Arch. Environ. Contam. Toxicol. 16, 137–147 (1987)

Archives of Environmental Contamination and Toxicology

Levels of Organochlorine Chemicals in Tissues of Beluga Whales (*Delphinapterus leucas*) from the St. Lawrence Estuary, Québec, Canada

D. Martineau*,**,[1], P. Béland*, C. Desjardins†, and A. Lagacé**

*Centre de Recherche en Ecologie des Pêches, Pêches et Océans Canada, 310 av. des Ursulines, Rimouski, Québec, Canada, G5L 3A1, **Faculté de Médecine Vétérinaire, Université de Montréal, C.P. 5000, St-Hyacinthe, Québec, Canada, J2S 7C6 and †Laboratoire régional d'Inspection, Pêches et Océans Canada, 1001 Pierre Dupuy, Longueuil, Québec, Canada, J4K 1A1

Abstract. High levels of organochlorine chemicals (OC) were found in the blubber of 26 stranded carcasses of beluga whales from an isolated population in the St. Lawrence Estuary (Québec, Canada). These compounds accumulated with age in both sexes, being consistently more concentrated in male tissue; high and variable concentrations were found in four juveniles. Lower levels in females are best explained through massive transfer to the newborn during lactation, resulting in juvenile OC concentrations equal to or higher than in adult males. Concentrations in the liver and kidney expressed on a lipid basis suggest dynamic OC exchange between tissues. The adipose tissue concentrations reported here were higher or equal to those found in some pinnipeds, in laboratory animals, and in domestic animals with severe reproductive failure. These findings suggest that OC contamination is a major factor in the non-recovery of the St. Lawrence beluga population over the last decades.

Several marine mammals of Eastern and Arctic Canada have been studied with regards to organochlorine contamination. Grey seals (*Halichoerus grypus;* Addison and Brodie 1977), ringed seals (*Pusa hispida;* Addison and Smith 1974), harp seals (*Pagophilus groenlandicus;* Addison *et al.* 1973, Frank *et al.* 1973), harbour seals (*Phoca vitulina;* Gaskin *et al.* 1973), harbour porpoises (*Phocoena phocoena;* Gaskin *et al.* 1971, 1982, 1983) and Arctic beluga (*Delphinapterus leucas;* Addison and Brodie 1973) were the objects of as many reports. In many cases, the animals sampled belonged to widely spread populations, many of which migrate extensively. Therefore, contaminant levels could not be related to any regional source, nor could potential effects on the population as a whole be evaluated. The same remarks hold true for most reports worldwide on contamination of marine mammals, a good number of which refer to isolated cases of animals from uncertain origin.

By contrast, St. Lawrence beluga whales (*Delphinapterus leucas*) form a small population of a few hundred whales indigenous to and restricted to the St. Lawrence Estuary and to parts of the Gulf of St. Lawrence. For most of the year, the population is centered at the junction of two important carriers of industrial effluents in North America, the Saguenay and the St. Lawrence rivers. Feeding studies carried out in the late 1930s (Vladykov 1946) confirmed that this long-lived species had a very broad food base, therefore suggesting that it should integrate the flow of non-degradable substances through the food webs of this system. Until recently however, this population had been neglected with regard to toxicological studies.

Since the Fall of 1982, dead animals have been collected from this population. Recent papers by Martineau *et al.* (1985, 1986) described lesions from two specimens, one of which was also included in a discussion of organochlorine profiling by Massé *et al.* (1986). Two of the above papers suggested that contamination may be an important factor for the health of the population. Herein are presented results of toxicological analyses from 26 carcasses from which tissue samples were taken. Although

[1] Address correspondence to Dr. D. Martineau, Dept. of Pathology, New York State College of Veterinary Medicine, Ithaca, New York, 14853-6401.

Table 1. Organochlorine contaminant concentrations (μg/g, wet weight) in the blubber of St. Lawrence belugas (n.d. = non determined; data on carcasses from Béland *et al.* 1986b)

Animal	Estimated time since death (days)	GLG	Sex	Lipids %	PCBs	ΣDDT	DDT	DDE	DDD
100-82	?	40	F	n.d.	27.09	5.317	1.928	2.409	0.980
1-82	1	55	M	n.d.	312.5	225.6	53.8	136.0	35.8
2-82	2–5	43+	M	70.2	202.1	147.6	46.43	83.12	17.93
3-82	2–5	59+	F	96.3	53.33	22.14	16.31	1.20	4.62
1-83	2–5	56+	F	n.d.	110.8	46.22	14.36	23.79	8.073
4-83	−2	(0)	M	48.1	55.541	36.134	9.761	18.232	8.141
13-83	1–3	23	M	85.0	103.4	50.51	10.842	33.026	6.639
14-83	4–15	39+	M	91.7	148.5	55.41	14.90	30.56	9.943
114-83	?	2.25	M	89.2	69.3	12.93	2.437	7.650	2.843
15-83	2–4	36	F	100.0	17.70	3.150	0.850	1.550	0.750
18-83	−2	33	M	96.3	269.8	111.8	28.09	65.07	18.64
102-84	30–	43+	F	n.d.	27.08	8.107	3.647	2.658	1.802
2-84	2–4	48+	M	100.0	214.7	103.03	28.29	55.21	19.53
4-84	2–4	36	F	100.0	21.86	4.538	1.898	1.812	0.828
4a-84	3–5	0	?	n.d.	5.664	1.166	0.233	0.530	0.403
5-84	4–10	30?	M	100.0	159.9	26.37	0.281	21.18	4.910
6-84	4–10	7	M	69.6	150.0	17.74	1.181	11.47	5.095
9-84	−2	3.5	F	81.3	576.0	94.86	14.10	52.96	27.80
10-84	4–10	30+	F	99.8	21.42	2.544	0.931	1.177	0.436
11-84	4–8	46+	F	100.0	74.80	14.84	2.873	7.409	4.560
13-84	2–4	44+	M	97.3	196.7	64.96	12.84	42.18	9.936
100-85	10–20	46+	M	83.1	265.9	87.31	13.25	57.24	16.82
107-85	5–20	51++	M	100.0	112.2	87.60	1.058	51.46	35.08
117-85	5–20	43+	M	100.0	276.9	170.5	42.465	103.2	24.794
1-85	2–4	39+	M	100.0	165.7	94.78	26.408	54.35	14.025
2-85	2	49	F	100.0	36.86	13.10	5.348	4.455	3.292

over 25 individual contaminants have been identified, only polychlorinated biphenyls (PCBs), DDT and metabolites will be dealt with here. Results are compared to those from other studies on marine mammals, and some possible effects of the observed high levels on the population are discussed, on the basis of evidence from laboratory studies on other mammals.

Materials and Methods

Strandings of Belugas in the St. Lawrence

This study relates to carcasses of dead animals that had drifted onto the shores of the St. Lawrence, and reported to the laboratory by the public. Every year, the program is advertised in the media, governmental authorities are contacted and all communities along the shore are visited and given attractive posters requesting their help. Every effort is made to locate any carcass brought to the attention of the laboratory. To this date, and covering a period of 39 months, we have sampled three belugas in 1982, plus one reported by another observer; seven in 1983, ten in 1984 and five in 1985 (Table 1). Of all animals collected, 15% were found in April–May, 62% from June to September, and 23% from October to December (Béland *et al.* 1987b). The high number of potential observers during summer as well as adverse ice conditions from December to April would favor this type of distribution.

Whenever possible, standard measurements were recorded, tissues were sampled for contaminant analyses and a few teeth were removed. Aging was done by counting growth layer groups (GLG) on longitudinal sections of teeth (Sergeant 1973), adopting the standard of two dentine layer groups per year (Brodie 1982; IWC 1982). Tooth erosion and cessation of growth near the end of life may sometimes concur in producing an underestimation of the true age of the animal; such cases are shown by a + symbol in Table 1. Blubber thickness was measured dorsally and ventrally, on a section near the insertion of the pectoral fins. Fresh carcasses were brought to a necropsy room for autopsy, where tissue samples were taken for histopathology, parasitology, bacteriology, and virology.

Toxicological Analyses

In the field or necropsy room, tissues for organochlorine analyses were placed in acetone-rinsed glass jars, covered with aluminum foil, and frozen. Chemical analyses were made at the Capitaine Bernier Laboratory, Regional Inspection, Department of Fisheries and Oceans Canada, in Longueuil, Québec, Canada. This laboratory has obtained an average accuracy for polychlorobiphenyls of 13% over the period 1974–1985 in the Federal Interdepartmental Check Program (FICP).

All samples were kept frozen at −20°C until needed. Samples were homogenized, using commercial meat grinders having a mesh width of 4mm. For the quantification of organochlorine compounds, the homogenized aliquot was extracted by an acetone-hexane (1:1) mixture before processing in a gel permeation chromatograph using a column of SX-3 beads as described by Johnson *et al.* (1976). The eluate was concentrated before adding to a 2% deactivated Florisil® column as originally described by Mills *et al.* (1972). Two fractions were collected. Fraction I eluted by hexane contained heptachlorobenzene, DDE, polychlorobiphenyls and mirex; fraction II eluted by a 50.0:0.35:49.65 mixture of hexane-acetonitrile-dichloromethane contained twelve other pesticides of interest (McLeod and Ritcey 1978). Polychlorobiphenyls were quantified by gas chromatography (Sherma and Beroza 1980; Freeman 1981), using 2% OV-17 + 2.6% OV-210 packed columns. Three peaks were used to calculate the total Aroclor 1254® contents: DDE 127, 147, 177 (Reynolds 1971). Average recoveries were in the 90–110% range. Polychlorobiphenyls were not corrected for recovery data. A more complete description of the procedures has been published elsewhere (Desjardins *et al.* 1983). Figures referred to in this paper are given in μg/g (ppm) of total tissue wet weight except when indicated on a tissue lipid basis. Total fat was determined as in the AOAC (1980).

Results and Discussion

Twenty-six animals were submitted to analysis. In Tables 1 and 2, concentrations of PCBs and DDT metabolites in the blubber, liver and kidney are given on a wet weight basis, along with tissue fat contents for converting to concentrations on a lipid basis. The latter have been plotted for the blubber and for each sex in Figure 1 (PCBs) and Figure 2 (ΣDDT).

All drifted beluga whales from the St. Lawrence that have been examined to date were highly contaminated with organochlorine compounds. The blubber concentrations of total DDT were two orders of magnitude higher than in beluga whales from the Mackenzie Delta (Addison and Brodie 1973). In that population, a PCB detection limit of 0.5 ppm could not be exceeded. In St. Lawrence belugas, PCB and DDT levels were one order of magnitude above those found in grey seals from Sable Island, Nova Scotia (Addison and Brodie 1977), in harp seals from the Gulf of St. Lawrence (Addison *et al.* 1973), and in beluga whales from the Baltic (Harms *et al.* 1978). They were similar to and often higher than those of ringed seals from the Bothnian Bay (Helle *et al.* 1976), of California sea lions (Delong *et al.* 1973), and of harbour porpoises from Danish waters (Andersen and Rebsdorff 1976). In our sample, blubber ΣDDT levels are similar to those of harbour seals from the Bay of Fundy and the Gulf of Maine as reported by Gaskin *et al.* (1973), but lower than those found in harbour porpoises from the same area (Gaskin *et al.* 1971).

Table 2. Organochlorine contaminant concentrations (μg/g, wet weight) in the liver and kidney of St. Lawrence belugas (n.d. = non determined)

Animal	Tissue	% lipids	PCBs	ΣDDT	DDT	DDE	DDD
1-83	liver	1.71	0.525	0.136	0.014	0.095	0.027
	kidney	n.d.	2.814	0.951	0.130	0.574	0.247
4-83	liver	2.87	2.126	1.465	0.021	1.031	0.413
	kidney	0.76	0.802	0.450	0.095	0.261	0.094
13-83	liver	1.50	0.923	0.419	0.044	0.262	0.113
	kidney	0.74	1.366	0.779	0.207	0.402	0.170
14-83	liver	0.96	3.342	1.419	<0.001	0.856	0.563
	kidney	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
15-83	liver	4.69	0.256	0.163	0.075	0.027	0.061
	kidney	5.50	1.189	0.237	0.054	0.096	0.087
18-83	liver	0.82	3.712	1.899	0.050	1.151	0.698
	kidney	1.92	2.218	0.611	0.015	0.388	0.208
2-84	liver	3.12	3.587	1.426	0.007	1.007	0.412
	kidney	2.76	1.573	0.559	0.010	0.423	0.126
4-84	liver	1.50	0.227	0.033	0.005	0.020	0.008
	kidney	6.47	1.072	0.189	0.052	0.098	0.039
5-84	liver	0.85	2.789	0.641	0.017	0.445	0.179
	kidney	4.00	3.085	0.746	0.004	0.595	0.147
6-84	liver	0.79	2.771	0.303	0.016	0.182	0.105
	kidney	5.06	6.220	1.046	0.008	0.558	0.480
9-84	liver	n.d.	71.58	7.985	0.105	5.414	2.466
	kidney	1.64	31.65	3.461	0.170	2.408	0.883
10-84	liver	0.42	2.120	0.041	0.008	0.022	0.011
	kidney	8.85	0.399	0.177	0.035	0.098	0.044
11-84	liver	1.32	0.972	0.170	0.030	0.095	0.045
	kidney	0.54	5.750	1.043	0.016	0.601	0.426
13-84	liver	0.67	3.256	0.921	0.032	0.517	0.372
	kidney	2.79	5.297	1.550	0.096	0.937	0.517
100-85	liver	4.79	5.112	2.345	0.011	1.617	0.717
	kidney	0.79	1.751	0.679	0.033	0.487	0.159
1-85	liver	2.85	8.550	3.258	<0.001	2.199	1.059
	kidney	2.17	3.022	1.179	0.096	0.693	0.390
2-85	liver	0.80	0.566	0.127	0.018	0.051	0.058
	kidney	3.17	1.586	0.423	0.137	0.159	0.127

However, as measured on a lipid basis, hepatic organochlorine levels from the present sample of animals are mostly lower than in California sea lions (Delong *et al.* 1973) and lower than in harbour seals from the Wadden Sea (Reijnders 1980). This discrepancy could arise from the following. After ingestion, PCBs are channelled initially to the liver where the less chlorinated biphenyls may be metabolized, and thereafter directed to the less densely perfused tissues such as blubber (Matthews and Kato 1979). Thus, the presence of these products in the liver is related at least in part to the time elapsed since ingestion of contaminated food. The animals in the present series had drifted after dying of natural causes, and their stomachs were invariably empty. Furthermore, post-mortem examinations showed chronic diseases in many of them (Martineau *et al.* 1985; Martineau *et al.*, submitt.).

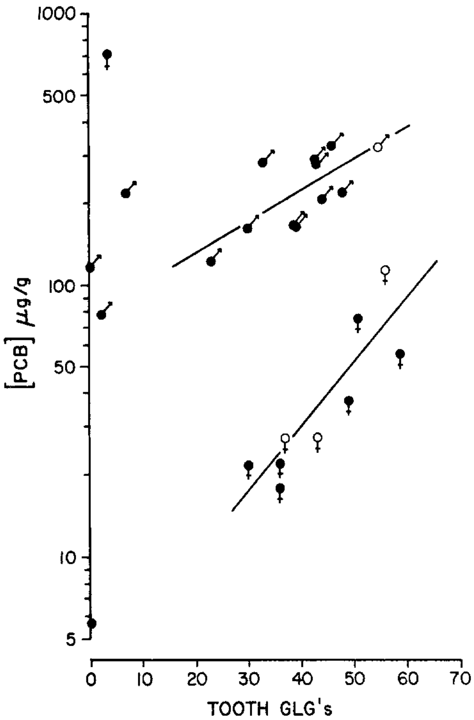

Fig. 1. Polychlorobiphenyl concentrations (µg/g, lipid basis) in the blubber of St. Lawrence belugas over age (expressed as the number of growth layer groups; Sergeant 1973). Regression lines for males, log PCB = 1.8838 + 0.0112 GLG (n = 11, sd = 0.1068), and for females, log PCB = 0.5282 + 0.0234 GLG (n = 9, sd = 0.1655), do not consider the five animals 3+ years and under. A few results available on a wet weight basis only (open symbols) were included in the computations, as they did not alter the results significantly

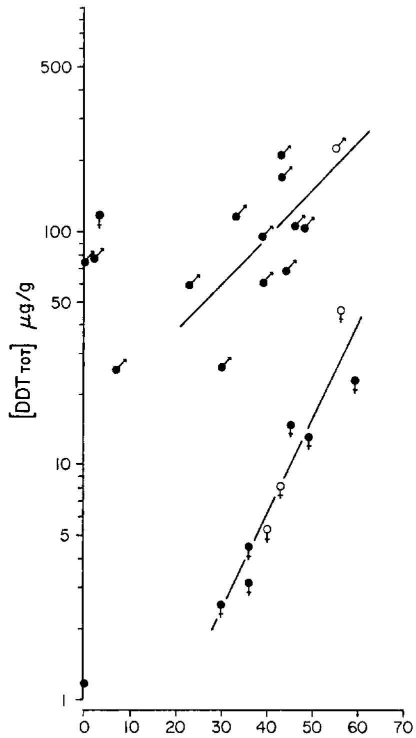

Fig. 2. DDT concentrations (µg/g, lipid basis) in the blubber of St. Lawrence belugas over age (expressed as the number of growth layer groups; Sergeant 1973). Regression lines for males, log DDT = 1.1745 + 0.0200 GLG (n = 11, sd = 0.0070), and for females, log DDT = −0.8680 + 0.0413 GLG (n = 9, sd = 0.0841), do not consider the five animals 3+ years and under. A few results available on a wet weight basis only (open symbols) were included in the computations, as they did not alter the results significantly

On the contrary, Delong *et al.* (1973) refer to animals that were killed for the purpose. It is not unlikely that they could have fed shortly before their death, therefore increasing their hepatic organochlorine concentrations, as opposed to belugas in the present study. Secondly, pinnipeds being smaller and more active mammals than monodonts, they could have a higher basal metabolic rate than beluga whales. Thirdly, trophic relationships may be invoked, as seals feed mostly on fish, while St. Lawrence belugas feed on a large array of invertebrates and fish (Vladykov 1946; Kleinenberg 1964). As organochlorines tend to concentrate at higher trophic levels, the average beluga meal may be less contaminated than the average meal taken by a seal. The long life span of the beluga would compensate for a lower rate of ingestion and result in a more dramatic accumulation in blubber.

Aguilar (1985) recently reviewed organochlorine contamination in cetaceans in the context of compartmentation in various tissues. His analysis showed that, in a long-term exposure, although the absolute amounts of lipophilic compounds are higher in the blubber, they maintain a certain proportionality with other organs. In the present sample, when expressed on a wet weight basis, concentrations are invariably higher in the blubber, being from 10 to 350 times those in the liver or kidney. However, when expressed on a lipid basis, the differences are equally often reversed, varying between 0.04 and 5.09 (Table 3). Overall, the distribution of organochlorines approaches proportionality with tissue fat contents, although still being slightly higher in the blubber. Individual differences are striking, but remain to be explained.

The measure of the organochlorine contamination of given populations of marine mammals through analyses of samples of drifted animals has been much criticized (Bergman 1981). In a study of seals from the Baltic, Bergman (1981) found no correlation between age and residue levels in a sample of animals found dead on the shore, while such a correlation existed for a sample of killed animals. The range and mean for organochlorine levels were

Table 3. Comparison of ratios of blubber to liver and of blubber to kidney organochlorine concentrations in St. Lawrence belugas, as measured (a) on a wet weight basis, and (b) on a lipid basis (n.d. = non determined).

	Wet weight				Lipid basis			
	Blubber/liver		Blubber/kidney		Blubber/liver		Blubber/kidney	
Animal	PCBs	DDT	PCBs	DDT	PCBs	DDT	PCBs	DDT
1-83	211.1	339.9	39.4	48.6	n.d.	n.d.	n.d.	n.d.
4-83	26.1	24.6	69.2	80.3	1.56	1.47	1.09	1.26
13-83	112.0	120.5	75.7	64.8	1.98	2.13	0.66	0.56
14-83	44.4	39.0	n.d.	n.d.	0.47	0.41	n.d.	n.d.
15-83	69.1	19.3	14.9	13.3	3.24	0.91	0.82	0.73
18-83	72.7	58.9	121.6	183.0	0.62	0.50	2.43	3.65
2-84	59.9	72.3	136.5	184.3	1.87	2.25	3.77	5.09
4-84	96.3	137.5	20.4	24.0	1.44	2.06	1.32	1.55
5-84	57.3	41.1	51.8	35.3	0.49	0.35	2.07	1.41
6-84	54.1	58.5	24.1	16.9	0.61	0.66	1.75	1.23
9-84	8.0	11.9	18.2	27.4	n.d.	n.d.	0.37	0.55
10-84	10.1	62.0	53.7	14.4	0.04	0.26	4.76	1.27
11-84	76.9	87.3	13.0	14.2	1.02	1.15	0.07	0.07
13-84	60.4	70.5	37.1	41.9	0.42	0.49	1.07	1.20
100-85	52.0	37.2	151.9	128.6	2.99	2.15	1.44	1.22
1-85	19.4	29.1	54.8	80.4	0.55	0.83	1.19	1.74
2-85	65.1	103.1	28.7	30.9	0.52	0.82	0.74	0.98
Averages	64.4	77.2	56.9	61.8	1.19	1.10	1.57	1.50

also higher for drifted carcasses, respectively, than for animals that had been killed. The author concluded that drifted animals did not appear to constitute a homogeneous group and therefore did not appropriately reflect contamination in a presumed population of origin. The situation is different in this study, where organochlorine levels allow to split the sample into three groups, namely suckling or recently weaned whales, other males, and other females. For the latter two groups, organochlorine levels in the bubbler relate well to both age and sex (Figures 1, 2).

In rats contaminated by hexachlorobiphenyl and submitted to a restricted food intake, the concentration of that compound decreased at the same time as the adipose tissue mass (Wyss *et al.* 1982). However in some seal species, Addison and Smith (1974), Donkin *et al.* (1981) and Drescher *et al.* (1977) reported that organochlorine concentrations were inversely proportional to blubber thickness. The first study demonstrated a weak correlation limited to males in the ringed seal (*Pusa hispida*). The second study showed a correlation for both sexes, but a stronger one for males, while in the third study, on the harbour seal (*Phoca vitulina*), sexes were not determined. Such a relationship was not found in harp seals (*Pagophilus groenlandicus;* Addison *et al.* 1973; Frank *et al.* 1973) nor has it been determined in cetaceans. The two former studies suggest that lipid metabolism and/or organochlorine deposition in fatty tissues are different in the male seal and therefore probably influenced by hormonal mechanisms and/or reproductive cycle. Moreover, blubber thickness in seals is known to depend on many factors, such as sex, lactation in females, behavioral cycles, season and variations in food intake (Lockley 1966; Bryden 1972; Ling 1974). In addition, organochlorine ingestion by marine mammals depends on seasonal changes in the body fat contents of prey fish, which may in turn show some relation to seasonal organochlorine flushing in the environment (Olsson *et al.* 1977). Consequently, a discussion correlating PCB levels with blubber thickness would also have to consider concurrent changes in prey availability and quality, physiological fasting and even behavioral and reproductive cycles altering lipid metabolism. The importance of the latter was demonstrated in pregnant rats contaminated with hexachlorobiphenyls; PCB concentrations in adipose tissue were highest on the day of birth (Vodicnik and Lech 1980).

In some mysticetes, blubber was clearly identified as an important energy store (Lockyer *et al.* 1985). By contrast, blubber thickness in odontocetes seems little influenced by fasting, apparently because blubber fat is not readily available as a source of calories (Andersen and Rebsdorff 1976; Britt and Howard 1983). In belugas, lower dorsal blubber thicknesses (4 to 11 cm, median 7 cm) were found than in a much larger sample of kills from the same population taken 50 years ago (7 to 27 cm, median 15 cm; Vladykov 1944). However, measurements were not taken at the same body site in both

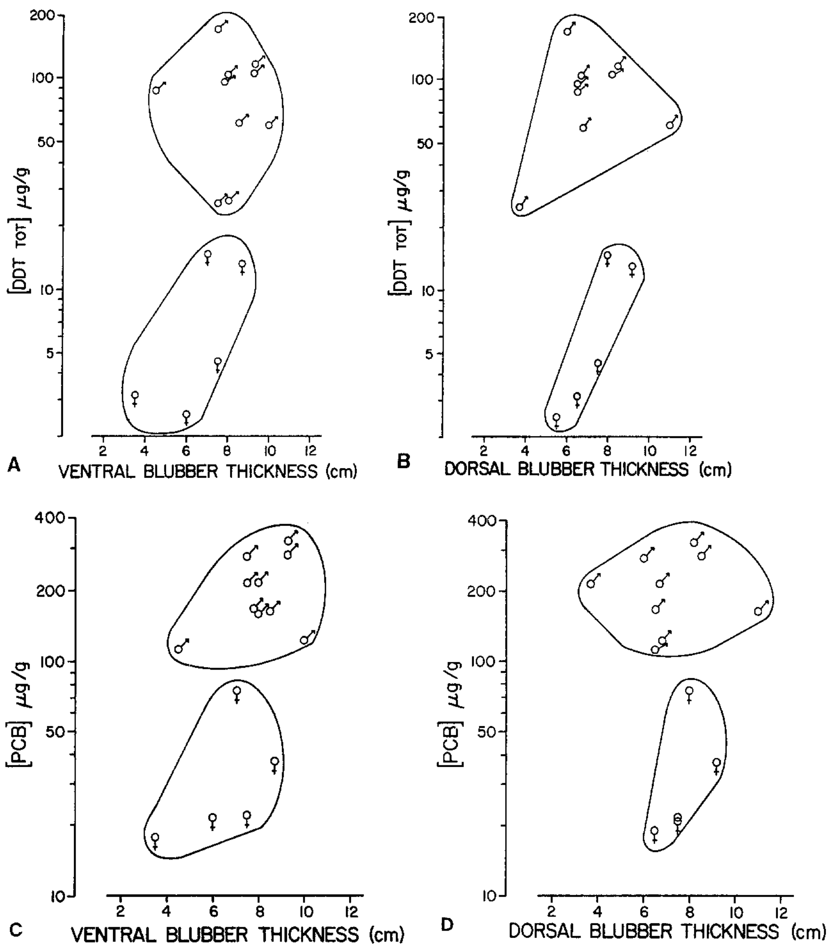

Fig. 3. Relationship between tissue thickness and organochlorine concentrations (μg/g, lipid basis) in the blubber of adult St. Lawrence belugas. a,b: DDT; c,d: PCBs

studies, and it was noted that blubber thicknesses varied from 9 to 19 cm over only a 75 cm distance on two carcasses. Kleinenberg *et al.* (1964) also stated that blubber thickness showed marked topographical variations. Nevertheless, it is obvious from the present sample of beluga whales that there is no inverse relationship between blubber thickness and organochlorine concentrations (Figure 3). Males had more blubber ventrally, while dorsally, thickness was more variable in males than in females; as contaminant levels in males are consistently higher than in females, emaciation alone would not explain organochlorine concentrations. In fact, the data would suggest that in adults of both sexes, organochlorine levels increase with blubber thickness (Figure 3).

The mean DDE/ΣDDT ratio determined from the blubber of 25 beluga whales is 0.52 (Figure 4 and Table 1). Ratios for the liver and kidney are respectively 0.60 ($n = 17$) and 0.59 ($n = 16$) (Table 2). These ratios are comparable to those from a recent analysis of cetacean blubber (Aguilar 1984). Following Aguilar, this would suggest that no recent significant input of DDT has occurred in the St. Lawrence estuary. Indeed, that author found that DDE/ΣDDT ratios in cetaceans over the last twenty years have slowly increased. This reflects the conversion of DDT to DDE in the environment and the trend towards an equilibrium between DDT and its metabolites that will be attained when the ratio will have reached approximately 0.60. Furthermore, in the present study, ΣDDT/PCB ratios are low, variable, and always inferior to 1 (Figure 5). This confirms the lack of recent DDT input into this ecosystem.

Ratios of individual DDT metabolites in various tissues from several species of cetaceans were compiled by Aguilar (1985). The data showed that metabolically more active tissues such as liver and kidney degraded DDT into its metabolites more completely than a reserve tissue such as blubber. In St. Lawrence beluga whales, such ratios were quite variable between individuals (Tables 1 and 2), but patterns became apparent when animals were grouped by sex and age categories. Thus in adults, the average proportion of non-metabolized DDT

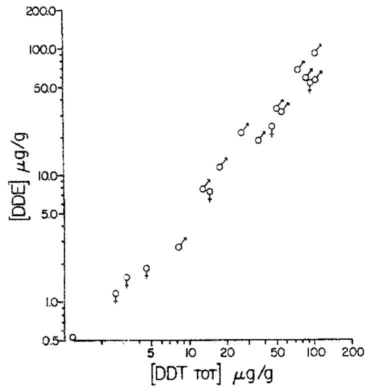

Fig. 4. Relationship between DDE and DDT concentrations (μg/g, wet weight) in the blubber of St. Lawrence belugas

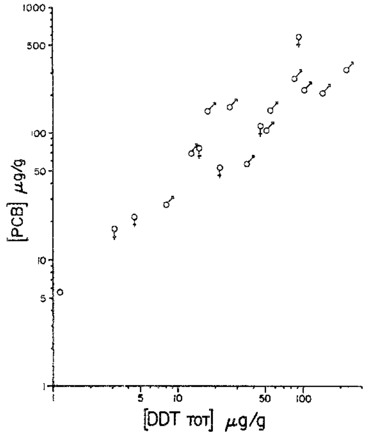

Fig. 5. Relationship between PCBs and DDT concentrations (μg/g, wet weight) in the blubber of St. Lawrence belugas

was two (females) to six (males) times higher in blubber than in kidney or liver. Considering only those individuals for which data is available for all three tissues (six females and eight males), two-way ANOVA shows that the proportion of non-metabolized DDT differs significantly between sexes ($p < 0.001$) and between tissues ($p < 0.001$). The between tissue difference is however not significant in females taken alone (one-way ANOVA, $p < 0.0960$), but is highly so in males ($p < .002$). This means that the dynamics of DDT metabolites are different in both sexes. The observed differences could result from several processes.

Firstly, the differences could originate from females having a diet richer in non-degraded DDT. They may be feeding at different locations and/or using food items produced at different levels of the food web. There is evidence that both exist and might therefore contribute to the observed differences. Thus, female and young are, at least in summer, generally segregated from males (Pippard 1985; Béland *et al.* 1987a). Vladykov (1946), based on extensive data from stomach contents, concluded that the diet of females and young in the St. Lawrence was in some respects different from that of adult males. Thus, while capelin and sand lance were staple foods for both sexes, males consumed more large fish of many species, while females and young preferred smaller food, and in particular benthic polychetes (*Nereis* sp.) and shrimps. Similar differences were also found in Alaska (Seaman *et al.* 1982) and in the USSR (Kleinenberg *et al.* 1964).

A second hypothesis would hold that the production of metabolites is enhanced in males as opposed to females. The degradation of DDT into its metabolites is favored by increased enzymatic activity. As ratios of non-metabolized DDT for all three tissues examined are lower in males than in females, they may indicate higher enzymatic activity in active tissues of males (liver and kidney), with a corresponding difference in so-called storage tissues (blubber). Causative factors for such an increased activity are a matter of conjecture, but the following could reasonably be invoked. On the one hand, there may be a basic difference between sexes in metabolic pathways for organochlorines (see also below); on the other hand, the differences could result from the effect of the much higher overall burden of organochlorines in males than in females. Several PCB isomers found in three animals of the present series (Massé *et al.* 1986) were shown to be hepatic inducers of many different cytochromes in rats (Safe 1984). By analogy therefore, a higher degradation rate of DDT would be expected in beluga males than in females, resulting in relative enrichment in DDE + DDD in males. Similarly, very young specimens ($n = 4$) have total PCB burdens as high or higher than those of adult males, and their DDT metabolite ratios are similar to those of males.

The presence of a possible effect of total PCB body burden on DDT metabolite enrichment could be tested by examining the relationship between the ratio of DDT over ΣDDT in liver or kidney and total body burden (Tables 1 and 2). PCBs in the blubber were used as a measure of total body burden, as they are thought to be major mono-oxygenase system inducers. Only male belugas were considered, as their higher burdens should provide a more adequate test, while avoiding the possibly confounding effects of pregnancy and lactation history. Linear correlations of the ratios in liver or kidney over log-transformed total burdens in the same animals are negative for both tissues as predicted, but not high enough to establish significance (liver $r = -0.46$, $n = 9$, $p > 0.10$; kidney $r =$

-0.63, n = 8, $0.05 < p < 0.10$). The high variance suggests that larger samples would be required to allow a definite answer to this question.

Finally, the higher ratios of non-metabolized DDT in females could originate from differences in the dynamics of organochlorine transport and accumulation in males and females. The transfer of organochlorines through placenta and mostly through lactation has been demonstrated in many mammal species (Platonow and Chen 1973; Fries *et al.* 1973; Addison and Brodie 1977; Vodicnik and Lech 1980). The overall lower burdens in female belugas in the present study can be inferred to result from a similar process, whereby females have an unloading mechanism unavailable to males. Then, the particular ratios of DDT metabolites in females would result from differential selection of various organochlorines during transfer processes.

The low PCB concentrations found in the tissues of a stillborn (animal 4a-84) reflect the minimal transplacental passage known to occur in some other mammals. However, the three other immatures under three years of age (Figures 1, 2 and Table 1) show high and variable tissular concentrations. For these animals, maternal milk had likely been a major or exclusive source of food since birth, as lactation in belugas lasts for 20–24 months (Brodie 1971). It seems probable that the variability of contaminant concentrations would be related to the ages, and consequent total burden, of their mothers. The transmammary transfer of organochlorines demonstrated in different mammal species obviously also occurs in beluga whales.

PCB concentrations in grey seal milk from Sable Island were 1.3 to 7.6 μg/g (Addison and Brodie 1977). The observation that female grey seals showed no increase in residue burdens with age was explained through an equilibrium between annual intake from food and excretion through lactation. By contrast, in the St. Lawrence, organochlorine burdens of female belugas increase exponentially with age, converging towards those of males (Figures 1, 2). Thus, although females are on average less contaminated than males, the annual rate of increase of their adipose tissue burden (1.11 for PCBs and 1.21 for ΣDDT) is higher than that of males (1.05 and 1.11 respectively). This indicates an overall higher input of contaminants in females, accounted for in part by the increase in food intake needed to sustain lactation (estimated at 30% from a single female in captivity; S. Hewlett, Vancouver Aquarium, Box 3232, Vancouver, V6B 3X8, pers. comm. 1986). In a review summarizing analyses from St. Lawrence sediments (Trépanier 1984), eighteen localities within the main beluga range averaged 0.07 μg/g of PCBs; two further localities within this range had 2.54 μg/g (Saguenay) and 27.4 μg/g (lower Estuary). It has been shown elsewhere that polychetes and shrimps can concentrate PCBs by factors of 10.8 to 1.9 (McLeese *et al.* 1980). The apparent differences in the diets of males and females may result in higher overall intakes of contaminants by females. Detailed studies of contaminant flow through the St. Lawrence food webs are needed in order to clarify this point.

There is an unfortunate lack of data on young females, but it would seem that the first pregnancies followed by lactation early in adulthood would result in a significant reduction in body burden, so as to bring females on a curve different from that of males. The role of additional pregnancies in reducing body burden would then be progressively less important with age. The reasons for this are unknown at the present time, but it may be a direct consequence of a natural and progressive reduction in age-specific birth rates, as is known to occur in Alaska (from 0.333 at age 6 to 0.125 by age 29; Burns and Seaman 1985). This phenomenon may perhaps be even more pronounced in the St. Lawrence.

Numerous studies have reported reproductive and hormonal dysfunctions in various mammals, including marine species, following PCB contamination (Table 4). In animals showing such dysfunctions, adipose concentrations were lower or equal to those of St. Lawrence beluga whales. Oestrogenic effects of PCBs were also demonstrated in birds (Platonow and Funnell 1971). DDT metabolites are similary potent oestrogenic substances in some mammals (Bitman and Cecil 1970).

The proportions of calves and juvenile beluga whales in the St. Lawrence are lower than in Alaskan populations (Burns and Seaman 1985; Béland *et al.* 1987a). If this difference is real, it would indicate a decreased birth rate and/or increased juvenile mortality. The degree of organochlorine contamination found in the population could explain a decreased birth rate, as comparable or lower concentrations have such an effect in other animals. Whether such is the causative factor in the present case cannot yet be demonstrated, and continued analysis and monitoring of the population are required to confirm the importance of such phenomena. The evidence would not be detectable by examination of ovaries since domestic animals ingesting PCBs and contaminated pinnipeds seemed to ovulate normally. No ovarian abnormalities could be detected in female beluga strandings from the St. Lawrence, except for a single case of Granulosa cell tumor (Martineau *et al.*, submitt.).

Table 4. Polychlorobiphenyl concentrations of adipose tissue in various mammals with associated reproductive and hormonal dysfunctions

Species	Concentration in adipose tissue (ppm)	Reproductive and hormonal dysfunctions	Study
Boars	13.7–245.1 n = 4, ww[a]	Decreased urinary excretion of natural steroids	Platonow *et al.* 1972
California sea lions[b]	17.1 (n = 4), 112.4 (n = 6) ww	Higher mean PCB and DDT concentration in animals aborting (112.4) than in animals normally pregnant (17.1 ppm)	DeLong *et al.* 1973
Mice	44–424 n = 23, ww	Prolonged oestrous cycle, decline in the number of implanted ova	Orberg and Kihlstrom 1973
Sows	4.1–19.8 n = 5, ww	Increased fetal deaths	Hansen *et al.* 1975
Ringed seals[b]	56 (n = 15), 77 (n = 26)	Unusually numerous non pregnant female seals had higher PCB concentrations (77 ppm) than pregnant seals (56 ppm)	Helle *et al.* 1976
Rhesus monkeys	71 n = 3	Embryonic resorption, abortion, stillbirth, irregular menstrual cycles	Barsotti *et al.* 1976
Minks	14–280 n = 149	Embryonic resorption	Jensen *et al.* 1977
Beluga whales[b]	5.7–576 n = 25, ww	———	This study

[a] wet weight basis
[b] indicates wild animals; others were experimental

Conclusions

A recent review of the St. Lawrence beluga population (Reeves and Mitchell 1984) indicated that it had decreased dramatically during the present century. Heavy hunting into the 1940s reduced the population from about 5000 to several hundreds. Although hunting was reduced considerably afterwards and the species has been completely protected since 1979, it is estimated that there are presently only a few hundred animals remaining (Pippard 1985; Béland *et al.* 1987a). Several factors, such as a physiological or behavioral limitation to reproduction, a skewed age composition, excessive predation, contamination, habitat destruction and limited food resources have been proposed to explain the failure of the population to recover over the last 35 years (Reeves and Mitchell 1984).

While some of the above factors remain open to speculation, predation and competition for food do not appear likely. Thus, the formerly occasional killer whale (*Orcinus orca*), the more likely predator, has been seen even more rarely in the last decade in spite of an increased interest in whale watching. While belugas are the cetaceans commonly found in the upstream half of their summer range, competition for food likely occurs in the downstream part of the range where seals, fin whales, minke whales and an odontocete school are found. However, in spite of uncertain population numbers, it can be estimated that the combined present biomasses of all cetaceans within the beluga range does not reach one-half of that of belugas alone at the turn of the century. On the other hand, the present study has evidenced contamination by compounds known to induce severe reproductive dysfunctions in many other animal species at similar and often lower adipose concentrations. It is therefore suggested that organochlorine contamination should be considered as a prime cause for the low recruitment observed in this population. As there is also ample evidence in the literature that PCBs are strong immunosuppressive agents (Safe 1984), they may also contribute to mortality. A description of the lesions found in the animals of the present series will be the subject of another paper.

Acknowledgments. We are much indebted to local residents and colleagues who reported and made carcasses available for study. We thank Dr. R. Laflèche, Dental School, Laval University, for his expertise towards reading growth layers on sectioned teeth. This work would have been incomplete without colleagues, staff and students at several institutions (Centre de Recherche en Ecologie des Pêches, Rimouski; Faculté de Médecine Vétérinaire, Univ. de Montréal, St. Hyacinthe; Laboratoire du M.A.P.A.Q., Rimouski; INRS-Océanologie, Rimouski) who helped with recovering carcasses and during necropsies; in particular, we thank Paul Robichaud, Jean Caron, Claude Soucy, Richard Plante, Minos Pagonis, as well as Yves Saintonge, Richard Bourassa and Jacques Lagacé. Finally, we thank Richard Maltais, Gérald Leblanc, François Brunet and Pierre

Arpin, technicians at the Capitaine Bernier Laboratory in Longueuil, for chemical analyses.

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Manuscript received May 23, 1986 and in revised form August 18, 1986.