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Organochlorines and mercury in pilot whale blubber consumed by Faroe islanders

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Abstract

Some 22 000 pilot whales (*Globicephala melaena*) were taken in the Faroe Islands between 1970 and 1992. It is known that tissues from these animals are widely consumed by the islanders. The position of these animals at the apex of a direct marine food chain renders them liable to accumulate toxic chemicals, such as metals and organochlorines. Although the consumption of contaminating metals in pilot whale tissues has been studied, the significance of blubber as a dietary source of organochlorine compounds has not been fully considered. This study reports levels of organochlorine and mercury contamination in the blubber of pilot whales taken in two Faroese kills. Published estimates of pilot whale tissue consumption are used to evaluate dietary organochlorine intake in relation to established national and international guidelines and clinical studies conducted in the North American Great Lakes.

Key words: Pilot whales; Whaling; Organochlorines; Diet; Mercury; Faroes

1. Introduction

The North Atlantic Faroese pilot whale drive fishery takes between 500 and 3000 whales annually, with some 15 000 whales having been killed between 1970 and 1984 (Julshamn et al., 1987). Between 1985 and 1992, the total catch was reported as 11 864 animals (Sanderson, K., pers. commun.) Tissues from these animals were widely consumed by many of the islands' population of 45 000. It has been estimated (Andersen et al. 1987) that between 1970 and 1984 consumption of pilot whale tissues on a per capita basis varied between 82 and 555 g/week with a mean value of 257 g calculated on the basis of pilot whale landing statistics and assuming a percentage ratio between meat, blubber and liver of 65:32:3 by vol.

These figures assume that the total catch is consumed in equal amounts by all of the islanders. Daily blubber consumption figures were also de-

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rived from a 1982 survey of 331 Faroese adults (Vestergaard and Zachariassen, 1987). Consumption of pilot whale muscle was estimated at 12 g/day and blubber consumption at 7 g/day (see also Grandjean et al., 1992).

The pilot whale, *Globicephalus melaena*, is a carnivore, feeding largely on squid but also on pelagic fish and crustaceans (Wiborg et al., 1982; Desportes and Mouritsen, 1988). Accordingly, these animals are at the apex of a direct marine food chain which renders them liable to bioaccumulate both toxic metals and organic chemicals.

Consumption of pilot whale blubber and muscle can therefore act as a source of organochlorine and metallic contaminants in the diet of Faroe Islanders. In a comparative study of human tissues sampled from the Norwegian and Faroe populations (Juhlshamn et al., 1989), it was found that median levels of mercury in kidney and liver of Faroese islanders were approximately ten times higher than in residents of Bergen in Norway. This difference was attributed by the authors to the high consumption of pilot whale tissues by the Faroese.

Further, Grandjean et al. (1992), in a study of a small Faroe Island fishing community, found that mercury exposures were primarily determined by consumption of pilot whale tissues rather than the fish component of the diet which was primarily cod.

As a result of the potential health risks associated with excessive dietary intake of mercury and cadmium, the Faroes Food and Environmental Institute has already warned that pilot whale flesh and blubber should only be eaten once a week. It is recommended that kidney and liver should not be consumed at all (Anon, 1985; Gibson-Lonsdale, 1990). It has, however, been reported that a 'large meal of whale meat and blubber twice a month still places Faroese people well within the suggested international maximum limits for the intake of substances such as heavy metals and PCBs' (Sanderson, 1991). Apart from the estimates of consumption given by Julshamn et al. (1987) and the limited survey outlined by Vestergaard and Zachariassen (1987) there are no data reporting the actual consumption of pilot whale tissues in the literature.

Additionally, the concentrations of persistent organochlorines which occur in pilot whales have only been sporadically reported and these have generally been sampled in areas other than the Faroe Islands. Data relating to pilot whales caught in the Faroe Islands are restricted to determinations of PCBs, DDT and DDE (Borrell, 1993; Borrell and Aguilar, in press). This paper reports on contaminant levels in the blubber of pilot whales taken in the fishery in 1986 and discusses the implications of consuming contaminated tissues in relation to established advisory limits and the findings of studies of contaminated fish consumption in the Laurentian Great Lakes.

2. Materials and methods

Samples of blubber were collected from pilot whales killed at Nordragota on the 11th July and at Leynar on the 12th July 1986 in the Faroe Islands. The sampling was conducted as an independent exercise. At Nordragota the total kill of 35 whales was sampled but at Leynar only 15 animals, part of the total kill, could be sampled. Where possible, samples were taken from the ventral surface of the whales, caudal to the head. At Leynar samples were also taken from the placenta of a pregnant female, her foetus and umbilical cord. Another pregnant female (blubber and placenta) and an isolated foetus were also sampled. The 'skinn' size of each whale — a traditional estimate of the size of each animal assigned by the hunters — was recorded for the Nordragota samples. One 'skinn' is approximately equivalent to 75 kg of usable meat and blubber and is essentially a subjective estimate based upon the measured length of the animal from blowhole to anus (Gibson-Lonsdale, pers. comm.; Sanderson, 1991). This measure is used to apportion shares of each whale carcass.

Samples were taken in precleaned screw top jars and rapidly deep frozen after collection. All samples were transported frozen at −20°C to the analysing laboratory. Analytical methods were as described in Johnston et al. (1991). Organochlorine analyses were conducted on approximately 0.7 g of tissue. This was homogenised using a grinding agent composed of acid washed sand and

Table 1
Concentrations of organochlorine compounds and mercury in samples of blubber tissue of pilot whales (*Globicephala melaena*) from Nordragota ($\mu g\ g^{-1}$ wet wt.) HEL, hexane extractable lipid; (Sample 3 was taken from the head of whale No. 2)

Sample No.	Whale No.	Size (skins)	%HEL	Gamma-HCH	H.EPOX.	DDE	Dieldrin	PCB	Hg
1	1	v	89.9	0.35	0.84	12.30	2.98	20.97	0.08
2	2	viii	66.8	0.30	0.62	9.49	2.22	11.53	0.24
3	2	viii	99.4	ND	0.73	11.94	2.74	9.32	ND
4	3	xiv	79.2	0.54	ND	30.27	1.69	28.74	0.61
5	4	x	82.1	0.22	0.40	1.58	0.70	6.40	0.61
6	5	viii	84.9	0.43	ND	18.98	1.75	19.49	0.12
7	6	vii	103.6	0.21	0.56	1.61	0.98	1.82	0.28
8	7	v	87.6	0.66	ND	21.11	2.21	23.82	0.09
9	8	xx	98.5	0.66	0.88	21.52	2.56	47.68	0.32
10	9	vi	91.1	0.39	ND	10.44	1.55	3.27	0.13
11	10	vi	87.1	0.23	ND	2.16	0.81	5.30	0.18
12	11	vii	107.2	0.22	ND	1.50	0.61	7.65	0.23
13	12	i	66.7	0.40	1.23	7.71	1.23	ND	ND
14	13	i	96.3	ND	0.62	10.53	1.69	4.90	ND
15	14	iv	89.1	0.54	0.96	26.76	2.08	38.57	ND
16	15	vii	69.2	0.12	ND	1.16	0.47	4.72	0.63
17	16	xx	88.0	0.52	0.78	17.86	2.18	32.67	0.26
18	17	v	86.5	ND	0.61	7.97	1.48	17.09	0.12
19	18	v	95.9	0.92	ND	19.04	2.59	34.66	ND
20	19	ix	99.2	0.34	ND	1.63	0.58	8.36	0.30
21	20	ix	89.6	0.50	ND	16.54	1.38	16.37	0.13
22	21	ix	91.2	0.22	ND	2.34	0.61	9.92	0.39
23	22	xii	81.0	0.44	0.35	1.23	0.57	7.13	0.39
24	23	ii	92.0	0.76	ND	6.46	1.96	35.04	ND
25	24	ix	89.1	0.52	ND	13.56	1.64	33.77	0.17
26	25	vi	90.7	0.25	ND	4.50	0.72	12.30	0.15
27	26	viii	94.3	0.26	ND	1.69	1.21	9.17	0.28
28	27	xv	86.7	0.34	ND	15.42	2.72	20.38	0.14
29	28a	vii	86.6	0.20	0.35	1.20	0.49	8.41	0.35
30	28b	vii	83.7	0.24	0.32	1.33	3.66	7.17	0.30
31	29	xx	85.8	ND	ND	17.64	2.80	21.66	0.26
32	30	ii	97.3	0.46	ND	15.29	3.55	21.19	ND
33	31	viii	93.1	0.43	ND	19.03	2.99	18.92	0.24
34	32	vi	90.5	0.44	ND	10.20	2.24	6.64	0.05
35	33	i	73.0	0.58	ND	26.89	5.15	29.03	0.26
36	34	viii	87.9	0.35	ND	13.19	2.56	7.65	0.06
37	35	v	89.7	0.43	ND	12.42	2.74	12.55	ND
38	36	v	88.4	0.41	ND	36.47	4.55	31.18	0.21
39	38	ix	91.3	0.24	0.40	1.63	0.61	8.70	0.12
Range			66.7–107.2	ND–0.92	ND–1.23	1.16–36.47	0.47–5.15	ND–47.68	ND–0.63
Mean			88.2	0.38	0.23	11.87	1.90	16.95	0.20

$n = 37$. ND, 'not detectable' — (i.e. below detection limits).
Detection limits were as follows: 0.005 $\mu g\ g^{-1}$ for gamma-HCH, 0.01 $\mu g\ g^{-1}$ for DDE, 0.01 $\mu g\ g^{-1}$ for dieldrin, 0.05 $\mu g\ g^{-1}$ for PCBs.

anhydrous sodium sulphate both held at 700°C for 5 h prior to use. The homogenates were repeatedly solvent extracted in a 1:1 pesticide analysis grade hexane/acetone mixture to give a final volume of 50 ml. Twenty-five milliliters of this was then evaporated to dryness to give the lipid weight for the sample and redissolved in 5 ml of hexane. One milliliter of extract was cleaned up by passing it through a column containing aluminium oxide previously held at 800°C for 4 h and subsequently deactivated by tumbling with 5% distilled water added. The column was eluted to give 5 ml of cleaned-up extract.

Analysis was performed on a Varian 3400 GC with electron capture detection using a 30-m DB210 capillary column, at a programmed temperature of 190°C. Identification and quantitation were achieved by comparison with a standard pesticide mixture and a PCB standard of Aroclor 1254. Analyte recovery was tested on samples spiked at the extraction stage as part of the analytical quality control procedures. Pesticide and PCB recoveries were equal to or greater than 94% in all cases. Detection limits were as follows: 0.005 $\mu g\ g^{-1}$ for gamma-hexachlorocyclohexane (lindane); 0.01 $\mu g\ g^{-1}$ for DDE and dieldrin and 0.05 $\mu g\ g^{-1}$ for PCBs.

Mercury analysis was conducted on approximately 2 g of material dried to constant weight over 72 h at 85°C. This was cold acid digested for 12 h in 10 ml of analytical grade nitric acid and brought to boiling for a final period of 1 h. Digests were made up to 25 ml volume and analyses conducted by cold vapour generation according to the method of Hatch and Ott (1968). Measurements were made on a Thermo-electron 151 background corrected atomic absorption spectrophotometer. Detection limits were established at 4 ppb wet weight. Recoveries of mercury from spiked samples were equal to or greater than 96% in all cases.

3. Results

Results for Nordragota are presented in Table 1 and for Leynar in Table 2 and are summarised below.

In samples from Nordragota, lindane (gamma-HCH) was above detection limits in all but four samples with a maximum recorded concentration of 0.92 $\mu g\ g^{-1}$ and a mean of 0.38. Heptachlor epoxide was found in 15 out of the 39 samples with a maximum value of 1.23 $\mu g\ g^{-1}$ and a mean of 0.23 $\mu g\ g^{-1}$. DDE and dieldrin were detectable in all blubber samples with ranges of 1.16–36.47 $\mu g\ g^{-1}$ (mean 11.87) and 0.47–5.15 $\mu g\ g^{-1}$ (mean 1.90), respectively. Polychlorinated biphenyls (PCBs) were detected in all but one sample with a maximum concentration of 47.68 $\mu g\ g^{-1}$ and a mean value of 16.95 $\mu g\ g^{-1}$. Mercury was detected in 30 of the samples from Nordragota with a recorded maximum value of 0.63 $\mu g\ g^{-1}$ and 0.20 $\mu g\ g^{-1}$ mean.

Results from the 16 adult blubber samples collected from Leynar are very similar to those from Nordragota. Lindane (gamma-HCH) was above detection limits in all but two samples with a maximum recorded concentration of 0.60 $\mu g\ g^{-1}$ and a mean of 0.30. Heptachlor epoxide was found in seven samples with a maximum value of 0.99 $\mu g\ g^{-1}$ and mean of 0.28 $\mu g\ g^{-1}$. DDE, dieldrin and PCBs were detectable in all blubber samples with ranges of 1.23–33.83 $\mu g\ g^{-1}$ (mean 15.9), 0.54–5.27 $\mu g\ g^{-1}$ (mean 2.12) and 8.42–64.19 $\mu g\ g^{-1}$ (mean 28.68), respectively. Mercury was also detected in all blubber samples taken from Leynar, range 0.08–0.54 $\mu g\ g^{-1}$ (mean 0.23). These contaminants were also present in all the foetal tissues, including samples of placenta, collected from this site (Table 2).

4. Discussion

4.1. Mercury contamination of blubber

The mercury concentrations in blubber reported in the present study are lower than, but comparable to, the mean value of 0.70 ± 0.28 $\mu g\ g^{-1}$ wet weight previously reported from nine pilot whales sampled in the Faroes in 1977 and 1978 (Juhlshamn et al., 1987). The blubber, however, appears to be a relatively unimportant reservoir of mercury in comparison to other tissues. Julshamn et al. (1987) additionally reported mean concentrations of total mercury as 3.3 ppm in muscle, ($N = 10$); 280 ppm in liver ($N = 8$); and 18 ppm in kidney, ($N = 6$) all as wet weight. There are few data published on the levels of mercury in pilot whales.

Table 2
Concentrations of organochlorine compounds and mercury in samples of blubber and other tissues of pilot whales [*Globicephala melaena*] from Leynar ($\mu g\ g^{-1}$ wet wt.). Samples 53 and 60 were from pregnant females.

Sample No. (Whale No.)	Notes	%HEL	Gamma-HCH	H.EPOX.	DDE	HEOD	PCB	Hg
40	Blubber	85.0	0.39	ND	33.83	4.09	27.63	0.12
41	Blubber	86.4	0.19	0.46	1.86	0.66	8.42	0.19
42	Blubber	83.8	0.48	ND	26.56	3.51	26.16	0.08
43	Blubber	82.2	0.51	ND	27.00	2.43	24.82	0.38
44	Blubber	92.8	0.60	ND	18.14	2.45	13.02	0.21
45	Blubber	89.4	0.22	0.27	1.23	0.54	11.09	0.28
46	Blubber	91.0	0.27	ND	18.61	1.77	38.95	0.19
47	Blubber	77.9	0.26	ND	17.21	1.74	37.61	0.54
48	Blubber	86.1	0.25	ND	28.41	5.27	51.68	0.08
49	Blubber	89.2	0.27	ND	25.84	2.36	48.11	0.34
50	Blubber	85.7	0.32	0.51	1.92	0.58	25.17	0.37
51 (23)	Blubber	82.7	ND	0.84	13.61	2.21	36.99	0.25
52 (13)	Blubber	92.8	0.25	ND	2.61	0.68	11.61	0.18
53 (18)	Blubber	91.7	0.58	0.97	19.73	2.50	28.47	0.13
54 (18)	Placenta of 53	1.8	0.27	0.23	0.56	0.35	11.72	10.8
55	Blubber of lactating female	88.5	ND	0.99	15.77	2.46	64.19	0.16
56	Mammary gland of 55	15.2	0.31	ND	3.95	0.79	14.61	4.51
57	Foetus I blubber	68.9	0.28	0.63	2.42	0.79	13.08	0.16
58	Cord of Foetus I	0.5	0.14	0.16	0.27	0.40	11.55	1.19
59	Cord of Foetus II	0.5	0.16	0.13	0.29	0.23	7.96	0.63
60	Blubber of mother II	86.0	0.25	0.45	2.14	0.64	10.06	0.23
61	Blubber of foetus II	48.5	0.30	0.42	1.68	0.48	11.50	0.89
62	Placenta of foetus II	1.0	0.24	0.20	0.38	ND	9.69	5.52
Range*		77.9–92.8	ND–0.60	ND–0.99	1.23–33.83	0.54–5.27	8.42–64.19	0.08–0.54
Mean*		87.0	0.30	0.28	15.9	2.12	28.68	0.23

*(Adult blubber samples)
$n = 16$. ND, Not detected (i.e. below detection limits); HEL, hexane extractable lipid.
Detection limits were as follows: 0.005 $\mu g\ g^{-1}$ for gamma-HCH, 0.01 $\mu g\ g^{-1}$ for DDE, 0.01 $\mu g\ g^{-1}$ for dieldrin, 0.05 $\mu g\ g^{-1}$ for PCBs.

Arima and Nagakura (1979) cite a mean mercury concentration in muscle of a sample of 12 long-finned pilot whales of 4.2 $\mu g\ g^{-1}$ (wet wt.) with a range of 3.0–5.2 $\mu g\ g^{-1}$. In addition, Muir et al. (1988) have reported the following mean mercury concentrations in the tissues of pilot whales stranded at Grand Beach, Newfoundland: blubber, 0.15 $\mu g\ g^{-1}$ ($N = 14$); kidney 9.09 $\mu g\ g^{-1}$ ($N = 15$); liver, 105 $\mu g\ g^{-1}$ ($N = 13$); muscle 3.26 $\mu g\ g^{-1}$ ($N = 15$). For another group stranded at Point Leamington ($N = 26$) they reported the following: blubber 0.13 $\mu g\ g^{-1}$, kidney 5.8 $\mu g\ g^{-1}$ liver 62.7 $\mu g\ g^{-1}$ and muscle 2.77 $\mu g\ g^{-1}$. These values were reported on a dry weight basis and sample moisture content ranged between 58 and 72.8% for the tissues analysed. Hence, the values found in the samples from the Faroe Islands in this study appear to be comparable to previously published data.

4.2. Organochlorine contamination of blubber

The organochlorine data, particularly the PCB levels (ranges 0.05–47.7 $\mu g\ g^{-1}$, mean 16.95 $\mu g\ g^{-1}$ at Nordragota and 8.4–64.19 $\mu g\ g^{-1}$, mean 26.68 $\mu g\ g^{-1}$ at Leynar) are consistent with other reported determinations. Determined organochlorine levels for long finned pilot whales are presented in Table 3 (Sources: Wagemann and Muir, 1984; Martin et al., 1987; Muir et al., 1988; Johnston and Stringer, 1988; Borrell and Aguilar, in press). The mean values of PCBs and total DDT reported by Borrell and Aguilar (in press) on a fresh weight basis for blubber of 90 pilot whales sampled in the Faroe Islands were 27.4 $\mu g\ g^{-1}$ and 18.78 $\mu g\ g^{-1}$, respectively. No ranges are given for these data but the mean values are consistent with the values reported in this study.

From published data it is apparent that the highest levels of contamination are found in samples obtained closest to industrialised areas. Nonetheless, levels found in the Faroese samples are high and this is consistent with the status of the waters of the North East Atlantic as the greatest global environmental reservoir of PCBs (Larsson, 1985).

4.3. Mercury contamination and blubber consumption

The World Health Organisation recommended a maximum weekly intake of organic mercury of 0.2 mg for a human of 70 kg weight. The European and Paris Commissions have adopted an Environmental Quality Objective based on EEC Directives 82/176 and 84/156 that fish flesh for human con-

Table 3
Levels of organochlorines determined from tissues of long finned pilot whales. Data taken from: Muir et al., 1988; Wagemann and Muir, 1984; Martin et al., 1987; Johnston and Stringer, 1988

Location	Approximate sampling date	Tissue	No. sampled	Mean DDT (ppm wet wt.)	Mean PCB (ppm wet wt.)
Southwest England	1975	Blubber	2	42.7	98.0
E. USA	1971–1975	Blubber	2	149.2	78.0
French coasts	1979	Blubber	7	94.2	189.0
Japan	1968–1976	Blubber	6	4.7	—
		Muscle	6	—	8.03
Shetland					
UK	1982	Blubber	4	—	42.0
UK	1982	Liver	4	—	1.05
Newfoundland	1980	Blubber	5 (m.)	11.9	9.03
Newfoundland	1980	Blubber	9 (f.)	4.70	3.46
Irish Sea, UK	1988	Blubber	1	—	15.6
Irish Sea, UK	1988	Liver	1	—	3.5
Faroes	1987	Blubber	184	16.45	23.53
Faroes (this study)	1986	Blubber	53	—	19.51

Key: m. = males; f. = females.

sumption should not contain more than 0.3 $\mu g\ g^{-1}$ of total mercury on a wet weight basis (MAFF, 1990). Standards adopted for maximum permitted mercury levels in seafood are 0.5 $\mu g\ g^{-1}$ in the US and Canada; 0.7 $\mu g\ g^{-1}$ in Italy, 1.0 $\mu g\ g^{-1}$ in Japan, Germany, Sweden and Switzerland and 1.5 $\mu g\ g^{-1}$ in Norway (Clark, 1989).

Mercury intake from the consumption of pilot whale tissues has been investigated in the Faroe Islands. Andersen et al. (1987) estimated that long-term intakes of total mercury and methyl mercury in the Faroes population were close to the critical level (WHO, 1976) at which negative health effects could be expected. They determined that between 1970 and 1984 the Provisional Tolerable Weekly Intake (PTWI) of total mercury (300 μg per person per week) was consistently exceeded while between 1979 and 1984 the methyl mercury PTWI (200 μg/person per week) was exceeded by between 2 and 4 times. Intake of cadmium was also found to exceed guideline levels by a factor of two in 1980 and 1981. Accordingly the authors recommended that the Faroes population should significantly restrict consumption of pilot whale tissues. Gibson-Lonsdale (1990) stated that pilot whale kidney and liver are no longer eaten in the Faroes for this reason, following Health Department warnings.

The mercury levels found in the blubber samples reported here contribute around 6% of the total dietary intake of mercury from consumption of pilot whales based on the relative proportions of tissues consumed (Andersen et al., 1987). Hence, consumption of blubber will not constitute a significant source of mercury to consumers of pilot whales in the Faroe Islands, relative to the consumption of other tissues, at the consumption levels given by Andersen et al. (1987) or Grandjean et al. (1992).

4.4. Organochlorine contamination and blubber consumption

By contrast, the consumption of whale blubber represents a significant exposure pathway for organochlorine compounds in Faroe Islanders. Using the formula given by Andersen et al. (1987) which suggests that 32% of the per capita consumption is attributable to a blubber component, then Faroe Islanders consumed between 26 and 178 g blubber weekly between 1970 and 1984 with an overall mean, over this period, of 82.84 g.

Using the PCBs as an example, then at a mean contamination level of 20.6 $\mu g\ g^{-1}$ this represents an intake of between 535.3 and 3666.8 μg of PCBs/person per week with a mean of 1706.6 μg. Assuming a mean adult body weight of 70 kg this translates to values between 1.09 and 7.48 $\mu g\ kg^{-1}$ body weight/day with a mean of 3.48 $\mu g\ kg^{-1}\ day^{-1}$. At the average consumption levels, reported from a survey of 331 Faroese adults (Grandjean et al., 1992), of 7 g of blubber daily, or 49 g/week, then the daily intake can be calculated as 2.06 $\mu g\ kg^{-1}$ body weight/day. This represents a mean annual intake of 48 mg of PCBs.

The US Food and Drug Administration (FDA) Tolerable Daily Intake (TDI) for PCBs (Swain, 1988) is set at 1 $\mu g\ kg^{-1}\ day^{-1}$ and this is widely accepted internationally. This is based upon a dose-response assessment derived from the study of human and animal toxicological data (Boyer et al., 1991). Even at the lowest consumption level, this TDI is exceeded in the Faroe Islands, while at higher levels of consumption reported in the literature the guideline could be exceeded by a factor of seven. On the basis of the average blubber consumption reported by Grandjean et al. (1992) guidelines were exceeded by a factor of two. Seven grams of blubber consumed per day equates to 196 g/month which is close to the maximum consumption of 200 g/month recommended by the Faroese Food and Environmental Institute (Hanusardottir, pers. comm.).

The United Nations Food and Agriculture Organisation and the World Health Organisation have also established 'Acceptable Daily Intakes' (ADIs) for various compounds (FAO/WHO, 1990). These are defined as the daily intake of a chemical which, during a lifetime, appears to be without appreciable risk and these are compiled in Table 5. The concentrations of PCBs and other organochlorines determined from the pilot whale samples may also be compared with permissible levels in food. Table 4 gives various national legal limits according to EPA (1987). For example, in the USA, 2 ppm PCBs in fish and shellfish is the official limit. This limit is statutorily enforceable (Boyer et al., 1991). In Canada, there is also a 2

Table 4
Compilation of regulatory limits for priority organochlorine pesticides in fish and fishery products ($\mu g\ g^{-1}$ fresh wt.) (Source: EPA 1987)

Location	PCBs	Aldrin/Dieldrin	DDT	DDE	H/H.epox	HCH
Canada	2.0	0.1	5.0	5.0	0.1	0.1
Denmark			2.0–5.0			
Germany		0.5–1.0			0.01	2.0
Iceland						0.5
Netherlands	5.0					
Sweden	2.0–5.0	0.1				0.2
Switzerland	1.0					
Thailand		0.3	5.0		0.3	0.5
United States	2.0	0.3	5.0	5.0	0.3	

ppm PCB guideline for fish and shellfish intended for human consumption. Currently, no limit exists in Danish territories for PCBs and there are no international standards laid down.

Table 5
Acceptable daily intake (ADI) (maximum daily intake of a chemical which during a life time appears to be without appreciable risk) (FAO/WHO, 1976) for organochlorine compounds (cited by Gunderson, 1988)

Organochlorine	FAO/WHO ADIs[a] (mg/kg body wt./day)
Aldrin and dieldrin	0.0001+
Aldrin	+
Dieldrin	+
BHC (alpha, beta, delta)	–
BHC (gamma) = Lindane	0.01[b]
Total Chlordane	0.0005+
Total DDT	0.02+
Total Heptachlor	0.0005+
Heptachlor	+
Heptachlor epoxide	+
PCBs (Aroclor 1221, 1242 and 1254)	–
Toxaphene	–

[a]Values obtained from: Vettorazzi, 1985; FAO, 1985; and FAO/WHO (1986a,b) — cited by Gunderson, 1988.
NB — ADIs are not established for all chemicals (marked '–' above). '+' refers to compounds for which the ADI includes other (related) compounds (i.e. some ADIs include several chemicals which may be related because of their formation during manufacturing or environmental degradation (e.g. the heptachlor group)).
[b]There has recently been cause for the ADI for lindane to be revised downwards to 0.008 mg/kg body wt./day (FAO/WHO, 1990).

Using the method of calculation described above, the equivalent figures for intake of DDE, a metabolite of DDT, based on a mean value in the blubber of 13.09 $\mu g\ g^{-1}$ are between 0.69 and 4.75 $\mu g\ kg^{-1}\ day^{-1}$ with a mean of 2.21 $\mu g\ kg^{-1}\ day^{-1}$ for Faroe Islanders. These values are lower than the WHO/FAO ADI for DDE of 20 $\mu g\ kg^{-1}\ day^{-1}$. For dieldrin, with a mean blubber value of 1.94 $\mu g\ g^{-1}$, the intake values fall between 0.1 and 0.7 $\mu g\ kg^{-1}\ day^{-1}$ with a mean value of 0.32 $\mu g\ kg^{-1}\ day^{-1}$. These equal or exceed the FAO/WHO ADI value of 0.1 $\mu g\ kg^{-1}\ day^{-1}$.

As noted by EPA (1987) the use of national consumption averages are not predictive of all subgroups and regions on a scale fine enough to address local situations of potential concern. In setting the tolerance levels for PCBs in foodstuffs, the FDA recognised (Boyer, 1991) that some sectors of the population could be exposed to PCB levels higher than the TDI as a result of habits that differed substantially from those represented in dietary studies undertaken.

It is likely that there will be a wide variation in consumption of pilot whale tissues by individual Faroe Islanders. Any broad estimates of consumption will inevitably be subject to uncertainties. For example, Faroese adults are recommended to restrict their consumption of pilot whale meat to 150–200 g/week corresponding to two large meals per month (Hanusardottir, pers. comm.). Grandjean et al. (1992), however, present data indicating that of 990 mothers surveyed in a study of fetal exposures to mercury, 266 (26.8%) ate in excess of

this, with 180 (18.1%) eating four or more whale dinners per month. The mean level of consumption of blubber reported from a separate survey (Vestergaard and Zachariassen, 1987) was 196 g/month, close to the recommended maximum consumption of 200 g/month. Similar extremes of blubber consumption may be expected in the population.

In addition, as noted by Gibson-Lonsdale (1990), the unpredictable geographic appearance of the whales leads to regional disparity in meat and blubber distribution. There is no formal system for distributing any excess. The internal, unofficial exchange between friends and families has not been subjected to empirical study (Sanderson, pers. comm.). Moreover, family shares of meat and blubber tend to be cut from single carcasses (Sanderson, 1991). Hence, although mean contamination levels provide an assessment of general population exposure, individual exposures are also likely to be somewhat variable due to the range of contaminant concentrations found in the pilot whale blubber. Variation in organochlorine intake could also occur as a result of seasonal changes in the lipid content of pilot whale tissues affecting absolute concentrations of lipophilic contaminants. The data provided by Lockyer (in press) indicate that blubber is the most stable lipid reservoir. Blubber lipid content varies between approximately 70–85%. Major variations occur in muscle and visceral fat lipid content which range between 1 and 45% and 50 and 70%, respectively. Hence, seasonal factors are not expected to result in major differences in blubber organochlorine burden although this deserves further evaluation.

Human uptake of contaminants and bioaccumulated levels are highly dependent upon dietary exposure. This is well illustrated by the variations found in broad regional exposures to toxic chemicals in the United States as evidenced by levels of organochlorine compounds in human adipose tissue (Phillips and Birchard, 1991). Given that estimates of PCB intake for Faroe Islanders show an increase over guideline values when calculated either on the basis of a national average consumption figure, or on the basis of a local survey, it is probable that some sectors of the population will exceed these guideline values very substantially.

A further important consideration is that several contaminants are detectable in the blubber. Hence, the potential interactive effects of the various organochlorines require evaluation since ADIs are generally determined on the basis that contaminants are present singly. Nonetheless, simply on a quantitative basis, the PCBs represent the greatest potential risk to consumers. PCBs are classified as probable human carcinogens and a range of potential acute and chronic toxic effects have been established in laboratory animal models.

4.5. Effects of polychlorinated biphenyls in humans

(a) Dose response relationships. PCBs have been shown to exert a range of effects on exposed mammalian species and the literature has been regularly reviewed. Dose-effect relationships have been reliably established in a number of animal models using a variety of toxicological endpoints (Kimbrough and Jensen, 1989). By contrast, establishing dose-effect relationships in humans has been limited by the fact that in most cases doses have necessarily been retrospectively estimated, in some cases from incomplete data sets (Boyer et al., 1991). Moreover PCBs have not been the only organochlorine chemicals present. Dose estimates are also complicated by the variety of congeners present in any one technical mixture. Phillips et al. (1989) have established a half-life of 4.8 years for Aroclor 1254 in humans but point out that any estimates of persistence are confounded by the differences in behaviour of individual congeners present in the technical mix. Significant variation in the ability of laboratories to successfully analyse PCBs and other organochlorines is a further complicating factor (e.g. Miskiewicz and Gibbs, 1992). Luotamo et al. (1991) note that actual profiles of commercial PCB mixes are rarely found in biological samples. The character of PCB contamination in humans reflects the nature and history of exposure (Burse et al., 1991). Recently, much attention has been focussed on the highly toxic co-planar PCB congeners (Tanabe et al., 1987; Hong et al. 1992) which appear particularly recalcitrant in the environment.

Nonetheless, it is now well established that PCBs can be readily detected in adipose tissues from humans sampled from the general community in industrialised countries. Greve and van

Zoonen (1990) concluded that the persistence and ubiquity of PCBs meant that, unlike other organochlorines, no fall in levels could be observed with time in the Netherlands. Age-related accumulation has been described for members of the general population in Germany (Brunn et al., 1990) and Phillips and Birchard (1991) report PCBs to be present in human adipose tissues from all regions of the United States.

(b) 'Yusho' and 'Yucheng'. The 'Yusho' incident in Japan and the 'Yucheng' poisoning in Taiwan both occurred as a result of the contamination of rice oil with thermally degraded PCBs (Safe, 1987). A variety of acute toxic effects including chloracne, respiratory problems, anorexia and fatigue were recorded. Some acute symptoms persisted for over 10 years after exposure and included developmental abnormalities in children born to affected mothers (Rogan et al., 1988; Tilson et al., 1990). Although these two incidents are widely regarded as attributable to PCBs, Tanabe et al. (1989) note that analysis of the oil and of human adipose tissue revealed the presence of polychlorinated dibenzofurans (PCDFs) and polychlorinated quaterphenyls (PCQs) as well as PCBs. Retrospective mean estimates of the intake of these chemicals in 141 'Yusho' patients are given by Kuratsune (1989) as 157 μg PCBs, 148 μg PCQs and 0.9 μg PCDFs/kg body weight per day. Estimated aggregate mean total intakes were 633 mg PCBs, 596 mg PCQs and 3.4 mg PCDFs, respectively.

(c) Occupational exposure. Equally, the growing body of literature on subacute effects from occupational exposure to PCBs contains few data concerning actual doses experienced. In most of the studies of occupational exposure, reviewed by Smith and Brown (1986), the dose has been inferred from a mean (time weighted average) personal air exposure although Maroni et al. (1981) consider that absorption through the skin constitutes the most important route of uptake under circumstances of occupational exposure. To date, studies of occupational morbidity and mortality have yielded equivocal results although subclinical biochemical alterations have been demonstrated (Smith and Brown, 1986).

(d) The Great Lakes studies. In the Laurentian Great Lakes particularly, contamination of fish with PCBs and other organochlorine chemicals is of considerable economic significance and considered to pose a potential hazard to humans (IJC, 1989; Fox, 1991).

These concerns stem from the widespread excedences of the US FDA Tolerance of 2 μg^{-1} for PCBs in the edible portions of fish and in fish meal during the late 1970s (Thomann et al., 1986). Consequently considerable research effort has been expended upon elucidation of effects in humans resulting from the consumption of contaminated sport fish. The Great Lakes population studies provide a useful comparator for the Faroe Islanders insofar as both populations have broadly comparable 'Western' life styles. In both cases consumption of aquatic resources represents a significant intake pathway for organochlorines. This contrasts with other communities, such as the Inuit, exposed to organochlorines and trace metals in their diet (Hansen, 1986; Dewailly et al., 1989) but where socioeconomic factors differ markedly from those in the Faroe Islands.

Fiore et al. (1989) demonstrated that in Wisconsin sport fish anglers, high levels of fish consumption were positively correlated with high serum PCB levels and suggested that these anglers could provide a suitable population for the long-term assessment of PCB and DDE associated morbidity and mortality. Sonzogni et al. (1991) further reported that the most predominant PCB congeners in the blood of the sampled population were the toxicologically significant mono and di-ortho analogues of the co-planar congeners 77 and 126. Anderson (1989) notes that a questionnaire survey (1973–1975) of sportfish anglers for acute and subacute illness (based on 'Yusho' symptoms) failed to establish a correlation between PCB serum levels and the prevalence of any of the seventeen symptoms investigated. The total annual intake of PCBs by consumers of 10.9 kg or more of lake sport fish per year was calculated as between 14.2 mg and 114.3 mg with a mean of 46.5 mg. Humphrey (1983) found that consumers of Lake Michigan fish could exceed the TDI by a factor of four and that average sport-fishermen consuming their catch could ingest some 200 mg of PCBs over a 4-year period.

Developmental effects have been studied in children born to Great Lakes sport-fish consumers. In these studies, a PCB dose in a sampled maternal population was estimated by assigning weighted values to Lake Michigan sport fish based upon average, contaminant levels for the species. The range of values obtained for 18 fish species was then normalised to the species containing the highest contaminant levels: lake trout, carp, salmon and catfish. Reported consumption of each type of fish was then used to calculate annual and cumulative consumption using the derived normalised unit of PCB-kg of fish (Jacobson et al., 1983). On average, 6.7 PCB-kg year^{-1} (Range 1.2–41.7 PCB-kg) for 16.1 years (Range 1.0–40 years) of contaminated fish were consumed by members of the study group. This was considered moderate to high consumption. The consumption of contaminated fish in these studies was at least 11.8 PCB-kg over 6 years.

Calculated consumption of contaminated fish was found to be a good predictor of PCB levels of maternal serum and maternal milk. Schwartz et al. (1983) reported that for every 0.45 PCB-kg of fish consumed at the highest rates of consumption (3–10 PCB-kg of fish/year or more) serum PCB levels increased in direct relation to consumption and duration of exposure. Maternal serum PCB levels increased by 0.15 ng ml^{-1} and milk PCB levels by 0.12 ng g^{-1}.

Fein et al. (1984) evaluated birth weight, length, gestational age and indices of neurological development in 313 newborn infants of which 242 were born to consumers of Lake Michigan sport fish. Seventy-one infants were born to mothers who did not consume Lake Michigan fish. Seventy-three potentially confounding variables including demographic factors, alcohol, caffeine and nicotine consumption were controlled for in the analysis of the data.

Overall, contaminated fish consumption was found to predict a smaller birth size on two measures: birth weight and head circumference. A shorter gestational period was also found as estimated by the Ballard examination. Infants born to contaminated fish consumers averaged 190 g less in birth weight, 0.6 cm less in head circumference and 4.9 days less in gestational age. This shortened gestation was also related to greater neuromuscular immaturity. All four outcomes were related to contaminated fish consumption in a dose-dependent fashion. These outcomes, however, could not be predicted from evaluation of fish consumption during pregnancy alone. Umbilical cord serum PCB levels >5 ng/l were associated with the observed infant effects. Co-exposure to other chemicals was not accounted for.

In a follow-up study of 123 infants from the same experimental cohort, tests conducted after 5–7 months showed that these infants performed poorly in visual recognition tests (Jacobson et al., 1985; Jacobson et al., 1990a) and that these observations were not necessarily related to the adverse neonatal outcomes observed at birth. A cord serum PCB level was a stronger predictor of poor test performance than overall contaminated fish consumption.

Approximately 75% of the children originally examined by Fein et al. (1984) were re-examined at age 5 by Jacobson et al. (1990a) who established that prenatal exposure assessed by cord serum PCB levels was associated with poorer performance in two specific tests involving short-term memory. Prenatal exposure was also associated with lower weight in a dose-dependent manner. There were, however, no indications of perceptual motor deficits or alteration of long-term memory. Exposure to polybrominated biphenyls (PBBs), lead and seven organochlorine pesticides were controlled for in the statistical analyses.

Although intrauterine exposure to PCBs is considered particularly harmful by Jacobson et al. (1983; 1990a), PCB exposure may also result from the mobilisation of PCBs into human breast milk. Such mobilisation has been widely reported (Skaare et al., 1988; Galetin-Smith, 1990; Greve and van Zoonen, 1990; Skaare and Polder, 1990; Franchi and Focardi, 1991). In the follow-up study of the original cohort studied by Fein et al. (1984), Jacobson et al. (1990b) found that the activity level was negatively related to child blood serum PCB level and also to the maternal milk level. A statistical evaluation of the data indicated that the effect of maternal milk was strongest in children of mothers with higher than average PCB levels and who had breast fed for at least 12 months.

Exposure from breast feeding was not related to cognitive performance although Jacobson et al. (1989) found that body burdens of PCBs in children studied at age 4 were primarily determined by nursing. Recently, Koppe et al. (1989) have hypothesised that late haemorraghic disease in newborn children may be related to organochlorine contamination of breastmilk fed to them.

The actual consumption of PCBs was not reported in these studies. At the time that the studies were initiated, however, Thomann et al. (1986) state that concentrations of PCBs in whole Lake Michigan fish were between 1 and 20 ppm wet weight. Contemporary data for median levels of PCBs in prepared Lake Michigan fish were reported by Humphrey (1988) as 3.01 $\mu g\ g^{-1}$ for lake trout, 1.4 $\mu g\ g^{-1}$ for chinook salmon, 0.87 $\mu g\ g^{-1}$ for coho salmon and 0.16 $\mu g\ g^{-1}$ for a mixed group of four other species.

In reviewing the Great Lakes studies, Anderson (1989) notes that the findings need further investigation and confirmation in other exposed groups. Despite the uncertainties in estimating actual doses in the Great Lakes studies from the available data, assuming that the dose was 3.0 ppm in each PCB-kg of fish consumed, the highest median value reported for prepared fish, the intake can be calculated at 3.6–125.1 mg annually with a mean of 20.1 mg. Mean cumulative consumption can be estimated at 323.6 mg over the mean duration of exposure of 16.1 years.

4.6. Faroe Islanders

On the basis of mean consumption figures for whale blubber coupled with mean contaminant levels, excedence of the internationally recognised US FDA TDI is evident. On this basis, a mean intake of PCBs of 48 mg/year can be calculated for Faroe Islanders which falls within the range, but above the mean value, estimated for intake of PCBs by consumers studied in the Great Lakes. Ranges in the amounts of blubber consumed and in the levels of contaminants found imply that the range of exposure in Faroe Islanders may be quite broad, with substantial excedences of guideline values at upper limits of consumption. Hence, the Faroe Islanders are exposed to dietary levels of PCBs comparable to those of Great Lakes sport-fish consumers. The need for a comparative study of the Faroes population to evaluate precise intake levels and effects is indicated and such a study could, in addition, usefully extend the Great Lakes work.

5. Conclusion

The work of Andersen et al. (1987) clearly demonstrated that on the basis of national average consumption figures, Faroese Islanders were exceeding the WHO/FAO Provisional Tolerable Weekly Intakes for mercury and cadmium. The data presented for mercury in this study suggest that blubber is not likely to be a significant source of mercury in comparison to other pilot whale tissues which the Faroese may consume. Blubber content of organochlorines, by contrast, is likely to represent a major source of these chemicals in consumers.

Since these figures are calculated on a national average basis and do not therefore differentiate between varying consumption levels by individual population sectors, nor variation in blubber contamination levels, it is likely that some communities will be more highly exposed than others. This implies that excedence of the guideline values may well be considerably higher in some areas than those calculated. The statement by Sanderson (1991) that no excedence of international guidelines results from a twice monthly large meal of blubber and meat is, therefore, highly questionable. On the assumption that this comprises 196 g of blubber, close to the maximum consumption recommended by the Faroes Food and Environmental Institute's maximum recommended consumption of 200 g/month, at average contaminant burdens this would breach the internationally accepted US FDA standard for PCBS and the WHO/FAO ADI for dieldrin.

Studies of the highly contaminated Laurentian Great Lakes have shown that a number of human reproductive and developmental abnormalities are associated with the moderate to high consumption of contaminated fish. The highest fish consumption of approximately 3–10 PCB-kg $year^{-1}$ in these studies equates with the Faroese national average consumption determined by Andersen et

al. (1987) of blubber at between 1.3 and 9.25 kg year^{-1} with a mean consumption of 4.3 kg. Recommended maximum consumption figures of 200 g of blubber per month equates to a consumption of 2.4 kg of blubber per year and average consumption, determined by survey, is 2.3 kg. Levels of contaminants in pilot whale blubber appear to be higher than those reported in the edible portions of Great Lakes fish.

Thus, exposure of the Faroe Islands population to PCBs and other organochlorines present in pilot whale blubber appears to be comparable to exposure in the Great Lakes populations consuming contaminated sport fish. The continuing consumption of organochlorine contaminated pilot whale blubber by the Faroe Islands population represents an opportunity to extend and confirm previous studies of the impact on humans of consumption of contaminated aquatic resources.

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7. References

Andersen, A., K. Julshamn, O. Ringdal and J. Morkore, 1987. Trace elements intake in the Faroe Islands II. Intake of Mercury and other elements by consumption of pilot whales (*Globicephalus meleanus*). Sci. Total Environ., 65: 63–68.

Anderson, H.A., 1989. General population exposure to environmental concentrations of halogenated biphenyls. In: R.D. Kimbrough and A.A. Jensen (Eds), Halogenated Biphenyls, Terphenyls, Naphthalenes, Dibenzodioxins and Related Products. Topics in Environmental Health, Volume 4. Elsevier Science Publishers, Amsterdam, pp. 325–380.

Anon, 1985. Statement from the Fisheries and Veterinary Department of the Torshavn Hygienic Institute, Faroe Islands, 2 pp.

Arima, S. and K. Nagakura, 1979. Mercury and selenium content of Odontoceti. Bull. Jpn. Soc. Sci. Fish., 45(5): 623–626.

Borrell, A., 1993. PCB and DDTs in blubber of cetaceans from the Northeastern North Atlantic. Mar. Poll. Bull., 26(3): 146–151.

Borrell, A. and A. Aguilar, (in press) DDT and PCB pollution in blubber and muscle of long-finned pilot whales from the Faroe Islands. Rep. Int. Whal. Commn., Special Issue, No. 14.

Boyer, I.J., C.J. Kokoski and P.M. Bolger, 1991. Role of FDA in establishing tolerable levels for dioxin and PCBs in aquatic organisms. J. Toxicol. Environ. Health, 33: 93–101.

Brunn, H., S. Georgi and J. Prucha, 1990. Polychlorierte Biphenyle (PCB) im menschlichen Fettgewebe. Z. Lebensm.-Unters. Forsch., 190: 108–111.

Burse, V.W., D.F. Groce, M.P. Korver, S.P. Caudill, P.C. McClure, C.R. Lapeza, S.L. Head, R.J. Schilling, J.A. Farrar, S.R. Ostrowski, B.K. Mortensen, J.P. Maher, H.L. Russell and S. Sievila, 1991. Evidence of an unusual pattern of polychlorinated biphenyls in the serum of some residents and canines in Paoli, Pennsylvania. J. Assoc. Off. Anal. Chem., 74(4): 577–586.

Clark, R.B., 1989. Marine Pollution, Second Edition. Clarendon Press, Oxford.

Desportes, G. and R. Mouritsen, 1988. Diet of the Pilot Whale, *Globicephala melas* around the Faroe Islands. ICES Report No. C.M. 1988/No. 12. (mimeo).

Dewailly, E., A. Nantel, J.-P. Weber and F. Meyer, 1989. High levels of PCBs in breast milk of Inuit women from Arctic Quebec. Bull. Environ. Contam. Toxicol., 43: 641–646.

EPA, 1987. Assessing Human Health Risks from Chemically Contaminated Fish and Shellfish: A Guidance Manual. US Environmental Protection Agency, Office of Water Regulations and Standards (WH-552), Washington DC Report No: EPA-503-89-002, 136 pp.

FAO/WHO, 1990. Pesticide Residues in Food — 1989. Evaluations 1989. Part II — Toxicology. Report of the Joint Meeting of the FAO Panel of Experts on Pesticide Residues in Food and the Environment and the WHO Expert Group on Pesticide Residues 1989. Food and Agriculture Organization of the United Nations, Rome, Italy. p. 261.

Fein, G.G., J.L. Jacobson, S.W. Jacobson, P.M. Schwartz and J.K. Dowler, 1984. Prenatal exposure to polychlorinated biphenyls: Effects on birth size and gestational age. J. Pediatr., 105: 315–320.

Fiore, B.J., H.A. Anderson, L.P. Hanrahan, L.J. Olson and W.C. Sonzogni, 1989. Sport fish consumption and body burden levels of chlorinated hydrocarbons: A study of Wisconsin anglers. Arch. Environ. Health, 44: 82–88.

Fox, G.A., 1991. Practical causal inference for eco-epidemiologists. J. Toxicol. Environ. Health, 33: 359–373.

Franchi, E. and S. Focardi, 1991. Polychlorinated biphenyl congeners, hexachlorobenzene and DDTs in human milk in central Italy. Sci. Tot. Environ., 102: 223–228.

Galetin-Smith Pavkov, S. and N. Roncevic, 1990. DDT and PCBs in human milk: Implication for breast feeding infants. Bull. Environ. Contam. Toxicol., 45: 811–818.

Gibson-Lonsdale, J.J., 1990. Pilot whaling in the Faroe Islands — its history and present significance. Mamm. Rev., 20(1): 44–52.

Grandjean, P., P. Weihe, P.J. Jorgensen, T. Clarkson, E. Cernichiari and T. Videro, 1992. Impact of maternal seafood diet on fetal exposure to mercury, selenium and lead. Arch. Environ. Health, 47(3): 185–195.

Greve, P.A. and P. van Zoonen, 1990. Organochlorine pesticides and PCBs in tissues from Dutch Citizens (1968–1986). Int. J. Environ. Anal. Chem., 38: 265–277.

Hansen, J.C. 1990. Human exposure to metals through consumption of marine foods: a case study of exceptionally high intake among Greenlanders. In: R.W. Furness and P.S. Rainbow (Eds), Heavy Metals in the Marine Environment. CRC Press, Florida.

Hatch, W.R. and W.L. Ott, 1968. Determination of submicrogram quantities of mercury by atomic absorbtion spectrophotometry. Anal. Chem., 40: 2085–2087.

Hong, C.S., B. Bush and J. Xiao, 1992. Co-planar PCBs in fish and mussels from marine and estuarine waters of New York State. Ecotoxicol. Environ. Safety, 23: 118–131.

Humphrey, H.E.B., 1983. Population studies of PCBs in Michigan residents. In: F.M. D'Itri and M. Kamrin (Eds), PCBs: Human and Environmental Hazards. Butterworth, Boston, pp. 299–310.

Humphrey, H.E.B., 1988. Chemical Contaminants in the Great Lakes: The Human Health Aspect. In: M.S. Evans (Ed.), Toxic Contaminants and ecosystem Health: A Great Lakes Focus. John Wiley and Sons, NY, pp. 153–165.

IJC, 1989. Fifth Biennial Report on Great Lakes Water Quality. International Joint Commission on the Great Lakes, Windsor, Ontario.

Jacobson, S.W., J.L. Jacobson, P.M. Schwartz and G.G. Fein, 1983. Intrauterine exposure of human newborns to PCBs: Measures of exposure. In: F.M. D'Itri and M. Kamrin (Eds), PCBs: Human and Environmental Hazards. Butterworth, Boston, pp. 311–343.

Jacobson, S.W., G.G. Fein, J.L. Jacobson, P.M. Schwartz and J.K. Dowler, 1985. The effect of intrauterine PCB exposure on visual recognition memory. Child Dev., 56: 853–860.

Jacobson, J.L., H.E.B. Humphrey, S.W. Jacobson, S.L. Schantz, M.D. Mullin and R. Welch, 1989. Determinants of polychlorinated biphenyls (PCBs), polybrominated biphenyls (PBBs) and dichlorodiphenyl trichloroethane (DDT) levels in the sera of young children. Am. J. Publ. Health, 79(10): 1401–1404.

Jacobson, J.L., S.W. Jacobson and H.E.B. Humphrey, 1990a. Effects of in utero exposure to polychlorinated biphenyls and related contaminants on cognitive function in young children. J. Pediatr., 116: 38–45.

Jacobson, J.L., S.W. Jacobson and H.E.B. Humphrey, 1990b. Effects of exposure to PCBs and related compounds on growth and activity in children. Neurotoxicol. Teratol., 12: 319–326.

Johnston, P.A. and R.L. Stringer, 1988. Greenpeace QMC: Technical note — contaminants in marine mammals from Liverpool Bay. October 27, 3 pp.

Johnston, P.A., R.L. Stringer and M.C. French, 1991. Pollution of UK estuaries: historical and current problems. Sci. Total Environ., 106: 55–70.

Julshamn, K., A. Andersen, O. Ringdal and J. Morkore, 1987. Trace elements intake in the Faroe Islands. I. Element levels in edible parts of pilot whales (*Globicephalus melaena*). Sci. Total Environ., 65: 53–62.

Julshamn, K., K.-J. Andersen, E. Svendsen, O. Ringdal and M. Egholm, 1989. Trace elements intake in the Faroe Islands. III. Element concentrations in human organs in populations from Bergen (Norway) and the Faroe Islands. Sci. Total Environ., 84: 25–33.

Kimbrough, R.D. and A.A. Jensen, 1989. (Eds), Halogenated biphenyls, terphenyls, naphthalenes, dibenzodioxins and related products. Topics in Environmental Health, Vol. 4. Elsevier Science Publishers, Amsterdam, 518 pp.

Koppe, J.G., E. Pluim and K. Olie, 1989. Breastmilk, PCBs, dioxins and vitamin K deficiency: discussion paper. J. R. Soc. Med., 82: 416–419.

Kuratsune, M., 1989. Yusho, with reference to Yu-Cheng. In: R.D. Kimbrough and A.A. Jensen (Eds), Halogenated Biphenyls, Terphenyls, Naphthalenes, Dibenzodioxins and Related Products. Topics in Environmental Health, Vol. 4. Elsevier Science Publishers, Amsterdam, pp. 381–400.

Larsson, P., 1985. Contaminated sediments of lakes and oceans act as sources of chlorinated hydrocarbons for release to water and atmosphere. Nature, 317: 347–349.

Lockyer, C., (in press) Seasonal changes in body fat condition of Northeast Atlantic pilot whales and the biological significance. Rep. Int. Whal. Commn., Special Issue, No. 14.

Luotamo, M., J. Jarvisalo and A. Aitio, 1991. Assessment of exposure to polychlorinated biphenyls: Analysis of selected isomers in blood and adipose tissue. Environ. Res., 54: 121–134.

MAFF, 1990. Monitoring and surveillance of non-radioactive contaminants in the aquatic environment, 1984–1987. Aquat. Environ. Monit. Rep., MAFF Direct. Fish. Res. Lowestoft, 22: 60.

Maroni, M., A. Colombi, S. Cantoni, E. Ferioli and V. Foa, 1981. Occupational exposure to polychlorinated biphenyls in electrical workers. I: Environmental and blood polychlorinated biphenyls concentrations. Br. J. Ind. Med., 38: 49–54.

Martin, A.R., P. Reynolds and M.G. Richardson, 1987. Aspects of the biology of pilot whales (*Globicephala melaena*) in recent mass strandings on the British coast. J. Zool., Lond., 21: 11–23.

Miskiewicz, A.G. and P.J. Gibbs, 1992. Variability in organochlorine analysis in fish: An interlaboratory study and its implications for environmental monitoring and regulatory standards. Arch. Environ. Contam. Toxicol., 23: 45–53.

Muir, D.C.G., R. Wageman, N.P. Grift, R.J. Norstrom, M. Simon and J. Lieu, 1988. Organochlorine chemical and heavy metal contaminants in white beaked dolphins (*Lagenorhynchus albirostris*) and pilot whales (*Globicepbala melaena*) from the coast of Newfoundland, Canada. Arch. Environ. Contam. Toxicol., 17: 613–629.

Phillips, D.L., A.B. Smith, V.W. Burse, G.K. Steele, L.L. Needham and W.H. Hannon, 1989. Half life of polychlorinated biphenyls in occupationally exposed workers. Arch. Environ. Health, 44 (6): 351–354.

Phillips, L.J. and Birchard, 1991. Regional variations in human

toxics exposure in the USA: An analysis based on the National Human Adipose Tissue Survey. Arch. Environ. Contam. Toxicol., 21: 159–168.

Rogan, W.J., B.C. Gladen, K.-L. Hung, L.-Y. Shih, J.S. Taylow, Y.-C. Wu, D. Yang, N.B. Ragan and C.-C. Hsu, 1988. Congenital poisoning by polychlorinated biphenyls and their contaminants in Taiwan. Science, 241: 334–336.

Safe, S., 1987. PCBs and Human Health. Environmental Toxin Series, Vol. 1. Springer-Verlag, Berlin, pp. 133–145.

Sanderson, K., 1991. Whales and whaling in the Faroe Islands. The Department of Fisheries, Faroe Islands, 29 pp.

Schwartz, P.M., S.W. Jacobson, G. Fein, J.L. Jacobson and H.A. Price, 1983. Lake Michigan fish consumption as a source of polychlorinated biphenyls in human cord serum, maternal serum, and milk. Am. J. Public Health, 73(3): 293–296.

Skaare, J.U., J.M. Tuveng and H.A. Sande, 1988. Organochlorine pesticides and polychlorinated biphenyls in maternal adipose tissue, blood, milk, and cord blood from mothers and their infants living in Norway. Arch. Environ. Contam. Toxicol., 17: 55–63.

Skaare, J.U. and A. Polder, 1990. Polychlorinated biphenyls and organochlorine pesticides in milk of Norwegian women during lactation. Arch. Environ. Contam. Toxicol., 19: 640–645.

Smith, A.B. and D.P. Brown, 1986. Polychlorinated biphenyls in the workplace In: J.S. Waid (Ed.) PCBs in the Environment, Vol. III. CRC Press Inc., Boca Raton, Florida, pp. 63–82.

Sonzogni, W., L. Maack, T. Gibson, D. Degenhardt, H. Anderson and B. Fiore, 1991. Polychlorinated biphenyl congeners in blood of Wisconsin sport fish consumers. Arch. Environ. Contam. Toxicol., 20: 56–60.

Swain, R.W., 1988. Human health consequences of consumption of fish contaminated with organochlorine compounds. Aquat. Toxicol., 11: 357–377.

Tanabe, S., N. Kannan, An. Subramanian, S. Watanabe and R. Tatsukawa, 1987. Highly toxic co-planar PCBs: Occurrence, source, persistency and toxic implications to wildlife and humans. Environ. Pollut., 47: 147–163.

Tanabe, S., N. Kannan, T. Wakimoto, R. Tatuskawa, T. Okamoto and Y. Masuda, 1989. Isomer-specific determination and toxic evaluation of potentially hazardous coplanar PCBs, dibenzofurans and dioxins in the tissues of "Yusho" PCB poisoning victims and in the causal oil. Toxicol. Environ. Chem., 24: 215–231.

Thomann, R.V., J.P. Connolly and N.A. Thomas, 1986. The Great Lakes ecosystem — modeling the fate of PCBs. In: J.S. Waid (Ed.), PCBs and the Environment, Vol. III. CRC Press Inc., Boca Raton, Florida, pp. 153–179.

Tilson, H.A., J.L. Jacobson and W.J. Rogan, 1990. Polychlorinated biphenyls and the developing nervous system: cross-species comparisons. Neurotoxicol. Teratol., 12: 239–248.

Vestergaard, T. and P. Zachariassen, 1987. Fodslukanning 1981–1982. Frodskaparrit, 33: 5–18.

Wagemann, R. and D.C.G. Muir, 1984. Concentration of heavy metals and organochlorines in marine mammals of Northern waters: Overview and evaluation. Canadian Technical Report of Fisheries and Aquatic Sciences No. 1279. 97 pp.

WHO, 1976. Environmental Health Criteria: Mercury. World Health Organisation, Geneva.

Wiborg K.F., J. Gjosaeter, I.M. Beck and P. Fossum, 1982. The Squid *Todarodes sagittatus* (Lamarck) Distribution and Biology in Northern waters. Aug 1981–Aug 1982 ICES Report No. C M 1982/K:30 (mimeo).