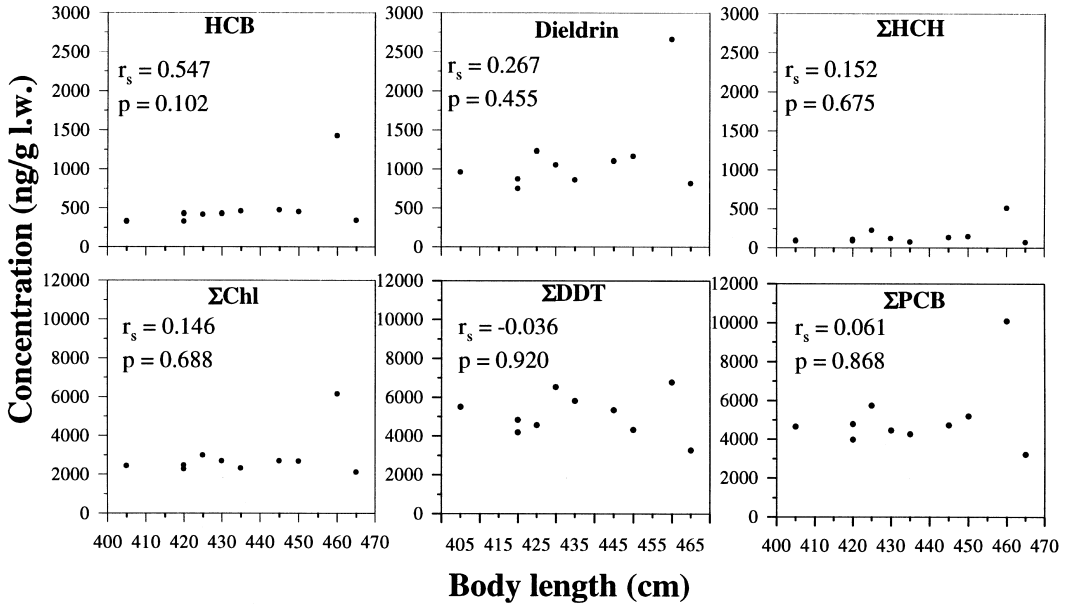

Fig. 2. Body length vs. OC concentration of major OC compounds in blubber of 10 male white whales from Svalbard. r_s = Spearman's rank correlation coefficient. *P*-values are two-tailed.

ΣHCH was dominated by the β-HCH isomer (Fig. 4) which accounted for almost half the HCH load (49.1 ± 7.0%). α-HCH and γ-HCH contributed 24.1 ± 5.6% and 26.8 ± 4.5% to ΣHCH, respectively. *trans*-Nonachlor was the predominant chlordane compound (51.9 ± 8.1%) in ΣChl, followed by the metabolite oxychlordane which accounted for 25.5 ± 3.4% (Fig. 4). *cis*-Nonachlor, heptachlor epoxide and *cis*-chlordane were minor contributors to ΣChl (9.0 ± 1.4%, 8.3 ± 1.0% and 5.4 ± 3.5%, respectively). The metabolite *pp'*-DDE was by far the most important contributor to ΣDDT, accounting for 76.7 ± 3.6% (Fig. 4). Another metabolite, *pp'*-DDD, was present in much smaller amounts (9.0 ± 2.3% of ΣDDT). The proportion of the parent compound *pp'*-DDT was 14.2 ± 1.9% of ΣDDT.

ΣPCB was dominated by the penta- and hexachlorobiphenyls (CBs) (32.8 ± 3.0% and 42.0 ± 2.9%, respectively, Figs. 4 and 5). Tetra and

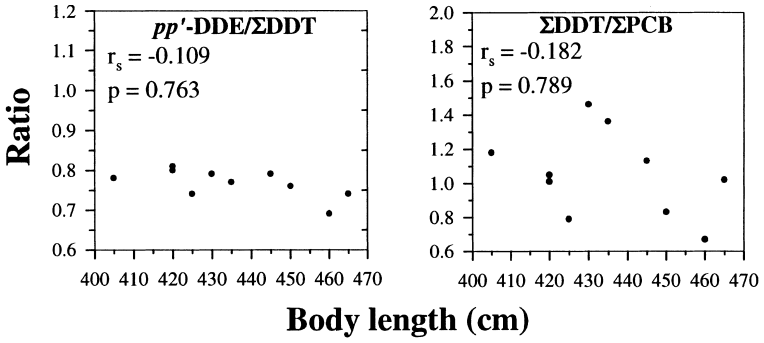

Fig. 3. Body length vs. selected OC ratios in blubber of 10 male white whales from Svalbard. r_s = Spearman's rank correlation coefficient. *P*-values are two-tailed.

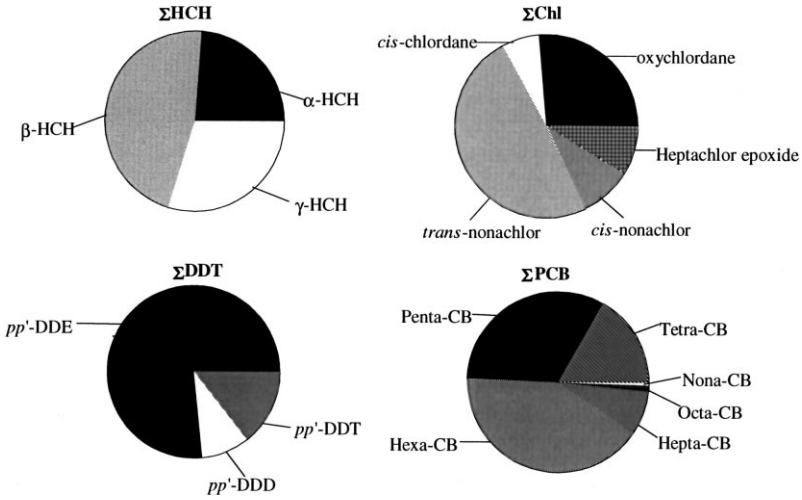

Fig. 4. Mean of the relative contributions of individual isomers/metabolites to the sum of the corresponding OC groups (ΣHCH, ΣChl, ΣDDT, ΣPCB) in blubber from 10 male white whales from Svalbard.

hepta-CBs contributed $15.7 \pm 3.6\%$ and $8.4 \pm 1.6\%$ to ΣPCB, respectively. Octa- and nona-CBs were minor contributors to ΣPCB ($0.9 \pm 0.4\%$ and $0.40 \pm 0.5\%$, respectively). The major PCB congeners were PCB-138 and PCB-153, which together made up 25% of the ΣPCB load. Other large contributors were PCB-52, -99, -101, -118 and -149. Altogether, these seven congeners constituted 66% of ΣPCB.

4. Discussion

White whales, like many other Arctic marine mammals, feed primarily on small fish (e.g. Heide-Jørgensen and Teilmann, 1994). In Svalbard, their principal prey includes polar cod (*Boreogadus saida*) and capelin (*Mallotus villosus*) (Dahl et al., 2000). Thus, it is perhaps not surpris-

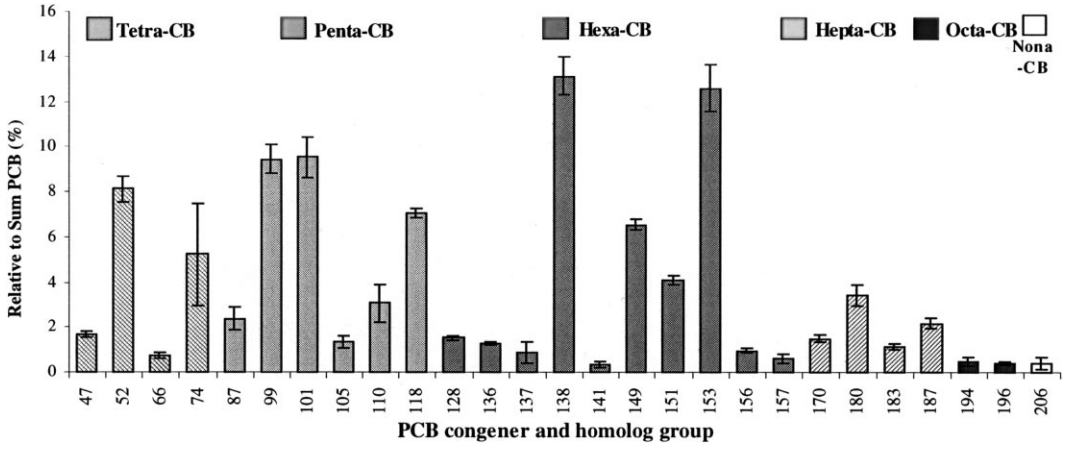

Fig. 5. Mean (±95% CI) of relative contribution (%) to ΣPCB of the 27 PCB congeners analysed in blubber from 10 male white whales from Svalbard. The PCB congeners are grouped according to homolog groups (degree of chlorination); tri-CB, trichlorobiphenyl; tetra-CB, tetrachlorobiphenyl; penta-CB, pentachlorobiphenyl; hexa-CB, hexachlorobiphenyl; hepta-CB, heptachlorbiphenyl; octa-CB, octachlorobiphenyl; nona-CB, nonachlorobiphenyl.

ing that the ranking of OC compounds relative contributions to ΣOC in this study (ΣDDT = ΣPCB > ΣChl ≫ Dieldrin > HCB > ΣHCH) is consistent with the OC pattern generally found in Arctic marine mammals (see Norstrom and Muir, 1994), including species from Svalbard and adjacent waters (Skaare, 1995; Kleivane et al., 1997; Kleivane and Skaare, 1998). This OC pattern largely reflects the different OC compounds' physical and chemical properties, and thus their bioaccumulative properties in organisms within the marine food chain (Hargrave et al., 1992; Borgå, 1997).

On a finer scale, some geographic patterns of relative prevalence of compounds are apparent. ΣHCH was dominated by β-HCH, which is considered to be the most persistent and bioaccumulative isomer (Howard, 1991; Turnbull, 1996). This is in contrast to reports from the Canadian Arctic for white whales, where α-HCH was the major contributor to ΣHCH (Muir et al., 1990). The same geographical HCH-pattern, with β-HCH dominating in Svalbard and α-HCH in Canada, has also been found in polar bears (Norstrom et al., 1988; Bernhoft et al., 1997). Technical-HCH consists of more β-HCH (5–14%) than Lindane (99.8% γ-HCH), which is easily converted to α-HCH (Barrie et al., 1992; Turnbull, 1996). This suggests that Svalbard's biota is more influenced by a technical-HCH source compared with the North-American Arctic.

The dominant chlordane compounds in this study were *trans*-nonachlor and the persistent metabolite, oxychlordane (Fig. 4). This is in accordance with reports in other Arctic top predators (Muir et al., 1992a; Norstrom and Muir, 1994), including white whales from other regions (Muir et al., 1990, 1996a).

The high *pp′*-DDE levels found in this study were not particularly surprizing. This compound is a product of the abiotic breakdown and biotic metabolism of the active pesticide *pp′*-DDT, and is the most persistent and bioaccmulative DDT compound (Aguilar, 1984). The *pp′*-DDE/ΣDDT ratio in the current study ranged from 0.69 to 0.81, which indicates an equilibrium between DDT input and conversion to DDE in the environment (Aguilar, 1984). Thus, the DDT input to Svalbard is likely via long-range transport of old DDT discharges that have been converted to DDE over time.

The PCB pattern in this study was consistent with that previously reported in marine mammals from Svalbard (Skaare, 1995), as well as white whales and narwhals (*Monodon monoceros*) from other Arctic regions (Muir et al., 1990, 1992b; Stern et al., 1994). However, the contributions of tetra- to hexa-CBs was high while hepta- to octa-CB contributions were low in Svalbard's white whales compared to their St Lawrence counterparts (Muir et al., 1990, 1996a). This might be caused by different PCB sources or dissimilar transport pathways (Wania and Mackay, 1993; Muir et al., 1996a,b). It might also be explained by variation in biological factors such as diet, age, migrations patterns, nutritional and reproductive condition, and metabolic capacity among individuals in the two white whale populations (Skaare, 1995; Muir et al. 1996a). Degradation of PCBs might be enhanced in the St Lawrence white whales because of induction of P450 enzymes by the higher PCB load; this might leave only the most recalcitrant congeners with higher chlorine content (Muir et al., 1990, 1996a). The PCB load in Svalbard's white whales might not be sufficiently high to cause enzyme induction and, therefore, the PCB pattern may retain the influence of the less highly chlorinated congeners.

The dominating PCB congeners in this study, PCB-153 and PCB-138, are consistent with their high prevalence in technical mixtures and high persistence in the biota (Boon et al., 1987; McFarland and Clarke, 1989). However, the PCB pattern also reflects the specific metabolic capacity of cetaceans to metabolize these compounds (Tanabe et al., 1988; Norstrom et al., 1992; White et al., 1994; Goksøyr, 1995). When comparing the result of this study to OC levels reported for adult, male marine mammals of other species from Svalbard and adjacent waters a mixed pattern emerges. White whales show two to three times higher ΣPCB and *pp′*-DDE concentrations compared with ringed seals (Daelemans et al., 1993; Wolkers et al., 1998). Svalbard white whales also have two to five times higher levels of OC compounds compared to harp seals (*Phoca groen-*

landica) from the southern Barents Sea (results from early and late in the moulting season; Kleivane et al., 1995a). However, polar bears from Svalbard show approximately five times higher ΣPCB and twice the ΣHCH level in subcutaneous fat compared with blubber of Svalbard white whales (Bernhoft et al., 1997; this study). In contrast, HCB, ΣChl and dieldrin are two to three times higher and ΣDDT is as much as 20 times higher in the white whales compared with polar bears (Bernhoft et al., 1997; this study). Levels of the different OC compounds are generally two to five times lower in white whales from Svalbard compared to harbour porpoises (*Phocoena phocoena*) inhabiting the Norwegian coastal waters (Kleivane et al., 1995b). Minke whales (*Balaenoptera acutorostrata*) in the Barents Sea and Norwegian coastal waters have OC levels that are similar to the white whales in this study (Kleivane and Skaare, 1998).

Geographical distribution, diet including trophic feeding level, and specific metabolic capacity to degrade OC compounds are probably the main factors explaining the observed interspecific patterns in OC levels. Cetaceans have been found to have a low OC metabolizing ability compared to seals (Tanabe et al., 1988; Norstrom et al., 1992), which is the most likely explanation for the higher OC levels in Svalbard white whale compared with seals from this area. Polar bears are predators that feed almost exclusively on seals (Stirling and Archibald, 1977). The bears' high and specific ability to metabolize many OCs, likely explains the lower concentrations of these compounds in these animals compared with white whales from the same area, except for the highly persistent PCBs which were five times higher in the polar bears (Bernhoft et al., 1997). The findings in the current study are consistent with the suggestions of lower ability in cetaceans to metabolize OC compounds compared with seals or polar bears (Tanabe et al., 1988; Norstrom et al., 1992; Norstrom and Muir, 1994; Skaare, 1995).

Some caution must be taken when comparing levels of OC contaminants among geographic areas when they are represented in different studies because of variations in biological parameters of the sampled animals, as well as differences in collection year and season, and inconsistencies in sampling and analytical methods. For instance, comparing the results from this study directly with OC levels found in white whales from the Canadian Arctic and St Lawrence (Muir et al., 1990, 1996a) may not be appropriate. Several of the blubber samples from Svalbard did not include tissue from the innermost layer. Different blubber-coring methods over the various years of sampling in our study might have biased the overall OC values upwards slightly. Aguilar and Borrell (1991) found that higher concentrations of toxins occurred in the outer blubber layer compared to deeper in toward the core in a study of baleen whales. However, this finding has been questioned by contrary data in a similar study by Gauthier et al. (1997) and toothed whales have thinner, less functionally and compositionally complex blubber than baleen whales (Aguilar, 1985), therefore, odontocetes might have a more homogeneous OC distribution in the blubber column. Differences were not clearly apparent between samples acquired using the different biopsy tools in this study, but we cannot rule out a potential bias in some of our samples. Ideally, all blubber samples from marine mammals used in ecotoxicology studies should include the entire column to avoid potential biases. Inconsistent presentation of concentrations, on wet vs. a lipid weight basis and summation of OC groups including variable numbers of compounds also confounds comparisons. Standardization among OC surveys is needed in order to confirm the existence of spatial OC trends in Arctic marine biota. Keeping this set of problems in mind, it is still possible to look cautiously at large-scale geographic patterns.

OC levels previously reported in stranded white whales from the St Lawrence Estuary were three to 15 times higher than in white whales from Svalbard (Muir et al., 1990, 1996a; see Table 2). This is as expected because the St Lawrence River and Estuary were heavily polluted from regional and local industry and agriculture during the 1970s, especially with PCBs, DDTs and Mirex (Martineau et al., 1987; Muir et al., 1996c). Svalbard is much further north and its remote location means that there have been few local sources.

Table 2
Mean (±S.D.) OC levels (ng/g) in blubber of male white whales from selected stocks

Locations[a]	Year	*N*	Lipid content (%)	Conc.	ΣPCB	ΣDDT	ΣChl	ΣHCH	Dieldrin	HCB
Svalbard[1]	1995–97	10	27.1	l.w.	5100 ± 1870	5110 ± 1090	2870 ± 1180	155 ± 130	1145 ± 550	507 ± 330
N. West Greenland (Nuussuaq)[2]	1989–90	54	89.1	w.w.	5580 ± 2500	4370 ± 1730	2600 ± 1160	390 ± 140	1040 ± 580	825 ± 477
Jones Sound (Canadian High Arctic)[3]	1984	8	88.4	w.w.	2530 ± 570	1960 ± 320	1870 ± 440	190 ± 90	340 ± 110	500 ± 210
Mackenzie Delta, Beaufort Sea (Canadian Arctic)[4]	1993–94	26	89.9	w.w.	5010 ± 1618	3433 ± 641	2486 ± 319	334 ± 94	410 ± 124	956 ± 128
E. Bering Sea (Alaska)[5]	1990	7	88.2	w.w.	4170 ± 644	3120 ± 571	2100 ± 179	320 ± 64	343 ± 54	956 ± 128
E. Chukchi Sea (Alaska)[6]	1990, 96	11	86.0	w.w.	5200 ± 900	3630 ± 900	2420 ± 460	330 ± 760	390 ± 86	810 ± 120
Cook Inlet (Alaska)[6]	1992–97	10	90.0	w.w.	1490 ± 700	1350 ± 730	560 ± 250	210 ± 70	92 ± 47	220 ± 93
SE. Baffin Island (Canadian Arctic)[7]	1993–94	6	91.2	w.w.	6794 ± 2171	6964 ± 3303	4236 ± 1669	515 ± 163	1441 ± 661	1305 ± 781
S. Hudson Bay (Canadian Arctic)[7]	1994	7	91.2	w.w.	6768 ± 2346	11 200 ± 8231	4167 ± 1585	442 ± 133	923 ± 360	460 ± 255
St Lawrence Estuary (Canada)[8b]	1987–90	15	82.1	l.w.	78 900	81 100	8400	468	1680	1350

[a1] Current study; [2] Stern et al. (1994); [3] Muir et al. (1990); [4] Muir (1994, 1996) compiled by Muir et al. (1997); [5] Becker et al. (1995) compiled by De March et al. (1998); [6] Krahn et al. (1999); [7] Muir et al. (1997); [8] Muir et al. (1996c).

[b] Only geometric mean was available; w.w., wet weight concentration; l.w., lipid weight concentration.

OC pollutants in Svalbard's waters occur mainly via long-range atmospheric transport (Oehme, 1991; Oehme et al., 1996). Few geographical differences in OC levels have been found between the Arctic white whale stocks in Canada, west Greenland and Alaska (Muir et al., 1990, 1997; Stern et al., 1994; Krahn et al., 1999). Svalbard whales appear to have OC concentrations that are somewhat lower than the Hudson Bay and south-eastern Baffin Island stocks, but generally higher than the Jones Sound (Canadian High Arctic) and Cook Inlet stock (Alaska). ΣHCH and HCB concentrations in Svalbard white whales are approximately half as high as the levels reported in north-western Greenland, the eastern Canadian Arctic (Beaufort Sea) and Alaska (E. Bering Sea and E. Chukchi Sea) (Stern et al., 1994; Muir et al., 1997; Krahn et al., 1999). This is in agreement with a study by Bernhoft et al. (1997) who found that Svalbard polar bears had lower ΣHCH and HCB levels compared to their Canadian counterparts (Norstrom et al., 1988). Dieldrin and ΣDDT concentrations in Svalbard white whales are similar to those from north-western Greenland, and somewhat higher (1.5–2 ×) than those found in the eastern Canadian Arctic and Alaska (Stern et al., 1994; Muir et al., 1997; Krahn et al., 1999). ΣPCB is similar in the various Arctic white whale populations. This pattern is also found for DDT concentrations reported in ringed seals (Luckas et al., 1990; Muir et al., 1992a; Wolkers et al., 1998) and polar bears (Norstrom et al., 1998).

The lack of relationship between body length and concentrations of different OC concentrations in this study is consistent with previous studies of white whales (Stern et al., 1994; Muir et al., 1996a,c) and narwhals (Muir et al., 1992b). These data contradict the general notion that cetacean males accumulate OC contaminants with increasing age (Aguilar and Borrell, 1988; Tanabe et al., 1994). However, our study adds little information to this issue because all of the sampled animals were adults, and the relatively small differences in their lengths may not be age-related.

The most dominant OC contaminants in this study, ΣDDT (3200–6800 ng/g l.w.) and ΣPCB (3200–10 000 ng/g l.w.), are considered among the most toxicologically important (Safe, 1990; Ahlborg et al., 1994). Heptachlor epoxide and especially dieldrin are also considered to be very toxic compounds because of their carcinogenic properties (Howard, 1991), and they were found in rather high concentrations in this study. However, the levels found in St Lawrence white whales that have been thought to cause chronic effects were much higher [ΣDDT (52 400–123 000 ppm w.w.) and ΣPCB (53 900–89 200 ppm w.w.)] levels than those found in the present study (Muir et al., 1990; Beláand et al., 1993). Likewise, an epizootic incident of the morbillovirus in striped dolphins (*Stenella coeruleoalba*) from the Mediterranean Sea that has been coupled to blubber concentrations of ΣPCB in the range 100 000–3 000 000 ng/g l.w. (Aguilar and Borrell, 1994), are several magnitudes above levels in the Svalbard white whale. Furthermore, ΣPCB levels in the Svalbard white whales are also an order of magnitude below the threshold that has been suggested to cause impaired reproduction in Baltic seals (50 000 ng/g l.w.; Bergman and Olsson, 1985). Thus, we conclude, that OC concentrations in Svalbard white whales, and in particular ΣDDT and ΣPCB, are well below levels that are suggested to cause effects in more polluted waters.

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