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Short communication

Trace elements, PCBs and organochlorine pesticides in New Zealand common dolphins (*Delphinus* sp.)

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Abstract

Trace elements, polychlorinated biphenyls (PCBs) and organochlorine (OC) pesticide levels were determined in tissues collected from stranded and bycaught common dolphins (*Delphinus* sp.) from New Zealand waters between 1999 and 2005. The concentrations of mercury (Hg), selenium (Se), chromium (Cr), zinc (Zn), nickel (Ni), cadmium (Cd), cobalt (Co), manganese (Mn), iron (Fe), copper (Cu), tin (Sn), lead (Pb), arsenic (As) and silver (Ag) were determined in blubber, liver and kidney tissue. PCBs (45 congeners) and a range of OC pesticides including dieldrin, hexachlorocyclohexane (HCH) and dichlorodiphenyltrichloroethane (DDT) and its metabolites DDE and DDD were determined in blubber samples. Cr and Ni were not detected in any of the samples and concentrations of Co, Sn and Pb were generally low. Concentrations of Hg ranged from 0.17 to 110 mg/kg wet weight. Organochlorine pesticides dieldrin, HCB, *o,p′*-DDT and *p,p′*-DDE were present at the highest concentrations. Sum DDT concentrations in the blubber ranged from 17 to 337 and 654 to 4430 µg/kg wet weight in females and males, respectively. Similarly, Σ45CB concentrations ranged from 49 to 386 and 268 to 1634 µg/kg wet weight in females and males, respectively. The mean transmission of ΣDDTs and ICES7CBs between a genetically determined mother–offspring pair was calculated at 46% and 42%, respectively. Concentrations of organochlorine pesticides determined in the present study are within similar range to those reported for Hector's dolphins (*Cephalorhyncus hectori*) from inshore New Zealand waters.

Keywords: Common dolphin; *Delphinus* sp.; Trace elements; PCBs; Organochlorine pesticides; DDT; Mother–calf transmission; New Zealand

1. Introduction

Toxicological studies have investigated pollutant levels in several marine fauna including fish (Hoekstra et al., 2003; Alquezar et al., 2006), birds (Ryan et al.,

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Table 1
Specimen details for common dolphins stranded and bycaught in New Zealand waters between 1999 and 2005

Reference	Date	Sex	Length (cm)	Weight (kg)	Age (yr)	Body condition	Source*	GPS location	Region
WS99-14_30447	19/07/1999	M	215.0	102.0	–	Moderate	S	41°17′S, 174°46′E	Wellington Harbour, Wellington
WS00-01_30890	17/12/1999	M	196.0	98.0	–	Mild	S	35°10′S, 174°20′E	Deep Water Cove, Northland
WB02-01_32789	14/10/2001	M	227.5	134.0	11.0	Mild	B	40°07′S, 174°01′E	SW coast, South Island
WS02-14_33100	14/03/2002	M	172.0	58.0	5.0	Moderate	S	40°51′S, 175°01′E	Waikanae, Wellington
WB03-04_34086	17/10/2002	M	206.0	102.0	8.0	Mild	B	39°53′S, 173°40′E	SW coast, South Island
WB03-17_34705	30/04/2003	M	178.5	76.0	3.5	Mild	B	40°21′S, 170°00′E	SW coast South Island
WB03-18_34712	30/04/2003	M	199.5	88.0	8.0	Mild	B	40°21′S, 170°00′E	SW coast South Island
WB04-04_35613	17/12/2003	M	226.0	119.0	10.5	Moderate	B	37°10′S, 174°05′E	W Coast, North Island
WS04-19_36305	05/08/2004	M	174.0	64.0	8.0	Moderate	S	35°27′S, 174°43′E	Opahi Bay, Hauraki Gulf, Auckland
WS04-28_36737	14/12/2004	F	195.0	76.0	5.5	Fresh	S	36°46′S, 174°40′E	Lucas Creek, Hauraki Gulf, Auckland
WS04-29_36738	14/12/2004	F	199.0	73.0	10.5	Fresh	S	36°46′S, 174°40′E	Lucas Creek, Hauraki Gulf, Auckland
WS04-30_36739	14/12/2004	M	118.0	18.2	1.0	Fresh	S	36°46′S, 174°40′E	Lucas Creek, Hauraki Gulf, Auckland
WS04-32_36745	17/12/2004	F	99.0	9.8	0.5	Fresh	S	36°46′S, 174°40′E	Lucas Creek, Hauraki Gulf, Auckland
WS04-33_36746	16/12/2004	F	195.0	64.0	7.0	Fresh	S	36°46′S, 174°40′E	Lucas Creek, Hauraki Gulf, Auckland
WS04-34_36747	18/12/2004	F	189.0	69.0	10.0	Fresh	S	36°46′S, 174°40′E	Lucas Creek, Hauraki Gulf, Auckland
WS04-35_36751	18/12/2004	F	200.0	66.3	8.0	Fresh	S	36°51′S, 174°49′E	Orakei Bay, Hauraki Gulf, Auckland
WS04-36_36752	18/12/2004	F	195.0	73.0	5.0	Fresh	S	36°51′S, 174°49′E	Orakei Bay, Hauraki Gulf, Auckland
WS05-06_36823	26/01/2005	M	220.0	80.0	–	Mild	S	35°25′S, 174°44′E	Warkworth, Hauraki Gulf, Auckland
WS05-26_37521	27/07/2005	M	160.0	47.0	6.0	Mild	S	36°50′S, 174°40′E	Waitemata Harbour, Auckland

* B = bycaught, S = stranded.

1988; Mallory et al., 2005; Rothschild and Duffy, 2005) and mammals (Hobbs et al., 2001; Haynes et al., 2005; Roots et al., 2005; Ylitalo et al., 2005). Organochlorine (OC) compounds such as polychlorinated biphenyls (PCBs) are both chemically and physically stable within the environment, and thus classified as Persistent Organic Pollutants (POPs) within the Stockholm Convention. The lipophilic nature of such chemicals facilitates their accumulation along food chains where, in the case of top predators, they may bio-accumulate to high concentrations. Many toxicological studies have focused on the pollutant burdens in cetaceans, particularly in small toothed (odontoceti) cetaceans (de Kock et al., 1994; Minh et al., 1999; Fossi et al., 2004; Karuppiah et al., 2005). The high metabolic rate and elevated trophic position of odontocetes within food webs increase their likelihood of accumulating persistent toxins, such as organochlorine pesticides. These factors, in combination with the longevity and the large proportion of lipids present within cetaceans, facilitate bioaccumulation, a phenomenon that in some populations has resulted in high levels of toxicity (Hayteas and Duffield, 2000).

Several studies have examined the biological effects of contaminants such as PCBs, OC pesticides and trace metals on marine mammal health and life history (Subramanian et al., 1987; Kuiken et al., 1994; Wells et al., 2005). Certain organochlorines (e.g. dieldrin, lindane) are known to be particularly toxic in the early developmental stages of life and have been identified as endocrine-disrupting chemicals. Such chemicals may interfere with the production and metabolism of hormones responsible for the maintenance of homeostasis and the regulation of reproduction processes (Reijnders and Aguilar, 2002). In marine mammals, persistent pollutants have also been associated with a variety of toxic effects including immune suppression and the development of infectious diseases (Kuiken et al., 1994; Jepson et al., 2005), reproductive impairment (Reijnders, 1986; Schwacke et al., 2002; Wells et al., 2005) and the generation of tumours (De Guise et al., 1994).

The occurrence and distribution of marine pollutants has been extensively studied in northern hemisphere cetaceans (Jarman et al., 1996; Law et al., 1996; McKenzie et al., 1997; Parsons et al., 1999; Siebert et al., 1999; Watanabe et al., 1999 Frodello et al., 2000; Hernandez et al., 2000; Hobbs et al., 2001; Berrow et al., 2002; Law et al., 2005), where waters are generally accepted to be more industrialized. However, less information is available on the contaminant loads of marine mammals found in southern hemisphere waters (Borrell and Aguilar, 1999; Kemper et al. 1994; Vetter et al. 2001), with a particular paucity for data relating to New Zealand.

A range of organochlorine pesticides have historically been used in New Zealand including dichlorodiphenyltrichloroethane (DDT), dieldrin, heptachlor, hexachlorobenzene (HCB), chlordane, hexachlorocyclohexane (HCH) and aldrin (Buckland et al., 1998). However,

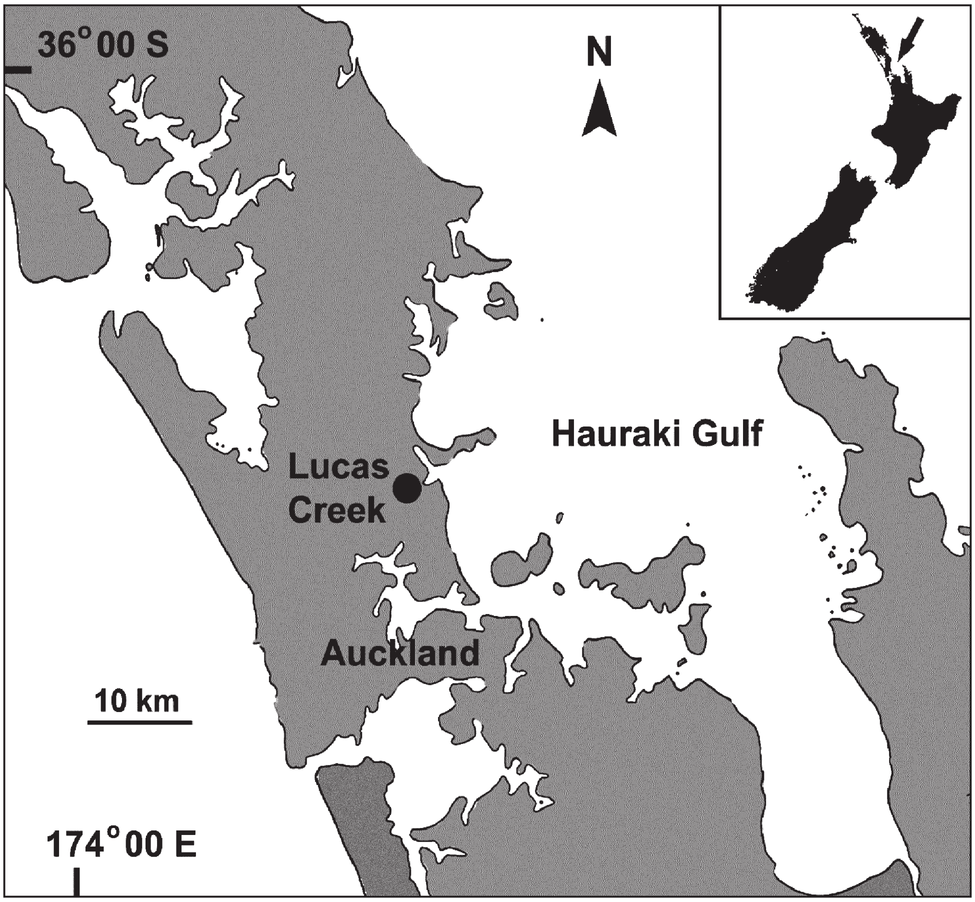

Fig. 1. Map showing Lucas Creek, Hauraki Gulf, New Zealand—the location of a mass-stranding of common dolphins used in the present study.

few published data relating to these compounds or their effects on biological systems within New Zealand can be found in the literature. Presented here are the first data relating to trace metals, PCBs and OC pesticide levels determined in New Zealand common dolphins (*Delphinus* sp.). We describe pollutant levels in males and females, and examine differences in pollutant loads between stranded and bycaught common dolphins. We investigate parent–offspring pollutant transfer between a genetically determined mother–calf pair, and discuss pollutant levels and their implications at the population level.

2. Materials and methods

2.1. Sample collection and storage

Tissue sampling was undertaken on fourteen stranded and five commercially bycaught common dolphins from New Zealand waters between 1999 and 2005 (Table 1). The majority of the stranded carcasses ($n = 8$) were recovered from a mass-stranding that occurred in the Hauraki Gulf (Fig. 1, approximate latitude 36°10′S to 36°60′S) during December 2004. Bycaught samples originated from animals incidentally killed in the mid-water trawl fishery for jack mackerel (*Trachurus* sp.) off the west coast of Auckland (37°10′S, 174°05′E) and in the South Taranaki Bight (40°10′S, 174°05′E).

All carcasses were examined and subjectively divided into three categories (*fresh, mild* and *moderate*) based on the degree of post-mortem autolysis evident (Table 1). Animals described as *fresh* typically live stranded and either subsequently died immediately prior to discovery (as determined by the presence of eye moisture and absence of rigour mortis) or were euthanased onsite by the Department of Conservation. *Mild* was assigned to carcasses which exhibited rigor mortis but which showed no obvious external signs of decomposition. Carcasses which showed early signs of decomposition (e.g., odour, skin degradation and/or loss) were deemed to be *moderate.* No animals exhibiting advanced stages of decomposition (e.g., tissue autolysis) were included in the present study. Pathological examination and sampling was conducted according to standard protocols (Geraci and Lounsbury, 1993). Prior to sampling, external measurements (cm) and body weight (kg) were recorded (Table 1), and where possible, teeth and skin samples collected for age and sex determination respectively. Six to eight teeth were removed from the lower jaw of each specimen and preserved in 70% ethanol. Skin samples were removed from the dorsal fin of each animal and preserved in 95% ethanol.

Tissue samples were taken for PCB and OC analysis using standard protocols (Kuiken et al., 1994; Jepson et al., 2005). In summary, cross sectional samples of blubber adjacent to the dorsal fin were excised from each carcass using a stainless steel knife. Samples were placed in a hexane-washed glass container with an aluminium or Teflon-lined cap and stored at −20 °C. Sampling for trace elements was conducted using methods described in Zhou et al. (2001). Blubber, liver and kidney tissue was collected during post-mortem examinations, and wrapped in aluminium foil prior to storage at −20 °C. Trace elements could only be assessed in animals that mass-stranded in the Hauraki Gulf, since liver and kidney tissues were not available from carcasses sampled outside of this region.

2.2. Age determination

Age was estimated by the examination of decalcified thin sections of teeth, following methods adapted from Slooten (1991). Due to the curvature of the teeth, care was taken to select teeth from which a complete section through the centre of the pulp could be obtained. Tooth sections were independently read by two observers (GWJ and PJD) at 16–80× magnification and the number of dentinal growth layers (GLGs) assigned by consensus between the readers.

2.3. Sex and mother–offspring identification

The sex of most individuals was determined by direct anatomical examination during necropsy. However, in order to ascertain sex in circumstances where a full post-mortem examination was not possible, molecular techniques were used, following methods outlined by Rosel (2003). Total genomic DNA (gDNA) was extracted from skin samples following a standard proteinase-K/phenol-chloroform procedure (Sambrook et al., 1989). Sex was determined by a multiplex PCR reaction using primers detailed in Rosel (2003).

In order to assess pollutant transfer between parent and offspring, parentage analyses using mitochondrial DNA (mtDNA) and microsatellite loci were performed to identify mother–calf pairs in a nursery group of common dolphins that mass-stranded in the Hauraki Gulf in December 2004 (see Table 1). A 397 base pair (bp) portion of the 5′ end of the mtDNA control region was sequenced using methods and primers detailed in Rosel et al. (1994), and multi-locus genotypes from a panel of 14 microsatellite loci were obtained as detailed in Coughlan et al. (2006) and Mirimin et al. (2006). The transfer rate of ΣDDTs and ICES7CBs between a mother and her calf was calculated using methods detailed in Borrell and Aguilar (2005).

2.4. Chemical analysis

Trace elements were determined in samples of blubber, liver and kidney by inductively-coupled plasma mass spectrometry (ICP-MS) or inductively-coupled plasma optical emission spectrometry (ICP-OES) using a Perkin Elmer ELAN 9000 and Perkin Elmer OPTIMA 3300 RL respectively. ICP-MS was used for those elements typically present at relatively low levels (e.g., tin, cobalt), whereas ICP-OES was used for elements that occur at higher concentrations (e.g., copper, zinc). Approximately 1 g of each tissue sample was digested in concentrated nitric acid with a trace of hydrofluoric acid. Full analytical quality control protocols were followed and all metals (other than chromium and silver) were analysed within the laboratory's International Accreditation New Zealand (IANZ) (No. 175).

Organochlorine pesticides and CBs were determined in blubber by high resolution gas chromatography-high resolution mass spectrometry (HRGC-HRMS). Extraction and quantification of hexachlorocyclohexanes; *alpha*-HCH, *beta*-HCH, *gamma*-HCH (lindane), dieldrin, heptachlor, heptachlor epoxide, *alpha*-chlordane, *gamma*-chlordane and DDT (plus metabolites *p, p′*-DDE, *p, p′*-DDD (also known as *p, p′*-TDE), *o, p′*-DDT, *p, p′*-DDT and 45 chlorobiphenyl congeners (CB1, CB3, CB4, CB15, CB19, CB28. CB37, CB44, CB49, CB52, CB54, CB70, CB74, CB77, CB81, CB99, CB101, CB104, CB105, CB110, CB114, CB118, CB123, CB126, CB138, CB153, CB155, CB156, CB157, CB167, CB169, CB170, CB180, CB183, CB187, CB188, CB189, CB194, CB196, CB199, CB202, CB205, CB206, CB208 and CB209) was conducted as follows:

2.4.1. Blubber extraction

Samples were thawed and a portion of the blubber tissue (approx. 10 g) was removed and chopped into small cubes (approx. 1 cm). The sample was accurately weighed and placed into a blender with powdered sodium sulphate and blended until the mixture was free-flowing. Each sample was subsequently packed into a

Table 2
Trace elements determined in the liver, kidney and blubber of stranded common dolphins from the Hauraki Gulf, New Zealand in December 2004 (mg/kg wet weight)

Reference	Sex	Tissue	Cr	Mn	Fe	Co	Ni	Cu	Zn	As	Se	Ag	Cd	Sn	Hg	Pb	Hg:Se
WS04-28_36737	F	Blubber	<0.1	0.085	9.4	<0.02	<0.1	0.43	25	0.81	6.9	<0.02	0.041	0.049	0.88	<0.01	0.05
WS04-29_36738	F	Blubber	<0.1	0.072	26	<0.02	<0.1	0.28	6.8	1.7	3.3	<0.02	0.038	0.043	0.86	0.012	0.10
WS04-30_36739	M	Blubber	<0.1	0.07	9.2	<0.02	<0.1	0.47	4.1	1.4	1.9	<0.02	<0.002	0.063	0.034	<0.01	0.01
WS04-32_36745	F	Blubber	<0.1	0.11	13	<0.02	<0.1	0.59	57	0.23	5.6	<0.02	<0.002	0.052	0.17	0.013	0.01
WS04-33_36746	F	Blubber	<0.1	0.07	7.5	<0.02	<0.1	0.53	100	0.89	20	<0.02	0.039	0.044	1.2	0.031	0.02
WS04-34_36747	F	Blubber	<0.1	<0.07	16	<0.02	<0.1	0.52	33	1.3	9.0	<0.02	0.036	0.035	1.0	0.011	0.04
WS04-34_36747	F	Liver	<0.1	3.7	180	<0.02	<0.1	8.6	44	0.32	19	0.73	7.3	0.09	50	<0.01	1.04
WS04-34_36747	F	Kidney	<0.1	0.78	150	0.021	<0.1	5.4	34	0.13	5.3	0.033	18	0.04	6.1	<0.01	0.45
WS04-35_36751	F	Blubber	<0.1	0.1	18	<0.02	<0.1	0.53	9.7	0.73	5.4	<0.02	0.19	0.052	1.3	<0.01	0.09
WS04-35_36751	F	Liver	<0.1	4.8	250	<0.02	<0.1	14	73	0.27	39	1.2	21	0.086	110	0.018	1.11
WS04-35_36751	F	Kidney	<0.1	0.69	140	0.031	<0.1	4.8	37	0.11	6.4	0.033	52	0.03	7.8	0.018	0.48
WS04-36_36752	F	Blubber	<0.1	0.084	18	<0.02	0.71	4.5	11	1.2	7.6	<0.02	0.12	0.062	1.7	0.014	0.09
WS04-36_36752	F	Liver	<0.1	3.7	190	<0.02	<0.1	7.9	60	0.17	18	0.48	3.4	0.083	53	0.74	1.16
WS04-36_36752	F	Kidney	<0.1	0.66	110	<0.02	<0.1	4.9	33	<0.07	5.3	0.033	17	0.05	8.1	0.15	0.60

* Hg:Se is the molar ratio of the concentrations of mercury and selenium.

Soxhlet extraction thimble. The blender container was cleaned between samples by though scrubbing with hot water and detergent, and rinsing with hot water, followed by acetone, toluene and dichloromethane. Before extraction, a range of isotopically labelled internal standards was added to each sample. Each sample was Soxhlet extracted with dichloromethane:hexane (1:1 v/v) for at least 16 h. The samples were evaporated to constant weight in a tared flask on a rotary evaporator. The lipid content was measured by difference. The samples were then subjected to clean-up as follows:

2.4.2. Organochlorine (OC) pesticides

The lipids were redissolved in hexane and partitioned with acetonitrile. The acetonitrile fraction was evaporated and residual lipids redissolved in hexane and chromatographed twice on 1 g columns of florisil, the pesticides being eluted with hexane:diethyl ether (82:18 v/v). The remaining lipids were removed by gel permeation chromatography (GPC) on a Phenomenex Envirosep ABC 300 × 7.8 mm GPC column using ethyl acetate:cyclohexane (1:1 v/v). The solvent was removed by nitrogen blowdown and the solution reconstituted in 100 μL of toluene containing the recovery standard ($^{13}C_{12}$–CB) and analysed by HRGC-HRMS using the accurate mass ions given in Buckland et al (1998).

2.4.3. Polychlorinated biphenyls (PCBs)

The lipids were dissolved in hexane and removed by chromatography on a reactive multi-column containing sodium silicate and sulfuric acid impregnated silica gel by elution with hexane. The hexane was removed by nitrogen blowdown and the residue reconstituted in 100 μL of nonane containing the recovery standards and analysed by HRGC-HRMS. All procedures for the analysis of PCBs followed the methods outlined in USEPA Method 1668A.

2.4.4. Instrumental

The HRGC-HRMS analyses were performed on an Agilent 6890 gas chromatograph equipped with a Phenomenex Zebron ZB5 60 m × 0.25 mm id × 0.25 μm phase thickness column using splitless injection, coupled to a Micromass Ultra high resolution mass spectrometer. All analyses were performed under the laboratory's IANZ accreditation (No. 131).

3. Results and discussion

3.1. Samples

A total of nineteen samples were analysed, comprising fourteen stranded and five bycaught specimens (Table 1).

Table 3
Lipid content and organochlorine pesticide levels determined in the blubber of stranded and bycaught common dolphins from New Zealand waters between 1999 and 2005 (percentage and μg/kg wet weight, respectively)

Reference	Sex	Lipid%	α-HCH	*β*-HCH	*γ*-HCH	HCB	Dieldrin	Heptachlor	Heptachlor-epoxide	α-Chlordane	*γ*-Chlordane	*p,p*'-DDE	*p,p*'-DDD	*o,p*'-DDT	*p,p*'-DDT	ΣDDT
WS99-14_30447	M	57	<0.8	2.3	<1.0	28.0	55.0	<0.6	24.0	33.0	<2.0	3900	140.0	250.0	140.0	4430
WS00-01_30890	M	58	<1.0	2.6	<1.0	23.0	59.0	<0.4	22.0	30.0	<2.0	3600	55.0	320.0	49.0	4024
WB02-01_32789	M	52	<1.0	<2.0	<2.0	14.0	19.0	<0.2	12.0	15.0	<1.0	2000	59.0	79.0	64.0	2202
WS02-14_33100	M	68	<1.0	1.5	<1.0	28.0	61.0	<0.2	11.0	11.0	<1.0	1100	120.0	43.0	120.0	1383
WB03-04_34086	M	62	<0.5	1.7	<0.7	16.0	34.0	<0.2	13.0	14.0	<1.0	1300	83.0	71.0	84.0	1538
WB03-17_34705	M	70	<0.7	1.6	<0.9	20.0	37.0	<0.6	8.3	10.0	<5.0	1200	52.0	76.0	81.0	1409
WB03-18_34712	M	64	<0.8	2.1	<1.0	22.0	32.0	<0.4	17.0	14.0	<2.0	1300	110.0	110.0	100.0	1620
WB04-04_35613	M	39	<1.0	<2.0	<2.0	8.6	19.0	<0.8	7.7	8.7	<4.0	1000	41.0	42.0	43.0	1126
WS04-19_36305	M	65	<1.0	<2.0	<2.0	12.0	54.0	<2.0	10.0	14.0	<10.0	740	60.0	35.0	16.0	851
WS04-29_36738	F	47	0.14	0.71	0.2	7.6	12.0	<0.3	<0.8	8.1	<3.0	150	8.1	21.0	14.0	193
WS04-30_36739	F	73	0.38	3.5	0.5	130.0	100.0	<0.6	18.0	36.0	NQ	460	28.0	130.0	36.0	654
WS04-33_36746	F	52	<0.2	<0.4	<0.2	5.3	6.5	<0.7	<1.0	<7.0	<6.0	69	6.7	10.0	8.5	94
WS04-34_36747	F	52	<0.2	<0.3	<0.2	3.1	4.2	<0.4	<0.6	<5.0	<4.0	110	7.6	13.0	9.7	140
WS04-35_36751	F	41	<0.08	0.29	0.11	12.0	9.7	<0.2	<0.5	2.8	<3.0	42	4.6	5.9	6.0	59
WS04-36_36752	F	51	0.16	1.4	0.24	16.0	21.0	<0.6	<1.0	11.0	<5.0	270	19.0	30.0	18.0	337
WS05-06_36823	M	57	<1.0	2.0	<2.0	12.0	64.0	<0.8	12.0	18.0	<5.0	970	87.0	58.0	42.0	1157
WS05-26_37521	M	76	<1.0	2.8	<2.0	28.0	87.0	<1.0	17.0	23.0	<7.0	780	66.0	47.0	23.0	916

NQ = Not quantified.

Males ($n=12$) ranged from one to eleven years of age and from 118 to 227.5 cm in total body length. Females ($n=7$) ranged from less than one to over 10 years of age and from 99 to 200 cm in total body length (Table 1). No sex bias was evident in the stranded samples, with both males and females comprising 50% ($n=7$) of the total sample size. Sex bias was evident in the bycaught samples, with males comprising 100% ($n=5$) of the sample set (Table 1). The majority of the samples ($n=11$) originated from animals that had stranded within the Hauraki Gulf, New Zealand (Fig. 1.). Post-mortem examinations revealed decomposition levels were generally low; fresh ($n=8$), mild ($n=7$) and moderate ($n=4$) (see Table 1).

3.2. Trace elements

Concentrations of trace elements in liver, kidney and blubber are given in Table 2. Of the trace elements determined, chromium and nickel were not detected in any of the samples and concentrations of cobalt, tin and lead were generally low. Liver concentrations of copper and zinc were within the range of concentrations thought to represent homeostatic control (3 to 30 and 20 to 100 µg/g wet weight, respectively (Law, 1996)). Concentrations of mercury ranged from 0.03 to 110 mg/kg wet weight with the lowest concentrations found in blubber and the highest in liver. Generally, mercury concentrations found in the blubber were relatively low, whereas mercury concentrations found in liver and kidney tissue were in line with those reported in other studies. The Hg:Se molar ratio in liver was approximately 1, reflecting the detoxification mechanism by which organic mercury is transformed and deposited as inert mercuric selenide (Law et al., 2001). Failure or overloading of this mechanism could result in toxic effects due to organic mercury ingested from prey (Law, 1996).

Much higher metal concentrations have been reported in three dolphin species from an area of South Australia affected by point source inputs, including a lead smelter (Lavery et al., in review). Inshore bottlenose dolphins (*Tursiops aduncus*) yielded the highest concentrations, whilst the levels of contamination in pelagic common dolphins (*Delphinus delphis*) were similar to those observed in the present study. In the Australian study, common dolphins showed maximum liver concentrations of cadmium, mercury and lead of approximately 11, 165 and 0.13 mg/kg wet weight, respectively. Maximum values in the present study were 21, 110 and 0.74 mg/kg wet weight in liver for cadmium, mercury and lead, respectively, with a maximum cadmium concentration in kidney of 52 mg/kg wet weight.

Comparatively high concentrations of cadmium in the liver and kidney of three stranded dolphins (WS04-34_36747, WS04-35_36751 and WS04-36_36752; 3.4 to 52 mg/kg wet weight) suggest a significant proportion of squid in the diet (Law, 1996). Stomach content analyses for these animals concur, revealing arrow squid (*Nototodarus* sp.) to be the most common prey item by percentage occurrence (Massey University, unpublished data). In contrast, a juvenile female common dolphin from the Gold Coast of Australia (RJM-04) yielded liver and kidney concentrations of 0.02 and 0.07 mg/kg wet weight, respectively, indicating a diet predominant in fish (Law et al., 2003). Apart from cadmium (0.38 mg/kg wet weight in liver) and detectable levels of chromium and nickel (0.21 and 0.31 mg/kg wet weight, respectively), the trace element concentrations observed in the New Zealand common dolphins were similar to those found in a 12 year old stranded adult female common dolphin from the UK (SW1998/104; Law et al., 2001). Mercury, selenium and arsenic levels observed herein for New Zealand common dolphins were in line with those previously reported for common dolphins (Law et al., 2006). A limited number of other data available from New Zealand, from an earlier investigation by Koemann et al. (1972), also concur. Zinc (30 to 40 mg/kg wet weight), arsenic (0.13 to 0.80 mg/kg wet weight), selenium (9.3 to 24 mg/kg wet weight) and mercury (35 to 72 mg/kg wet weight) concentrations in the liver of common dolphins reported by Koemann et al. (1972) were similar to those observed in the present study. Only cadmium (0.21 to 1.6 mg/kg wet weight) concentrations differed to those reported herein, again most likely resulting from dietary differences between the sampled dolphins.

3.3. Organochlorines and PCBs

Concentrations of organochlorine pesticides and chlorobiphenyls in blubber are listed in Tables 3 and 4 respectively. Of the organochlorines determined, dieldrin (up to 100 µg/kg wet weight), HCB (up to 130 µg/kg wet weight), o,p'-DDT (up to 320 µg/kg wet weight) and p,p'-DDE (up to 3900 µg/kg wet weight) were present at the highest concentrations. Sum DDT concentrations (p,p'-DDE + p,p'-DDD + o,p'-DDT + p,p'-DDT) ranged from 17 to 4430 µg/kg wet weight. The maximum value is considerably higher than the concentrations found in the UK (Law et al., 2001) and Australian (Law et al., 2003) common dolphins, whose ΣDDT concentrations (the three p,p'-isomers only) were 690 and 548 µg/kg wet weight, respectively.

Organochlorine concentrations in the blubber of reproducing female dolphins are usually lower than those

Table 4
Chlorinated biphenyl levels determined in the blubber of stranded and bycaught common dolphins from New Zealand waters between 1999 and 2005 (μg/kg wet weight)

Reference	Sex	Lipid%	CB1	CB3	CB4	CB15	CB19	CB28	CB37	CB44	CB49	CB52	CB54	CB70
WS99-14_30447	M	57	0.0096	0.0081	<0.033	0.053	NQ	0.8	0.06	0.57	0.8	8.5	<0.02	0.53
WS00-01_30890	M	58	<0.024	<0.016	<0.17	<0.052	NQ	0.35	<0.033	0.16	0.62	10	NQ	0.12
WB02-01_32789	M	52	<0.007	0.0097	<0.045	<0.043	NQ	0.27	<0.027	0.5	0.54	3.3	<0.064	0.12
WS02-14_33100	M	68	0.015	0.015	<0.04	<0.046	NQ	3.0	<0.031	1.3	2.1	7.2	<0.038	0.76
WB03-04_34086	M	62	0.007	0.0076	<0.029	<0.025	NQ	0.35	<0.024	0.56	0.67	3.3	<0.017	0.13
WB03-17_34705	M	70	0.024	0.026	<0.078	0.03	NQ	0.96	0.036	0.85	1.3	5.6	NQ	0.51
WB03-18_34712	M	64	0.013	0.019	<0.042	<0.032	NQ	0.7	0.029	0.36	0.61	4.6	<0.033	0.21
WB04-04_35613	M	39	0.015	0.018	<0.074	<0.038	NQ	0.33	<0.02	0.1	0.19	2.7	NQ	0.066
WS04-19_36305	M	63	0.012	0.018	<0.086	<0.041	NQ	0.81	0.055	1.0	2.5	7.7	NQ	0.73
WS04-28_36737	F	43	0.0012	0.0014	0.006	0.0065	0.0056	0.17	0.0073	0.15	0.21	0.38	0.0033	0.26
WS04-29_36738	F	47	NQ	0.0011	0.0055	0.006	0.0055	0.16	0.0088	0.16	0.23	0.74	0.002	0.2
WS04-30_36739	M	73	0.003	0.0023	0.012	0.01	0.026	0.99	0.013	1.1	1.2	4.7	0.0066	0.42
WS04-32_36745	F	44	0.0022	0.0015	0.017	0.007	0.022	0.65	0.0084	1.0	0.96	3.7	0.0061	0.6
WS04-33_36746	F	52	NQ	0.0014	0.0076	0.0047	0.0073	0.19	0.0057	0.11	0.21	0.42	0.0021	0.34
WS04-34_36747	F	52	0.0016	0.0014	0.0083	0.0072	0.0073	0.28	0.0092	0.14	0.26	0.5	0.004	0.24
WS04-35_36751	F	41	0.0013	0.00096	0.0098	0.004	0.0095	0.29	0.0057	0.14	0.27	0.59	0.0023	0.23
WS04-36_36752	F	51	NQ	0.0015	0.016	0.0066	0.02	0.58	0.0084	0.36	0.73	2.2	0.0047	0.43
WS05-06_36823	M	57	0.023	0.029	NQ	<0.058	NQ	1.5	<0.054	0.55	2.2	12	NQ	0.3
WS05-26_37521	M	76	0.014	0.019	0.11	<0.032	NQ	4.0	0.048	1.7	4.7	14	NQ	1.6

Reference	Sex	Lipid%	CB74	CB77	CB81	CB99	CB101	CB104	CB105	CB110	CB114	CB118	CB123	CB126
WS99-14_30447	M	57	3.1	0.039	0.046	37	32	0.013	8.8	1.3	0.32	29	1.2	0.15
WS00-01_30890	M	58	3.1	0.069	<0.029	49	28	0.017	8.3	0.71	0.24	30	1.4	0.2
WB02-01_32789	M	52	1.7	<0.04	<0.039	19	18	<0.01	7.6	1.2	0.53	26	0.85	0.11
WS02-14_33100	M	68	7.3	0.13	0.11	31	43	0.009	17	6.2	0.89	54	1.2	0.097
WB03-04_34086	M	62	1.9	0.03	0.047	15	22	0.0077	7.4	2.1	0.42	25	0.7	0.073
WB03-17_34705	M	70	2.9	0.057	0.066	16	28	0.0093	7.7	2.2	0.42	29	0.75	0.082
WB03-18_34712	M	64	2.5	0.034	0.047	19	27	0.0084	8.9	1.3	0.45	32	0.71	0.061
WB04-04_35613	M	39	1.9	0.031	0.017	17	17	0.0042	5.7	0.56	0.23	19	0.47	0.066
WS04-19_36305	M	63	5.6	0.15	0.061	43	39	0.018	16	2.5	0.9	75	1.3	0.24
WS04-28_36737	F	43	0.23	0.022	0.012	1.4	1.8	0.0023	0.63	0.77	0.041	2.2	0.06	0.026
WS04-29_36738	F	47	0.31	0.019	0.016	2.0	3.0	0.0028	0.98	0.61	0.078	3.1	0.12	0.033
WS04-30_36739	M	73	1.8	0.042	0.093	8.1	18	0.014	4.8	3.4	0.39	15	0.5	0.11
WS04-32_36745	F	44	1.7	0.052	0.083	10	17	0.013	4.8	2.8	0.44	17	0.63	0.12
WS04-33_36746	F	52	0.37	0.023	0.015	2.6	2.6	0.0022	1.1	0.64	0.071	3.5	0.11	<0.06
WS04-34_36747	F	52	0.38	0.027	0.016	3.1	2.9	0.0028	1.3	0.8	0.083	4.3	0.12	0.031
WS04-35_36751	F	41	0.37	0.019	0.014	2.4	2.3	0.002	0.96	0.64	0.058	3.3	0.082	0.017
WS04-36_36752	F	51	1.1	0.038	0.043	7.4	8.9	0.0059	3.4	1.5	0.26	11	0.36	0.061
WS05-06_36823	M	57	6.1	0.13	0.053	70	51	0.032	16	1.7	0.83	79	1.6	0.25
WS05-26_37521	M	76	11	0.21	0.12	114	93	0.041	26	6.3	1.5	109	2.3	0.3

Reference	Sex	Lipid%	CB138	CB153	CB155	CB156	CB157	CB167	CB169	CB170	CB180	CB183	CB187	CB188
WS99-14_30447	M	57	202	246	0.25	3.9	2.1	2.9	<0.05	48	126	28	69	0.19
WS00-01_30890	M	58	298	444	0.43	5.4	2.7	3.8	<0.043	76	221	44	134	0.36
WB02-01_32789	M	52	128	188	0.12	8.5	2.2	6.0	<0.04	40	116	20	71	0.17
WS02-14_33100	M	68	134	171	0.17	11	2.5	6.1	0.083	30	73	15	46	0.13
WB03-04_34086	M	62	85	107	0.11	6.0	1.6	3.3	0.075	20	56	10	36	0.095
WB03-17_34705	M	70	103	142	0.084	6.1	1.6	5.7	0.073	26	72	12	46	0.11
WB03-18_34712	M	64	111	143	0.13	6.0	1.9	4.2	0.049	28	78	14	47	0.13
WB04-04_35613	M	39	111	141	0.1	3.7	1.6	2.9	<0.03	30	79	14	50	0.14
WS04-19_36305	M	63	209	295	0.23	14	3.6	12	<0.15	44	94	16	99	0.47
WS04-28_36737	F	43	7.0	8.4	0.029	0.53	0.14	0.35	0.014	2.8	9.2	1.8	5.3	0.017
WS04-29_36738	F	47	15	16	0.098	0.94	0.27	0.76	0.026	6.1	19	3.4	9.9	0.036
WS04-30_36739	M	73	56	61	0.35	2.7	0.93	2.0	0.059	12	36	7.8	22	0.075
WS04-32_36745	F	44	93	110	0.36	2.9	1.1	2.9	0.05	18	48	11	31	0.1
WS04-33_36746	F	52	13	14	0.063	0.82	0.2	0.43	0.023	5.7	13	2.4	6.4	0.023
WS04-34_36747	F	52	18	21	0.09	1.0	0.29	0.72	0.012	5.4	15	3.2	9.2	0.035
WS04-35_36751	F	41	9.6	10	0.037	0.62	0.15	0.33	0.012	2.4	6.4	1.4	3.9	0.015
WS04-36_36752	F	51	45	46	0.22	2.6	0.73	1.7	0.035	16	42	8.7	24	0.084
WS05-06_36823	M	57	342	527	0.3	12	4.4	11	<0.056	74	170	31	163	0.69
WS05-26_37521	M	76	324	462	0.39	17	5.2	18	0.11	56	128	24	139	0.73

Reference	Sex	Lipid%	CB189	CB194	CB196	CB199	CB202	CB205	CB206	CB208	CB209	Σ45CBs	ΣICES7 [a]
WS99-14_30447	M	57	2.0	13	16	0.73	3.8	0.28	1.2	0.86	1.0	892	644
WS00-01_30890	M	58	3.5	19	20	0.65	5.2	0.43	1.2	0.92	0.64	1414	1031
WB02-01_32789	M	52	2.1	12	11	0.34	3.4	0.33	1.3	0.69	0.4	691	480
WS02-14_33100	M	68	1.4	7.4	8.9	0.2	2.3	0.34	0.97	0.42	0.16	686	485
WB03-04_34086	M	62	1.0	5.1	6.1	0.21	1.8	0.19	0.72	0.4	0.19	421	299
WB03-17_34705	M	70	1.3	5.6	6.2	0.2	2.0	0.2	0.66	0.31	0.14	528	381
WB03-18_34712	M	64	1.3	7.7	7.5	0.27	2.4	0.28	0.89	0.47	0.22	553	396
WB04-04_35613	M	39	1.4	8.4	8.9	0.22	2.5	0.3	0.9	0.44	0.21	522	370
WS04-19_36305	M	63	2.2	8.7	8.3	0.28	4.6	0.32	0.99	0.67	0.27	1010	721
WS04-28_36737	F	43	0.17	1.8	1.6	0.062	0.38	0.055	0.32	0.21	0.31	48.9	29.2
WS04-29_36738	F	47	0.32	3.2	2.7	0.11	0.74	0.089	0.46	0.4	0.48	91.8	57.0
WS04-30_36739	M	73	0.46	2.0	1.9	0.12	0.84	0.071	0.21	0.19	0.086	268	192
WS04-32_36745	F	44	0.58	2.0	1.9	0.13	1.0	0.067	0.16	0.14	0.048	386	289
WS04-33_36746	F	52	0.18	3.0	2.4	0.05	0.54	0.075	0.5	0.35	0.58	76.1	46.7
WS04-34_36747	F	52	0.23	1.8	1.6	0.074	0.52	0.051	0.25	0.21	0.22	93.4	62.0
WS04-35_36751	F	41	0.11	1.2	1.1	0.046	0.29	0.04	0.27	0.22	0.45	50.3	32.5
WS04-36_36752	F	51	0.7	6.6	5.7	0.25	1.6	0.16	0.94	0.7	0.96	243	156
WS05-06_36823	M	57	4.0	18	20	0.55	7.8	0.66	2.0	1.3	0.65	1634	1183
WS05-26_37521	M	76	2.8	10	11	0.36	6.0	0.38	0.9	0.73	0.2	1597	1134

NQ = Not quantified.

[a] The seven CB congeners included on the list developed by the International Council for the Exploration of the Sea (ICES) for comparative purposes are CB28, CB52, CB101, CB118, CB138, CB153 and CB180.

of adult males as a result of transplacental and lactational transfer of these lipophilic contaminants to the calves. In this case, ΣDDT concentrations in the blubber ranged from 17 to 337 and 654 to 4430 μg/kg wet weight in females and males, respectively. Similarly, Σ45CB concentrations, the sum of the 45 congeners determined, ranged from 49 to 386 and 268 to 1634 μg/kg wet weight in females and males, respectively.

Borrell and Aguilar (2005) studied the transmission of organochlorine pesticides and chlorobiphenyls from a common dolphin mother to her calf, and estimated the degree of transfer. In the present study, parentage analyses allowed the identification of one mother–offspring pair (WS04-29_36738 and WS04-30_36739 respectively). In terms of blubber concentrations, ΣDDT and ICES7CB levels were both 3.4 times higher in the calf (a male yearling) compared to his respective mother (a 10.5 years old lactating female). The mean transmission of ΣDDTs and ICES7CBs between mother and offspring was calculated at 46% and 42% respectively. This is similar to that observed by Borrell and Aguilar (2005) who reported respective transmissions of 46% and 55% between a mother–calf pair off the south-western Mediterranean coast of Spain. In the present study, examination of the females ovaries revealed multiple corpora scars (Stockin, unpublished data), thus suggesting this may not have been her first calf. This is significant since offloading of contaminants via lactation is typically greater for the first born than for subsequent offspring (Borrell and Aguilar, 2005).

Previously, experimental data on aquatic mammals has been collated to derive dose–response relationships for the adverse health effects of PCB exposure (Kannan et al., 2000). The resulting dose–response relationships, based on experimental studies of PCB-induced immunological and reproductive effects in seals and otters have led to a proposed blubber total PCBs (based on the Aroclor 1254 formulation) threshold concentration for adverse health effects in all marine mammals of 17 mg/kg lipid weight. As analyses are now conducted on a congener basis, a conversion factor of 3× the sum ICES 7 congeners (Table 4) and the total PCB concentration has been established to allow comparisons (Jepson et al., 2005). This relationship was used to estimate the increased risk of infectious disease mortality in porpoises from the UK (Hall et al., 2006). Applying the conversion factor to the data for the New Zealand dolphins, the overall range of total PCB concentrations is from 0.2 to 6.2 mg/kg lipid weight, well below the toxic effects threshold derived by Kannan et al. (2000).

The ICES list of seven congeners was derived specifically to allow comparisons to be made across datasets in which, overall, different suites of congeners were determined. In this case, the ICES7 concentrations ranged from 29 to 289 and 192 to 1183 μg/kg wet weight in females and males, respectively. Comparative concentrations in the UK and Australian common dolphins were 4340 and 650 μg/kg wet weight, respectively (Law et al., 2001, 2003). As the UK animal was an adult female, 4340 μg/kg wet weight represents a concentration ca. 15× higher than the highest equivalent concentration in the female New Zealand dolphins. There are a limited number of other data available from New Zealand, from an earlier investigation by Jones et al. (1999). In that study, ICES7 CB concentrations in two adult male common dolphins were 227 and 1315 μg/kg wet weight — both within the range established in the present study. In one adult male dusky dolphin (*Lagenorhyncus obscurus*) the ICES7 CB concentration was 810 μg/kg wet weight and in six Hector's dolphins (*Cephalorhyncus hectori*) this ranged from 447 to 706 and 319 to 1916 μg/kg wet weight in females and males, respectively.

There are differences in the ICES7CB/ΣDDT ratios at separate locations, reflecting the inconsistent patterns of use and sources of pollutants in each area. In the common dolphins from this study and those from Australia (Law et al., 2003) and the UK (Law et al., 2001), such variations can be observed. In the common dolphin from the UK the ratio is 6.3, whilst in Australia it is 1.19 and in New Zealand ranges from 0.15 to 1.7 (mean value 0.52). This reflects a greater use of PCBs in Europe than in Australasia and a relatively heavy use of DDT in agriculture in New Zealand. As outlined by Aguilar (1984), the ratio of the concentration of *p,p′*-DDE to the sum of the concentrations of *p,p′*-DDT, *p,p′*-DDD and *p,p′*-DDE (the DDE/ΣDDT ratio) can be used to identify recent inputs of DDT as it degrades to DDE over time. In the New Zealand common dolphins this ratio ranges from 0.7 to 0.91, indicating that these contaminants result from historic usage of DDT in New Zealand agriculture. Jones and Giesy (2000) suggest such use has resulted in many agricultural soils in New Zealand having higher concentrations of pesticides (particularly DDT) than those seen in "background" soils. Additionally, these authors noted that the concentrations of organochlorines that have accumulated in some coastal cetacean species, such as Hector's dolphins, were close to those suspected to cause adverse effects in other animal species, but that the risks posed to open ocean marine mammals were small.

Coastal marine mammals such as Hector's and bottlenose dolphins (*Tursiops* spp.) are subject to many known stressors including coastal anthropogenic impacts. It is widely acknowledged that dolphins living in near-shore waters close to agricultural and industrial activity tend to accumulate higher concentrations of toxins (O'Shea, 1999). Conversely, the current conception for oceanic species such as common dolphins is that wider habitat usage places them at lower risk from inshore activities such as point source pollution, inshore fisheries and recreational vessel traffic. However, in the present study, CB concentrations for common dolphins span a similar range to those reported for Hector's dolphins. This may reflect higher usage of coastal waters by New Zealand common dolphins, thus highlighting the potential vulnerability of this species to coastal anthropogenic impacts.

Confounding factors are known to alter toxin loads and require consideration when examining the containment burdens of animals that have stranded, possibly due to ill health. We acknowledge toxin levels in specimens are variable with age and gender, and may change as a result of several different mechanisms, including decomposition (Borrell and Aguilar, 1990) and depletion of lipid reserves with disease or starvation (O'Shea, 1999). However, the examination of apparently healthy bycaught specimens, and our consideration of lipid content and body condition in the present study, alleviates many of these concerns. Thus, the authors believe the toxicity levels reported herein fairly represent pollutant levels present in New Zealand common dolphins.

4. Conclusions

The tendency for marine mammals such as common dolphins to accumulate and carry high burdens of environmental contaminants make these animals suitable bioindicators with which to monitor marine pollution in New Zealand waters. Despite this, few studies have described contaminant levels in New Zealand cetaceans. Our findings suggest concentrations of organochlorine pesticides in New Zealand common dolphins are within a similar range to those reported for Hector's dolphins, an inshore species typically considered to be more susceptible to coastal anthropogenic impacts. We also reveal pollutant transmission levels between a mother–calf pair concur with those previously reported for this species in northern hemisphere waters.

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