Water Qual. Res. J. Canada, 2001
Volume 36, No. 2, 233–251

Effects of Endocrine Disrupting Chemicals on Wildlife in Canada: Past, Present and Future

GLEN A. FOX*

Canadian Wildlife Service, National Wildlife Research Centre, Environment Canada, Hull, Quebec K1A 0H3

Within a few years of their widespread introduction into the environment, there were major concerns about the effects of man-made chemicals on wildlife in Canada. Biologists have found evidence of endocrine disruption in at least 16 species of birds (particularly fish-eating and predatory species), 3 species of terrestrial mammals and 1 whale, 1 reptile and several amphibians. These effects were seen in the Great Lakes, areas polluted with pulp and paper effluents, the forests of New Brunswick, a variety of agricultural habitats, acidified lakes in Nova Scotia, major rivers and inshore coastal waters, and the Arctic. They include decreased reproductive success, gross congenital abnormalities and interference with developmental/regenerational processes; thyroid dysfunction; metabolic abnormalities; feminization/demasculinization; alterations in sex ratios; altered brain development and behaviour; and altered immune function. These sublethal effects have the potential to alter an individual's quality of life, its survival, or reproductive fitness. Morbidity and mortality are rarely detected and grossly underestimated in wildlife populations and we are likely to have underestimated the occurrence, prevalence and severity of endocrine disruption in wildlife. There is no formal existing program in Canada for gathering long-term evidence for determining trends in the incidence and severity of effects in organisms. All of the information on endocrine disruption in Canadian wildlife reported in this review was gathered as a result of academic research or research-based monitoring. Much greater effort must be made to systematically detect and identify a much broader range of man-made chemicals, both persistent and nonpersistent, and their metabolites in wildlife tissues and environmental media. These efforts should be complimented by parallel measurements of various biological activities of extracts of the same wildlife tissues and environmental media. This monitoring information would then point to specific locations, effluents, agricultural practices, industrial processes, and products where investigations of effects should be conducted and to species and functions potentially at risk.

Key words: **endocrine disruptors, wildlife, fish-eating birds, snapping turtles, beluga, amphibians, Great Lakes, Canada**

Introduction

When World War II ended, the chemical industry turned to production and distribution of a variety of chemicals for widespread application in industrial and domestic settings, where they were enthusiastically

* Corresponding author; glen.fox@ec.gc.ca

adopted by manufacturing, agriculture, forestry and public health sectors, and by the general public. The 1948 Nobel Prize in chemistry was awarded to Paul Müller for the development of DDT. By the 1950s, researchers were becoming aware of unexpected and undesirable effects of the widespread use of DDT and other man-made chemicals. In 1953, the delegates to the Federal-Provincial Wildlife Conference agreed

> *that this conference recommends that the appropriate federal department take the lead in bringing together manufacturers and users of chemicals and wildlife authorities with the objective of bringing about the greatest benefit from these chemicals with no damage to wildlife resources, or where this is not feasible, with as little damage as possible.* (Hurtig 1961a)

By 1960, the level of concern of the delegates was much greater, and they approved a much stronger and precautionary motion:

> *WHEREAS very discouraging reports are continually being received in respect to great damage being done to most if not all forms of wildlife, including the more or less uncontrolled use of herbicides and insecticides, THEREFORE BE IT RESOLVED that immediate and appropriate action be taken by all game management agencies in Canada and the United States of America to thoroughly investigate and ascertain the* [actual effect] *on human life and wildlife in the use of these insecticides and to take, if warranted, immediate steps to stop, modify, or in some way control the use thereof in order to ensure that the said use will not be* [harmful] *to human life, wildlife or fish.* (Hurtig 1961b)

Clearly, the delegates recognized the inadequate state of knowledge regarding the fate and effects of man-made chemicals released into the environment and were concerned about sublethal and lethal effects. Then came the publication of Rachel Carson's insight-filled classic *Silent Spring* in 1962. In response to provincial and public pressures, the Canadian Wildlife Service (CWS) hired J. Anthony (Tony) Keith in 1965 to initiate their pesticide program, which later became the Wildlife Toxicology Division. Evidence gathered in early CWS studies played a large part in the 1969 decision of the federal cabinet to limit the use of DDT in Canada. In announcing restrictions that would cut use of the pesticide by about 90%, then Prime Minister Pierre Elliott Trudeau made the point that

> *these [DDT] residues are mostly at low levels but have concentrated sufficiently in a few populations of birds and fish to cause reproductive failures and the elimination of a few bird populations over large parts of their range.* (Office of the Prime Minister 1969)

Thus, unlike the United States, evidence from wildlife and concerns for their declining populations played a pivotal role in the regulation of DDT in Canada.

In a 1969 editorial entitled "The DDE Affair", Tony Keith recognized the potential for environmental contaminants to interfere with endocrine functions. He wrote

> *But now it is clear that DDE itself is a thoroughly capable steroid degrader, and there is little doubt that it is by this means it is a cause of the declines of those populations of birds which have substantial DDE residues and abnormally thin eggshells. This gives a most unfortunate new significance to those recent studies documenting the universal contamination of the biosphere with DDE. For here is a consistent component of the diet and bodies of many entire populations of higher vertebrates that increases enzyme activity in the liver* [and can critically interfere with endocrine functions.] (Keith 1969)

However, another 20 years elapsed before Theo Colborn assembled a multidisciplinary group of experts at the Wingspread Conference Center, in Racine, Wisconsin, in July 1991 to discuss "chemically induced alterations in sexual development: the wildlife/human connection". It was there that the term "endocrine disruptor" was coined and discussions begun that would lead to a paradigm shift in toxicology and the way we regulate chemicals. Most of the evidence used in Colborn's meta-analysis that led her to her "endocrine disruption" hypothesis was for wildlife, and much of it was collected by Canadian scientists, many of whom were members of the CWS Wildlife Toxicology Division.

Evidence, Past and Present, of Endocrine Disruption in Canadian Wildlife

What follows is my attempt to present the evidence for endocrine disruption in Canadian wildlife, grouped by region or habitat. For another review, see Colborn and Thayer (2000).

The Great Lakes

In the 1950s to 1970s, complete (bald eagles [*Haliaeetus leucocephalis*] and double-crested cormorants [*Phalacrocorax auritus*]) or partial (common terns [*Sterna hirundo*] and herring gulls [*Larus argentatus*]) reproductive failure was observed in Lake Ontario due to thinning of eggshells, embryotoxicity and/or inadequate parental behaviour (reviewed in Fox 1993, Grasman et al. 1998). Overt effects on reproductive success are no longer evident. The bald eagle was still not attempting to nest on Lake Ontario by the mid-1990s. However, the cormorant population increased from a low of three nests in two colonies in 1973 to over 9,000 nests in 11 colonies in 1991 (Weseloh et al. 1995). On the Canadian shores of the other Great Lakes, productivity of bald eagles did not change between 1982 and 1996 and was sufficient to maintain an increasing population (Donaldson et al. 1999).

A female-biased sex ratio in adult herring gulls and feminization of herring gull embryos were found in Lake Ontario in the 1970s (Fox 1992). Abnormalities in gonadal histology occurr in gull chicks in the late 1990s (Kelly et al. 1999). When de Solla et al. (1998) measured a sexually dimorphic character (ratio of precloacal length to the posterior lobe of the plastron = PPR) and circulating levels of testosterone and estrogen in snapping turtles (*Chelydra serpentina*) at two reference sites and three sites on lakes Ontario and Erie, they found decreases in the PPR (indicating feminization) in male turtles from the Great Lakes sites. At one Lake Ontario site, the magnitude of this response was such that the PPRs of a significant proportion of the males overlapped with females. These differences were not reflected in circulating hormone levels. The PPR in turtles may be equivalent to the anogenital distance in laboratory rodents, a morphological measure that is known to be very sensitive to the effects of *in utero* exposure to estrogens and anti-androgens (Kelce et al. 1995).

Hatching success of eggs in Great Lakes double-crested cormorant colonies in 1986–1988 was highly correlated with their dioxin-like activity but not total PCB concentrations in extracts of their eggs (Tillitt et al. 1992). Developmental abnormalities have been observed in nine species of fish-eating birds in the Great Lakes (reviewed in Fox 1993; Grasman et al. 1998) The prevalence of bill defects in double-crested cormorant showed marked spatial variation in 1979–1987 of 2.7 to 52.1 per 10,000 (Fox et al. 1991). In a follow-up study, for the period 1988–96, spatial variation was not significant, and Georgian Bay was the only region to show a significant decrease in prevalence between the two periods (Ryckman et al. 1998). Developmental deformities have also been observed in hatchling snapping turtles (Bishop et al. 1991, 1998b). The deformity rates in turtle hatchlings in 1986–89 and 1989–91 did not differ significantly.

A survey of contaminant levels in livers of mink (*Mustela vison*) harvested from 15 southern Ontario townships in 1988 and 1999 revealed that PCB concentrations were sufficiently high in some areas to potentially affect the reproductive fitness of the populations (Haffner et al. 1998).

Goitre was observed in adult herring gulls from 1974 to 1991 (Moccia et al. 1986; Fox et al. 1998). Histologically, these goiters are characterized by microfollicular hyperplasia. There was significant spatial variation in thyroxine concentrations in plasma of prefledgling herring gulls and Caspian terns (*Sterna caspia*) in 1992–93 (Grasman et al. 1996). In 1991, adult herring gulls were hypothyroxinemic, and their thyroxine levels were inversely correlated with PCB, but not PCDD, concentrations in their livers (Fox et al., in prep.). Currently, thyroids of pipping herring gull embryos from Great Lakes sites highly contaminated with PCBs have significantly decreased thyroid hormone content on a per mg and per gland basis, relative to the reference site. Prefledglings from these sites have significantly increased thyroid gland mass and decreased hormone content/mg gland. The net effect of these alterations is no difference in hormone content of the thyroid gland. Their plasma thyroid hormone concentrations indicate there is no alteration in whole-body thyroid hormone status (McNabb et al. 2001).

An in-depth investigation of immune function in young fish-eating birds in 1992–2000, employing in vivo and in vitro assays found thymic atrophy in herring gull chicks (Grasman et al. 1998) and a marked decrease in T-cell-mediated immunity in Caspian tern (Grasman et al. 1996; Grasman and Fox 2001) and herring gull chicks (Grasman et al. 1996). The degree of immunosuppression increased as PCB concentrations in eggs or plasma increase. In young Caspian terns, antibody production was increased in an exposure-related manner with plasma PCBs (Grasman and Fox 2001).

The activity of key metabolic enzymes and basal plasma corticosterone levels of herring gull embryos collected from Great Lakes colonies in 1997 were decreased with increasing concentrations of PCDDs and PCDFs in their yolk sacs (Lorenzen et al. 1999). These finding suggest that current levels of contaminants may be affecting the hypothalmo-pituitary-adrenal axis in some fish-eating birds. They are consistent with the wasting syndrome that has been observed in Caspian terns (Ludwig et al. 1993), Forster's terns (*Sterna forsteri* [Harris et al. 1993]) and herring gull chicks (Grasman et al. 1996) at highly contaminated sites in the early 1970s–1998.

We believe that the majority of these effects were caused by persistent and bioaccumulative halogenated hydrocarbons, including DDE, polychlorinated biphenyls (PCBs), hexachlorobenzene and 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD). The sediment record suggests that deposition rates for most of these contaminants peaked between 1960 and 1980 (Durham and Oliver 1983).

Pulp and Paper Mill Effluents

The Strait of Georgia receives wastes from what is probably the greatest concentration of forest products industries in Canada, including nine large kraft pulp and paper mills which discharge their effluent into the strait. Pulp mills that use pentachlorophenol-treated wood chips and high concentrations of chlorine in the bleaching process discharge TCDD and other PCDDs and PCDFs in their effluent (Elliott et al. 1989).

The great blue heron (*Ardea herodias*) is a wading bird that feeds on small inshore fish and is a year-round resident of the Strait of Georgia. Herons which feed in the proximity to pulp mill discharges are exposed to TCDD and other contaminants through the food chain. Since the early 1970s, CWS has been monitoring contaminants in heron eggs collected from several colonies in the Strait. In 1987, it was observed that no chicks fledged from 57 active nests in the colony near Crofton, and that this reproductive failure accompanied an approximate four-fold increase in the concentration of 2,3,7,8-TCDD in their eggs (Elliott et al. 1989). Remedial processes implemented by the Crofton mill between 1988 and 1990 led to a decrease in the amounts of many PCDDs and PCDFs released, and this was reflected in a rapid decline in contaminant levels in heron eggs and an increase in reproductive success of the Crofton colony (Whitehead et al. 1992; Elliott et al., in press). The spatial and temporal variation in PCDDs and PCDFs in heron eggs provided the opportunity for scientists to undertake a variety of investigations that revealed the presence of subcutaneous edema, decreased body and yolk mass and tibia length (Hart et al. 1991; Sanderson et al. 1994) and the degree of intercerebral asymmetry (Henshel et al. 1995) in chicks were significantly correlated with increasing concentrations of TCDD and TCDD-EQ concentrations in sibling eggs. From 1991 to 1995, Elliott studied breeding bald eagle populations near pulp mills in the Strait of Georgia and at a variety of other coastal sites. Productivity was lower at 9 of 16 sites near the Crofton and Powell River mills. However, for the whole data set, there was no significant relationship between productivity and I-TEQs or any TCDD-like compounds in the nestling plasma (Elliott and Norstrom 1998). In 1992, eggs were collected, artificially incubated, and the hatchlings examined for biochemical and histological effects. Yolk sacs of chicks collected near pulp mills showed they were exposed to elevated concentrations of potent embryotoxic PCDD and PCDF congeners, relative to those from the reference nests (Elliott et al. 1996). In 5 of 16 eggs from pulp mill sites, the embryos failed; 4 were from the vicinity of the Powell River mill. All but one of the 8 eggs from non-pulp mill sites hatched. Other than edema in one chick from the Powell River area, and markedly elevated hepatic P4501A protein, EROD and BROD activities in chicks from pulp mill versus reference sites, no other symptoms of TCDD-like exposure were found.

Harris and Elliott (2000) studied nest box populations of tree swallows (*Tachycineta bicolor*) to investigate the reproductive success and uptake of pulp mill-related chlorinated hydrocarbons from emergent aquatic insects at locations upstream and downstream of pulp mills on the Fraser and Thompson rivers in British Columbia in 1994, after mill upgrades to bleaching and wastewater facilities. Nest success was one reproductive measure that showed substantial reductions on both rivers. Failures were largely due to parental abandonment, and while poor parental attentiveness and nest abandonment have been associated with chlorinated hydrocarbon tissue concentrations in other studies, their one-year assessment was insufficient to establish any link with exposure to pulp mill effluent.

Wayland et al. (1998) studied nest box populations of tree swallows at sites upstream and downstream of two bleached kraft pulp mills on the Wapiti River near Grand Prairie, Alberta, and the North Saskatchewan River near Prince Albert, Saskatchewan, from 1993 to 1996. They observed a trend towards improved reproduction and larger nestlings at downstream sites, which may reflect a greater food supply in response to nutrient enrichment below the outfall. They noted that plasma estradiol concentrations were lower in incubating females downstream from the Grand Prairie mill in one of two years. Adult ranch mink chronically fed a diet based on fish, captured immediately below the Prince Albert mill in 1993, and effluent from the mill for 26 weeks showed no detrimental effects on a broad set of pathological, biochemical, hematological, behavioral or reproductive variables (Smits et al. 1995). However, they had a significantly suppressed delayed type hypersensitivity immune response (Smits et al. 1996).

The Forests of Eastern Canada

The forest industry has aerially applied vast amounts of pesticides in eastern Canada to control epidemics of defoliating lepidopteran insects. In excess of 17 million kg of chemical pesticides were applied to New

Brunswick forests alone between 1952 and 1992 (calculated from Freedman 1989). The insecticides applied were DDT, phosphamidon, fenitrothion, aminocarb, and trichlorphon. Between 1952 and 1965, about 5.75 million kg of DDT were applied to New Brunswick's forests (Pearce 1975). Between 1958 and 1963, there was a highly significant inverse correlation between the breeding success of woodcock (*Philohela minor*) and the amount of DDT applied (Wright 1965). The woodcock is a forest floor-dwelling bird that feeds on earthworms and other soil invertebrates. From 1954 to 1958, the ratio of young-of-the-year to adults in the fall hunter-killed woodcock in sprayed areas was 39% of that in unsprayed areas, and between 1959 and 1963 the ratio was depressed by at least 50% (Wright 1960, 1965). In two years where the composition of the fall hunter-kill in New Brunswick was compared to that in Nova Scotia, where no spraying had occurred, the breeding success of New Brunswick woodcock was 48% lower (Wright 1965). The impact on reproduction was not due to eggshell thinning (Dilworth et al. 1972) and must reflect embryo or juvenile mortality.

Fenitrothion was the principal insecticide applied for budworm control in the forests of eastern North America through the 1970s and 1980s. Between 1975 and 1986, an average of 1.76 million hectares of New Brunswick forest were sprayed annually (calculated from Freedman 1989). In an intensive study of a marked population of white-crowned sparrows (*Zonotrichia albicollis*), Busby et al. (1990) documented extensive adult mortality, territorial abandonment, inability to defend a territory, disruption of incubation and clutch desertion, resulting in a 75% reduction in productivity following an experimental fentrothion spray application which produced levels of cholinesterase inhibition similar to those measured after operational sprays. This may be another example of the higher level endocrine effects of organophosphates reported by Rattner et al. (1982).

Terrestrial Agricultural Landscapes

In the grain growing areas of the Prairie Provinces, large amounts of a wide variety of chemical biocides have been applied to control insects (especially grasshoppers), to inhibit the growth of fungi on seeds and seedlings, and to kill weeds (Fyfe et al. 1976a). In 1966, CWS initiated a long-term study of two predatory species of grassland birds, the prairie falcon (*Falco mexicanus*) and the merlin (*Falco columbarius*) in southwestern Saskatchewan and southeastern Alberta. At the time, numbers of both species were declining and reproduction of the more contaminated members of both populations was significantly decreased. Both species carried significant residues of organochlorine pesticides, PCBs and mercury accumulated from local food webs. Data collected between 1966 and 1973 showed that as eggshells thinned and DDE levels in eggs increased, fewer young were produced by both species (Fyfe et al. 1976a). In the case of the merlin, more contaminated individuals defended their nests less aggressively. It is noteworthy that the last breeding record of the *anatum* race of the closely related peregrine falcon (*Falco peregrinus*) in North America south of the boreal forest and east of the Rocky Mountains occurred in southern Alberta in 1972 (Fyfe et al. 1976b).

Fruit production is a large agricultural industry in five Canadian provinces, and almost all orchards are frequently and intensively sprayed with insecticides (synthetic pyrethroids, organophosphates and carbamates) and fungicides (dithiocarbmates and myclobutamils), acaricides, herbicides and growth regulators, applied singly and as mixtures (Bishop et al. 1998a). In orchards, birds are exposed to in-use chemicals as well as residues of historically used chemicals (organochlorine pesticides and lead arsenate). Bishop et al. (2000) conducted an intensive study of the reproduction of eastern bluebirds (*Sialia sialis*) and tree swallows nesting in nonsprayed (but historically contaminated with organochlorines) and actively sprayed apple orchards in southern Ontario, 1988–1994. Bluebird eggs contained much higher levels of organochlorines (>90% DDE) than tree swallows, and there was a significant increase in unhatched bluebird eggs as organochlorine levels (mainly DDE) increased. Both species experienced significant declines in reproduction (particularly in egg survival) with increased exposure and toxicity from the suite of pesticides used during the 1990s, but the reductions in reproductive rates did not exceed 14% in any year. In an intensive study of reproduction, hormone levels, testicular histology and immune function of tree swallow chicks in pesticide-sprayed apple orchards in Ontario, 1994–95, Bishop et al. (1998a) found immunomodulatory effects (stimulation, delayed thymic involution). There was no effects on sex steroid hormones, but there was a significant positive correlation between triiodothyronine (T3) levels and the number of mixtures of sprays applied during incubation through chick-rearing (Bishop et al. 1998b). There was also a marginally significant increase in the occurrence of a disrupted Sertoli cell population on the seminiferous tubule basement membranes as the number of mixtures of pesticides sprayed during chick rearing increased (Bishop et al. 1998c).

Long-term and extremely high past application rates, seasonally freezing temperatures and infrequent cultivation make old orchard habitat in northern locations some of the most DDT-contaminated environments in North America. American robin (*Turdus migratorius*) eggs collected in orchards in southern Ontario and the Okanagan Valley of British Columbia bioaccumulated DDE in the local soil-earthworm-robin food chain to concentrations equal to or exceeding those found in raptor eggs in the early 1970s! In the Okanagan, high average DDE (soil: 5.2 mg/kg; earthworms: 52 mg/kg; robin egg: 484 mg/kg dry weight) concentrations confirmed that previously recorded contamination was common in the region (Harris et al. 2000). To determine whether robins suffer reproductive effects from such high *in ovo* exposures to DDE (a proven antiandrogen), chicks from nests in the Okanagan orchards and a reference site were raised in captivity, bred and their reproductive behavior and performance studied. Preliminary evidence suggests that there are transgenerational effects on behavior and that there may be a threshold concentration at which such effects occur (Smith et al. 2000).

Wetlands in Agricultural Landscapes

Frogs and toads are universally dependent on temporary or permanent wetlands for breeding and early life stage development. With the decline in natural wetlands, those in agricultural landscapes are becoming increasingly important to amphibians and other wildlife. However, the toxic pesticides that are applied to crops and anthropogenic nitrogen pollution enter wetlands in agricultural landscapes through runoff or percolation and have the potential to adversely affect wildlife. Rouse et al. (1999) concluded that nitrate in many agricultural ecosystems exceeds concentrations that are toxic to amphibians. Preliminary evidence suggests that agricultural runoff reduces survival and hatching success of red-legged frogs (*Rana aurora*) in the lower Fraser Valley of British Columbia (de Solla et al., submitted). In the St. Lawrence River valley, in 1992 and 1993, a markedly higher incidence of hind limb deformities was observed in metamorphosing frogs caught in agricultural sites exposed to pesticide runoff relative to those in ponds in more pristine areas (Ouellet et al. 1997). Harris et al. (1998a) concluded that wetlands in apple orchards in southern Ontario can provide viable breeding habitat for two frog species, based on information obtained on wild adults and young-of-the-year over the short term. However, they caution that the changes in rates of survival, deformities and growth of caged tadpoles exposed to the pesticide formulations that were applied suggested that the recorded pesticide exposure at orchard sites could contribute to a suite of environmental stressors acting on populations to reduce developmental success (Harris et al. 1999b).

Major Rivers and Inshore Coastal Waters

The northern gannet (*Sula bassanus*) colony on Bonaventure Island in the Gulf of St. Lawrence accounts for more than 50% of the species' population in North America. After steadily increasing from the turn of the century until 1966, the population dropped from 21,000 pairs in 1969 to 16,400 pairs in 1976 and then rose to 21,100 pairs in 1984. Between 1966 and 1974 hatching success varied from 36–58%, but increased to 78–89% from 1976 to 1984 (Chapdelaine et al. 1988). Unhatched eggs in 1969 had no outer calcified layer, a mean thickness 20% less than the pre-1947 mean, and a DDE concentration of 31 mg/kg (Elliott et al. 1988). There was a highly significant negative relationship between log DDE and eggshell thickness. The Gulf of St. Lawrence receives water from the major river drainages in the four Maritime Provinces. Over 6.6 million kg of DDT were used in New Brunswick and Quebec from 1952 to 1969 to control forest insect pests (Nigam 1975). Extensive use of DDT to control forest insect pests around the Gulf of St. Lawrence ceased in 1969, leading to reductions in concentrations of its metabolites in the Gulf of St. Lawrence ecosystem. As DDE levels declined, eggshell thickness increased, and hatching success improved, allowing the gannet colony to increase to its size.

In the 1980s and 1990s, abnormalities in reproduction, and immune and adrenal function and a high prevalence of tumors have been observed in an isolated population of beluga whales (*Delphinapterus leucas*) in the St. Lawrence estuary (Béland et al. 1993; De Guise et al. 1994). The tissues of these animals contain high concentrations of a wide variety of persistent bioaccumulative toxic substances, including PAHs.

Bishop and Gendron (1998) reported that the frequency of skeletal deformities such as oligodactyly and polydactyly in the mudpuppy (*Necturus maculosus*) significantly increased with higher exposure to chlorinated hydrocarbons in the St. Lawrence and Ottawa rivers. This suggests that these pollutant concentrations in the mid-1990s may be sufficient to interfere with developmental or regenerational processes in this aquatic salamander. A suppressed corticosterone response to ATCH challenge was also found in this species at more contaminated sites (Gendron et al. 1997).

Harding et al. (1999) assessed the concentrations and effects of chlorinated hydrocarbon contaminants in wild mink and river otter (*Lutra canadensis*) on the Columbia and Fraser River systems in British Columbia, 1994–1996. Contaminant levels were low relative to other North American populations. However, reproductive tract abnormalities were observed in 5 of 38 males collected. There was a significant correlation between total PCB concentrations and baculum length in juvenile mink. Henny et al. (1996) found significant correlations between PCBs and other organochlorine contaminants and the mass of the testes and baculum in river otters collected along the Columbia River in Oregon. A recent survey of chlorinated hydrocarbon concentrations in otter scats from the coast of British Columbia revealed the presence of significant quantities of PCDDs, PCDFs and PCBs (Elliott 2000). Concentrations of PCBs in scats from some locations near Victoria exceeded the adverse effect level for reproduction.

Blubber biopsy samples were used to assess contaminant burden in two discrete populations of killer whales (*Orcinus orca*) which frequent the coastal waters of British Columbia (Ross et al. 2000). They revealed that the resident and transient populations off British Columbia are among the most contaminated cetaceans in the world, and that the majority of killer whales in this region are at risk to PCB-induced toxic effects, particularly on the immune and endocrine systems.

Acidified Lakes in Atlantic Canada

Reproductive success of common loons (*Gavia immer*) nesting on acidified lakes in Nova Scotia is decreased markedly as mercury concentrations in their tissues and forage fish increases (Burgess et al. 1998; Evers et al. 1998) due to alterations in the biogeochemical cycle of mercury. Reproductive failure occurs early in the cycle (predominantly before laying), suggesting disruption of the hypothalamic-pituitary-gonadal axis. These findings confirm the results of a study of mercury-contaminated loons on the Wabigoon-English River system in northwestern Ontario in the 1970s (Barr 1986). In that case, the source of the waterborne mercury contamination was a chlor-alkali plant.

Arctic

There is considerable evidence that persistent bioaccumulative pollutants, including numerous organochlorines, cadmium and mercury, have increased in recent times in northern environments as a result of long-range transport and deposition from temperate zones. According to Norstrom (2000), the polar bear (*Ursus maritimus*) is among the most highly organochlorine-contaminated of arctic mammals due to its almost exclusive diet of ringed seals (*Phoca hispida*). A collaborative study comparing the effects of persistent organic pollutants on polar bears from Svalbard and western Hudson Bay suggests there are adverse effects on plasma retinol concentrations, thyroid hormone transport and immune function (Norstrom 2000). Data from Svalbard bears suggest that plasma concentrations of total IgG are significantly decreased with increased PCB levels (Bernhoft et al. 2000). The statistical analysis for the Canadian bears, the assessment of the responses of individual bears to immunization, characterization of hydroxy-PCB metabolites, and rates and patterns of testosterone metabolism are nearing completion.

A study of chicks of the Black Guillemot (*Cepphus grylle*), a fish-eating seabird nesting in a PCB-contaminated fjord on the Labrador coast, revealed no alterations of thyroid function, but severe exposure-related suppression of T-cell mediated immune response (Grasman et al. 2000), and an increased incidence of gonadal abnormalities (Kelly et al. 2000).

There is growing evidence that seaduck population declines are real, but the causes of these declines are unclear. Pollution, particularly with heavy metals, is one hypothesized stress factor. Seaducks are exposed to significant concentrations of cadmium and mercury through their aquatic diet. Preliminary results suggest that elevated cadmium concentrations in the tissues of common eiders (*Somateria mollissima*) nesting on Southampton Island may decrease the stress response (Wayland et al. 1999) and cell-mediated immunity (Wayland et al. 2000) of incubating females.

Conclusions

Clearly, wildlife are effective sentinels of the real world endocrine disrupting effects of man-made chemicals. Man-made chemicals play no part in the normal biochemistry of living organisms — they are new forces in population ecology. Many are biocides by design. Like natural enemies and exploitation, some man-made chemicals are capable of killing individuals or altering reproductive performance. Like parasites, the impacts of man-made chemicals are influenced by factors which increase exposure or decrease resistance. Like adverse weather and parasites, man-made chemicals can have sublethal effects on individuals which alter their survival and reproductive performance. However, laboratory and field studies have shown that these man-made toxicants are unique in the variety of their modes of action and the diversity of their sublethal effects — effects which are frequently unexpected and often delayed. The capability of man-made chemicals to drastically affect populations is best illustrated by the dramatic simultaneous decline of the peregrine falcon on two continents. This species' remarkable resistance to human persecution and exploitation has been well documented. Yet, as the pioneer raptor ecologist Tom Cade put it:

> *Down through the centuries, not all the falcon trappers, egg collectors, war messengers concerned for their messenger pigeons, or misguided gunmen have been able to affect a significant reduction in the numbers of falcons. But the simple laboratory trick of adding a few chlorine molecules to a hydrocarbon and the massive application of this unnatural class of chemicals to the environment can do what none of these grosser, seemingly more harmful agents could do.* (Cade 1968)

What is more, DDT brought about the decline or extripation of disparate populations of peregrines within 10 to 20 years of its widespread use.

The forms of endocrine disruption that have been observed in Canadian wildlife include decreased reproductive success, gross congenital abnormalities and interference with developmental/regenerational processes; thyroid dysfunction; metabolic abnormalities; feminization/demasculinization; alterations in sex ratios; altered brain development and behavior; and altered immune function. These are all sublethal effects which could alter an individual's quality of life, its survival or reproductive fitness. Several of these effects are known to contribute to morbidity of human populations in the Great Lakes basin. In the St. Lawrence beluga, their cumulative effect may be sufficient to prevent population recovery. Elsewhere in the Great Lakes, they have been reported to affect populations of bald eagles (Bowerman et al. 1995) and Caspian terns (Mora et al. 1993). To fully understand the potential or actual impact of such effects, one must thoroughly understand the life history and demography of the species in question. Such an understanding is lacking for all the wildlife species in which such effects were detected in Canada. However, long-term studies of sparrowhawks (*Accipiter nisus*) in Britain have provided sufficient understanding for the development of realistic population models. These models clearly show that sublethal effects were important during population declines due to exposure to highly toxic cyclodiene insecticides (Sibly et al. 2000). Sublethal toxic effects must be important; they affect an individual's quality of life, metabolism and energetics, reproductive fitness, suceptibility to disease and predators, and genotype. Sick, reproductively incapable, or otherwise dysfunctional individuals in wildlife populations do not visit doctors or hospitals for treatment of their complaint. If, or when such individuals die, there is no record. Hence, morbidity and mortality are rarely detected and grossly underestimated in wildlife populations. Therefore we greatly underestimate the occurrence, prevalence and severity of endocrine disruption in any wildlife population we study; the individuals we see or study are the survivors. I believe that we must look to the impacts of endocrine disruption on human populations, where accurate records of morbidity and mortality are kept, to grasp their potential significance to wildlife.

Within a few years of their widespread introduction into the environment, there were major concerns about the effects of man-made chemicals on wildlife in Canada. These concerns have been addressed through regulatory actions, the creation of a special division within the Canadian Wildlife Service with federal responsibilities for toxicological issues and, to some extent, by the research the division and university scientists have conducted. The reactions of the chemical industry, agriculturists and, to some extent, the regulatory community to the ecological findings of the 1950s and '60s to Rachel Carson's dire warnings in *Silent Spring* and to Theo Colborn's endocrine disruption hypothesis have progressed from "we do not believe it" to "it is of no importance anyway" and finally to "we knew it all the time". In his 1970 editorial in a special issue of the *Canadian Field-Naturalist* devoted to the peregrine falcon, Joseph Hickey wondered whether there would be a peregrine falcon surviving on the North American continent when the agricultural chemical industry finally reported, "we knew it all along" (Hickey 1972). Today, Joe must be pleased that, because of his pioneer efforts and many other scientists, and the regulatory community, there are recovering populations of peregrine falcons in North America. What is more, I believe that we have turned a corner, and the chemical industry and agriculturalists, etc., have moved on to a fourth stage, that of "how are we going to predict and prevent these nontarget sublethal injuries?". The endocrine disruption issue in particular has brought us all to the same table.

In this context, it is noteworthy that all of the information on endocrine disruption in Canadian wildlife reported in this review was gathered as a result of academic research or research-based monitoring. It was short term, localized and not systematic, and as such is incapable of providing an accurate or comprehensive picture. While there are a few long-term monitoring programs to document gross trends in concentrations of pollutants in the environment, *there is no formal existing program in Canada for gathering long-term evidence for determining trends in the incidence and severity of effects in organisms.* In his 1969 editorial "The DDE Affair", Tony Keith recognized the inadequacy of such research in providing the information required for resource management and chemical regulation. He said:

> *The central significance [of the DDE affair] for environmental pollution research is that it has hopelessly failed. For these decisive studies showing critical DDE effects in Europe and North America come after 20 years of large-scale DDT use and small-scale side-effect research, and at the point where, in these countries, DDT use is declining anyway and ironically for different reasons, principally the development of resistant strains of insects. If environmental pollution research cannot be supported to a much greater and more coherent extent in the future, and so achieve results in time spans that have any relevance to the problems, then this research is merely a sop to the conscience of the scientific community, a pretense with no real function in resource management.* (Keith 1969)

Things have not changed! Yes, we now have wildlife toxicologists working in all regions of Canada, well-equipped and competent analytical and biochemical laboratories, and a national federal toxics program, but they are grossly under-resourced to detect and assess biological effects and new chemicals of concern. From 1985–1997, World Wildlife Fund Canada, in partnership with Environment Canada and industry, maintained the Wildlife Toxicology Fund to support environmental toxicology research projects initiated by scientists at Canadian universities. This funding, and more recently the Toxic Substances Research Initiative, have made a significant contribution to the funding of the research results reported here. *However, their focus is research.* Monitoring is long term and systematic, and transcends the short-term thinking of politicians. It is what is needed, however, if we want to prevent another DDE affair. Since analytical methods are specific to a particular chemical or groups of structurally related chemicals, we only measure what we are looking for. Much greater effort must be made to detect and identify a much broader range of man-made chemicals (including nonpersistent chemicals) and their metabolites in wildlife tissues and environmental media. These efforts should be complimented by parallel measurements of various biological activities of extracts of the same wildlife tissues and environmental media. This monitoring information would then point us to specific locations, effluents, agricultural practices, industrial processes and products where investigations of effects should be conducted and to species and functions potentially at risk. However, such "levels" and "effects" monitoring require a recognition of their value and the appropriate fiscal responsibility, plus the political and public will to make them happen. Fifty years have passed since wildlife agencies first raised concerns, but environmental monitoring in Canada today is still not supported sufficiently or coherently, and our broad-brush efforts often seem no more than a sop to the conscience of the scientific community and the public. Must we wait for a silent spring?

References

Barr JF. 1986. Population dynamics of the Common Loon (*Gavia immer*) associated with mercury-contaminated waters in northwestern Ontario. Occassional Paper No. **36**, Canadian Wildlife Service. Supply and Services Canada, Ottawa.

Béland P, De Guise S, Girard C, Lagacé A, Martineau D, Michaud R, Muir DCG, Norstrom RJ, Pelletier E, Ray S, Shugart LR. 1993. Toxic compounds and health and reproductive effects in St. Lawrence beluga whales. J. Great Lakes Res. **19:**766–775.

Bernhoft A, Skaare JU, Wiig O, Derocher AJ, Larsen HJS. 2000. Possible immunotoxic effects of organochlorines in polar bears (*Ursus maritimus*) at Svalbard. J. Toxicol. Environ. Health **59:**561–574.

Bishop CA, Boermans HJ, Ng P, Campbell GD, Struger S. 1998a. Health of tree swallows (*Tachycineta bicolor*) nesting in pesticide-sprayed apple orchards in Ontario, Canada. I. Immunological parameters. J. Toxicol. Environ. Health, **55A:**531–559.

Bishop CA, Collins B, Mineau P, Burgess NM, Read WF, Risely C. 2000. Reproduction of cavity-nesting birds in pesticide-sprayed apple orchards in southern Ontario, Canada, 1988–1994. Environ. Toxicol. Chem. **19:**588–599.

Bishop, CA, Gendron AD. 1998. Reptiles and amphibians: shy and sensitive vertebrates of the Great Lakes basin and St. Lawrence River. Environ. Monit. Assess. **53:**225–244.

Bishop CA, Ng P, Brooks RJ, Kennedy S, Stegeman JJ, Norstrom RJ. 1998b. Environmental contamination and developmental abnormalities in eggs and hatchlings of the common snapping turtle (*Chelydra serpentina serpentina*) from the Great Lakes-St. Lawrence River basin (1989–1991). Environ. Pollut. **99**: 1–14.

Bishop CA, van der Kraak GJ, Ng P, Smits JEG, Hontela A. 1998c. Health of tree swallows (*Tachycineta bicolor*) nesting in pesticide-sprayed apple orchards in Ontario, Canada. II. Sex and thyroid hormone concentrations and testis development. J. Toxicol. Environ. Health **55A:**561–581.

Bowerman WW, Giesy JP, Best DA, Kramer VJ. 1995. A review of factors affecting productivity of bald eagles in the Great Lakes region: implications for recovery. Environ. Health Perspect. **103** (Suppl. 4)**:**51–59.

Burgess NM, Evers DC, Kaplan JD, Duggan M, Kerekes JJ. 1998. Mercury and reproductive success of common loons breeding in the Maritimes, p 104–109. *In* Burgess N, Beauchamp S, Brun G, Clair T, Roberts C, Rutherford L. Tordon R, Vaidya O (ed.), Mercury in Atlantic Canada: a progress report. Environment Canada, Sackville, NB.

Busby DG, White LM, Pearce PA. 1990. Effects of aerial spraying of fenitrothion on breeding white-crowned sparrows. J. Appl. Ecol. **27:**743–755.

Cade TJ. 1968. The gryfalcon and falconry. Living Bird 7**:**237–240.

Chapdelaine G, Laport P, Nettleship DN. 1988. Population, productivity and DDT contamination trends of northern gannets (*Sula bassanus*) at Bonaventure Island, Quebec, 1976–1984. Can. J. Zool. **65:**2922–2926.

Colborn T, Thayer K. 2000. Aquatic ecosystems: harbingers of endocrine disruption. Ecol. Applic. **10:**949–957.

De Guise S, Lagacé A, Béland P. 1994. True hermaphroditism in a St. Lawrence beluga whale (*Delphinapterus leucas*). J. Wildlife Diseases **30:**287–290.

De Solla SR, Bishop CA, van der Kraak G, Brooks RJ. 1998. Impact of organochlorine contamination on levels of sex hormones and external morphology of common snapping turtles (*Chelydra serpentina serpentina*) in Ontario, Canada. Environ. Health Perspect. **106:**253–260.

De Solla SR, Pettit KE, Bishop CA, Cheng KM, Elliott JE. Effects of agricultural runoff on hatching success, deformities and survivorship of native amphibians in the Lower Fraser Valley, British Columbia, Canada (submitted to Environ. Toxicol. Chem.).

Dilworth TG, Keith JA, Pearce PA, Reynolds LM. 1972. DDE and eggshell thickness in New Brunswick woodcock. J. Wildlife Manage. **36:**1186–1193.

Donaldson GM, Shutt JL, Hunter P. 1999. Organochlorine contamination in bald eagle eggs and nestlings from the Canadian Great Lakes. Arch. Environ. Contam. Toxicol. **36:**70–80.

Durham RW, Oliver BG. 1983. History of Lake Ontario contamination from the Niagara River by radiodating and chlorinated hydrocarbon analysis. J. Great Lakes Res. **9:**160–168.

Elliott JE. 2000. Chlorinated hydrocarbons in otter (*Lutra canadensis*) scats from the coast of British Columbia. Abstract, SETAC-2000, Nashville, TN, 12–16 November, 2000.

Elliott JE, Butler RW, Norstrom RJ, Whitehead PE. 1989. Environmental contaminants and reproductive success of great blue herons *Ardea herodias* in British Columbia, 1986–87. Environ. Pollut. **59:**91–114.

Elliott JE, Harris ML, Wilson LK, Whitehead PE, Norstrom RJ. Monitoring temporal and spatial trends in polychlorinated dibenzo-*p*-dioxins (PCDDs) and dibenzofurans (PCDFs) in eggs of great blue heron (*Ardea herodias*) on the coast of British Columbia, Canada, 1983–1998. Ambio, in press.

Elliott JE, Norstrom RJ. 1998. Chlorinated hydrocarbon contaminants and productivity of bald eagle populations on the pacific coast of Canada. Environ. Toxicol. Chem. **17:**1142–1153.

Elliott JE, Norstrom RJ, Keith JA. 1988. Organochlorines and eggshell thinning in northern gannets (*Sula bassanus*) from eastern Canada, 1968–1984. Environ. Pollut. **52:**81–102.

Elliott JE, Norstrom RJ, Lorenzen A, Hart LE, Philibert H, Kennedy SW, Stegeman JJ, Bellward GD, Cheng KM. 1996. Biological effects of polychlorinated dibenzo-p-dioxins, dibenzofurans and biphenyls in bald eagle (*Haliaeetus leucocephalus*) chicks. Environ. Toxicol. Chem. **15:**782–793.

Evers DC, Kaplan JD, Meyers MW, Reaman PS, Braselton WE, Major A, Burgess N, Scheuhammer AM. 1998. Geographic trend in mercury measured in common loon feathers and blood. Environ. Toxicol. Chem. **17:**173–183.

Fox GA. 1993. What have biomarkers told us about the effects of contaminants on the health of fish-eating birds in the Great Lakes? The theory and a literature review. J. Great Lakes Res. **19:**722–736.

Fox GA. 1992. Epidemiological and pathobiological evidence of contaminant-induced alterations in sexual development in free-living wildlife, p 147–158. *In* Colborn T, Clement C. (ed.), Chemically-induced alterations in sexual and functional development; the wildlife/human connection. Princeton Science Publ. Co., Inc, Princeton, NJ.

Fox GA, Collins B. Hayakawa, Weseloh DV, Ludwig JP, Kubiak TJ, Erdman TC. 1991. Reproductive outcomes in colonial fish-eating birds: a biomarker for developmental toxicants in Great Lakes food chains. II. Spatial variation in the occurrence and prevalence of bill defects in young double-crested cormorants in the Great Lakes, 1979–1987. J. Great Lakes Res. **17:**158–167.

Fox GA, Trudeau S, Won H, Grasman KA. 1998. Monitoring the elimination of persistent toxic substances from the Great Lakes; chemical and physiological evidence from adult herring gulls. Environ. Monit. Assess. **53:**147–168.

Freedman B. 1989. Environmental ecology: the impacts of pollution and other stresses on ecosystem structure and function. Academic Press, Inc., San Diego, CA.

Fyfe RW, Risebrough RW, Walker W. 1976a. Pollutant effects on the reproduction of the prairie falcons and merlins of the Canadian prairies. Can. Field-Nat. **90:**346–355.

Fyfe RW, Temple SA, Cade TJ. 1976b. The 1975 North American peregrine falcon survey. Can. Field-Nat. **90:**228–273.

Gendron AD, Bishop CA, Fortin R, Hontela A. 1997. *In vivo* testing of the functional integrity of the corticosterone-producing axis in mudpuppy (Amphibia) exposed to chlorinated hydrocarbons in the St. Lawrence and Ottawa Rivers. Environ. Toxicol. Chem. **16:**1694–1706.

Grasman KA, Fox GA. 2001. Associations between altered immune function and organochlorine contamination in young Caspian terns (*Sterna caspia*) from Lake Huron, 1997–1999. Ecotoxicology **10:**101–114.

Grasman KA, Fox GA, Burgess NM, Kuzic ZA, Jeffrey D. 2000. Effects of polychlorinated biphenyls on the immune systems of black guillemots from Labrador. Abstract, SETAC-2000, Nashville, TN.

Grasman KA, Fox GA, Scanlon PF, Ludwig JP. 1996. Organochlorine-associated immunosupression in prefledgling Caspian terns and herring gulls from the Great Lakes: an ecoepidemiological study. Environ. Health Perspect. **104** (Suppl. 4)**:**829–842.

Grasman KA, Scanlon PF, Fox GA. 1998. Reproductive and physiological effects of environmental contaminants in fish-eating birds of the Great Lakes: a review of historical trends. Environ. Monit. Assess. **53:**117–145.

Haffner GD, Glooschenko V, Straughan CA, Hebert CE, Lazar R. 1998. Concentrations and distributions of polychlorinated biphenyls, including non-ortho congeners, in mink populations from southern Ontario. J. Great Lakes Res. **24:**880–888.

Harding L, Harris ML, Stephen CR, Elliott JE. 1999. Reproductive and morphological condition of wild mink (*Mustela vison*) and river otter (*Lutra canadensis*) in relation to chlorinated hydrocarbon contamination. Environ. Health Perspect. **107:**141–147.

Harris HJ, Erdman TC, Ankley GT, Lodge KB. 1993. Measures of reproductive success and polychlorinated biphenyl residues in eggs and chicks of Forster's terns on Green Bay, Lake Michigan, Wisconsin — 1988. Arch. Environ. Contam. Toxicol. **25:**304–314.

Harris ML, Bishop CA, Struger J, van den Heuvel MR, van der Kraak GJ, Dixon DG, Ribley B, Bogart JP. 1999a. The functional integrity of northern leopard frog (*Rana pipiens*) and green frog (*Rana clamitans*) populations in orchard wetlands. I. Genetics, physiology, and biochemistry of breeding adults and young-of-the-year. Environ. Toxicol. Chem. **17:**1338–1350.

Harris ML, Bishop CA, Struger J, Ripley B, Bogart JP. 1999b. The functional integrity of northern leopard frog (*Rana pipiens*) and green frog (*Rana clamitans*) populations in orchard wetlands. II. Effects of pesticides and eutrophic conditions on early life stage developments. Environ. Toxicol. Chem. **17:**1351–1363.

Harris ML, Elliott JE. 2000. Reproductive success and chlorinated hydrocarbon contamination in tree swallows (*Tachycineta bicolor*) nesting along rivers receiving pulp and paper mill effluent discharges. Environ. Pollut. **110:**307–320.

Harris ML, Wilson LK, Elliott JE, Bishop CA, Tomlin AD, Henning KV. 2000. Transfer of DDT and metabolites from fruit orchard soils to American robins (*Turdus migratorius*) twenty years after agricultural use of DDT in Canada. Arch. Environ. Contam. Toxicol. **39:**205–220.

Hart LE, Cheng KM, Whitehead PE,Shah RM, Lewis RJ, Ruschkowski SR, Blair RW, Bennett DW, Bellward GD, Bandiera SM. 1991. Effects of dioxin contamination on the growth and development of great blue heron embryos. J. Toxicol. Environ. Health **32:**331–344.

Henny CJ, Grove RA, Hedstrom OR. 1996. A field evaluation of mink and river otter on the lower Columbia River and the influence of environmental contaminants. Final Report to the Lower Columbia River Bi-State Water Quality Program (Portland, OR). National Biological Service, Forest and Rangeland Ecosystem Science Center, Corvallis, OR.

Henshel DS, Martin JW, Norstrom R, Whitehead P, Steeves JD, Cheng KM. 1995. Morphometric abnormalities in brains of great blue heron hatchlings exposed in the wild to PCDDs. Environ. Health Perspect. **103** (Suppl 4)**:**61–66.

Hickey JJ. 1970. Peregrine falcons, pollutants, and propaganda. Can. Field-Nat. **84:**207–208.

Hurtig H. 1961a. Personal responsibility in association with pesticide use. *In* 1961 Wildlife Management Papers, Canadian Wildlife Service, Ottawa, p. 40.

Hurtig H. 1961b. Personal responsibility in association with pesticide use. *In* 1961 Wildlife Management Papers, Canadian Wildlife Service, Ottawa, p. 40–41.

Keith, JA. 1969. The DDE affair. Canadian Field-Nat. **83:**89–90.

Kelce WR, Stone CR, Laws SC, Gray LE, Kemppainnen JA, Wilson EM. 1995. Persistent DDT metabolite p,p′-DDE is a potent androgen receptor agonist. Nature **375:**581–585.

Kelly ME, Grasman KA, Burgess NM, Kuzyk ZA. 2000. Altered reproductive development in PCB-exposed black guillemot chicks. Abstract, SETAC-2000, Nashville TN.

Kelly ME, Grasman KA, Fox GA. 1999. Gonadal histology of Great Lakes herring gulls. Abstract, SETAC-1999, Philadelphia, PA.

Lorenzen A, Moon TW, Kennedy SW, Fox GA. 1999. Relationships between environmental organochlorine contaminant residues, plasma corticosterone concentrations, and intermediary metabolic enzyme activities in Great Lakes herring gull embryos. Environ. Health Perspect. **107:**179–186.

Ludwig JP, Auman HJ, Kurita H, Ludwig ME, Campbell LM, Giesy JP, Tillitt DE, Jones P, Yamashita N, Tanabe S, Tatsukawa R. 1993. Caspian tern reproduction in the Saginaw Bay ecosystem following a 100-year flood event. J. Great Lakes Res. **19:**96–108.

McNabb FMA, Fowler LA, Parsons CM, Grasman KA, Fox GA. 2001. Thyroid function in herring gulls from PCB-contaminated Great Lakes sites. Abstract, Society for Integrative and Comparative Biology, Chicago, IL, Jan. 2001.

Moccia RD, Fox GA, Britton A. 1986. A quantitative assessment of thyroid histopathology of herring gulls (*Larus argentatus*) from the Great Lakes and a hypothesis on the causal role of environmental contaminants. J. Wildlife Diseases **22:**60–70.

Mora MA, Auman HJ, Ludwig JP, Giesy JP, Verbrugge DA, Ludwig ME. 1993. Polychlorinated biphenyls and chlorinated insecticides in plasma of Caspian terns: relationships with age, productivity, and colony site tenacity in the Great Lakes. Arch. Environ. Contam. Toxicol. **24:**320–331.

Nigam PC. 1975. Chemical insecticides. *In* Prebble ML. (ed.), Aerial control of forest insects in Canada. Department of the Environment, Ottawa, Cat. No. F023/19/1975.

Norstrom RJ. 2000. Effects of persistent organic pollutants on polar bears, p 208–221. *In* Kalhok S. (ed.) Synopsis of research conducted under the 1999–2000 Northern Contaminants Program, Department of Indian Affairs and Northern Development, Ottawa.

Office of the Prime Minister. 1969. Statement by the Prime Minister on DDT and other organochlorine pesticides. Press release, House of Commons, Ottawa, Nov 3, 1969.

Ouellet M, Bonin J, Rodrigue J, DesGranges J-L, Lair S. 1997. Hind limb deformities (ectromelia, ectrodactyly) in free-living anurans from agricultural habitats. J. Wildlife Diseases **33:**95–104.

Pearce P. 1975. Effects on birds, p 306–313. *In* Prebble ML (ed.), Aerial control of forest insects in Canada. Edited by. Department of the Environment, Ottawa.

Rattner BA, Sileo L, Scanes CG. 1982. Oviposition and plasma concentrations of LH, progesterone, and corticosterone in bobwhite quail (*Colinus virginianus*) fed parathion. J. Reprod. Fert. **66:**147–155.

Ross PS, Ellis GM, Ikonomou MG, Barret-Lennards LG, Addison RF. 2000. High PCB concentrations in free-ranging Pacific killer whales, *Orcinus orca*: effects of age, sex, and dietary preference. Mar. Pollut. Bull. **40:**504–515.

Rouse JD, Bishop CA, Struger J. 1999. Nitrogen pollution: an assessment of its threat to amphibian survival. Environ. Health Perspect. **107:**799–803.

Ryckman DP, Weseloh DV, Hamr P, Fox GA, Collins B, Ewins PJ, Norstrom RJ. 1998. Spatial and temporal trends in organochlorine contamination and bill deformities in double-crested cormorants (*Phalacrocorax auritus*) from the Canadian Great Lakes. Environ. Monit. Assess. **53:**169–195.

Sanderson JT, Elliott JE, Norstrom RJ, Whitehead PE, Hart LE, Cheng KM, Bellward GD. 1994. Monitoring biological effects of polychlorinated dibenzo-p-dioxins, dibenzofurans, and biphenyls in great blue heron chicks (*Ardea herodias*) in British Columbia. J. Toxicol. Environ. Health **41:**435–450.

Sibly RM, Newton I, Walker CH. 2000. Effects of dieldrin on population growth rates of sparrowhawks 1963–1986. J. Appl. Ecol. **37:**540–546.

Smith LK, Elliott JE, Wilson L, Cheng KM. 2000. Effects of *in ovo* DDT exposure on behavior and reproduction in the American robin (*Turdus migratorius*). Abstract, SETAC-2000, Nashville, TN.

Smits JEG, Blakley BR, Wobeser GA. 1996. Immunotoxicity studies in mink (*Mustela vison*) chronically exposed to dietary bleached kraft pulp mill effluent. J. Wildlife Diseases **32:**199–206.

Smits JEG, Wobeser GA, Schieffer HB. 1995. Physiological, reproductive and pathological effects of dietary bleached pulp mill effluent on mink (*Mustela vison*). Environ. Toxicol. Chem. **14:**2095–2105.

Tillitt DE, Ankley GT, Giesy JP, Ludwig JP, Kurita-Matsuba H, Weseloh DV, Ross PS, Bishop CA, Sileo L, Stromborg KL, Larson J, Kubiak TJ. 1992. Polychlorinated biphenyl residues and egg mortality in double-crested cormorants from the Great Lakes. Environ. Toxicol. Chem. **11:**1281–1288.

Wayland M, Gilchrist HG, Dickson L, Braune B, Bollinger T, Marchant T, James C. 1999. Contaminants in arctic seas ducks, p 213–222. *In* Kalhok S (ed.) Synopsis of research conducted under the 1998–1999 Northern Contaminants Program, Department of Indian Affairs and Northern Development, Ottawa.

Wayland M, Gilchrist HG, Dickson L, Braune B, Bollinger T, Marchant T, Smits JEG. 2000. Contaminants in arctic sea ducks, p 237–243. *In* Kalhok S. (ed.) Synopsis of research conducted under the 1999–2000 Northern Contaminants Program, Department of Indian Affairs and Northern Development, Ottawa.

Wayland M, Trudeau S, Marchant T, Parker D, Hobson KA. 1998. The effect of pulp and paper mill effluent on an insectivorous bird, the tree swallow. Ecotoxicology 7**:**237–251.

Weseloh DV, Ewins PJ, Struger J, Mineau P, Bishop CA, Postupalsky S, Ludwig JP. 1995. Double-crested cormorants of the Great Lakes: changes in population size, breeding distribution and reproductive output between 1913 and 1991. Colonial Waterbirds **18** (Spec. Publ. 1)**:**48–59.

Whitehead PE, Norstrom RJ, Elliott JE. 1992. Dioxin levels in eggs of great blue herons (*Ardea herodias*) decline rapidly in response to process changes in a nearby kraft pulp mill. Dioxin '92. Organohalogen Compounds **9:**325–328.

Wright BS. 1960. Woodcock reproduction in DDT-sprayed areas of New Brunswick. J. Wildlife Manage. **24:**419–420.

Wright BS. 1965. Some effects of heptachlor and DDT on New Brunswick woodcocks. J Wildlife Manage. **29:**172–182.