FRASER RIVER ACTION PLAN

Chlorinated Compounds In Wildlife From The Fraser River Basin

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Abstract

Chlorinated hydrocarbons (dichlorodiphenyltrichloroethane, DDT; polychlorinated biphenyls, PCBs; polychlorinated dibenzo-*p*-dioxins, PCDDs; polychlorinated dibenzofurans, PCDFs) have been measured in eggs of Great Blue Heron (*Ardea herodias*), Double-crested Cormorants (*Phalocrocorax auritus*), Pelagic Cormorants (*P. pelagicus*), Bald Eagles (*Haliaeetus leucocephalus*), and Osprey (*Pandion haliaetus*) as well as the livers of American Mink (*Mustela vison*) and River Otters (*Lontra canadensis*) in the Fraser River Basin between 1977 and 1993. The most common chlorinated hydrocarbons detected in wildlife samples from the Fraser Basin were PCBs and DDE, followed by PCDDs and PCDFs. Non-ortho PCBs and the dioxin congener 2378-TCDD probably pose the greatest health risk to the wildlife species sampled. In general, samples from the upper Fraser River (osprey, mink, otter) were less contaminated than those from the lower Fraser River (heron, cormorants, eagles). Among the birds studied in the lower Fraser River, the highest toxic burdens were observed in eagles, followed by herons and cormorants. Concern over adverse effects on human health and the environment lead to regulatory restrictions on use and production of these chlorinated hydrocarbons. Tissue contaminant levels have generally fallen since those regulations were implemented; however, many chlorinated hydrocarbons still persist in the environment and pervade the foodchain. Ecotoxicology studies suggest that the contaminant levels in wildlife species discussed in this report are currently not having an adverse effect on reproduction, although some biochemical responses (which are biomarkers of a mediated toxic response by the organism) were induced in eagles and herons. The significance of this induction is not well understood.

Résumé

Entre 1977 et 1993, on a dosé les hydrocarbures chlorés (dichlorodiphényltrichloroéthane, DDT; biphényles polychlorés, BPC; polychloro-dibenzo-*p*-dioxines, PCDD; polychloro-dibenzofuranes, PCDF) dans les oeufs de grands hérons (*Ardea herodias*), de cormorans à aigrettes (*Phalocrocorax auritus*), de cormorans pélagiques (*P. pelagicus*), d'aigles à tête blanche (*Haliaeetus leucocephalis*) et de balbuzards (*Pandion haliaetus*), ainsi que dans le foie de visons (*Mustela vison*) et de loutres de rivière (*Lontra canadensis*) du bassin du Fraser. Les hydrocarbures chlorés les plus communs décelés dans les échantillons fauniques de ce bassin étaient les BPC et les DDE, suivis des PCDD et des PCDF. Les BPC non-ortho et la dioxine 2378-TCDD représentent probablement la menace la plus grave pour la santé des espèces fauniques échantillonnées. De façon générale, les échantillons provenant du bassin supérieur du Fraser (balbuzards, vison, loutres) étaient moins contaminés que ceux prélevés dans le bassin inférieur (hérons, cormorans, aigles). Parmi les oiseaux étudiés dans le bassin inférieur, les charges toxiques les plus fortes se retrouvaient chez les aigles, suivis des hérons et des cormorans. Le risque appréhendé d'effets néfastes sur la santé humaine et l'environnement a conduit à la réglementation de l'utilisation et de la production de ces hydrocarbures chlorés. Depuis la mise en oeuvre de cette réglementation, les concentrations de contaminants dans les tissus ont généralement baissé; cependant, de nombreux hydrocarbures chlorés demeurent encore dans l'environnement et pénètrent dans la chaîne alimentaire. Des études d'écotoxicologie laissent supposer que les concentrations de contaminants chez les espèces fauniques examinées ici n'ont pour l'instant aucun effet négatif sur la reproduction, même si certaines réponses biochimiques (biomarqueurs d'une réponse toxique indirecte par l'organisme) ont été induites chez les aigles et les hérons. La portée de cette induction reste encore mal comprise.

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List of Abbreviations

Chlorinated Hydrocarbons

DDT	- dichlorodiphenyltrichloroethane
DDE	- dichlorodiphenyldichloroethane
DDD	- tetrachlorodiphenylethane
PCB	- polychlorinated biphenyl
PCB #77	- 3,3',4,4'-tetrachlorinated biphenyl
PCB #126	- 3,3',4,4',5-pentachlorinated biphenyl
PCB #169	- 3,3',4,4',5,5'-hexachlorinated biphenyl
PCB #118	- 2,3',4,4',5-pentachlorinated biphenyl
PCB #105	- 2,3,3',4,4'-pentachlorinated biphenyl
PCB #157	- 2,3,3’,4,4’5’ -hexachlorinated biphenyl
PCDD(s)	- polychlorinated dibenzo-*p*-dioxin(s)
TCDD(s)	- tetrachlorodibenzo-*p*-dioxin(s)
PnCDD(s)	- pentachlorodibenzo-*p*-dioxin(s)
HxCCD(s)	- hexachlorodibenzo-*p*-dioxin(s)
HpCDD(s)	- heptachlorodibenzo-*p*-dioxin(s)
OCDD(s)	- octachlorodibenzo-*p*-dioxin(s)
PCDF(s)	- polychlorinated dibenzofuran(s)
TCDF(s)	- tetrachlorodibenzofuran(s)
PnCDF(s)	- pentachlorodibenzofuran(s)
HxCDF(s)	- hexachlorodibenzofuran(s)
HpCDF(s)	- heptachlorodibenzofuran(s)
OCDF(s)	- octachlorodibenzofuran(s)
2,4-D	- dichlorophenoxyacetic acid
2,4,5-T	- trichlorophenoxyacetic acid
TCP	- tetrachlorophenol
PCP	- pentachlorophenol
PAH	- polychlorinated aromatic hydrocarbon

Miscellaneous

#	- number
%	- percent
&	- and
<	- less than

>	- greater than
µg/g	- microgram per gram
µg/kg	- microgram per kilogram
Adt	- air dried tonne
A*h*	- hepatic cytosolic receptor protein
AHH	- aryl hydrocarbon (benzo [a] pyrene) hydroxylase
Aroclor 1016	- technical mixture of PCBs
Aroclor 1242	- technical mixture of PCBs
Aroclor 1254	- technical mixture of PCBs
BAEA	- Bald Eagle
BCNH	- Black-crowned Night Heron
BROD	- benzyloxyresorufin-O-deethylase
CAQU	- California Quail
CEPA	- Canadian Environmental Protection Act
Cl	- chlorine
Clophen A50	- technical mix of PCBs
CWS	- Canadian Wildlife Service, Environment Canada
CYP1A	- cytochrome P450 1A cross-reactive proteins
DCCO	- Double-crested Cormorant
EC_1	- dose-effect models congener specific 1% effect level
EC_{50}	- dose-effect models congeners specific median effect level
EROD	- ethoxyresorufin-O-deethylase
FRAP	- Fraser River Action Plan
GBHE	- Great Blue Heron
GVRD	- Greater Vancouver Regional District
HDV I/M	- heavy duty vehicle inspection and maintenance
ICBC	- Insurance Corporation of British Columbia
kg	- kilogram
kg/ADt	- kilogram per air dried tonne
km	- kilometre
LC_{50}	- lethal concentration which results in mortality of 50% of test population
LD_{50}	- lethal dose which results in mortality of 50% of test population
LOEL	- lowest-observed-effect-level
m^3/day	- cubic metres per day
m^3/yr	- cubic metres per year
mg/kg	- milligram per kilogram
MOELP	- (British Columbia) Ministry of the Environment, Lands, and Parks
n	- statistical sample size
ng/kg	- nanogram per kilogram
NOBW	- Northern Bobwhite Quail
NOEL	- no-observed-effect-level
O_2	- oxygen
p	- statistical probability
pers. comm.	- personal communication

pg/g	- picagrams per gram
ppb	- parts per billion, same as µg/kg
ppm	- parts per million, same as mg/kg
ppt	- parts per trillion, ng/kg
Pt.	- Point
r	- statistical correlation coefficient
S.E.	- statistical standard error
TCDD-Eq	- TCDD Equivalent
TCDD-TEQs	- TCDD toxic equivalent
TEF	- TCDD toxic equivalence factor
TEQ	- TCDD toxic equivalent
TEQ_{WHO}	- TCDD toxic equivalent, World Health Organization
TUDO	- Turtle Dove
U.S.	- United States
UBC	- University of British Columbia
WHO	- World Health Organization
wt.	- weight

Executive Summary

The Fraser River Basin, one of the largest and most productive watersheds in Canada, is composed of diverse and productive ecosystems that support internationally significant populations of birds, notably waterfowl, shorebirds, and raptors. The Fraser River delta and surrounding area is highly urbanized and home to 1.5 million people. The productive woodlands and grasslands of the upper basin have also resulted in extensive development of land for forest industry and agricultural uses.

Chlorinated hydrocarbons commonly detected in the environment of the Fraser River Basin include dichlorodiphenyltrichloroethane (DDT), polychlorinated biphenyls (PCBs), polychlorinated dibenzo-*p*-dioxins (chlorinated dioxins or PCDDs), and polychlorinated dibenzofurans (chlorinated furans or PCDFs). Because large quantities of these compounds were released into the environment, concern over adverse effects on human health and the environment led to regulatory restrictions on their use and production. The use of DDT and PCBs in Canada was significantly reduced by the early 1970s. In 1992, the federal government set national standards that have resulted in virtual elimination of the most toxic PCDD and PCDF congeners in pulp mill effluent. However, many chlorinated hydrocarbons still persist in the environment and pervade the food chain.

Concentrations of the major metabolite of DDT, dichlorodiphenyldichloroethane (DDE), PCBs, PCDDs and PCDFs have been measured in the eggs of Great Blue Herons (*Ardea herodias*), Double-crested Cormorants (*Phalocrocorax auritus*), Pelagic Cormorants (*P. pelagicus*), Bald Eagles (*Haliaeetus leucocephalus*), and Ospreys (*Pandion haliaetus*) as well as the livers of American Mink (*Mustela vison*) and River Otters (*Lontra canadensis*) from the Fraser River Basin between 1977 and 1993. Tissue contaminant levels have generally fallen since regulations restricting the emissions of these pollutants were implemented.

The most common chlorinated hydrocarbons detected in the samples collected in the early 1990s were PCBs and DDE, followed by PCDDs and PCDFs. Non-ortho PCBs and the dioxin congener 2378-TCDD probably pose the greatest health risk to the wildlife species sampled. In general, samples from the upper Fraser River (osprey, mink, otter) were less contaminated than those from the lower Fraser River (heron, cormorants, eagles). Among the birds studied from the lower Fraser River, the highest toxic burdens were observed in Bald Eagles followed by herons and cormorants. Differences in contaminant levels detected in various wildlife species must be interpreted carefully since variability can be caused by differences in diet, foraging behaviour, and physiology in addition to proximity to contaminant sources. The overall effect of these contaminant levels on population health and long-term productivity is still not clear. Productivity studies suggest that the contaminant levels currently detected in the wildlife species discussed in this report are not adversely affecting reproduction. However, present contaminant levels do induce some biochemical responses, such as increases in cytochrome P-450 proteins in livers of Bald Eagles and Great Blue Herons. The significance of this induction is debatable; however, it is a biomarker of an A*h*-receptor mediated toxic response by the organism. In some laboratory species, other A*h*-receptor mediated effects, particularly the immune system, occur at levels lower than those associated with P-450 induction.

The levels of chlorinated hydrocarbon contaminants in various indicator species in the Fraser River Basin have been determined. We recommend that monitoring continue for some sentinel species, particularly Great Blue Heron. Egg shells should also continue to be stored in the Canadian Wildlife Service National Specimen Bank. We also recommend some further research on the toxicological implication, particularly on endocrine system endpoints, from those contaminants reported here and for other compounds, such as nonyl-phenols, known to be released into the Fraser River Basin.

1. Introduction

Chlorinated hydrocarbon compounds, such as dichlorodiphenyldichloroethane (DDE), polychlorinated biphenyls (PCBs), polychlorinated dibenzo-*p*-dioxins (chlorinated dioxins or PCDDs), and polychlorinated dibenzofurans (chlorinated furans or PCDFs) have been detected in wildlife samples from all parts of the biosphere, including the Arctic and Antarctic. These contaminants are highly toxic; even at low non-lethal levels, they can effect survival or reproduction. One of the best-known examples is the rapid decline of Peregrine Falcon, Bald Eagle, and other birds of prey populations caused by DDT-related effects on egg shell thickness in the 1950s and 1960s. Most of these compounds, or the processes that produce them, have been banned or strictly controlled in North America; however, some sources remain. PCBs, for example, are still used in old electrical equipment and spills occur, while DDT is used in other countries, and not all sources of PCDDs and PCDFs have been controlled. Moreover, due to their persistence and tendency to bioaccumulate, residues are still present in the environment. Assessment of levels in wildlife to identify existing and new threats will therefore be necessary for the foreseeable future.

Monitoring of British Columbia wildlife for organochlorine contamination began in the late 1960s (Ohlendorf *et al*., 1978; Elliott *et al*., 1989a). Monitoring of chlorinated PCDD and PCDF contamination of wildlife in British Columbia began in 1982. Initial studies found high levels of HxCDDs, PnCDDs, and TCDDs in eggs from the Great Blue Heron colony near the University of British Columbia (UBC). These studies led to an intensive monitoring program that measured chlorinated hydrocarbons in a number of wildlife indicator species throughout British Columbia and, in particular, the Fraser River Basin, where health advisories were issued against consumption of contaminated fish.

In 1991, three levels of government (federal, provincial, and municipal) joined in a Fraser Basin Management Program, of which the Fraser River Action Plan (FRAP) is the federal government's contribution. It involves the clean-up of pollution, the restoration of biological productivity, and the development of partnerships for management of the Fraser River Basin in the future. Since the distribution of these compounds throughout the basin is ecologically important, it is essential to direct clean-up. This report reviews the levels of major chlorinated hydrocarbons in various wildlife indicator species in the Fraser River Basin between 1977 and 1993. This report is intended to be used as reference material for other scientists conducting research in the Fraser River Basin, as well as the concerned public.

The report briefly describes the Fraser River Basin focusing mainly on local point sources of chlorinated hydrocarbons. Characteristics of specific contaminants are briefly reviewed. The levels of DDE, PCBs, PCDDs, and PCDFs detected in the eggs of Great Blue Herons, Double-crested Cormorants, Pelagic Cormorants, Bald Eagles and Ospreys as well as the liver of American Mink and River Otters collected from the Fraser River Basin between 1977 and 1993 are summarized. Geographic differences and temporal trends in the data are discussed and the contaminant levels observed in the various indicator species compared. Tentative conclusions are drawn regarding the potential health risks that the observed levels of chlorinated hydrocarbons pose to the wildlife populations. The report concludes with recommendations for further action.

Concentrations of chlorinated hydrocarbons in water, sediment, and other ecosystem indicators such as shellfish, fish and waterfowl are not discussed in this report. This information is available in Mah *et al*. (1989), Whitehead *et al*. (1990), Drinnan *et al*. (1991), and Vermeer *et al*. (1993). Heavy metal residues are also not discussed; mercury and lead concentrations in seabirds, waterfowl, and eagles are presented in Noble and Elliott (1986), Elliott *et al*. (1992), Kennedy and Nadeau (1993), and Elliott *et al*. (in prep).

2. Fraser River Basin

The Fraser River Basin is one of the largest and most productive watersheds in Canada draining approximately one quarter of British Columbia (Figure 1). It supports nearly two million people (about 65% of British Columbia's population), and is the location of more than 75% of the industrial activity of the province. In terms of provincial totals, the Fraser River Basin provides 48% of commercial forest area, 60% of mining operations, 80% of gross provincial product, and 66% of total household income (Paquet, 1994).

For the purposes of this paper it can be divided into two main sections: the upper and lower basins. The upper basin drains the interior plateau and surrounding mountains, finally cutting the Fraser River Canyon through the Cascade Range, ending at Hope, 1,300 km downstream from its source. The upper basin has extensive forests and grasslands, important salmon spawning habitat, high recreational value. A total of 137,000 people, almost half of the region's population, live in the two major cities, Kamloops and Prince George (British Columbia Regional Index, 1995). The main economic activities in the area are forestry, ranching, mining, and tourism.

The lower basin extends from Hope through to the Strait of Georgia, encompassing an area of 1.1 million hectares (British Columbia Regional Index, 1995). It is a diverse and productive ecosystem including coastal forests, floodplains, delta marshes and streams. The mild climate, suitable habitat, and abundant food sources attract a variety of wildlife. Butler and Campbell (1987) estimated that as many as 1.4 million birds use the estuary including internationally significant populations of a variety of waterfowl and raptors. The Fraser River delta also contains Canada's third largest metropolitan area supporting 1.5 million people, approximately one half of the population of British Columbia (British Columbia Regional Index, 1995). Between 1986 and 1991, the population of the Vancouver area increased 9.6% (British Columbia Regional Index, 1995). Although a substantial amount of farmland has been lost to other uses over the years, it remains the leading farming district by a wide margin. The 214,307 acres of farmland (7.8% of total acreage) produce 83% of vegetables, 85% of small fruits, and 68% of potatoes grown province-wide (British Columbia Regional Index, 1995). The Lower Mainland accounted for 53% of the provincial industrial output in 1981. Eighty-eight percent of the Lower Mainland industrial activity is within the Greater Vancouver Regional District (GVRD), which accounts for 47% of the total provincial output (GVRD, 1988a). To protect this agricultural, urban, and industrial land from flooding, the river has been extensively diked and channelled.

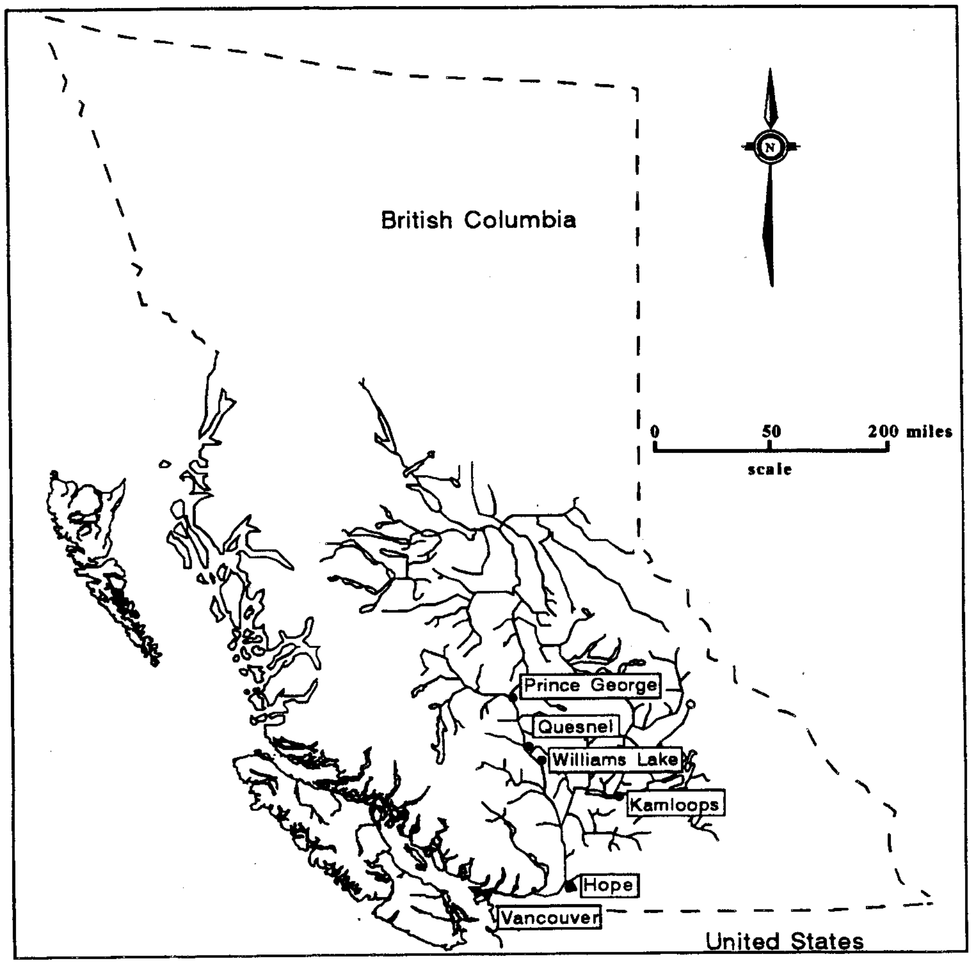


Figure 1. Fraser River Basin.

3. <u>Sources of chlorinated hydrocarbons</u>

3.1. Uses and methods of production

3.1.1. *Dichlorodiphenyltrichloroethane (DDT)*

Dichlorodiphenyltrichloroethane (DDT) is probably the most infamous chemical ever produced. It was used as a broad spectrum insecticide from the 1940s to the early 1970s. At the peak of its production in 1969, almost 2 million pounds of DDT was used annually in Canada (Statistics Canada, 1971). The development of insect resistance, along with regulation to curtail use because of its long persistence and bioaccumulation in the food chain, resulted in dramatic reduction in the use of DDT. Restrictions were first implemented in the early 1970s, with further controls imposed throughout the 1970s and 1980s (Noble and Elliott, 1986). DDT is still currently used in many parts of the world, such as Latin America (Elliott and Noble, 1993).

3.1.2. *Polychlorinated biphenyls (PCBs)*

Polychlorinated biphenyls (PCBs) are a mixture of compounds that were predominantly used as coolant-insulants and heat transfer agents in a number of electrical products such as transformers and capacitors. Previously they had a wide variety of applications and were used in many products such as printing inks, adhesives, and paints. First produced in 1929, PCBs were recognized as a global environmental contaminant in the late 1960s. Although PCBs were never manufactured in Canada, their production in the United States was significantly reduced in 1970, and completely banned in North America in 1977 (Canadian Council of Resource and Environment Ministers, 1986). However, PCBs continue to enter the environment during disposal of electrical equipment, accidental spills, and leakage from landfills (Tanabe *et al*., 1987).

3.1.3. *Polychlorinated dibenzo-p-dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs)*

There are 75 chlorinated PCDD congeners and 135 PCDF congeners. They are present as trace impurities in a variety of compounds; their presence is neither intentional nor desired (Eisler, 1986). There are three primary sources: creation during combustion of certain organic materials in the presence of chlorine; presence as impurities in some chemicals; creation by certain industrial processes and discharge in air emissions or liquid effluent. Stringent regulations have reduced the release of PCDDs and PCDFs into the environment from most of these sources.

1. <u>Combustion</u> Common sources include emissions from the incineration of municipal and industrial wastes. Emissions from combustion sources vary depending on the type of fuel, combustion temperature, residence time at temperature, and many other factors (Hites, 1990). On average, there is a similar likelihood of all PCDDs and PCDFs being produced, except for the OCDFs and TCDDs (Hagenmaier *et al*., 1986). Less chlorinated PCDDs and PCDFs partition into the vapor phase, a process that is highly temperature dependent (Hites, 1990).

Gasoline and diesel engine exhaust are also sources of PCDD emissions (Douben *et al*., 1995; Hagenmaier, 1994; Jones, 1995). Douben *et al*. (1995) ranked mobile engine exhaust as a relatively small contributor to total PCDD emissions in the United Kingdom. However, this estimate may be understated (Jones, 1995), especially in Europe where leaded gasoline with chlorine scavengers was used much later than in North America (Norstrom, pers. comm.). Dominant congeners in heavy duty diesel truck emissions are 2378 HxCDD, HpCDD and OCDD (Jones, 1995).

2. Synthesis of industrial chemicals The synthesis of some organic chemicals (such as certain herbicides and chlorophenol-based wood protection products) produces specific congeners as impurities that are extremely toxic and bioaccumulate in fish and wildlife (Ramel, 1978; Rappe, 1984). Strictly controlled conditions during the production of some of these chemicals (e.g. 2,4-D and 2,4,5-T) have significantly reduced the amount of such impurities. In the case of wood protection chemicals, production has decreased as alternate wood protection chemicals have become available. None of these commercial chemicals is currently manufactured in Canada (Government of Canada, 1990) and chlorophenol-based wood protection products are heavily restricted in British Columbia. However, the large number of sites in British Columbia where chlorophenols were used has left a legacy of contamination in soils and adjacent aquatic and marine sediments.

3. Forestry-related industries In British Columbia until recently, forest-related industries were an important source of PCDDs and PCDFs. The effluent of kraft pulp mills that used molecular chlorine bleaching contained PCDDs and PCDFs, particularly, 2378-TCDD and 2378-TCDF (Kuehl *et al*., 1987; Norstrom, 1988). The presence of defoaming agents containing chemical precursors of PCDDs and PCDFs contributed to their formation. Chlorophenols, used to protect lumber and preserve wood, contain a host of PCDDs and PCDFs but predominantly OCDDs and HpCDDs (Miles *et al*., 1985). Stacking treated lumber without protection from the rain has been responsible for chlorophenols in receiving waters around sawmills (Krahn *et al*., 1987; Environmental Protection Service, 1979). The use of chlorophenol contaminated woodchips in the production of pulp by kraft mills produces HxCDDs and PnCDDs (Luthe *et al*., 1990; Norstrom *et al*., 1988). Pulp and paper mills routinely dispose of effluent sludge and hogged waste wood fuel (hog fuel) by incineration in their power boilers. Even chips from salt water-soaked logs may provide the chlorine necessary for production of PCDDs and PCDFs in these boilers. Although scrubbers retain 70% of the PCDDs and PCDFs in the emissions, some are still released into the environment (Bovar-Concord Environmental, 1994). Since the late 1980s, the concentration of contaminants in effluent from forestry-related industries has been severely reduced; details are discussed in Section 3.2.1. Specific PCDD and PCDF congeners associated with particular production processes are summarized in Table 1.

3.2. Sources of contaminants in the Fraser River Basin

Until they were controlled in the late 1980s and 1990s, large concentrations of chlorinated hydrocarbons were released into the Fraser River. Legislation has substantially reduced the concentrations of contaminants allowed in industrial effluent. The effluent from industries, and the pesticides and wood treatment products that were primary sources are comparatively clean today, but other sources still exist. It is necessary to know both the past and present sources of contamination in the Fraser River Basin to explain the contaminant levels still observed in the local wildlife populations.

Table 1. Principal PCDD and PCDF congeners produced from various sources.

Source	Major Congeners	Reference
Combustion		
municipal incinerators	all except OCDFs, TCDD	Hagenmaier *et al*. (1986)
diesel engine emissions	2378 HxCDD HpCDD OCDD	Jones (1995)
Production of industrial chemicals		
Chlorophenol synthesis		
2,4-D	1268-, 1379-TCDD	Van den Berg *et al*. (1987)
2,4,5-T	2378-TCDD	Van den Berg *et al*. (1987)
Tetrachlorophenol (TCP)	2468-TCDD	Van den Berg *et al*. (1987)
Pentachlorophenol (PCP)	HxCDDs	Van den Berg *et al*. (1987)
PCB technical mix	PCDFs	Kociba and Cabey (1985) & Van den Berg *et al*. (1987)
Forestry-related industries		
Kraft pulp mills with chlorine bleaching process	2378-TCDD/F, 1278-TCDF, OCDDs	Norstrom (1988) & Kuehl *et al*. (1987)
Sawmills using chlorophenols	OCDDs, HpCDDs	Miles *et al*. (1985)
Pulp production using chlorophenol treated wood	HxCDDs, PnCDDs	Luthe *et al*. (1990) & Norstrom *et al*. (1988)
Hog fuel/sludge incineration	2378-TCDD/F, 23478 PnCDF, 12378-PnCDD	Bovard-Concord Environmental (1994)

3.2.1. *Point sources*

The residential population and industrial activities in the Fraser River Basin have an impact on the quality of the aquatic environment. Point sources include: sewage treatment plant outflows, landfill leachates, emissions from hog fuel burners and municipal incinerators, and effluent from industrial operations. The primary sources of chlorinated hydrocarbons in the Fraser River Basin have been the industrial operations, in particular, forestry-related industries.

On average, just over one million m^3/day of effluent was discharged into the Fraser River estuary during 1990 to 1992. Approximately 91% (992,939 m^3/day) was released from the four wastewater treatment facilities, 0.03% (373 m^3/day) from private municipal sewage treatment plants, and 8.6% (93,522 m^3/day) from industrial discharge (Moore, 1994).

<u>Sewage outflows</u>

There are 33 sewage treatment plants in the Fraser River Basin, of which four are located in the lower basin. Ninety-five percent of the municipal waste discharge volumes comes from cities and towns in its lower reaches (Paquet, 1994). Sewage is comprised of effluent from municipal sewage treatment plants, urban run-off, as well as domestic and industrial waste water (Lamparski *et al*., 1984; Horstmann *et al*., 1992). In 1993 to 1994, the Federal Environmental Protection Branch inspected 10 of the 33 plants; of the 40 inspections, 23 were found in compliance, for an overall rating of 58% (Paquet, 1994). Two of the four plants in the Fraser River estuary were not in compliance (GVRD, 1993).

<u>Landfills</u>

There are six active landfills in the Fraser River estuary which discharge leachate into the local sewage systems. Landfill leachate accounts for, on average, 1.2% and 1.5% of total flows of the Annacis and Lions Gate Sewage Treatment Plants, respectively (GVRD, 1988b).

<u>Emissions from hog fuel burners and municipal incinerators</u>

Most pulp and paper mills dispose of the hogged waste wood fuel and sludge by incineration in their steam generating boilers. Organic contaminants in the boiler flue gas are captured in the multiclones and scrubbers; combined capture efficiency is greatest for PCDDs and PCDFs (70%), followed by chlorophenols, chlorobenzenes, and PAHs (Bovard-Concord Environmental, 1994).

In 1986, four British Columbia municipalities incinerated their refuse, none of which was located in the Fraser River Basin (British Columbia Ministry of Environment and Parks, 1988). However, the Greater Vancouver Regional District (GVRD) opened an incinerator facility in Burnaby in 1988.

<u>Effluent from industrial operations</u>

A variety of industries discharge waste into the Fraser River including: some domestic type effluent from hotels and restaurants not connected to the sewage system, forest, food, metal, cement, miscellaneous and uncontaminated cooling water (Fraser River Estuary Study, Water

Quality, 1979). The greatest volume of effluent and the primary contributors to chlorinated hydrocarbons released into the Fraser River are the forestry related industries.

There are 26 active pulp and paper mills in British Columbia, of which ten are located on the Fraser and Thompson Rivers. They are the Prince George and Intercontinental Pulp and Paper Mills #1 and #2 and Northwood Pulp and Paper in Prince George, the Quesnel Pulp and the Cariboo Pulp and Paper in Quesnel, the Weyerhaeuser Pulp Mill in Kamloops, and the Burnaby Paperboard Division, Scott Paper Western Mfg. Division, Island Paper Mills Company, and Newstech Recycling in the Lower Mainland. Historically, pulp and paper mills released substantial amounts of chlorinated hydrocarbons into the local environment, but recent process changes enforced by new federal and provincial regulations has resulted in major declines in the contaminant levels in effluent. Between 1988 and 1993, an average reduction of 78% in chlorinated hydrocarbons generated per tonne of pulp produced in British Columbia was measured (MOELP, 1994). In 1993, the annual average chlorinated hydrocarbon discharge for all mills was 1.4 kg/ADt (air dried tonne). Twelve of the 17 mills using chlorine bleaching have met the regulations of 1.5 kg/ADt, formally required by 1995. The level of PCDDs and PCDFs in effluent from mills has also declined dramatically. Between 1990 and 1993, the average total daily PCDD discharge from all bleached kraft mills across the province was approximately 7.3 mg, a reduction of 85% since 1990. Similarly, the average daily discharge of PCDFs across the province was about 73 mg, roughly 86% lower than in 1990 (MOELP, 1994).

Sawmills have contributed to local pollutant levels by using biocides to preserve wood products and protect lumber for export. The 19 wood preservation facilities in British Columbia use approximately 4,500 tonnes of wood preservation chemicals annually (Paquet, 1994). Although the more hazardous products were largely removed from the Canadian market in November 1989, PCDDs and PCDFs associated with chlorophenols are still present in the foodchain in the Fraser estuary (Harfenist *et al*., 1993).

There are 93 sites registered in the PCB inventory of British Columbia (Paquet, 1994). Environmental Protection Branch of Environment Canada inspected six regulated facilities in the Fraser River Basin; three were out of compliance for some of the criteria on the inspectors check list (Paquet, 1994). Although PCBs have been prohibited in anti-sapstains (Pest Control Production Act Revisited), regulations of the Canadian Environmental Protection Act (CEPA) established in 1988, still permit 5 ppm PCBs in road oiling, 50 ppm PCBs in old electrical equipment, the release of 1 gram per day of PCBs from commercial, manufacturing, and processing activities, and 6 ppb in stormwater runoff in British Columbia (Paquet, 1994).

3.2.2. *Non-point sources*

Non-point sources of toxins in the Fraser River Basin include: storm water runoff, pesticides used in agriculture and forestry, leaching of PAHs from railway ties and pilings, and gasoline and diesel engine exhaust.

Storm water run-off

Storm water includes rain water and snowmelt that has run off streets, roofs, and open areas as well as water that has been collected by storm drains and ditches. There are over 500 municipal storm sewage water outfalls, ditches and streams in the Fraser River estuary (Lidstone *et al*., 1993). Storm water can contain toxic materials such as PCBs that are carried from paved areas in street sediment, but limited bioassays reported that storm water was not acutely toxic to fish (Ferguson and Hall, 1979). Untreated storm water discharges to the river contributed about the same amount of contaminant as primary treated municipal effluents (Fraser River Estuary Study, Water Quality, 1979).

Pesticides used in agriculture and forestry

Residues of agricultural pesticides may enter aquatic systems through run-off from fields, direct deposition in ditches or streams, accidental spills, improper storage, drift and contaminated ground water (Hagen, 1990). The annual precipitative run-off from agricultural land in the Fraser River estuary has been estimated at 155,803,000 m^3/yr (Hagen, 1990). Currently, the greatest health concerns are associated with organophosphate and carbamate pesticides; they are not discussed in this report. However, historically, several pesticides, which contained PCDD and PCDF residues, were used in large quantities.

In 1982, 2,4-D (amine and ester formulations) was the most commonly used herbicide in forestry, accounting for 55% of all pesticide by area treated (2,802 hectares treated), or 74% by weight (Humphries, 1982). By 1990, the proportion had dropped to 1.7% of the area treated with herbicides, with 1,052 hectares treated with 2,4-D (Humphries, 1990), and 1% in 1991 (Environment Canada and MOELP, 1993). In 1991, 2,4-D accounted for 3% of all agricultural pesticide active ingredients, 7% of all domestic pesticide sales excluding wood preservatives, 7% of pesticides used in landscaping, and 1% of all forestry pesticides (Environment Canada and MOELP, 1993).

Leaching of PAHs from railway ties and pilings

Heavy duty wood preservative chemicals provide long-term protection against fungi, insects, and marine borers for wood used in exposed situations (eg. patio decks, railway ties, pilings). Anti-sapstain chemicals provide short-term protection against fungal growth, which can cause staining and hence reduce marketability, of cut lumber. In 1991, creosote accounted for 53% of heavy duty wood preservation chemicals, and pentachlorophenol (PCP) accounted for 26%. PCP was previously used extensively for anti-sapstain protection of lumber, but by 1991 was no longer used for this purpose, having been replaced by alternatives without PCDD and PCDF contamination. In 1991, 1,691,000 kg of creosote (33.6% of all pesticide active ingredients) and 790,000 kg of PCP (15.6% of all active ingredients) were used in British Columbia. The British Columbia storm water run-off regulations allow 6.0 ppb of PCP (Paquet, 1994).

Gasoline and diesel engine exhaust

The existing Air Care program regulates emissions from light duty vehicles; heavy duty vehicles are currently exempt from any form of emissions testing. In 1994, over 18,000 heavy duty vehicles were registered in the Lower Fraser Valley, the majority of which were most likely commercial and fuelled by diesel. The Insurance Corporation of British Columbia estimated another 35,000, mostly diesel heavy duty trucks visit the area each year and probably contribute about 25% of the total heavy duty vehicle emissions in the area (HDV I/M Task Force Report, 1994).

Although gasoline and diesel engine exhaust are known sources of PCDD emissions, a 1994 report by the Heavy Duty Vehicle Inspection and Maintenance Task Force concluded that the most significant heavy duty vehicle emissions in the Lower Fraser Valley, when compared to the total emissions inventory, were fine particulate matter and nitrogen-oxides from heavy duty diesel vehicles; PCDD levels were not reported.

4. Toxicity of chlorinated hydrocarbons

4.1. Relative toxicity of various PCB, PCDD and PCDF congeners

The relative toxicity of the 209 PCB congeners is determined by the chlorine substitution pattern of the polychlorinated biphenyl ring (Figure 2). Congeners without chlorines at the ortho positions (eg. PCB #77, 126, 169) are highly toxic but make up a small percentage of the total PCBs produced (Tanabe *et al*., 1987; Safe, 1990). Mono-ortho chlorine substituted PCBs (eg. PCB #105, 118) are less toxic than non-ortho coplanar PCBs, but make up a larger percentage of the total PCBs produced.

There are 75 PCDD congeners and 135 PCDF congeners. The relative toxicity of the congeners is determined by their molecular structure with the most toxic congeners halogated in the 2,3,7, and 8 positions (Figure 2). In general, highly chlorinated congeners are less toxic than congeners with fewer chlorines and PCDFs are less toxic than the corresponding PCDDs, except for 23478-PnCDF that is very toxic (Safe, 1990).

4.2. Mode of action and general toxicological effects

The more toxic forms of chlorinated hydrocarbons have common characteristics that account for their similar behaviour in the environment and similar toxicological responses. Although most are not acutely toxic to wildlife at the levels normally encountered in environmental samples, they are very lipophilic, highly persistent, and biomagnify in the foodchain. Predatory animals, such as raptorial and fish-eating birds, can become highly contaminated due to biomagnification through the food chain and their tendency to select impaired and potentially contaminated prey (Noble and Elliott, 1986; Noble *et al*., 1993)

DDT and its metabolites effect reproduction in avian species by interference of DDE with the metabolism of calcium which reduces shell thickness and quality (Cooke *et al*., 1982).

PCDDs and PCDFs, non-ortho PCBs, and some mono-ortho PCBs have a similar mode of action: they bind to a cytosolic receptor protein referred to as the *Ah* receptor (Safe, 1984; Landers and Bunce, 1991). The *Ah* receptor appears to mediate a wide range of

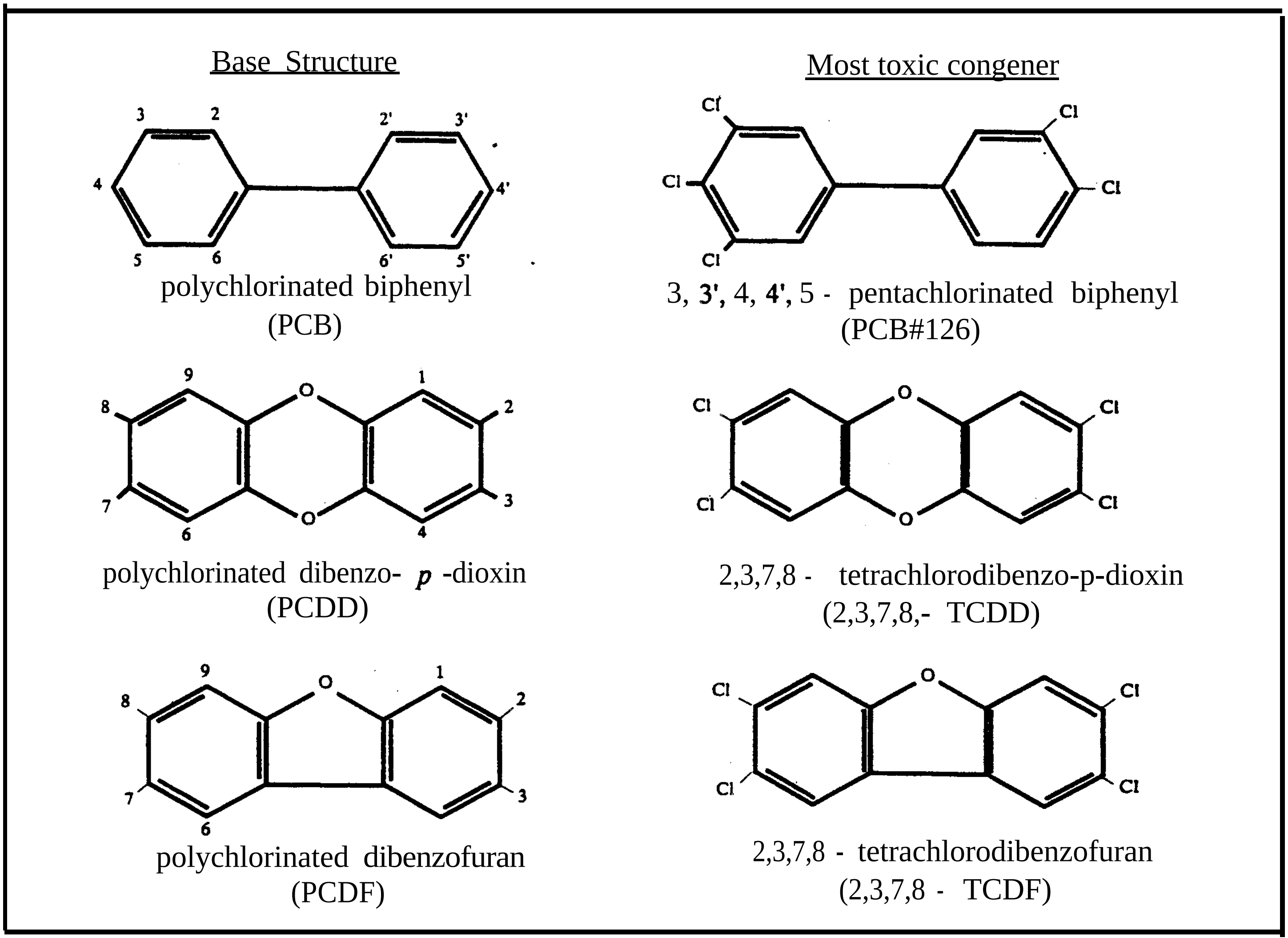


Figure 2. **Chemical structure of various PCBs, PCDDs and PCDFs. Numbers highlight position of chlorine substitution on biphenyl ring.**

toxicological effects including developmental, reproductive, and immunological (Safe, 1984; Peterson et al., 1993). Toxicological effects of PCBs in wildlife were reviewed by Hansen (1987) and Bosveld and Van den Berg (1994). Toxicological effects of PCDDs and PCDFs in wildlife were reviewed by the Ontario Ministry of the Environment (1985), Eisler (1986), and Gilbertson et ***al.*** (1991).

Although the prevailing model of dioxin-like toxicity, the toxic equivalent scheme (TEQs, see Section 4.4), assumes additivity, synergistic effects have been demonstrated. Bimbaum et ***al.*** (1985) showed that although some PCBs resulted in a low incidence of cleft palate deformities in mice, mixtures of 2378-TCDD and PCB #157 caused a ten-fold increase in the incidence of cleft palate.

palate.

4.3. Critical concentrations of contaminants in wildlife

Considerable literature has been published on the toxic effects of chlorinated hydrocarbons in wildlife. During the 1960s and 1970s, a time period of intense pesticide and industrial contaminant use, many birds (particularly raptors) were found dead or dying from acute exposure to toxins. Consequently, research was conducted to determine the acute toxicity of a range of contaminants in various wildlife species. Doses of the major chlorinated hydrocarbons that are acutely toxic to various wildlife species are summarized in Table 2.

Table 2. Summary of critical levels of major chlorinated hydrocarbons that cause acute toxicity in various wildlife species.

Chlorinated Hydrocarbon **Toxicity**	**Contaminant Level**	**Tissue**	**Animal**	**Reference**
DDT				
acute oral LD_{50}[1]	595 mg/kg	diet	CAQU[2],male,6months	Hudson *et al*. (1984)
acute oral LD_{50}	>2,240 mg/kg	diet	Mallard,female,3months	Hudson *et al*. (1984)
DDE				
acute	100 mg/kg	liver	raptor	Cooke *et al*. (1982)
	250 mg/kg	brain	raptor	Noble & Elliott (1986)
PCBs				
acute	100 mg/kg	liver	raptor	Cooke *et al*. (1982)
	500-3000 mg/kg	brain	raptor	Noble & Elliott (1986)
acute oral LD_{50}	>2,000 mg/kg	diet	NOBW[3],male,12months	Hudson *et al*. (1984)
acute oral LD_{50}	>>2,000 mg/kg	diet	Mallard,male,8-9months	Hudson *et al*. (1984)
TCDD				
acute oral LD_{50}	15 µg/kg	diet	NOBW,male,7months	Hudson *et al*. (1984)
acute oral LD_{50}	>810 µg/kg	diet	TUDO[4],male,adult	Hudson *et al*. (1984)
acute oral LD_{50}	>>108 µg/kg	diet	Mallard,1week	Hudson *et al*. (1984)

(1) LD_{50} - lethal dose which results in mortality of 50% of test population
(2) CAQU - California Quail
(3) NOBW - Northern Bobwhite
(4) TUDO - Ringed Turtle Dove

Most data in the literature related contaminant levels measured in tissues or eggs with reproductive performance, survivorship, and population trends (Noble *et al*., 1993). Different wildlife species have varying sensitivities to particular contaminants. Contaminant levels that are defined in the literature as 'critical' concentrations that adversely affect survival or reproduction in the wildlife species discussed in this report have been summarized in Tables 2 and 3. Specific details are presented in the section of the report dealing with the individual indicator species (Section 6).

4.4. TCDD Toxic Equivalent (TEQs)

Several PCB, PCDD and PCDF congeners are widely distributed and can often be found in biological samples from nominally "pristine" locations. Until recently, it has been difficult to assess the overall effect of the total contaminant burden in the animal. A methodology has been developed which assesses the toxicity of complex mixtures of these chemicals based on the structure activity relationships of the various congeners. Each congener is assigned a 'TCDD Toxic Equivalence Factor' (TEF). It is a number between 0 and 1, which relates its toxicity to that of the most toxic congener 2378-TCDD which has a value of 1. The concentration of each compound detected in the sample is multiplied by the TEF and the products are summed to produce the 'TCDD Toxic Equivalent' (TEQs). The TEFs for the most important and common polychlorinated aromatic hydrocarbons are listed by Safe (1990) and more recently by Ahlborg *et al*. (1994). TEF values for selected PCBs (determined by the World Health Organization, WHO) and PCDDs and PCDFs (determined by Safe) used for this report are listed in Table 4.

5. Methods

For this paper, we reviewed results of studies conducted by the Canadian Wildlife Service in the Fraser River Basin and associated marine environments for the period between 1977 and 1993. The studies reviewed included the collection of eggs of Great Blue Herons, Double-crested Cormorants, Pelagic Cormorants, Bald Eagles, and Ospreys as well as livers of American Mink and River Otters from the Fraser River Basin. Each of the eggs collected in a season was taken from a different nest. The mink and otter carcasses were obtained from local trappers; only animals trapped within 10 metres of the main river bank were sampled.

Egg contents and tissues were placed into acetone and hexane rinsed glass jars with a chemically cleaned foil liner between the lid and the jar, or wrapped in aluminum foil. Samples were stored frozen until analysis. Residues of DDE, PCBs, PCDDs and PCDFs were determined by the Canadian Wildlife Service at the National Wildlife Research Centre in Hull, Quebec. Detailed methods for the collection and storage of the specimens as well as the analytical methods are described in Norstrom *et al*. (1990) and Turle *et al*. (1991). All of the collection is currently stored in the Canadian Wildlife Service National Specimen Bank at the National Wildlife Research Centre. Concentrations of some PCB, PCDD and PCDF congeners were not measured for many of the earlier samples since the analytical methodology was not available. Residues are expressed on a wet weight basis.

The data are presented as geometric means since residue concentrations are often skewed and commonly exhibit a log normal distribution (Ohlendorf *et al*., 1978). TEQs were calculated using the PCDD and PCDF congeners which substantially contributed to the overall toxic burden. Statistical comparisons of some geographic and temporal trends were not possible due to the small size of the datasets. This report contains some previously unpublished data collected by the Canadian Wildlife Service.

Table 3. Summary of critical levels of major chlorinated hydrocarbons in eggs and tissues of various wildlife species which cause sub-lethal toxicity, biochemical effects, egg shell thinning, reproductive and behavioural problems.

Chlorinated Hydrocarbon Species	Contaminant Level	Tissue	Effect	Reference
DDE				
GBHE[1]	-	egg	shell thickness neg. correlated with log DDE levels	Blus *et al*. (1980)
BCNH[2]	8.0 mg/kg	egg	shell thinning	Custer *et al*. (1983)
Cormorant	3-5 mg/kg	egg	7-14% shell thinning	Heinz *et al*. (1985)
	10 mg/kg	egg	20% shell thinning & severe reproductive failure	Pearce *et al*. (1979)
BAEA[3]	4.0 µg/g	egg	10% shell thinning	Wiemeyer *et al*. (1993)
	approx. 16 µg/g	egg	15% shell thinning	Wiemeyer *et al*. (1993)
	<3.6 µg/g	egg	'normal' productivity	Wiemeyer *et al*. (1993)
	3.6-6.3 µg/g	egg	50% of normal productivity	Wiemeyer *et al*. (1993)
	>6.3 µg/g	egg	25% of normal productivity	Wiemeyer *et al*. (1993)
Osprey	4.2 mg/kg	egg	15% shell thinning & egg breakage	Poole (1989), Wiemeyer *et al*. (1988)
	>5-10 mg/kg	egg	improper embryo development	Poole (1989)
Raptor[4]	10 mg/kg	liver	sub-lethal toxicity - hyperthyroidism, change in heart wt. & function, altered production of adrenal corticosterone	Cooke *et al*. (1982)
	< 4 mg/kg	liver	behavioural effects	Cooke *et al*. (1982)
Mink	100 mg/kg DDT & 100 mg/kg DDT + 50 mg/kg DDE	diet	no reproductive failure	Aulerich & Ringer (1970), Duby (1970)

Table 3. cont...

Chlorinated Hydrocarbon Species	Contaminant Level	Tissue	Effect	Reference
PCBs				
BCNH	4.5 mg/kg	egg	reduced embyronic growth	Hoffman *et al*. (1986)
	-	embryo	enhanced CYP1A, EROD, BROD activity correlated with PCB concentration, r^2=0.42[5]	Rattner *et al*. (1994)
DCCO[6]	-	embryo	TCDD-Eq (derived from EROD activity of treated H4IIE rat hepatoma cells) correlated with egg mortality rate, r^2=0.703[7]	Tillitt *et al*. (1992)
	-	egg	EROD induction, reduced yolk weight	Sanderson *et al*. (1994a)
Osprey	>25 mg/kg	egg	no observable effect in productivity	Poole (1989)
Raptor	10 mg/kg	liver	sub-acute toxicity - increased thyroid wt.	Cooke *et al*. (1982)
	50 mg/kg	egg	reproductive effects	Noble *et al*. (1993)
Mink	11.0 µg/g Aroclor[8]	brain	mortality	Aulerich *et al*. (1973)
	4.2 µg/g Aroclor[8]	liver		
	4.5 µg/g Aroclor[8]	kidney		
	79 µg/g Aroclor 1254	28 day diet	LD_{50}	Aulerich *et al*. (1986)
	7 µg/g Aroclor 1254	28 day diet	sub-lethal toxicity - decreased appetite & body wt. kidney & heart wt. decrease, liver & adrenal wt. increase	Hornshaw *et al*. (1986)
	2 µg/g Aroclor 1254	diet	reproductive failure	Byrne (1974)
	2.0 µg/g Aroclor 1254	liver	adult female - no effect	Wren *et al*. (1987)

Table 3. cont...

Chlorinated Hydrocarbon Species	Contaminant Level	Tissue	Effect	Reference
—				
	1.75 µg/g Aroclor 1254	liver	5 week kit nursed by above female - reduced growth and survival	Wren *et al*. (1987)
	2.4 µg/g PCBs = 0.22 µg/g PCB153= 200 pg/g TCDD-TEQ[9]	tissue	EC_{50} kit survival	Leonards *et al*. (1994)
	1.3 µg/g PCB	tissue	no-observed-effect[10] , kit survival	Leonards *et al*. (1994)
	20 µg/g Aroclor 1242	28 day diet	no enzyme induction	Shull *et al*. (1982)
	20 µg/g Aroclor 1016	28 day diet	induced CYP1A, BaP hyroxylase, ECOD	Shull *et al*. (1982)
	2 mg non- & mono-ortho fractions of Clophen A50	daily diet during reproduction season	enhanced EROD 2-3x in females & 30x in kits of treated females	Brunstrom (1992)
	1.64 mg Aroclor 1254	79-94 day diet	enhanced AHH 2x in adults	Brunstrom (1992)
PCDDs/PCDFs				
GBHE	-	embryo	reduced body, stomach, intestine wt. negatively correlated with TCDD, r>0.48, p<0.01	Sanderson *et al.* (1994b)
	-	embryo	EROD induction positively correlated with TCDD	Bellward *et al.* (1990) & Sanderson *et al*. (1994b)
	150-200 ng/kg	chick	subcutaneous edema	Hart *et al*. (1991)
	<100 ng/kg 2378-TCDD	chick	no observable effects including EROD induction	Sanderson *et al*. (1994b)
BAEA	-	egg yolk sac	CYP1A, EROD, BROD induction positively correlated with 2378-TCDD/F	Elliott *et al*. (1996)
	100 ng/kg TEQ_{WHO}	egg yolk sac	NOEL, CYP1A induction biomarker	Elliott *et al*. (1996)

Table 3. cont...

Chlorinated Hydrocarbon Species	Contaminant Level	Tissue	Effect	Reference
	210 ng/kg TEQ_{WHO}	egg yolk sac	LOEL, CYP1A induction biomarker	Elliott *et al*. (1996)
Mink	4.2 µg/kg 2378-TCDD	28 day oral	LD_{50}, adult male	Hochstein *et al*. (1988)
	4.3 µg/kg 2378-TCDD	diet	28 day diet LC_{50}, adult female	Hochstein *et al*. (1988)
	0.1 µg/kg 2378-TCDD	diet	12 day diet LC_{50}, neonate	Aulerich *et al*. (1988)
	5 ng /kg 2378-TCDD	diet	reproductive	Elliott & Whitehead (1989b)

(1) GBHE - Great Blue Heron
(2) BCNH - Black-crowned Night Heron
(3) BAEA - Bald Eagle
(4) Raptor species not identified
(5) $\log_{10}$EROD = 1.43+0.49$\log_{10}$PCB, n=30, p<0.0001)
(6) DCCO - Double-crested Cormorant
(7) Y=0.067X+13.1
(8) dose = 30 µg/g (10 µg/g of Aroclors 1242, 1248, 1254)
(9) TEF system, Safe (1993)
(10) no-observed-effect = EC_1

Table 4. TCDD Toxic Equivalence Factors (TEFs) for various PCDDs, PCDFs, and PCBs.

Chlorinated Compound	TCDD Toxic Equivalence Factor
PCDDs	
2378-TCDD	1.0
12378-PnCDD	0.5
123678-HxCDD	0.1
123789-HxCDD	0.1
123478-HxCDD	0.1
1234678-HpCDD	0.01
OCDD	0.001
PCDFs	
2378-TCDF	0.1
23478-PnCDF	0.5
12378-PnCDF	0.05
123478-HxCDF	0.1
234678-HxCDF	0.1
123678-HxCDF	0.1
1234678-HpCDF	0.01
1234789-HpCDF	0.01
non-ortho PCBs	
#77	0.0005
#126	0.1
#169	0.01
mono-ortho PCBs	
#118	0.0001
#105	0.0001

(1) PCDDs and PCDFs from Safe (1990); PCBs established by the WHO in Ahlborg *et al*. (1994).

6. <u>Chlorinated hydrocarbons detected in various indicator species</u>

The concentrations of DDE, PCBs, PCDDs and PCDFs detected in all samples are summarized in this section. Possible sources of contaminants are discussed. Geographic differences and temporal trends in the contaminant levels detected for each species are also discussed. The biological significance of the contaminant levels found in each species is evaluated. Comments are made on the population status of each species in the Fraser River Basin.

6.1. Great Blue Herons (*Ardea herodias*)

Great Blue Heron feed mainly in shallow water, and sometimes in meadows, on a wide variety of small fish, marine invertebrates and small mammals (Verbeek and Butler, 1989). Observations of herons foraging at Iona Island and Westham Island in 1991 showed that Pacific staghorn sculpin, starry flounder, and threespine stickleback were the principal prey items (Harfenist *et al*., 1993).

Great Blue Heron are excellent local indicator species for marine, estuarine, and aquatic environments. In coastal British Columbia, Great Blue Herons are resident throughout the year (Butler, 1991); for example Butler *et al*. (1995) determined that the average distance between 18 coastal colonies and their main foraging sites was 2.9 km (S.E. = 0.6). Since herons eat mainly first year age class fish (Whitehead *et al*., 1992; Harfenist *et al*., 1993), most of the contaminant burden in the fish prey will have been acquired locally. Therefore, the contaminants detected in the herons eggs are acquired from local sources.

6.1.1. *Concentrations of chlorinated hydrocarbons*

Great Blue Heron eggs were collected from seven colonies in the lower Fraser River Basin between 1977 and 1993. Chlorinated hydrocarbons were detected in all eggs. The major contaminants were PCBs, followed by DDE and HxCDD, PnCDD and TCDD.

<u>Total PCBs</u>

Concentrations of total PCBs detected in the eggs ranged from 0.1 to 18 mg/kg (Figures 3 and 4). In 1977, the total PCBs detected in eggs from the UBC colony were at least two-fold higher than the eggs collected from colonies at Nicomekl, Coquitlam and Pt. Roberts. Comparable residues were detected in eggs collected from five colonies in both 1989 and 1990 (less than 3.5 mg/kg). The concentration of total PCBs declined in a logarithmic fashion between 1977 and 1993.

The PCBs detected in the heron eggs during the late 1970s most likely originated from industrial sources in the lower Fraser River. The comparatively higher concentrations detected in the eggs from the UBC colony were likely due to the herons foraging near the Fraser River estuary, downstream of numerous discharges.

The steep initial reduction in total PCB residues between 1977 and 1983 likely resulted from regulations implemented in 1977 that prohibited the production of equipment

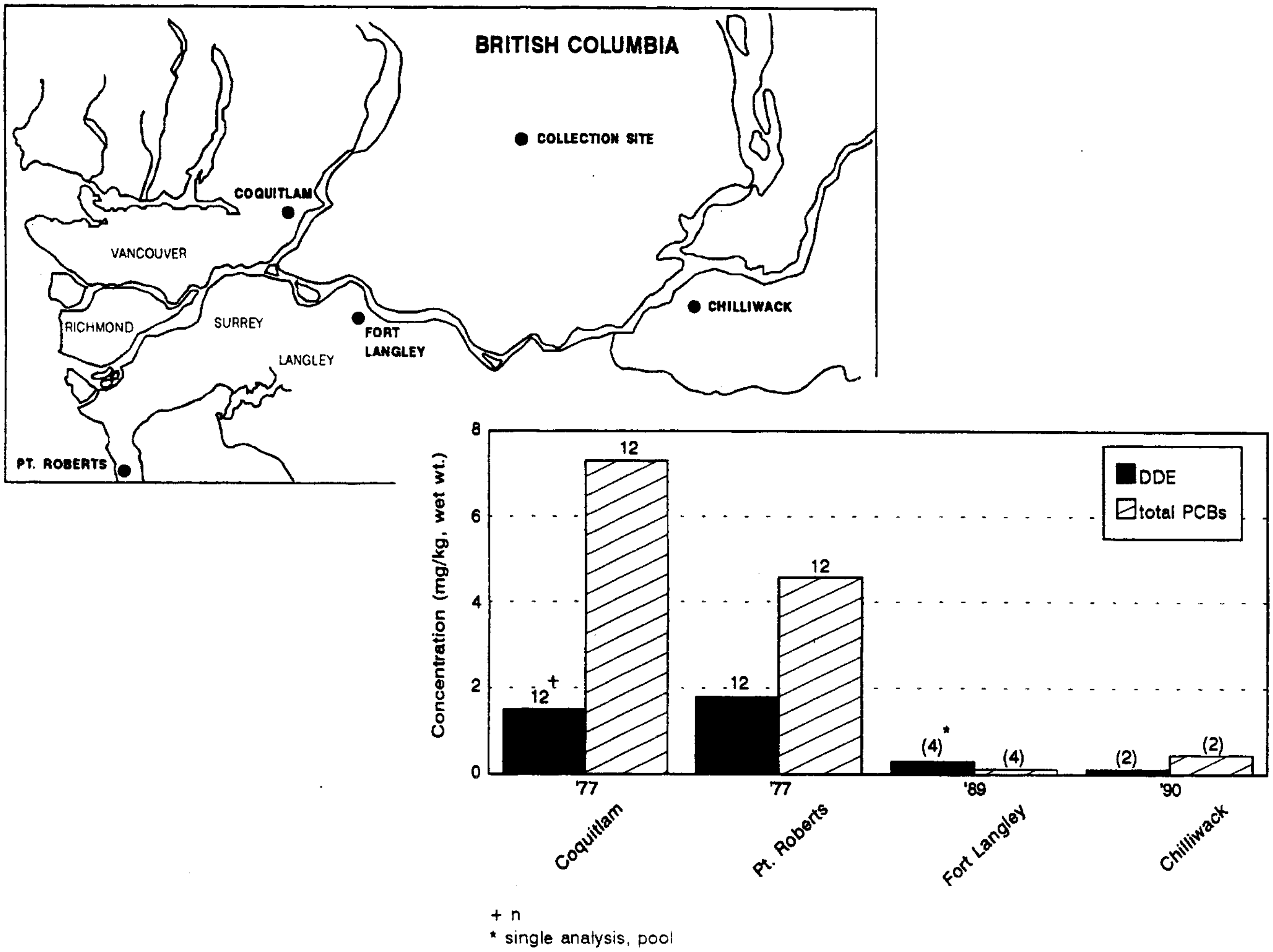


Figure 3. DDE and total PCBs (geometric means) in Great Blue Heron eggs: Fraser River Basin, 1977 to 1990.

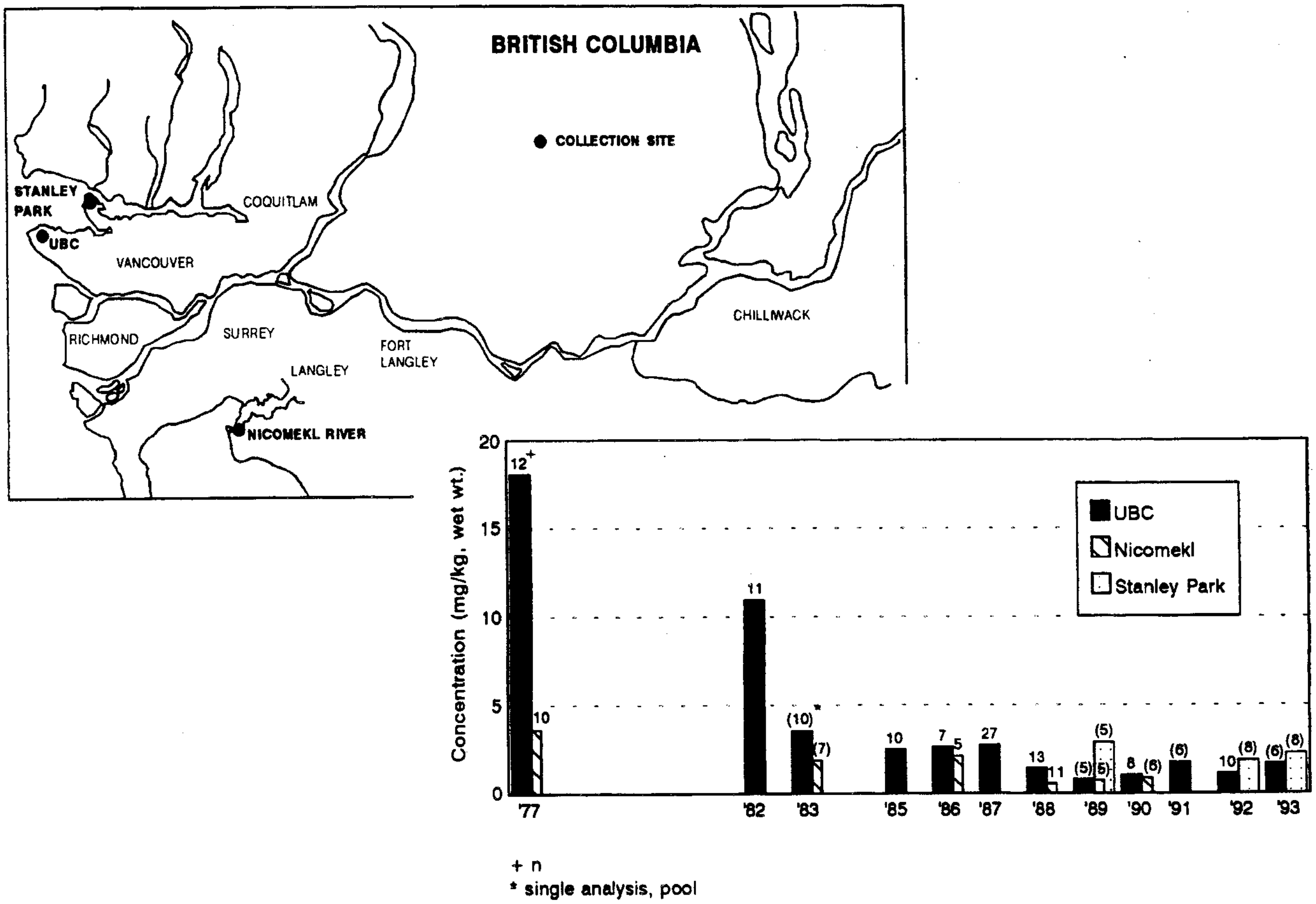


Figure 4. Total PCBs (geometric means) in Great Blue Heron eggs: UBC, Nicomekl and Stanley Park colonies, 1977 to 1993.

containing PCBs. The relatively consistent concentration of PCBs detected in eggs collected between 1983 and 1993 may reflect some ongoing low-level input and atmospheric deposition of PCBs currently circulating in local and world-wide ecosystems (Tanabe *et al*., 1987).

PCB levels greater than 100 ppm wet weight in liver are acutely toxic (Cooke *et al*., 1982). Reproductive effects have been observed in birds of prey whose eggs has greater than 50 mg/kg wet weight (Noble *et al*., 1993). Sub-lethal effects occur at much lower exposure levels; for example, total PCBs at 4.5 mg/kg were associated with reduced embryonic growth in Black-crowned Night Herons in San Francisco harbour (Hoffman *et al*., 1986). Rattner *et al*. (1994) measured enhanced enzyme levels (hepatic cytochrome P-450 cross reactive proteins, CYP1A; ethoxyresorufin-O-deethylase, EROD; benzyloxyresorufin-O-deethylase, BROD) with elevated PCB levels in embryos.

<u>DDE</u>

DDE residues measured in the eggs ranged from 0.2 to 2.5 mg/kg (Figures 3 and 5). Similar concentrations of DDE were measured in eggs collected from different colonies within the same year. Concentrations decreased over time except for the eggs collected from the UBC and Stanley Park colonies in 1993.

The major source of DDE found in heron eggs is most likely DDT used in agriculture in the 1960s. Comparable levels of DDE among colonies in 1977 suggests similar DDE exposure at heron colonies throughout the lower Fraser River Basin. The constant decline of DDE reflects the slow elimination of DDT from the local environment since its ban in early 1970s such as reported previously (Whitehead, 1989) and observed in eggs of fish-eating species from relatively DDE polluted environments in many parts of North America (Anderson *et al*., 1975; Mineau *et al*., 1984; Elliott *et al*., 1988). The comparably high levels of DDE measured in the eggs from the UBC and Stanley Park colonies in 1993 can not currently be accounted for, but monitoring efforts will continue.

Great Blue Herons in Oregon and Washington displayed a negative correlation between log DDE concentrations and eggshell thickness (Blus *et al*., 1980). Custer *et al*. (1983) determined that 8.0 mg/kg wet weight DDE was the critical threshold to cause eggshell thinning in Black-crowned Night Herons. The DDE concentrations presently detected in the Great Blue Herons in the lower Fraser River should not adversely affect their reproduction.

<u>PCDDs and PCDFs</u>

The major PCDDs detected in eggs collected from the UBC, Nicomekl and Stanley Park colonies were 123678-HxCDD, followed by 12378-PnCDD and 2378-TCDD (Figures 6, 7, and 8). In 1982, with the exception of 2378-TCDD, the PCDD and PCDF concentrations measured in eggs from the UBC colony were over three-fold higher than the levels detected in other samples. Between 1983 and 1990, PCDD and PCDF concentrations in eggs from the UBC colony were almost two-fold higher than eggs from either the Stanley Park or Nicomekl colonies. Since 1992, comparable levels of PCDDs and PCDFs have been detected among colonies. In general, the sum of PCDDs and PCDFs have decreased over

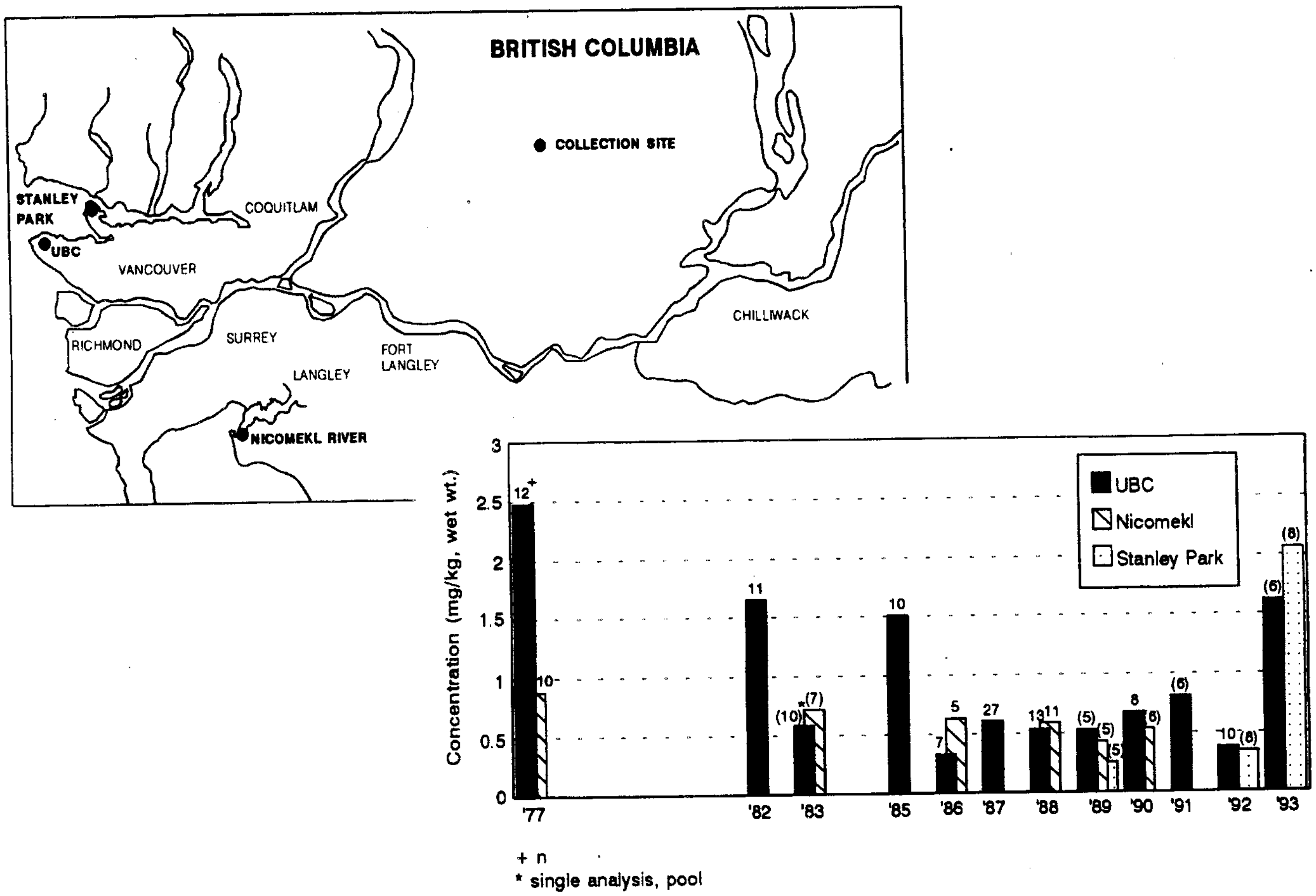


Figure 5. DDE (geometric means) in Great Blue Heron eggs: UBC, Nicomekl and Stanley Park colonies, 1977 to 1993.

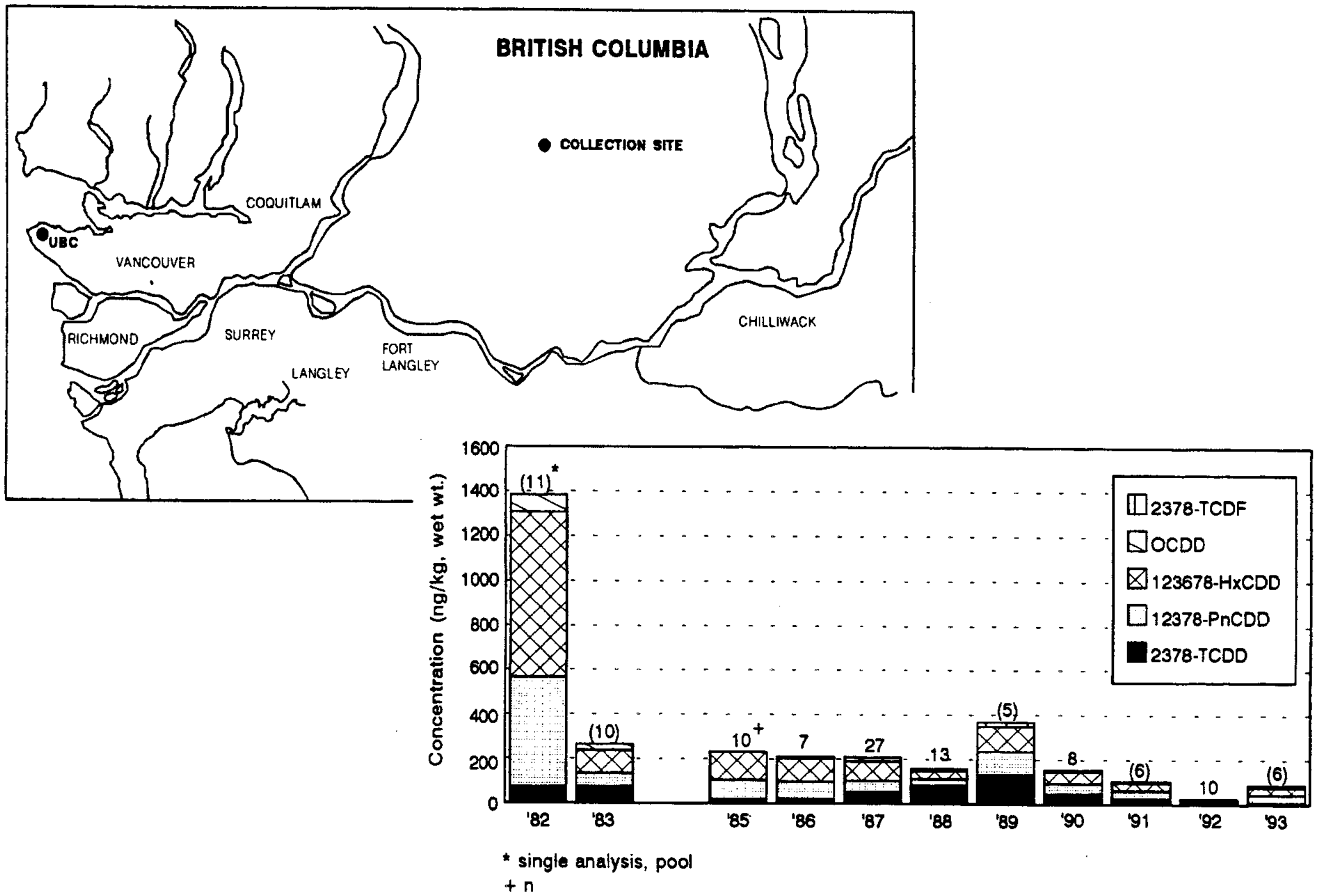


Figure 6. Major PCDDs and PCDFs (geometric means) in Great Blue Heron eggs: UBC colony, 1982 to 1993.

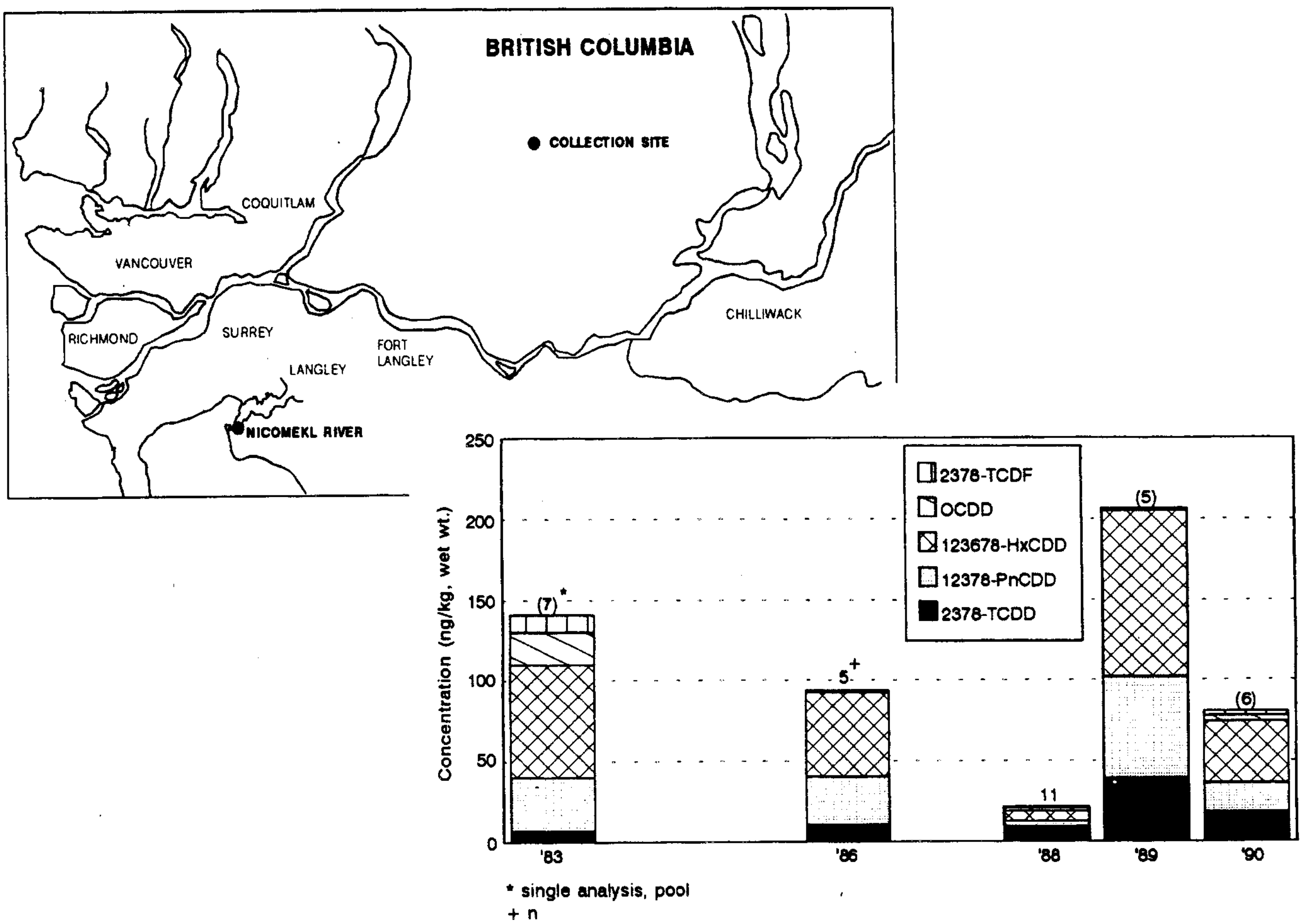


Figure 7. Major PCDDs and PCDFs (geometric means) in Great Blue Heron eggs: Nicomekl colony, 1983 to 1990.

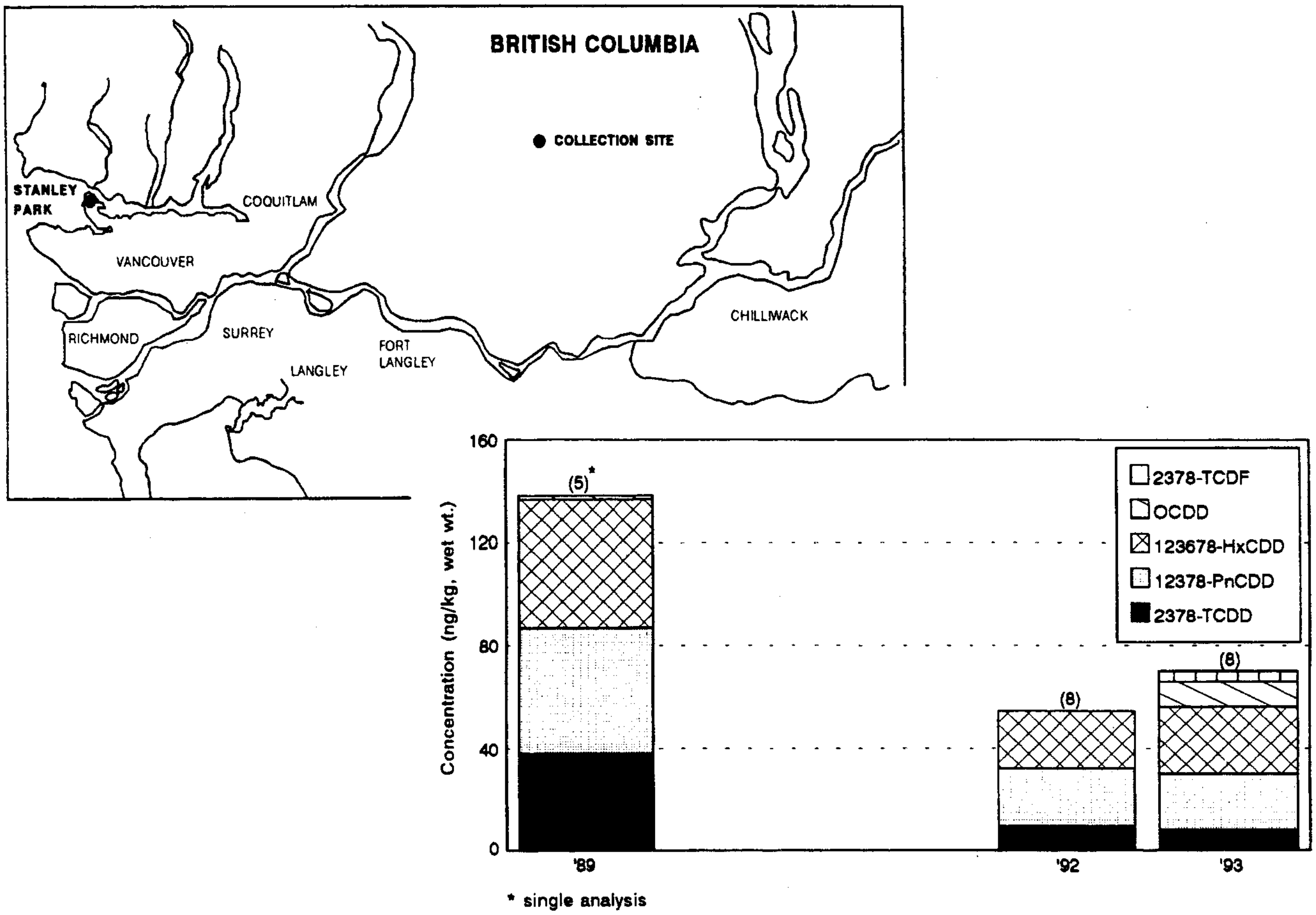


Figure 8. Major PCDDs and PCDFs (geometric means) in Great Blue Heron eggs: Stanley Park colony, 1989 to 1993.

time, although concentrations of 2378-TCDD increased between 1983 and 1989, after which they declined.

The types of PCDD and PCDF congeners detected in the eggs suggest multiple sources: pulp and paper mills; wood preservation facilities; and other sources such as municipal incinerators. In general, lower chlorinated compounds are predominately associated with pulp and paper mill effluent, moderately chlorinated compounds (5 to 7 Cls) with PCPs, and highly chlorinated compounds with incineration

The contaminant levels detected in the heron eggs also relate to both voluntary changes by industry and the implementation of various regulations. In 1987, pulp and paper mills began in eliminate defoamers, adjust the bleaching process, and some mills started substituting $Cl^{-}O_2^{++}$. The use of anti-sapstains was banned in 1989. CEPA regulations limiting the discharge and use of PCP-contaminated chips were implemented in 1992. Secondary treatment of effluent was also mandated under the Fisheries Act in 1992. The relationship between PCDD and PCDF levels in fish-eating birds exposed to pulp and paper mill contaminants are further discussed in Whitehead *et al*. (1992) and Elliott *et al*. (1995a).

Geographically, the highest concentrations of PCDDs and PCDFs were detected in the eggs from the UBC colony. The UBC colony is situated in the Fraser estuary where TCDDs and TCDFs in sediments from upstream mills are settling out. Higher-chlorinated compounds likely originated from chlorophenol sources located on the North Arm of the Fraser River immediately upstream from the colony.

A Great Blue Heron colony near the bleached kraft pulp mill at Crofton in the Strait of Georgia, British Columbia had complete reproductive failure in 1987, following an almost threefold increase in TCDD (66 to 210 ng/kg), as well as a smaller increase in HxCDD, levels in eggs (Elliott *et al*., 1989c). However, Elliott *et al*. (1989c) concluded the probable cause of failure could have been disturbance by humans and eagles, rather than TCDD-induced embryotoxicity, since the contaminant levels overlapped with those from 1986, when nest success was approximately 60%. Hart *et al*. (1991) found one third of heron hatchlings at the Crofton heron colony in 1988 exhibited subcutaneous edema, which has been associated with high concentrations of halogenated aromatic hydrocarbons (Gilbertson *et al*., 1991). Embryo weight and the weight of specific body organs have been negatively correlated with TCDD concentrations ($r>0.48$, $p<0.01$) (Hart *et al*., 1991; Sanderson *et al*., 1994b).Comparatively, EROD activity in heron eggs has been positively correlated with TCDD concentrations ($r=0.70$, $p<0.001$, $n=54$) (Bellward *et al*., 1990; Sanderson *et al*., 1994b). Sanderson *et al*. (1994b) also found that Great Blue Heron chicks showed no observable adverse effects (including EROD induction) when exposed to < 100 ng/kg 2378-TCDD.

<u>Total toxic burden (TEQs)</u>

The major contributors to the TCDD-like toxic burden in heron eggs collected from the lower Fraser River Basin are the non-ortho PCBs, 2378-TCDD, 12378-PnCDD, and 123678-HxCDD (Figures 9, 10, and 11). Comparison of residues detected in eggs collected

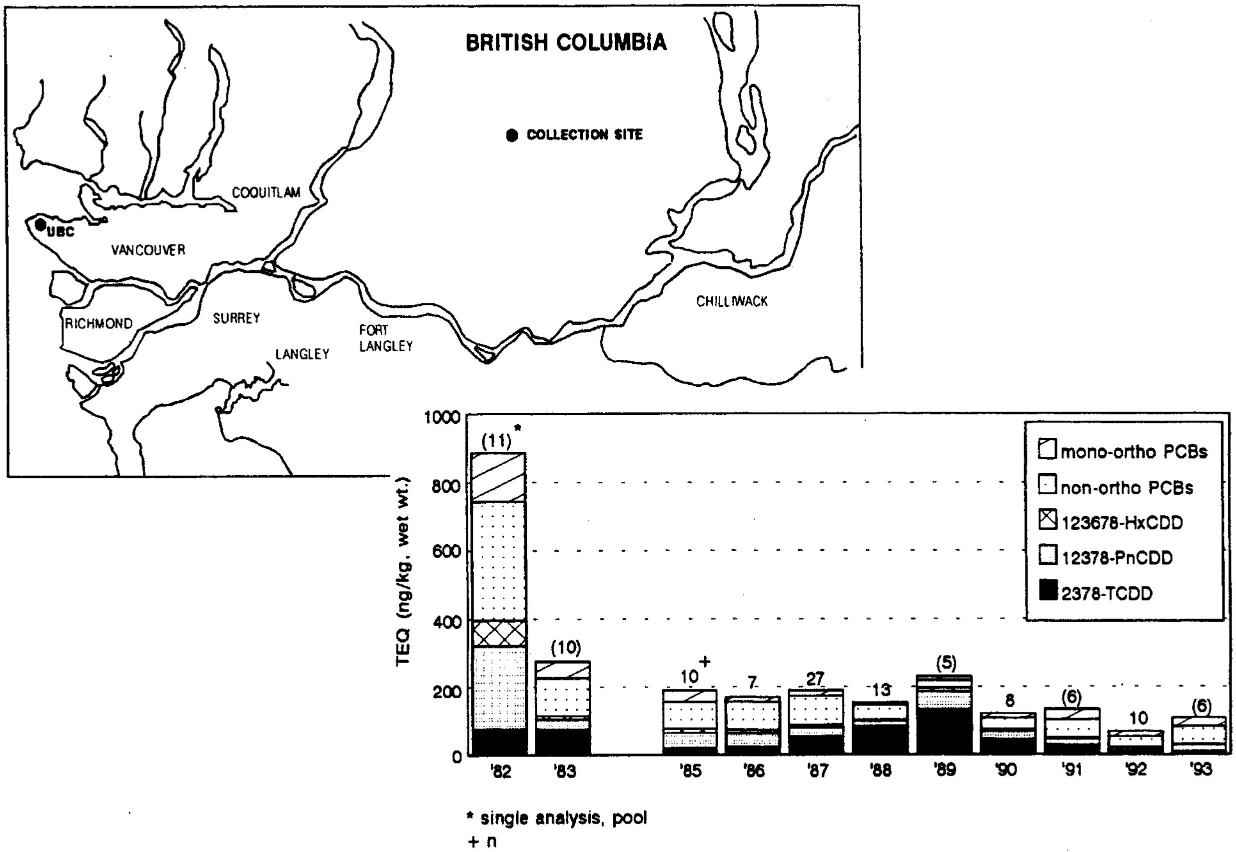


Figure 9. Toxic equivalents (TEQs) for major PCDDs, PCDFs and PCBs in Great Blue Heron eggs: UBC colony, 1982 to 1993.

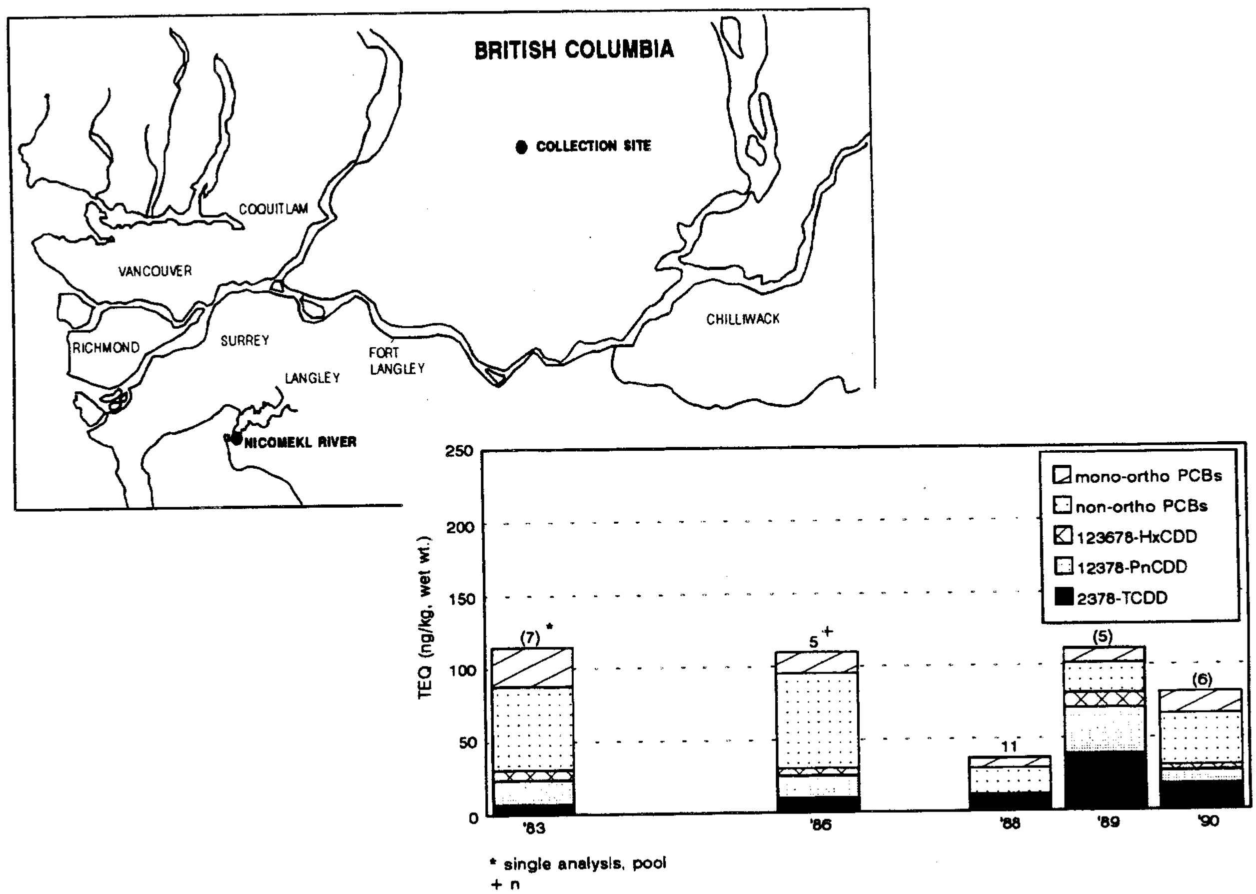


Figure 10. Toxic equivalents (TEQs) for major PCDDs, PCDFs and PCBs in Great Blue Heron eggs: Nicomekl colony, 1983 to 1990.

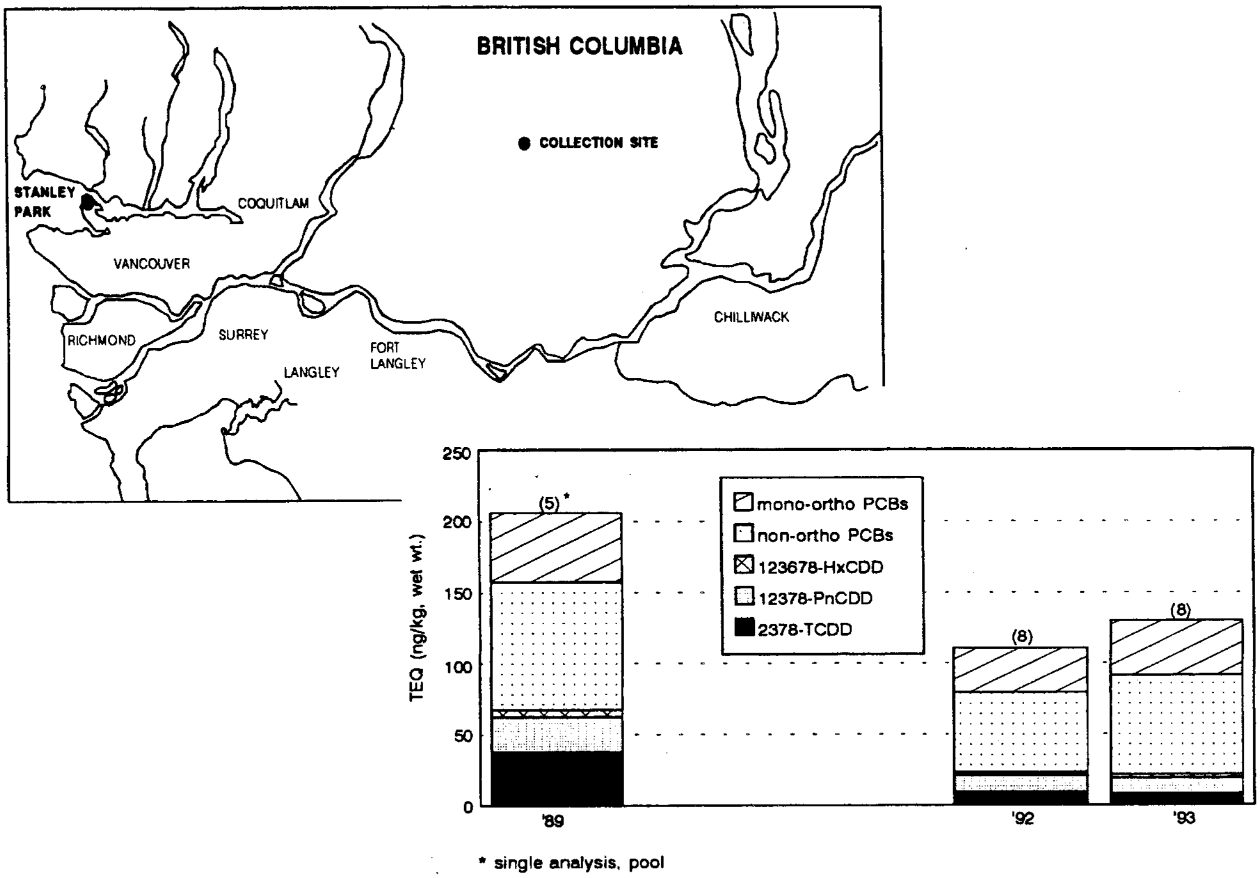


Figure 11. Toxic equivalents (TEQs) for major PCDDs, PCDFs and PCBs in Great Blue Heron eggs: Stanley Park colony, 1989 to 1993.

within the same year showed that the most contaminated colony was Stanley Park, followed by UBC and then Nicomekl. The total toxic burden in eggs from all colonies appear to be quite stable over time particularly since 1982. Efforts are still ongoing to establish the no-observed-effect-limit (NOEL) and lowest-observed-effect-limit (LOEL) of TEQs for Great Blue Herons based on existing data from south-western British Columbia.

6.1.2. *Population status*

There is some data on reproductive success of Great Blue Herons in coastal British Columbia since 1977 (Forbes *et al*., 1985; Simpson *et al.*; 1987; Butler, 1989). However, these studies overestimated fledging success because they used data only from successful nesting attempts (Butler, 1992). Based on the fledging success of 4,510 Great Blue Heron nests in 31 colonies on the Pacific Coast of British Columbia between 1987 and 1992, Butler *et al*. determined an average of 1.7 herons fledged per nesting attempt.

The number of active heron nests in the Lower Mainland have apparently increased since the early 1980s (599 active nests in the early 1980s, 787 nests in 1992, 940 nests in 1993, and 961 nests in 1994) (Gebauer, 1995). Although there is still insufficient data to determine if heron populations in the Fraser River estuary are stable (Hitchcock, pers. comm.), based on current data, the contaminant levels detected in heron eggs are not effecting overall breeding rates.

6.2. Double-crested Cormorants (*Phalocrocorax auritus*) and Pelagic Cormorants (*P. pelagicus*)

Coastal breeding Double-crested and Pelagic Cormorants prefer small demersal fish such as the anchovy, surfperch, silverside and drum families which school near the bottom of shallow feeding grounds (Robertson, 1974; Ainley *et al*., 1981). Pelagic Cormorants select small non-schooling fish of rocky reefs and bottoms feeding primarily on inshore neritic waters whereas Double-crested Cormorants feed on schooling prey that occurred from the surface to near, but not on, bottoms having no relief (Ainley *et al*., 1981). In the lower Fraser River, breeding Pelagic Cormorants are only found at Prospect Pt. and Second Narrows whereas Double-crested Cormorants breed only Delta. Detailed locations of their seasonal feeding areas are not known.

Double-crested and Pelagic cormorants are year-round residents on the B.C. coast. Since both species are sensitive to disturbances, shifts in colony sites are common (Seigel-Causey and Litvinenko, 1993). These movements may alter their foraging locations, potentially affecting the specific contaminants detected in their eggs. Seasonal movement patterns and breeding activities of Double-crested Cormorants in the Strait of Georgia are currently being studied by Moul *et al*. (1995) and Sullivan (1995).

6.2.1. *Concentrations of chlorinated hydrocarbons*

The locations of cormorant colonies from which eggs were collected are shown in Figure 12. Double-crested Cormorant eggs were collected from the Fraser River estuary between 1985 and 1992; Pelagic Cormorant eggs were sampled from the colony at Prospect Pt. in 1989. In general, the contaminant levels were similar for the two species and concentrations in the Double-crested Cormorant eggs decreased over time. The most abundant chlorinated hydrocarbons were PCBs, followed by DDE and HxCDD.

<u>Total PCBs</u>

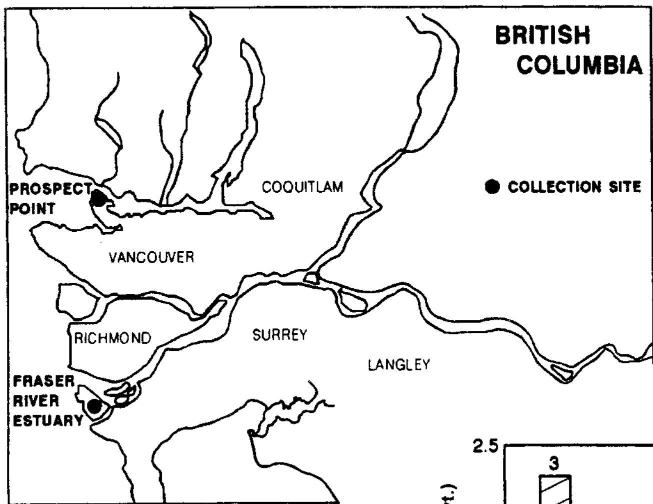
Total PCB burdens in Double-crested Cormorant eggs from the Fraser River estuary were higher in 1985 and 1988 (average 2.3 mg/kg) than in 1990, 1991 and 1992 (average 0.8 mg/kg). Concentrations in the Pelagic Cormorant eggs from Prospect Pt. in 1989 were about two-fold greater than the Double-crested Cormorants in 1990 to 1992 but lower than the levels detected in the Double-crested Cormorant eggs collected from the Fraser River estuary in 1985 and 1988 (Figure 12). Pelagic Cormorants generally have lower organochlorine levels than the Double-crested Cormorants (Elliott *et al*., 1989a); thus, the relatively elevated PCBs at the Prospect Pt. colony may indicate higher environmental level of PCBs in Burrard Inlet than in the Fraser River estuary. Recent PCB levels in Great Blue Heron are also higher at the Stanley Park colony which forage partially in Burrard Inlet than the UBC colony which forage the Fraser estuary. The slow rates of decline in PCB concentrations reflect the persistence of these compounds as well as ongoing low-level inputs and atmospheric deposition of PCBs currently circulating in local and world-wide ecosystems.

Double-crested Cormorant egg mortality has been positively correlated with TCDD-Eqs (derived from EROD activity of treated H4IIE rat hepatoma cells) (Tillitt *et al*., 1992) Sanderson *et al*. (1994a) noted reduced yolk weight and EROD induction in PCB contaminated Double-crested Cormorant eggs from areas of the Strait of Georgia.

<u>DDE</u>

Like total PCBs, concentrations of DDE in the Double-crested Cormorant eggs from the Fraser River estuary were higher in 1985 and 1988 (average 0.5 mg/kg) compared to eggs collected in 1990, 1991, and 1992 (less than 0.22 mg/kg). DDE levels in the Pelagic Cormorant eggs collected from Prospect Pt. in 1989 were similar to the concentrations detected in the Double-crested Cormorant eggs collected between 1990 and 1992 (Figure 12).

The low concentrations and comparable levels of DDE detected among cormorant colonies suggest similar DDE exposure throughout the lower Fraser River Basin. The constant decline shows the slow elimination of DDT from the local environment since its ban in the 1970s such as reported previously (Whitehead, 1989) and observed in eggs from fish-eating species from relatively DDE polluted environments in many parts of North America (Anderson *et al*., 1975; Mineau *et al*., 1984; Elliott *et al*., 1989c).



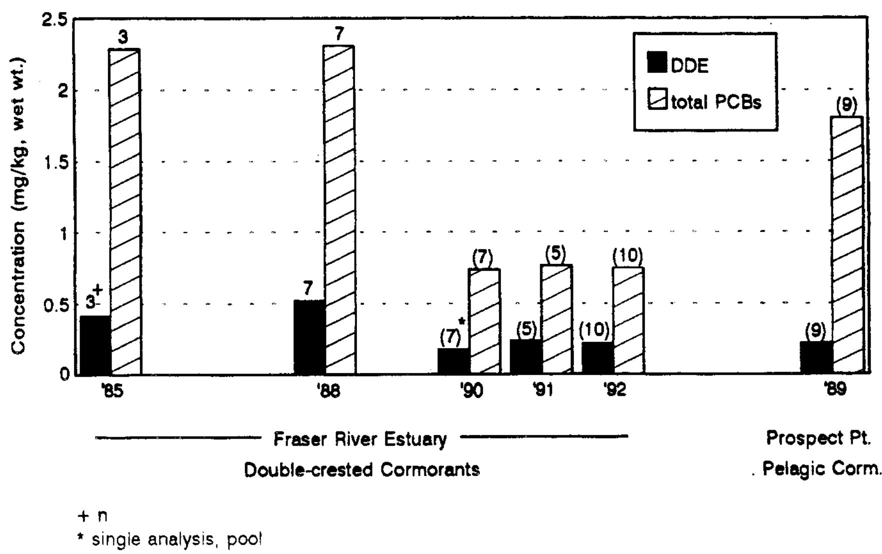


Figure 12. DDE and total PCBs (geometric means) in Double-crested Cormorants eggs: Fraser River Estuary, 1985 to 1992 and Pelagic Cormorant eggs: Prospect Pt., 1989.

PCDDs and PCDFs

The major PCDD congeners detected in the cormorant eggs were 123678-HxCDD, 12378-PnCDD and 2378-TCDD (Figure 13), exactly as found in herons (Section 6.1). In the Fraser River estuary, total concentrations of PCDDs were comparatively high in 1985 and 1988, 290 and 170 ng/kg, respectively. From 1990 to 1992, total levels fell to less than 52 ng/kg. Total PCDD levels in the Pelagic Cormorant eggs collected from Prospect Pt. in 1989 were similar to levels in the Double-crested Cormorant eggs collected from the estuary in 1985. This sharp drop after 1990 reflects both changes in the bleaching process at the upstream kraft pulp mills and the banning of chlorophenols as anti-sapstain agents.

Total toxic burdens (TEQs)

The largest contributors to the toxic burden in cormorant eggs are the non-ortho PCBs and 2378-TCDD. The TEQs of eggs collected from the Fraser River Delta are comparable between colonies and steadily decreased between 1985 and 1992 (Figure 14). The biological significance of contaminants currently detected in the cormorants is not known. However, discussions are ongoing to determine the NOEL and LOEL values in cormorants based on data collected from south-western British Columbia.

6.2.2. *Population status*

Double-crested Cormorant populations in the Strait of Georgia have increased from 203 nesting pairs in 1960 to 1,981 pairs in 1987, whereas Pelagic Cormorant populations have been fairly stable with 2,149 nesting pairs in 1975 and 2,356 in 1987 (Vermeer *et al*., 1989a). These population trends suggest the level of contaminants currently detected in cormorant eggs have not adversely affected their overall reproductive success in the Strait of Georgia.

6.3. Bald Eagles (*Haliaeetus leucocephalus*)

Bald Eagles are opportunistic predators and scavengers. On the Pacific coast, their diet consists of primarily marine and estuarine birds and fish (Knight *et al*., 1991). Vermeer *et al*. (1989b) found that the prey items located below Bald Eagle nests in the Gulf Islands were primarily birds (52%) followed by fish (34%), marine invertebrates (12%) and mammals (3%). However, this data, based on findings below nests, may be biased towards birds and mammals as fish remains are less persistent. Eagles may acquire high concentrations of contaminants due to the biomagnification of toxins through the food chain and their tendency to select impaired and potentially contaminated prey (Noble *et al*., 1993).

Telemetry studies conducted in Alaska showed that resident birds do not move further than 160 km from their nest, even during the winter (Schempf, pers. comm.). Therefore, the contaminants detected in the eggs of resident eagles are mainly from local sources, except where vectored through non-resident prey.

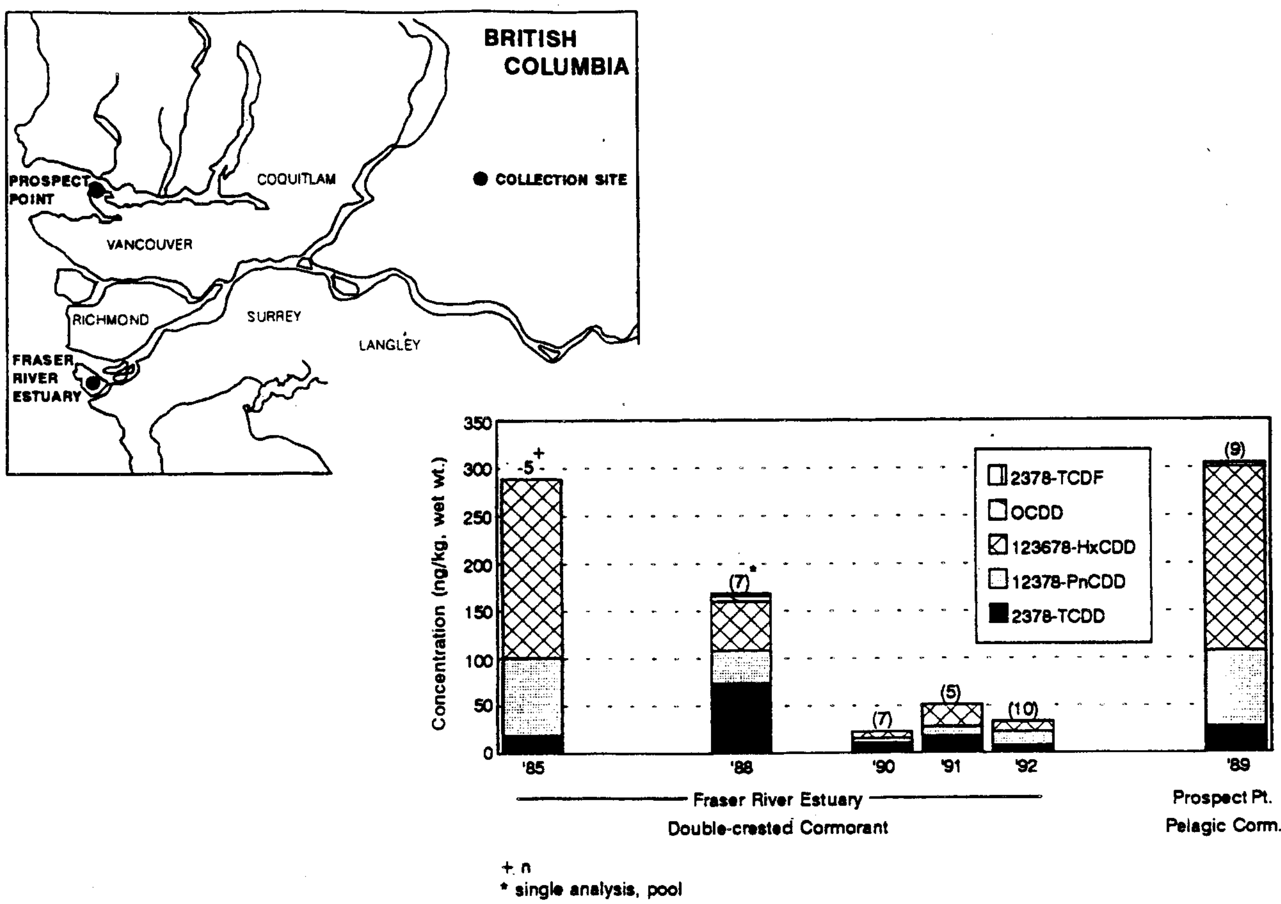


Figure 13. Major PCDDs and PCDFs (geometric means) in Double-crested Cormorant eggs: Fraser River Estuary, 1985 to 1992 and Pelagic Cormorant eggs: Prospect Pt., 1989.

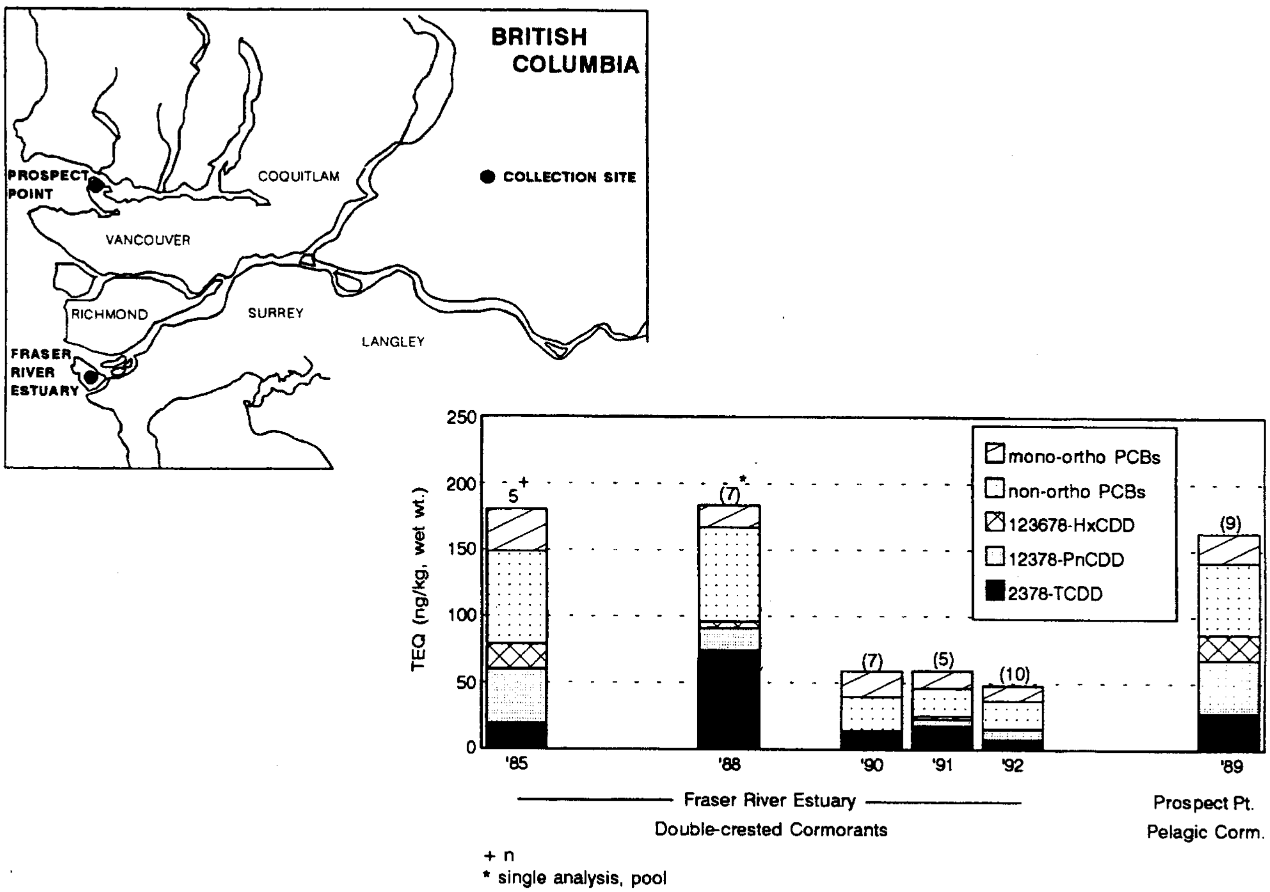


Figure 14. Toxic equivalents (TEQs) for major PCDDs, PCDFs and PCBs in Double-crested Cormorant eggs: Fraser River Estuary, 1985 to 1992 and Pelagic Cormorant eggs: Prospect Pt., 1989.

6.3.1. *Concentrations of chlorinated hydrocarbons*

Chlorinated hydrocarbons were detected in each of the six Bald Eagle eggs collected in the lower Fraser River Basin in 1990 and 1991. The most prevalent contaminants were DDE and PCBs followed by 2378-TCDD and 2378-TCDF.

<u>DDE</u>

Eggs collected in the vicinity of the Fraser River estuary (Annacis Island, Brunswick Pt.) contained higher levels of DDE than eggs collected further upstream at Island 20, Cheam Island and Agassiz Bridge (Figure 15). One exception is the egg collected from Chahalis Flats which although located upstream of the Fraser River estuary, also had the highest DDE residues (4.9 mg/kg). The DDE residues in eagle eggs reflects agricultural use of DDT in the lower Fraser. DDE levels in Great Blue Heron eggs were also highest at colonies in the Fraser Delta (Elliott *et al.*, 1989c; Whitehead, 1989).

A number of studies have shown that DDE adversely affects the breeding success and productivity of Bald Eagles. Wiemeyer *et al.* (1993) reported production of young was normal when eggs contained greater than 3.6 µg/g DDE wet weight, was nearly halved between 3.6 to 6.3 µg/g, and halved again when concentrations exceeded 6.3 µg/g. Although Bald Eagles appear to be less sensitive to DDE-induced shell thinning than a number of other species (Wiemeyer *et al.*, 1988), 15% egg shell thinning has been associated with 16 µg/g DDE in eggs (Wiemeyer *et al.*, 1993). Based on the results of these studies, it is unlikely that current DDE residues would cause serious shell thinning in Bald Eagles breeding in the Fraser River estuary. However, Elliott *et al.* (1996), based on extrapolation from trends in heron eggs, estimated that average DDE levels in eagle eggs in the late 1970s would have been about 25 mg/kg, which would have effected productivity.

<u>Total PCBs</u>

The highest concentration of PCBs, 6.2 mg/kg, was detected in the egg from Annacis Island (Figure 15). In general, eggs collected near the Fraser River estuary contained higher levels of PCBs compared to eggs collected further upstream. One exception was the egg collected from Chahalis Flats which had higher PCB levels than the egg collected from Brunswick Pt. The PCBs detected in the eagle eggs most likely originated from local industrial sources which accumulated in the food chain. The geographic distribution observed in the eggs generally reflects areas where PCBs have been more heavily used.

<u>PCDDs and PCDFs</u>

The major PCDDs and PCDFs in eagle eggs were 2378-TCDD and 2378-TCDF followed by 123678-HxCDD and 12378-PnCDD (Figure 16). Notably higher concentrations of PCDDs and PCDFs were measured in the eggs from Annacis Island and Chahalis Flats compared to other nesting sites.

Upstream pulp mills and heavy past use of chlorophenol wood preservatives at industrial sites account for the high levels of 2378-TCDD and TCDF detected in the eggs. Since 2378-TCDF should be cleared quickly from the body (Elliott *et al.*, 1996), the elevated

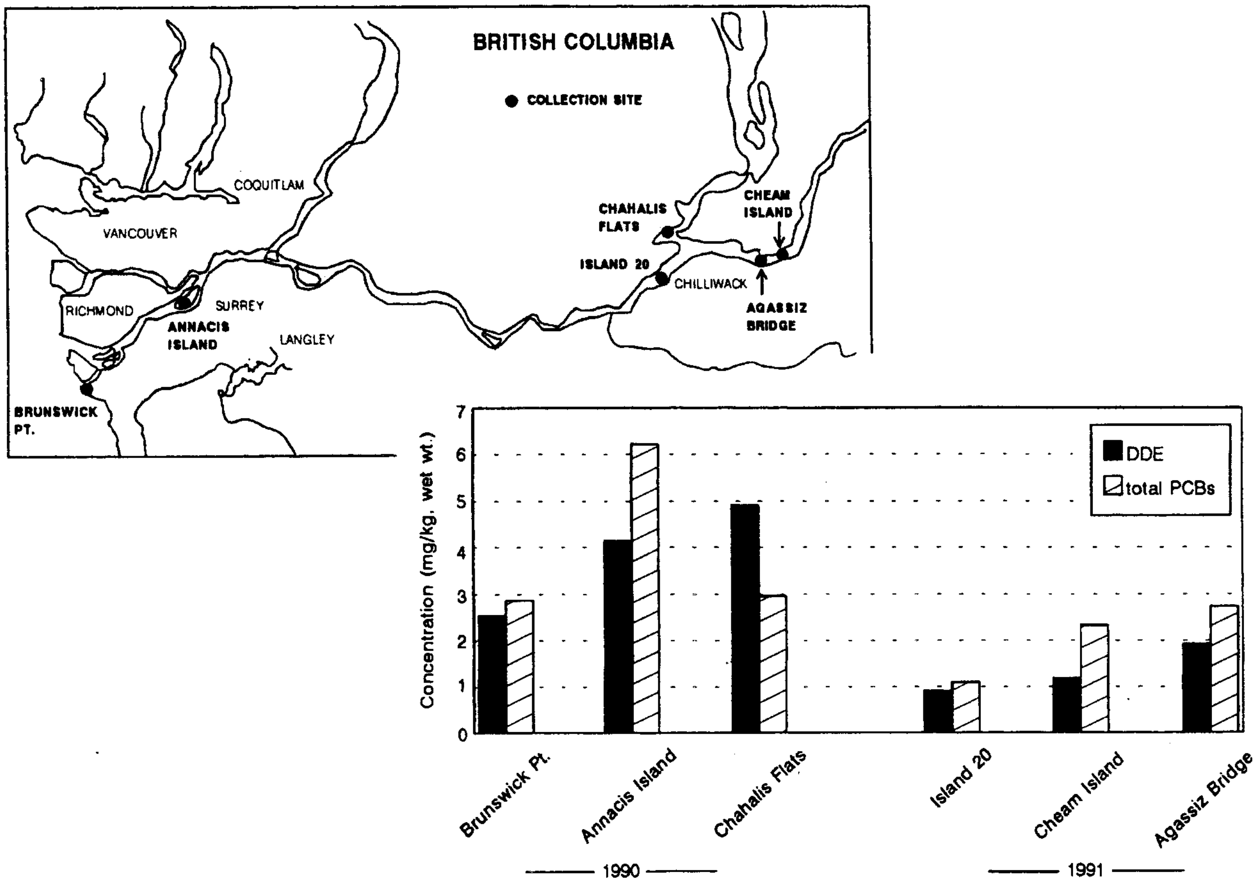


Figure 15. DDE and total PCBs in individual Bald Eagle eggs: Lower Fraser River, 1990 and 1991.

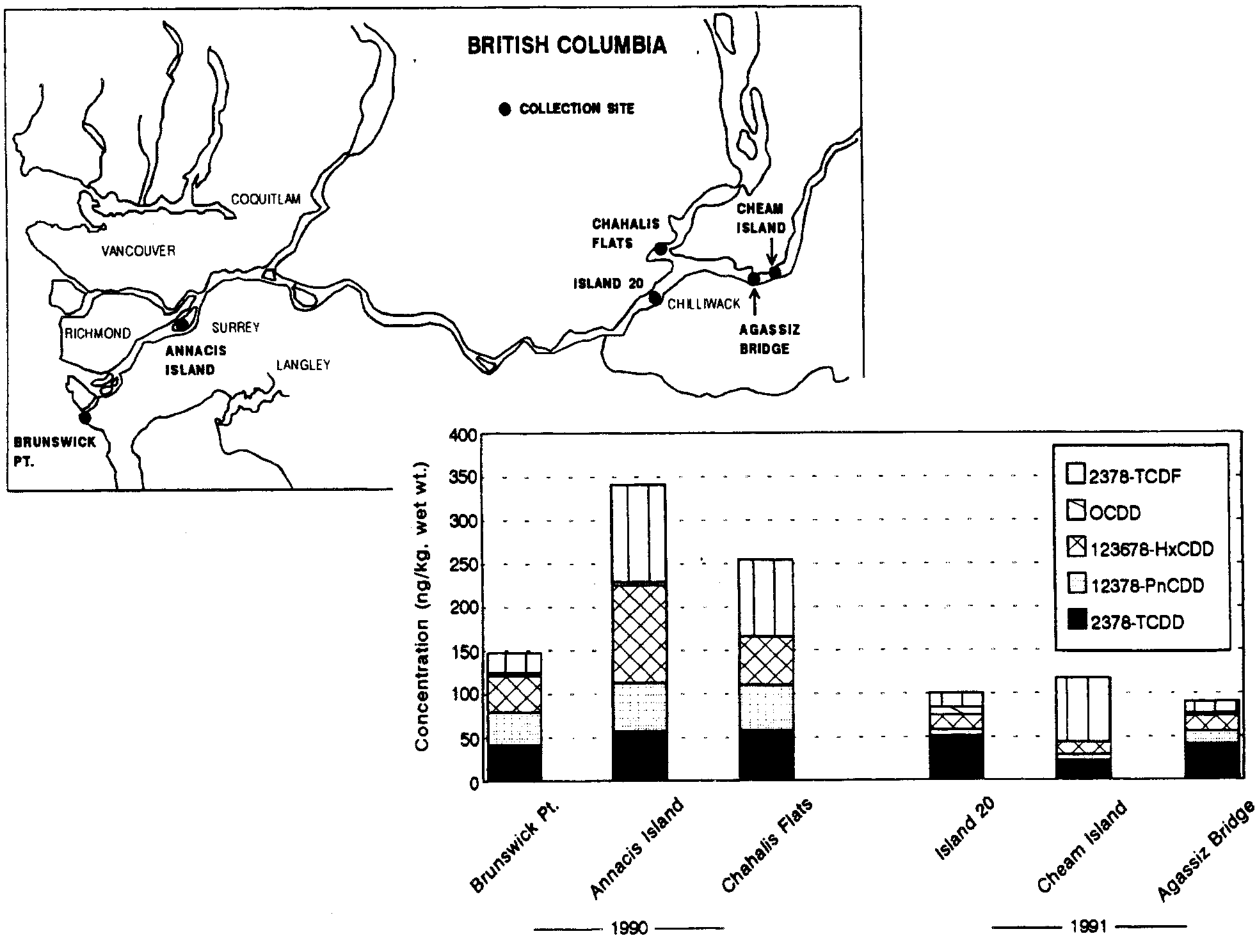


Figure 16. Major PCDDs and PCDFs in individual Bald Eagle eggs: Lower Fraser River, 1990 and 1991.

levels in the eggs indicate the deposition of lipid soluble contaminants from recent exposure, likely from consuming prey which was contaminated by local pulp mill discharges.

Total toxic burden (TEQs)

The largest contributor to the toxic burden observed in the Bald Eagle eggs was 2378-TCDD followed by non-ortho PCBs (Figure 17). The biological effects of PCDDs, PCDFs, and PCBs on Bald Eagle chicks on the southern coast of British Columbia have been discussed in detail by Elliott *et al*. (1996). Although no significant correlation was found between PCDD, PCDF, and PCB levels in Bald Eagle eggs and morphological, physiological, or histological parameters such as chick growth, edema, or density of thymic lymphocytes, positive correlations were found with the CYP1A, EROD, and BROD activities in Bald Eagle chicks and 2378-TCDD, 2378-TCDF and toxic equivalents ($TEQs_{WHO}$) in egg yolk sacs. Using hepatic CYP1A induction as a biomarker, a NOEL of 100 ng/kg and a LOEL of 210 ng/kg $TEQs_{WHO}$ on a whole weight basis have been suggested for Bald Eagle chicks.

6.3.2. *Population status*

Historically, DDE has been primarily responsible for reductions in shell thickness and productivity of Bald Eagle populations in North America from the 1950s to the 1970s. As DDE residues have declined, reproductive success and hence the size of some Bald Eagle populations have substantially improved (United States Fish and Wildlife Service, 1990). DDE levels have likely declined substantially in eagle eggs. However, based on extrapolation from trends in heron eggs, Elliott *et al*. (1996) estimated average DDE levels in eagle eggs in the late 1970s would have been about 25 mg/kg, which would have effected productivity. Bald Eagle productivity in the lower Fraser Valley at Delta and in the Strait of Georgia is well above bare levels needed to sustain a stable population (Elliott, 1995). Current productivity data suggests the possible adverse effects of chlorinated hydrocarbons on eagle reproductive success may be out-weighed by other environmental factors such as an increase in important prey items (Vermeer *et al*., 1989b). Although other toxicants such as lead (Elliott *et al*., 1992) and anticholinesterase pesticides (Elliott *et al*., 1995b) are causing substantial mortality among eagles wintering in the Fraser Delta, the existing population combined with high rates of breeding success means that the population is still stable (Elliott *et al*., 1995b).

6.4. Osprey (*Pandion haliaetus*)

Osprey are almost exclusively piscivorous, feeding mainly on slow-swimming fish at the surface. Although opportunistic, the bulk of their diet usually is composed of only several species of fish (Poole, 1989). The principal fish consumed by Osprey on the South Thompson and Thompson Rivers are whitefish and suckers (Machmer and Steeger, 1992). Osprey may acquire contaminants that have bioaccumulated in the foodchain since they consume some of the larger fish species. The Osprey which breed in the upper Fraser River winter in the tropics from the United States gulf coast states to Ecuador. Therefore, Ospreys could also acquire contaminants during migration or on these wintering areas in Latin America.

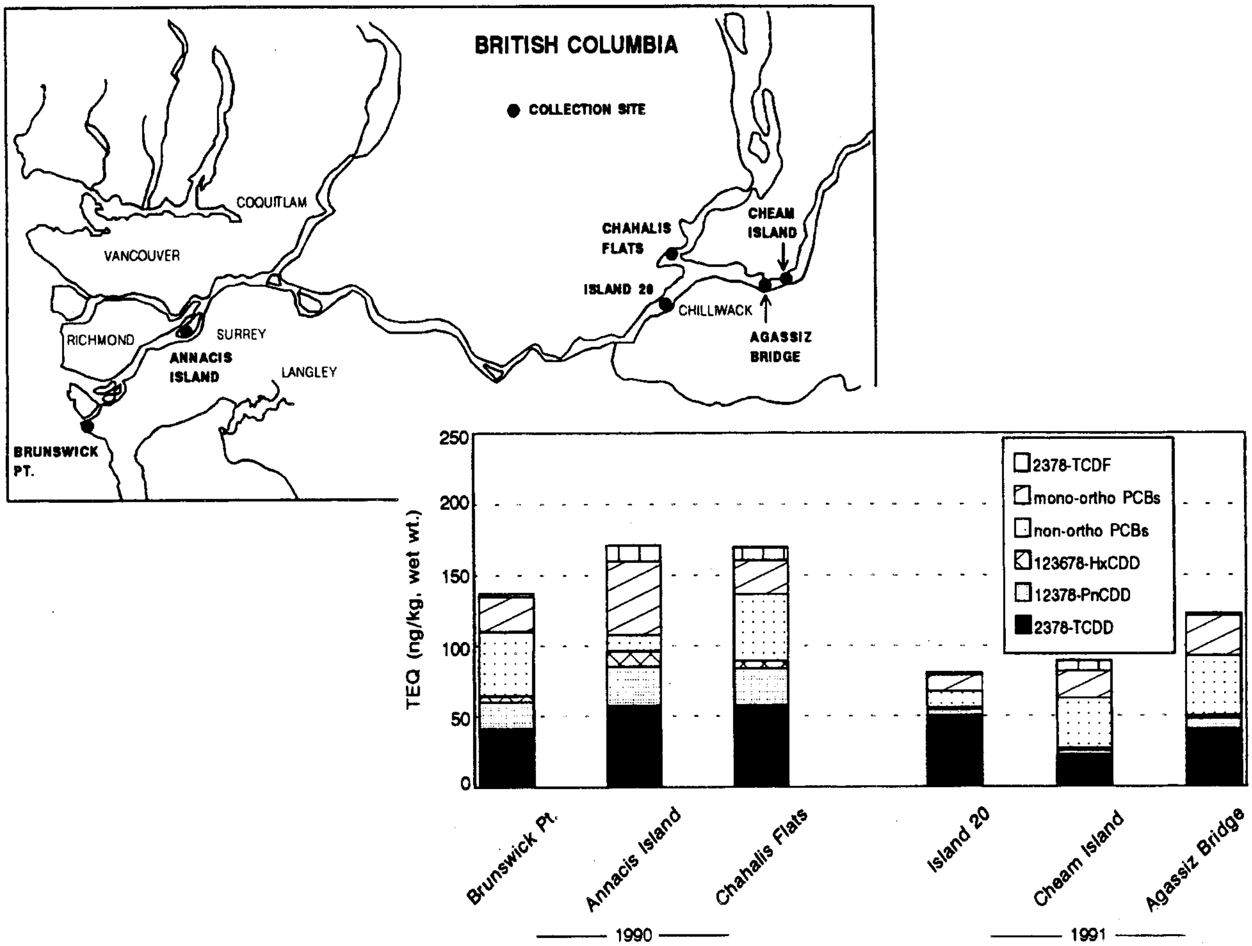


Figure 17. Toxic equivalents (TEQs) for major PCDDs, PCDFs and PCBs in individual Bald Eagle eggs: Lower Fraser River, 1990 and 1991.

6.4.1. *Concentrations of chlorinated hydrocarbons*

In 1991, eggs were collected from nests upstream and downstream of the pulp mill on the Thompson River near Kamloops, and downstream of the mills at Quesnel on the Fraser River. In 1992, nests on the Thompson and Fraser Rivers were resampled as well as nests on the Nechako River about 100 km west of the pulp mills at Prince George. Chlorinated hydrocarbons, predominantly DDE, PCBs and OCDDs were detected in each of the eggs.

<u>DDE</u>

DDE concentrations measured in the Osprey eggs were relatively consistent among nesting sites; average concentrations ranged from 1.1 to 2.1 mg/kg (Figure 18). However, variability was extremely great within nesting sites, ranging at some sites from 0.7 to 14 mg/kg. Of the 35 nests sampled on the Fraser River, 6 nests (17%) had DDE levels greater than the critical level for some eggshell thinning (4.2 mg/kg). Similar trends were observed in Osprey eggs collected in the Columbia River Basin in 1991 and 1992, with eggs from 9 of the 40 nests sampled (23%) having DDE burdens greater than the critical threshold value.

DDT is still used in Latin American countries. Therefore, Osprey could be acquiring DDE from prey consumed at their southern winter feeding grounds. However, it is possible that the Osprey have accumulated some of their burden from residues persisting in the local environment from DDT used during the 1950s and 1960s.

A number of studies have shown that DDE adversely affects Osprey productivity. Osprey eggs routinely break when shells are about 16% thinner than normal (Poole, 1989). Wiemeyer *et al*. (1988) found 15% eggshell thinning was associated with 4.2 mg/kg DDE. Eggs with DDE concentrations exceeding 5 to 10 mg/kg will also result in hatching failure due to improper development of the embryo (Poole, 1989). The high concentration of DDE measured in some of the eggs could have some effect on individual hatching success.

<u>Total PCBs</u>

Concentrations of PCBs detected in Osprey eggs collected from various nesting sites in the upper Fraser River in 1991 and 1992 ranged from 0.13 to 0.42 mg/kg (Figure 18). Variability among individual nests within a nesting site was great, ranging at some sites from 0.055 to 1.1 mg/kg. The lowest level was measured in eggs from the Nechako River. Although residues were consistently higher in 1992 for the nests that were sampled in both 1991 and 1992, temporal trends can not be ascertained from data collected over two years. PCB levels in Osprey eggs collected in the Fraser River system are much lower than those collected in the Columbia River system (unpublished data).

As PCBs are industrial contaminants, they are probably mainly from local sources rather than from their wintering grounds in Latin America.

Based on published data, Ospreys appear to tolerate high PCB exposure with no observable effect on reproductive success. For example, PCB levels in eggs from an Osprey colony in Massachusetts averaged 25 mg/kg yet both productivity and adult survival in the colony were above normal levels (Poole, 1989).

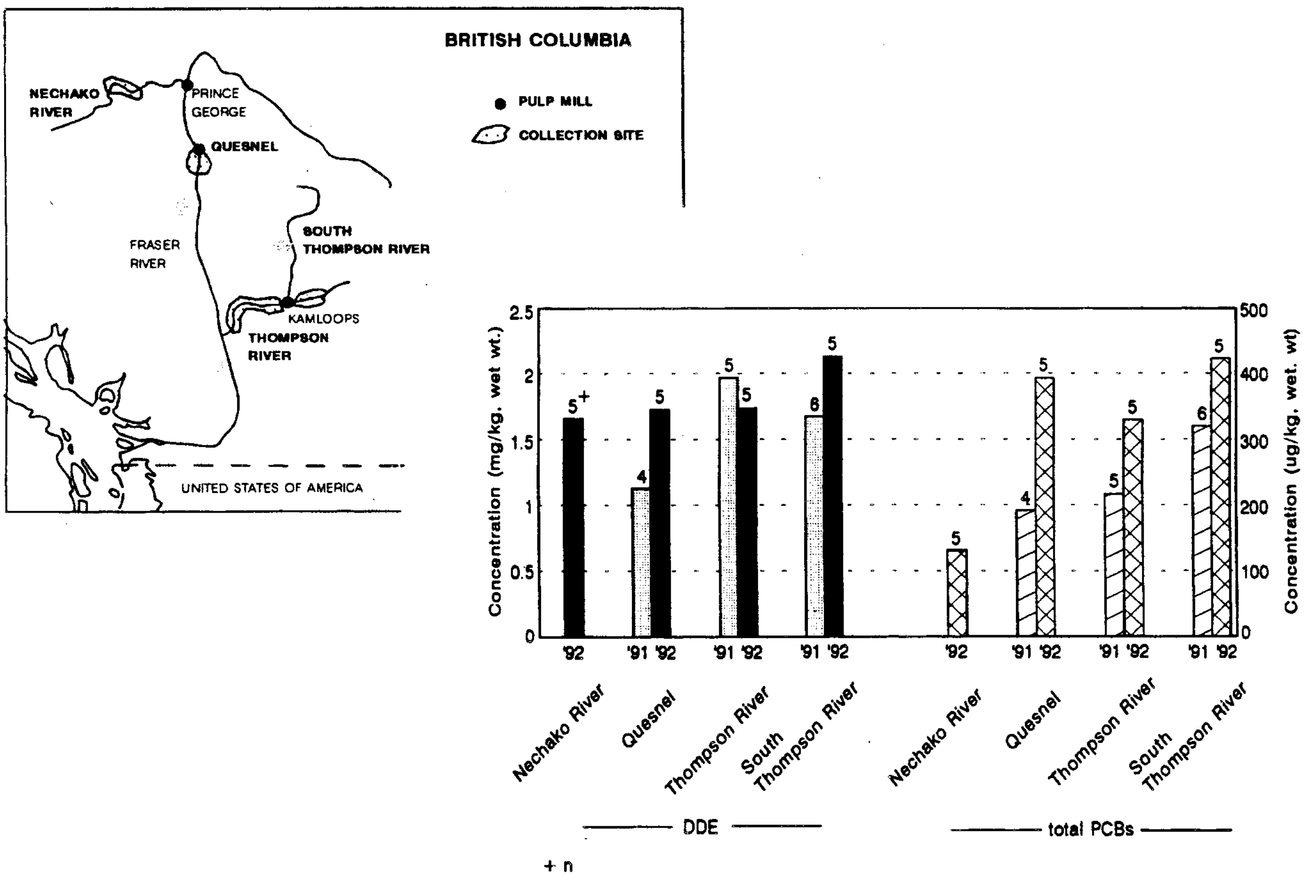


Figure 18. DDE and total PCBs (geometric means) in Osprey eggs: Nechako, Thompson and Fraser Rivers, 1991 and 1992.

PCDDs and PCDFs

The principle PCDDs detected in the Osprey eggs were OCDDs (Figure 19); eggs from some of the nests (about 20% of the total) had unusually high levels of OCDDs and HpCDDs. Concentrations in eggs collected from the South Thompson River were more than two-fold higher than other nesting sites that had levels ranging from 60 to 170 ng/kg. Comparatively high levels of 2378-TCDD were detected in the eggs from the Thompson River relative to other sampling sites; the highest concentration of 2378-TCDD was measured in the eggs collected in 1991 downstream of the bleached kraft pulp mill at Kamloops.

Usually high levels of OCDDs were found in some Osprey eggs while other eggs contained only trace amounts (Whitehead *et al*., 1993). It is not likely that the elevated levels are related to urban development or industrial activity near the nesting sites since often nests with high contaminant levels were located only short distances from other nests which contained only trace amounts (Whitehead *et al*., 1995). Also, OCDD and HpCDD levels in fish foraged on by the Osprey were below detection limits (Mah *et al*., 1989). This suggests Osprey may be accumulating highly chlorinated contaminants from their southern wintering grounds, or during migration (Whitehead *et al*., 1995). However, some of the mink and otter sampled from the upper Fraser River Basin also had elevated levels of OCDDs. Because these mammals do not migrate great distances, there must be a local source. Possible sources of OCDDs include: sawmills or wood preservation sites using chlorophenols; and effluent and/or incinerator emissions from bleached kraft pulp mills which use chlorophenol treated wood (Miles *et al*., 1985; Norstrom, 1988; Kuehl *et al*., 1987). However, the source of highly chlorinated contaminants in the upper Fraser River Basin is currently not clear. Investigation into possible sources and the sporadic accumulation of OCDDs in particular individuals of various wildlife populations in the upper Fraser River Basin is recommended.

The source of the 2378-TCDD detected in eggs from the Thompson River is almost certainly the pulp mill located upstream of the collection site at Kamloops (Whitehead *et al*., 1993, Whitehead *et al*., 1995). In 1988, before PCDD and PCDF abatement was implemented, muscle tissue of largescale sucker, mountain whitefish and northern squawfish collected downstream of the Kamloops pulp mill was contaminated with up to 60 ng/kg 2378-TCDD and 704 ng/kg 2378-TCDF (Mah *et al*., 1989). About half of the fish delivered to the nest by Osprey foraging in the area were either whitefish or sucker (Machmer and Steeger, 1992). Osprey eggs collected downstream of the Kamloops mill had higher levels of 2378-TCDD and 2378-TCDF than eggs collected above mill sites. The Kamloops pulp mill reduced the level of PCDDs and PCDFs in its effluent in early 1990. Similar programs conducted at mills on the Fraser River reduced 2378-TCDD output by 90% (Derksen, pers. comm.), and within a year there was a significant reduction in PCDD and PCDF levels were observed in juvenile chinook salmon (Servizi, 1993). Although PCDD and PCDF levels in the Thompson River have likely fallen considerably since 1990, and 2378-TCDD levels in Osprey eggs collected downstream of the Kamloops mill were lower in 1992 compared to 1991, measurable levels of PCDDs and PCDFs were still detected in most of the Osprey eggs sampled in 1992.

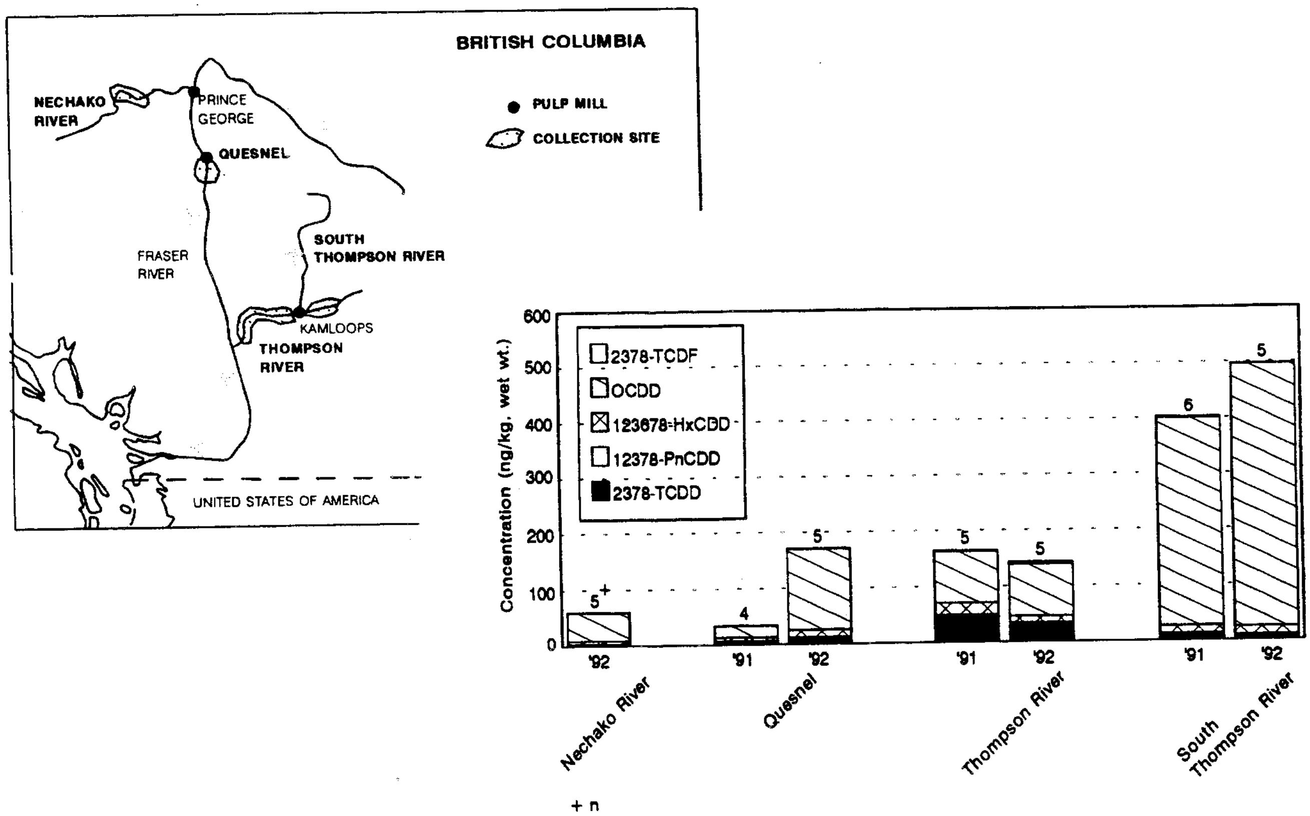


Figure 19. Major PCDDs and PCDFs (geometric means) in Osprey eggs: Nechako, Thompson and Fraser Rivers, 1991 and 1992.

Total toxic burdens (TEQs)

The principal contaminants contributing to the toxic burden of Osprey in the upper Fraser and Thompson Rivers are non-ortho PCBs with 2378-TCDD also contributing substantially to the toxic load of eggs collected downstream of the pulp mills at Quesnel and Kamloops (Figure 20). The toxicological effects of these contaminants on Ospreys are not known, but is currently being investigated in a co-operative study between the Canadian Wildlife Service, the United States Fish and Wildlife Service, and the University of British Columbia.

6.4.2. *Population status*

The productivity of Osprey nesting along the Fraser River has not been determined. However, the reproductive success of Ospreys nesting along the upper Columbia River has been monitored and no statistically significant difference was found in the number of young fledged per active nest located upstream and downstream of pulp mills (unpublished data). Since Osprey populations are recovering and returning to many areas of Canada (Ewins, 1995), it is likely that populations are also on the rise in British Columbia.

6.5. American Mink (*Mustela vison*) and River Otter (*Lontra canadensis*)

American Mink are semi-aquatic carnivores that feed on a wide variety of prey. Because they are highly opportunistic, their diet is extremely variable by season and location (Wren, 1991). In general, the predominant food items are small mammals, followed by fish (Erlinge, 1969; Gilbert and Nancekivell, 1982). Crayfish may also be common prey items (Wren, 1991).

River Otter are also semi-aquatic carnivores that feed largely on fish, with crustaceans (primarily crayfish), birds, insects, amphibians, and mammals comprising smaller portions (Wren, 1991). Although they feed on a wide variety of fish, individual species are preyed upon in direct proportion to their availability, and inversely proportional to their swimming ability. Otter tend to prefer larger fish that are not very agile and less able to find hiding spots than smaller fish, except when smaller fish are more abundant (Erlinge, 1969).

Both species are year-round residents found along much of the Fraser River watershed. The average home range of mink is about 2 km in length (Gerell, 1970; Birks and Linn, 1982). Otter have a larger home range than mink; females with young averaging about 7 km in diameter and males about 15 km diameter (Wren, 1991). Male Otters often travel extensively, with an average nightly distance of 9 to 10 km (Erlinge, 1967). Contaminants detected in mink and otter would be from either (a) a local source, or (b) pollutants that travelled downstream and contaminated local prey, or (c) fish that became contaminated downstream and migrated upstream.

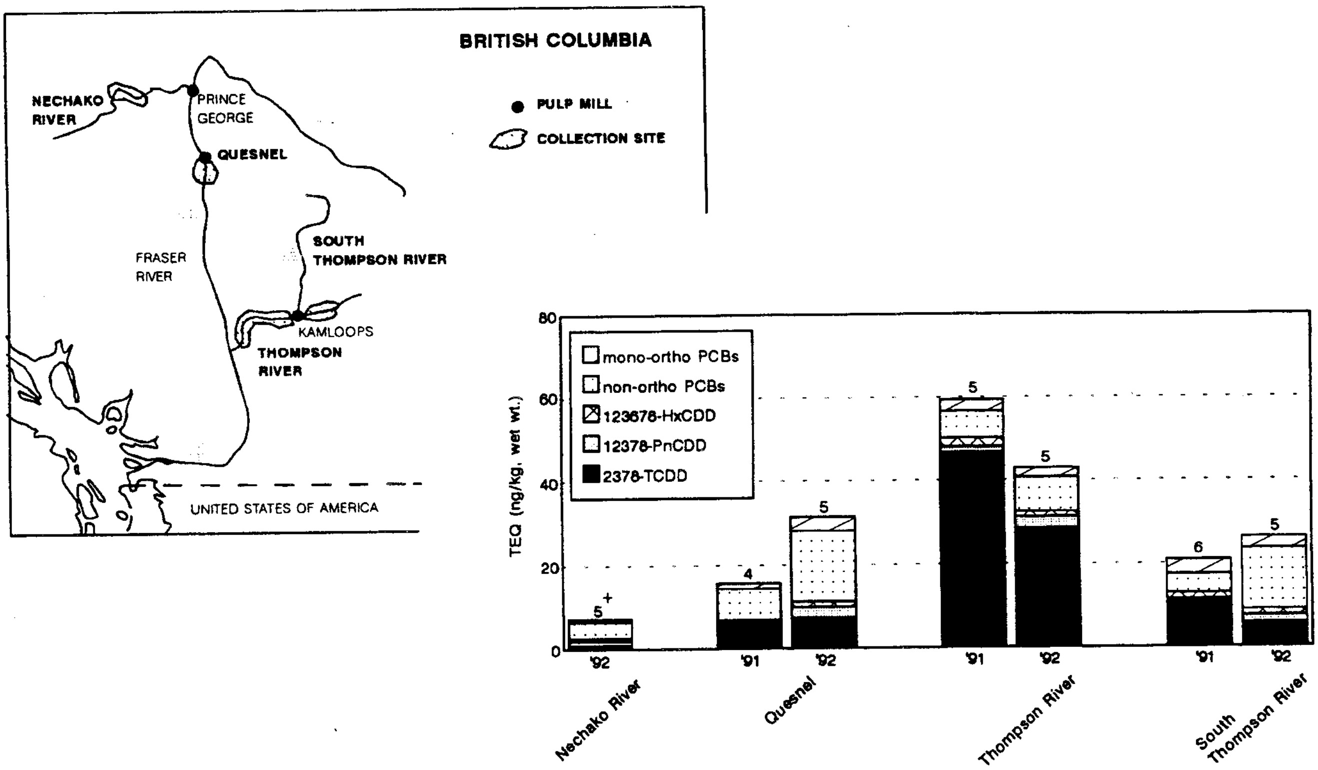


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Figure 20. Toxic equivalents (TEQs) for major PCDDs, PCDF and PCBs in Osprey eggs: Nechako, Thompson and Fraser Rivers, 1991 and 1992.

6.5.1. *Concentrations of chlorinated hydrocarbons*

Mink and otters were collected from the upper Fraser River in the fall and winter of 1991. Three mink were trapped on the shore of Cale Creek where it joins the Fraser River. Since substantially different levels of PCDDs and PCDFs were detected in the liver of each mink, the residue data for each animal is reported separately. The mink are labelled Cale Creek #1, #2, and #3.

Low concentrations of chlorinated hydrocarbons were detected in all of the mink and otters with the exception of the mink from Stone Creek Canyon that had higher levels of PCBs, and two of the three mink from Cale Creek which had higher levels of OCDDs and 1234678-HpCDD.

DDE

Very low concentrations of DDE residues (up to 0.007 mg/kg) were detected in the liver of mink and otter from the upper Fraser River Basin in 1991 (Figure 21). Such low DDE levels in mink indicate that even atmospheric input of global environmental pollutants, such as DDT, is minimal in the upper Fraser River Basin.

Prevailing westerly winds mean that Asia is the primary source of atmospheric pollutants to the west coast of North America (Elliott *et al*., 1989a). Most atmosperically transported pollutants are likely deposited in the heavy precipitation that falls to the west of the Coast Mountains. Fallout of atmospheric pollutants to the upper Fraser River Basin, thus, would be greatly reduced.

Mink are less sensitive to DDT than other chlorinated hydrocarbons. No reproductive effects were observed in mink fed either 100 ppm DDT and 50 ppm DDD, or 100 ppm DDT (Aulerich and Ringer, 1970; Duby, 1970). The levels of DDE detected in the mink and otter in the upper Fraser River would not affect their reproductive success.

Total PCBs

PCB concentrations were generally low (average 0.01 mg/kg) except for the mink from Stone Creek Canyon that had 0.15 mg/kg (Figure 21). The PCB levels detected in most of the samples are comparable to concentrations detected in otter from Alberta (Somers, 1985), which are low by global standards. However, PCB residues detected in liver of mink and otter from the Fraser River Basin were low compared to specimens collected from the Columbia River Basin in 1990 which had an average PCB concentration of 0.375 mg/kg (range: 0.019 - 1.5 mg/kg, n=14) (Elliott and Henny, unpublished data). The reason for the comparatively higher levels of PCBs in the mink from Stone Creek Canyon is not clear. Perhaps that mink ate dietary items higher on the foodchain that had bioaccumulated more PCBs compared to other prey.

Mink are extremely sensitive to PCBs. Mortality occurred in mink with 11.0, 4.2, and 4.5 µg/g Aroclor in brain, liver, and kidney tissue, respectively (Aulerich *et al*., 1973). Aulerich *et al*. (1986) calculated a LD_{50} of 79 µg/g when mink were fed Aroclor 1254 for 28 days; sub-lethal effects were observed when mink were fed a 28 day diet of 7 µg/g Aroclor

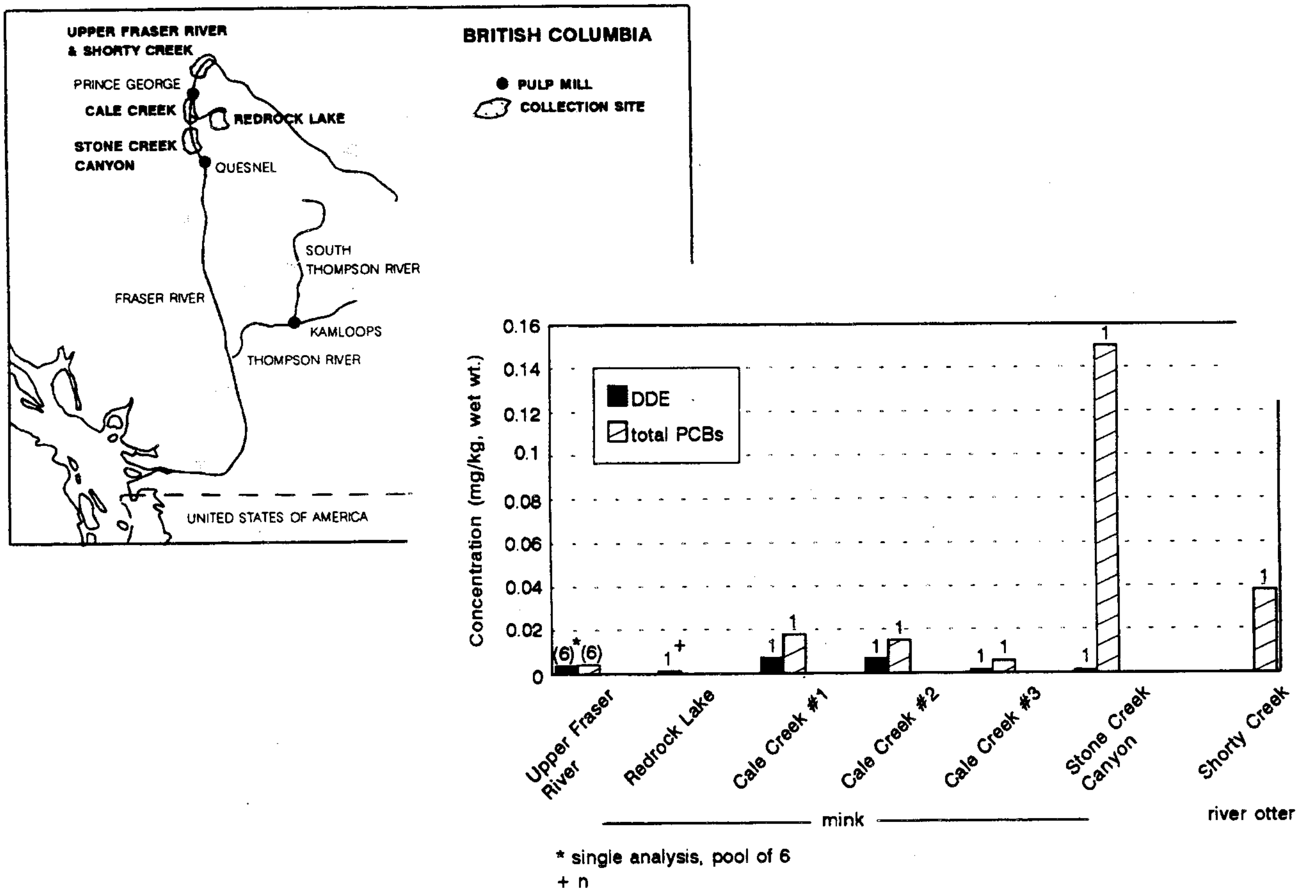

Figure 21. DDE and total PCBs in the liver of American Mink and River Otters: Fraser River Basin, 1991.

1254 (Hornshaw *et al*., 1986). A diet of 2 µg/g Aroclor 1254 resulted in reproductive failure (Byrne, 1974). Although no enzyme induction occurred when mink were fed a 28 day diet of 20 µg Aroclor 1242, a 28 day diet of 20 µg Aroclor 1016 induced CYP1A, BaP hydroxylase and ECOD (Shull *et al*., 1982). AHH was elevated two-fold in adults fed a 79-94 day diet of 1.64 mg Aroclor 1254 (Brunstrom, 1992). A daily diet of 2 mg non-ortho and mono-ortho fractions of Clophen A50 during the reproductive season enhanced EROD two to three fold in females and thirty times in kits of treated females (Brunstrom, 1992). Similarly, although no effects were visible in female mink with 2.0 µg/g Aroclor 1254 in their liver tissue, kits nursed for 5 weeks by these females had 1.75 µg/g Aroclor 1254 in liver as well as reduced growth and survival (Wren *et al*., 1987). Leonards *et al.* (1994) calculated an EC_{50} for kit survival of 2.4 µg PCB/g tissue or 200 pg TCDD-TEQs/g tissue (TEFs from Safe, 1993) and a no effects level for kit survival, EC_1, of 1.3 µg PCB/g tissue. Based on the available toxicity data, PCB concentrations detected in mink and otter in the upper Fraser River Basin should not affect their reproductive success.

PCDDs and PCDFs

The principle PCDDs detected in the mink and otters were OCDDs and 1234678-HpCDD (Figure 22). Although the concentrations of PCDDs and PCDFs in the majority of the mink and otter were below approximately 50 ng/kg, two mink from Cale Creek had considerably higher levels of both OCDDs and 1234678-HpCDD. Comparatively, concentrations of higher chlorinated dioxins were also greater in Ospreys from the upper Fraser River Basin (Section 6.4).

As discussed previously (Section 3.2), sources of higher chlorinated dioxins are not known, but possible sources include: sawmills or wood preservation sites using chlorophenols; and effluent and/or incinerator emissions from bleached kraft pulp mills using wood treated with chlorophenols. Mink and otter usually consume a variety of prey from a local area. The variability observed in contaminant levels among mink probably reflects either the prevalence of specific preferred prey that is highly contaminated or local feeding areas (i.e. an uncontaminated tributary upstream of a contaminated stream or mainstream of the river).

Mink are among the most sensitive species to the toxic effects of TCDDs. A study with adult ranch mink determined a 28 day oral LD_{50} of 4.2 µg TCDD/kg body weight and a 28 day dietary LC_{50} of 4.3 ng/kg for male and female mink respectively (Hochstein *et al.*, 1988). Neonatal mink are also more sensitive to PCDDs, daily injections of 0.1 µg TCDD/kg body weight for 12 days resulted in significant loss in body weight and 50% mortality within 10 weeks (Aulerich *et al*. 1988). A risk assessment conducted by the Canadian Wildlife Service for environmental assessment hearings for a new pulp mill development in northern Alberta indicated that mink feeding on a diet containing greater than 5 ng/kg TCDD or toxic equivalent could suffer reproductive effects. The sensitivity of mink and otter to HpCDDs and OCDDs is not known, but in general, highly chlorinated hydrocarbons are much less toxic than lower chlorinated hydrocarbons (Table 4).

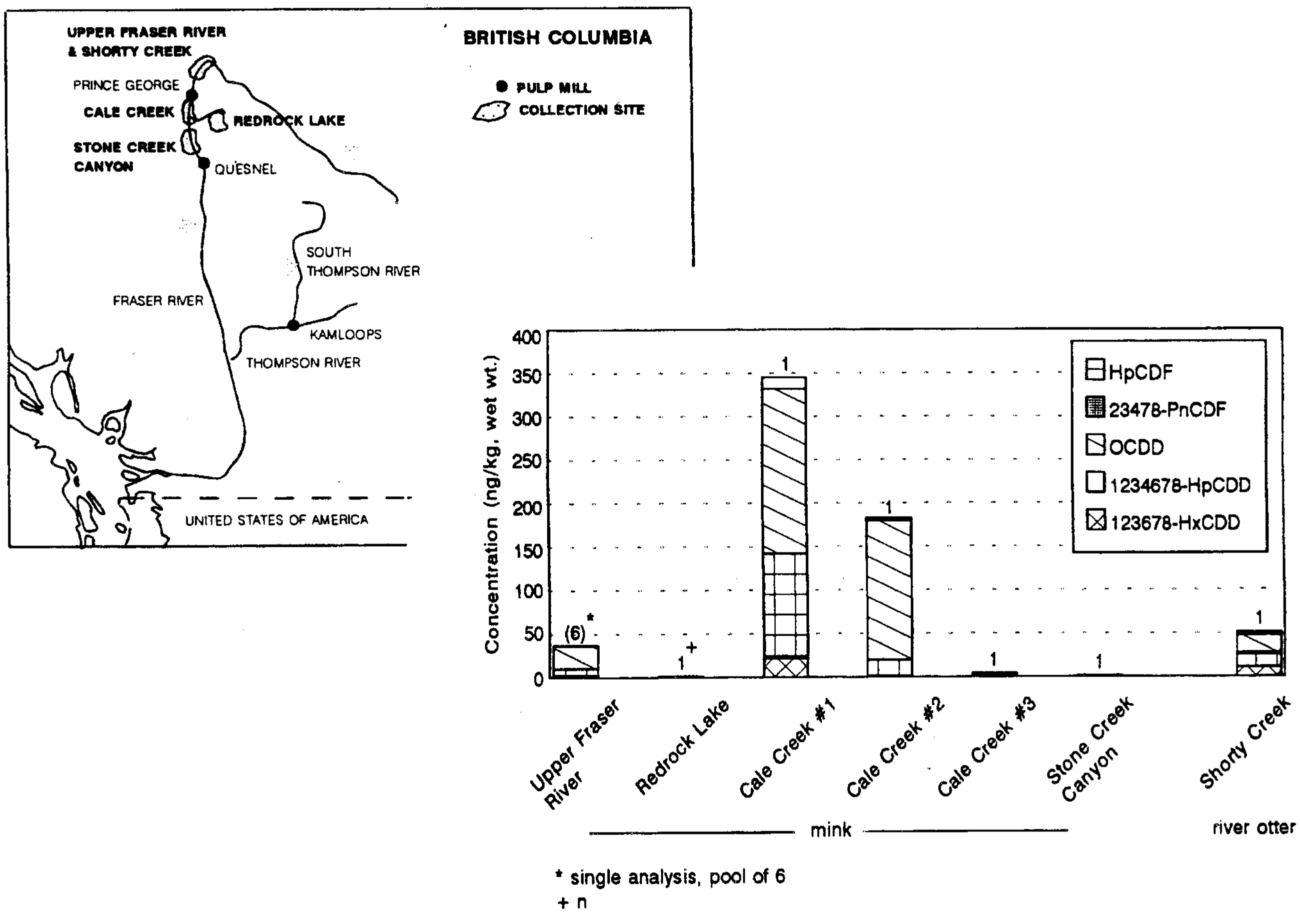

Figure 22. Major PCDDs and PCDFs in the liver of American Mink and River Otters: Fraser River Basin, 1991.

Total toxic burdens (TEQs)

The toxic burden of the mink and otter collected from the upper Fraser River was low, with TEQs up to 5 ng/kg (Figure 23). The major contributor to the toxic load varied among locations, with 123678-HxCDD the largest contributor at Cale Creek #1 and mono-ortho PCBs dominant at Stone Creek Canyon; non-ortho PCBs contributed significantly to the toxic burden in every sample.

6.5.2. *Population status*

Data on population status of mink and otter in the upper Fraser River is not available, but there is no reason to suggest that populations with minimal contaminant body burden are not reproducing normally. Toxic chemicals have been implicated in the population decline of otter in several locations including Britain (Chanin and Jefferies, 1978), Sweden (Sandegren *et al.*, 1980; Erlinge, 1980; Olsson and Sandegren, 1983), and Georgia (Halbrook *et al.*, 1981), and mink and otter in Oregon (Henny *et al.*, 1981), and the Great Lakes area (Wren, 1991). Studies are underway to determine PCB exposure of River Otter in the Fraser estuary.

7. Prevalence and distribution of chlorinated hydrocarbons in the Fraser River Basin and their relevance to local wildlife populations

Chlorinated hydrocarbons contaminate the food chain by moving from contaminated soils or suspended solids (most are hydrophobic and rapidly adsorb to sediment particles or organic molecules in suspension) to invertebrates to fish to fish-eating birds or mammals. Species differences in contamination are primarily determined by variations in diet, movements outside the breeding area, and physiology (Elliott and Noble, 1993). In this section, the relative toxic burden observed in each of the indicator species are compared. Geographic differences and temporal trends are discussed. Conclusions are drawn about the toxicological significance of the contaminant levels detected in the wildlife in the Fraser River Basin.

7.1. DDE

The highest recent concentrations of DDE were detected in the eggs of Bald Eagles (0.9 to 4.9 mg/kg, average 2.6 mg/kg), followed by the Great Blue Heron (0.1 to 2.5 mg/kg, average 0.9 mg/kg), Osprey (1.1 to 2.1 mg/kg, average 1.7 mg/kg), and Cormorants (0.2 to 0.5 mg/kg, average 0.3 mg/kg). The lowest levels of DDE were measured in liver of mink (0.001 to 0.007 mg/kg, average 0.004 mg/kg) and otter (below detection limit).

The contaminant levels detected in the various wildlife indicator species can be explained by differences in preferred prey and foraging areas. Although herons and cormorants consume prey of similar size, herons tend to feed closer to shore than cormorants. Bald Eagles consume primarily larger fish, waterfowl and gulls that have accumulated elevated levels of DDE from their diet. Osprey also prey on larger fish species and likely also have accumulated significant concentrations of DDE from their wintering

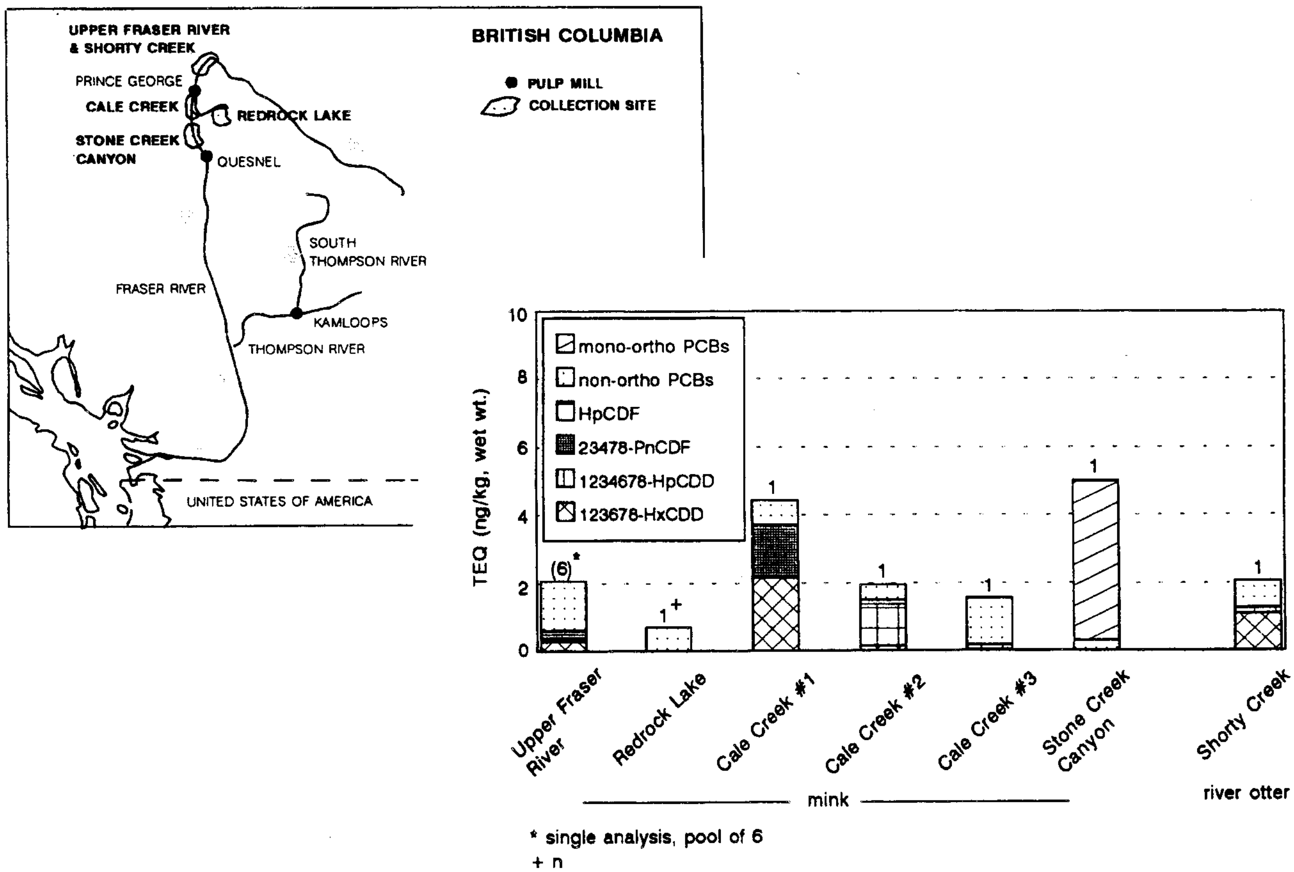

Figure 23. Toxic equivalents (TEQs) for major PCDDs, PCDFs and PCBs in American Mink and River Otters: Fraser River Basin, 1991.

areas in Latin America, where DDT is still used. Mink and otter tend to forage on a variety of smaller local prey. The low DDE burden detected in these species may reflect the comparatively low levels in their prey and lack of local sources.

It is unlikely that current concentrations of DDE would cause seriously affect productivity of any of the indicator species discussed here. Currently there is increasing concern over chick exposure to hormone-mimic compounds like DDE (Colborn *et al*., 1993). The long term effects are not clear, but should be less than in the past given the indication of general decrease in exposure. Extrapolating from declining levels of DDE in herons, the concentrations of DDE in the 1960s and 1970s were probably high enough to have effected egg shell quality and therefore reproduction of Bald Eagles at that time (Elliott *et al*., 1996).

7.2. PCBs

Eggs collected from the heron colony at UBC between 1977 and 1982 contained the highest concentrations of PCBs (11 to 18 mg/kg, average 15 mg/kg). After 1982, the PCB levels detected in the indicator species were considerably lower with the highest concentrations found in the Bald Eagles (1.1 to 6.2 mg/kg, average 3.0 mg/kg), followed by herons (0.1 to 3.6 mg/kg, average 1.7 mg/kg), cormorants, (0.7 to 2.3 mg/kg, average 1.4 mg/kg), Osprey (130 to 420 µg/kg, average 290 µg/kg), and finally mink and otter tissues (below detection limit to 150 µg/kg, average 32 µg/kg; and 38 µg/kg, respectively).

The overall reduction of PCBs undoubtedly results from regulation of the use of PCBs in 1977 and the ban on the importation of equipment containing PCBs legislated in 1980. However, PCB-containing electrical equipment continues to be used, PCBs are still measured in many industrial effluents, and spills from stored or in-use equipment do occur.

Overall, higher levels of PCBs were measured in indicator species in the lower basin foodchain compared to the upper basin foodchain. This may reflect larger historic outputs of PCBs in the lower Fraser River basin compared to the upper reaches since more industries were present in the lower basin. In the Fraser River estuary, since 1982, contaminant burdens have been highest in eggs of Bald Eagles, followed by the UBC and Stanley Park heron colonies and the Double-crested Cormorants (although the difference between contaminant levels in the heron and cormorant eggs is small). Relative concentrations of PCBs detected in various wildlife indicator species may be partially explained by their preferred prey. Bald Eagles forage on large prey such as gulls, grebes and waterfowl that would have accumulated higher levels of PCBs from their diet. Great Blue Herons and cormorants forage on smaller fish that may not have bioaccumulated PCBs from their local environment. Cormorants also move around more in the winter and therefore exposure may be greater or less depending on their wintering site.

Indicator species in the upper reaches of the Fraser River basin (mink, otter, and Osprey) have extremely low PCB levels. This indicates that the upper Fraser basin appears to be among the "cleanest" systems at least in North America. Most of the organochlorines and PCBs in the mink and otter likely come from atmospheric input. Because of the influence of the coastal mountain range in trapping weather patterns from the west, airborne contaminants from industrial and agricultural sources in Asia are probably greatly reduced upon reaching the upper Fraser River Basin. Since mink and otter forage on a variety of small fish and animals, the low level of PCBs detected in their tissues most likely reflect the low concentrations in their local environment. Compared with PCB burden measured in mink in other areas such as Alberta and the Great Lakes, the mink in the upper Fraser River basin are clean (Somers, 1985; Wren, 1991). The relatively low PCB burden in the Osprey may be a result of the fewer PCBs being released into their local environment although they may have acquired some burden from southern wintering grounds in the case of individuals that wintered in the southern United States or industrial areas of Mexico for example.

Toxicology data suggest that the PCB levels currently detected in most of the wildlife indicator species studied in this report would not adversely effect overall productivity. However, the consequences of long term exposure to the PCB levels found in Bald Eagles are not clear. In particular, chronic low level PCB exposure can have subtle effects on the immune and reproductive systems (Safe, 1990).

7.3. PCDDs and PCDFs

Concentrations of the various PCDDs and PCDFs detected in wildlife can often be interpreted, given adequate information on foraging grounds and proximity to pollutant sources, their diet, the behaviour of the specific contaminants in the environment, and their capacity to accumulate in wildlife. Acquisition of such data requires a commitment to long term research and monitoring studies of both the ecology of the sentinel species and toxicology of the chemicals of concern. In general, the higher chlorinated PCDDs and PCDFs are more persistent in the environment than the less chlorinated congeners, but some of the less chlorinated congeners are more toxic and accumulate more readily in wildlife.

The most prevalent congeners in heron and cormorant eggs collected from the Fraser River estuary were 123678-HxCDD, 12378-PnCDD and 2378-TCDD. Those congeners most likely entered the environment in runoff from wood treatment plants that used chlorophenols, effluent from pulp and paper mills which used chlorophenol treated wood, and (especially 2378-TCDD and 2378-TCDF) from kraft mill effluent using chlorine bleaching.

The highest level of total PCDDs and PCDFs were detected in the heron colony at UBC in 1982 (almost 1400 ng/kg). After 1982, the levels measured in all of the heron and cormorant eggs were considerably lower (23 to 370 ng/kg, average 150 ng/kg). The apparent drop during 1982 to 1987 is difficult to explain, since PCPs were still in use and kraft mill pulp production was increasing. The 1982 results may be at lest partially an analytical artifact as the methodology had only recently been developed at that time. By 1990, when pulp mills on the upper Fraser and Thompson River began decreasing chlorine use in anticipation of new regulations, 2378-TCDD levels began to decline in eggs of some species (herons, cormorants). Trends from about 1991 to 1992 are the result of the restrictions implemented on the allowable levels of PCDDs and PCDFs in pulp mill effluent and the removal of chlorophenols from the market. In general, the levels were highest in eggs from the heron colony at UBC, followed by the Double-crested Cormorant eggs, and then the heron eggs from the Nicomekl and Stanley Park colonies. Herons and cormorants tend to forage in specific locations. The overall higher levels observed in the heron colony at UBC is likely attributable to the settling out of contaminants adsorbed to fine particulate effluent from upstream mills (TCDD and TCDF) and chlorophenol sources located on the North Arm of the Fraser River immediately upstream from the colony (higher-chlorinated compounds). The cormorants could be acquiring their contaminants from effluent from the numerous pulp and paper mills in the Strait of Georgia and Howe Sound. In addition, herons and cormorants at some colonies may prefer specific prey that could be more contaminated than the fish preferred by birds at other colonies.

Although the Bald Eagle eggs collected in the lower Fraser River had detectable concentrations of 123678-HxCDD and 12378-PnCDD, 2378-PCDD and 2378-PCDF were present in the highest concentrations in most eggs. This difference is probably the result of eagles foraging on larger prey. The less chlorinated congeners tended to accumulate in the food chain thereby exposing eagles to a higher burden compared to heron and cormorants that feed on smaller fish species.

The most "prevalent" PCDDs and PCDFs detected in the Osprey eggs collected from the upper Fraser River were OCDDs. We hypothesize that the Osprey may have accumulated these contaminants from their equatorial wintering grounds; the finding of OCDDs being released into the environment (i.e. from combustion processes) in Osprey wintering areas would support this hypothesis. An alternative hypothesis is that an OCDD source, as yet unidentified, is present in the upper Fraser River Basin. The comparatively higher levels of 2378-TCDD detected in eggs from the Thompson River probably originated from effluent from upstream pulp and paper mills.

The major PCDD congeners detected in the mink and otter collected from the upper Fraser River were OCDDs and HpCDDs. Since organochlorine and PCB levels in most of the mink and otter are very low and likely are obtained from atmospheric input (Section 7.2), finding only some of the mink with elevated levels of PCDDs must indicate local sources rather than atmospheric input. Also, since mink and otter are year-round residents, highly chlorinated congeners would be obtained from feeding on the local foodchain. The ultimate sources are not clear.

Restrictions on the concentrations of contaminants allowed to be released into the environment have resulted in a reduction of chlorinated hydrocarbon residues in the wildlife indicator species discussed in this report. Productivity studies of Great Blue Herons and Bald Eagles recently conducted in the Fraser River Basin suggest that reproductive success is not adversely affected by current PCDD and PCDF burdens.

7.4. Total toxic burden (TEQs)

The largest contributors to the total toxic burden in the wildlife species discussed in this report are non-ortho PCBs and 2378-TCDD. The prevalence of the non-ortho PCBs in wildlife 15 years after regulatory restrictions reflects their abundance and resilient behaviour in the environment as well as their tendency to accumulate in foodchains. The contribution of 2378-TCDD to the overall toxic load reflects its high toxicity, its tendency to bioaccumulate, and the importance of pulp mill sources in the Fraser River.

The eggs collected from the lower Fraser River (heron, cormorants and eagles) contained higher contaminant levels than the tissue samples collected in the upper Fraser River (Osprey, mink, and otter). This trend reflects the increased industrialization progressing from the upper to lower reaches of the river.

Among the wildlife species presented in this report, the highest burden was observed in the heron eggs collected in the early 1980s. Unfortunately, samples were not collected from other wildlife species until the mid-1980s. Considering comparative contaminant levels detected in various wildlife species in the early 1990s, contaminant levels in the early 1980s would also probably have been very high in predatory birds such as Bald Eagles.

The total toxic burden measured in the most recently collected samples for each species were highest in the Bald Eagles, followed by the herons, cormorants, Osprey and finally mink and otter. Differences in contaminant levels can be explained by their relative position in the foodchain, their diet, and preference to forage in particular areas.

Non-ortho PCBs are not as acutely toxic to wildlife as the PCDD congener 2378-TCDD. This chemical was the principal contributor to the toxic burden measured in Bald Eagles in the lower reaches of the Fraser River as well as Osprey nesting downstream of the pulp mills near Kamloops in the upper Fraser River.

The biological significance of the concentrations of 2378-TCDD detected in these species is not clear. Productivity studies of herons and Bald Eagles suggest the current levels are not adversely effecting populations. However, the sublethal effects are difficult to monitor. Some level of monitoring of sentinel indicator species should continue.

8. Recommendations for future research

Contamination of the Fraser River has been monitored by measuring the chemical residues in eggs and tissue from a variety of wildlife since the early 1980s. Studies to date suggest total burden of chlorinated hydrocarbons in local wildlife populations has decreased, particularly in recent years. The toxic burden in tissues (i.e. eggs in birds; livers in mammals) is closely connected to their relative position in the foodchain, their diet, and preference to forage in particular areas. Differences in the behaviour of individuals can result in a large range of contaminant levels detected in their tissues. It is important to continue monitoring the toxic burden of these species in order to further establish temporal trends. Collection and specimen banking of tissue of selected indicator species, such as Great Blue Herons, will also permit retrospective determination of any newly identified toxicant.

The toxicity of PCDDs and PCDFs varies significantly among species, including the those monitored in the Fraser Valley. The biological significance of the low concentrations of the chemicals detected in many of these indicator species is not well understood. Further research should be conducted to determine the effects of these contaminant levels on overall health of the individual as well as the productivity and reproductive success of the population. Cumulative and possible synergistic effects of exposure to and uptake of multiple chemicals, combined with other human-derived stresses such as habitat loss and disturbance are also not clear. Research studies should be initiated to address these issues.

Specific monitoring and research projects which would help further assess the impact of chlorinated contaminants on local wildlife in the Fraser River Basin include:

1. Egg collection of select wildlife species from specific sites, such as the Great Blue Heron colony at UBC, should be continued in order to further monitor contaminant trends.

2. The health of otter and possibly mink populations in the Fraser River Estuary should be assessed. Contaminant burdens should be determined and related to potential disruption of the endocrine system.

3. The foraging behaviour (location, habitat, diet) of some indicator species (herons, cormorants, eagles) should be further investigated and the data related to the relative concentrations of various chlorinated compounds observed in their eggs.

4. Telemetry studies to determine the migratory route and wintering area of the osprey breeding in the upper Fraser Basin should be initiated in order to determine potential sources of highly chlorinated compounds found in their eggs.

5. Continued monitoring of bald eagle reproductive success in the lower Fraser valley will provide an ongoing indication of the population levels. Cumulative and/or synergistic effects of chlorinated contaminants would also be effectively addressed using the bald eagle as an indicator.

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