Chlorinated hydrocarbon contaminants in marine birds of the temperate North Pacific

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Abstract

North Pacific marine birds are continuously exposed to chlorinated hydrocarbon pollutants transported by atmospheric and oceanic processes from distant sources. Birds breeding and/or wintering in coastal areas that receive effluent from manufacturing plants or concentrated runoff from agricultural areas are further exposed to chemicals from local sources. Because of their lipophilicity, chlorinated hydrocarbons and their metabolites tend to bioaccumulate in animal tissues, including those of marine birds. Species differences in contamination are determined primarily by variation in diet, movements outside the breeding area, and physiology. In the 1960s and 1970s breeding populations of Brown Pelicans *Pelecanus occidentalis*, Double-crested Cormorants *Phalacrocorax auritus*, Common Egrets *Casmerodius albus*, and Common Murres *Uria aalge* at a number of sites on the southern California coast were severely impacted by DDT contamination via its effects on eggshell thickness. Hazardous levels of DDE and PCBs were still present during the 1980s in eggs of a number of species breeding in San Diego and San Francisco bays. In the Strait of Georgia, British Columbia, high levels of polychlorinated dioxins were correlated with sublethal effects in Great Blue Herons *Ardea herodias*. In Puget Sound, Washington, a number of sublethal biomarkers associated with chlorinated hydrocarbon contaminants were reported in breeding Glaucous-winged Gulls *Larus glaucescens*. Some species, particularly Leach's Storm-Petrels *Oceanodroma leucorhoa* and Double-crested Cormorants, have proven to be effective indicators for monitoring temporal and geographical trends in environmental levels of persistent contaminants. First reports of previously undetected chlorinated hydrocarbons such as tris-(chlorophenyl)methanol and a new polybrominated aromatic contaminant in seabirds attest to the ongoing usefulness of such monitoring.

Résumé

Les oiseaux marins du Pacifique Nord sont continuellement exposés à la contamination par les hydrocarbures chlorés, transportés par les courants marins et atmosphériques à partir de sources éloignées. Les oiseaux qui nichent ou qui hivernent dans les régions côtières, sont également exposés aux sources locales de contaminants chimiques, lorsque ces régions reçoivent les effluents des usines de fabrication ou le ruissellement abondant des entreprises agricoles. La lipophilie fait que les hydrocarbures chlorés et leurs métabolites ont tendance à s'accumuler dans les tissus animaux, y compris ceux des oiseaux marins. Les écarts de contamination entre les espèces s'expliquent surtout par leurs régimes alimentaires, par leurs déplacements en dehors des aires de reproduction et par leur physiologie. Dans les années 1960 et 1970, les populations d'oiseaux nicheurs des espèces Pélican brun *Pelecanus occidentalis*, Cormoran à aigrettes *Phalacrocorax auritus*, Grande aigrette *Casmerodius albus* et Marmette de Troïl *Uria aalge*, en différents endroits de la côte sud de la Californie, ont fortement souffert de la contamination par le DDT, ce produit chimique affectant l'épaisseur de la coquille des oeufs. Dans les années 1980, des concentrations nocives de DDE et de BPC étaient toujours présentes dans les oeufs de différentes espèces d'oiseaux nichant dans les baies de San Diego et de San Francisco. Dans le détroit de Georgie, en Colombie-Britannique, on a corrélé les fortes concentrations de dioxines polychlorées et des effets subletaux observés chez le Grand héron *Ardea herodias*. Dans le détroit de Puget (État de Washington), on a aussi relevé un certain nombre d'indicateurs biologiques subletaux, qui démontrent la contamination des reproducteurs de l'espèce Goéland à ailes grises *Larus glaucescens* par les hydrocarbures chlorés. Certaines espèces, particulièrement le Petrel cul blanc *Oceanodroma leucorhoa* et le Cormoran à aigrettes, sont réputées d'excellents indicateurs pour la surveillance des tendances temporelles et spatiales de la contamination par les substances rémanentes. Les premiers rapports faisant état de la contamination des oiseaux marins par des substances jusqu'alors non détectées, comme le tris-(chlorophényle)méthanol et un nouveau contaminant polybromé aromatique, témoignent du bien-fondé constant des activités de surveillance

1. Introduction

Widespread contamination of marine food chains by chlorinated hydrocarbons has been documented since the 1960s (Moore and Tatton 1965; Risebrough et al. 1967, 1968; Potts 1968). On the Pacific coast of North America, reduced reproductive success due to the eggshell-thinning effects of DDE were reported for a number of marine bird populations (Risebrough et al. 1971; Faber et al. 1972; Gress et al. 1973). Partly as a consequence of such findings, uses of persistent organochlorine pesticides and industrial chemicals including polychlorinated biphenyls (PCBs) were severely restricted by legislative and regulatory actions in Canada, the United States, and many other countries. There have also been a number of international agreements to reduce chemical contamination of the world's oceans (United Nations Environment Programme 1985). Those actions decreased the chlorinated hydrocarbon

levels of inshore food chains at a number of sites (Blus 1982; Dyck and Kraul 1984; Nisbet and Reynolds 1984; Olsson and Reutergardh 1986; Elliott et al. 1988a; Pearce et al. 1989); however, levels of most measured contaminants did not decrease significantly from the late 1960s to the late 1980s in offshore seabird species from both the North Atlantic and North Pacific (Elliott et al., in press).

In recent years, advances in analytical technology have revealed widespread environmental contamination of marine and aquatic food chains by polychlorinated dibenzodioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs) resulting from industrial processes such as the chlorine bleaching of wood pulp and the use of chlorophenolic biocides (Kuehl et al. 1987). In the Pacific Northwest of North America, where many forest industry operations are located, elevated PCDD and PCDF levels in eggs and tissues of coastal populations of fish-eating birds have been related to forest industry sources in British Columbia (Elliott et al. 1989a; Whitehead et al. 1990). Tanabe (1988) has suggested that the highly toxic non-ortho PCB congeners may pose an even greater threat than dioxins to the health of marine organisms. There is a need, therefore, to continue to study the levels and effects of chlorinated hydrocarbons in marine organisms.

There have been a number of reviews of the exposure and effects of environmental pollutants on marine birds (Bourne 1976; Ohlendorf et al. 1978; Noble and Elliott 1986). In this paper we cover available data on chlorinated hydrocarbon contaminants in marine birds from the temperate North Pacific. Marine birds included in this review are primarily those populations that both breed and winter in the marine environment, although we excluded marine shorebirds. Chemical concentrations are on a wet-weight basis unless stated otherwise.

2. Exposure of marine birds to chlorinated hydrocarbons

2.1. Influence of atmospheric and oceanic transport

Levels of contamination vary geographically on the basis of local inputs, such as agricultural runoff and industrial effluents, and the degree of exposure to chemicals transported by atmospheric, fluvial, and oceanic processes from more distant sources. The data on seabirds indicate that some coastal areas, such as San Diego Bay, California, are still highly contaminated with residues of persistent organochlorine pesticides even 10–20 years after severe restrictions on the production and use of most of those chemicals.

Prevailing winds in the North Pacific are mainly from the west, northwest, or southwest; thus, atmospherically borne contaminants in offshore areas of the eastern Pacific are expected to have a mainly Asian origin. Asian sources were suggested by Elliott et al. (1989b) and Ohlendorf and Harrison (1986) to explain elevated HCH contamination in some samples. DDT compounds in air and water samples from the North Pacific were higher than those in the North Atlantic (Bidleman et al. 1981; Norstrom and Muir 1988) or the Gulf of Mexico (Chang et al. 1985).

Organochlorines associated with organic particulate matter are also carried by ocean currents. The Kuroshio Current flows eastward from Japan to the Canadian coast, where it splits to become the north-flowing Alaskan Current and the south-flowing California Current. The California Current Extension and the North Equatorial Current flow westwards just south of the Hawaiian Islands to the Philippines. These currents may have influenced the south-to-north declining gradient in contaminant levels in Hawaii (Ohlendorf and Harrison 1986). The Davidson Current, which flows seasonally northwards along the coast of California, may be responsible for transport of contaminants to western Vancouver Island (Elliott et al. 1989b).

2.2. Influence of diet, seasonal movements, and physiology

Sympatric species differ markedly in contamination, clearly demonstrating the importance of diet, movements outside the breeding area, and physiology. Organochlorines biomagnify in food chains, and thus species that feed on fish rather than zooplankton are expected to accumulate higher residues (see Henny et al. 1982; Ohlendorf et al. 1982; Noble and Elliott 1986). However, the elevated levels in storm-petrels (e.g., Henny et al. 1982) show that foraging behaviour also plays a role. Storm-petrels obtain their diet of mainly small invertebrates from the oily surface microlayer of the ocean where organochlorines, particularly the more volatile compounds such as toxaphene and HCH, are deposited (Seba and Corcoran 1969).

Finally, as suggested by Ohlendorf and Harrison (1986) and demonstrated by Henny et al. (1990), exposure to contaminants on the wintering areas of marine birds can be significant, and result in elevated residues in eggs. Further discussion of the influence of diet and seasonal movements of Pacific marine birds on patterns of contaminant accumulation are found in Ohlendorf et al. (1985), Ohlendorf and Harrison (1986), and Elliott et al. (1989b).

2.3. Organochlorine pesticides

Organochlorine pesticides are synthetic compounds used widely to control agricultural and forest pests and the transmission of vector-borne diseases. The most abundant pesticide derivative is DDE, the major metabolite of DDT (dichlorodiphenyltrichloroethane). Other compounds detected include DDD, DDT, dieldrin, heptachlor epoxide, mirex, photomirex, toxaphene, oxychlordane, cis- and trans-chlordane, cis- and trans-nonachlor, endrin, HCH (hexachlorocyclohexane), and HCB (hexachlorobenzene) isomers. Selected data mainly from previously unreviewed reports of organochlorines in Pacific seabird eggs are presented in Table 1.

DDT, a broad-spectrum insecticide, was first used in North America in the 1940s in public health campaigns to control lice (Carson 1962). From the 1940s until the early 1970s, large quantities of DDT were sprayed on forests against insect pests in British Columbia (Nigam 1975) and in the northwest United States (Henny 1977). Restrictions on the use of most organochlorine pesticides (i.e., DDT, dieldrin, endrin, heptachlor, HCH, and toxaphene) in Canada and the United States were first implemented in the early 1970s, with further controls imposed throughout the 1970s and 1980s (Noble and Elliott 1986). Heptachlor continued to be used in Oregon until 1974 (Henny et al. 1983) and significant amounts of chlordane, lindane, dicofol, and toxaphene were used until the early 1980s in California (Ohlendorf and Miller 1984). A few minor applications of chlordane, lindane, dieldrin, and heptachlor (e.g., seed treatment, termite control) are still permitted in Canada and the United States. In Mexico, restrictions on the use of DDT, BHC, dieldrin, and heptachlor were imposed in 1980 (Burton and Philogene 1986).

There is less information on organochlorine use in Asian countries bordering the North Pacific. Since the 1950s, DDT and BHC have been used extensively on rice, cotton, and vegetable crops, but in the 1970s, many countries replaced them

Table 1
Organochlorine and PCB residue levels in eggs of North Pacific seabirds

Location	Species[a]	Yr.[b]	Organochlorine residues (mg/kg wet weight) DDE	DDT	DIEL[c]	HE[c]	TOX[c]	OXY[c]	HCH[c]	HCB[c]	PCBs[d]	Ref.[e]
California												
Channel Is.	BRPE	69	72									1
Channel Is.	DCCO	69	32								*3.7	2
Channel Is.	DCCO	69	24								*1.8	2
San Diego Bay	CATE	81	9.3	ND	+	+	+	+			1.7	3
San Diego Bay	ELTE	81	3.8	ND	ND	ND	ND	ND			1.6	3
Farallon Is.	COMU	68	41	ND	0.003						*23	4
Farallon Is.	ASSP	69	43	2.0	0.02						*26	5
San Francisco area	GBHE	70	8.1	+							*3.8	6
San Francisco area	COEG	70	19	+							*3.0	6
San Francisco area	BCNH	82	2.8	+	+	+	+	0.22			6.7	7
San Francisco area	SNEG	82	2.0	+	+	ND	+	0.11			4.3	7
San Francisco area	FOTE	82	2.9	ND	ND		ND	+			5.6	7
San Francisco area	CATE	81	2.6	+	+		+	ND			1.8	7
San Francisco Bay	BCNH	83	7								4.1	8
Oregon												
Southern Oregon	DCCO	79	4	ND	+	ND	ND	+			1.3	9
Southern Oregon	PECO	79	0.8	ND	ND	+	ND	ND			0.68	9
Oregon	LESP	79	2	0.15	ND	ND	0.33	ND			1.3	9
Northern Oregon	FTSP	79	1	0.67	ND	ND	0.67	0.22			5.1	9
Southern Oregon	PIGU	79	0.6	ND	ND	ND	ND	ND			0.33	9
Northern Oregon	TUPU	79	0.2	ND	ND	ND	ND	ND			0.51	9
Southern Oregon	COMU	79	0.1	ND	ND	ND	ND	ND			0.52	9
Oregon	WEGU	79	1	ND	+	ND	ND	ND			0.47	9
Washington												
Puget Sound	DCCO	84	0.6		ND	ND		+			1.8	10
British Columbia												
Strait of Georgia	DCCO	85	0.50	++	0.006	0.004		0.014	0.012	0.30	3.8	11
Strait of Georgia	DCCO	85	0.46	++	0.004	0.003		0.012	0.010	0.014	2.9	11
Strait of Georgia	PECO	85	0.27	++	0.027	0.008		0.018	0.027	0.083	1.5	11
Strait of Georgia	GWGU	77	0.48	++	0.009	0.014		0.001	0.007	0.023	1.78	11
W. Vancouver I.	PECO	85	0.27	++	0.023	0.009		0.014	0.030	0.018	0.61	11
W. Vancouver I.	LESP	85	0.73	++	0.008	0.004		0.015	0.020	0.026	2.74	11
W. Vancouver I.	GWGU	73	0.57		0.014	0.033		0.030	0.012	0.013	0.59	11
Hecate Strait	RHAU	85	0.63	++	0.012	0.009		0.23	0.018	0.034	0.61	11
Queen Charlotte Is.	LESP	83	0.65	++	0.013	0.030		0.018	0.038	0.019	0.79	11
Queen Charlotte Is.	FTSP	83	1.68	++	0.040	0.030		0.015	0.050	0.063	3.9	11
Queen Charlotte Is.	ANMU	86	0.69	++	0.004	0.009		0.014	0.38	0.05	1.1	11
Alaska												
Gulf of Alaska	NOFU	73–76	0.484	0.048	0.005	+	ND	0.033		0.008	0.483	12
Gulf of Alaska	PECO	73–76	0.134	0.005	0.020	0.007	0.081	0.009		0.011	0.206	12
Gulf of Alaska	GWGU	73–76	0.380	+	0.015	0.014	+	0.018		0.028	0.637	12
Gulf of Alaska	MEGU	73–76	0.660	0.005	0.025	0.010	ND	0.033		ND	1.33	12
Gulf of Alaska	BLKI	73–76	0.06	0.001	0.003	0.002	0.003	0.021		0.013	0.182	12
Gulf of Alaska	TBMU	73–76	0.147	ND	0.010	0.007	+ +	0.019		0.060	0.259	12
Gulf of Alaska	COMU	73–76	0.426	0.010	0.013	0.008	+	0.025		0.014	0.609	12
Gulf of Alaska	TUPU	73–76	0.297	0.010	0.013	0.008	+	0.018		0.042	0.508	12
Chukchi Sea	BLKI	73–76	0.033	ND	+	0.002	ND	0.010		0.031	0.365	12
Chukchi Sea	TBMU	73–76	0.166	ND	0.004	0.006	0.003	0.021		0.008	0.307	12
Chukchi Sea	COMU	73–76	0.141	ND	+	0.004	ND	0.023		0.111	0.182	12
Aleutian Is.	LESP	73–76	0.337	0.019	0.002	+	0.050	0.016		0.018	0.508	12
Aleutian Is.	FTSP	73–76	0.602	0.140	0.032	0.001	0.101	0.050		0.0141	1.60	12
Aleutian Is.	GWGU	73–76	3.72	0.030	0.078	0.026	ND	0.304		0.112	1.94	12
Aleutian Is.	TBMU	73–76	0.145	ND	ND	ND	ND	0.001		0.062	0.307	12
Aleutian Is.	COMU	73–76	0.175	ND	0.007	ND	ND	0.005		0.058	0.168	12
Aleutian Is.	ANMU	73–76	1.61	ND	ND	0.004	0.030	0.005		0.002	0.829	12
Aleutian Is.	ALTE	73–76	0.432	ND	0.011	0.004	0.029	0.000		0.005	0.324	12
Bering Sea	NOFU	73–76	0.307	0.015	0.008	0.005	0.067	0.051		0.055	0.387	12
Bering Sea	BLKI	73–76	0.033	ND	+	0.003	0.025	0.034		0.033	0.472	12
Bering Sea	COMU	73–76	0.204	0.013	0.007	ND	0.010			0.000	0.209	12
Bering Sea	TUPU	73–76	0.266	0.002	0.020	0.004	ND	0.012		0.004	0.423	12
Hawaii												13
Maui I.	HAPE	70	0.45									14
Oahu I.	WTSH	80	0.50	0.054	ND	ND	ND	ND	ND	ND	0.10	14
Oahu I.	RFBO	80	0.034	ND	+	ND	ND	ND	ND	ND	ND	14
Oahu I.	SOTE	80	0.015	ND	ND	ND	ND	ND	ND	ND	0.086	14
Midway I.	WTSH	80	0.30	0.027	ND	ND	ND	ND	+	+	0.11	14
Midway I.	RFBO	80	0.021	ND	ND	ND	ND	ND	ND	ND	ND	14
Midway I.	SOTE	80	0.012	ND	ND	ND	ND	ND	ND	ND	ND	14

Note: Blank spaces mean the compound was not reported in the reference. *ND means not detected.* "+" means the compound was detected in less than 50% of the eggs analyzed, "++" means the compound was detected in greater than 50% of the eggs analyzed, but no mean was reported or could be determined.

(Continued)

Table 1 (continued)

[d] **Species letter codes**

BRPE	Brown Pelican	DCCO	Double-crested Cormorant	CATE	Caspian Tern
ELTE	Elegant Tern	COMU	Common Murre	ASSP	Ashy Storm-Petrel
GBHE	Great Blue Heron	COEG	Common Egret	BCNH	Black-crowned Night-Heron
SNEG	Snowy Egret	FOTE	Forster's Tern	PECO	Pelagic Cormorant
LESP	Leach's Storm-Petrel	PIGU	Pigeon Guillemot	TUPU	Tufted Puffin
WEGU	Western Gull	GWGU	Glaucous-winged Gull	RHAU	Rhinoceros Auklet
FTSP	Fork-tailed Storm-Petrel	ANMU	Ancient Murrelet	NOFU	Northern Fulmar
MEGU	Mew Gull	BLKI	Black-legged Kittiwake	TBMU	Thick-billed Murre
ALTE	Aleutian Tern	HAPE	Hawaiian Petrel	WTSH	Wedge-tailed Shearwater
RFBO	Red-footed Booby	SOTE	Sooty Tern		

[b] Yr = year(s) of collection.

[c] DIEL = dieldrin, HE = heptachlor epoxide, TOX = toxaphene, OXY = oxychlordane, HCH = sum of a-HCH and b-HCH isomers, HCB = hexachlorobenzene.

[d] Based on a 1:1 ratio of Aroclors 1254 and 1260 unless preceded by an asterix indicating Aroclor 1260 only.

[e] **References**

1 = Risebrough et al. 1971
2 = Gress et al. 1973
3 = Ohlendorf et al. 1985
4 = Gress et al. 1971
5 = Coulter and Risebrough 1973
6 = Faber et al. 1972
7 = Ohlendorf et al. 1988
8 = Hoffman et al. 1986
9 = Henny et al. 1982
10 = Henny et al. 1989
11 = Elliott et al. 1989b
12 = Ohlendorf et al. 1982
13 = King and Lincer 1973
14 = Ohlendorf and Harrison 1986

with organophosphates. As in North America, agricultural uses of organochlorine pesticides are subject to regulation in most western Pacific countries, but the degree of compliance varies. The People's Republic of China manufactures organochlorine pesticides; however, the production and use of two, DDT and BHC, were banned there in 1983 (Wolfe et al. 1984). In Japan, production and use of DDT and BHC were prohibited in 1971, but the use of chlordane for termite control was continued until the late 1980s (Tanabe et al. 1989). Korea also prohibited the use of DDT in the early 1970s (Phillips and Tanabe 1989). However, in Hong Kong (where many pesticides are formulated) there appears to be continued input of DDT into coastal waters despite restrictions imposed in 1988.

2.4. Polychlorinated biphenyls (PCBs)

It has been estimated that more than 1 billion (10^9) kg of PCBs were produced worldwide. Currently the amount of PCBs in use and stored in landfills, waste dumps, and so on is more than double the amount that has already been lost to the environment (Tanabe 1988). Therefore, if more stringent measures are not taken to destroy stored PCBs or remove them permanently from circulation, significant inputs to the environment will continue for many years.

Since the presence of PCBs in Pacific seabirds was first documented by Risebrough et al. (1968), a number of studies have reported levels of PCBs and other chlorinated hydrocarbons in Pacific marine birds. In recent years, Ohlendorf and Harrison (1986) reported the presence of PCBs in eggs of Red-footed Booby *Sula sula* and Wedge-tailed Shearwater *Puffinus pacificus* but not in those of Sooty Tern *Sterna fuscata* (limit of detection 0.02 mg/kg, wet weight) collected in 1980 from the Hawaiian Islands. Elevated PCB levels have been found in wildlife breeding in San Francisco Bay, including Black-crowned Night-Herons *Nycticorax nycticorax* (Hoffman et al. 1986; Ohlendorf and Marois 1990), Caspian Terns *Sterna caspia* (Ohlendorf et al. 1988a), and Surf Scoters *Melanitta perspicillata* (Ohlendorf et al. 1990). Elliott et al. (1989b) reported PCB levels of 1.11 mg/kg in Ancient Murrelets *Synthliboramphus antiquus* collected in 1986 from the Queen Charlotte Islands, British Columbia. Henny et al. (1990) reported a significant increase in total PCB concentrations in Western Grebes *Aechmophorus occidentalis* wintering in Puget Sound, Washington (1.94 mg/kg in October to 6.37 mg/kg in February).

2.5. Polychlorinated dibenzodioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs)

Neither PCDDs nor PCDFs were deliberately produced commercially. Rather they are produced either as by-products during the synthesis of other chemicals, such as chlorophenolic biocides, or during combustion of chlorine-containing wastes.

Incineration of municipal and industrial wastes is the major global source of dioxins (Hites 1990). PCDDs produced by combustion can be transported long distances and subsequently deposited in soils and lake sediments. Czuczwa et al. (1984) showed that combustion was the main source of PCDDs and PCDFs in Great Lakes sediments. Although combustion produces a fairly uniform mixture of PCDD and PCDF isomers, physical and chemical atmospheric processes favour the deposition and accumulation of less toxic higher chlorinated compounds, which therefore tend to dominate in sediments (Hites 1990). Elevated contamination by more toxic isomers such as 2378-TCDD was until recently always associated with use, production, or waste storage of chlorophenoxy acid herbicides, particularly 245-T (see Baughman and Meselson 1973; Fanelli et al. 1980; Powell 1984). Recent studies have shown that effluents from kraft pulp mills using the chlorine bleaching process are contaminated by 2378-TCDD and 2378-TCDF (Kuehl et al. 1987), which has resulted in contamination of fish and wildlife in receiving waters (Rogers et al. 1989; Elliott et al. 1989a). The use of tetrachlorophenol-contaminated woodchips by kraft mills results in the formation of hexachlorinated dioxins during wood digestion and their presence in effluents (Luthe et al. 1990) and in fish and wildlife in receiving waters (Elliott et al. 1989a). PCDD and PCDF contamination from kraft pulp mills and forest industry chlorophenol use has particular implications for North Pacific marine organisms because of the large number of wood-processing sites along the coast from California to Alaska. This is especially true for the Strait of Georgia (Waldichuk 1988).

There are limited published data on PCDD/PCDF levels in marine birds of the North Pacific. Ivory gull *Pagophila eburnea* eggs collected from Seymour Island in the western Canadian Arctic contained 5 ng/kg 2378-TCDD in 1976 and 8 ng/kg in 1987 (Elliott et al., in press). Leach's Storm-Petrel *Oceanodroma leucorhoa* eggs collected in 1987 from the Storm Islands, located off the northeast coast of Vancouver Island, British Columbia, showed no detectable PCDD or PCDF levels (limit of detection from 2 to 10 ng/kg). Jarman et al. (1989) reported low levels of PCDDs and PCDFs (all measured

isomers <10 ng/kg, wet weight) in breast muscle of Western Gulls *Larus occidentalis* from the Channel Islands off the California coast. By way of comparison, in the North Sea of Holland, Van den Berg et al. (1987) reported elevated PCDD and PCDF levels in livers of Great Cormorants *Phalacrocorax carbo* and other estuarine birds. The mean level of 23478-PnCDF, a compound associated with PCBs, was 1096 ng/kg in a cormorant liver sample. Recently, Cederberg et al. (1991) reported 27 ng/kg 2378-TCDD, 45 ng/kg 12378- PnCDD, 59 ng/kg 123678-HxCDD, and 136 ng/kg PnCDF (on a wet-weight basis, calculated from lipid weight results, assuming 17% lipid in Common Murre *Uria aalge* eggs [Noble and Elliott 1986]) in Common Murre eggs from the Baltic Sea. Chlorinated hydrocarbon levels were much lower in murre eggs from the North Atlantic outside the Baltic Sea.

In the Strait of Georgia, elevated levels of 2378-TCDD and 2378-TCDF have been reported in marine populations of Great Blue Herons *Ardea herodias*. The highest mean 2378-TCDD level, 210 ng/kg (range in 10 eggs, 67–444 ng/kg), was found in 1987, in eggs from a colony known to be foraging on tidal mudflats impacted by effluent from a large kraft pulp mill (Elliott et al. 1988b, 1989a). Elevated levels of penta- and hexa-substituted PCDDs were also found in heron eggs and were attributed to chlorophenol contamination, particularly of woodchips used by pulp mills. The highest recorded mean level of 123678-HxCDD was also found in heron eggs from Crofton, British Columbia, in 1986, 544 ng/kg (range in 10 eggs, 13–1298 ng/kg). A similar pattern of dioxin and furan contamination was found in eggs of Double-crested Cormorants *Phalacrocorax auritus* breeding in Howe Sound, British Columbia, a fjord contiguous with the Strait of Georgia, that receives effluent directly from two bleached-kraft pulp mills (Whitehead et al., in press). Double-crested Cormorant eggs from Mandarte Island in the Strait of Georgia, distant from industrial pollutant sources, were also contaminated with PCDDs and PCDFs, indicating widespread contamination of the marine food chains of the strait during the 1980s (Elliott et al., in press).

Waterfowl wintering near pulp mills on the British Columbia coast were also found to be contaminated with PCDDs and PCDFs (Whitehead et al. 1990, in press). Highest levels were in Western Grebes with 120 ng/kg 2378-TCDD and Red-necked Grebes *Podiceps grisegena* with 910 ng/kg 123678-HxCDD collected near a kraft pulp mill at Port Alberni (Vermeer et al., in press). The elevated PCDD and PCDF levels in waterfowl tissues resulted in a 1990 advisory by the Canadian federal health department recommending limited human consumption of Common Merganser *Mergus merganser* and Surf Scoter taken from the Port Alberni area (Whitehead et al. 1990).

2.6. Other chlorinated compounds

Jarman et al. (1990) reported finding tris-(chlorophenyl) methanol in eggs of seabirds from the west coast of North America and a number of other locations. A pooled sample of 11 Leach's Storm-Petrel eggs collected in 1987 from the Storm Islands, British Columbia, contained a previously unidentified penta-brominated aromatic compound at an estimated concentration of 80 µg/kg (Elliott et al., in press). Poly-brominated biphenyls were believed to be the source of this chemical.

2.7. Geographical patterns

In general, contaminant residues in North Pacific seabirds declined in a gradient from California in the south to Alaska in the north, with high concentrations in scattered hot spots such as industrialized estuaries (Risebrough et al. 1971; Gress et al. 1973; Henny et al. 1982; Ohlendorf et al. 1982, 1985, 1988; Elliott et al. 1989b). Mid-ocean and offshore locations tended to be less contaminated (Ohlendorf and Harrison 1986), although that pattern may be changing over time as inputs from point sources are reduced and contaminants are dispersed by long-range transport (Elliott et al. 1989b). There are few data from the western Pacific, although marine birds with highly contaminated tissues have previously been reported in Japan (Fujiwara 1974).

2.7.1. *California*

In the late 1960s, Brown Pelicans *Pelecanus occidentalis* and Double-crested Cormorants in the Channel Islands in southern California were found to be laying thin-shelled eggs heavily contaminated with DDE (Risebrough et al. 1971; Gress et al. 1973). Cassin's Auklet *Ptychoramphus aleuticus* eggs from nearby colonies contained DDE residues in excess of 10 mg/kg (Risebrough et al. 1967). Near San Francisco, investigations of nesting failures in Great Egrets *Casmerodius albus* and Great Blue Herons revealed thin eggshells, high levels of DDE and dieldrin in unhatched eggs, and dead adults in the colony (Faber et al. 1972). High concentrations of DDT related compounds in California coastal wildlife have been attributed to losses from the pesticide manufacturing industry.

Organochlorine levels in Californian marine birds were not reported again until the early 1980s when Ohlendorf et al. (1985) analyzed eggs of Caspian Terns and Elegant Terns *Sterna elegans* collected in San Diego Bay in 1981 (Table 1). Although DDE concentrations exceeded 30 mg/kg in at least one Caspian Tern egg, both species contained few residues other than DDE, PCBs, and trans-nonachlor. Local sources were suspected as fish brought to Caspian Tern chicks contained up to 3.0 mg/kg DDE. Three Forster's Tern *S. forsteri* eggs contained 3.7 mg/kg DDE and 1.8 mg/kg PCBs.

Ohlendorf et al. (1988) analyzed eggs of Black-crowned Night-Herons, Snowy Egrets *Egretta thula*, and Caspian and Forster's terns collected from the San Francisco area in 1982 (Table 1). DDE residues were highest in Caspian Terns; other pesticides showed no pattern among species. Sixteen percent of eggs of Black-crowned Night-Herons collected at six Californian colonies (including three in San Francisco Bay) between 1982 and 1983 contained DDE residues in excess of 8.0 mg/kg (Ohlendorf and Marois 1990). Some night heron eggs were also high in chlordane metabolites (up to 0.92 mg/kg oxychlordane, 2.0 mg/kg cis-chlordane, and 1.9 mg/kg trans-nonachlor).

2.7.2. *Oregon and Washington*

Henny et al. (1982) reported organochlorine levels in eggs of 11 species of marine birds collected in 1979–80 from the coast of Oregon (Table 1). One egg contained 12 mg/kg DDE; most samples were much less contaminated. Only two species, Common Murres and Western Gulls, exhibited statistically significant although minor (<6%) eggshell thinning. Toxaphene was detected in most (83%) of the storm-petrel eggs, but not in cormorants, alcids, gulls, or shorebirds. Dieldrin and chlordane-related residues were poorly represented, and no geographic trends were apparent.

Eggs of Double-crested Cormorants from northern Washington in 1984 contained low levels of DDE and other organochlorines (Henny et al. 1989). However, Glaucous-winged Gulls *Larus glaucescens* breeding in Puget Sound exhibited minor eggshell thinning, hepatic defects (Calambokidis et al. 1985), and abnormal oviduct development (Fry et al. 1987). Henny et al. (1990) found that Western Grebes overwintering in Puget Sound accumulated significant but not toxic amounts of DDE, PCBs, chlordanes, and HCB.

2.7.3. British Columbia

Eggs and tissues of marine birds were first sampled in British Columbia in 1968. Since that time, eggs of selected seabirds—storm-petrels, cormorants, Rhinoceros Auklets *Cerorhinca monocerata*, and Glaucous-winged Gulls—have been sampled regularly by the Canadian Wildlife Service (CWS). In addition, eggs and tissues of Great Blue Herons, alcids, and sea ducks have been analyzed periodically. Results are reported in Ohlendorf et al. (1978), Elliott et al. (1989b), Whitehead (1989), Noble (1990), and Elliott et al. (in press).

All eggs contained DDE, with the highest levels (12 mg/kg) occurring in Fork-tailed Storm-Petrels *Oceanodroma furcata*. DDE residues in Leach's Storm-Petrels, Double-crested and Pelagic *Phalacrocorax pelagicus* cormorants, Glaucous-winged Gulls, and Rhinoceros Auklets ranged from less than 0.5 mg/kg to 4.1 mg/kg (Noble and Elliott 1986).

Dieldrin, heptachlor epoxide, oxychlordane, and HCB were present in eggs of all seabirds sampled, at levels between 0.01 and 0.1 mg/kg.

Levels of organochlorines in eggs of Pigeon Guillemots *Cepphus columba* collected in 1970, Rhinoceros Auklets collected between 1970 and 1990, and Ancient Murrelets collected between 1968 and 1986 were low (Elliott et al. 1989b, in press). However, residues of b-HCH in excess of 0.5 mg/kg in some Ancient Murrelet eggs from the Queen Charlotte Islands were the highest values reported for any Pacific seabird.

In the late 1960s and early 1970s, tissues of a number of Pacific coast alcids were analyzed for organochlorine content. Levels in bodies of Marbled Murrelet *Brachyramphus marmoratus*, Cassin's Auklet, and Tufted Puffin *Fratercula cirrhata* were low, whereas levels of DDE in bodies of Ancient Murrelets were as high as 5.9 mg/kg.

2.7.4. Alaska

Ohlendorf et al. (1982) reported organochlorine levels in eggs of 19 Alaskan seabird species collected between 1973 and 1976. All eggs contained DDE, 84% contained oxychlordane, and 83% contained HCB. Concentrations were generally low, with the highest mean levels reported in Glaucous-winged Gull eggs from Bogoslof Island in the Aleutian chain (5.2 mg/kg DDE, 0.21 mg/kg dieldrin, 0.25 mg/kg oxychlordane, 0.19 mg/kg HCB, 0.037 mg/kg heptachlor epoxide, 0.075 mg/kg cis-chlordane, and 0.026 mg/kg cis-nonachlor). DDT, DDD, and toxaphene were highest in eggs of Fork-tailed Storm-Petrels at 0.14, 0.16, and 0.10 mg/kg, respectively. Eggs of Arctic Terns *Sterna paradisaea* contained low concentrations of all compounds, and Black-legged Kittiwakes *Rissa tridactyla* were found to have low concentrations of DDE relative to other species, as reported previously by Peakall and Nettleship (1987).

2.7.5. Hawaii

Black-footed Albatross *Diomedea nigripes* and Laysan Albatross *D. immutabilis* collected on Midway Island in the early 1970s revealed low levels of DDT compounds, dieldrin, and PCBs (Fisher 1973). DDE was similarly low in eggs of Hawaiian Petrels *Pterodroma phaeopygia* (King and Lincer 1973).

Eggs of three Hawaiian seabird species collected in 1980 from four colonies were analyzed for organochlorines (Ohlendorf and Harrison 1986). Wedge-tailed Shearwaters contained higher concentrations of DDE (Table 1) than Red-footed Boobies or Sooty Terns, and were the only eggs that contained the parent DDT. PCBs occurred frequently, and dieldrin and mirex rarely in eggs of shearwaters and boobies. No heptachlor epoxide, toxaphene, or chlordane compounds were detected.

Only Red-footed Boobies are resident; therefore, contamination in this species best represents contamination in the local environment. The authors considered that the lower levels of most contaminants in Sooty Terns reflected the smaller proportion of fish in their diet, and that the HCH and HCB in tern eggs may have originated from their wintering area in the western Pacific.

2.7.6. North Pacific and Japan

We could not find data on organochlorines other than PCBs in marine birds from any Asian countries. DDE concentrations in carcasses of Short-tailed Shearwaters *Puffinus tenuirostris* collected in Japan and the northern North Pacific were low, ranging from 0.027 to 0.059 mg/kg.

Subcutaneous fat from three Thick-billed Murres *Uria lomvia* collected in the North Pacific Ocean was analyzed for total chlordane, total DDT, and HCH isomers (Kawano et al. 1986). Concentrations were 0.10, 1.8, and 0.24 mg/kg, respectively, on a lipid-weight basis.

2.7.7. Comparisons with the Atlantic coast and other sites

There have been a number of published comparisons of organochlorine contamination of marine birds from sites in the Atlantic and Pacific oceans (Ohlendorf et al. 1978; Ohlendorf and Fleming 1988; Elliott et al. 1989b). Recent analyses showed that eggs of seabirds foraging in offshore locations in British Columbia had higher concentrations of DDE and HCH, but lower concentrations of PCBs, dieldrin, oxychlordane, and HCB, than eggs of the same or ecologically similar Atlantic species (Elliott et al. 1989a). In contrast, eggs of seabirds from inshore locations on the Pacific coast were generally less contaminated than eggs from inshore areas on the Atlantic coast.

In recent years, PCBs in eggs from Pacific coastal colonies were generally lower than those from either Great Lakes or Atlantic coast colonies and were comparable to those in the prairies (Table 2). Higher PCB levels in cormorant eggs particularly from the Great Lakes-St. Lawrence basin reflect the greater concentration of heavy industry compared to the Pacific coast. Atmospheric transport by prevailing winds and regional deposition of PCBs from the North American mainland, including re-emission from the Great Lakes (Eisenreich et al. 1981; Bidleman et al. 1981), would have a major influence on PCB levels in food chains of the Atlantic coast.

2.8. Temporal trends

We are unaware of any long-term data on contaminants in Pacific seabirds, other than those collected in British Columbia by CWS. Between the late 1960s and late 1980s, only DDT compounds showed significant declines; trends for dieldrin, oxychlordane, heptachlor epoxide, and HCB were inconsistent among species and locations.

Table 2
PCB levels in Double-crested Cormorant eggs from North American locations, 1977–1990 (mg/kg, wet weight)

Location	Year	No. analyzed/ no. collected[a]	PCBs[b]	Reference[c]
British Columbia				
Crofton	1990	1/12	3.30	1
Mandarte I.	1990	1/11	2.37	1
Chain I.	1990	1/9	5.17	1
Fraser Estuary	1990	1/7	1.64	1
Christie I.	1990	1/9	2.01	1
Washington				
Colville I.	1984	36/36	2.19	2
Protection I.	1984	11/11	1.37	2
Oregon				
Hunter I.	1979	10/10	1.32	3
Maheur NWR[d]	1980	6/6	1.27	2
Klamath NWR	1977	5/5	2.59	2
Umatilla NWR	1978/79	8/8	6.76	2
Lake Ontario				
Hamilton Harbour	1989	1/10	20.1	4
Lake Superior				
Cone I.	1989	1/10	9.0	4
Quebec				
Île aux Pommes	1988	5/5	6.0	5
New Brunswick				
Manawagonish I.	1988	5/5	4.95	5
Newfoundland				
Great Barasway I.	1988	5/5	9.70	5

[a] 1/N indicates single analyses of a pool of N eggs; other values are geometric means.
[b] Based on estimate of total PCBs, i.e., 1:1 Aroclor 1254:1260 for Canadian Wildlife Service data.
[c] 1 - Elliott et al. unpubl. data.
2 - Henny et al. 1989
3 - Henny et al. 1982
4 - Bishop et al. 1992
5 - Canadian Wildlife Service, unpubl. data.
[d] NWR means National Wildlife Refuge.

Levels of organochlorines in Leach's Storm-Petrels reflect surficial contamination of the offshore environment. Between 1970 and 1990, only DDE concentrations in those eggs declined significantly (Elliott et al., in press). In eggs of Rhinoceros Auklets collected in 1970, 1985, and 1990, only DDE declined initially. HCB and oxychlordane, but not dieldrin or b-HCH, declined between 1985 and 1990. A similar pattern was found in eggs of inshore-feeding Double-crested Cormorants. DDE and dieldrin declined between 1977 and 1988 in eggs of Great Blue Herons (Whitehead 1989) and in Pelagic Cormorants between 1973 and 1984 (Elliott et al. 1989b).

Total PCBs (as 1:1 Aroclor 1254:1260) in Great Blue Heron eggs decreased significantly from 17.5 mg/kg in 1977 to 2.89 mg/kg in 1987 at a colony in Vancouver (Whitehead 1989). Between 1973 and 1985, PCBs decreased significantly in both Pelagic and Double-crested cormorant eggs from Mandarte Island, British Columbia (Elliott et al. 1989b, in press). Between 1970 and 1990, PCB concentrations in Rhinoceros Auklet eggs from Lucy Island, British Columbia, also declined from 2.01 to 0.33 mg/kg. In contrast to the declining PCB levels reported from coastal and inshore species discussed above, levels of PCBs in eggs of the Leach's Storm-Petrel collected from the west coast of Vancouver Island did not change between 1970 (1.09 mg/kg) and 1990 (1.05 mg/kg).

Recent results show substantial declines in all PCDD and PCDF levels in both herons and cormorant eggs from colonies in the Strait of Georgia, subsequent to efforts by the forest industry to reduce sources of dioxin pollution (Whitehead et al. 1992, in press).

3. Biological effects on marine birds

3.1. Organochlorine pesticides

Incidences of seabird mortality due to organochlorine contamination were reviewed by Ohlendorf et al. (1978) and Noble (1990); relatively few instances of mortality in North Pacific seabirds were attributed to organochlorine poisoning. However, most mortality goes undetected or uninvestigated. Scott et al. (1975) attributed the mass mortality of Common Murres off the coast of Oregon in 1969 to DDE, and dieldrin was implicated in the deaths of Common Egrets in California in 1969, where brain levels exceeded 7.0 mg/kg (Faber et al. 1972). No seabird deaths investigated by CWS in western Canada could be conclusively linked to organochlorine pesticides (Noble and Elliott 1986).

Exposure to organochlorines may affect reproduction; the negative relationship between DDE residues and eggshell thickness has been well established in a number of birds, including marine species (Blus et al. 1974, 1980; Custer et al. 1983; Elliott et al. 1988a). Pelecaniformes are relatively sensitive to the effects of DDE on eggshell thickness and subsequent hatching failure (Keith and Gruchy 1972; Cooke 1973). Pacific populations of Brown Pelicans and Double-crested Cormorants were severely affected in the past (Risebrough et al. 1971; Gress et al. 1973). Since the 1980s residues in Double-crested Cormorant eggs from Washington and British Columbia have been low, and populations flourishing (Elliott et al. 1989b; Henny et al. 1989). Morrison (1979) found only minor eggshell thinning in North Pacific tropicbirds and boobies during the 1950s and 1960s.

Few organochlorine-related effects have been established in procellariids, which forage in offshore locations. Organochlorines have been linked to shell thinning in Ashy Storm-Petrels *Oceanodroma homochroa* on the Farallon Islands (Coulter and Risebrough 1973). Based on data from eastern Canada, Pearce et al. (1979) calculated a critical value of 12 mg/kg DDE to cause significant eggshell thinning in Leach's Storm-Petrels. Residues of that magnitude were not detected in any Pacific populations of Leach's Storm Petrels, but were found in some Fork-tailed Storm Petrel eggs (Henny et al. 1982; Elliott et al. 1989b).

Organochlorine-related effects have been reported regularly in Ardeidae. Faber et al. (1972) reported 15% eggshell thinning and a high rate of reproductive failure in Common Egrets in northern California, whereas the productivity of sympatric Great Blue Herons appeared to be normal. A negative correlation between log DDE concentration and eggshell thickness has been shown for Great Blue Herons from Oregon and Washington (Blus et al. 1980). Pesticide residues in eggs of this species in British Columbia were not considered to be an important factor in observed reproductive problems (Whitehead 1989). Custer et al. (1983) calculated a critical value of 8.0 mg/kg DDE to cause significant eggshell thinning in Black-crowned Night-Herons. Ohlendorf and Marois (1990) found that a number of night-heron eggs from California exceeded that value.

Among charadriiformes, terns appear to be more susceptible to the eggshell-thinning effects of DDE than gulls or alcids (Keith and Gruchy 1972; Fox 1976). More than 25% of eggs laid by Caspian Terns breeding in San Diego Bay in 1981 failed to hatch, or died during pipping (Ohlendorf et al.

Table 3
Populations of North Pacific marine birds reported to have experienced reproductive problems or sublethal effects other than eggshell thinning

Species	Location	Years	Effect	Contaminant	SR[a]	Reference
Double-crested Cormorant	California	1969–72	Reproductive failures, severe EST[b]	DDE 32 ppm	Y	Risebrough et al. 1971
Double-crested Cormorant	British Columbia	1989	Bill deformities	Dioxins, PCBs	N	Elliott et al., unpubl. data
			EROD[c] induction	PCBs	Y	J.T. Sanderson et al., unpubl. data
Brown Pelican	California	1969–72	Reproductive failures, severe EST	DDE 71 ppm	Y	Gress et al. 1973
Western Gull	California	1972	Reproductive failures, 20% EST	Not measured		Hunt and Hunt 1973
Glaucous-winged Gull	Washington	1984	Hepatic atrophy, minor EST	Not measured		Calambokidis et al. 1985
Caspian Tern	California	1981	25% hatching failure	DDE 9.3 ppm	N	Ohlendorf et al. 1985
Great Egret	California	1967–70	Reproductive failures, 15% EST	DDE about 25 ppm	Y	Faber et al. 1972
Great Blue Heron	British Columbia	1987–88	Reproductive failures	Dioxins	N	Elliott et al. 1989b
Great Blue Heron	British Columbia	1988	EROD induction	TCDD	Y	Bellward et al. 1990
			Edema, reduced embryonic weight	TCDD	Y	Hart et al. 1991
Black-crowned Night-Heron	California	1983	Reduced embryonic weight	PCBs	Y	Hoffman et al. 1986

[a] SR = Y, if there was a significant relationship between contaminant residues and effect; otherwise, N.
[b] EST = eggshell thinning.
[c] EROD = ethoxyresorufin-O-deethylase.

1985). Although DDE residues in eggs averaged 9.3 mg/kg and were inversely associated with eggshell thickness, residues were not significantly related to hatching success.

Population declines associated with eggshell thinning and DDE residues have been reported in Common Murres (Gress et al. 1971). However, organochlorines do not appear to have been an important factor in reproductive failures or population declines in some British Columbia alcids (Elliott et al. 1989b; but see Nelson and Myers 1976). Cassin's Auklets in the Farallon Islands appeared to be unaffected by DDE residues in excess of 10 mg/kg (Risebrough et al. 1967).

Investigations of pollutant effects on Glaucous-winged Gulls in Puget Sound, Washington, revealed evidence of 10% eggshell thinning, liver enlargement, and hepatic lesions, but no effects on hatching, fledging, or growth rates (Calambokidis et al. 1985). Hunt and Hunt (1973) reported reproductive failure and 20% eggshell thinning in Western Gulls from southern California.

Table 3 lists studies where bioeffects other than eggshell thinning were related to organochlorine residues in eggs or tissues of marine birds from the North Pacific. Comparatively few studies report sublethal effects of compounds other than DDE. In the Atlantic, dieldrin has been implicated in the hatching failure of Northern Gannets *Sula bassana* (Chapdelaine et al. 1987; Elliott et al. 1988a), shags *Phalacrocorax aristotelis* (Potts 1968), and Brown Pelicans (Blus et al. 1974). However, concentrations of organochlorines tend to be correlated with each other (Ohlendorf et al. 1978), often making it difficult to distinguish among their effects.

Ohlendorf and Fleming (1988) compared published levels of environmental contaminants in eggs of aquatic birds collected in San Francisco Bay with levels associated with adverse effects. Based on these criteria, hazardous residues of DDE occurred in Black-crowned Night-Herons, Snowy Egrets, and Caspian and Forster's terns. None of the levels found in seabird eggs collected since 1980 in Hawaii (Ohlendorf and Harrison 1986) or British Columbia (Noble and Elliott 1986; Elliott et al. 1989b) were high enough to cause significant effects on reproduction.

Apart from DDE, the toxicological significance of most organochlorines to marine birds has not been established. This is important because, although DDT compounds are generally decreasing, other compounds such as HCH and toxaphene may be increasing (Elliott et al., in press). Although some Pacific seabird populations may be in decline, other factors, including oil (Burger and Fry, this volume), heavy metals (Ohlendorf, this volume), deforestation, and introduced predators (Vermeer and Sealy 1984), may be involved.

3.2. Polychlorinated biphenyls

PCB mixtures are not acutely toxic to birds when compared to many other environmental chemicals (Hill and Camardese 1986). PCBs may have poisoned a number of Great Cormorants in the Netherlands (Koeman et al. 1973). The role of PCBs and other environmental contaminants in seabird "wrecks" or mass mortality incidences is not clear; most wrecks are associated with bad weather impairment of foraging efficiency, resulting in mass starvation. The subsequent mobilization of lipid reserves results in increased concentrations of lipophilic contaminants in the bloodstream and eventually the brain. Elevated PCB levels were reported in a die-off of Common Murres off the Oregon coast in 1969 (Scott et al. 1975); yet in another die-off of 100 000 murres in Alaska, contaminant levels in starved birds were low (Bailey and Davenport 1972).

Chronic effects of PCBs are common to many species and include weight loss, immunotoxicity, liver enlargement and other signs of hepatotoxicity such as porphyria, induction of drug-metabolizing enzymes, and reproductive toxicity (Kimbrough 1974; Safe 1984).

A field study of Black-crowned Night-Herons in San Francisco Bay showed an association between total PCBs and lower embryonic weight (Hoffman et al. 1986). Ohlendorf and Fleming (1988) determined that PCB levels in Snowy Egrets, Black-crowned Night-Herons, Caspian Terns, and Forster's Terns were high enough to adversely affect avian health or reproduction.

The toxic symptoms caused by PCBs are also common to the PCDDs, the PCDFs, and the polybrominated biphenyls. This group of compounds is often referred to collectively as polyhalogenated aromatic hydrocarbons (PHAHs). The toxicity of individual PHAH compounds varies greatly with the molecular structure. The most toxic compound is 2378-TCDD which is often used as a model for studying the effects of these chemicals. The most toxic furan and biphenyl congeners all exhibit a similar structure to 2378-TCDD. Many of the toxic effects caused by this class of compound are believed to be

mediated by a cytosolic receptor found in many tissues, known as the *Ah* receptor (Landers and Bunce 1991).

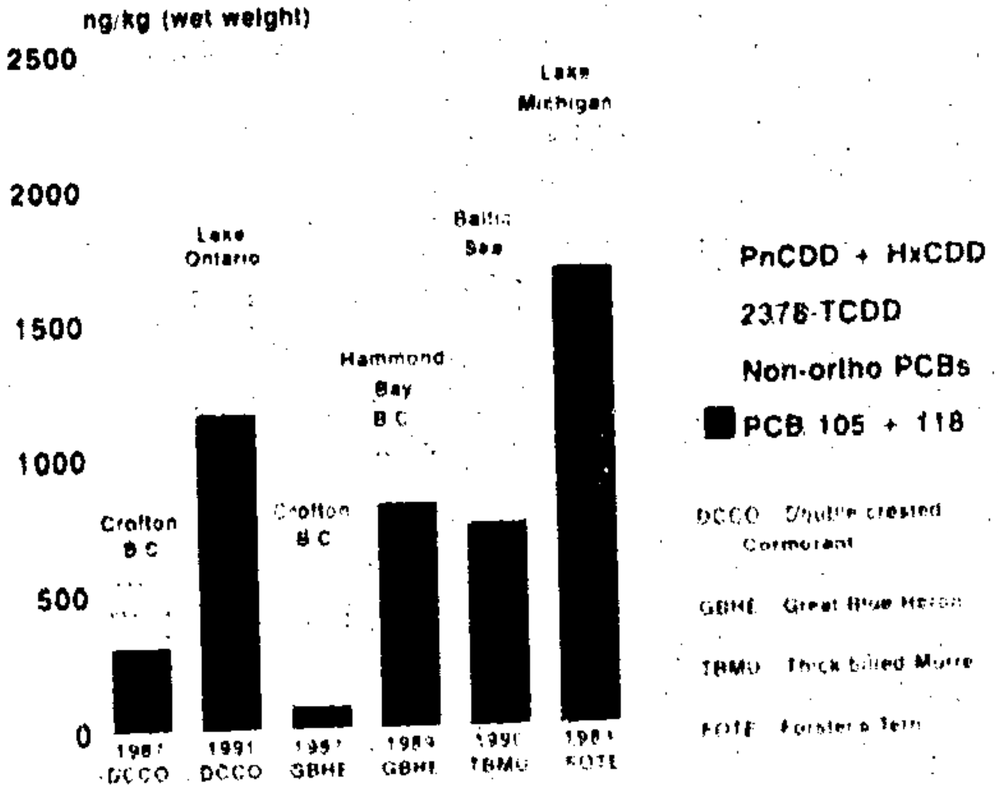
Study of PHAH structure–activity relationships has produced a scheme for determining the toxicity of complex mixtures of these chemicals through use of TCDD Toxic Equivalence Factors (TEFs). Each individual compound is assigned a TEF between 0 and 1 relative to 2378-TCDD. Analytically determined concentrations are multiplied by the TEF, the results summed to produce the TEQs. Safe (1990) recently published a set of TEFs for the most important PHAH compounds. Kubiak et al. (1989) determined TCDD toxic equivalences in a Forster's Tern population in Green Bay, Michigan, that was exhibiting reduced breeding success. They attributed most of the toxicity to two PCB congeners, 3,3′,4,4′,5-pentachlorobiphenyl (PCB No. 126) and 2,3,3′,4,4′-pentachlorobiphenyl (PCB No. 105). In studies of North Pacific marine mammals, Tanabe et al. (1987) reported that the highly toxic non-ortho substituted PCBs were of greater toxicological concern than dioxins and furans.

Figure 1 compares TCDD TEQ concentrations in eggs of fish-eating birds from different sites. The various PCB congeners contribute >90% of the TEQ load in birds from locations other than British Columbia. In eggs of Great Blue Herons collected in 1987 from Crofton, British Columbia, the PCBs contributed less than 20% of the total TEQs. This appears, however, to have been an unusual situation. Herons at Crofton feed year-round in a shallow bay dominated by a large kraft pulp and paper mill with virtually no other current or historical industrial discharges. In contrast, eggs of cormorants collected in 1989 from the Crofton area had PCB levels that made up more than 60% of the total TEQs. Although resident on the British Columbia coast, cormorants do not remain year-round in the Crofton area (Elliott, unpubl. data). In Great Blue Heron eggs collected from Hammond Bay, approximately 100 km north of Crofton, where there is apparently greater PCB contamination of the food chain, PCBs accounted for greater than 80% of the TEQs.

3.3. Polychlorinated dibenzodioxins and polychlorinated dibenzofurans

The environmental hazard of dioxins, including toxicity to wildlife, has been reviewed by Eisler (1986) and Ontario Ministry of the Environment (1985). Acute avian toxicity of 2378-TCDD ranges from 15 µg/kg body weight in Northern Bobwhite *Colinus virginianus* to greater than 810 µg/kg body weight in the Ringed Turtle-Dove *Streptopelia risoria* (Hudson et al. 1984). In chicken embryos, mortality, teratogenic effects, and edema occur in eggs injected with as little as 10 ng/kg 2378-TCDD (Verrett 1970), while 6.4 ng/kg caused a 21% increase in the incidence of cardiovascular malformations (Cheung et al. 1981). The LD_{50} for 2378-TCDD in the chicken embryo was reported as 250 ng/kg egg when administered into the air sac (Allred and Strange 1977). In domestic turkey embryos, non-ortho PCB congeners that bind the *Ah* receptor and thus act by a similar toxic mechanism to 2378-TCDD also cause gross deformities (Brunstrom and Lund 1988). However, in embryos of other avian species, such as Ring-necked Pheasants *Phasianus colchicus* and Eastern Bluebirds *Sialia sialis*, sublethal effects including gross deformities, edema, liver lesions, and inhibition of lymph and thymus development were not observed; rather mortality was the most sensitive endpoint (Martin et al. 1989; Nosek et al. 1992). The LD_{50} for 2378-TCDD was 1100 ng/kg egg in the pheasant embryo and between 1000 and 10 000 ng/kg egg in the Eastern Bluebird

Figure 1
Total TCDD Toxic Equivalence Factors (TEQs) in eggs of fish eating birds from a number of locations. PCB No. 105 (2,3,3′,4,4′-PCB) and No. 118 (2,3,4,4′,5-PCB) are two of the more important mono-ortho PCB congeners in environmental samples. The non-ortho PCBs were: No. 77 (3,3′,4,4′ TCB), No. 126 (3,3′,4,4′,5-PCB), and No. 169 (3,3′,4,4′,5,5′ HCB). PnCDD and HxCDD include 2,3,7,8-substituted penta- and hexa-substituted dioxins.



embryo, in both cases via albumin injection (Martin et al. 1989; Nosek et al. 1992).

There are few published studies of chronic effects of dioxin on birds. Kenaga and Norris (1983) reported that a diet containing 0.3 or 3 ng/kg TCDD in a formulation of 245 T fed to bobwhites for 18 weeks produced no effects on egg production or survival of embryos. However, 50% mortality did occur within five days at a dietary level of 167 ng/kg. Nosek et al. (1992) reported that Ring-necked Pheasants dosed with 1.0 µg/kg/week of 2378-TCDD for 10 weeks exhibited mortality and signs of wasting syndrome; egg production was also reduced, and fewer than 2% of eggs hatched.

Field studies of dioxin effects on wildlife are limited. Great Lakes Herring Gull eggs collected from Lake Ontario in the early 1970s, and subsequently reanalyzed, contained greater than 2000 ng/kg 2378-TCDD (Elliott et al. 1988c). Although Herring Gull productivity at that time was poor, eggs also contained high levels of other known embryotoxins, including PCBs and HCB (Mineau et al. 1984; Bishop et al. 1992). Although complete failure of a Great Blue Heron colony in 1987 at Crofton, British Columbia, coincided with a threefold increase in mean levels of 2378-TCDD in eggs, no statistically significant relationship between contaminant levels and reproductive outcome was determined for herons (Elliott et al. 1989a). Heron embryos, collected in 1988 at colonies with high, intermediate, and low levels of PCDD and PCDF contamination and incubated artificially, did not exhibit any significant differences in hatching success among the three sites. There were, however, a number of sublethal effects, correlated with 2378-TCDD levels, including induction of hepatic EROD (ethoxyresorufin-O-deethylase) activity, edema, and lower embryonic weight (Bellward et al. 1990; Hart et al. 1991). Between 1987 and 1989, in all three instances where mean levels of 2378-TCDD in heron eggs exceeded 150 ng/kg, the colony failed or had significantly reduced productivity (Whitehead et al. 1991). Although disturbance by people and/or Bald Eagles *Haliaeetus leucocephalus* appears to be the main

cause of heron colony failure on the British Columbia coast (Norman et al. 1989), intensive observation of heron nests showed that mean time spent incubating was lower and there was greater between-nest variability in incubation time at a contaminated versus a control heron colony in 1988 (Moul 1990). The possibility exists, therefore, of a contaminant-related effect on adult incubation behaviour. Chemically mediated aberrant parental behaviour has been reported for a number of species in both laboratory (Peakall and Peakall 1973; McArthur et al. 1983) and field (Cooke et al. 1976; Mineau et al. 1984; Kubiak et al. 1989) studies.

4. Outlook

Since the impact of persistent contaminants, particularly DDE and PCBs, on marine birds was first detected and publicized, the situation has improved dramatically in former hot spot areas such as southern California. However, the later finding of high levels of PCDDs and PCDFs in marine birds in British Columbia attests to the need for ongoing monitoring programs. These findings demonstrate clearly the value of marine birds as indicators of both levels and effects of persistent pollutants. Future investigations of the impact of chlorinated hydrocarbons (and other pollutants) on seabirds should be improved by the development of more sensitive analytical methodologies. Ongoing development and increased use of biomarkers, such as induction of cytochrome P-450 enzymes, should improve early detection and quantification of sublethal effects of chemical stressors on physiological functions.

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Literature cited

Allred, P.M.; Strange, J.R. 1977. The effects of 2,4,5-trichlorophenoxyacetic acid and 2,3,7,8-tetrachlorodibenzo-p-dioxin on developing chicken embryos. Arch. Environ. Contam. Toxicol. 5:483–489.

Bailey, E.P.; Davenport, G.H. 1972. Die-off of Common Murres on the Alaska peninsula and Unimak Island. Condor 74:213–219.

Baughman, R.; Meselson, M. 1973. An analytical method for detecting TCDD (dioxin): levels of TCDD in samples from Vietnam. Environ. Health Perspect., 5:27–35.

Bellward, G.D.; Norstrom, R.J.; Whitehead, P.E.; Elliott, J.E.; Bandiera, S.M.; Dworschak, C.; Chang, T.; Forbes, S.; Cadario, B.; Hart, L.E.; Cheng, K.M. 1990. Comparison of polychlorinated dibenzodioxin levels with hepatic mixed-function oxidase induction in Great Blue Herons. J. Toxicol. Environ. Health 30:33–52.

Bidleman, T.F.; Christensen, E.J.; Billings, W.N.; Leonard, R. 1981. Atmospheric transport of organochlorines in the North Atlantic gyre. J. Mar. Res. 39:443–464.

Bishop, C.A.; Weseloh, D.V.; Burgess, N.M.; Struger, J.; Norstrom, R.J.; Turle, R.; Logan, K.A. 1992. An atlas of contaminants in fish-eating colonial birds of the Great Lakes (1970–1988). Vol. 1. Can. Wildl. Serv. Tech. Rep. Ser. 152. 400 pp.

Blus, L.J. 1982. Further interpretation of the relation of organochlorine residues in Brown Pelican eggs to reproductive success. Environ. Pollut. 28:15–33.

Blus, L.J.; Neely, B.S. Jr.; Belisle, A.A.; Prouty, R.M. 1974. Organochlorine residues in Brown Pelican eggs: relation to reproductive success. Environ. Pollut. 7:81–91.

Blus, L.J.; Henny, C.J.; Kaiser, T.E. 1980. Pollution ecology of breeding Great Blue Herons in the Columbia Basin, Oregon and Washington. Murrelet 61:63–71.

Bourne, W.R.P. 1976. Seabirds and pollution. Pages 403–502 *in* Johnson, R. (ed.), Marine pollution. Academic Press, London.

Brunstrom, B.; Lund, J. 1988. Differences between chick and turkey embryos in sensitivity to 3,3′,4,4′-tetrachlorobiphenyl and in concentration/affinity of the hepatic receptor for 2,3,7,8-tetrachlorodibenzo-p-dioxin. Comp. Biochem. Physiol. 91C:507–512.

Burton, D.K.; Philogene, B.J.R. 1986. An overview of pesticide usage in Latin America. Unpublished report. Canadian Wildlife Service, Ottawa. 122 pp.

Calambokidis, J.; Speich, S.M.; Peard, J.; Steiger, G.H.; Cubbage, J.C.; Fry, D.M.; Lowenstine, L.J. 1985. Biology of Puget Sound marine mammals and marine birds: population health and evidence of pollution effects. NOAA Tech. Memo., NOS OMA 18. 159 pp.

Carson, R. 1962. Silent spring. Houghton Mifflin Co., Boston, MA.

Cederberg, T.; Storr-hansen, E.; Cleeman, M.; Dyck, J. 1991. Organochlorine pollutants in guillemot eggs from the Baltic sea and northern Atlantic—polychlorinated dibenzo-p-dioxins, dibenzofurans, biphenyls and pesticides. DIOXIN '91, 23–27 Sept. 1991, Chapel Hill, NC.

Chang, L.W.; Atlas, E.; Glam, C. 1985. Chromatographic separation and analysis of chlorinated hydrocarbons and phthalic acid esters from ambient air samples. Int. J. Environ. Anal. Chem. 19:145–153.

Chapdelaine, G.; Laporte, P.; Nettleship, D.N. 1987. Population, productivity, and DDT contamination trends of Northern Gannets (*Sula bassanus*) at Bonaventure Island, Quebec, 1967–1984. Can. J. Zool. 65:2922–2926.

Cheung, M.; Gilbert, E.; Peterson, R.E. 1981. Cardiovascular teratogenicity of 2,3,7,8-tetrachlorodibenzo-p-dioxin in the chick embryo. Toxicol. Appl. Pharmacol. 61:197–204.

Cooke, A.S. 1973. Shell thinning in avian eggs by environmental pollutants. Environ. Pollut. 4:85–152.

Cooke, A.S.; Bell, A.A.; Prestt, I. 1976. Egg shell characteristics and incidence of shell breakage for Grey Herons (*Ardea cinerea*) exposed to environmental pollutants. Environ. Pollut. 11:59–84.

Coulter, M.C.; Risebrough, R.W. 1973. Shell thinning in eggs of Ashy Storm-Petrel (*Oceanodroma homochroa*) from the Farallon Islands. Condor 75:254–255.

Custer, T.W.; Hensler, G.L.; Kaiser, T.E. 1983. Clutch size, reproductive success, and organochlorine contaminants in Atlantic coast Black-crowned Night-Herons. Auk 100:699–710.

Czuczawa, J.M.; McVeety, B.D.; Hites, R.A. 1984. Polychlorinated dibenzo-p-dioxins and dibenzofurans in sediments from Siskiwit Lake, Isle Royale. Science 226:568–569.

Dyck, J.; Kraul, I. 1984. Environmental pollutants and shell thinning in eggs of the guillemot *Uria aalge* from the Baltic sea and the Faeroes, and a possible relation between shell thickness and sea water salinity. Dan. Ornithol. Foren. Tidsskr. 78:1–14.

Eisenreich, S.J.; Looney, B.B.; Thornton, J.D. 1981. Airborne organic contaminants in the Great Lakes ecosystem. Environ. Sci. Technol. 15:30–38.

Elsler, R. 1986. Dioxin hazards to fish, wildlife and invertebrates: a synoptic review. U.S. Fish Wildl. Serv. Biol. Rep. 85 (1.8). 37 pp.

Elliott, J.E.; Norstrom, R.J.; Keith, J.A. 1988a. Organochlorines and eggshell thinning in Northern Gannets (*Sula bassanus*) from eastern Canada, 1968–198 Environ. Pollut. 52:81–102.

Elliott, J.E.; Butler, R.W.; Norstrom, J.E.; Whitehead, P.E. 1988b. Levels of polychlorinated dibenzodioxins and polychlorinated dibenzofurans in eggs of Great Blue Herons (*Ardea herodias*) in British Columbia, 1983–87: possible impacts on reproductive success. Can. Wildl. Serv. Prog. Note 176. 7 pp.

Elliott, J.E.; Norstrom, R.J.; Kennedy, S.W.; Fox, G.A. 1988c. Trends and effects of environmental contaminants determined from analysis of archived wildlife samples. Pages 131–142 *in* Wise, S.A.; Zeisler, R.; Goldstein, G.M. (eds.). Progress in environmental specimen banking. U.S. Natl. Bur. Stand. Spec. Publ. 740.

Elliott, J.E.; Butler, R.W.; Norstrom, J.R.; Whitehead, P.E. 1989a. Environmental contaminants and reproductive success of Great Blue Herons, *Ardea herodias*, in British Columbia, 1986–87. Environ. Pollut. 59:91–114.

Elliott, J.E.; Noble, D.G.; Norstrom, R.J.; Whitehead, P.E. 1989b. Organochlorine contaminants in seabird eggs from the Pacific coast of Canada, 1971–1986. Environ. Monit. Assess. 12:67–82.

Elliott, J.E.; Noble, D.G.; Norstrom, R.J.; Whitehead, P.E.; Simon, M.; Pearce, P.A.; Peakall, D.B. In press. Patterns and trends of organic contaminants in Canadian seabird eggs, 1968–1990. *In* Walker, C.R., Livingstone, D. (eds.), Persistent pollutants in marine ecosystems. Pergamon. Oxford, U.K.

Faber, R.A.; Risebrough, R.W.; Pratt, H.M. 1972. Organochlorines and mercury in Common Egrets and Great Blue Herons. Environ. Pollut. 3:111–122.

Fanelli, R.; Castelli, M.G.; Martelli, G.P.; Noseda, A.; Garattini, S. 1980. Presence of 2,3,7,8-tetrachlorodibenzo-p-dioxin in wildlife living near Seveso, Italy: a preliminary study. Bull. Environ. Contam. Toxicol. 24:460–462.

Fisher, H.I. 1973. Pollutents in North Pacific albatrosses. Pac. Sci. 27:220–225.

Fox, G.A. 1976. Eggshell quality: its ecological and physiological significance in a DDT-contaminated common tern population. Wilson Bull. 88:459–477.

Fry, D.M.; Toone, C.K.; Speich, S.M.; Peard, R.J. 1987. Sex ratio skew and breeding patterns of gulls: demographic and toxicological considerations. Stud. Avian Biol. 10:26–43.

Fujiwara, K. 1974. Environmental and food contamination with PCBs in Japan. Department of Hygienic Chemistry, Kyoto City Institute of Public Hygiene. 45 pp.

Gress, F.; Risebrough, R.W.; Sibley, F.C. 1971. Shell thinning in eggs of the Common Murre, *Uria aalge*, from the Farallon Islands, California. Condor 73:368–369.

Gress, F.; Risebrough, R.W.; Anderson, D.W.; Kiff, L.F.; Jehl, J.R. 1973. Reproductive failures of double-crested cormorants in southern California and Baja California. Wilson Bull. 85:197–208.

Hart, L.E.; Cheng, K.M.; Whitehead, P.E.; Shah, R.M.; Lewis, R.J.; Ruschkowski, S.R.; Blair, R.W.; Bennett, D.C.; Bandiera, S.M.; Norstrom, R.J.; Bellward, G.D. 1991. Dioxin contamination and growth and development in Great Blue Heron embryos. J. Toxicol. Environ. Health 32:331–344.

Henny, C.J. 1977. Birds of prey, DDT, and tussock moths in the Pacific Northwest. Trans. North Am. Wildl. Nat. Resour. Conf. 42:397–411.

Henny, C.J.; Blus, L.J.; Prouty, R.M. 1982. Organochlorine residues and shell thinning in Oregon seabird eggs. Murrelet 63:15–21.

Henny, C.J.; Blus, L.J.; Stafford, C.J. 1983. Effects of heptachlor on American Kestrels in the Columbia Basin, Oregon. J. Wildl. Manage. 47:1080–1087.

Henny, C.J.; Blus, L.J.; Thompson, S.P.; Wilson, U.W. 1989. Environmental contaminants, human disturbance and nesting of Double-crested Cormorants in northwestern Washington. Colon. Waterbirds 12:198–206.

Henny, C.J.; Blus, L.J.; Grove, R.A. 1990. Western Grebe, *Aechmophorus occidentalis*, wintering biology and contaminant accumulation in Commencement Bay, Puget Sound, Washington. Can. Field-Nat. 104:460–472.

Hill, E.F.; Camardese, M.B. 1986. Lethal dietary toxicities of environmental contaminants and pesticides to *Coturnix*. U.S. Fish Wildl. Serv. Tech. Rep. 2. 28 pp.

Hites, R.A. 1990. Environmental behaviour of chlorinated dioxins and furans. Acc. Chem. Res. 23:194–201.

Hoffman, D.J.; Rattner, B.A.; Bunck, C.M.; Krynitsky, A.; Ohlendorf, H.M.; Lowe, R.W. 1986. Association between PCBs and lower embryonic weight in Black-crowned Night-Herons in San Francisco Bay. J. Toxicol. Environ. Health 19:383–391.

Hudson, R.H.; Tucker, R.K.; Haigele, M.A. 1984. Handbook of toxicity of pesticides to wildlife. U.S. Fish Wildl. Serv. Resour. Publ. 153. 90 pp.

Hunt G.L., Jr.; Hunt, M.W. 1973. Clutch size, hatching success and eggshell thinning in Western Gulls. Condor 75:483–486.

Jarman, W.M.; Chang, R.R.; Esperanza, C.C.; Stephans, R.D.; Lithicum, J. 1989. PCDDs and PCDFs levels in California Peregrine Falcons, eggs, and selected prey species. DIOXIN '89, 17–22 Sept. 1989, Toronto.

Jarman, W.M.; Burns, S.A.; Bacon, C.A.; Risebrough, R.W. 1990. Levels of Tris(chlorophenyl)methanol in birds and mammals from North America, the Arctic, the Antarctic and Australia. SETAC '90, 11–15 Nov. 1990. Arlington VA.

Kawano, M.; Matsushita, S.; Inoue, T.; Tanaka, H.; Tatsukawa, R. 1986. Biological accumulation of chlordane compounds in marine organisms from the northern North Pacific and Bering Sea. Mar. Pollut. Bull. [illegible]

Keith, J.A.; Gruchy, I.M. 1972. [illegible] levels of chemical pollutants in North American birdlife. Proc. Int. Ornithol. Congr. 15:[illegible]

Kenaga, E.E.; Norris, L.A. 1983. Environmental toxicity of TCDD. Pages 277–[illegible] *in* Tucker, R.E.; Young, A.L.; Gray, A.P. (eds.), Human and environmental risks of chlorinated dioxins and related compounds. [illegible] New York.

Kimbrough, R.D. 1974. The toxicity of polychlorinated polycyclic compounds and related compounds. CRC Crit. Rev. Toxicol. 2:[illegible]

King, W.B.; Lincer, J.L. 1973. DDE residues in the endangered Hawaiian Dark-rumped Petrel [illegible]. Condor 75:[illegible]

Koeman, J.H.; Van Velzen-Blad, H.C.; de Vries, R.; Vos, J.G. 1973. Effects of PCB and DDE [illegible] experimental study. J. Reprod. Fert. (Suppl.) [illegible]

Kubiak, T.J.; Harris, H.J.; Smith, L.M.; Schwarz, T.R.; Stalling, D.L.; Trick, J.A.; Sileo, L.; Docherty, D.E.; Erdman, T.C. 1989. Microcontaminants and reproductive impairment of the Forster's Tern on Green Bay, Lake Michigan – 1983. Arch. Environ. Contam. Toxicol. [illegible]

Kuehl, D.W.; Cook, P.M.; Batterman, A.R.; Lothenbach, D.; Butterworth, B.C. 1987. [illegible] dibenzofurans from contaminated Wisconsin river sediment [illegible] Chemosphere 16:[illegible]

Landers, J.P.; Bunce, N.J. 1991. The Ah receptor and the mechanism of dioxin toxicity. Biochem. J. 276:[illegible]

Luthe, C.E.; Berry, R.M.; Voss, R.H.; Fleming, B.I. 1990. Dioxins and furans in pulp mill questions – research answers. Pulp and Paper Research Institute of Canada. Misc. Rep. [illegible] pp.

Martin, S.; Duncan, L.; Thiel, D.; Peterson, R.E.; Lemke, M. 1989. Evaluation of the effects of dioxin-contaminated sludges on Eastern Bluebird and tree swallows. Nekoosa Papers Inc., Port Edwards, WI.

McArthur, M.L.B.; Fox, G.A.; Peakall, D.B.; Philogene, B.J.R. 1983. Ecological significance of behavioral and hormonal abnormalities in breeding ring doves fed an organochlorine chemical mixture. Arch. Environ. Contam. Toxicol. 12:343–353.

Mineau, P.; Fox, G.A.; Norstrom, R.J.; Weseloh, D.V.; Hallett, D.J.; Ellenton, J.A. 1984. Using the Herring Gull to monitor levels and effects of organochlorine contamination in the Canadian Great Lakes. Pages 425–452 *in* Nriagu, J.O.; Simmons, M.S. (eds.). Toxic contaminants in the Great Lakes. J. Wiley & Sons, New York.

Moore, N.W.; Tatton, J.O'G. 1965. Organochlorine insecticide residues in the eggs of sea birds. Nature 207:42–43.

Morrison, M.L. 1979. Eggshell thickness changes in Pacific Ocean Phaethontidae and Sulidae. Condor 81:209.

Moul, I. 1990. Environmental contaminants, disturbance and breeding failure at a Great Blue Heron colony on Vancouver Island. M.Sc. thesis, Univ. of British Columbia, Vancouver.

Nelson, R.W.; Myers, M.T. 1976. Declines in populations of Peregrine Falcons and their seabird prey at Langara Island, British Columbia. Condor 78:282–293.

Nigam, P.C. 1975. Chemical insecticides. Pages 8–24 *in* Prebble, M.L. (ed.), Aerial control of forest insects in Canada. Department of Environment, Ottawa. 329 pp.

Nisbet, I.C.T.; Reynolds, L.M. 1984. Organochlorine residues in Common Terns and associated estuarine organisms, Massachusetts, U.S.A., 1971–81. Mar. Environ. Res. 11: 33–66.

Noble, D.G. 1990. Environmental contaminants in Canadian seabirds. State of the Environment Rep. 90-2. Environment Canada, Ottawa.

Noble, D.G.; Elliott, J.E. 1986. Environmental contaminants in Canadian seabirds, 1968–95: trends and effects. Can. Wildl. Serv. Tech. Rep. Ser. 13. Ottawa.

Norman, D.; Breault, A.M.; Moul, I.E. 1989. Bald Eagle incursions and predation at Great Blue Heron colonies. Colon. Waterbirds 12:215–217.

Norstrom, R.J.; Muir, D.C.G. 1988. Long-range transport of organochlorines in the arctic and sub-arctic: evidence from analysis of marine mammals and fish. Pages 83–112 *in* Schmidtke, N.W. (ed.), Toxic contamination in large lakes. Vol. 1. Lewis Publ., Chelsea, MI.

Nosek, J.A.; Sullivan, J.R.; Hurley, S.S.; Craven, S.R.; Peterson, R.E. 1992. 2,3,7,8-tetrachlorodibenzo-p-dioxin toxicity in Ring-necked Pheasant hens. J. Toxicol. Environ. Health 35:187–198.

Ohlendorf, H.M.; Fleming, W.J. 1988. Birds and environmental contaminants in San Francisco and Chesapeake bays. Mar. Pollut. Bull. 19:487–495.

Ohlendorf, H.M.; Harrison, C.S. 1986. Mercury, selenium, cadmium, and organochlorines in eggs of three Hawaiian seabird species. Environ. Pollut. 11:169–191.

Ohlendorf, H.M.; Marois, K.C. 1990. Organochlorines and selenium in California night-heron and egret eggs. Environ. Monit. Assess. 15:91–104.

Ohlendorf, H.M.; Miller, M.R. 1984. Organochlorine contaminants in California waterfowl. J. Wildl. Manage. 48:867–877.

Ohlendorf, H.M.; Risebrough, R.W.; Vermeer, K. 1978. Exposure of marine birds to environmental pollutants. U.S. Fish Wildl. Serv. Wildl. Res. Rep. 9. 40 pp.

Ohlendorf, H.M.; Bartonek, J.C.; Divoky, G.J.; Klaas, E.E.; Krynitsky, A.J. 1982. Organochlorine residues in eggs of Alaskan seabirds. U.S. Fish Wildl. Serv. Spec. Sci. Rep. 245. 41 pp.

Ohlendorf, H.M.; Schaffner, F.G.; Custer, T.W.; Stafford, C.J. 1985. Reproduction and organochlorine contaminants in terns at San Diego Bay. Colon. Waterbirds 8:42–53.

Ohlendorf, H.M.; Custer, T.W.; Lowe, R.W.; Rigney, M.; Cromartie, E. 1988. Organochlorines and mercury in eggs of coastal terns and herons in California, USA. Colon. Waterbirds 11:85–94.

Ohlendorf, H.M.; Marois, K.C.; Lowe, R.W.; Harvey, T.E. 1990. Trace elements and organochlorines in Surf Scoters from San Francisco Bay, 1985. Environ. Monit. Assess. 15:91–104.

Olsson, M.; Reutergardh, L. 1986. DDT and PCB pollution trends in the Swedish aquatic environment. Ambio 15:103–109.

Ontario Ministry of Environment. 1985. Polychlorinated dibenzo-p-dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs). Scientific Criteria Doc. 4-84. Toronto.

Peakall, D.B.; Nettleship, D.N. 1987. Organochlorine residue levels in three high Arctic species of colonially-breeding seabirds from Prince Leopold Island. Mar. Pollut. Bull. 18:434–438.

Peakall, D.B.; Peakall, M.L. 1973. Effect of a polychlorinated biphenyl on the reproduction of artificially and naturally incubated dove eggs. J. Appl. Ecol. 10:863–868.

Pearce, P.A.; Peakall, D.B.; Reynolds, L.M. 1979. Shell thinning and residue of organochlorines and mercury in seabird eggs, eastern Canada, 1970–76. Pestic. Monit. J. 13:61–67.

Pearce, P.A.; Elliott, J.E.; Peakall, D.B.; Norstrom, R.J. 1989. Organochlorine contaminants in eggs of seabirds in the northwest Atlantic, 1968–84. Environ. Pollut. 56:217–235.

Phillips, D.J.H.; Tanabe, S. 1989. Aquatic pollution in the Far East. Mar. Pollut. Bull. 20:297–303.

Potts, G.R. 1968. Success of eggs of the Shag on the Farne Island, Northumberland, in relation to their content of dieldrin and p,p-DDE. Nature 217:1232–1284.

Powell, R.L. 1984. Dioxin in Missouri: 1971–1983. Bull. Environ. Contam. Toxicol. 33:648–654.

Risebrough, R.W.; Menzel, D.B.; Martin, D.J.; Olcott, H.S. 1967. DDT residues in Pacific seabirds: a persistent insecticide in marine food chains. Nature 216:589–591.

Risebrough, R.W.; Reiche, P.; Peakall, D.B.; Herman, S.G.; Kirven, M.N. 1968. Polychlorinated biphenyls in the global ecosystem. Nature 220:1098–1102.

Risebrough, R.W.; Sibley, F.C.; Kirven, M.N. 1971. Reproductive failure of the Brown Pelican on Anacapa Island in 1969. Am. Birds 25:8–9.

Rogers, I.H.; Levings, C.D.; Lockhart, W.L.; Norstrom, R.J. 1989. Observations on overwintering juvenile Chinook salmon (*Oncorhynchus tshawytscha*) exposed to bleached kraft mill effluent in the upper Fraser river, British Columbia. Chemosphere 19:1853–1868.

Safe, S. 1984. Polychlorinated biphenyls (PCBs) and polybrominated biphenyl (PBBs): biochemistry, toxicology and mechanisms of action. CRC Crit. Rev. Toxicol. 13:319–369.

Safe, S. 1990. Polychlorinated biphenyls (PCBs), dibenzo-p-dioxins (PCDDs), dibenzofurans (PCDFs), and related compounds: environmental and mechanist considerations which support the development of toxic equivalency factors (TEFs). CRC Crit. Rev. Toxicol. 21:51–88.

Scott, J.M.; Wiens, J.A.; Claeys, R.R. 1975. Organochlorine levels associate with a Common Murre die-off in Oregon. J. Wildl. Manage. 39:310-320.

Seba, D.B.; Corcoran, E.F. 1969. Surface slicks as concentrators of pesticides in the marine environment. Pestic. Monit. J. 3:190–193.

Tanabe, S. 1988. PCB problems in the future: foresight from current knowledge. Environ. Pollut. 50:5–28.

Tanabe, S.; Kannan, N.; Subramanian, A.; Watanabe, S.; Tatsukawa, R. 1987. Highly toxic coplanar PCBs: occurrence, source, persistency and toxic implications to wildlife and humans. Environ. Pollut. 47:147–163.

Tanabe, S.; Kannan, N.; Fukushima, M.; Okamoto, T.; Wakimoto, T.; Tatsukawa, R. 1989. Persistent organochlorines in Japanese coastal waters: an introspective summary from a Far East developed nation. Mar. Pollut. Bull. 20:344–352.

United Nations Environment Programme. 1985. Protection of the marine environment against pollution from land-based sources. UNEP Working Group 120/3 (Part IV): Environment Policy and Law. Elsevier Science Publ. B.V., North Holland.

Van den Berg, M.; Blank, F.; Heeremans, C.; Wagenaar, H.; Olie, K. 198 Presence of polychlorinated dibenzo-*p*-dioxins and polychlorinated dibenzofurans in fish-eating birds and fish from the Netherlands. Arch. Enviro Contam. Toxicol. 16:149–158.

Vermeer, K.; Sealy, S.G. 1984. Status of nesting seabirds of British Columbi Pages 29–40 *in* Croxall, J.P.; Evans, P.E.H.; Schreiber, R.W. (eds.), Status and conservation of the world's seabirds. Int. Counc. Bird Preserv. Tech. Publ. 2. Cambridge, U.K.

Vermeer, K.; Cretney, W.J.; Elliott, J.E.; Norstrom, R.J.; Whitehead, P.E. In press. Elevated polychlorinated dibenzodioxin and dibenzofuran concentrations in grebes, ducks and their prey near Port Alberni, British Columbia, Canada. Mar. Pollut. Bull.

Verrett, M.J. 1970. Witness statement. Pages 190–203 *in* Hearings before the Subcommittee on Energy, Natural Resources and the Environment of the Committee on Commerce, U.S. Senate. U.S. Gov. Printing Office. Serial 91-60. Washington, DC.

Waldichuk, M. 1988. The nature and extent of marine contamination caused by land-based sources in Canada. Pages 75–135 *in* Wells, P.G.; Gratwick, J. (eds.), Canadian conference on marine environmental quality: proceedings. 29 Feb. – 3 Mar. 1988, Halifax, N.S. Published by the International Institute for Transportation and Ocean Policy Studies, Halifax.

Whitehead, P.E. 1989. Toxic chemicals in Great Blue Heron *Ardea herodias* eggs in the Strait of Georgia. Pages 177–183 *in* Vermeer, K.; Butler, R.W. (eds.), The ecology and status of marine and shoreline birds in the Strait of Georgia, British Columbia. Can. Wildl. Serv. Spec. Publ., Ottawa.

Whitehead, P.E.; Elliott, J.E.; Norstrom, R.J.; Vermeer, K. 1990. PCDD and PCDF contamination of diving ducks in the Strait of Georgia, British Columbia, Canada. *In* Hutzinger, O.; Fiedler, H. (eds.), DIOXIN '90 - EPRI Seminar. Ecoinforma Press, Bayreuth, Germany.

Whitehead, P.E.; Norstrom, R.J.; Elliott, J.E. 1992. Dioxin levels in eggs of Great Blue Herons *Ardea herodias* decline rapidly in response to process changes in a nearby kraft pulp mill. DIOXIN '92, 24–28 Aug. 1992. Finnish Institute of Occupational Health, Helsinki, Finland.

Whitehead, P.E.; Harfenist, A.; Elliott, J.E.; Norstrom, R.J. In press. Levels of polychlorinated dibenzo-p-dioxins and polychlorinated dibenzofurans in waterbirds of Howe Sound, British Columbia. Proceedings of The Howe Sound Environmental Workshop, 30 Sept.–3 Oct. 1991, Bowen Island, B.C. Can. Tech. Rep. Fish. Aquat. Sci.

Wolfe, D.A.; Champ, M.A.; Cross, F.A.; Kester, D.R.; Park, P.K.; Swanson, R.L. 1984. Marine pollution research facilities in the People's Republic of China. Mar. Pollut. Bull. 15:207–212.