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COMPARISON OF ORGANOCHLORINE CONTAMINANTS AMONG SEA OTTER (*ENHYDRA LUTRIS*) POPULATIONS IN CALIFORNIA AND ALASKA

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Abstract—Organochlorine pesticides, polychlorinated biphenyls (PCBs) including non–*ortho* PCBs, polychlorinated dibenzo-*p*-dioxins (PCDDs), and polychlorinated dibenzofurans (PCDFs) were measured in sea otter liver tissue from California, southeast Alaska, and the western Aleutian archipelago collected between 1988 and 1992. Average total dichlorodiphenyltrichloroethane concentrations for California otters (850 μg/kg wet weight) were over 20 times higher than in Aleutian otters (40 μg/kg) and over 800 times higher than otters from southeast Alaska (1 μg/kg). Levels for total PCBs in Aleutian otters (310 μg/kg) were 1.7 times higher than levels in California otters (190 μg/kg) and 38 times higher than otters from southeast Alaska (8 μg/kg). Levels for PCDD and PCDF were extremely low in all otter populations. Levels of PCBs in Aleutian and Californian otters are abnormally high when compared with southeast Alaskan otters. The source of PCBs to the Aleutian Islands remains unclear and vital to understanding the potential impacts to sea otters.

Keywords—Sea otter Organochlorine Pesticide Polychlorinated biphenyls Polychlorinated dibenzo-*p*-dioxins

INTRODUCTION

Sea otters (*Enhydra lutris*) were once distributed across the North Pacific Rim from the Kuril Islands to Baja California and numbered between 150,000 and 300,000 individuals. Fur harvesting during the 18th and 19th centuries reduced the world population to an estimated 1,000 to 2,000 individuals in about a dozen widely separated colonies [1]. Sea otters have partially recovered through the efforts of legal protection, although recovery rates have differed among these remnant colonies. The California otter population is increasing at approximately 5 to 7% per year, compared with 17 to 20% per year for the more northerly populations [2]. Elevated mortality is the most likely reason for this depressed population growth rate; however, the cause or causes of elevated mortality remain unclear [3].

Organochlorine contaminants have been implicated in the decline of several marine mammal species [4,5]. A widely used pesticide, dichlorodiphenyltrichloroethane [1,1-bis(4-chlorophenyl)-2,2,2,-trichloroethane] and its metabolite, *p,p′*-DDE (DDE), have been associated with adverse effects in marine mammals, including possible reproductive changes and the early termination of pregnancy in California sea lions (*Zalophus californianus*) [6]. Polychlorinated biphenyls (PCBs) are also widespread environmental contaminants [7]. Non-*ortho* PCBs (NOPCBs) are of particular concern because they are isostereomers of toxic polychlorinated dibenzo-*p*-dioxin (PCDDs) and polychlorinated dibenzo furan compounds (PCDFs) [7]. Common toxic responses of NOPCBs in rodents include weight loss, thymic atrophy, dermal problems, hepatic damage, teratogenicity, induction of 3-methylcholanthrene hepatic microsomal enzymes, and reproductive problems [7].

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Although the effect(s) of environmental contaminants in sea otters is unstudied, confamilial species such as mink (*Mustela vison*) and ferret (*M. putorius furo*) have been shown to be extremely sensitive to several organochlorines (OCs) [8–21]. Minks exposed to PCB levels as low as 0.64 parts per million (ppm) (causing liver PCB levels of 1.2 ppm) in their feed caused nearly complete reproductive failure [8]. Because levels of PCBs exceeding 1.2 ppm in sea otter livers have been documented in California sea otters [22,23], these compounds are of concern. In addition, minks exposed to environmental PCB levels have exhibited symptoms of PCB intoxication, including anorexia, listlessness, nervousness, seizures, bloody stools, fatty liver, kidney degeneration, gastric ulcers [11,13], hepatic changes including microsomal mixed function oxidase induction [24], and decreased birth weights, growth rates, and survival of kits [9,13]. Polychlorinated biphenyls have been implicated in mink declines in New York State near Lake Ontario [25] and in the decline of river otters throughout much of Europe [26].

Unlike most other marine mammals that migrate extensively, sea otters provide an unusual opportunity for ecotoxicological studies because both the otters and their principal prey are relatively sedentary (the equilibrium density of otters was estimated to be 20–30/km^2 at Amchitka Island [2]); thus, their contaminant burdens should reflect conditions of local habitats. Sea otters provide an unusual opportunity to study a high-trophic level marine consumer inhabiting highly industrial to extremely remote habitats.

In this study, we report levels of OC pesticides, PCBs (including NOPCBs), PCDDs, and PCDFs, in sea otters from central California, southeast Alaska, and the western Aleutian Islands. These sea otter populations provide comparative data from apparently stable populations (Adak and Amchitka, both

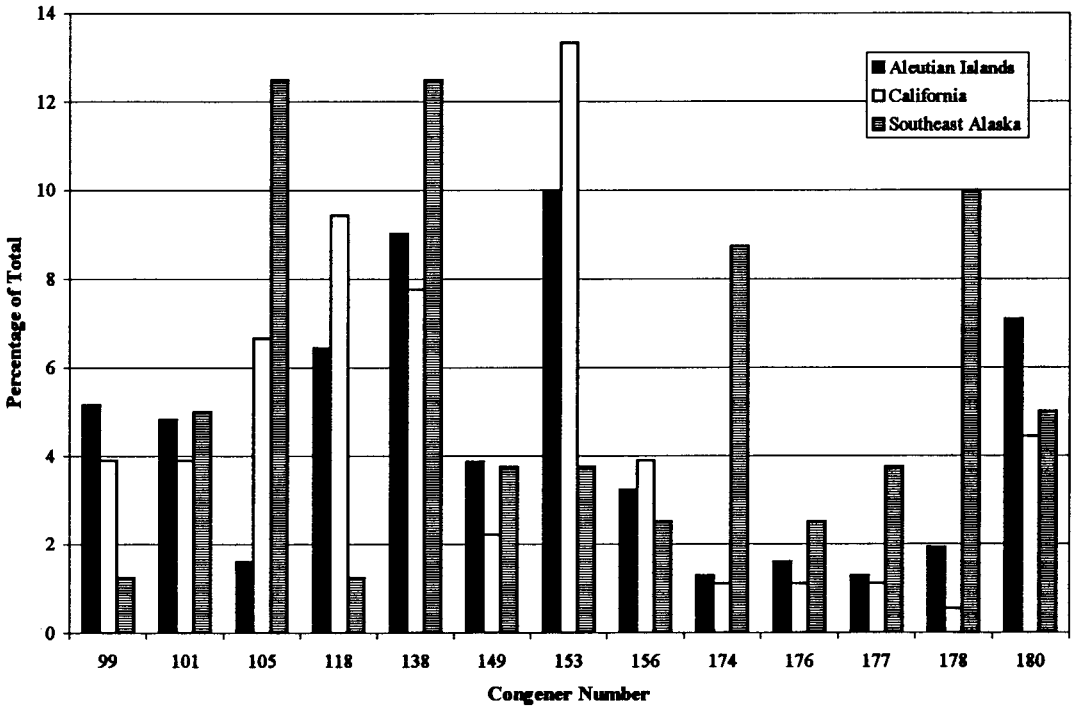

Fig. 1. Polychlorinated biphenyl congener composition for sea otter livers collected from California, the western Aleutian Islands, and southeast Alaska, 1988–1992.

of which were thought to be at or near equilibrium density during our study) and from a population growing near the species' maximum potential (southeast Alaska). The California sea otter was chosen to determine if exposure to organochlorines (OCs) could be impeding the growth of the California sea otter population.

This particular comparison was chosen for several reasons. First, we had access to tissue samples from each region. In addition, the California population occurs in close proximity to various industrial and agricultural pollutant sources and for many years has been characterized by a depressed rate of population increase. The habitat in southeast Alaska was purportedly less contaminated, being more remote from industrial/agricultural pollutants and upstream from convective inputs from California by virtue of the strong southward flowing California Current. The Aleutians are oceanic islands, far distant from any known industrial/agricultural activity and human population centers. High densities of sea otters occur throughout most of the central and western Aleutians. In addition, we discuss potential sources and the ecotoxicological implications of these compounds to sea otter populations.

METHODS

Sample collection

Our analyses were done on liver tissue samples obtained from beach-cast sea otter carcasses or recently killed individuals. When possible, we selected adult males to reduce the variability in toxicological data from female marine mammals [5] (Table 1). All tissue samples in California were taken from beach-cast sea otter carcasses provided by the California Department of Fish and Game. The California sea otter tissue used in our analyses were from adult males collected between 1988 and 1991. Animals from southeast Alaska were acquired by native hunters at Middle Island and Ogden Bay during May 1991. Most of our material from the Aleutian Islands was obtained from beach-cast carcasses at Adak and Amchitka islands during 1991 and 1992. A sample also was obtained from one otter that drowned during capture operations at Amchitka Island in July 1992. Because it was difficult for us to obtain usable material from the Aleutian Islands, our samples include three adult males and four female otters of varying ages (Table 1). Otter tissues were wrapped in aluminum foil bags and immediately frozen until analysis. That we obtained several samples from beach-cast carcasses could complicate data interpretation because the degree of emaciation in these otters is unknown.

Organochlorine and polychlorinated biphenyl determination

Tissue samples were spiked with internal standards, ground with anhydrous sodium sulfate, extracted with 1:1 dichloromethane (DCM):hexane and reduced by rotary evaporation. Samples were cleaned and separated using Florisil® (1 cm i.d. $\times$ 27 cm, 0.5% distilled water deactivated, 60/100 mesh) into three fractions, eluting sequentially with 100% hexane for frac-

Table 1. Number and sex class of sea otter liver samples analyzed for organochlorine compounds from California, the Aleutian Islands, and southeast Alaska

	California 1988–1991	Southeast Alaska 1991	Aleutian Islands	
			Adak 1991–1992	Amchitka 1992
Male				
Adult	5	7	1	1
Young	3	0	1	0
Female				
Adult	1	0	1	1
Young	0	0	2	0

Table 2. Levels of chlorinated organics in sea otter liver tissue from California, the Aleutian Islands, and southeast Alaska collected between the years 1988 and 1992[a]

	% Water	% Lipid	ΣPCB	ΣDDT	Σchlordane	ΣNOPCB[b]	ΣPCDD	ΣPCDF	*p,p′*-DDE	HCB	β-HCH	Dieldrin	TCP methanol
Aleutian (n = 7)													
Mean	71	3	310	36	15	0.028	0.001	0.001	36	2	5	3	ND
Std Dev	2	2	480	500	97	0.076	0.001	ND	500	4	16	5	
California (n = 9)													
Mean	69	4	190	850	31	0.036	0.004	0.001	840	2	9	1	4
Std Dev	2	1	350	380	56	0.017	0.004	ND	380	1	21	11	7
Southeast Alaska (n = 7)													
Mean	68	3	8	1	1	0.004	0.001	ND	1	1	6	2	ND
Std Dev	5	1	14	3	1	0.002			3	ND	3	4	

[a] Values are reported in micrograms per kilogram wet weight. ΣPCB, total polychlorinated biphenyl (PCB) levels; ΣDDT, total dichlorodiphenyltrichloroethane levels; ΣNOPCB, total non-*ortho* PCB levels; ΣPCDD, total polychlorinated dibenzo-*p*-dioxin levels; ΣPCDF, total polychlorinated dibenzofuran levels; *p,p′*-DDE, dichlorodiphenylethylene; TCP methanol, Tris(4-chlorophenyl) methanol; ND, not detected.
[b] Congeners 37, 77, 81, 126, 169, and 189.

tion 1, 30% DCM:hexane for fraction 2, and 50% DCM:hexane for fraction 3.

Samples were analyzed on a Hewlett-Packard (HP) 5890 series II gas chromatograph (GC) equipped with a ^{63}Ni electron-capture detector and a HP 7673A automatic sampler (Hewlett Packard, Avondale, PA USA). Two 30-m, 0.25-mm i.d., 0.25-mm film thickness, DB-5 and DB-17 columns (J&W Scientific, Folsum, CA, USA) were used to provide dual-column confirmation of analyte retention times. Analyte peak retention times were compared with those of known standards run under identical conditions to assure accurate peak identification. Instrumental internal standards (GCI standards) were utilized to correct for volume differences between samples. In addition, Standard Reference Material (organic mussel tissue, *Mytilus eludes*, 1974) from the National Institute of Standards and Technology was analyzed to assure quality control.

Non–ortho *polychlorinated biphenyl, polychlorinated dibenzo*-p-*dioxin, and polychlorinated dibenzofuran determination*

The tissues were extracted according to the previously mentioned procedures, frozen, and shipped to the Canadian Wildlife Service in Hull, Quebec, Canada, for analyses. Prior to analysis, extracts were spiked with a cocktail of ^{13}C-labeled NOPCB, PCDD, and PCDF internal standards. Extracts were eluted with 50% DCM:hexane in automated gel-permeation chromatograph (GPC, 60 g Biobeads SX-3) [27]. The desired fraction was reduced and transferred to the top of a 10-g alumina column (1 cm i.d. × 12 cm long, basic alumina, activated, Fisher Scientific, Fairlawn, NJ, USA) and eluted with 50% DCM:hexane, reduced, and further cleaned on a carbon column (packed with 45 mg AX-1 carbon 0.65 cm i.d. × 6 cm). Samples were eluted sequentially with a forward wash of hexane (discarded), followed by a second wash of 100% DCM (which contained organochlorine pesticides and most PCB congeners), and then back eluted with 100% toluene (which contained NOPCBs, PCDDs, and PCDFs). The back-eluted fraction was then reduced and separated by Florisil (1.2% deactivated).

Samples were spiked with PCB 112 and then analyzed on a VG AUTOSPEC double-focusing, high-resolution mass spectrometer linked to a (VG Instruments, Wiesbaden, Germany) HP 5890 series II high-resolution GC equipped with a Carlo Erba CTC-A200S autosampler (Carlo Erba, Milan, Italy). The mass spectrometer was operated in the Selected Ion Monitoring (VOLTAGE SIR) mode with the following conditions: EI 70 EV; source temperature = 280°C; interface temperature = 280°C. Polychlorinated dibenzo-*p*-dioxins/PCDFs were analyzed with a resolution of 10,000, whereas the resolution for NOPCBs was 7,000.

In addition to internal and recovery standards, an external standard mixture was analyzed under conditions identical to those of our samples. Relative response factors for each native analyte and ^{13}C-labeled compound were calculated daily. The linearity of the instrument was tested with a five-point calibration. Residue levels for NOPCBs, PCDDs, and PCDFs were determined by using an internal standard quantitation method on a VAX workstation station 4000 operating TRACES and DIOXIN programs (VG Instruments). Recoveries for ^{13}C-labeled NOPCBs, PCDDs, and PCDFs were calculated and were considered acceptable between 70 and 110%. Peak identities were confirmed by, monitoring accurate masses (four decimal places) while running the GCMS in high-resolution mode, proper chlorine isotope ratio (<20% deviation from the correct ratio) for the two strongest ions in the M+ cluster, and the proper retention time (<5-s deviation from the retention time obtained for authentic standard). Chlorinated diphenyl ethers can interfere with PCDFs analyses and were proven absent by routinely monitoring one mass in each chromatographic window.

RESULTS

Total DDT (ΣDDT = *p,p′*-DDT, *p,p′*-DDE, and *p,p′*-DDD) levels were highest in sea otters from California (850 μg/kg), intermediate in otters from the Aleutians (40 μg/kg), and lowest in otters from southeast Alaska (1 μg/kg; Table 2). Differences among locations differed significantly (one-way analysis of variance [ANOVA], $p < 0.05$). Levels of other pesticides, including chlordanes, HCB, β-HCH, and dieldrin, were very low in otters from both California and Alaska (Table 2). The levels of HCB (2 μg/kg), β-HCH (9 μg/kg), and dieldrin (1 μg/kg) are highest in California otters, yet these values could not be shown to differ significantly among regions (one-way ANOVA, $p > 0.05$). Tris (4-chlorophenyl) methanol (TCP methanol) was only detected in California otters (Table 2).

Total PCB levels (ΣPCB) in sea otters from both the Aleutians (310 μg/kg) and California (190 μg/kg) were significantly higher than those in southeast Alaska (geometric mean: 8 μg/kg, one-way ANOVA, F = , $p < 0.05$) (Table 2). Though

Aleutian otters had the highest ΣPCB levels, they could not be shown to differ significantly from levels in California otters. In addition, PCB congeners were determined for each otter population. The primary congeners in each population included PCB 99, 101, 105, 118, 138, 149, 153, 156, 174, 176, 177, 178, and 180.

Otters from each population were also analyzed for NOPCBs, including congeners 37, 77, 81, 126, 169, and 189. Total NOPCB levels were highest in otters from California (geometric mean: 33 μg/kg), but the three populations could not be shown to differ significantly (one-way ANOVA, $p > 0.05$) (Table 2).

Unlike PCBs, PCDDs and PCDFs occurred at very low to undetectable levels in otters from all three regions. Levels of PCDDs and PCDFs were not significantly different in otters from California and the Aleutian Islands ($p > 0.05$). The southeast Alaskan otter population was not included in the statistical comparison because of extremely low or undetectable levels in that region (Table 2).

Ratios of ΣDDT/ΣPCB and ΣDDT/Σchlordane are similar for Aleutian (0.10 and 2.40, respectively) and southeast Alaskan otters (0.24 and 1.26, respectively), whereas both these ratios were an order of magnitude greater in California otters (4.3 and 29, respectively). Ratios of DDE/DDT were similar in both the Aleutians (26.0) and southeast Alaska (11.0), whereas in California the DDE/DDT ratio was nearly two orders of magnitude greater (1,200). Total chlordane/ΣPCB ratios were similar among all three otter populations.

When comparing PCDD congeners among the three regions, we found that in California otters the greatest toxic equivalents (TEQs) resulted from 1,2,3,7,8-PeCDD, whereas in the Aleutians the greatest contributor is 2,3,7,8-TCDD. When comparing PCDF congeners we found a similar pattern with 2,3,7,8-TCDF contributing the greatest risk. The only TEQ reported for southeast Alaskan otters for either PCDDs or PCDFs was OCDD (0.0010). NOPCB congener 126 comprises 25 and 16% of ΣTEQ profile from the California and Aleutian otters, respectively. Interestingly, congener 156 contributes the most to the total TEQs of the California and Aleutian otters (28 and 41%, respectively). When comparing the percent contribution of PCDDs/PCDFs, mono- and di-*ortho*, and NOPCBs to total TEQs, we find that NOPCBs contribute 26% to the total in California otters, 17% in Aleutian otters, and 49% in southeast Alaskan otters (Table 3).

DISCUSSION

Pesticides

Levels of ΣDDT were significantly higher in California sea otters when compared to levels in otters from either the Aleutians or southeast Alaska. The source of DDT in the California ecosystem is most likely from the extensive use and production of DDT from the 1950s through the early 1970s in this region [28]. The ΣDDT levels we found in livers of California otters (geometric mean 850 μg/kg wet weight) are not exceptionally high when compared to reported levels in other marine mammals from this region [6,29]. Rote [22] found levels of DDE in otter livers collected along the central California coast between 1968 and 1973 ranged from 300 to 13,000 μg/kg, with a geometric mean of 2,200 μg/kg. DeLong et al. [6] reported ΣDDT levels in blubber of California sea lions of 820,000 μg/kg wet weight and associated this with premature births. Similarly, Le Boeuf and Bonnell [30] reported ΣDDT concentrations of 911,000 μg/kg wet weight in blubber from the same species. The ΣDDT we found in California otters are roughly three orders of magnitude less than this. Although this comparison is between species and tissue matrices, the difference is noteworthy. It appears as though the California ecosystem is much less contaminated with DDT than in the past. This conclusion is supported by a recent study by Lieberg-Clark et al. [29], who reported a decrease of over two orders of magnitude of ΣDDT levels in California sea lions from the 1970s.

Although the levels of DDT in the California ecosystem have declined by orders of magnitude from the 1970s, the coastal ecosystem is still much more contaminated than either Alaskan ecosystem in this study. The ratios of DDE/DDT for California otters are comparable with air samples from equatorial regions where DDT is still used extensively [31] and are similar to ratios for other marine mammals from California [32]. The difference in the DDE/DDT ratios between Aleutian and California otters is noteworthy and suggests that the source of DDT to California is both older and enhanced in this region. The results of ΣDDT we report for California otters, although most likely not problematic, indicate that the California ecosystem is still contaminated with DDT compounds nearly 25 years after it was banned from use in the United States.

The levels ΣDDT we report in California sea otters do not appear to be toxicologically significant. Minks exposed to 11 mg DDT per day for 66 consecutive days did not experience significant decreases in the number of whelps born or other extreme physiological effect [10]. ΣDDT levels we report in California otters, although significantly higher than Alaskan otters, are below levels associated with reproductive problems in this closely related species [10].

The levels we report of other organochlorine pesticides, including chlordanes, HCB, β-HCH, and dieldrin, were all extremely low in each otter population, and these would appear to pose little or no toxicological threat to otters. The only pesticide compound detected exclusively in California otters (Table 2) was TCP methanol. This result is consistent with the potential source of TCP methanol; it is possibly a metabolite of a contaminant in technical DDT [33]. The source of TCP methanol must therefore be associated with DDT use, production, and disposal, all of which were extensive in the California ecosystem [28].

*Polychlorinated biphenyls, polychlorinated dibenzo-*p*-dioxins, and polychlorinated dibenzofurans*

Levels of ΣPCB were highest in the Aleutian Islands, followed by California and southeast Alaska (Table 2). The Aleutian Islands and California were not shown to differ significantly, but both were significantly higher than southeast Alaska. This result is surprising, considering the remote and presumably pristine nature of the western Aleutian Islands. Potential sources of PCBs to the western Aleutian archipelago include local contamination sources (point-source input) and/or atmospheric/oceanic currents supplying the Aleutian Islands. Adak and Amchitka Islands form part of the boundary between the Pacific Ocean and Bering Sea and are located approximately 2,000 km west of the Alaskan mainland and 1,000 km east of the Kamchatka Peninsula. Although remote, the islands of Adak and Amchitka have a long history of military occupation since World War II, which may be contributing a point source of PCBs to these ecosystems.

Additional sources of PCB to the western Aleutian Islands could involve atmospheric deposition and transport by oceanic currents [31]. Air and oceanic currents tend to move in a

Table 3. Mean values calculated 2,3,7,8-TCDD equivalences (TEQs) and geometric mean values for sea otter livers collected from California, the Aleutian Islands, and southeast Alaska[a]

		California		Aleutian		Southeast Alaska	
	TEF	CA mean[b]	TEQ[c]	AL mean[b]	TEQ[c]	SE mean[b]	TEC[c]
PCDDs and PCDFs							
2,3,7,8-TCDD	1.0000	0.58	0.58	0.48	0.48	ND	ND
1,2,3,7,8-PeCDD	0.5000	1.82	0.91	0.51	0.26	ND	ND
1,2,3,4,7,8-HxCDD	0.1000	0.96	0.10	0.21	0.021	ND	ND
1,2,3,6,7,8-HxCDD	0.1000	1.57	0.16	0.71	0.071	ND	ND
2,3,4,6,7,8-HxCDD	0.1000	0.35	0.04	0.13	0.013	ND	ND
1,2,3,4,6,7,8-HpCDD	0.0100	1.17	0.01	0.36	0.004	ND	ND
OCDD	0.0010	1.33	0.001	1.0	0.001	1.02	0.0010
2,3,7,8-TCDF	0.5000	0.13	0.065	0.21	0.11	ND	ND
1,2,3,7,8-PeCDF	0.0500	0.08	0.004	0.05	0.003	ND	ND
2,3,4,7,8-PeCDF	0.0500	0.55	0.028	0.52	0.026	ND	ND
1,2,3,4,7,8-HxCDF	0.1000	ND	ND	ND	ND	ND	ND
1,2,3,7,8,9-HxCDF	0.1000	ND	ND	ND	ND	ND	ND
1,2,3,6,7,8-HxCDF	0.1000	ND	ND	ND	ND	ND	ND
2,3,4,6,7,8-HxCDF	0.1000	ND	ND	0.11	0.011	ND	ND
1,2,3,4,6,7,8-HpCDF	0.0100	0.12	0.001	0.11	0.0011	ND	ND
1,2,3,4,7,8,9-HpCDF	0.0100	ND	ND	ND	ND	ND	ND
OCDF	0.0010	0.32	0.0003	0.16	0.0002	ND	ND
Sum of PCDD/F TEQs			1.9		0.99		0.0010
Mono, di-*ortho* PCBs							
PCB 105	0.0001	12400	1.2	5220	0.5	1000	0.10
PCB 118	0.0001	17200	1.7	20310	2.0	100	0.01
PCB 156	0.0005	6500	3.3	9700	4.9	200	0.10
PCB 167/128	0.00001	5900	0.06	22150	0.22	100	0.00
PCB 170/190	0.0001	2800	0.28	9840	1	100	0.01
PCB 180	0.00001	7700	0.08	22150	0.22	400	0.004
Sum of mono-, and di-*ortho* PCB TEQs			6.6		8.8		0.23
Non-*ortho* PCBs							
PCB 37		0.82		ND		0.70	
PCB 77	0.0005	1.49	0.0007	1.05	0.0005	1.41	0.0007
PCB 81		0.53		0.44		0.25	
PCB 126	0.1000	30	3.0	19	1.9	2.2	0.22
PCB 169	0.0100	2.52	0.025	2.33	0.023	ND	ND
PCB 189		1.40		2.37		ND	
Sum of non-*ortho* PCB TEQs			3.0		1.9		0.22
Total TEQ			12		12		0.44
% PCDD/PCDF TEQs			16%		8%		0.2%
% Mono, di-*ortho* PCBs TEQ			58%		75%		51%
% non-*ortho* PCB TEQs			26%		17%		49%

[a] Levels are reported in nanograms per kilogram of wet weight. TCDD, tetrachlorodibenzo-*p*-dioxin; TEQ, toxic equivalent; PCDD, polychlorinated dibenzo-*p*-dioxin; PCDF, polychlorinated dibenzo furan; OCDD, *ortho*-chlorodibenzo-*p*-dioxin; OCDF, *ortho*-dichlorobenzo furan; PCB, polychlorinated biphenyl. PCDD/F TEFs from [41], PCB TEFs from [42]; ND, value not detected in samples.
[b] CA mean, AL mean, SE mean = geometric mean values for compounds from California, the Aleutian Islands, and southeast Alaska, respectively.
[c] TEQs resulted from multiplying TEFs by the corresponding geometric means within the geographical region.

clockwise rotating gyre in the North Pacific [34], possibly transporting anthropogenic chemicals from the lower latitudes in Asia to the Aleutian archipelago. Although PCBs have been banned in North America, many Third World countries on the Asian continent still use these materials [31]. Ratios of ΣDDT/ΣPCB for both Alaskan otter populations are comparable to those found in air samples [31] from arctic regions, supporting an atmospheric source of OCs to the Aleutian Islands.

The comparatively low ΣPCB levels in southeast Alaska demonstrate that levels found in sea otters from the Aleutian Islands and California are abnormally high. Elevated PCB levels in these two populations are of concern because reproductive problems and reduced kit survival in minks have been correlated with PCB exposure at environmental levels [8,10–14]. Recent data suggest that high preweaning pup mortality accounts for the impaired growth rate of the California sea otter population [35]. Monson [36] found a similarly high rate of pup mortality at Amchitka Island. It appears that our comparison of PCBs in California otters with those in the Aleutian Islands involves two ''ecotoxicologically'' similar populations, whereas the actual valid control population is southeast Alaska.

The major PCB congeners in both California and Aleutian Islands include the congeners 105 (IPUAC), 118, 138, 153, and 180. This is not surprising because these congeners seem to bioaccumulate in marine food chains and appear as a common theme throughout marine mammal literature [4,37]. However, from a toxicological point of view, two congeners, 126 and 156, combine to have over 50% of the total TEQs in the California and Aleutian otters. In Dutch otters the NOPCBs dominate the TEQ profiles; however PCB, 156 also contributes substantially to the total TEQs [15].

The levels of ΣPCDD and ΣPCDDF are either below 5 ng/kg or not detected in all three otter populations (Table 2).

Because these levels are so low in each region, a point source of these compounds to any of the regions is highly improbable. More likely, the source of PCDDs and PCDFs to each region is atmospheric/oceanic. Because the levels are extremely low and the TEQ data imply the most risk being associated with NOPCBs, PCDDs and PCDFs are most likely not toxicologically significant to sea otters in California and Alaska.

Sea otters from both California and the Aleutian Islands had detectable levels of many OCs, which alone do not appear toxicologically significant, however synergistically, but may be affecting the health of these otters. Synergistic effects of PCB exposure with HCB and dieldrin accompanied by environmental stresses (i.e., severe cold or food limitations) have been documented in the confamilial mink [9,26,38–40]. Potential synergistic effects of contaminants are relatively unstudied and may play a significant role in the health of sea otters.

It appears that current levels of DDTs and other organic pesticides do not adversely affect the well-being of sea otters in California or Alaska nor are levels of PCDDs and PCDFs of present concern. However, PCB levels are sufficiently high to be of concern to sea otters in California and the western Aleutian Islands. Because the California ecosystem is located in close proximity to industry and agriculture, elevated levels of PCBs in California sea otters is not surprising. However, the source(s) of PCBs to the western Aleutian Islands is less clear. Otter samples in this study were selected solely from islands with a history of military presence. Without chemical and toxicological data from islands within the Aleutian archipelago that did not have a military occupation, the question involving the source of PCBs to the Aleutian Islands remains unanswered. In addition, the effects of PCB exposure to otters in California and the Aleutian Islands remain unclear until a mechanism of action and ecological data are more closely linked in these otter populations.

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