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CONTAMINANTS IN WILDLIFE INDICATOR SPECIES FROM THE FRASER RIVER BASIN

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The Fraser River Basin originally contained large areas of pristine riparian and wetland habitat important for many resident and migratory wildlife species. The intensive use and alteration of these habitats by agricultural, urban and industrial development have lead to concerns about the viability and health of local wildlife populations. Scientists have focused their research on a small number of wildlife species which are used to gauge the health of the entire wildlife community. These species are known as 'indicator species.'

Discrete breeding and/or resident populations of four bird species and two mammal species have been extensively monitored along the Fraser River since the early 1980s. These species embody many of the elements that characterize a good indicator species (Moul *et al.* 1996). In an assessment of 57 indigenous amphibian, bird and mammal species, four birds (great blue heron, osprey, bald eagle and tree swallow) and two mammals (river otter and mink) monitored in our studies achieved among the highest scores for potential use as indicator species in the Fraser River Basin (Moul *et al.* 1996). Ranking criteria were based on published knowledge and the practicality of sample collection. Criteria included: (1) fundamental attributes of residency, distribution, abundance and diet; (2) current understanding and documentation of natural history, such as home range, habitat, migration and historic abundance; (3) ease of collecting samples; (4) suitability of each species for laboratory study; and (5) availability of knowledge on contaminant responses. Five of the six species selected (great blue heron, osprey, bald eagle, mink, river otter) had comparatively high scores also because they have been monitored in the Fraser River Basin before. Consequently, data are available on their contaminant levels and biology. Each of these species has its utility as an indicator affected by its distinct habitat requirements and regional distribution as described below.

The great blue heron was selected as an indicator species in the Lower Fraser Valley because it is a year-round resident with a dominantly piscivorous diet. Herons eat mostly one-year-old fish or younger (Harfenist *et al.* 1993), which implies that contaminants acquired through the food chain originate locally. Double-crested cormorant, while not evaluated in the Moul *et al.* (1996) report because of its marine habitat preference, was also added to the Lower Fraser Valley indicator group because it possesses many characteristics of a good indicator species, and there is a breeding colony in the Fraser delta. Heron and cormorant eggs were collected from breeding colonies in the estuary (Fig. 1). Cormorants, however, were found to forage beyond the Fraser estuary to the Strait of Georgia; thus they better represent the condition of the southern Georgia Basin, rather than specifically the Fraser estuary. Herons have been regularly monitored since the late 1970s; cormorants have been sampled periodically since 1985.

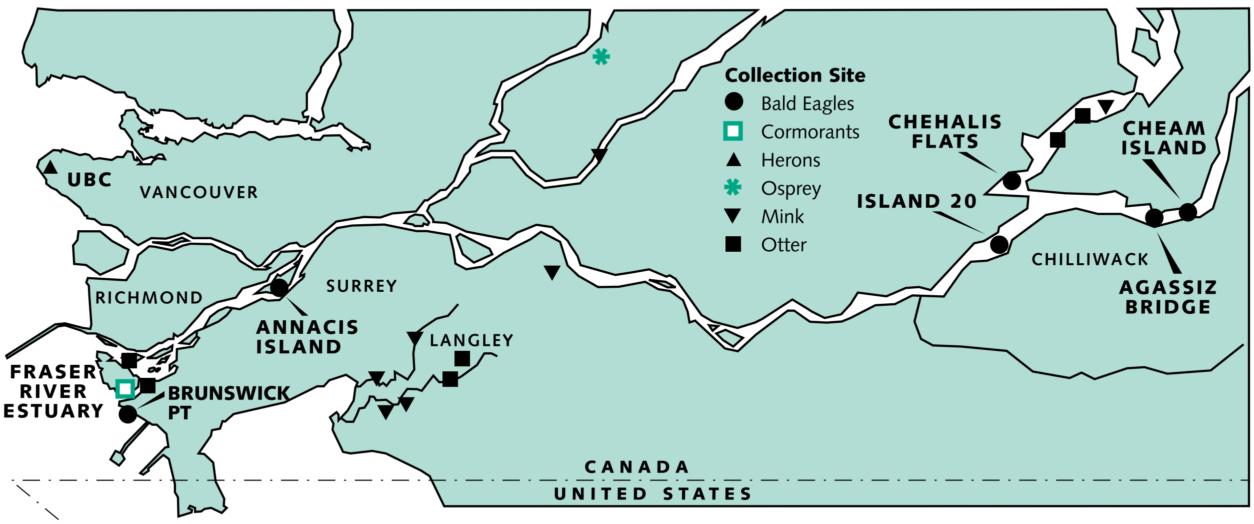

Figure 1. *Collection sites of selected indicator species in the Lower Fraser Valley, 1990–97.*

The two raptor species chosen for contaminant surveys occupy all major habitats in B.C. The bald eagle is a year-round resident along the coast, and also uses the upper reaches of the Fraser River for breeding. It has been useful as an indicator species in the lower basin, largely because of its scavenging habits, which include the selection of impaired (often contaminated) prey (Noble *et al.* 1993). The osprey breeds along the full length of the river, but migrates to southern foraging grounds, primarily in Latin America, for the winter. Osprey predominantly eat large, slow-moving fish (*e.g.* largescale sucker). As these fish feed along the stream bottom, and Fraser River sediments have substantial levels of persistent chlorinated compounds (Mah *et al.* 1989; Hatfield Consultants 1995; Brewer *et al.* 1998), osprey represent the top of an aquatic food chain that is potentially biomagnifying historical and ongoing inputs of contaminants. While osprey, cormorants and herons have direct, essentially exclusive links to aquatic food webs, the more opportunistic bald eagles prey on aquatic (fish), semi-aquatic (ducks), and terrestrial (birds, small mammals) species. This will increase the diversity of contaminants and contaminant sources to which the eagles are exposed. Bald eagle eggs and chicks collected from nests along the river in the Fraser Valley in 1990–91 (Fig. 1) were analyzed for contaminants. Dead or dying bald eagles were also collected from 1989 to 1994 by a variety of agencies, autopsied for cause of death, and assayed for contaminants in liver tissue (Elliott *et al.* 1996a). Osprey eggs were collected from nests near Kamloops, Quesnel and Prince George from 1991 through 1997 for contaminant analysis, while eggs collected in 1995 and '96 were hatched in captivity to study the impact of pulp mill contaminants on embryo development (Fig. 2).

The tree swallow was added to the monitoring regime only recently (1994) as the fifth bird indicator species. These swallows nest along the full length of the Fraser River, and feed heavily on insects over the

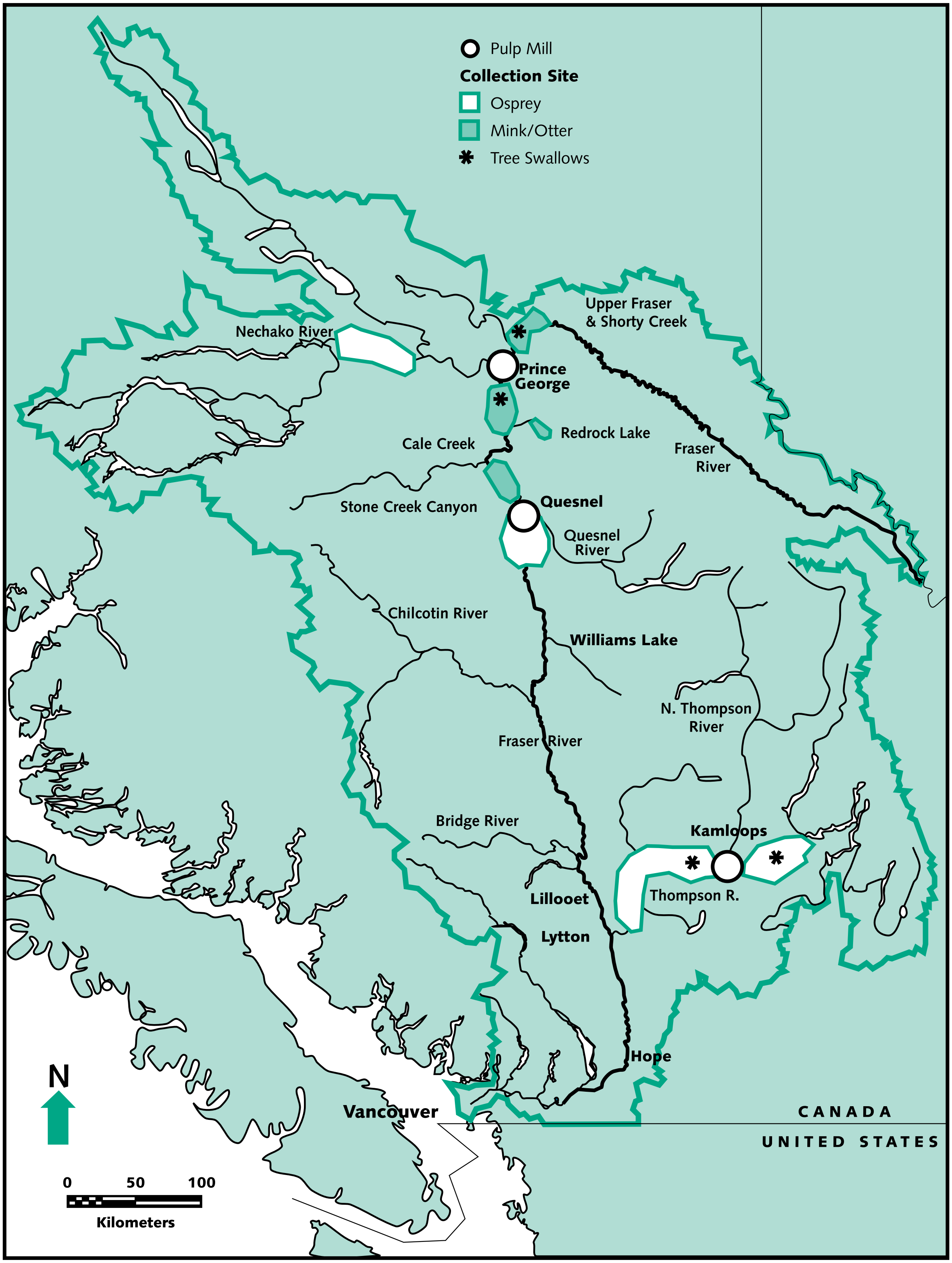

Figure 2. *Collection sites of selected indicator species in the upper reaches of the Fraser River Basin, 1990–97.*

river, especially during emergence events (of chironomids, for example). Because of their insectivorous nature, we hypothesized that they would be good indicators of the effects of persistent as well as less persistent contaminants on the river. Tree swallow nestlings were collected near pulp mills at Kamloops and Prince George (Fig. 2) from nest boxes placed specifically to attract birds for the study.

Finally, mink and river otter were chosen as mammalian indicator species. They have relatively small home ranges, without the migratory behaviour characteristic of some of the bird species. Also, they potentially prey on different species than birds. Mink are opportunistic predators (Wise *et al.* 1981). River otter likely have tighter ties to aquatic food webs, since the majority of their diet consists of fish with only minor seasonal variation (Melquist and Dronkert 1987). Mink and otter were initially only collected from the upper reaches of the river (Fig. 2), but surveys were expanded in 1995 to include collections in the Lower Fraser Valley (Fig. 1).

LEVELS AND EFFECTS OF CONTAMINANTS BY CHEMICAL GROUP

The contaminant families most frequently measured in wildlife from the basin are discussed in the following subsections. While most contaminant groups could be detected in each species, each indicator species is exposed to different levels in their food and have differing biochemical responses to this exposure. In addition, species and class differences in sensitivity varied the risk associated with some compounds. For instance, mercury residues were predominant in more piscivorous species, because fish are a major environmental source (via bioaccumulation pathways) of mercury (Fimreite *et al.* 1971). Lead and anticholinesterase pesticide residues, on the other hand, were predominant in bald eagles, because routes of uptake are typically through waterfowl foraging on agricultural and foreshore areas (see Wilson *et al.* 1999). However, the persistent chlorinated compounds, including polychlorinated-*para*-dibenzo-dioxins (PCDDs), polychlorinated dibenzofurans (PCDFs), polychlorinated biphenyls (PCBs) and organochlorine (OC) pesticides were detected in all species.

Organochlorine Pesticides

Contamination of B.C. wildlife with OC pesticides is a consequence of local persistence of historically applied compounds, long-range atmospheric transport of compounds from jurisdictions still allowing their use or having deposits from past use that are still volatile, and exposure on wintering grounds in Latin America. Only a handful of OCs are still registered for use in Canada (*e.g.* endosulfan, dicofol) and they are not very persistent nor readily bioaccumulated (Fry 1995). The most prevalent OC pesticide detected in wildlife from the Fraser River Basin was dichlorodiphenyldichloroethane (DDE), a more persistent breakdown product of dichlorodiphenyltrichloroethane (DDT).

The bald eagle and osprey showed higher egg concentrations of DDE than the other species (Table 1; see end of chapter); established toxicity levels were attained or surpassed in some nests. Wiemeyer *et al.* (1993) reported that 3.6 to 6.3 mg/kg wet weight DDE in bald eagle eggs would inhibit the production of young to approximately half normal values. Two of six eggs collected in 1990–91 (from Annacis Island and Chehalis Flats, downstream of Harrison and Agassiz) contained DDE within that range, and mean eggshell thickness was less than the pre-1947 average for the Pacific Northwest (Elliott *et al.* 1996b). Both findings suggest that DDE might still limit productivity of some eagle nests in the Lower Fraser Valley.

Survey data suggest that bald eagles currently have stable populations in the basin. Productivity calculations indicate that reproductive success is well above levels needed to sustain populations (Elliott and Norstrom 1998). However, in a five-year survey of nest success across coastal B.C., DDE plasma concentrations in chicks were weakly related to productivity declines (Elliott and Norstrom 1998), implying that DDE might influence provincial bald eagle populations in a limited fashion. In addition, back calculations based on current DDE levels in eagle and heron eggs and historical concentrations in heron eggs, suggest that breeding populations of bald eagles in the 1970s would have been affected by DDE levels around 25 mg/kg in eggs (Elliott *et al.* 1996b). Although DDE concentrations still fall into the range for effects in bald eagles, we suggest that any minor impacts at the population-level have been offset by other factors, including regulation of lead and specific pesticides causing mortalities in the Lower Fraser Valley (Elliott *et al.* 1992; Elliott *et al.* 1997), reduced persecution and more attention to protection of habitat, particularly nest trees (Elliott *et al.* 1996b).

Concentrations of DDE in osprey eggs showed high variability among individual eggs within study areas, although there were no significant differences in mean concentrations among study areas (Fig. 3). Twenty percent (10 of 51) of eggs contained DDE concentrations >4.2 mg/kg, the value associated with 15 per cent eggshell thinning in osprey (Wiemeyer *et al.* 1988), while eight per cent (4 of 51 eggs) contained >10 mg/kg which is associated with improper embryo development (Poole 1989). There was also consistent evidence of egg shell thinning in the eggs with elevated DDE levels (>4.2 mg/kg)(Machmer *et al;* in prep). Findings to date suggest that DDE might limit productivity of some osprey nesting at sites along the Fraser River. Reproductive success of osprey nesting along the Fraser River has been analyzed from the perspective of examining possible effects of exposure to pulp mill contaminants in populations breeding upstream and downstream of bleached kraft pulp mill sites (see section 2.4.3: Machmer *et al.,* in prep). These data do not differentiate eggs with varying DDE levels; consequently, potential effects of DDE exposure on osprey productivity are not currently available.

Most of the DDE acquired by ospreys during the breeding season likely entered the environment via long-range atmospheric transport as there was minimal historic use of its parent substance, DDT, in most of the Fraser Basin (Elliott *et al.*, in press). However, while Donald *et al.* (1993) and Macdonald *et al.* (1999) found that long-range atmospheric transport and deposition of pesticides could result in relatively high concentrations in trout and burbot in remote alpine headwater lakes, the latter study found even higher levels in lakes near agriculture (*e.g.* Nicola Lake). In agricultural areas where DDT was historically used in large quantities, the primary source of DDE for osprey is likely the ongoing release of DDT compounds from soil. This theory is supported by elevated DDE levels measured in osprey eggs collected near Pitt River, which is adjacent to an area of intensive horticultural activity in the Lower Fraser Valley. The most probable source of high DDE levels in osprey from the upper Fraser is exposure on their wintering grounds. Osprey nesting in the Fraser Basin likely migrate south to winter along the west coast of

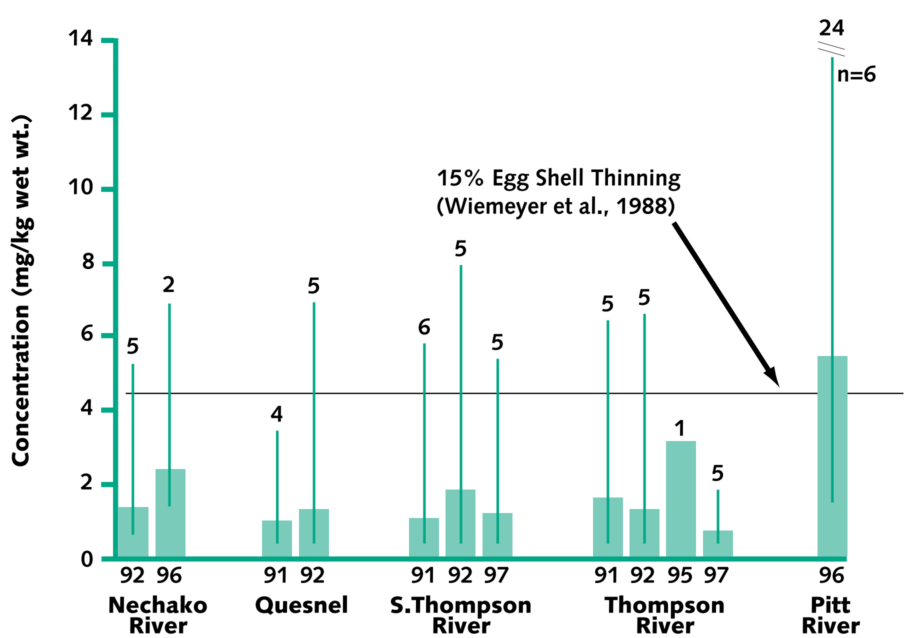

Figure 3. DDE in osprey eggs collected in the Fraser River Basin, 1991–97. Values expressed as geometric means with 95% confidence interval error bars (mg/kg wet wt.). Number above bar graph represents the number of eggs analyzed per site per year.

Mexico and other Central American countries where areas of intensive agriculture are located next to common osprey roosting sites. Although the DDE:DDT ratios are extremely variable, there is no indication of recent DDT input. Even ospreys with DDE concentrations >10 mg/kg had relatively high DDE:DDT ratios, suggesting that DDE exposure on wintering grounds is also from historic rather than recent DDT usage. The high degree of individual variability within sites also may be caused by differences in diet or related food chain factors (Elliott *et al.*, in press).

Other indicator species showed lower levels of DDE, and likely were not affected by measured concentrations (Table 1; see end of chapter). Great blue heron surveys indicate that residual values have been declining from peaks seen in the late 1970s (Fig. 4). Although past agricultural use probably has some influence on body burdens in wildlife from the Lower Fraser Valley and in the Nicola Lake area (Macdonald *et al.* 1999), it appears that atmospheric transport from regions where DDT is still used (*e.g.* Asia) might be responsible for the widespread occurrence of DDE even in areas without agriculture. Elliott *et al.* (1989a) found elevated DDE concentrations in eggs of seabirds along the B.C. coast. DDE plasma concentrations from bald eagle chicks collected on the west coasts of Vancouver and Langara islands were also elevated (Elliott and Norstrom 1998). These trends suggest a substantial DDE input from atmospheric sources as has been noted in the assessment of organochlorine accumulation in selected headwater lakes in the basin (Macdonald *et al.* 1999).

A suite of other OC pesticides was consistently detected in wildlife at low levels (Table 1). Chlordanes (trans- and cis-nonachlors, transchlordane, oxychlordane) were commonly found in eggs and adult livers, along with dieldrin, heptachlor epoxide, chlorobenzenes, hexachlorocyclohexane and mirex. Organochlorine residues detected in mink and river otter were not notably different in the agricultural setting of the Lower Fraser Valley from other collection sites along the river. This observation, along with the presence of compounds like mirex, suggests that contaminant sources are primarily atmospheric. None of the compounds in this suite of pesticides poses a threat to wildlife health at concentrations detected.

Polychlorinated Biphenyls (PCBs)

PCBs is the generic term for a family of 209 congeners that contain a variable number of chlorine atoms on a biphenyl molecule. Their major use was in electrical transformers. While all uses in Canada were significantly curtailed in 1977, up to 40% of the PCBs imported are not accounted for and have been dispersed in the environment (Moore and Walker 1991). Unfortunately, chemicals in this family are relatively persistent and semi-volatile. A major route of re-distribution of PCBs existing in soils, sediments and waste

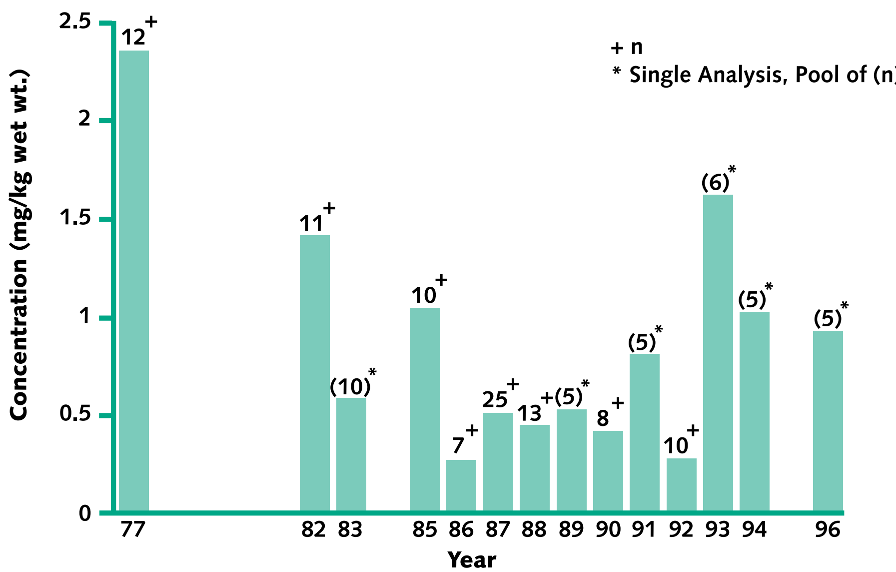

Figure 4. *DDE (geometric means, mg/kg wet wt.) in great blue heron eggs collected from the colony at the University of British Columbia Endowment Lands in the Lower Fraser Valley, 1977–96. Number above bar graph represents the number of eggs analyzed per year.*

dumps occurs via the atmosphere and so deposition can contain PCBs from sources nearby or from sites on other continents in the Northern Hemisphere.

The PCB family of contaminants poses more of a risk to mustelids than birds. Mink reproductive success is extremely affected by exposure to small amounts of PCBs (Aulerich and Ringer 1970; Heaton *et al.* 1995; Leonards *et al.* 1994; 1995), and the threatened or extinct status of otter in most European countries has been linked to PCB contamination (Mason 1989; Smit *et al.* 1994). Mink and river otter from the Fraser River Basin showed low levels of total PCBs (Elliott *et al.* 1998a) compared to those seen in the Columbia River Basin (Table 2; see end of chapter). It is difficult to categorically state whether these levels in the Fraser Basin have an effect on these species. A negative correlation found between juvenile male mink baculum (penis bone) length and liver Aroclor 1260 (a subset of PCBs) levels found using the Columbia and Fraser data set, suggests there could be impacts on reproductive success in some locations (Harding *et al.* 1999; Henney *et al.,* in prep.). As the Fraser sample data formed the "normal" range for baculum length and the low range for PCBs, we cannot assess its potential impact in mink at these lower levels. It is interesting to note, however, that the non-ortho (77, 126, 169) PCB congeners contributed consistently and substantially to toxic body burdens (as measured with toxic equivalent units [TEQs]) in these mammals (see Fig. 13 in PCDD/F section, *Mink and river otter*).

Long-term monitoring of heron colonies in the Fraser estuary suggests that egg concentrations of PCBs declined precipitously a few years after strict regulations on use were imposed (1977), and fluctuated at lower levels from the early 1980s until 1994. Eggs collected in 1996 had the highest levels measured since 1983, however, it is too early to determine if this is an increasing trend (Fig. 5). PCBs in great blue heron eggs in 1977 and 1982 might have been sufficient to affect embryonic development, based on a lowest-observed-effect level (LOEL) for black-crowned night herons of 4.5 mg/kg (Hoffman *et al.* 1986), but more recently measured values were below toxicity thresholds. Similarly, recent total PCB residues in cormorants, raptors and tree swallows from the Fraser River Basin were all low and toxicologically inconsequential (Table 2). The early high concentrations in herons, however, imply that raptorial species in the basin probably also had toxicologically relevant residues during that period (1977–1982). Toxicological studies on osprey embryos from the upper Fraser River, undertaken in the 1995 and 1996 breeding seasons, showed that biochemical responses (hepatic EROD induction, elevated plasma vitamin A) were positively correlated with total PCB and PCB 126. No similar relationship was observed with hatching success. The implications for individual and population health are not known.

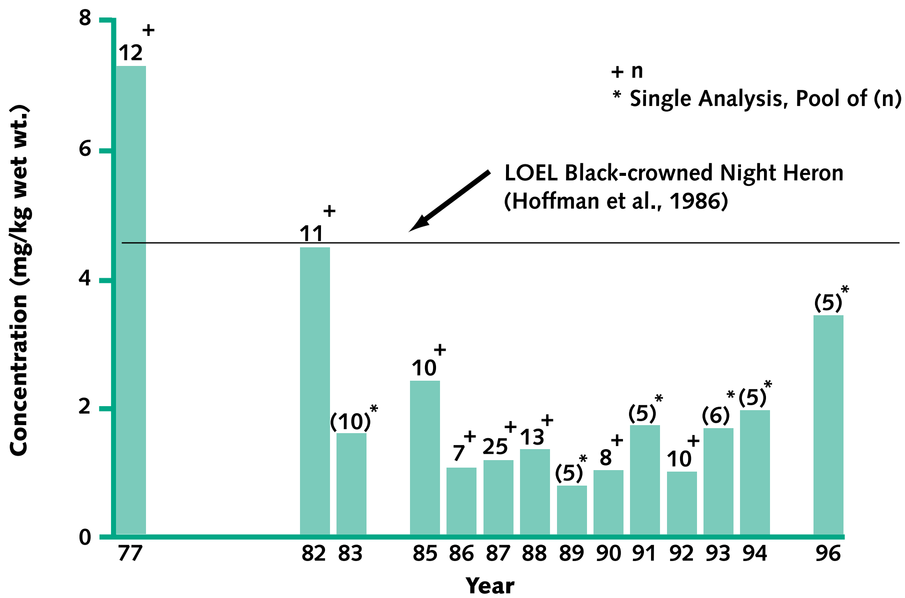

Figure 5. Total PCBs (geometric means, mg/kg wet wt.) in great blue heron eggs collected from the colony at the University of British Columbia, 1977–96. Number above bar graph represents the number of eggs analyzed per site per year.

LOEL = lowest-observed-effect level.

Even though total PCB residues in wildlife have dropped since monitoring began, the most toxic non-ortho congeners still contribute substantially to TEQs (see PCDD/F section). In fact, the relative contribution of specific PCB congeners to TEQs has increased over time for some species, reflecting the decreased presence of the more highly toxic PCDD/F congeners in the Fraser River. Thus, the toxic stress contributed by PCBs to wildlife will extend further into the future than the dioxins and furans released, in the past, from pulp mills.

Mercury

Mercury residues can persist from past industrial and agricultural releases. However, recent concerns focus on mobilization of mercury from man-made reservoirs (Hughes *et al.* 1997) and deposition of atmospherically transported mercury from coal-fired utilities and municipal waste incineration (Swain *et al.* 1992).

Higher mercury levels observed in bald eagles compared to great blue herons from the Fraser estuary may reflect dietary differences. A greater proportion of the eagles' diet is larger, older fish which may have accumulated more mercury than the year-old fish preferred by herons. Osprey from inland waterways have less mercury contamination than coastal eagles (Fig. 6) even though mercury concentrations are usually higher in freshwater versus marine environments (Elliott *et al.* 1989b) because eagles, in addition to consuming fish, also feed on other fish-eating birds thereby bioaccumulating a larger amount from the local food chain. Mercury concentrations in osprey eggs from the upper Fraser River were quite uniform among sites and were comparable to those reported in the literature for ospreys nesting on naturally formed lakes and rivers (Elliott *et al.,* in press). Based on the levels in osprey eggs from the Thompson and eagle eggs from coastal B.C. (Elliott *et al.,* in press), past mercury discharges from pulp mills have not had a lasting impact on local food chains.

Liver tissue in mink and river otter had substantially higher mercury levels than eggs from the fish-eating birds (Fig. 6). Mink and river otter collected from the Lower Fraser Valley showed greater average mercury values than those collected near Prince George, but the variability amongst individuals was very high. Two juvenile male mink from the upper basin were actually the most contaminated, with 4.2 and 4.3 mg/kg wet weight mercury in liver tissue (Harding *et al.* 1998). Although such concentrations would be insufficient to produce mercury poisoning (>24 mg/kg; Wren 1991), they might be within the range of chronic toxicity. Wren *et al.* (1987) discovered that a combination of 0.5 mg/kg methylmercury

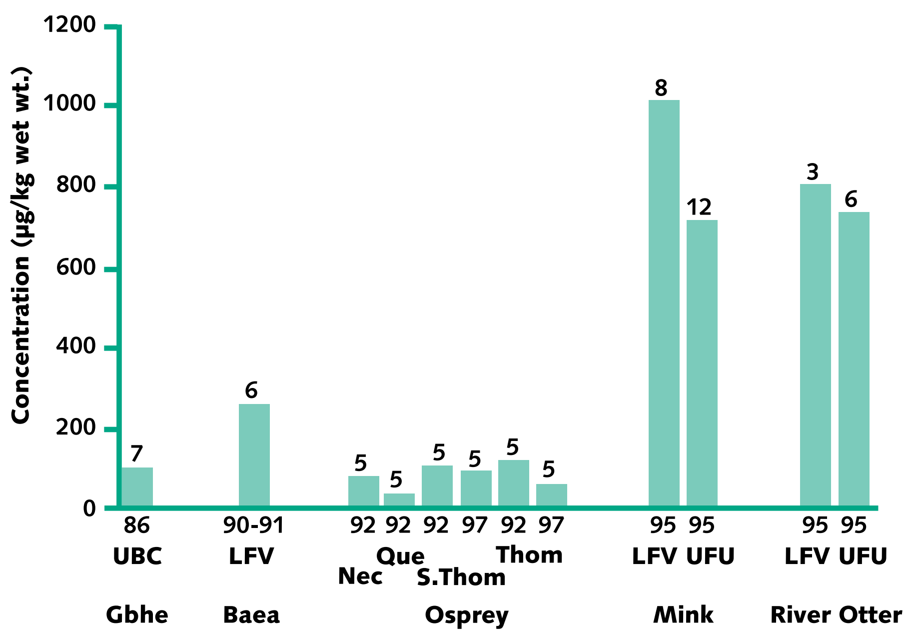

Figure 6. *Mercury (geometric means, µg/kg wet wt.) in eggs of great blue heron (gbhe), bald eagle (baea) and osprey, and in liver tissue of mink and otter from the Fraser River Basin. Number above bar graph represents the number of individuals analyzed.*

UBC=University of British Columbia in the Lower Fraser Valley; LFV=Lower Fraser Valley; nec=Nechako River; Que=Quesnel; S.Thom=S. Thompson River; Thom = Thompson River; UFV=upper Fraser Valley and 0.5 mg/kg Aroclor 1254 reduced mink kit survival even when neither compound, tested independently, elicited a measurable response. Since mink and river otter from this region were also typically contaminated with low levels of PCBs, the combined presence of mercury and PCBs could pose some risk to mustelids along the Fraser River. An examination of reproductive success and the presence of these contaminants over a wider area would be necessary for further confirmation.

Polychlorinated Dibenzo-p-dioxins (PCDDs), Dibenzofurans (PCDFs) and Toxic Equivalent Units (TEQs)

The major incentive for initiating residue studies of polychlorinated dibenzo-*p*-dioxins (PCDDs) and dibenzofurans (PCDFs) contamination in B.C. wildlife was to evaluate impacts of the persistent pollutants from the pulp and paper industry on wildlife at the top of the aquatic food chain. As monitoring progressed over the years, other PCDD/F sources, such as sites of past chlorophenolic-based wood preserving chemical production, were also identified as probable contributors to contaminants in wildlife. Much of the monitoring results are already published (Elliott *et al.* 1989a, b; 1996a, b, c; 1998a, b; in press; Elliott and Norstrom 1998; Sanderson *et al.* 1994a, b; Wilson *et al.* 1996).

Concentrations of major PCDD and PCDF congeners detected in key indicator species are listed in Tables 3 and 4 (see end of chapter), respectively. The major findings (residual patterns, associated toxicity) are summarized by species in the following subsections. The toxic equivalent (TEQ) methodology (using WHO Toxic Equivalent Factors [TEFs]; Van den Berg *et al.*, submitted) was used to evaluate the toxicological relevance of body burdens of PCDDs, PCDFs and PCBs.

Great blue herons and double-crested cormorants

A dramatic decline in egg concentrations of PCDDs and PCDFs between 1982 and 1989 was observed in the great blue heron colony near the University of British Columbia (UBC). The most toxic congener, 2,3,7,8-TCDD, however, peaked in 1989 and then dropped to near or below the detection limit by 1992. The timing of these declines (post-1990) suggests that implementation of chlorine dioxide substitution at upstream pulp mills was effective in reducing the discharge and subsequent concentrations of the tetrachlorinated congeners in prey and that body burdens in eagles responded quickly to these declines. The shorter data set for double-crested cormorants followed the same pattern from 1985, but as no eggs were collected in 1989, we cannot confirm the peak in 2,3,7,8-TCDD found in great blue heron that year.

Concentrations of pentachloro-dibenzo-*p*-dioxins (PnCDDs) and hexachloro-dibenzo-*p*-dioxins (HxCDDs) also declined in both species over time. Heron egg 1,2,3,6,7,8-HxCDD values dropped from hundreds of ng/kg in the early 1980s to an average of 23 ng/kg for 1991–94. The production of HxCDDs has been associated with the synthesis of pentachlorophenol (PCP) (Van den Berg *et al.* 1987) and use of chlorophenolic-treated wood in pulp mills. Reductions in HxCDD burdens in wildlife attest to the effectiveness of PCP use restrictions implemented in 1989.

Investigations of biological effects associated with contaminant burdens in B.C. heron eggs (Sanderson *et al.* 1994a) estimated that herons nesting at the UBC colony in 1988 were negatively impacted by PCDD/F contamination, but that reductions in body burdens over the next several years improved chick health. TEQs for great blue heron eggs were very high in the early 1980s (Fig. 7), TEQ contributions from PCDDs far exceeding those from PCBs. TEQs declined dramatically in 1983 but peaked again in 1989 when 2,3,7,8-TCDD releases in pulp mill effluents were at their maximum. Post-1990, TEQs were substantially reduced and PCBs dominated the total toxic burden in both herons and cormorants.

Although attempts to conduct toxicological studies on double crested cormorants, similar to those conducted for great blue heron, were unsuccessful in the cormorant colony from the Fraser River Estuary (Sanderson *et al.* 1994b), total TEQs in these cormorants (approximately 150 ng/kg in the ’80s and 50 ng/kg in the ’90s; Fig. 8) were always lower than the TEQs correlated with effects in a Lake Ontario cormorant colony. This implies that contamination from the Strait of Georgia, where the cormorants prefer to hunt, was not sufficient to affect their breeding success. Limited surveys of nest activity suggest that both herons (Gebauer 1995) and cormorants (Vermeer *et al.* 1989) have increased their use of the estuary for breeding.

Bald eagles

Six individual bald eagle eggs collected from the Lower Fraser Valley in 1990–91 contained higher total residues than heron and cormorant eggs collected in the same years. Although the proportions of 2,3,7,8-TCDD to 2,3,7,8-TCDF were comparable to those in residues found in fish tissues collected during that time period (Raymond *et al.* 1999), the levels of TCDF were much greater than those detected in other estuarine bird species. Together, the two congeners, most reflective of pulp mill contaminant sources, contributed 40–80 per cent of the total PCDD/F burden. In yolk sacs of chicks collected in 1992, the tetra-congeners contributed 11–25 per cent of the total PCDD/Fs. Investigations in other bird species suggest that 2,3,7,8-TCDF is eliminated quickly from the body (Norstrom *et al.* 1976); therefore, the elevated concentrations detected in bald eagle eggs and chicks were probably the result of recent exposure to pulp mill contaminated prey and direct deposition of contaminated lipids into the yolk (Elliott *et al.* 1996d).

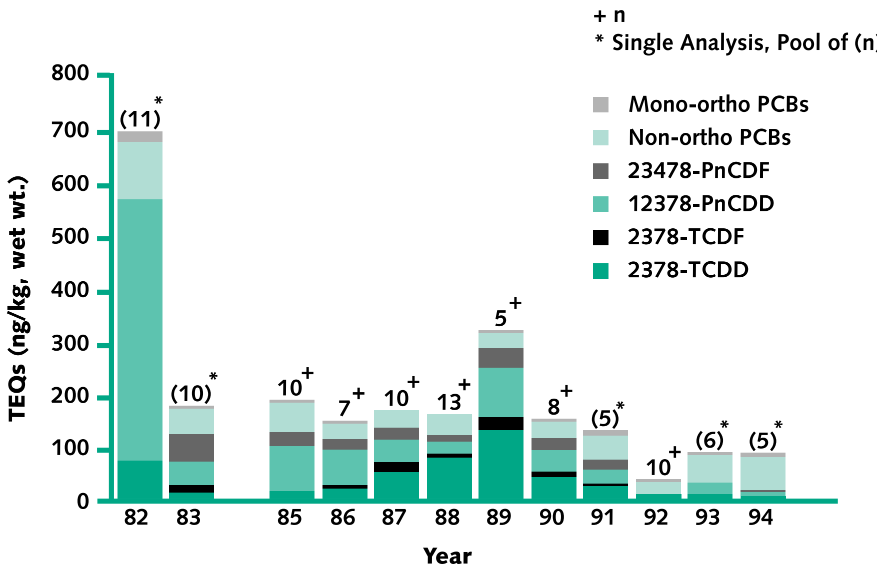

Figure 7. *Toxic equivalent units (TEQs) for major PCDDs, PCDFs and PCBs in great blue heron eggs collected from the University of British Columbia colony in the Lower Fraser Valley, 1982–1994. Calculations used WHO-TEFs. Number above bar graph represents the number of eggs analyzed per year.*

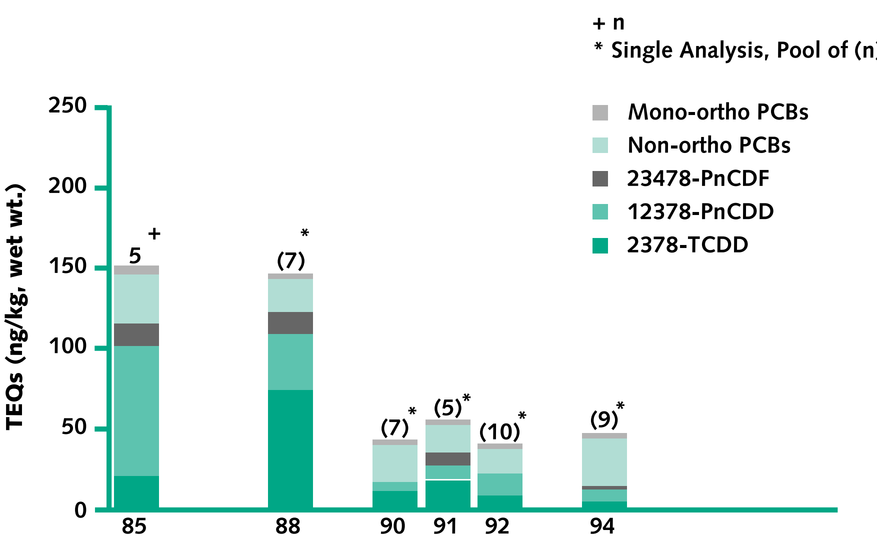

Figure 8. *Toxic equivalent units (TEQs) for major PCDDs, PCDFs and PCBs in double-crested cormorants eggs from the Fraser River Estuary, 1985–1994. Calculations used WHO-TEFs. Number above bar graph represents the number of eggs analyzed per year.*

A reasonable database on exposure and biological effects of PCDDs and PCDFs in bald eagles has been produced for B.C. (Elliott *et al.* 1996a, d; Elliott and Norstrom 1998). The no-observed-effect level (NOEL; 100 ng/kg TEQs) was exceeded in five of six eggs collected from the Lower Fraser Valley, while a lowest-observed-effect level (LOEL; 210 ng/kg TEQs) was exceeded in two eggs (Fig. 9). As suggested above, 2,3,7,8-TCDD was a major contributor to total TEQs, indicating that pulp mill effluents were still a dominant source of toxicity for this species in the early 1990s—probably via contaminated prey (*e.g.* fish). In nestling blood samples collected in 1993–94, the pattern of contamination was more skewed toward PCBs instead of TCDD (Elliott *et al.* 1996d; Elliott and Norstrom 1998). The PCDD/F burdens did not appear to affect reproductive success in Fraser Valley populations between 1990 and 1994. Most recent surveys (1992–97) estimated that average productivity of bald eagles in the lower basin was higher (1.1–1.2 young/nest) than the minimum 0.7 young/nest needed to maintain a population (Elliott and Norstrom 1998).

Osprey

The osprey, along with three other species (tree swallows, mink, river otter) collected from the upper reaches of the basin, were predominantly contaminated with octachloro-dibenzo-*p*-dioxin (OCDD) (Table 4). In particular, OCDD concentrations were high (374–472 ng/kg) in South Thompson River osprey, and still high, but markedly lower (91–94 ng/kg) in the Thompson River group, except for the single osprey egg sampled in 1995. OCDDs also contributed 60–85 per cent of the PCDD/F burden in Quesnel, Nechako and Pitt River eggs, although the absolute values were lower. Variability in values was extremely high, even between neighbouring nests, perhaps suggesting that minor differences in diet or feeding areas (*e.g.* mainstream versus tributaries or neighbouring lakes) have major repercussions for contaminant loads (Elliott *et al.* 1998b). A significant correlation between pentachlorophenol and both OCDD and 1,2,3,4,6,7,8-heptachloro-dibenzo-*p*-dioxin (HpCDD) concentrations in Thompson River eggs (OCDD: r=0.870, p<0.01; HpCDD: r=0.891, p<0.01; Fig. 10) strongly implicates a pentachlorophenol source for this contamination. The hypothesis is further supported by the pattern of trace HpCDFs in osprey eggs, which is characteristic of a pentachlorophenol source (Elliott *et al.* 1998b). Numerous primary timber processing facilities—major users of pentachlorophenols—are also located along the South Thompson and Thompson Rivers around the Kamloops area where the highest amount of elevated PCDDs were measured in osprey eggs.

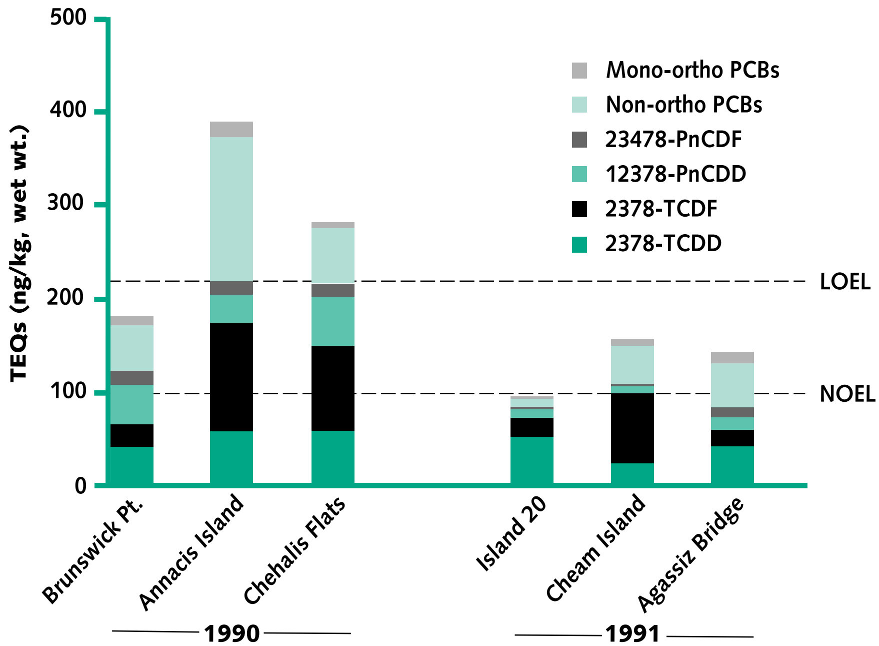

Figure 9. *Toxic equivalent units(TEQs, ng/kg wet wt) for major PCDDs, PCDFs and PCBs in individual bald eagle eggs from the Lower Fraser Valley, 1990–1991. Calculations used WHO-TEFs.*

NOEL: no-observed-effect level; LOEL: lowest-observed-effect level

There was also a strong upstream/downstream trend in the lower chlorinated PCDD/Fs in the Thompson River osprey eggs. Concentrations of 2,3,7,8-TCDD and 2,3,7,8-TCDF were greater in eggs collected downstream compared to upstream nests near the Kamloops pulp mill in 1991. As a result of changes in bleaching technology (chlorine dioxide substitution) at the mill, by 1997, concentrations of 2,3,7,8-TCDD, and -TCDF were significantly lower than in previous years in nests sampled downstream. Osprey along this stretch of the Thompson eat primarily mountain whitefish and largescale sucker, two species shown to carry large burdens of TCDD and TCDF during a 1988 fish survey (Mah *et al.* 1989). Between 1988 and 1992, concentrations of 2,3,7,8-TCDD in mountain whitefish muscle samples collected downstream of the Kamloops mill decreased from 61 to 1.7 ng/kg, while 2,3,7,8-TCDF levels dropped from 390 to 8.2 ng/kg (Hatfield Consultants Ltd. 1995). More recent samples in 1994 show levels have stopped declining with 2, 3, 7, 8-TCDD and -TCDF concentrations of 1.1 and 10.6 ng/kg, respectively (Raymond *et al.* 1999). The large-scale sucker is a bottom-feeder whose close association with sediments likely results in the bioaccumulation of PCDD/Fs deposited in sediments from mill effluent. In addition, mountain whitefish eat benthic invertebrates which in 1993 exhibited elevated tissue levels of OCDD and 2,3,7,8-TCDF downstream at Agassiz (Richardson and Levings 1996).

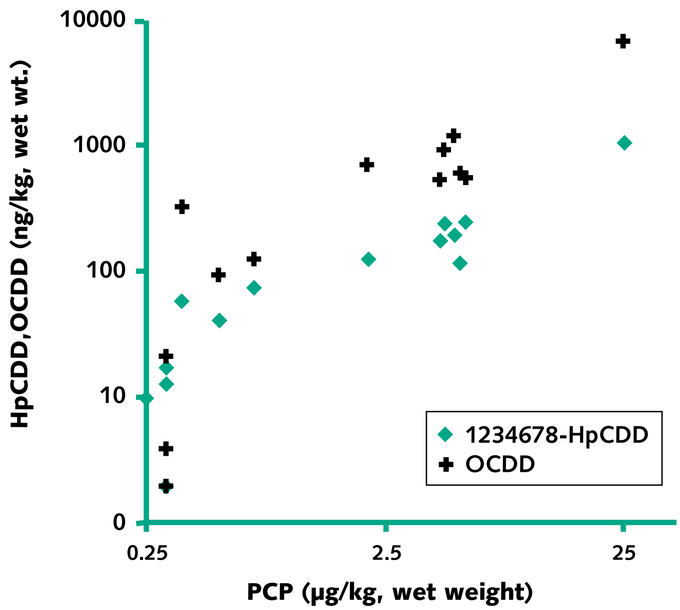

Figure 10. *Relationship between concentration of PCP (pentachlorophenol) and 1,2,3,4,6,7,8-HpCDD and OCDD in osprey eggs from the Fraser River Basin of British Columbia, 1991 (OCDD, $r=0.870$, $p<0.01$; HpCDD, $r=0.891$, $p<0.01$).*

Embryonic toxicological studies were undertaken in 1995 and 1996 breeding seasons which involved osprey chicks from upstream and downstream of the pulp mill at Kamloops along the South Thompson and Thompson rivers and in the Nechako and Pitt rivers (Elliott *et al.*, in prep). No differences were observed in laboratory hatching success between these areas. Hepatic EROD activity was greater (2.8-fold) in chicks collected downstream of Kamloops compared to the Nechako site, and correlated positively with total PCBs and PCB 126, but not 2,3,7,8-TCDD. A hepatic protein (cytochrome P4501A-like) was detected in all samples and correlated with EROD activity. Preliminary results suggest osprey chicks in the upper Fraser River exhibited toxicological response to chlorinated contaminant exposure. While no differences were observed in laboratory hatching success between the upstream and downstream study areas, eggs which did not hatch had, on average, a two-fold higher level of DDE compared to eggs which did hatch, regardless of collection area. The implication for individual and population health are not known.

The osprey eggs accumulated lower TEQs (Fig. 11) than the herons and bald eagles, principally because the TEF for OCDD is relatively small. Highest TEQs were calculated for osprey eggs collected downstream of the pulp mill at Kamloops, and, not surprisingly, those were dominated by contributions from 2,3,7,8-TCDD. Preliminary analyses showed that osprey clutch size, estimated in 1991, ’92, ’94 and ’97 surveys, was always greater in upstream versus downstream nests on the Thompson River. Similarly, fledgling success, estimated in 1992, ’94 and ’97, was always greater in upstream nests; but only fledgling success in 1992 was significantly greater in upstream nests (Machmer *et al.,* in prep). The low productivity of osprey downstream of pulp mills at Kamloops on the Thompson River was not caused by poor embryonic survival, as shown in the embryonic toxicological study where there was no difference in hatching success between areas. However, the study did show a positive correlation between biological effects (hepatic EROD induction, elevated plasma vitamin A) in the chicks and elevated PCB concentrations in the yolk sac suggesting that contaminants may influence reproductive success by adversely affecting chick survival.

Tree swallows

Tree swallow nestlings were collected in 1994 from nest boxes placed upstream and downstream of the pulp mills at Kamloops on the South Thompson and Thompson rivers and Prince George on the upper Fraser River. Analyses of pooled whole body homogenates detected elevated PCDDs and PCDFs in swallows downstream of Prince George mills (Table 4). The presence of highly chlorinated congeners, as well as the detection of pentachlorophenol (Table 5), is consistent with patterns shown in all other monitored wildlife species from the upper basin, and supports the suggestion that benthic prey species are accumulating highly chlorinated PCDDs from sediments exposed to runoff from areas of high chlorophenol use (Elliott *et al.*, 1998b). Kamloops samples and those upstream of Prince George showed negligible contamination of PCDD/Fs, yet all homogenates had detectable levels of 2,3,7,8-TCDF, suggesting that nestlings from both upstream and downstream of the mills were being fed contaminated insects.

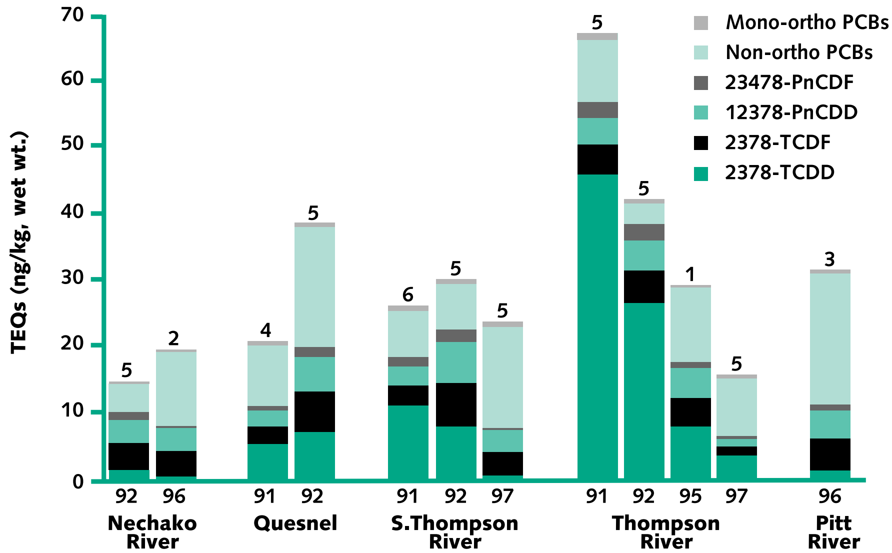

Figure 11. Toxic equivalent units (TEQs) for major PCDDs, PCDFs and PCBs in osprey eggs from the Fraser River Basin, 1991–97. Calculations used WHO-TEFs. Numbers above bars represent the number of eggs analyzed.

Tree swallow whole body homogenates had higher ratios of TCDF:TCDD compared to levels detected in eggs of most other birds studied (Table 4). TCDF is cleared rapidly by birds (Norstrom *et al.* 1976; Braune and Norstrom 1989), thus, the detection of this compound in birds collected near bleached kraft pulp mills results from dietary exposure to high concentrations in prey. TCDF may be directly deposited to the yolk from dietary lipids during the period of rapid yolk deposition in ovarian follicles (Elliott *et al.* 1998b). The comparatively high levels of TCDF detected in the tree swallows may also reflect the swallows' position in the food chain. Tree swallows consume emergent insects which acquire high burdens of TCDF from sediments. Fairchild *et al.* (1992) reported that emergent insects exported up to 2.1 per cent of TCDF burden in sediment one year after chemical addition (Elliott *et al.* 1998b). Raptorial species, such as ospreys, forage higher in the food chain, consuming prey which may have already degraded some of their TCDF burden. Alternatively, the different concentrations of TCDF detected in the various indicator species

Table 5. Relationship between concentrations of highly chlorinated PCDDs (OCDD and 1234678-HpCDD) and Pentachlorophenol in tree swallow nestlings (whole body homogenates) collected from upstream and downstream of pulp mills in British Columbia, 1994.

SITE[a]	N[b]	OCDD & 1234678-HpCDD (mg/kg wet wt)	PENTACHLOROPHENOL (µg/kg wet wt)
S. Thompson River (u/s Kamloops)	3p	4.89	NA[c]
Thompson River (d/s Kamloops)	11p	4.3	<1
upper Fraser River (u/s Pr. George)	18p	3.78	<1
upper Fraser River (d/s Pr. George)	14p	70.45	4

[a] u/s - upstream; d/s - downstream
[b] number of nestlings pooled in single analysis
[c] NA - not analyzed

may reflect the relative metabolic capability (inducible levels of a hepatic cytochrome P450 1A-like protein and associated ethoxyresorufin-o-deethylase activity) of the species to break down these contaminants. Osprey chicks had an enhanced ability to metabolize chlorinated hydrocarbons (Elliott and Trudeau, unpublished data) compared to bald eagle chicks (Elliott *et al.* 1996d) even though eagles were exposed to higher concentrations of contaminants (Elliott *et al.* 1998b).

The dominant PCDD/Fs in the tree swallows were the higher chlorinated congeners which, because of their low TEF value, did not contribute substantially to the overall TEQ values (Fig. 12). Non-ortho PCBs were a major contributor to tree swallow TEQ values, suggesting PCBs may be toxicologically more relevant than PCDD/Fs. Productivity data have not yet been analyzed, so the biological relevance of chlorinated contaminant exposure in these swallow populations cannot be assessed.

Mink and river otter

The difference in magnitude of contamination was substantial amongst otter groups collected from the Lower Fraser Valley in 1995 (Table 3) and the upper basin in 1991 and 1995 (Table 4). Total concentrations of PCDD/Fs, particularly OCCD, were at least 20-fold greater in two river otter composites from the Lower Fraser Valley compared to the otter composite from the upper Fraser River. Liver tissue from mink collected near Prince George and Quesnel had low levels of PCDD/Fs; only one individual (Cale Crk #1, Table 4), with elevated concentrations of 1,2,3,4,6,7,8-HpCDD (123 ng/kg) and OCDD (186 ng/kg), was roughly as contaminated with the highly chlorinated congeners as the Fraser Valley river otters. The dominant PCDD/F congener, as in osprey eggs and tree swallow nestlings, was OCDD, which suggests that the contaminant source for these mammals is similar and likely related to pentachlorophenol use. Otter from the Lower Fraser Valley had higher concentrations of higher chlorinated contaminants than otter from upper reaches of the Fraser River, which may reflect the greater prevalence of industrial activities along tributaries in the Lower Fraser Valley.

Mink and river otter from the Fraser River had low TEQs (Fig. 13). Mink TEQs were all low and dominated by PCBs, except for the one individual discussed above. All values were considerably lower than the 160 ng/kg TEQs estimated as a critical body residue for mink reproduction by Leonards *et al.* (1995). River otter TEQs were low in samples collected from the upper Fraser River; one composite collected in the Lower Fraser Valley was comparatively more contaminated, especially with 2,3,4,7,8-PnCDF and 1,2,3,6,7,8-HxCDD.

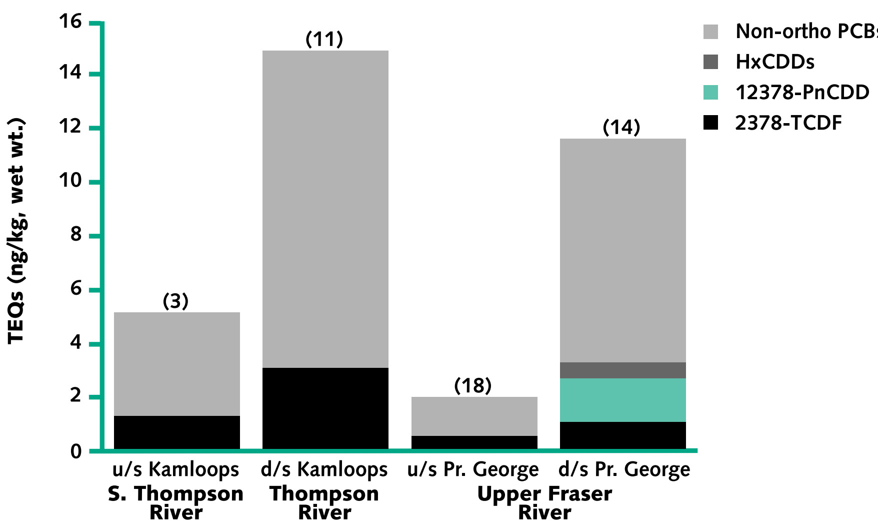

Figure 12. Toxic equivalent units(TEQs) for major PCDDs, PCDFs and PCBs in tree swallows collected upstream and downstream of pulp mills at Kamloops on the Thompson River system and Prince George on the Fraser River, 1994. Calculations used WHO-TEFs. Numbers above bars represent the number of whole body homogenates analyzed as a single pool.

Comparison of PCDD/F burdens and patterns with other regions

Elliott *et al.* (1996c) compared PCDD and PCDF residues in North American wildlife, and found that B.C.

populations were often among the most contaminated of monitored groups. However, heron, cormorant and bald eagle eggs collected in the Fraser River Basin were less contaminated than those in other areas of B.C., for example, colonies near coastal pulp mills such as Crofton on Vancouver Island, which was the most contaminated of all monitored colonies.

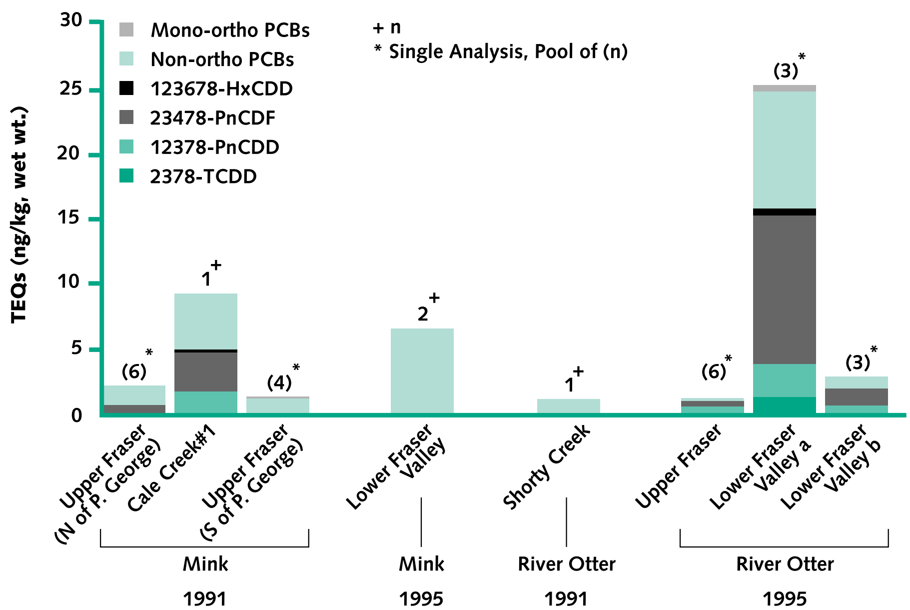

Figure 13. Toxic equivalent units (TEQs) for major PCDDs, PCDFs and PCBs in liver tissue of mink and river otter from the Fraser River Basin, 1991 and 1995. Calculations used WHO-TEFs.

Osprey, mink and river otter collected from the Fraser River Basin were generally less contaminated than Columbia River populations (Elliott *et al.* 1998 a, b; Harding *et al.* 1998). However, OCDD concentrations seen in wildlife from the upper Fraser River Basin were usually higher than those from Columbia River samples (except for two river otter and mink pools from the lower Columbia). Although mink and otter were collected proximate to pulp mills on both rivers, there was no pattern suggestive of pulp mill influences in the Fraser River animals, compared to the strong mill effluent patterns (higher levels downstream compared to upstream of mill site) in animals near Castlegar on the Columbia River as well as mink from Quebec (Champoux 1996). Fraser River mustelids had lower concentrations of PCDD and PCDFs compared to Quebec mink.

SUMMARY OF CONTAMINANT TRENDS IN SELECTED INDICATOR SPECIES

Great Blue Heron

Chlorinated contaminants (OC, PCBs, PCDD/Fs) in eggs from the great blue heron colony at the University of British Columbia in the Lower Fraser Valley have been monitored since the late 1970s.

DDE levels have declined from peaks in the late 1970s. The major source of DDE is from historical use of DDT for agricultural activities, although atmospheric sources also contribute to body burdens. Present levels are below toxic thresholds.

Concentrations of total PCBs in eggs declined a few years after regulatory controls (1977) and have fluctuated at lower levels from the early 1980s to 1994. While eggs collected in 1996 had the highest levels measured since 1983, it is too early to determine if this is an increasing trend. PCB concentrations measured in heron eggs in the late 1970s/early 1980s might have affected embryonic development, but more recent values are below toxicity thresholds.

A substantial decline in PCDD/F levels in heron eggs has been observed since 1989. In particular, concentrations of 2,3,7,8-TCDD/F dropped after implementation of chlorine dioxide substitution at upstream pulp mills, and levels of PnCDDs and HxCDDs diminished after bans on, and reduced usage of, chlorophenolic antisapstains and wood preservatives.

Investigations of biological effects associated with contaminant burdens in heron eggs estimated that, in 1989, herons were affected by PCDD/F contamination, but reductions in body burdens over the next several years greatly improved population health, including reproductive success.

Double-crested Cormorants

Eggs from the double-crested cormorant colony located at the Fraser estuary have been monitored for chlorinated contaminants since the mid 1980s.

Similar to trends in great blue herons, levels of DDE, PCBs and PCDD/Fs have declined in double-crested cormorant eggs since restrictions on their use were implemented.

Extensive biological studies have not been carried out for double-crested cormorants in the Fraser estuary colony. However, TEQs in the Fraser estuary colony were lower than TEQs correlated with biological effects in Lake Ontario, suggesting present levels of chlorinated contaminants are toxicologically inconsequential.

Bald Eagles

Eggs from bald eagle nests around the Lower Fraser Valley were collected in 1990 and 1991 and analyzed for chlorinated contaminants. Plasma from eagle nestlings was collected during the same time period.

Bald eagles had higher DDE concentrations than other species in the Lower Fraser Valley. Several eggs contained DDE levels within the range associated with reproductive effects, and mean eggshell thickness was less than values reported in the pre-DDT era (earlier than 1947). Although DDE concentrations may still be limiting productivity of some eagles nesting in the Lower Fraser Valley, impacts at the population level have been offset by other positive factors (such as abundant food supply) leading to increased survival of juveniles and adults.

Total PCB residues in eagle eggs collected in the early 1990s were below toxic thresholds.

Bald eagle eggs collected from the Lower Fraser Valley in 1990–91 contained higher total PCDD/F residues compared to those detected in ’90–91 heron and cormorant eggs. The proportion of 2,3,7,8-TCDD/F in eagle eggs was much greater than that detected in other estuarine birds, but levels were comparable to residues in fish tissues collected during that time period. As 2,3,7,8-TCDF is eliminated quickly from the body, the elevated levels detected in eagle eggs and chicks likely reflect recent exposure to pulp mill contaminated prey and direct deposition of contaminated lipids into the yolk.

The major contributor to total TEQs was 2,3,7,8-TCDD indicating that pulp mill effluent was still a dominant source of toxicity to eagles in the early 1990s. The PCDD/F burdens do not appear to affect reproductive success in Fraser Valley populations based on average productivity measured between 1990–94.

While bald eagle eggs had higher mercury levels than eggs from herons in the Lower Fraser Valley and osprey from the upper reaches of the Fraser River, levels were still below toxic thresholds.

Osprey

Osprey egg collections have been conducted primarily upstream and downstream of pulp mill sites along the Fraser and Thompson rivers since 1991.

Similar to eagle eggs, osprey eggs contained comparatively high amounts of DDE relative to other species. Twenty percent of eggs had concentrations exceeding toxic thresholds. Eggs with elevated DDE levels also showed evidence of egg shell thinning. Laboratory hatching success in an embryotoxicological study found that unhatched eggs contained, on average, two-fold higher DDE levels than eggs which hatched. These data suggest that DDE might be limiting productivity of some osprey nesting along the Fraser and Thompson rivers. Spatial trends suggest the primary local source of DDE for osprey is likely ongoing soil release of DDT compounds present from past usage in agricultural areas where DDT was historically used in large quantities. The high degree of individual variability within sites is likely caused by differences in diet, or related food chain factors, or by differences in exposure outside of breeding grounds.

Total PCBs, specifically PCB 126, were positively correlated with biochemical responses (hepatic EROD induction, elevated plasma vitamin A) in osprey embryotoxicity studies conducted in 1995 and 1996. The implications for individual and population health are not known.

Ospreys, along with other species collected from the upper reaches of the basin, were predominately contaminated with OCDDs. The higher chlorinated PCDDs are believed to have originated from a pentachlorophenol (PCP) source as there is a significant correlation between PCP and OCDD/HpCDD levels in eggs from the Thompson River. Tetra-chlorinated congeners, most reflective of pulp mill sources, were higher in eggs collected downstream of the pulp mill at Kamloops, although levels have declined since 1991 as a result of implementation of chlorine dioxide substitution at the mill.

TEQs for osprey were "low" compared to other species, principally because the TEF for OCDD is relatively small. Highest TEQs were observed in eggs collected downstream of pulp mills. Productivity (clutch size, fledgling success) was lower in downstream nests compared to upstream nests, although this difference has declined. However, embryonic studies undertaken in 1995 and 1996 showed no difference in hatching success betwen the two areas. The downstream chicks exhibited physiological responses to chlorinated contaminants (hepatic EROD induction, elevated plasma vitamin A); these responses were correlated to total PCBs and PCB 126 and not 2,3,7,8-TCDD. It is possible that PCBs negatively affect subsequent development characteristics like fledgling success.

Tree Swallows

Tree swallow nestlings were collected from nest boxes constructed upstream and downstream of pulp mills in the upper Fraser River in 1994.

DDE and total PCB levels were below toxic thresholds.

The dominance of highly chlorinated PCDD congeners relative to the other congeners and their correlation with PCP suggests that insect larvae are biomagnifying these compounds from contaminated runoff where chlorophenol-treated wood is used or manufactured.

Tree swallows had lower TEQs compared to osprey also collected from the same area, due to the lower concentration of tetra-congeners, especially 2,3,7,8-TCDD, associated with pulp mill contaminant sources. The toxicological significance of TEQs in whole-body homogenates is unknown.

Mink and River Otter

Mink and river otter were collected from the upper and lower reaches of the Fraser River in 1991 and 1995; liver tissues were analyzed for chlorinated contaminant residues.

DDE values were low compared to other species sampled in the same area and were toxicologically irrelevant.

Mink and otter from the Fraser Basin had low levels of PCBs compared to the Columbia River Basin.

Compared to river otter from the upper reaches of the Fraser River, otter from the Lower Fraser Valley had significantly higher concentrations of total PCDD/F, which were dominated by higher-chlorinated congeners, possibly reflecting the greater prevalence of industrial activities.

Similarly, river otter from the lower Fraser River had higher TEQs compared to those collected from upper reaches. All TEQ values in mink were below the critical body burden which is known to affect reproduction.

The highest mercury levels of all species tested were detected in river otter. Whether these levels pose a health risk to otters is unknown. However, the combined presence of mercury and PCBs could pose some risk to mink along the Fraser River.

General Conclusions on the Extent of Contamination in Fraser Basin Aquatic-Based Wildlife

Monitoring trends showed that during the 1980s, some indicator species in the Fraser River Basin had levels of chlorinated contaminants above concentrations associated with toxicological impairment. Contaminant levels declined very rapidly in the environment after regulations were implemented to eliminate chlorinated contaminants in mill effluent. However, contaminants still pose concern for a number of indicator species, notably DDE in osprey, TCDD in bald eagles and possibly PCBs in mink and otter. In addition, the world is a dynamic place and atmospheric transportation of specific contaminants may result in an increased concentration in local wildlife even though the chemical is banned from use in our country. Alternatively, it is possible that specific hot spots remain to be discovered (*e.g.* Cale Creek).

RESEARCH AND MONITORING RECOMMENDATIONS

Monitoring of contamination in indicator species should be continued to assess the persistence and bioavailability of contaminants in the Fraser River Basin.

Productivity and reproductive success of eagles, cormorants and herons at the presently monitored sites should continue on a three- to five-year rotation to take advantage of the baseline and trend data established over the last 20 years. This data would also provide information on broader ecological trends related to habitat disturbance and food chain productivity that need environmental management action beyond that required to control toxic contamination.

The utility of otter and mink as indicator species should be further examined. While they both “sample” the environment at different scales and points in the food chain, their use in combination might be very advantageous in the long term. The possibility of using live trapping for obtaining blood samples and observing condition factors is also attractive.

Although we have made advances in assessing the impacts of less-chlorinated dioxins and furans, the movement and toxicity of higher-chlorinated dioxins in the local food chains are not well understood. There is a need to evaluate these congeners which continue to be released to the environment from combustion sources, wood preserved with pentachlorophenol solutions and pulp mills.

Ospreys breeding on hydroelectric reservoirs can accumulate mercury levels two- to three-fold higher than birds on naturally formed water bodies (Hughes *et al.* 1997). A study of mercury accumulation in ospreys breeding on hydroelectric reservoirs in the Fraser Basin is warranted.

Ospreys eggs at DDE ‘hot spots’ (*e.g.* Nicola Lake, Moose Lake) should be assessed for DDE levels, fledgling success and general productivity.

REFERENCES

Aulerich, R. J. and R. K. Ringer. 1970. Some effects of chlorinated pesticides on mink. *American Fur Breeder* 46: 10–11.

Braune, B. M. and R. J. Norstrom. 1989. Dynamics of organochlorine compounds in herring gulls. 3. Tissue distribution and bioaccumulation in Lake Ontario gulls. *Environmental Toxicology and Chemistry* 8: 957–968

Brewer, R., M. Sekela, S. Sylvestre, T. Tuominen and G. Moyle. 1998. Contaminants in Bed Sediments from 15 Reaches of the Fraser River Basin. Aquatic and Atmospheric Sciences Division, Environment Canada, Vancouver, B.C. DOE-FRAP 1997-37.

Champoux, L. 1996. PCBs, dioxins and furans in hooded merganser (*Lophodytes cucullatus*), common merganser (*Mergus merganser*) and mink (*Mustela vison*) collected along the St. Maurice River near La Tuque, Quebec. *Environmental Pollution* 92: 147–153.

Donald, D. B., R. Bailey, R. Crosley, D. C. G. Muir, P. Shaw and J. Syrgiannis. 1993. *Polychlorinated Biphenyls and Organochlorine Pesticides in the Aquatic Environment Along the Continental Divide Region of Alberta and British Columbia*. Environment Canada, Regina Sask, 98 pp.

Elliott, J. E., D. G. Noble, R. J. Norstrom and P. E. Whitehead. 1989a. Organochlorine contaminants in seabird eggs from the Pacific coast of Canada, 1971–1986. *Enviromental Monitoring and Assessment* 12: 67–82.

Elliott, J. E., R. W. Butler, R. J. Norstrom and P. E. Whitehead. 1989b. Environmental contaminants and reproductive success of great blue herons, *Ardea herodias*, in British Columbia, 1986–87. *Environmental Pollution* 59: 91–114.

Elliott, J. E., K. M. Langelier, A. M. Scheuhammer, P. H. Sinclair and P. E. Whitehead. 1992. *Incidence of Lead Poisoning in Bald Eagles and Lead Shot in Waterfowl Gizzards from British Columbia, 1988–91*. Environment Canada. Canadian Wildlife Service Progress Note No. 200. 7 pp.

Elliott, J. E., L. K. Wilson, K. M. Langelier and R. J. Norstrom. 1996a. Bald eagle mortality and chlorinated hydrocarbon contaminants in livers from British Columbia, Canada, 1989–1994. *Environmental Pollution* 94(1): 9–18.

Elliott, J. E., R. J. Norstrom and G. E. J. Smith. 1996b. Patterns, trends, and toxicological significance of chlorinated hydrocarbon and mercury contaminants in bald eagle eggs from the Pacific coast of Canada, 1990–1994. *Archives of Environmental Contamination and Toxicol*ogy 31: 354–367.

Elliott, J. E., P. E. Whitehead, P. A. Martin, G. D. Bellward and R. J. Norstrom. 1996c. Persistent pulp mill pollutants in wildlife. In: M.R. Servos, K.R. Munkittrick, J.H. Carey, and G.J. Van Der Kraak, eds., *Environmental Fate and Effects of Pulp and Paper Mill Effluents*. St. Lucie Press, Delray Beach, Fla.

Elliott, J. E., R. J. Norstrom, A. Lorenzen, L. E. Hart, H. Philibert, S. W. Kennedy, J. J. Stegeman, G. D. Bellward and K. M. Cheng. 1996d. Biological effects of polychlorinated dibenzo-*p*-dioxins, dibenzofurans, and biphenyls in bald eagle (*Haliaeetus leucocephalus*) chicks. *Environmental Toxicology and Chemistry* 15(5): 782–793.

Elliott, J. E., L. K. Wilson, K. M. Langelier, P. Mineau and P. H. Sinclair. 1997. Secondary poisoning of birds of prey by the organophosphate insecticide, phorate. *Ecotoxicology* 6: 219–231.

Elliott, J. E. and R. J. Norstrom. 1998. Chlorinated hydrocarbon contaminants and productivity of bald eagle populations on the Pacific coast of Canada. *Environmental Toxicology and Chemistry* 17: 1142–1153.

Elliott, J. E., C. J. Henny, M. L. Harris, L. K. Wilson and R. J. Norstrom. 1998a. Chlorinated hydrocarbons in livers of American mink (*Mustela vison*) and river otter (*Lutra canadensis*) from the Columbia and Fraser River basins, 1990–1992. *Environmental Monitoring and Assessment* 00: 1–25.

Elliott, J. E., M. Machmer, C. J. Henny, L. K. Wilson and R. J. Norstrom. 1998b. Contaminants is Ospreys from the Pacific Northwest: I Trends and patterns in polychlorinated dibenzo-p-dioxins and dibenzofuran contaminants in eggs and plasma. *Archives of Environmental Contamination and Toxicol*ogy 35: 620–631.

Elliott, J. E., M. Machmer, L. K. Wilson and C. J. Henny. In press. Contaminants in Ospreys from the Pacific Northwest: II. Organochlorine Pesticide, Polychlorinated Biphenyl and Mercury Contaminants, 1991–1997. *Archives of Environmental Contamination and Toxicol*ogy.

Elliott, J. E., L. K. Wilson, C. Henny, S. Trudeau, S. W. Kennedy, F. A. Leighton and K. M. Cheng. In prep.(a) Biological effects of chlorinated hydrocarbon contaminants in osprey chicks.

Elliott, J. E. and S. Trudeau. Unpublished data. Canadian Wildlife Service, Environment Canada, Delta, B.C.

Fairchild, W. L., D. C. G. Muir, R. S. Currie and A. L. Yarechewski. 1992. Emerging insects as a biotic pathway for movement of 2,3,7,8-tetrachlorodibenzofuran from lake sediments. *Environmental Toxicology and Chemistry* 11: 867-872.

Fry, D. M. 1995. Reproductive effects in birds exposed to pesticides and industrial chemicals. *Environmental Health Perspectives* 103(Suppl.7): 165–171.

Gebauer, M. B. 1995. *Status and Productivity of Great Blue Heron (Ardea herodias) in the Lower Fraser River Valley in 1992, 1993, and 1994*. Report prepared by Enviro-Pacific Consulting, Surrey, B.C., for BC Ministry of Environment, Lands and Parks, Surrey, B.C.

Harding, L., M. L. Harris and J. E. Elliott. 1998. Heavy and trace metals in wild mink (*Mustela vison*) and river otter (*Lutra canadensis*) captured on rivers receiving metals discharges. *Bulletin of Environmental Contamination and Toxicology*. 61: 600-607.

Harding, L., M. L. Harris, C. Stephen, and J. E. Elliott. 1999 (in press). Reproductive and morphological condition of wild mink (*Mustela vison*) and river otter (*Lutra canadensis*) in relation to chlorinated hydrocarbon contamination. *Environmental Health Perspectives* 107(2).

Harfenist, A., P. E. Whitehead, W. J. Cretney and J. E. Elliott. 1993. *Food Chain Sources of Polychlorinated Dioxins and Furans to Great Blue Herons (Ardea herodias) Foraging in the Fraser River Estuary, British Columbia*. Technical Report Series No. 169. Canadian Wildlife Service, Pacific and Yukon Region, Delta, B.C.

Hatfield Consultants Ltd. 1995. *Weyerhauser (Kamloops Pulp Mill) Environmental Effects Monitoring (EEM) Pre-Design Reference Document*. Hatfield Consultants, West Vancouver, B.C.

Heaton, S. N., S. J. Bursian, J. P. Giesy, D. E. Tillitt, J. A. Render, P. D. Jones, D. A. Verbrugge, T. J. Kubiak and R. J. Aulerich. 1995. Dietary exposure of mink to carp from Saginaw Bay. I. effects on reproduction and survival and potential risks to wild mink populations. *Archives of Environmental Contamination and Toxicol*ogy 28: 334–343.

Henny, C. J., R. A. Grove and O. R. Hedstrom. In prep. Hypoplastic reproductive organs related to xenobiotic compounds in young male river otter from the Columbia River.

Hoffman, D. J., B. A. Rattner, C. M. Bunck, A. Krynitsky, H. M. Ohlendorf and R. W. Loew. 1986. Association between PCBs and lower embryonic weight in black-crowned night herons in San Francisco Bay. *Journal of Toxicology and Environmental Health* 19: 383–391.

Hughes K. D., P. J. Ewins and K. E. Clark. 1997. A comparison of mercury levels in feathers and eggs of osprey (*Pandion haliaetus*) in the North American Great Lakes. *Archives of Environmental Contamination and Toxicology* 33: 441–452

Leonards, P. E. G., M. D. Smit, A. W. J. J. de Jongh and B. van Hattum. 1994. *Evaluation of Dose-response Relationships For the Effects of PCBs on the Reproduction of Mink* (*Mustela vison*). Report No. R-94/6, Institute for Environmental Studies, Vrije Universiteit, Amsterdam, The Netherlands.

Leonards, P. E. G., T. H. de Vries, W. Minnaard, S. Stuijfzand, P. de Voogt, W. P. Cofino, N. M. van Straalen and B. van Hattum. 1995. Assessment of experimental data on PCB-induced reproduction inhibition in mink, based on an isomer- and congener-specific approach using 2,3,7,8-tetrachlorodibenzo-*p*-dioxin toxic equivalency. *Environmental Toxicology and Chemistry* 14(4): 639–652.

Macdonald, R. W., D. P. Shaw and C. B. J. Gray. 1999. Contaminants in lake sediments and fish. In: C. Gray and T. Tuominen, eds. *Health of the Fraser River Aquatic Ecosystem: A Synthesis of Research Conducted Under the Fraser River Action Plan*. Environment Canada, Vancouver, B.C. DOE FRAP 1998-11.

Machmer, M. M., J. E. Elliott, L. K. Wilson and C. J. Henny. In prep. Contaminants in ospreys from the Pacific Northwest: III. An ecotoxicological assessment.

Mah, F. T. S., D. D. MacDonald, S. W. Sheehan, T. M. Tuominen and D. Valiela. 1989. *Dioxins and Furans in Sediment and Fish from the Vicinity of Ten Inland Pulp Mills in British Columbia*. Environment Canada, Conservation and Protection, North Vancouver, B.C.

Mason, C. F. 1989. Water pollution and otter distribution: a review. *Lutra* 32: 97–131.

Melquist, W. E. and A. E. Dronkert. 1987. River otter. In: M. Novak, J.A. Baker, M.E. Obbard, and B. Malloch, eds. *Wild Furbearer Management and Conservation in North America*. Ontario Trappers Association, Ontario Ministry of Natural Resources, Toronto, Ont .

Moore, D. R. J. and S. L. Walker. 1991. Canadian Water Quality Guidelines for Polychlorinated Biphenyls in Coastal and Estuarine Waters. Environment Canada. Ottawa. Scientific Series No. 186. 61pp.

Moul, I. E., L. M. Nichol, J. M. Landucci and M. Hanawa. 1996. *Assessment of Potential Indicator Species for Monitoring Environmental Contamination of the Fraser River Basin, British Columbia*. Environment Canada, Vancouver, B.C. DOE FRAP 1996-06.

Noble, D. G., J. E. Elliott and J. L. Shutt. 1993. *Environmental Contaminants in Canadian Raptors, 1965–1989*. Technical Report Series No. 91. Canadian Wildlife Service, Ottawa, ON, Canada.

Norstrom, R. J., R. W. Riseborough and D. J. Cartwright. 1976. Elimination of chlorinated dibenzofurans associated with polychlorinated biphenyls fed to mallards (*Anas platyrhynchos*). *Toxicology and Applied Pharmacol*ogy 37: 217–228.

Poole, A. F. 1989. *Ospreys: A Natural and Unnatural History*. Cambridge University Press, Cambridge, England.

Raymond, B. A., D. P. Shaw and K. Kim. 1999. Fish Health Assessment. In: C. Gray and T. Tuominen, eds. *Health of the Fraser River Aquatic Ecosystem: A Synthesis of Research Conducted Under the Fraser River Action Plan*. Environment Canada, Vancouver, B.C. DOE FRAP 1998-11.

Richardson, J. S. and C. D. Levings. 1996. Chlorinated organic contaminants in benthic organisms of the lower Fraser River, British Columbia. *Water Quality Research Journal of Canada*. 31: 153–162.

Sanderson, J. T., J. E. Elliott, R. J. Norstrom, P. E. Whitehead, L. E. Hart, K. M. Cheng and G.D. Bellward. 1994a. Monitoring biological effects of polychlorinated dibenzo-*p*-dioxins, dibenzofurans, and biphenyls in great blue heron chicks (*Ardea herodias*) in British Columbia. *Journal of Toxicology and Environmental Health* 41: 435–450.

Sanderson, J. T., R. J. Norstrom, J. E. Elliott, L. E. Hart, K. M. Cheng and G. D. Bellward. 1994b. Biological effects of polychlorinated dibenzo-*p*-dioxins, dibenzofurans, and biphenyls in double-crested cormorant chicks (*Phalacrocorax auritus*). *Journal of Toxicology and Environmental Health* 41: 247–265.

Smit, M. D., P. E. G. Leonards, B. van Hattum and A. W. J. J. de Jongh. 1994. *PCBs in European Otter (Lutra lutra) Populations*. Report No. R-94/7. Institute for Environmental Studies, Vrije Universiteit, Amsterdam, The Netherlands.

Swain, E. B., D. R. Engstrom, M. E. Brigham, T. A. Henning and P. L. Brezonik. 1992. Increasing rates of atmoshperic mercury deposition in midcontinental North America. *Science* 257: 784–787.

Van den Berg, M., F. Blank, C. Heeremans, H. Wagenaar and K. Olie. 1987. Presence of polychlorinated dibenzo-*p*-dioxins and polychlorinated dibenzofurans in fish-eating birds and fish from the Netherlands. *Archives of Environmental Contamination and Toxicol*ogy 16: 149–158.

Van den Berg, M., M., L. Birnbaum, A. T. C. Bosveld, B. Brunström, P. Cook, M. Feeley, J. P. Giesy, A. Hanberg, R. Hasegawa, S.W. Kennedy, T. Kubiak, J. C. Larsen, F. X.R. van Leeuwen, A. K. D. Liem, C. Nolt, R. E. Peterson, L. Poellinger, S. Safe, D. Schrenk, D. Tillitt, M. Tysklind, M. Younes, F. Wærn, T. Zacharewski. Submitted. Toxic Equivalency Factors (TEFs) for PCBs, PCDDs, PCDFs for Humans and Wildlife. *Environmental Health Perspectives*.

Vermeer, K., K. H. Morgan and G. E. J. Smith. 1989. Population trends and nesting habitat of double-crested and pelagic cormorants in the Strait of Georgia. In: K. Vermeer and R.W. Butler, eds. *The Ecology and Status of Marine and Shoreline Birds in the Strait of Georgia, British Columbia*. Special Publication. Canadian Wildlife Service, Pacific and Yukon Region, Delta, B.C.

Wiemeyer, S. N., C. M. Bunck and A. J. Krynitsky. 1988. Organochlorine pesticides, polychlorinated biphenyls, and mercury in osprey eggs—1970–79—and their relationships to shell thinning and productivity. *Archives of Environmental Contamination and Toxicology* 17: 767–787.

Wiemeyer, S. N., C. M. Bunck and C. J. Stafford. 1993. Environmental contaminants in bald eagle eggs—1980–84—and further interpretations of relationships to productivity and shell thickness. *Archives of Environmental Contamination and Toxicology* 24: 213–227.

Wilson, L. K., J. E. Elliott and P. E. Whitehead. 1996. *Chlorinated Compounds in Wildlife From the Fraser River Basin*. Environment Canada, Vancouver, B.C. DOE FRAP 1996-09.

Wilson, L., M. Harris and J. Elliott. 1999. Impact of agricultural pesticides on birds of prey in the Lower Fraser Valley. In: C. Gray and T. Tuominen, eds. *Health of the Fraser River Aquatic Ecosystem: A Synthesis of Research Conducted Under the Fraser River Action Plan*. Environment Canada, Vancouver, B.C. DOE FRAP 1998-11.

Wise, M. H., I. J. Linn and C. R. Kennedy. 1981. A comparison of the feeding biology of mink *Mustela vison* and otter *Lutra lutra*. *Journal of Zoology* (London) 195: 181–213.

Wren, C. D. 1991. Cause-effect linkages between chemicals and populations of mink (*Mustela vison*) and otter (*Lutra canadensis*) in the Great Lakes basin. *Journal of Toxicology and Environmental Health* 33: 549–585.

Wren, C. D., D. B. Hunter, J. F. Leatherland and P. M. Stokes. 1987. The effects of polychlorinated biphenyls and methylmercury, singly and in combination on mink. II: reproduction and kit development. *Archives of Environmental Contamination and Toxicology* 16: 449–454.

Table 1. Concentrations of organochlorine pesticides (µg/kg wet weight) in wildlife from the Fraser River Basin. Values are geometric means (of 'N' samples) or one analysis of a sample composite.

SPECIES	SITE	YEAR	N[a]	DDE	DDT	OXY[b]	t-NONA-CHLOR	DIELDRIN	HE[c]	HCB[d]	β-HCH[e]	MIREX
Great blue heron (eggs)	UBC	1977	12	2354	66	10	NA[f]	73	64	20	NA	22
		1982	11	1424	77	50	70	72	38	NA	10	12
		1983	10p	582	NA	23	22	20	35	NA	NA	NA
		1985	10	1047	19	37	112	49	32	17	3	16
		1986	7	273	9	11	27	10	7	11	NA	5
		1987	25	510	24	25	60	21	16	15	NA	6
		1988	13	447	9	17	36	17	13	13	4	9
		1989	5p	531	16	21	47	19	17	14	5.2	4
		1990	8	415	12	11	27	13	6	7.0	4.0	4
		1991	5p	808	22	28	43	13	8	15	11	13
		1992	10	280	1	11	30	10	5	10	6.0	4
		1993	6p	1622	19	31	59	54	14	14	11	11
		1994	5p	1020	31	27	34	18	52	9.4	5.6	7
		1996	5p	926	10	22	44	58	28	13	<0.1[g]	11
Double-crested cormorant (eggs)	Estuary	1985	5	464	5	12	NA	4	3	14	6	8
		1988	7	531	1	3	2	2	1	15	5	5
		1990	7p	176	1	6.2	<0.3	6	2	7.7	7.0	3
		1991	5p	236	<0.1	6.2	<0.3	5	2	8.6	5.6	4
		1992	10p	222	2	5.6	<0.3	2	2	9.4	4.6	3
		1993	10p	382	2	5.2	0.4	1	2	9.4	4.1	3
		1994	9p	411	1	6.0	0.3	1	2	7.6	4.2	2
Bald eagle (eggs)	Brunswick Point	1990	1	2536	NA	33	135	66	69	16	11	9.7
	Annacis Island	1990	1	4142	NA	82	165	91	38	42	32	38
	Chehalis Flats	1990	1	4912	NA	31	234	44	6.5	18	3.1	14
	Island 20	1991	1	898	9.1	42	82	24	11	21	7.2	<1.1
	Cheam Island	1991	1	1171	<0.4	35	125	20	4.0	30	<0.5	17
	Agassiz Bridge	1991	1	1916	11	22	151	21	10	37	6.7	15

Table 1. Continued from previous page.

SPECIES	SITE	YEAR	N[a]	DDE	DDT	OXY[b]	t-NONA-CHLOR	DIELDRIN	HE[c]	HCB[d]	β-HCH[e]	MIREX
Osprey (eggs)	Nechako R.	1992	5	1384	20	1.9	0.8	3.5	2.8	1.1	3.2	0.8
		1996	2	2357	10	3.5	<1.1	0.5	3.4	<0.8	0.5	<1.0
	Quesnel	1991	4	1014	NA	<8.5	NA	0.1	1.8	0.2	NA	NA
		1992	5	1309	3.1	2.7	<0.5	5.2	3.0	0.1	0.4	2.0
	S. Thompson R.	1991	6	1174	NA	<6.7	NA	0.1	1.6	1.2	NA	NA
		1992	5	1800	28	3.7	1.2	1.5	2.4	3.0	1.3	3.3
		1997	5	1194	10	0.8	0.7	0.1	0.5	0.5	<1	1.4
	Thompson R.	1991	5	1646	NA	0.1	NA	0.7	2.7	0.6	NA	NA
		1992	5	1294	15	4.2	2.3	2.2	4.1	2.6	4.2	2.8
		1995	1	3173	18	3.8	<1.1	<0.2	2.5	<0.8	5.8	<1
		1997	5	706	1.1	1.8	0.3	0.2	0.4	0.3	<1	0.5
	Pitt R.	1996	3	5429	<0.2	3.7	<1.1	0.3	1.4	<0.8	0.2	<1
Tree swallow (nestlings)	S. Thompson R.	1994	3p	1783	ND	1.4	<0.1	0.8	0.7	<0.05	<0.05	<0.15
	Thompson R.	1994	11p	87	ND	1.5	<0.1	1.2	ND	<0.05	<0.05	<0.15
	upper Fraser R. - u/s	1994	18p	90	ND	1.3	<0.1	0.4	0.4	2.2	<0.05	<0.15
	upper Fraser R. - d/s	1994	14p	165	ND	1.5	<0.1	0.3	1.1	<0.05	<0.05	<0.15
River otter (liver)	Shorty Creek	1991	1	<1.0	<1.0	1.7	<1.0	1.3	<1.0	2.7	<1.0	<1.0
	upper Fraser R.	1995	6	<2	<2	NA	NA	<2	5.8	NA	NA	NA
	Lower Fraser Valley	1995	3	9.4	<2	NA	NA	<2	5.0	NA	NA	NA
Mink (liver)	upper Fraser R.	1991	6p	3.7	<1.0	1.7	<1.0	<1.0	<1.0	<1.0	<1.0	<1.0
	Cale Creek 1	1991	1	6.9	<1.0	>0.1[h]	<1.0	<1.0	<1.0	<1.0	<1.0	>0.1
	Cale Creek 2	1991	1	9.0	<1.0	1.0	<1.0	>0.1	<1.0	1.0	<1.0	<1.0
	Cale Creek 3	1991	1	1.4	<1.0	3.6	<1.0	1.2	<1.0	<1.0	<1.0	<1.0
	Redrock Lake	1991	1	1.3	<1.0	<1.0	<1.0	2.3	<1.0	<1.0	<1.0	<1.0
	Stone Ck Canyon	1991	1	4	<1.0	<1.0	<1.0	<1.0	<1.0	<1.0	<1.0	<1.0
	upper Fraser R.	1995	12	6	1.4	NA	NA	1.4	<2	NA	NA	NA
	Lower Fraser Valley	1995	8	31	<2	NA	NA	4.3	<2	NA	NA	NA

[a] *'p' = one pool of listed number of individual eggs or whole body homogenates*
[b] *Oxy = oxychlordane*
[c] *HE = heptachlor epoxide*
[d] *HCB = hexachlorobenzene*
[e] *b-HCH = b-hexachlorocyclohexane*
[f] *NA = not available*
[g] *Detection limit is contaminant specific therefore listed individually as '< particular value'*
[h] *Non-quantifiable peak observed below the calculated minimum detection limit, but still indicating presence of the compound*

Table 2. Concentrations of total PCBs in wildlife from the Fraser River Basin. Values are geometric means (of 'N' samples) or one analysis of a sample composite.

SPECIES	SITE	YEAR	N[a]	TOTAL PCBS (mg/kg wet wt.)	NON-ORTHO PCBS (ng/kg wet wt.)			MONO-ORTHO PCBS (µg/kg wet wt.)	
					77	126	169	105	118
Great blue heron (eggs)	UBC	1977	12	7.31	NA[b]	NA	NA	NA	NA
		1982	11	4.44	NA	1055	132	107	536
		1983	10p	1.57	NA	392	54	37	190
		1985	10	2.39	NA	582	77	42	297
		1986	7	1.05	NA	273	41	30	113
		1987	25	1.23	NA	314	45	3.4	18
		1988	13	1.37	NA	345	49	9.3	35
		1989	5p	0.79	NA	212	33	0.2	97
		1990	8	1.00	178	319	47	17	118
		1991	5p	1.76	NA	436	60	33	265
		1992	10	1.01	NA	263	39	22	102
		1993	6p	1.70	NA	423	58	24	230
		1994	5p	1.92	988	653	29	41	275
		1996	5p	3.41	NA	NA	NA	NA	NA
Double-crested cormorant (eggs)	Estuary	1985	5	1.23	NA	336	48	27	141
		1988	7	0.88	NA	213	34	13	151
		1990	7p	0.74	57	252	32	16	92
		1991	5p	0.76	NA	172	30	16	111
		1992	10p	0.75	NA	167	29	17	96
		1994	9p	0.71	11	317	27	12	78
Bald eagle (eggs)	Brunswick Point	1990	1	2.87	282	446	84	44	202
	Annacis Island	1990	1	6.21	605	1070	221	91	429
	Chehalis Flats	1990	1	2.96	291	462	87	46	194
	Island 20	1991	1	1.09	110	113	11	18	100
	Cheam Island	1991	1	2.33	230	346	62	36	191
	Agassiz Bridge	1991	1	2.73	269	420	78	45	240

Table 2. Continued from previous page.

SPECIES	SITE	YEAR	N[a]	TOTAL PCBS (mg/kg wet wt.)	NON-ORTHO PCBS (ng/kg wet wt.)			MONO-ORTHO PCBS (µg/kg wet wt.)	
					77	126	169	105	118
Osprey (eggs)	Nechako R.	1992	5	0.10	21	37	18	0.8	6.6
		1996	2	0.35	49	87	23	0.9	1.6
	Quesnel	1991	4	0.27	35	74	15	6.2	22
		1992	5	0.29	39	166	26	1.1	32
	S. Thompson R.	1991	6	0.37	50	55	25	7.8	28
		1992	5	0.38	32	138	37	4.0	23
		1997	5	0.25	54	94	26	4.9	15
	Thompson R.	1991	5	0.30	16	59	11	5.7	22
		1992	5	0.29	37	77	15	4.5	18
		1995	1	0.23	8.6	27	94	<1.4[c]	15
		1997	5	0.20	33	71	11	3.8	11
	Pitt R.	1996	3	0.44	44	179	23	0.3	26
Tree swallow (nestlings)	S. Thompson R.	1994	3p	<0.1	59	9	0.8	<0.15	<0.2
	Thompson R.	1994	11p	0.03	163	35	2.0	<0.15	7.2
	upper Fraser R. - u/s	1994	18p	<0.1	18	4.6	1.0	<0.15	<0.2
	upper Fraser R. - d/s	1994	14p	0.02	80	42	2.8	<0.15	<0.2
River otter (liver)	Shorty Creek	1991	1	38	3.5	5.8	18	<0.3	<0.3
	upper Fraser R.	1995	6p	0.02	2	1	6	<0.02	<0.03
	Lower Fraser Valley a	1995	3p	0.2	7	68	36	3.3	1.5
	Lower Fraser Valley b	1995	3p	0.2	2	6	9	<0.02	0.4
Mink (liver)	upper Fraser R.	1991	6p	4.1	4.2	14.4	<1.8	<0.1	<0.1
	Cale Creek 1	1991	1	18	3.3	39.3	4.6	<0.1	<0.1
	Cale Creek 2	1991	1	24	3.4	13.5	3.2	<0.1	<0.1
	Cale Creek 3	1991	1	5.5	2.6	6.7	4.0	<0.1	<0.1
	Redrock Lake	1991	1	<1	4.4	6.8	3.1	<0.1	<0.1
	Stone Ck Canyon	1991	1	5.0	3.3	2.9	1.3	17	29
	Lower Fraser Valley	1995	2	0.02	18	62	<5	0.4	1.4

[a] *'p' = one pool of listed number of individual eggs or whole body homogenates*
[b] *NA = not available*
[c] *Detection limit is contaminant specific therefore listed individually as '< particular value'*

Table 3. Major PCDDs and PCDFs (ng/kg wet wt.) in heron, cormorant and eagle eggs and mustelid liver tissue collected from sites within the lower Fraser River Basin. Values are geometric means (of 'N' samples), one analysis of a composite/pool of samples or one analysis of an individual.

SPECIES	SITE	YEAR	N[a]	2378-TCDD	12378-PnCDD	123678-HxCDD	1234678-HpCDD	OCDD	2378-TCDF	23478-PnCDF	TEQS[b]
Great blue heron (eggs)	UBC	1982	11p	76	491	740	31	76	NA[e]	NA	700
		1983	10p	15	59	104	ND[c]	27	11	48	179
		1985	10	22	85	125	2.0	0.3	0.2	18	191
		1986	7	25	74	104	3.5	5.9	3.0	16	150
		1987	10	55	49	83	1.3	0.2	17	16	169
		1988	13	86	27	37	2.8	3.5	5.7	10	165
		1989	5	133	103	112	3.0	1.9	19	38	319
		1990	8	46	47	52	0.6	0.8	7.5	18	153
		1991	5p	29	34	33	<3[d]	<5	4.0	15	132
		1992	10	9.2	5	8.3	0.1	0.1	0.3	0.003	40
		1993	6p	9.7	35	33	14	11	<2.8	<2.7	92
		1994	5p	4.4	12	18	<0.8	13	1.2	4.5	95
Double-crested cormorant (eggs)	Estuary	1985	5	20	81	188	0.04	<7.2	ND	13	152
		1988	7p	74	34	52	8	6	3	14	151
		1990	7p	11	5	7	<1	<10	<2	<2	44
		1991	5p	18	10	23	<6	<10	<1	8	56
		1992	10p	8	14	11	<0.1	<5.2	<1	<3.5	42
		1994	9p	3.3	8.2	11	1.0	1.4	<1.3	2.3	48
Bald eagle (eggs)	Brunswick Point	1990	1	42	37	42	<2	<3	23	13	182
	Annacis Island	1990	1	58	55	112	9	<4	112	12	390
	Chehalis Flats	1990	1	58	52	55	<6	<10	89	14	282
	Island 20	1991	1	51	7	17	<6	<9	16	<3	96
	Cheam Island	1991	1	23	6	15	<6	<10	73	2	157
	Agassiz Bridge	1991	1	41	15	18	<3	<3	13	10	143
River otter (liver)	Lower Fraser Valley (a)	1995	3p	1	2	57	120	160	0.5	15	26
	Lower Fraser Valley (b)	1995	3p	<0.2	0.6	4.1	39	180	<0.2	1.5	3.0
Mink (liver)	Lower Fraser Valley	1995	2	<1.9	<3.7	<4.0	<7	<13	<1.9	<1.7	

[a] 'p' = one pool of listed number of individual eggs or whole body homogenates
[b] TEQs include all PCDD/F congeners, non-ortho PCBs 77, 126, 169 and mono-ortho PCBs 105, 118 except for Great Blue Heron and Double-crested Cormorants which don't include CB 77. Calculations were completed using WHO-TEFs
[c] ND = not detected; detection limit not specified
[d] Detection limit is contaminant specific therefore listed individually as '< particular value'
[e] NA - not analyzed

Table 4. *Major PCDDs and PCDFs (ng/kg wet wt.) in osprey eggs, nestling tree swallow (whole body homogenates) and mustelid liver tissue collected from sites within the upper Fraser River Basin. Values are geometric means (of 'N' samples) or one analysis of a sample composite.*

SPECIES	SITE	YEAR	N[a]	2378-TCDD	12378-PnCDD	123678-HxCDD	1234678-HpCDD	OCDD	2378-TCDF	23478-PnCDF	TEQS[b]
Osprey (eggs)	Nechako River	1992	5	1.4	3.7	13	54	148	0.5	0.9	13
		1996	5	0.5	3.5	7.0	32	69	0.2	0.3	17
	Quesnel	1991	4	5.5	2.3	5.3	16	33	<1.0[c]	0.7	20
		1992	5	7.3	5.3	12	36	143	0.4	1.5	35
	S. Thompson River	1991	6	11	2.6	14	92	354	1.4	0.9	27
		1992	5	8.0	6.4	16	133	475	0.7	1.6	35
		1997	5	0.6	3.3	8.1	48	198	0.5	0.3	18
	Thompson River	1991	5	46	4.2	21	79	112	2.5	2.4	64
		1992	5	27	4.9	11	38	88	3.7	2.4	49
		1995	1	7.8	4.2	16	101	513	2.8	0.7	22
		1997	5	3.6	1.2	22	26	97	0.9	0.4	16
	Pitt River	1996	3	1.6	4.2	5.0	1.6	3.2	<0.4	0.9	28
Tree Swallow (nestlings)	S. Thompson River	1994	3p	<0.8	<0.8	<1.2	2.2	2.7	1.3	<0.9	5.2
	Thompson River	1994	11p	<0.9	<0.7	1.2	1.9	2.4	3.2	<1	14.9
	upper Fraser River - u/s	1994	18p	<1.1	<0.6	<0.5	1.7	2.1	0.6	<0.2	1.9
	upper Fraser River - d/s	1994	14p	<1.6	1.8	5.7	36	34	1.1	<0.8	11.8
Mink (liver)	upper Fraser River	1991	6p	<0.6	<0.3	2.9	8.0	26	<0.9	0.5	2.2
	Cale Creek 1	1991	1	<1.6	1.8	22	123	187	<0.8	3.1	9.3
	Cale Creek 2	1991	1	<1.9	<1.8	1.5	18	16	<1.8	<0.9	1.6
	Cale Creek 3	1991	1	<0.7	<0.8	<0.4	1.1	3.5	<0.8	<0.2	0.3
	Redrock Lake	1991	1	<0.4	<1	<0.8	<1.3	1.6	<0.4	<0.6	0.9
	Stone Creek Canyon	1991	1	<0.7	<0.9	<0.6	<1.2	1.4	<0.9	<0.4	2.4
River Otter (liver)	Shorty Creek	1991	1	<1	<1	11	16	20	<1.1	<0.4	0.9
	upper Fraser River	1995	6p	<0.2	0.5	0.6	2.3	5.3	<0.2	0.3	1.0

[a] *'p' = one pool of listed number of individual eggs, whole body homogenates or livers*
[b] *TEQs for all species include all PCDD/F congeners, non-ortho PCBs 77, 126, 169 and mono-ortho PCBs 105, 118 except for Great Blue Heron and Double-crested Cormorants which don't include CB 77. Calculations were completed using WHO-TEFs*
[c] *Detection limit is contaminant specific therefore listed individually as '< particular value'*