CHEMICAL RESIDUES IN WATERFOWL AND GAMEBIRDS HARVESTED IN CANADA, 1987-95

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ABSTRACT

This report reviews data on chemical residues in Canadian waterfowl and gamebirds collected from across the country between 1987 and 1995 in order to: (i) obtain recent data on contaminants in gamebirds so that Health Canada could assess the risk to human health from eating those birds, and (ii) identify any potential avian health concerns related to the contaminant residue levels. Over 800 pools of waterfowl, gamebird and seabird muscle, egg and liver tissues were analysed for chlorobenzenes, chlordanes, hexachlorocyclohexanes, DDTs, mirex, dieldrin, octachlorostyrene, PCBs, arsenic, selenium, mercury, lead and cadmium. Selected subsets of samples were also analysed for polychlorinated dibenzo-*p*-dioxins (PCDDs), dibenzofurans (PCDFs), toxaphene or for ^{137}Cs activity. These contaminants were either not detected or were found at very low levels in the majority of samples. The highest levels of most of the contaminants measured were found in those birds feeding at the higher trophic levels such as mergansers, loons and gulls. ΣDDT and ΣPCB were the compounds most frequently found above trace levels in pectoral muscle followed by ΣCBz and ΣCHL. ΣHCH and OCS were detected the least often. The highest DDE, ΣPCB and ΣCHL levels found were 2.5 $mg \cdot kg^{-1}$, 2.4 $mg \cdot kg^{-1}$ and 0.17 $mg \cdot kg^{-1}$ wet weight respectively. Mercury and selenium were found in the majority of samples at generally low levels. In only a few cases did contaminant levels approach or exceed levels associated with potential avian health effects or potential concern for human or animal consumers of the birds.

RÉSUMÉ

Ce rapport contient des données sur les résidus chimiques chez la sauvagine et les oiseaux considérés comme gibier au Canada prélevés dans tout le pays entre 1987 et 1995 afin i) d'obtenir des données récentes sur la contamination des oiseaux considérés comme gibier de sorte que Santé Canada puisse évaluer le risque que présente la consommation de ces oiseaux pour la santé humaine, et ii) de déterminer les préoccupations éventuelles pour la santé humaine associées aux concentrations de contaminants. Plus de 800 échantillons de muscles, d'œufs et de tissus du foie d'espèces de sauvagine, d'oiseaux considérés comme gibier et d'oiseaux de mer ont été analysés afin de déceler la présence examinés de chlorobenzènes, de chlordanes, d'hexachlorocyclohexanes, de DDT, de mirex, de dieldrine, d'octachlorostyrène, de BPC, d'arsenic, de sélénium, de mercure, de plomb et de cadmium. Des sous-échantillons ont également été analysés afin de déceler la présence de dibenzo-*p*-dioxines polychlorées (PCDD), de dibenzofurannes polychlorés (PCDF), de toxaphène ou de césium-137 (^{137}Cs). Ces contaminants n'ont pas été détectés ou l'ont été à des concentrations très faibles dans la plupart des échantillons. Les concentrations les plus élevées de la plupart des contaminants mesurés ont été trouvées dans les oiseaux appartenant aux niveaux trophiques supérieurs comme les harles, les plongeons et les goélands. Les ΣDDT et les ΣBPC sont les composés qu'on a trouvés le plus souvent au-dessus de l'état de trace dans les muscles pectoraux, suivis des ΣCBz et des ΣCHL. Les ΣHCH et l'OCS ont été détectés le moins souvent. Les concentrations de DDE, de ΣBPC et de ΣCHL les plus élevées étaient en poids frais de 2,5 $mg \cdot kg^{-1}$, 2,4 $mg \cdot kg^{-1}$ et 0,17 $mg \cdot kg^{-1}$, respectivement. Le mercure et le sélénium ont été trouvés dans la plupart des échantillons à des concentrations généralement faibles. Dans seulement quelques cas, les niveaux de contamination approchaient ou dépassaient ceux associés à des effets éventuels sur la santé des oiseaux ou à une éventuelle préoccupation pour la santé des consommateurs humains ou celle des prédateurs d'oiseaux.

Summary

To address concerns about the presence of environmental contaminants in gamebirds, particularly waterfowl, harvested for human consumption, a national survey was initiated in 1988: (i) to obtain recent data on contaminants in gamebirds so that Health Canada could assess the risk to human health from eating those birds, and (ii) to identify species and locations where contaminant residue levels may indicate potential avian health risks. This report deals primarily with the chemical residue data and associated potential avian health effects but also summarizes the evaluation done by Health Canada to address the concerns of human consumers.

Species sampled were representative of the birds most frequently shot by hunters. Between 1987 and 1995, 3,957 birds (pooled into 834 composite samples) representing 44 species were collected from 126 sites across Canada. Pooled samples of pectoral muscle were analysed for chlorobenzenes (CBz), hexachlorocyclohexanes (HCH), chlordanes (CHL), DDTs, mirex, dieldrin, octachlorostyrene (OCS), PCBs, arsenic, selenium, mercury, lead and cadmium. Selected subsets of samples were also analysed for polychlorinated dibenzo-*p*-dioxins (PCDDs), dibenzofurans (PCDFs), toxaphene or for Cesium-137 (^{137}Cs) activity. The species of waterfowl were categorized based on their feeding habits: grazers (geese and swans), surface-feeding (dabbling) ducks, diving ducks (bay and sea ducks), fish-eating ducks (mergansers) and loons. Also collected were woodcock, snipe, grouse, ptarmigan, alcids, fulmars and gulls.

The concentrations of most organochlorine and PCB compounds in the pectoral muscle of the birds in this survey were relatively low compared with earlier studies reported in the literature. Although several significant differences in organochlorine levels among geographical areas were identified, these differences were scattered among trophic groups and chemical classes with no strong, consistent pattern emerging and usually occurred where median residue concentrations were very low. In general, the higher concentrations were found in eastern and northern Canada. There was also a general trend for increased organochlorine levels at higher trophic levels, most notably for DDE. Organochlorine levels were lowest in the upland gamebirds (i.e. grouse, ptarmigan), geese and swans, and highest in the fish-eating birds (mergansers and loons).

The alcids (murres, guillemots, puffins and dovekies) had organochlorine levels similar to those found in the dabbling ducks. Gulls had organochlorine levels that were comparable to those found in the mergansers and loons. There was, however, relatively little variation among species in any one trophic group except for the Oldsquaw which consistently had the widest range of organochlorine values found in the sea ducks. This may be linked to the fact that, of the sea duck species surveyed, Oldsquaw are the only species to overwinter in relatively large numbers in the Great Lakes - St. Lawrence River system, an area known to be contaminated with organochlorines.

ΣDDT (mainly DDE) and ΣPCB were the compounds most frequently found above trace levels in pectoral muscle followed by ΣCBz (mainly hexachlorobenzene) and ΣCHL. ΣHCH and OCS were detected the least often. The highest DDE level was found in a single Hooded Merganser (2.5 $mg \cdot kg^{-1}$ wet weight (ww)) from Labrador. The highest ΣPCB concentration (2.4 $mg \cdot kg^{-1}$ ww) was found in a single Common Merganser from northern Ontario. The highest ΣCHL concentration (0.17 $mg \cdot kg^{-1}$ ww) was in a pool of Common Loons from northern Ontario. The highest concentration of ΣCBz (0.12 $mg \cdot kg^{-1}$ ww) was found in a pool of Common Loons from northern Québec. Oxychlordane was generally the dominant chlordane breakdown product making up 30-50% of chlordanes, followed by heptachlor epoxide (20-30%) and *trans*-nonachlor (15-20%). Four PCB congeners (153, 138, 180 and 118) constituted approximately half or more of the PCBs in the waterfowl analysed.

Concentrations of all organochlorines and PCB congeners measured in eggs were less than 1 $mg \cdot kg^{-1}$ ww with the exception of ΣPCB in Herring Gull eggs from northern Québec (2.6 $mg \cdot kg^{-1}$ ww) and Red-breasted Merganser eggs from Great Slave Lake (1.6 $mg \cdot kg^{-1}$ ww).

Although generally not detected, measurable concentrations of PCDDs and PCDFs were found in loons, mergansers, scoter, scaup, Oldsquaw and Ring-necked Ducks. The highest levels of 2,3,7,8-TCDD and 1,2,3,7,8-PnDD were found in Lesser Scaup from Lac St-Pierre, Québec (9.7 $ng \cdot kg^{-1}$ and 12.5 $ng \cdot kg^{-1}$ ww, respectively). The highest level of 2,3,7,8-TCDF was found in Surf Scoter from the Fraser River Delta, British Columbia (13 $ng \cdot kg^{-1}$ ww). These pools are associated with areas of particular concern for

organochlorine pollution. Total toxaphene levels in most of the 12 pools of waterfowl pectoral muscle measured were estimated as less than 0.50 $mg \cdot kg^{-1}$ ww. The highest toxaphene level found was 9.4 $mg \cdot kg^{-1}$ ww in a pool of Oldsquaw from Teslin Lake, Yukon.

Only a very few samples out of a total of 834 pools of birds from across the country exceeded extant organochlorine or PCB threshold levels for risk to birds, or to wildlife or human consumers of the birds. Most of the muscle pools in this survey contained ΣPCB levels that were less than the 0.1 $mg \cdot kg^{-1}$ ww level which the International Joint Commission (IJC) believes can pose a health risk to wildlife consumers at higher trophic levels and well below levels associated with adverse effects in laboratory studies, and all but one pool of Common Merganser pectoral muscle (2.4 $mg \cdot kg^{-1}$ ww) were below the lowest guideline for human consumption of fish (2.0 $mg \cdot kg^{-1}$ ww). Most of the DDE/DDT residue levels found in this study were at least an order of magnitude less than Health Canada human consumption guidelines for poultry, although several pools of mergansers and loons exceeded IJC guidelines for hazard to fish-eating wildlife. Levels of dieldrin, mirex, HCB and toxaphene found in this study were also lower than those associated with adverse avian health effects found in other studies. PCB and organochlorine levels measured in eggs were generally quite low and unlikely to have toxic effects on the embryos.

Levels of cadmium, mercury, selenium and arsenic in pectoral muscle were, for the most part, either low (< 1 $mg \cdot kg^{-1}$ ww) or below detection limits. Unusually elevated levels of lead measured in several pools of birds were due to the presence of tiny fragments of lead shot embedded in the tissue. Arsenic was detected at low levels (generally ≤ 0.20 $mg \cdot kg^{-1}$ ww) in all of the woodcock, snipe and seabird samples but was detected in only 20-60% of pools of birds in other trophic categories, mainly in birds from eastern Canada. The highest level of arsenic (3.3 $mg \cdot kg^{-1}$ ww) was measured in a single Mallard from Labrador. Arsenic was detected in only 16 of 36 egg pools analysed and the highest level found was 0.27 $mg \cdot kg^{-1}$ ww. The highest level of cadmium was measured in King Eider from the Northwest Territories (0.71 $mg \cdot kg^{-1}$ ww). Cadmium was found at higher levels in the seabirds than in the waterfowl samples but was not detected in any of the eggs sampled. Mercury was almost always below detection limits in muscle of

ptarmigan, grouse, geese and swans. Most of the samples of other species were well below the 1.0 $mg \cdot kg^{-1}$ ww level of mercury associated with potential adverse health effects in waterfowl. The highest level of total mercury (1.9 $mg \cdot kg^{-1}$ ww) was found in a single Red-throated Loon from the Northwest Territories. Several pools of liver from western Arctic dabbling ducks and sea ducks had mercury levels that exceeded 1.0 $mg \cdot kg^{-1}$ ww, the highest being 3.8 $mg \cdot kg^{-1}$ ww in a pool of Surf Scoter from the Northwest Territories. Mercury was detected in most of the egg pools analysed but at very low levels ($\leq$ 0.50 $mg \cdot kg^{-1}$ ww). Selenium was detected in over 75% of all the pools of birds measured (generally < 2 $mg \cdot kg^{-1}$ ww), the highest being in a single Black Scoter from Québec (10 $mg \cdot kg^{-1}$ ww). Selenium was detected in all 33 egg pools sampled (maximum 1.0 $mg \cdot kg^{-1}$ ww). The highest selenium level (23.0 $mg \cdot kg^{-1}$ ww) in liver was found in a pool of Common Eider from the Northwest Territories.

^{137}Cs activity, measured in 238 pools (37 species), was generally < 3 $Bq \cdot kg^{-1}$ ww. Geese and swans had the highest activity, but grouse and ptarmigan also had elevated activity. Marine species such as alcids, kittiwakes and sea ducks had very low activity.

Overall the organochlorines and metals measured in this study were either below detection limits or were found at very low levels. In only 14 pools (1.7%) of the 834 pools analysed did organochlorine contaminant levels approach or exceed levels that could pose a threat to the health of the birds or to animal consumers of the birds. The highest concentrations of most contaminants were found in those species feeding at the higher trophic levels such as mergansers, loons and gulls. Based on the data from this study and some earlier Ontario data, Health Canada advised that contaminant levels found in samples of pectoral muscle of ducks, geese and other gamebirds analysed do not pose a health hazard to human consumers.

Sommaire

Afin de répondre aux préoccupations concernant la présence de contaminants chez les oiseaux considérés comme gibier, particulièrement la sauvagine, capturés pour la consommation humaine, une étude nationale a été entreprise en 1988 pour : i) obtenir des données récentes sur la contamination des oiseaux considérés comme gibier afin que Santé Canada puisse évaluer le risque entourant la consommation de ces oiseaux pour la santé humaine et ii) déterminer les espèces et les endroits où les concentrations de contaminants peuvent présenter des risques éventuels pour la santé des oiseaux. Ce rapport porte principalement sur les données concernant les résidus chimiques et leurs effets éventuels sur la santé des oiseaux; il résume également l'évaluation faite par Santé Canada pour répondre aux préoccupations des consommateurs.

Les espèces ont été échantillonnées parmi celles que récollent le plus fréquemment les chasseurs. Entre 1987 et 1995, 3 957 oiseaux (regroupés en 834 échantillons composites) représentant 44 espèces ont été prélevés dans 126 sites dans l'ensemble du Canada. Des échantillons de muscles pectoraux ont été analysés afin de déceler la présence de chlorobenzènes (CBz), d'hexachlorocyclohexanes (HCH), de chlordanes (CHL), de DDT, de mirex, de dieldrine, d'octachlorostyrène (OCS), de BPC, d'arsenic, de sélénium, de mercure, de plomb et de cadmium. Certains sous-échantillons ont également été analysés afin de déceler la présence de dibenzo-*p*-dioxines polychlorées (PCDD), de dibenzofurannes polychlorés (PCDF), de toxaphène ou de césium-137 (^{137}Cs). Les espèces de sauvagine ont été classées selon leurs régimes alimentaires : brouteurs (oies et cygnes), alimentation de surface (canards barboteurs), canards plongeurs (canards de baie et de mer), canards piscivores (harles) et plongeons. On a également prélevé des échantillons de bécasses, de bécassines, de gélinottes, de lagopèdes, d'alcidés, de fulmars et de goélands.

Les concentrations de la plupart des pesticides organochlorés et de BPC dans les muscles pectoraux des oiseaux de cette étude étaient relativement faibles comparativement à celles d'études antérieures. Bien que plusieurs différences importantes entre les concentrations de pesticides organochlorés et de BPC selon les régions géographiques aient été notées, ces différences étaient dispersées parmi les groupes

trophiques et les catégories de produits chimiques, sans qu'un modèle uniforme solide se présente et habituellement là où des concentrations moyennes de résidus étaient très faibles. En général, on a constaté des concentrations supérieures de contaminants dans l'est et dans le nord du Canada. Des concentrations de composés organochlorés plus élevées ont également été constatées, en règle générale, chez les niveaux trophiques supérieurs, notamment pour le DDE. Ces mêmes concentrations étaient les plus faibles chez le gibier à plumes sédentaire (p. ex., gélinottes, lagopèdes), les oies et les cygnes, et les plus élevées chez les oiseaux piscivores (harles et plongeons). Les alcidés (marmettes, guillemots, macareux et mergules) présentaient des concentrations de produits organochlorés semblables à celles constatées chez les canards barboteurs. Les goélands présentaient des niveaux d'organochlorés analogues à ceux observés chez les harles et les plongeons. On a toutefois constaté une variation relativement faible entre les espèces d'un même niveau trophique, sauf pour l'Harelde kakawi qui présentait uniformément les plus fortes concentrations de composés organochlorés trouvées chez les canards de mer. Ce phénomène peut être associé au fait que, parmi les espèces de canards de mer étudiées, l'Harelde kakawi est la seule espèce à hiverner en assez grand nombre dans le bassin des Grands Lacs et du fleuve Saint-Laurent, région connue comme étant contaminée par les composés organochlorés.

Les ΣDDT (surtout le DDE) et les ΣBPC étaient les composés qu'on trouvait le plus souvent au-dessus de l'état de trace dans les muscles pectoraux, suivis des ΣCBz (surtout l'hexachlorobenzène) et des ΣCHL. Les ΣHCH et l'OCS ont été détectés le moins fréquemment. La concentration de DDE la plus élevée provient d'un seul Harle couronné (2,5 $mg \cdot kg^{-1}$ poids frais (pf)) du Labrador. La concentration de ΣBPC la plus élevée (2,4 $mg \cdot kg^{-1}$ pf) a été constatée chez un Grand Harle du nord de l'Ontario. La concentration de ΣCHL la plus élevée (0,17 $mg \cdot kg^{-1}$ pf) se trouvait chez un groupe de Plongeons huards du nord de l'Ontario. La concentration de ΣCBz la plus élevée (0,12 $mg \cdot kg^{-1}$ pf) a été constatée chez un groupe de Plongeons huards du nord du Québec. L'oxychlordane était généralement le métabolite dominant du chlordane, constituant de 30 à 50 p. 100 des chlordanes, suivi de l'époxyde d'heptachlore (20 à 30 p. 100) et du *trans*-nonachlore (15 à 20 p. 100). Quatre congénères de BPC (nº 153, 138, 180 et 118) constituaient environ la moitié ou plus des BPC dans les échantillons de sauvagine analysés.

Les concentrations dans les œufs de tous les pesticides organochlorés et des congénères de BPC étaient inférieures à 1 $mg \cdot kg^{-1}$ pf à l'exception des ΣBPC dans les œufs de Goélands argentés du nord du Québec (2,6 $mg \cdot kg^{-1}$ pf) et les œufs de Harles huppés du Grand lac des Esclaves (1,6 $mg \cdot kg^{-1}$ pf).

Bien qu'elles ne soient généralement pas détectées, des concentrations mesurables de PCDD et de PCDF ont été trouvées dans les plongeons, les harles, les macreuses, les Fuligules milouinans, les Hareldes kakawi et les Fuligules à collier. Les concentrations les plus élevées de 2,3,7,8-TCDD et 1,2,3,7,8-PnDD ont été constatées dans le Petit Fuligule du Lac Saint-Pierre (Québec) (9,7 $ng \cdot kg^{-1}$ et 12,5 $ng \cdot kg^{-1}$ pf, respectivement). La concentration la plus élevée de 2,3,7,8-TCDF a été signalée chez la Macreuse à front blanc du delta du fleuve Fraser (Colombie-Britannique) (13 $ng \cdot kg^{-1}$ pf). Ces résultats sont associés à des régions où la pollution par les composés organochlorés est particulièrement préoccupante. Les concentrations totales de toxaphène de la plupart des 12 groupes de muscles pectoraux de la sauvagine mesurés ont été estimées à moins de 0,50 $mg \cdot kg^{-1}$ pf. La concentration de toxaphène la plus élevée était de 9,4 $mg \cdot kg^{-1}$ pf dans un groupe d'Hareldes kakawi du lac Teslin (Yukon).

Sur un total de 834 groupes d'oiseaux provenant de tout le pays, très peu d'échantillons dépassaient les valeurs seuils des pesticides organochlorés ou de BPC posant un risque pour les oiseaux, les espèces sauvages ou pour les êtres humains qui consomment ces oiseaux. La plupart des échantillons groupes de muscles présentaient des concentrations de BPC inférieures à 0,1 $mg \cdot kg^{-1}$ pf qui, selon la Commission mixte internationale (CMI), peuvent poser un risque pour la santé des espèces sauvages des niveaux trophiques supérieurs. Ces concentrations sont aussi inférieures à celles associées aux effets néfastes démontrés lors d'études en laboratoire. De plus, les concentrations dans les muscles pectoraux de tous les échantillons d'oiseaux sauf ceux du Grand Harle (2,4 $mg \cdot kg^{-1}$ pf), étaient inférieures à celles de la ligne directrice la plus basse pour la consommation de poisson (2,0 $mg \cdot kg^{-1}$ pf) par l'homme. Les résidus de DDE et de DDT observées lors de cette étude étaient au moins d'un ordre de grandeur inférieur à la directive de Santé Canada pour la consommation humaine de volaille, bien que plusieurs groupes de harles et de plongeons aient dépassé les lignes directrices de la CMI pour les dangers posés par les espèces de sauvagine piscivores. Les concentrations de dieldrine, de

mirex, de HCB et de toxaphène observées lors de cette étude étaient également inférieures à celles associées aux effets néfastes sur la santé des oiseaux rapportés pars d'autres études. Les concentrations de BPC et de pesticides organochlorés mesurées dans les œufs étaient généralement très basses et peu susceptibles d'avoir des effets toxiques sur les embryons.

Les concentrations de cadmium, de mercure, de sélénium et d'arsenic dans les muscles pectoraux étaient, pour la plupart, faibles (< 1 $mg \cdot kg^{-1}$ pf) ou en dessous des limites de détection. Les concentrations exceptionnellement élevées de plomb mesurés dans plusieurs échantillons d'oiseaux étaient attribuables à la présence de minuscules fragments de grenaille de plomb incrustés dans la chair. Les concentrations d'arsenic étaient généralement inférieures ou egales à 0,20 $mg \cdot kg^{-1}$ pf dans tous les échantillons de bécasses, de bécassines et d'oiseaux de mer; il a toutefois été détecté dans 20 à 60 p. 100 des échantillons des autres catégories trophiques, principalement chez les oiseaux de l'est du Canada. La concentration d'arsenic la plus élevée (3,3 $mg \cdot kg^{-1}$ pf) a été mesurée dans un Canard colvert provenant du Labrador. L'arsenic a été détecté dans seulement 16 des 36 groupes d'œufs analysés et la concentration la plus élevée était de 0,27 $mg \cdot kg^{-1}$ pf. La concentration de cadmium la plus élevée a été constatée chez un Eider à tête grise des Territoires du Nord-Ouest (0,71 $mg \cdot kg^{-1}$ pf). Les concentrations de cadmium étaient plus élevées chez les oiseaux de mer comparativement à la sauvagine; il n'a cependant pas été détecté dans aucun échantillon d'œuf. Le mercure était presque toujours sous les limites de détection dans les muscles des lagopèdes, des gélinottes, des oies et des cygnes. La plupart des échantillons des autres espèces étaient bien en deçà des concentrations de mercure de 1,0 $mg \cdot kg^{-1}$ pf associées à des effets néfastes possibles sur la santé de la sauvagine. La concentration totale la plus élevée de mercure (1,9 $mg \cdot kg^{-1}$ pf) se trouvait chez un Plongeon catmarin des Territoires du Nord-Ouest. Plusieurs échantillons de foie de canards barboteurs et de canards de mer de l'ouest de l'Arctique présentaient des concentrations de mercure supérieures à 1,0 $mg \cdot kg^{-1}$ pf, la plus élevée étant à 3,8 $mg \cdot kg^{-1}$ pf dans un échantillons de Macreuses à front blanc des Territoires du Nord-Ouest. Le mercure a été détecté dans la plupart des échantillons d'œufs analysés mais à des concentrations très basses (0,50 $mg \cdot kg^{-1}$ pf). Le sélénium a été détecté dans plus de 75 p. 100 de tous les échantillons d'oiseaux (généralement < 2 $mg \cdot kg^{-1}$ pf), le plus élevé étant dans une Macreuse noire du Québec (10 $mg \cdot kg^{-1}$ pf). Le sélénium a été signalé dans les

33 échantillons d'œufs (maximum 1,0 $mg \cdot kg^{-1}$ pf). La concentration de sélénium la plus élevée (23,0 $mg \cdot kg^{-1}$ pf) dans le foie se trouvait dans un groupe d'Eiders à duvet des Territoires du Nord-Ouest.

L'activité du ^{137}Cs, mesurée dans 238 échantillons (37 espèces) était généralement inférieure à 3 $Bq \cdot kg^{-1}$ pf. Les oies et les cygnes présentaient le niveau d'activité le plus élevé; celui des gélinottes et des lagopèdes étaient toutefois aussi élevé. Les espèces marines comme les alcidés, les Mouettes tridactyles et les canards de mer présentaient une activité très faible.

Dans l'ensemble, les composés organochlorés et les métaux mesurés dans cette étude étaient en dessous des limites de détection ou leurs concentrations étaient très faibles. Dans seulement 14 (1,7 p. 100) des 834 échantillons analysés les concentrations de contaminants organochlorés approchaient ou dépassaient celles qui pourraient poser une menace pour la santé de ces oiseaux ou de leurs prédateurs. Les concentrations les plus élevées de la plupart des contaminants se trouvaient chez les espèces appartenant aux niveaux trophiques supérieurs comme les harles, les plongeons et les goélands. D'après les données de cette étude et certaines données antérieures recueillies en Ontario, Santé Canada a indiqué que les concentrations de contaminants trouvées dans les échantillons de muscles pectoraux des canards, des oies et d'autres espèces d'oiseaux considérés comme gibier ne présentent aucun danger pour la santé des consommateurs humains.

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1.0 Introduction

There has been an ongoing concern about the presence of environmental contaminants in gamebirds, particularly waterfowl, harvested for consumption by both recreational and subsistence hunters in Canada (Pearce and Baird, 1970; Wong, 1985; Nin.Da.Waab.Jig Research Centre, 1986; Whitehead *et al.,* 1990; Kuhnlein, 1991; Kinloch *et al.*, 1992; Kearney *et al.,* 1995; Egede, 1995; Usher *et al.*,1995; Receveur *et al.,* 1996; Dewailly *et al.*, 1996; Jensen et al., 1997; Berti *et al.*, 1998). A review by Braune *et al.* (1991), which included Canadian data on contaminants in gamebirds available up to August 1988, indicated that there was a lack of recent data with which to evaluate the extent and potential impacts of contaminant residues in harvested birds. Most residue surveys of Canadian gamebirds have focused on mercury, DDT (and its metabolites) and PCB contamination. Historically, mercury has caused the most concern, with several studies from the early 1970s indicating high levels in edible tissues (Fimreite *et al*., 1971; Vermeer *et al*., 1973; Annett *et al*., 1973; Fimreite, 1974; Desai-Greenaway and Price, 1976; Pearce *et al*., 1976). Studies on organochlorine compounds in waterfowl collected from the Detroit River (Smith *et al*., 1985) and Lake St. Clair (Hebert *et al*., 1990) have also suggested concern for the human consumer. Despite regulatory measures taken in the 1970s to reduce discharges of mercury and organochlorine compounds such as PCBs and DDT into the environment, organochlorine and metal contamination continues to be a problem in some areas (Jacknow *et al*., 1986; Braune *et al*., 1991).

There has long been a potential for radionuclide contamination in the Canadian environment due to past atmospheric nuclear testing, the operation of nuclear facilities and accidents such as Three Mile Island and Chernobyl but relatively little monitoring has been carried out in Canada's natural environment, beyond the measurement of fallout in air and precipitation (Taylor *et al.,* 1988; Joshi, 1991; Thomas *et al.,* 1992, Health and Welfare Canada, 1987). Most of the studies that have been done were on the lichen-

caribou food chain (MacDonald *et al.,* 1996) or in the aquatic environment (Joshi, 1991). Since Cesium-137 (^{137}Cs) tends to be retained in muscle tissue of free-living waterfowl (Potter *et al.,* 1989) but very little data exist for wild birds, a subset of the birds collected for this study were made available for ^{137}Cs determination.

To address the concerns raised about contaminant levels in harvested birds, a national survey was initiated in 1988. The objectives of the survey were: (i) to obtain recent data on contaminants in gamebirds so that Health Canada could assess the potential health risk to human consumers, and (ii) to identify species and locations where contaminant residue levels may indicate potential avian health risks. The survey progressively focused on different regions across Canada during 1988-95. After completing preliminary collections nation-wide during 1988-90, collections focused on Ontario in 1990-91, Québec in 1991-92, Atlantic Canada (Newfoundland & Labrador, Nova Scotia, Prince Edward Island and New Brunswick) in 1992-93, the Prairie provinces (Manitoba, Saskatchewan and Alberta) and the Northwest Territories in 1993-94, and British Columbia and the Yukon Territory in 1994-95.

2.0 Methods

2.1 Sample collection

Species and areas to be sampled were selected by reviewing the statistics of estimated retrieved kill of waterfowl for Canadian provinces and territories compiled through the Canadian Wildlife Service's Species Composition and National Harvest Surveys (Boyd, 1985; Dickson and Métras, 1987; Legris and Lévesque, 1991; Lévesque *et al*., 1993). The data from those surveys were divided into 10-minute blocks of latitude and longitude and analysed to determine the frequently hunted species and the most heavily hunted areas. There were few data for the remote northern areas of Canada and there was little information on the traditional native use of wildlife as a food resource since native hunters are not included in the annual licenced hunter surveys. Although geese dominate the native waterfowl harvest in many areas, ptarmigan, grouse and diving/fish-eating species are also often taken (Prevett *et al*., 1983; Gamble, 1987; Alton Mackey and Moore Orr, 1988; James Bay and Northern Québec Native Harvesting Research Committee, 1988; Kuhnlein, 1991; Wein *et al*., 1991; Cameron and Weis, 1993; Tobias and Kay, 1993; Berkes *et al*., 1994; Wein and Freeman, 1995; Bromley, 1996). Therefore, the species sampled were chosen to be representative of the birds most frequently shot by hunters and included ptarmigan, grouse and diving/fish-eating birds, particularly from northern areas.

Five to ten birds, representative of the hunted population, were requested per species per collection. Most birds were shot using lead shot during the fall hunting season. Those collections taken outside of the fall hunting season were collected as part of the native harvest or under Canadian Wildlife Service Scientific Collection Permits. Birds were individually placed in labelled plastic bags, frozen whole, and shipped to the National Wildlife Research Centre (NWRC), Hull, Québec, for processing. Although the main

waterfowl collection period spanned 1988-95, preliminary collections made in Ontario during 1987 have also been included in the data set.

2.2 Sample processing

All birds were processed at the NWRC laboratories. The pectoral muscle was chosen as representative of the edible portion of the bird. Tests to determine the variability in organochlorine concentration among different sections of pectoral muscle concluded that a median section was representative of the whole (Canadian Wildlife Service, 1990). Therefore, equal amounts of the median sections of right pectoral muscle from each bird in the pool were homogenized together, placed in solvent-cleaned glass jars for organochlorine residue analyses and acid-rinsed polyethylene vials for metal analyses, and stored at -20°C. Pools were created by taking equal aliquots from each bird of a given species collected from a specific location during a period of no more than four weeks were to make a composite sample for analysis. Pools varied in size depending on availability of specimens. A pool could contain a mix of birds of different age and sex, but of the same species, intended to represent the mix of birds harvested.

It has been shown that any differences between the means of individual analyses and the results from pooled samples are generally less than the normal analytical variation (Turle and Collins, 1992). Comparative analyses of individual and pooled samples of Lesser Snow Goose, Mallard, Black Duck, Greater Scaup, Common Eider, King Eider, Glaucous Gull and Herring Gull from this survey support that conclusion.

The remainder of the right pectoral muscle, the left pectoral muscle with subcutaneous fat and skin attached, liver, kidney, adipose fat, one wing and one femur from each bird were archived at -40°C in NWRC’s Specimen Bank for possible future analysis.

Samples destined for ^{137}Cs analysis were not homogenized, rather equal amounts of muscle strips from the median sections of right pectoral muscle from each bird in the pool were packed together in a 100 mm plastic petrie dish and stored at -20°C.

2.3 Chemical residue analyses

Samples were analysed for chlorobenzenes (ΣCBz = 1,2,4,5- and 1,2,3,4-tetrachlorobenzene, pentachlorobenzene and hexachlorobenzene), hexachlorocyclohexanes (ΣHCH = α-, β- and γ-hexachlorocyclohexane), chlordanes (ΣCHL = oxychlordane, *trans*-chlordane, *cis*-chlordane, *trans*-nonachlor, *cis*-nonachlor and heptachlor epoxide), DDTs (ΣDDT = *p,p'*-DDE, *p,p'*-DDD and *p,p'*-DDT), mirex (ΣMirex = photo-mirex and mirex), dieldrin, octachlorostyrene (OCS), and PCBs as Aroclor 1254:1260 (1:1 mixture) and on a congener-specific basis. The number of PCB congeners analysed increased over time and therefore, total PCBs (ΣPCB) were standardized as the sum of 32 PCB congeners reported according to IUPAC numbers (Ballschmiter and Zell, 1980): 28, 31, 44, 52, 60, 66/95, 87, 97, 99, 101, 105, 110, 118, 138, 141, 146, 153, 170/190, 171, 172, 174, 180, 182/187, 183, 194, 195, 201, 203, 206. ΣPCB for some of the pools is underestimated due to fewer than the 32 congeners being measured at the time of their analysis.

Sample aliquots were extracted, cleaned up and analysed using gas chromatography with electron capture detector for organochlorine compounds according to the procedure detailed in Norstrom and Won (1985). PCB congeners were analysed according to the method described in Norstrom *et al*. (1988). Analyses were carried out by the Great Lakes Institute, University of Windsor, Windsor, Ontario, or by NWRC.

A subset of samples was analysed for polychlorinated dibenzo-*p*-dioxins (PCDDs)

and dibenzofurans (PCDFs) by gas chromatography-mass spectrometry as described in Norstrom and Simon (1991). Later PCDD and PCDF and non-ortho PCB analyses were done according to the method of Moisey and Wakeford (1995). Analyses were carried out by NWRC or by Zenon Environmental Laboratories, Burnaby, British Columbia.

A limited number of samples were also analysed for toxaphene by low resolution mass spectrometry using electron capture negative ionization methodology coupled with a gas chromatograph adapted from Zhu and Norstrom (1993). Identification of toxaphene congeners was based on the isotope ratio of the two monitored ions in each M-Cl cluster. These analyses were carried out by NWRC.

Samples were also analysed for arsenic (As), selenium (Se), total mercury (Hg), lead (Pb) and cadmium (Cd). Samples destined for metal determinations were digested in mineral acids according to standard techniques (Association of Official Analytical Chemists, 1990). Arsenic and selenium determinations were made on acid digests using one of two analytical methods: inductively coupled plasma - mass spectrometry (ICP-MS) or graphite furnace atomic absorption spectrophotometry (GFAAS). Total mercury determinations on acid digests were completed using the technique of cold vapour atomic absorption spectrophotometry (CVAAS). Although mercury values in pectoral muscle are reported as total mercury it should be noted that most muscle tissue mercury occurs as methyl mercury (Scheuhammer *et al.*, 1998). Cadmium determinations were made on the digests using flame atomic absorption spectrophotometry (AAS). Lead determinations were made on acid digests using either GFAAS or AAS. Samples collected prior to 1990 were analysed by NWRC or by Mann Testing Laboratories, Mississauga, Ontario. Samples collected during 1990-93 were analysed by NWRC (Hg, Cd, Pb) and by Fenwick Laboratories (now Philip Analytical Services Corporation), Halifax, Nova Scotia (As, Se), and 1993-95 samples were all analysed at NWRC (Hg, Cd, Pb, As, Se).

Quality assurance for all residue analyses included the analysis of replicate samples,

reagent blanks, spiked tissue homogenates, and appropriate Standard Reference Materials (Turle and Norstrom, 1987; Wakeford and Turle, 1997).

The ^{137}Cs activity was determined with a solid state gamma-ray detector and a multi-channel analyser by the Fallout and Reactors Section, Radiation Protection Branch, Health Canada, Ottawa. The detector was a high-purity germanium detector in a co-axial configuration, mounted vertically; the petrie dishes containing the samples were placed directly on the aluminum case of the detector, centred on the cylindrical axis. In this arrangement, the size and shape of the sample are critical to detection efficiency and it is for this reason that the dishes were packed full, to achieve a reproducible volume and shape. Any slight differences in sample density are inconsequential. Each sample was counted for at least 22 hours. The specific activity of each sample was calculated from the determined activity and the exact weight.

2.4 Data presentation

All data are presented on a wet weight (ww) basis. The primary rationale for this approach is that it most accurately reflects the exposure of consumers to contaminants in the foods that they are eating. With respect to the metal residue data, earlier residue analyses were done on a wet weight basis whereas the later analyses were done on a dry weight basis. Therefore, all metals data were standardized to a wet weight basis and percent water given to facilitate conversions.

With respect to organochlorine data, the differing lipid and moisture content between eggs and pectoral muscle must be considered. Peakall and Gilman (1979) have shown that during incubation of eggs, percent water changes only marginally whereas percent lipid decreases by as much as 50%. Therefore, expressing contaminant concentration in the egg on a lipid weight basis may introduce considerable variation, depending on the

incubation state of the egg. Hebert and Keenleyside (1995) tested the relationship between organochlorine residue levels and percent lipid in Herring Gull eggs from the Great Lakes and found no correlation between the two.

Hebert and Keenleyside (1995) also proposed an approach for determining whether organochlorine residue data should be statistically analysed on a wet weight or lipid weight basis. The first step is to test for a relationship between wet weight residue levels and percent lipid. If no correlation is found to exist, then a wet weight basis is used for data reporting.

We tested the relationship between contaminant level and percent lipid separately for each trophic grouping and contaminant using regression analysis (SAS Institute, 1988). As well, the data were plotted on a trophic level and species basis (SAS Institute, 1988). Only 9 of 136 comparisons were significant at $\alpha = 0.05$ and there were no patterns of significance evident with respect to trophic level, species or residue. We concluded that both the reporting and the statistical analyses of these data should be done on a wet weight rather than a lipid weight basis.

Most of the graphical data in this report are presented as Box and Whisker Plots (McGill *et al*., 1978) which consist of a box enclosing the 25th and 75th percentiles of the data and bisected by the median of the data (50th percentile). The "whiskers" mark the 10th and 90th percentiles, and any data points outside the "whiskers" are indicated by bullets (Tilling *et al*., 1984) .

2.5 Data analysis

No correction was made for varying sex or age ratios within the pools. While there is a considerable literature addressing the effects of age and sex on contaminant uptake

(Heath, 1969; Baskett, 1975; Anderson and Hickey, 1976; White and Kaiser, 1976; Holt *et al*., 1979; Hebert *et al*., 1990; Gebauer and Weseloh, 1993), there is no general consensus on which effects exist and under what conditions. Sexual differences in contaminant levels were assumed to be negligible in juvenile birds. Since the majority of the birds were collected during the fall hunting season, which starts 3 to 4 months after egg-laying, it is assumed that any potential difference in contaminant levels in adult males and females due to females eliminating contaminants into eggs will have been equilibrated through rapid uptake of contaminants. Although the literature is inconclusive, a variety of studies provide evidence to support this assumption for organochlorines (Holt *et al*., 1979; Gebauer and Weseloh, 1993) as well as metals (Peterson and Ellarson, 1976; Holt *et al*., 1979; Silver and Nudds, 1995) and selenium (Heinz *et al*., 1990; Ohlendorf *et al*., 1990). Adults hunted in the spring were collected prior to egg-laying so the females had not yet eliminated contaminants into the eggs. It is also assumed that differences in contaminant levels between juvenile and adult birds collected in the fall will also have been minimized through rapid uptake of contaminants during the months since hatch for organochlorines (Holt *et al*., 1979; Hebert *et al*., 1990; Gebauer and Weseloh, 1993) as well as metals (Peterson and Ellarson, 1976; White and Finley, 1978; Holt *et al*., 1979) and selenium (Heinz *et al*., 1990; Ohlendorf *et al*., 1990). Similar assumptions were made for ^{137}Cs as described in Braune and Marshall (in prep.).

For purposes of comparing waterfowl collected from different sites, Canada was divided into 11 regions (Figure 1) based on a combination of political boundaries, migration pathways and geological boundaries. Collection locations shown in Figure 1 are identified in Table 1. The species of waterfowl have been categorized based on their general food or foraging habits (Robbins *et al*., 1983; Ehrlich *et al*., 1988): grazers (geese and swans), dabbling ducks, diving ducks (bay and sea ducks), fish-eating ducks (mergansers) and loons. See Table 2 for a list of species in each group. Insectivores, terrestrial browsers, alcids, fulmars and gulls are also included in the table. A summary of

the breeding season diet for each of these groups (Figure 2) was based on data from Ehrlich *et al*. (1988) which was adjusted for the relative abundances of each species within its trophic group.

Individual organochlorine compounds were grouped into related classes for data presentation: polychlorinated biphenyls (ΣPCB), technical chlordane-related compounds (ΣCHL), chlorobenzenes (ΣCBz), DDT metabolites (ΣDDT), hexachlorocyclohexanes (ΣHCH), mirex and photo-mirex (ΣMirex), octachlorostyrene (OCS), and dieldrin. PCBs were also presented as Aroclor 1254:1260 (1:1) to facilitate comparisons with earlier literature values.

The limit of quantitation (LQ) for organochlorine and PCB data was set at 0.001 $mg \cdot kg^{-1}$ although for many of the parameters, trace amounts between 0.0001 $mg \cdot kg^{-1}$ (DL, the detection limit) and 0.001 $mg \cdot kg^{-1}$ (the limit of quantitation) were present. For purposes of data analysis, it was preferable to use the measured trace values, where available, rather than use a constant replacement value such as 1/2 LQ for values below the limit of quantitation (Porter *et al*., 1988). Statistical analyses were not carried out for metals because of the extreme variability of detection limits among samples and because of the high ratio of non-detects to detects in the database. Statistical analyses were not carried out for dioxins and furans because of the small number of analyses.

Because pooling masks a large part of the variance in the data and the number of individuals in each pool was variable, normal statistical methods were not appropriate. Randomization tests (Collins, 1995) were used to test for differences in contaminant levels among geographical areas of Canada by trophic group, and to test for differences in residue concentrations among the trophic groups on a nation-wide basis (Sokal and Rohlf, 1981; SAS Institute, 1988). For each trophic group, three tests were done to compare geographical areas: (i) east (regions 1-7, 10) vs. west (regions 8, 9, 11); (ii) north (regions

10 & 11) vs. south (regions 1-9); and (iii) northeast (region 10) vs. northwest (region 11) vs. southeast (regions 1-7) vs. southwest (regions 8 & 9). Randomization tests were not carried out on any data block where more than 40% of the values were non-detects. All tests were performed on residue data on a wet weight basis.

3.0 Results

A total of 3,957 individual birds (excluding coots and seabird species, i.e. alcids, fulmars, gulls) representing 44 species was collected from 126 sites across Canada between 1987 and 1995 (Tables 1 and 2, Figure 1). Birds were pooled into 834 composite samples for analyses of chemical residues. All data are presented on a wet weight (ww) basis unless otherwise indicated.

3.1 Organochlorines and PCBs

3.1.1 Pectoral muscle

The groups of organochlorine compounds most frequently found above trace level in pectoral muscle of all birds were ΣDDT (mainly DDE) and ΣPCB followed by ΣCBz (mainly HCB) and ΣCHL (Tables 3 and 4 and Figure 3). Those birds (i.e. grouse, ptarmigan and geese) whose diet consists mainly of terrestrial vegetation had the lowest percentage of detectable organochlorine residues. The greatest percentage of detectable organochlorine residues was found in the fish-eating groups (mergansers and loons). ΣHCH and OCS were detected the least often (29% and 23% respectively) of all the organochlorines measured.

The highest DDE concentrations in waterfowl pectoral muscle were found in a single Hooded Merganser (2.5 $mg \cdot kg^{-1}$) from Goose Bay (Labrador), in a pool of Common Loons (1.3 $mg \cdot kg^{-1}$) from Weagamow Lake (northwestern Ontario) and in a single Red-throated Loon (1.1 $mg \cdot kg^{-1}$) from Long Lake (NWT) (Tables 3 and 4).

The highest ΣPCB concentration in pectoral muscle (2.4 $mg \cdot kg^{-1}$) was found in a

single Common Merganser from the Sault Ste. Marie area of northern Ontario (Tables 3 and 4). Elevated levels (>1.0 $mg \cdot kg^{-1}$) were also found in Common and Red-breasted Mergansers from the James Bay area of central Québec (1.0, 1.5 & 2.3 $mg \cdot kg^{-1}$), and from the Great Whale (1.6 $mg \cdot kg^{-1}$) and Ungava (1.7 $mg \cdot kg^{-1}$) areas of northern Québec. High levels were found in Common Loons from Big Trout (1.0 $mg \cdot kg^{-1}$) and Weagamow (1.2 $mg \cdot kg^{-1}$) Lakes in northwestern Ontario and from Inukjuaq (1.2 $mg \cdot kg^{-1}$) on the eastern shore of Hudson Bay. The only high levels found outside of the fish-eating birds was in a pool of Oldsquaw (1.1 $mg \cdot kg^{-1}$) from Arviaq on the western shores of Hudson Bay.

The highest ΣCHL concentration (0.17 $mg \cdot kg^{-1}$) in pectoral muscle was in one of the pools of Common Loon from Weagamow Lake in northwestern Ontario. The highest concentration of ΣHCB (0.12 $mg \cdot kg^{-1}$) was found in a pool of Common Loon from the Salluit/Ivujivik area of northern Québec (Tables 3 and 4).

In the terrestrial browsers, dabbling ducks, bay ducks, sea ducks and mergansers, oxychlordane was the dominant chlordane breakdown product making up 30 - 50% of chlordanes (Figure 4), followed by heptachlor epoxide (20-30%) and *trans*-nonachlor (15-20%). In the grazers and insectivores, heptachlor epoxide was the dominant chlordane making up 45-60% of the total. The loons were the most idiosyncratic with *trans*-nonachlor making up 50% of the chlordane residues.

Four PCB congeners (153, 138, 180 and 118) constituted approximately half or more of the PCBs in the various waterfowl groups. Five PCB congeners (182, 153, 138, 180 and 118) constituted over half the PCBs in the insectivores, and three PCB congeners (49, 153 and 118) constituted over half the PCBs in terrestrial browsers. PCB congener data are presented in Table 5.

The distribution of PCB homolog groups was very similar across all trophic groups of birds (Figure 5) with the hexachlorobiphenyls being the dominant group closely followed by the penta- and heptachlorobiphenyls. The only variations were in the grazers and terrestrial browsers where the tri- and tetrachlorobiphenyls were a little more common. In contrast the higher chlorinated groups were more common in the insectivores.

3.1.1.1 Geographical variation in organochlorine and PCB concentrations

Several significant differences among geographical areas in organochlorine levels in trophic groups were identified by the randomization tests (Tables 6 and 7). These differences were scattered among trophic groups and chemical classes with no strong, consistent pattern emerging. In most cases where significant differences were indicated, median residue concentrations were very low. Where significant differences were determined between east vs west, higher concentrations were found in the east in all cases except for ΣCBz in grazers and ΣHCH in loons. For the north vs south comparisons, the higher concentrations were found in the north except for ΣCHL in loons.

The geographical distribution of the data was graphed only for sea ducks which provided a data block with adequate samples in each geographical area as well as adequate numbers of detectable values for illustrative purposes. Graphical presentation of selected organochlorine classes shows that, although the median concentration values are quite low, the range of outliers is generally greater in the east (Figure 6). A comparison of north vs south (Figure 7) is less clear although there is a slightly wider range of outliers in the "north" plots. It is evident that the greatest ranges of residue concentrations are in the northeast (NE) and the southeast (SE) (Figure 8).

3.1.1.2 Trophic variation in organochlorine and PCB concentrations

Randomization tests showed some significant differences in organochlorine levels among trophic groups, most notably for DDE (Tables 6 and 7). The general trend was for increased organochlorine levels at higher trophic levels such as the sea ducks, fish-eating ducks (mergansers) and loons. The one interesting exception is for DDE in dabbling ducks from the west, and more specifically, the northwest. The result is probably driven by a single elevated DDE value in a pool of Northern Shoveler from the Yukon (Table 4). The distribution of organochlorine residue data for the different trophic groups is presented in Figure 9.

3.1.1.3 Species variation in organochlorine and PCB concentrations

A comparison of median values for organochlorine concentrations in pectoral muscle indicates relatively little variation among species in any one trophic group, for the most part, although some species show a wider range of levels than other species (Figures 10 to 15).

Common and other loon species (Pacific, Red-throated and Yellow-billed Loons) consistently showed higher median organochlorine levels than the mergansers (Figures 10f to 15f). Loons generally eat more and larger fish than do mergansers (Figure 2) and this may explain the slightly higher median values. Greater and Lesser Scaup generally had higher median levels and a wider range in values than the other bay ducks (Figures 10e-15e). This may be attributed to the differences in animal content in the diets of the various species of bay ducks, with Greater and Lesser Scaup having the highest amount of animal content in their diet, and Ring-necked Ducks, Canvasbacks and Redheads including increasingly higher proportions of plant matter in their diets (Bellrose, 1980).

There was little variation in median levels of organochlorines among species of dabbling ducks (Figures 10c-15c) and sea ducks (Figures 10d-15d). Some species of dabbling ducks showed a wider range in values than others and this seemed to vary with chemical compound. Therefore, the range of values probably reflects geographical differences rather than dietary differences. Of the sea ducks, the Oldsquaw consistently had the widest range of organochlorine values (Figures 10d to 15d). This may be linked to the fact that they are the only sea duck species surveyed to overwinter in relatively large numbers in the Great Lakes - St. Lawrence River system (Bellrose, 1980), an area known to be contaminated with organochlorines (Allen *et al.*, 1991; Kearney *et al.*, 1995). As well, since Oldsquaw probably dive deeper for food than any other duck (Bellrose, 1980), they may be utilizing a food base inaccessible to other species.

Organochlorine levels were generally very low in the upland gamebirds (Figures 10a to 15a), and geese and swans (Figures 10b to 15b). The range of values for some species probably reflect geographical differences rather than dietary differences. Of the upland gamebird species, American Woodcock had higher median levels of ΣPCB (Figure 10a), and to a much lesser extent, ΣCHL and dieldrin (Figures 12a & 13a), than the other species surveyed, whereas Common Snipe had higher median levels of DDE (Figure 11a). American Woodcock feed almost exclusively on worms (Ehrlich *et al.*, 1988) and may be picking up PCBs and other contaminants through the worms from the soil or directly from the soil during feeding. Common Snipe feed mainly on insects and aquatic invertebrates in wetland areas, whereas the ptarmigan and grouse species all feed on insects and various forms of terrestrial vegetation (Ehrlich *et al.*, 1988). Therefore, variation in organochlorine values among the upland gamebird species surveyed are likely due to dietary differences.

3.1.1.4 Seabirds and other species

Over the course of this study 21 pools of seabirds and other species were also collected, mainly at the request of local communities. The organochlorine and PCB congener data from these samples are presented in Tables 8 and 9. The muscle of the single American Coot collected from British Columbia had no detectable levels of any of the measured organochlorines. The highest levels of contaminants were found in Glaucous Gulls and Herring Gulls from the eastern Arctic. Levels of all measured organochlorines in gulls were comparable to those found in the mergansers and loons except for ΣCHL which was found at a level of 0.60 $mg \cdot kg^{-1}$ in a pool of Glaucous Gulls from Digges Island (Table 8). The alcids (murres, puffins and dovekies) had organochlorine levels similar to those found in the dabbling ducks (Tables 4, 8 and 9).

3.1.2 Eggs

Concentrations of all organochlorines (Table 10) and PCB congeners (Table 11) measured in eggs were less than 1 $mg \cdot kg^{-1}$ with the exception of ΣPCB in Herring Gull eggs from northern Québec and Red-breasted Merganser eggs from Great Slave Lake (Table 10).

The geographical distribution of eggs of Common Eider (12 pools from the eastern Arctic and 5 pools from the western Arctic) and King Eider (5 pools from the eastern Arctic) allowed us to compare organochlorine residues. However the patterns and levels of HCB, ΣHCH, ΣCHL, DDE, dieldrin and ΣPCB were virtually identical for both species and both regions. All residues were below 0.1 $mg \cdot kg^{-1}$.

3.2 PCDDs and PCDFs

3.2.1 Pectoral muscle

A subset of the waterfowl collected was analysed for PCDDs and PCDFs. Measurable concentrations of PCDDs were found in loons, mergansers, scoter, scaup, Oldsquaw and Ring-necked Ducks (Table 12). The highest levels of 2,3,7,8-TCDD and 1,2,3,7,8-PnDD were found in Lesser Scaup from Lac St-Pierre, Québec (9.7 $ng \cdot kg^{-1}$ and 12.5 $ng \cdot kg^{-1}$, respectively), and in Common Loons from Big Trout/Weagamow Lakes, northern Ontario (6.0 $ng \cdot kg^{-1}$ and 8.0 $ng \cdot kg^{-1}$, respectively). Measurable concentrations of PCDFs were found in loons, mergansers, scoter, scaup, eider and Mallards (Table 12). The highest levels of 2,3,7,8-TCDF were found in Surf Scoter from Westham Island, British Columbia (13 $ng \cdot kg^{-1}$) followed by Lesser Scaup from Lac St-Pierre, Québec (8.6 $ng \cdot kg^{-1}$), Common Mergansers from the Miramichi River area, New Brunswick (6.0 $ng \cdot kg^{-1}$), and Mallards from Westham Island, British Columbia (6.0 $ng \cdot kg^{-1}$). Except for the Mallards from Westham Island, no PCDDs or PCDFs were detected in any of the dabbling ducks or geese. The British Columbia data from this study as well as additional dioxin/furan data from British Columbia are discussed in greater detail in Elliott and Martin (1998).

3.2.2 Eggs

Only three collections of eggs, all from Great Slave Lake, Northwest Territories, were analysed for PCDDs and PCDFs (Table 12). No PCDFs were detected in any of the eggs, whereas PCDDs were detected in eggs of Red-breasted Mergansers and Herring Gulls.

3.3 Toxaphene

Toxaphene was measured in 12 pools of pectoral muscle taken from waterfowl: Oldsquaw (USOX 69809), Barrow's Goldeneye (USOX 65315 & 66025), Bufflehead (USOX 69600), Northern Shoveler (USOX 69372), Mallards (USOX 64505, 65998, 66041 & 69886), Northern Pintail (USOX 69708), Lesser Scaup (USOX 66038), and White-winged Scoter (USOX 68856). One pool of Mallards was from Alberta (USOX 64505) and the rest of the birds were from the Yukon. The total toxaphene level found in pectoral muscle of the pool of Oldsquaw (from Teslin Lake) was 9.4 $mg \cdot kg^{-1}$. Both pools of Barrow's Goldeneye had 1.7 $mg \cdot kg^{-1}$ of total toxaphene. The pool of Northern Shoveler contained 1.5 $mg \cdot kg^{-1}$ and the pool of Bufflehead contained 0.80 $mg \cdot kg^{-1}$ of total toxaphene. The total toxaphene levels in the rest of the pools were estimated as less than 0.5 $mg \cdot kg^{-1}$.

3.4 Metals

3.4.1 Pectoral muscle

Levels of cadmium, mercury, selenium and arsenic in pectoral muscle were, for the most part, either low (< 1 $mg \cdot kg^{-1}$) or below detection limits (Tables 13 and 14). Unusually elevated levels of lead were measured in several pools of birds. Follow-up analyses of selected pools on an individual bird basis comparing right and left pectoral muscles confirmed that tiny fragments of metallic lead, visually undetectable, were embedded in localized areas of the tissue of certain birds (Scheuhammer *et al*., 1998) indicating that fragments of lead shot, rather than biologically incorporated lead, were causing the elevated lead levels.

Arsenic was detected in all pools of woodcock and snipe analysed, but levels were

low ($\leq$ 0.20 mg·kg^{-1}). Arsenic was detected in only 20-60% of pools of birds in other trophic categories, and was found mainly in birds from eastern Canada. The highest level of arsenic (3.3 mg·kg^{-1}) was measured in a single adult Mallard from Labrador (Table 14).

Cadmium was detected in less than 45% of all the pools of birds analysed, mainly in sea ducks and loons, as well as woodcock, snipe and terrestrial browsers (Table 13 and Figure 16). The highest level of cadmium was measured in King Eider from Hall Beach, Northwest Territories (0.71 mg·kg^{-1}).

Mercury was detected in over 50% of dabbling, bay and sea ducks analysed, and in all of the fish-eating ducks and loons analysed (Table 13). Mercury was not detected in any of the terrestrial browsers, in only 40% of the woodcock and snipe, and in less than 2% of the geese. The highest level of total mercury (1.9 mg·kg^{-1}) was measured in a single adult Red-throated Loon from the Northwest Territories (Table 14). The distribution of mercury across trophic levels is shown in Figure 17 and the distribution across species is shown in Figure 18. Many of the data presented in these figures are non-detects that have been set to one-half the detection limit for graphing.

Selenium was detected in over 75% of all the pools of birds measured including woodcock, snipe and terrestrial browsers (Table 13). The highest level of selenium measured (10.1 mg·kg^{-1}) was in a single adult Black Scoter from Québec (Table 14). Most of the sample pools measured, however, contained < 2 mg·kg^{-1} of selenium. The distribution of selenium across trophic levels is shown in Figure 17 and the distribution across species is shown in Figure 19. As with the mercury data, non-detects have been set to one-half the detection limit for graphing.

3.4.1.1 Seabirds and other species

The only metal found above the detection limit in muscle of the American Coot was selenium (0.38 $mg \cdot kg^{-1}$) (Table 15).

Arsenic was found in all of the seabird muscle samples at levels that were high relative to the waterfowl species sampled. The maximum found was 1.5 $mg \cdot kg^{-1}$ in a pool of Herring Gull from Inukjuaq, Québec (Table 15).

Cadmium was found at generally higher levels in the seabirds than in the waterfowl samples, although the highest level found (0.55 $mg \cdot kg^{-1}$ in a Thick-billed Murre pool from Digges Island) was less than the highest level found in waterfowl. Mercury was present at quantifiable levels in all but one of the seabird pools with a maximum level of 0.51 $mg \cdot kg^{-1}$ in Glaucous Gulls from Kuujjuarapik, Québec (Table 15).

Selenium was found in all of the seabird pectoral muscle samples at levels similar to those in the mergansers and loons but well below the levels found in sea ducks (Figure 19 and Table 15).

3.4.2 Eggs

Arsenic was detected in only 16 of 36 egg pools analysed and the highest level found was only 0.27 $mg \cdot kg^{-1}$ (Table 16). Cadmium was not detected in any of the eggs sampled and lead was detected in only one pool of Mew Gull eggs (0.01 $mg \cdot kg^{-1}$) from Teslin Lake, Yukon (Table 16). Mercury was detected in all but two of the 36 egg pools. The highest level of mercury (0.50 $mg \cdot kg^{-1}$) found was in a pool of Black Guillemot eggs from the Ungava region of northern Québec (Table 16). Selenium was detected in all of

33 egg pools analysed and the highest level was 1.0 $mg \cdot kg^{-1}$ (Table 16) in a pool of Common Eider eggs from Labrador.

3.4.3 Liver

Several large pools of dabbling ducks and sea ducks from the Arctic were repooled to separate out males, females and juveniles. The resulting liver pools were analysed for mercury and selenium. The results are presented in Table 17 and Figure 20. The highest levels of selenium (23.0 $mg \cdot kg^{-1}$) were found in a pool of Common Eider from Arviaq, Northwest Territories, and in a pool of Oldsquaw from Fort Good Hope, Northwest Territories. The highest level of mercury (3.8 $mg \cdot kg^{-1}$) was found in a pool of Surf Scoter from Fort Good Hope, Northwest Territories. Overall, eiders and scoters had the highest levels of selenium. Mercury levels did not show much variation across species except in the Green-winged Teal, Northern Pintail and White-winged Scoter which had quite low levels (Figure 20).

3.5 Cesium-137

^{137}Cs was measured in 238 pools of pectoral muscle from 1,894 birds representing 37 species. The majority of the pools of birds contained <3 $Bq \cdot kg^{-1}$ of ^{137}Cs (Table 14). Geese and swans generally contained the highest levels of ^{137}Cs measured in this study, but grouse and ptarmigan also had higher levels of ^{137}Cs. Marine species such as the alcids and kittiwakes had extremely low ^{137}Cs levels (Table 14) as did some of the more marine duck species such as Oldsquaw, the eiders and Black Scoter.

4.0 Discussion

4.1 Organochlorines and PCBs

The use of persistent chlorinated pesticides, such as DDT, dieldrin and HCB, was common in the 1970s, but is now prohibited in North America, and the use of PCBs is severely restricted (Barrie *et al*., 1992; Rodrigue *et al*., 1992; Rodrigue *et al*., 1994). Many of these chlorinated pesticides are, however, still used in Central and South America where many migratory birds spend the winter (Barrie *et al*., 1992). In addition, there is considerable evidence that long range transport of some of these pollutants from European and Asian sources provides major inputs of contaminants into the Arctic areas of Canada (Barrie *et al.*, 1997, de March *et al.*, 1998, Dietz *et al.*, 1998). There is also concern that waterfowl are exposed to organochlorine contaminants on the migratory staging areas and breeding grounds in Canada and the United States due to the widespread distribution and extreme persistence of some of the organochlorine residues, particularly DDT and its metabolites and PCBs (Smith *et al.,* 1985; Blus *et al*., 1987; Foley and Batcheller, 1988; Gebauer *et al*., 1992; Elliott *et al.*, 1994; Hebert *et al.*, 1994; Weseloh *et al*., 1994; Mullie *et al*., 1996; Bredlin *et al*., 1997).

4.1.1 Pectoral muscle

4.1.1.1 Organochlorine levels relative to other studies

There is some indication that birds run a higher risk of exposure to elevated levels of organochlorine compounds in the eastern flyways than in the western ones (see section 3.1.1 and Figures 6 to 8). There are numerous eastern North American staging and overwintering areas such as the Great Lakes-St. Lawrence River, maritime Canada, the U.S. eastern seaboard and the Gulf of Mexico that are known to be contaminated with

chemical residues (Kim *et al.*, 1984; Kim *et al.*, 1985; P. Lane and Associates, 1986, 1991; Cain, 1988; Allen *et al*., 1991). Several studies have shown that organic contaminants can accumulate in waterfowl introduced into contaminated sites in relatively short periods of time (Dobos *et al*., 1991; Rodrigue *et al*., 1992, 1994; Gebauer and Weseloh, 1993; Custer *et al.*, 1996;).

As part of the National Pesticide Monitoring Program, the U.S. Fish and Wildlife Service systematically measured organochlorine residues in wings of Mallards and Black Ducks shot by hunters throughout the continental United States. These surveys were conducted every two to three years from the mid-1960s to the mid-1980s (see Heath, 1969; Heath and Hill, 1974; White and Heath, 1976; White, 1979; Cain, 1981; Prouty and Bunck, 1986). The highest mean residue levels in the 1981-82 survey occurred in Black Ducks and Mallards from the Atlantic Flyway. This pattern was also evident in the 1984-85 wing survey data as was the continuing decline in concentrations of all residues (Schmidt and Bunck, 1995). The declines in residue levels have also been documented by Heinz *et al.* (1994) and Weseloh *et al.* (1995).

A similar pattern of organochlorine decline was found in this study relative to earlier work (Braune *et al.*, 1991). The concentrations of most organochlorine and PCB compounds measured in the pectoral muscle of the birds in this survey were relatively low (Tables 3, 8, 9 and 12). By contrast Smith *et al*. (1985) found that carcasses of wintering Lesser Scaup, Greater Scaup and Common Goldeneye collected from the Detroit River in 1981 had organochlorine levels about an order of magnitude greater than levels we found in fall birds of those same species from southern Ontario including the Detroit River/Lake St. Clair region.

Although ΣPCB and ΣDDT were most frequently detected, residues of other organochlorine compounds were also found, particularly in those species feeding at

higher trophic levels. Levels of ΣPCB found in spring collections of whole body Oldsquaw from Rankin Inlet in 1991-94 (Johnstone *et al*., 1996) (Table 18) were, for the most part, two to three orders of magnitude greater than residue levels measured in pectoral muscle of Oldsquaw from the Maritimes or the western Arctic in our study (Table 4). It should also be noted, however, that since residue concentrations may vary among tissues, comparisons between whole body and pectoral muscle levels should be treated with caution. Oldsquaw containing higher levels of residues in the current study were collected from sites which are all centrally situated around the Hudson Bay - Great Lakes area and those birds may belong to the population which is known to overwinter in the Great Lakes - St. Lawrence system.

A statewide collection of waterfowl was made in Michigan, mainly during the 1988, 1989 and 1990 hunting seasons (Michigan Department of Natural Resources, 1993). All samples were analysed for ΣPCB. The majority of the samples showed no detectable levels of these contaminants in the pectoral muscle. The highest ΣPCB levels found in samples of pectoral muscle (without skin) were as follows: Mallard - 0.27 $mg \cdot kg^{-1}$ ww, Lesser Scaup - 0.03 $mg \cdot kg^{-1}$ ww, Greater Scaup - 0.07 $mg \cdot kg^{-1}$ ww. ΣPCB levels in pectoral muscle of some scaup from Québec, Ontario, Manitoba, southern British Columbia, Yukon and the western Arctic (Table 4) came close to or exceeded levels found in the Michigan scaup (Table 18). All of the ΣPCB levels in Mallards from the present Canadian survey (Table 4) were much lower than the highest levels found in the Michigan Mallards (Table 18).

Levels of ΣPCB in goose and dabbling duck species from this study were all lower than those found in geese and dabblers from Lower New York by Kim *et al.* (1984) and Kim *et al*. (1985) (Table 18).

Levels of DDE found in spring collections of whole body Northern Pintail and

Oldsquaw from Rankin Inlet in 1991-94 (Johnstone *et al*., 1996) (Table 18) were at least twice as great and, for the most part, two to three orders of magnitude greater than residue levels measured in pectoral muscle of Northern Pintail and Oldsquaw from our study (Table 4). As mentioned earlier, however whole body versus pectoral muscle comparisons should be treated cautiously.

Levels of DDE in geese were generally lower than those found in geese from the Chicago area in 1987 (Amundson, 1988) except for some birds collected in Labrador, eastern James Bay, the eastern Arctic, northern and southwestern Ontario and Saskatchewan (Quill Lakes area).

The highest levels of DDE in pectoral muscle (without skin) in Mallards from the 1988 - 1990 Michigan study (Michigan Department of Natural Resources, 1993) were 0.07 and 0.02 $mg \cdot kg^{-1}$, respectively. Some Mallards from eastern James Bay, northern and southwestern Ontario, Saskatchewan, Yukon and southern British Columbia contained DDE levels similar to or greater than levels found in the Michigan Mallards. The highest DDE level found in Mallards from the Canadian survey was 0.65 $mg \cdot kg^{-1}$ in birds collected in the Dawson area of the Yukon (Tables 3 and 4).

Levels of dieldrin found in spring collections of whole body Northern Pintail and Oldsquaw from Rankin Inlet in 1991-94 (Johnstone *et al*., 1996) (Table 18) were generally much higher than residue levels measured in pectoral muscle of Northern Pintail and Oldsquaw from our study (Table 4). Oldsquaw collected from Lac Waswanipi (Matagami area), Niagara River area and several sites in the eastern Arctic (Baker Lake, Arviaq, Sanikiluaq) (Table 4) contained dieldrin concentrations that fell within the range of concentrations measured in the birds from Rankin Inlet (0.02-4.0 $mg \cdot kg^{-1}$ ww). As with PCBs, the Oldsquaw containing higher levels of dieldrin residues may belong to the population which is known to overwinter in the Great Lakes - St. Lawrence system

(Tables 4 and 18).

Canada Geese collected from sites in eastern James Bay, the eastern Arctic, northern and southwestern Ontario and the Fraser River delta in this survey contained dieldrin levels in the pectoral muscle (Table 4 - converted to lipid weight basis) that were, in some cases, up to an order of magnitude higher than in skin fat of Canada Geese collected from the Chicago area in 1987 (Amundson, 1988).

The Mallards collected in the 1988 - 1990 statewide collection of waterfowl in Michigan (Michigan Department of Natural Resources, 1993) were analysed for a range of chlorinated hydrocarbons. The majority of the samples showed no detectable levels of these contaminants in the pectoral muscle. The highest levels of dieldrin in pectoral muscle (without skin) in Mallards was 0.07 $mg \cdot kg^{-1}$. All of the dieldrin levels in Mallards from the Canadian survey were much lower than the highest levels found in the Michigan Mallards.

Median values of the chlorobenzenes and hexachlorocyclohexanes were very low for most of the groups in this study (Table 3) and the highest values found (0.12 $mg \cdot kg^{-1}$ in a pool of Common Loon from Digges Island, NWT) were much lower than those found in the Detroit River in 1981 by Smith *et al*. (1985).

Hebert *et al*. (1994) have shown that levels of PCDDs in Great Lakes Herring Gulls declined between 1981 to 1984 and (along with PCDFs) have remained relatively constant since then indicating that equilibrium has been reached in the bioaccessible compartments of the Great Lakes. Elliott and Martin (1998) found a decline in PCDD and PCDF levels in Grebes and sea ducks on the British Columbia coast from 1989 to 1993 due the introduction of changes in pulp mill bleaching processes and restrictions on the use of chlorophenols. Dioxins and furans were generally not detected or only detected at

trace levels in our study but only a small subset of pools was analysed for these compounds.

The mirex levels found in this study (Table 4) are generally very low and comparable to levels found in other studies (Table 18). However, the highest levels found in our study (in pools of Oldsquaw, Red-breasted Merganser and Common Loon which could have spent time on the Great Lakes) are higher than those found in other studies (Table 18).

4.1.1.2 Effects of organochlorine residues

Muscle tissue is not often used to define toxic effects thresholds and most studies do not report the tissue levels associated with dietary concentrations or observed effects, but we have summarized the available literature on residue levels in muscle and other tissues associated with observed toxic effects (Table 19). It should be noted that residue levels in body tissues may not always have reached a steady-state in experimental dosing studies. Therefore, muscle residue levels may not necessarily provide an accurate measure with which to associate the described symptoms in nature.

In a dosing study on pheasants, Dahlgren *et al*. (1972) found a mean concentration of 140 $mg \cdot kg^{-1}$ ww of PCBs (as Aroclor 1254) in the muscle of pheasants that died. In a dosing study on chickens, Platenow *et al*. (1973) found a mean concentration of 120 $mg \cdot kg^{-1}$ ww of PCBs (as Aroclor 1254) in the muscle of cockerels that died (Table 19). In a chronic dietary exposure study on Ringed Turledoves, Peakall and Peakall (1973) attributed increased embryonic mortality to decreased parental attentiveness in birds with muscle PCB (Aroclor 1254) levels of 8 $mg \cdot kg^{-1}$ ww (Table 19). Induction of hepatic cytochrome P450-associated monooxygenases occurred in sentinel Mallards containing ΣPCB concentrations of 1.7-3.5 $mg \cdot kg^{-1}$ ww in pectoral muscle (Custer *et al*., 1996) although these birds also had low levels of chlordanes. Most of the muscle pools in the

present survey contained ΣPCB levels that were less than 1 mg·kg^{-1} ww (Table 4). The exceptions to this occurred for several merganser, loon and Oldsquaw pools from eastern Canada which contained ΣPCB concentrations of 1.0-2.4 mg·kg^{-1} ww in pectoral muscle. These levels are still much lower than most of those associated with adverse effects in other studies. As Schwartz and Stalling (1991) have pointed out, however, caution must be exercised when associating ΣPCB concentrations as measured in wild birds with toxicity effects observed in laboratory Aroclor feeding studies, because Aroclor mixtures change substantially in the environment and through the food chain.

In a study of dietary exposure of dieldrin on survival and growth in Mallard ducklings, Nebeker *et al*. (1992) found significant growth impairment and some mortality starting at muscle tissue levels of 5 mg·kg^{-1} (Table 19). The maximum value of dieldrin found in the current study was 0.17 mg·kg^{-1} in a pool of Common Mergansers (Tables 3 and 4).

Hyde *et al*. (1973) found no apparent effects on adults or hatching success associated with adult pectoral muscle tissue levels of mirex as high as 37 mg·kg^{-1} although there was reduced duckling survival at this level (Table 19). These levels far exceed anything found in the current study.

Adult Japanese quail dosed with HCB exhibited hyperactivity and increased porphyrin excretion when muscle HCB levels were 6 mg·kg^{-1} in females and 7 mg·kg^{-1} in males (Fletcher, 1972, cited in Wiemeyer, 1996). The health and general condition of chickens exposed to dietary HCB were not affected when muscle HCB levels were 12 mg·kg^{-1} (Avrahami and Steele, 1972, cited in Wiemeyer, 1996). All the muscle pools analysed in this study contained HCB residues that were well below 1 mg·kg^{-1} (Table 3 and 4).

The frequency of occurrence of toxaphene in the tissues and eggs of birds is low (Wiemeyer, 1996). Mehrle *et al*. (1979) found that duckling growth was depressed and spinal development impaired in Black Ducks when carcasses contained about 40 $mg \cdot kg^{-1}$ of toxaphene (Table 19). Haseltine *et al*. (1980) found no survival or reproductive effects associated with 0.5 $mg \cdot kg^{-1}$ carcass residue levels in a study of Black Ducks. We did not measure levels of toxaphene in waterfowl carcasses, but the highest total toxaphene level found in pectoral muscle was 9.4 $mg \cdot kg^{-1}$ ww in Oldsquaw from the Yukon and 7 of the 12 analyses were <0.5 $mg \cdot kg^{-1}$.

4.1.2 Eggs

Organochlorine and PCB levels measured in eggs collected in this survey were generally quite low and unlikely to cause toxic effects. With the exception of ΣPCB, all of the egg pools analysed in this survey contained organochlorine residue levels well below 1 $mg \cdot kg^{-1}$ (Table 10). Table 18 summarizes residue levels in tissues reported in the literature. Table 19 summarizes effects associated with tissue residue levels.

Hatching success was not affected in dietary PCB exposure studies on Mallards where PCB levels in the eggs were measured at 23 $mg \cdot kg^{-1}$ ww (as Aroclor 1254) (Custer and Heinz, 1980) (Table 19) and at 105 $mg \cdot kg^{-1}$ ww (as Aroclor 1242) (Haseltine and Prouty, 1980) (Table 19). Hoffman *et al*. (1996) have concluded from the literature that decreased hatching success is associated with total PCB concentrations of 1-5 $mg \cdot kg^{-1}$ ww in eggs of chickens and 8-25 $mg \cdot kg^{-1}$ ww in eggs of terns, cormorants, doves and eagles. Eisler (1986) concluded that total PCB levels in excess of 16 $mg \cdot kg^{-1}$ ww were frequently associated with PCB poisoning in birds. The only egg pools that exceeded 1 $mg \cdot kg^{-1}$ ΣPCB were Herring Gulls from northern Québec (2.7 $mg \cdot kg^{-1}$ ww) and Red-breasted

Mergansers from Great Slave Lake (1.6 $mg \cdot kg^{-1}$ ww). As Schwartz and Stalling (1991) have pointed out, however, caution must be exercised when associating ΣPCB concentrations as measured in wild birds with toxicity effects observed in laboratory Aroclor feeding studies, because Aroclor mixtures change substantially in the environment and through the food chain. The ΣPCB levels measured in eggs in this survey are unlikely to be indicative of toxic effects.

The inverse relationship between DDE residues in eggs and eggshell thickness and associated lowered reproductive success has been well documented (see review by Blus, 1996). Eggs of Black Ducks with DDE residues of 46 $mg \cdot kg^{-1}$ ww showed 18% shell thinning (Longcore *et al*., 1971). Jehl (1973), Blus *et al*. (1974), King *et al*., (1977) (cited in Wiemeyer, 1996), and Blus *et al*. (1979) have all documented a relationship between DDE levels of 1-3 $mg \cdot kg^{-1}$ or more and eggshell thinning and depressed productivity in the Brown Pelican (*Pelecanus occidentalis*), which appears to be the most sensitive species. Fox *et al*. (1980) found a small but significant inverse correlation between DDE levels in Common Loon eggs (mean of 5.8 $mg \cdot kg^{-1}$) and eggshell thickness together with structural abnormalities in the shells (Table 19). All of the egg pools analysed in the current survey contained DDE concentrations less than 1 $mg \cdot kg^{-1}$ (Table 10).

Hyde *et al*. (1973) showed that Mallard duckling survival was reduced when eggs contained 277 $mg \cdot kg^{-1}$ ww of mirex although there was no apparent effect on egg hatchability at this level (Table 19). These levels far exceed anything found in the current study.

No adverse effects on hatchability of eggs were observed when the HCB level was about 105 $mg \cdot kg^{-1}$ in chicken eggs (Avrahami and Steele, 1972, cited in Wiemeyer, 1996) (Table 19) or about 35 $mg \cdot kg^{-1}$ ww in Japanese quail eggs (Fletcher, 1972, cited in

Wiemeyer, 1996). The reproduction of Canada Geese appeared normal when maximum HCB residues in eggs were 2.97 $mg \cdot kg^{-1}$ (Blus *et al*., 1984). All egg HCB levels in the current study were less than 0.10 $mg \cdot kg^{-1}$ (Table 10).

In a study on the effects of heptachlor epoxide on nesting success of Canada Geese, Blus *et al*. (1984) showed that nesting success declined when eggs contained greater than 10 $mg \cdot kg^{-1}$ heptachlor epoxide. Hatchability of eggs from Ring-necked Pheasants (*Phasianus colchicus*) exposed to dietary γ-HCH was unaffected when eggs contained γ-HCH residues of about 10 $mg \cdot kg^{-1}$ (Ash and Taylor, 1964, cited in Wiemeyer, 1996). When lindane (γ-HCH) was substituted for heptachlor as a seed treatment in the Columbia Basin, Oregon, no HCH isomers were found in eggs or tissues of Canada Geese nesting nearby (Blus *et al*., 1984). All of the eggs in the current study had residue levels of heptachlor epoxide and HCHs well below 0.10 $mg \cdot kg^{-1}$ (Table 10).

In a study of dioxins in Wood Duck eggs, White and Seginak (1994) found that for Wood Ducks, which they suggest are more sensitive to dioxins and furans than other avian species, 14-35 $\mu g \cdot kg^{-1}$ ww of 2,3,7,8-TCDD in eggs was indicative of decreased productivity. TCDD was detected at trace levels in two of the three egg pools analysed in the current survey in Herring Gulls and Red-breasted Mergansers.

4.2 Metals

4.2.1 Pectoral muscle

Pectoral muscle is not generally considered to be a major target tissue for bioaccumulation of heavy metals. Rather, metal residues tend to accumulate in liver and kidney. Therefore care must be taken when trying to interpret the significance of muscle tissue levels of metals (Scheuhammer *et al.*, 1998).

4.2.1.1 Metal levels relative to other studies

Elevated natural levels (particularly in water) of arsenic occur in many parts of North and South America where migrating birds could be exposed: Chile (Antofagasta), Argentina (Cordoba), Mexico (Lagunera), United States (California, Oregon, Ohio, Alaska) and Canada (Nova Scotia, New Brunswick, Alberta) (Azcue, 1995).

Arsenic levels were generally low in this study (Table 14) and those that were elevated (above 1.0 $mg \cdot kg^{-1}$) were associated with areas where high natural levels could be expected due to regional geology (NRCC, 1978; Azcue, 1995). The levels were comparable to those found in other studies (Table 20).

Fish with Hg concentrations that exceeded an estimated threshold of 0.4 $mg \cdot kg^{-1}$ for reproductive impairment in loons and which are small enough for loons to eat occur in up to 30% of Ontario lakes (Scheuhammer and Blancher, 1994). Barr (1986) found that mercury levels in muscle of Common Loons averaged 1.2 $mg \cdot kg^{-1}$ under normal conditions and up to 4.6 $mg \cdot kg^{-1}$ in lakes near a mercury pollution source (Table 20). Loons and mergansers had the highest mercury levels of all species in this study (Table 13).They were, however comparable to levels found in these species in other studies (Table 20).

A statewide collection of waterfowl was made in Michigan during 1988, 1989 and 1990 (Michigan Department of Natural Resources, 1993). Mallards were analysed for mercury residues. The highest level found in pectoral muscle (without skin) was 0.048 $mg \cdot kg^{-1}$ ww. Many of the Mallards from across Canada had similar or higher mercury levels (Table 14). The highest mercury level (0.34 $mg \cdot kg^{-1}$) found in pectoral muscle of Mallards in the present Canadian survey was from a collection of birds from British Columbia.

4.2.1.2 Effects of metal residues

Heinz *et al.* (1987) showed that pectoral muscle levels of 1.3-1.4 $mg \cdot kg^{-1}$ selenium (associated with diets of 100 ppm) were associated with embryonic deformities, reduced hatching and adult mortality (Table 21). While levels in this study often exceeded those levels it should be remembered that metals levels in body tissues may not always have reached a steady-state in experimental dosing studies, therefore, muscle levels may not necessarily provide an accurate measure with which to associate the described symptoms in nature.

Only in the mergansers and loons did maximum mercury levels (1.5-1.9 $mg \cdot kg^{-1}$) match or exceed levels associated with behavioural changes in nesting Mallards by Heinz (1979) but did not equal the levels (>3 $mg \cdot kg^{-1}$) that Barr (1986) found to be associated with reduced reproductive success in loons (Table 21).

4.2.2 Eggs

Neither lead nor cadmium were detected in any of the eggs analysed in this study. Since neither lead nor cadmium are readily transferred to eggs (Burger and Gochfield, 1991), this was the expected result.

All mercury and selenium levels found in this study were well below egg levels associated with adverse effects in experimental studies (Table 21).

4.2.3 Liver

Mercury and selenium levels in livers of dabbling ducks and sea ducks from this study (Table 17) were in line with levels found in wild birds by other workers (Table 20).

For both mercury and selenium, the highest values found were in the range of tissue levels associated with toxic effects in some experimental dosing studies (Table 21) but, as mentioned earlier, it is often difficult to transfer information gained from laboratory studies to field situations or to infer toxicological effects from tissue levels (Scheuhammer *et al.,* 1998).

4.3 Cesium-137

Analysis of the ^{137}Cs data for species with adequate pool numbers across a wide geographical distribution showed a definite grouping of the data by longitude, with higher ^{137}Cs levels in the birds from the east (Braune and Marshall, in prep). This was particularly striking for both Canada and Lesser Snow Geese. The east-west split in the data appears to coincide approximately with the western boundary of the Canadian Shield. A detailed discussion of these data appears in Braune and Marshall (in prep.).

4.4 Critical levels for avian and wildlife health effects

Many different attempts have been made to determine critical levels of contaminants in various matrices. These can take the form of guidelines, objectives or regulations used to trigger some form of action. They may also be scientific assessments of levels beyond which harm is likely to occur to the contaminated matrix or to those who consume it. We have summarized some of these levels for organochlorines and metals relevant to Canadian waterfowl and gamebirds in Tables 22 and 23. In the following discussion we refer to individual or pooled samples from this study which exceed some of these threshold levels (Table 24). It is important to remember that these constitute only a very few samples out of a total of 834 pools of birds from across the country. The vast majority of these samples had very low or undetectable levels of the contaminants studied.

4.4.1 Organochlorines

The lowest critical levels for PCBs are the IJC guideline of 0.100 $mg \cdot kg^{-1}$ and the USEPA guideline of 0.160 $mg \cdot kg^{-1}$ for fish (Table 22). Beyond these levels the agencies involved believe there may be risk to wildlife which consume the fish. These guidelines are for fish which are eaten in large quantities by many other species. Birds, particularly the fish-eating birds, are a much smaller part of other animals' diets so the fish guidelines can be taken as cautionary for birds. Most of the birds from this study were well below these levels and all but one pool of Common Mergansers (2.4 $mg \cdot kg^{-1}$) were below the human consumption guideline for PCBs in commercial fish (2.0 $mg \cdot kg^{-1}$).

Nebeker *et al*. (1992) determined a 24-day no-observed-adverse-effect level (NOAEL) and a lowest-observed-adverse-effect level (LOAEL) of $<1\ mg \cdot kg^{-1}$ ww for dieldrin in muscle (Table 22). The maximum value of dieldrin found in this study was 0.17 $mg \cdot kg^{-1}$ in a pool of Common Mergansers (Tables 3 and 4).

All DDE/DDT residue levels found in this study were less than half and most were at least an order of magnitude less than published human consumption guidelines for birds (Table 22). There were, however, several pools, particularly among the mergansers and loons which exceeded the IJC guideline of 1 $mg \cdot kg^{-1}$ for hazard to fish-eating wildlife (IJC, 1987).

Fox *et al*. (1980) calculated that egg levels of 47 $mg \cdot kg^{-1}$ DDE would be required to produce a 20% thinning in Common Loon eggshells, characteristic of declining populations (Table 22). None of the eggs, of any species, in this study exceeded 1.0 $mg \cdot kg^{-1}$.

There were only two pools, one of Lesser Scaup from the St. Lawrence and one of Surf Scoter from the Fraser River delta that exceeded the Michigan Dept. of Natural Resources dioxin trigger level (10 ng·kg $^{-1}$) for setting advisories on consumption of fish (Michigan Department of Natural Resources, 1993) (Table 22). Both of these pools are associated with areas of particular concern for organochlorine pollution.

The Michigan Dept. of Natural Resources trigger level for fish consumption advisories for mirex (Michigan Department of Natural Resources, 1993) (Table 22) was only exceeded by four pools from this study: Oldsquaw from Arviaq, Northwest Territories (0.22 mg·kg $^{-1}$), Red-breasted Mergansers from Kangiqsualuujjuaq, Québec (0.27 mg·kg^{-1}), and Common Loons from Weagamow Lake (0.13 mg·kg $^{-1}$) and Big Trout Lake (0.11 mg·kg $^{-1}$) in northwestern Ontario.

4.4.2 Metals and cesium-137

Only 31 samples from this study exceeded the normal background levels of arsenic in animal tissues in Canada of <0.3 mg·kg $^{-1}$ fresh weight (NRCC, 1978)(Table 23). We could not find any published critical threshold levels for arsenic in muscles of wild birds against which to compare our data.

The highest cadmium level found in either pectoral muscle or egg in this study (0.7 mg·kg $^{-1}$) was well below the 2 mg·kg $^{-1}$ which Eisler (1985) considered to be evidence of cadmium contamination in wild birds.

Although Eisler (1987) considers tissue mercury levels greater than 1.0 mg·kg $^{-1}$ to be presumptive evidence of a mercury problem (Table 23), Thompson (1996) suggests that

levels in excess of 20-30 $mg \cdot kg^{-1}$ ww may be indicative of mercury poisoning (Table 23). Although 23 (19%) of dabbling duck and sea duck liver pools from the western Arctic and the Yukon had mercury levels (Table 17) that exceeded 1.0 $mg \cdot kg^{-1}$ the highest level found was 3.8 $mg \cdot kg^{-1}$ in a liver pool of Surf Scoter from Fort Good Hope, NT, which is an order of magnitude lower than the critical level range suggested by Thompson (1996).

Based on studies mainly with Mallards, Heinz (1996) suggests that liver selenium levels of 3 $mg \cdot kg^{-1}$ are associated with adverse reproductive effects in laying females (Table 23). However, he also cautions that measurements of selenium in liver outside the breeding season may not be useful in predicting reproductive effects. Most of the birds in this survey were collected outside of the breeding season. Heinz (1996) also states that liver levels greater than 10 $mg \cdot kg^{-1}$ in young or adult birds may be indicative of important sublethal effects and concentrations in excess of 20 $mg \cdot kg^{-1}$ may affect survival (Table 23). Twenty-two liver pools (18%) from this study had selenium levels that were higher than 10 $mg \cdot kg^{-1}$ and 5 pools were greater than 20 $mg \cdot kg^{-1}$ (Table 14, 17, 23, 24). All of these pools were sea ducks. Sea ducks and other marine birds often have elevated levels of selenium possibly because of the nature of selenium cycling in the marine environment and they appear to have evolved mechanisms to deal with these elevated levels (Eisler, 1987).

The highest level of ^{137}Cs activity found in this study was 46 $Bq \cdot kg^{-1}$; the radiation exposure from this burden of radiocesium is about one-third that which the animals receive from the naturally occurring potassium in their bodies. Most of the levels of ^{137}Cs we found were <3 $Bq \cdot kg^{-1}$, on the same order as those in domestic livestock (Health and Welfare Canada, 1987).

4.5 Human consumption guidelines

Canada does not issue consumption guidelines for wild ducks for any contaminants on a regular basis. The discovery that waterfowl collected throughout New York State were contaminated with high levels of PCBs and pesticide residues (Baker *et al*., 1976; Kim *et al*., 1984; Kim *et al*., 1985) prompted the development of waterfowl consumption advisories which have been issued annually to New York hunters since 1981. In 1987, the state of Wisconsin began to issue consumption advisories warning people to avoid eating certain waterfowl species from sections of several Lake Michigan tributaries. Michigan, too, has established "trigger levels" for establishing wildlife consumption advisories (Michigan Department of Natural Resources, 1993).

In Canada, consumption advisories and hunting season closures for waterfowl and game birds have been issued on a site- and species-specific basis. In 1969, the fall hunting season on pheasants and partridges was closed in Alberta due to mercury contamination. In 1970, the hunting season for woodcock was closed in New Brunswick because of high levels of DDT found in pectoral muscle of birds (Pearce and Baird, 1970). In 1971, an advisory was issued to hunters in northwestern Ontario that they should avoid eating Common Goldeneye, Blue-winged Teal and Mallards from the Wabigoon River system west of Dryden due to high mercury levels found in the birds. In 1990, an advisory was issued recommending that the livers of Common Mergansers, Surf Scoters and Western Grebes from the Port Alberni area of British Columbia not be eaten due to high levels of dioxins in the livers of those species (Whitehead *et al.*, 1990; Elliott and Martin, 1998).

As a result of the residue data collected for waterfowl and other game birds since 1987, which are presented in this report, several consumption advisories were issued. Based on residue data from this study as well as some earlier Ontario data from 1985-86 (Braune *et al*., 1991), Health Canada advised that contaminant levels found in samples of

pectoral muscle of ducks, geese and other gamebirds analysed do not pose a health hazard to human consumers. Some concerns were raised with respect to elevated levels of total chlordane, PCBs and mirex found in a single composite of two individual pectoral muscles of Glaucous Gulls from Kuujjuarapik, Québec, and levels of lead in pectoral muscle of one Mallard from Labrador, and one pool of American Wigeon and one pool of Northern Shoveler from Alberta. Health Canada suggested, however, that it is unlikely that the Glaucous Gulls are consumed to a degree that would pose a hazard to the consumer and, as already mentioned earlier, the elevated lead levels are likely due to undetected fragments of lead shot left embedded in the flesh after hunting and cleaning.

Health Canada also suggested that *"... there are a number of precautionary measures that could be taken to minimize exposure to chemical contaminants resulting from consumption of waterfowl. For example, cleaning and inspection of waterfowl as soon as possible after they are shot and careful removal of any visible lead shot would help to minimize exposure to lead. In addition, the use of cooking techniques which would eliminate as much fat as possible, particularly in the case of fish-eating waterfowl, would help to minimize exposure to organochlorine contaminants."*

Waterfowl consumption advisories are generally based on levels of contaminants in raw, uncooked tissue. Cooking studies on poultry (Liska *et al*., 1967; Ritchey *et al*., 1967, 1969), ducks (Smoczynski and Amarowicz, 1986) and geese (Amundson, 1988), however, all conclude that cooking reduces organochlorine residue concentrations, mainly through leaching of the fat into the drippings. It has also been shown that removal of skin and fat from Mallards reduced PCB levels by 60-90% (Amundson, 1987). These studies have led to the general advice given to hunters by the New York State Department of Health and the Wisconsin Division of Health, that all skin and visible fat should be removed prior to cooking, and drippings or stuffing discarded because they may contain organochlorine residues.

All of the 237 pools tested for ^{137}Cs activity in the current study contained less than 46 $Bq \cdot kg^{-1}$, most less than 3 $Bq \cdot kg^{-1}$, and so fall well below the allowable limit of 1000 $Bq \cdot kg^{-1}$ in commercially traded food set by the Codex Alimentarius Commission of the Food and Agriculture and World Health Organizations (FAO/WHO, 1995).

5.0 Conclusion

In this study, over 800 pools of waterfowl, gamebird and seabird muscle, egg and liver tissues were analysed for a range of organochlorine and metal contaminants. These contaminants were either not detected or found at very low levels in the majority of these samples. The highest levels of the various contaminants were found in those birds feeding at the higher trophic levels such as mergansers, loons and gulls. Of 834 pools of waterfowl muscle analysed for organochlorines only 14 pools (1.7 %) had contaminant levels that could pose a potential threat to the health of either the birds or to human or animal consumers of the birds (Table 24). Based on its evaluation of residue levels in pectoral muscle, Health Canada has recommended that these waterfowl are safe to eat and precautionary food preparation methods will limit contaminant exposure even further.

Literature Cited

Allen, R. J., A. J. Ball, V. W. Cairns, G. A. Fox, A. P. Gilman, D. B. Peakall, D. A. Piekarz, J. C. Van Oostdam, D. C. Villeneuve and D. T Williams. 1991. Toxic Chemicals in the Great Lakes and Associated Effects. Environment Canada, Department of Fisheries and Oceans, Health and Welfare Canada, Ottawa. 755p.

Alton Mackey, M. G. and R. D. Moore Orr. 1988. The seasonal nutrient density of country food harvested in Makkovik, Labrador. Arctic 41(2):105-108.

Amundson, D. S. 1988. Organochlorine pesticides and PCBs in Edible tissues of Giant Canada Geese from the Chicago area. M.Sc. Thesis, University of Illinois, Chicago IL. 98p.

Amundson, T. E. 1987. Environmental Contaminant Monitoring of Wisconsin Wild Game 1985-1986. Bureau of Wildlife Management, Wisconsin Department of Natural Resources, Madison WI. 72p.

Anderson, D. W. and J. J. Hickey. 1976. Dynamics of storage of organochlorine pollutants in Herring Gulls. Environ. Pollut. 10:183-200.

Annett, C. S., F. M. D'Itri, J. R. Ford and H. H. Prince. 1975. Mercury in fish and waterfowl from Ball Lake, Ontario. J. Environ. Quality 4:219-222.

Ash, J. S. and A. Taylor. 1964. Further trials on the effects of gamma BHC seed dressing on breeding pheasants. Game Res. Assoc. Ann. Rep. 4:14-20. (Cited in Wiemeyer, 1996)

Association of Official Analytical Chemists. 1990. Official Methods of Analysis. Volume One: Agricultural Chemicals; Contaminants; Drugs. 15th ed. *Edited by:* K. Helrich. Association of Official Analytical Chemists, Inc. Arlington, Virginia. 684p.

Avrahami, M. and R. T. Steele. 1972. Hexachlorobenzene. II. Residues in laying pullets fed HCB in their diet and the effects on egg production, egg hatchability, and on chickens. N.Z.J. Agric. Res. 15:482-488. (Cited in Wiemeyer, 1996)

Azcue, J. M. 1995. Environmental significance of elevated natural levels of arsenic. Environ. Rev. 3:212-221.

Baker, F. D., C. F. Tumasonis, W. B. Stone and B. Bush. 1976. Levels of PCB and trace

metals in waterfowl in New York State. NY Fish and Game J. 23:82-91.

Ballschmiter, K. and M. Zell. 1980. Analysis of polychlorinated biphenyls (PCBs) by glass capillary gas chromatography. Composition of technical Aroclor- and Clophen-PCB mixtures. Fresenius Zeitschrift für Analytische Chemie 302:20-31.

Barr, J. F. 1986. Population dynamics of the Common Loon (*Gavia immer*) associated with mercury contaminated waters in northwestern Ontario. CWS Occas. Paper 56. 25p.

Barrie, L. A., D. Gregor, B. Hargrave, R. Lake, D. Muir, R. Shearer, B. Tracey and T. Bidleman. 1992. Arctic contaminants: sources, occurrence and pathways. Sci. Tot. Environ. 122:1-74.

Barrie, L., R. Macdonald, T,. Bidleman, M. Diamond, D. Gregor, R. Semkin, W. Strachan, M. Alaee, S. Backus, M. Bewers, C. Gobeil, C. Halsall, J. Hoff, A. Li, L. Lockhart, D. Mackay, D. Muir, J. Pudykiewicz, K. Reimer, J. Smith, G. Stern, W. Schroeder, R. Wagemann, F. Wania, M. Yunker. 1997. Chapter 2. Sources, Occurrence and Pathways. In. Jensen, J., K. Adare and R. Shearer. (eds). 1997. Canadian Arctic Contaminants Assessment Report. Indian and Northern Affairs Canada, Ottawa. pp 25-182.

Baskett, T. S. 1975. Residues in fish, wildlife and estuaries. Pestic. Monit. J. 9:67-78.

Bellrose, F. C. 1980. Ducks, Geese and Swans of North America. 3rd ed. Stackpole Books, Harrisburg, PA. 540p.

Berkes, F. 1982. Waterfowl management and northern native peoples with reference to Cree hunters of James bay. Musk-Ox 30: 23-35.

Berti, P. R., O. Receveur, H. M. Chan and H. V. Kuhnlein. 1998. Dietary exposure to chemical contaminants from traditional food among adult Dene/Métis in the western Northwest Territories, Canada. Environm. Res., Sect. A 76:131-142.

Blus, L. J. 1996. DDT, DDD, and DDE in Birds. In: Environmental Contaminants in Wildlife: Interpreting Tissue Concentrations. *Edited by*: W.N. Beyer, G.H. Heinz and A.W. Redmon-Norwood. SETAC Special Publication Series, CRC Press Inc, Boca Raton, USA. pp. 49-71.

Blus, L. J., A. A. Belisle and R. M. Prouty. 1974. Relations of the Brown Pelican to certain environmental pollutants. Pestic. Monit. J. 7:181-194.

Blus, L. J., C. J. Henny, D. J. Lenhart and T. E. Kaiser. 1984. Effects of heptachlor- and lindane-treated seed on Canada Geese. J. Wildl. Manage. 48:1097-1111.

Blus, L. J., C. J. Henny, C. J. Stafford and R. A. Grove. 1987. Persistence of DDT and metabolites in wildlife from Washington state orchards. Arch. Environ. Contam. Toxicol. 16:467-476.

Blus, L. J., T. J. Lamont and B. S. Neely, Jr. 1979. Effects of organo-chlorine residues on eggshell thickness, reproduction, and population status of Brown Pelicans (*Pelecanus occidentalis*) in South Carolina and Florida, 1969-76. Pestic. Monit. J. 12:172-184.

Boyd, H. 1985. The Reported Kill of Ducks and Geese in Canada and the USA, 1974-82. Canadian Wildlife Service Occasional Paper Number 55. 20p.

Braune, B. M., and H. Marshall. Radiocesium in game birds harvested in Canada. In Prep.

Braune, B. M., M. P. Wong, J.-C. Belles-Isles and W. K. Marshall. 1991. Chemical Residues in Canadian Game Birds. Canadian Wildlife Service Technical Report Series No. 124, Ottawa. 375p.

Bredlin, K. A., N. M. Burgess, D. B. McCorquedale and D. L. Harris. 1997. Volunteer Surveys of Waterbirds of the Sydney, Nova Scotia Area, Including the Sydney Tar Ponds, 1993-1996. Canadian Wildlife Service Technical Report Series No. 273. Ottawa. 40p.

Bromley, R. G. 1996. Characteristics and management implications of the spring waterfowl hunt in the Western Canadian Arctic, Northwest Territories. Arctic 49:70-85.

Burger, J. and M. Gochfield. 1991. Cadmium and lead in Common Terns (Aves *Sterna hirundo*): relationships between parents and eggs. Environ. Monitor. Assess. 16:253-258.

Cain, B. W. 1981. Nationwide residues of organochlorine compounds in wings of adult Mallards and Black Ducks. Pestic. Monit. J. 15:128-134.

Cain, B. W. 1988. Wintering waterfowl habitat in Texas: shrinking and contaminated. In: Waterfowl in Winter. *Edited by*: M. W. Weller. University of Minnesota Press, Minneapolis. pp 583-596.

Cain, B. W., L. Sileo, J. C. Franson and J. Moore. 1983. Effects of dietary Cadmium on Mallard ducklings. Environ. Res. 32:286-297.

Camardese, M. B., D. J. Hoffman, L. J. LeCaptain and G. W. Pendleton. 1990. Effects of arsenate on growth and physiology in Mallard ducklings. Environ. Toxic. and Chem. 9:785-795.

Cameron, M. and I. M. Weis. 1993. Organochlorine contaminants in the country food diet of the Belcher Island Inuit, Northwest Territories, Canada. Arctic 46:42-48.

Canadian Wildlife Service. 1990. Distribution of organochlorines in breast muscle of waterfowl. Contaminants Project TOX-CE-3 Summary. CWS Unpublished Internal Report, Ottawa.

Collins, B. 1995. RERAND2: a program to perform rerandomization analysis for a one-way design. Documentation and User's Guide. Unpublished CWS Report. Environment Canada.

Custer, T. W. and G. H. Heinz. 1980. Reproductive success and nest attentiveness of Mallard ducks fed Aroclor 1254. Environ. Pollut. 21:313-318.

Custer, T. W., D. W. Sparks, S. A. Sobiech, R. K. Hines and M. J. Melancon. 1996. Organochlorine accumulation by sentinel Mallards at the Winston-Thomas sewage treatment plant, Bloomington, Indiana. Arch. Environ. Contam. Toxicol. 30:163-169.

Dahlgren, R. B., R. J. Bury, R. L. Linder and R. F. Reidinger, Jr. 1972. Residue levels and histopathology in Pheasants given polychlorinated biphenyls. J. Wildl. Manage. 36:524-533.

de March, B. G. E., C. A. De Wit and D. C. G. Muir. 1998. Chapter 6. Persistent Organic Pollutants. In: AMAP. AMAP Assessment Report: Arctic Pollution Issues. Arctic Monitoring and Assessment Programme (AMAP), Oslo, Norway. pp. 183-372.

Desai-Greenaway, P. and Price, I. M. 1976. Mercury in Canadian fish and wildlife used in the diets of native people. Toxic Chemicals Division, Canadian Wildlife Service Manuscript Report no. 35. 61p.

Dewailly, É., P. Ayote, C. Blanchard, S. Bruneau, S. Carrier and B. Holub. 1996. Health risk assessment and elaboration of public health advices concerning food contaminants in Nunavik: Final Report. Québec Centre for Public Health,

Beauport (Québec), Canada.

Dickson, K. and L. Métras. 1987. Migratory birds killed in Canada during the 1986 season. Canadian Wildlife Service Progress Notes Number 171. 42p.

Dietz, R., J. Pacyna and D. J. Thomas. 1998. Chapter 7. Heavy Metals. In: In: AMAP. AMAP Assessment Report: Arctic Pollution Issues. Arctic Monitoring and Assessment Programme (AMAP), Oslo, Norway. pp. 183-372.

Dobos, R. Z., D. S. Painter and A. Mudroch. 1991. Contaminants in wildlife utilizing confined disposal facilities. Int'l J. of Environ. and Pollut. 1:73-86.

Egede, I. 1995. Inuit food and Inuit health: contaminants in perspective. Avativut/Ilusivut Newsletter 2(1)December:1-3.

Ehrlich, P. R., D. S. Dobkin and D. Wheye. 1988. The Birder's Handbook: A field guide to the natural history of North American birds. Simon & Schuster Inc., New York. 785p.

Eisler, R. 1985. Cadmium Hazards to Fish, Wildlife and Invertebrates: A Synoptic Review. U.S. Fish Wildl. Serv. Biol. Rep. 85(1.2). 46p.

Eisler, R. 1986. Polychlorinated Biphenyl Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review. U.S. Fish Wildl. Serv. Biol. Rep. 85(1.7). 72p.

Eisler, R. 1987. Mercury Hazards to Fish, Wildlife and Invertebrates: A Synoptic Review. U.S. Fish Wildl. Serv. Biol. Rep. 85(1.10). 90p.

Elliott, J. E. and P. A. Martin. 1998. Chlorinated hydrocarbon contaminants in Grebes and sea ducks wintering on the coast of British Columbia, Canada: 1988-1993. Environ. Monitor. and Assess. 53:337-362.

Elliott, J. E., P. A. Martin, T. W. Arnold and P. H. Sinclair. 1994. Organochlorines and reproductive success of birds in orchard and non-orchard areas of central British Columbia, Canada. 1990-91. Arch. Environ. Contam. Toxicol. 26:435-443.

Fan, A. M., S. A. Book, R. R. Neutra and D. M. Epstein. 1988. Selenium and human health implications in California's San Joaquin Valley. J. Toxicol. Environ. Health 23:539-559.

FAO/WHO. 1995. Codex Alimentarius. General requirements, Section 6.2, Guideline

levels for radionuclides in foods following accidental nuclear contamination for use in international trade. Joint FAO/WHO Food Standards Programme, Rome.

Fimreite, N., W. N. Holsworth, J. A. Keith, P. A. Pearce and I. M. Gruchy. 1971. Mercury in fish and fish-eating birds near sites of industrial contamination in Canada. Canadian Field-Naturalist 85:211-220.

Fimreite, N. 1974. Mercury contamination of aquatic birds in northwestern Ontario. J. Wildl. Managem. 38:120-131.

Finley, M. T. and R. C. Stendell. 1978. Survival and reproductive success of Black Ducks fed methylmercury. Environ. Pollut. 16:51-64.

Fletcher, M. R. 1972. Effects of hexachlorobenzene on Japanese Quail. Proc. West. Assoc. State Game Fish Comm. 52:374-383. (Cited in Wiemeyer, 1996)

Foley, R. E. and G. R. Batcheller. 1988. Organochlorine contaminants in Common Goldeneye wintering on the Niagara River. J. Wildl. Manage. 52:441-445.

Fox, G., K. S. Yonge and S. G. Sealy. 1981. Breeding performance, pollutant burden and eggshell thinning in Common Loons (*Gavia immer*) nesting on a boreal forest lake. Ornis Scand. 11:243-248.

Franson, J. C., P. S. Koehl, D. V. Dirksen, T. C. Rothe, C. M. Bunck and J. F. Moore. 1995. Heavy metals in sea ducks and mussels from Misty Fjords National Monument in southeast Alaska. Environ. Monitor. and Assess. 36:149 - 167.

Friend, M. 1985. Interpretation of criteria commonly used to determine lead poisoning problem areas. USFWS Fish and Wildlife Leaflet 2. 4p.

Gamble, R. L. 1987. Native Harvest of Wildlife in Keewatin Region, Northwest Territories for the Period October 1985 to March 1986 and a Summary for the Entire Period of the Harvest Study from October 1981 to March 1986. Canadian Data Report of Fisheries and Aquatic Sciences No. 688. 85 p.

Gebauer, M. B., R. Z. Dobos and D. V. Weseloh. 1992. Waterbird surveys at Hamilton Harbour, Lake Ontario, 1985-1988. J. Great Lakes Res. 18:420-439.

Gebauer, M. B. and D. V. Weseloh. 1993. Accumulation of organic contaminants in sentinel Mallards utilizing confined disposal facilities at Hamilton Harbour, Lake Ontario, Canada. Arch. Environ. Contam. Toxicol. 25:234-243.

Haseltine, S. D., M. T. Finley and E. Cromartie. 1980. Reproduction and residue accumulation in Black Ducks fed toxaphene. Arch. Environ. Contam. Toxicol. 9:461-471.

Haseltine, S. D. and R. M. Prouty. 1980. Aroclor 1242 and reproductive success of adult Mallards (*Anas platyrhynchos*). Environ. Res. 23:29-34.

Health and Welfare Canada. 1979. Methylmercury in Canada: Exposure of Indian and Inuit Residents to Methylmercury in the Canadian Environment. Health and Welfare Canada, Medical Services Branch, Ottawa. 200p.

Health and Welfare Canada. 1987. Environmental radioactivity in Canada, 1986. Health and Welfare Canada, Environmental Health Directorate Pub. 87-EHD-136, Ottawa., 121p.

Health Canada. 1998. The Food and Drugs Act and Regulations, with Amendments to 17Dec97. Ottawa. Division 15, Table II.

Heath, R. G. 1969. Nationwide survey of residues of organochlorine pesticides in wings of Mallards and Black Ducks. Pestic. Monit. J. 3:115-123.

Heath, R. G. and S. A. Hill. 1974. Nationwide organochlorine and mercury residues in wings of adult Mallards and Black Ducks during the 1969-70 hunting season. Pestic. Monit. J. 7:153-164.

Hebert, C. E., G. D. Haffner, I. M. Weis, R. Lazar and L. Montour. 1990. Organochlorine contaminants in duck populations of Walpole Island. J. Great Lakes Res. 16:21-26.

Hebert, C. E. and K. A. Keenleyside. 1995. To normalize or not to normalize? Fat is the question. Environ. Toxicol. Chem. 14:801-807.

Hebert, C. E., R. J. Norstrom, M. Simon, B. M. Braune, D. V. Weseloh and C. R. Macdonald. 1994. Temporal trends and sources of PCDDs and PCDFs in the Great Lakes: herring gull egg monitoring, 1981-1991. Environ. Sci. Technol. 28:1268-1277.

Hebert, C. E., D. V. Weseloh, L. Kot and V. Glooschenko. 1994. Organochlorine contaminants in a terrestrial foodweb on the Niagara Peninsula, Ontario, Canada. Arch. Environ. Contam. Toxicol. 26:356-366.

Heinz, G. H. 1979. Methylmercury: Reproductive and behavioural effects on three generations of Mallard ducks. J. Wildl. Manage. 43:394-401.

Heinz, G. H. 1996. Selenium in birds. In: Environmental Contaminants in Wildlife: Interpreting Tissue Concentrations. *Edited by*: W.N. Beyer, G.H. Heinz and A.W. Redmon-Norwood. SETAC Special Publication Series, CRC Press Inc, Boca Raton, USA. pp. 447-458.

Heinz, G. H., D. J. Hoffman and L. H. Gold. 1989. Impaired reproduction of Mallards fed an organic form of selenium. J. Wildl. Manage. 53:418-428.

Heinz, G. H., D. J. Hoffman, A. J. Krynitsky and D. M. G. Weller. 1987. Reproduction in Mallards fed selenium. Environ. Toxicol. and Chem. 6:423-433.

Heinz, G. H., D. S. Miller, B. J. Ebert and K. L. Stromborg. 1994. Declines in organochlorines in eggs of Red-breasted Mergansers from Lake Michigan, 1977-1978 versus 1990. Environ. Monitor. and Assess. 33:175-182.

Heinz, G. H., G. W. Pendleton, A.J. Krynitsky and L.G. Gold. 1990. Selenium accumulation and elimination in Mallards. Arch. Environ. Contam. and Toxicol. 19:374-379.

Hoffman, D. J., C. P. Rice and T. J. Kubiak. 1996. PCBs and Dioxins in Birds. In: Environmental Contaminants in Wildlife: Interpreting Tissue Concentrations. *Edited by*: W. N. Beyer, G. H. Heinz and A. W. Redmon-Norwood. SETAC Special Publication Series, CRC Press Inc, Boca Raton, USA. pp. 165-207.

Holt, G., A. Froslie and G. Norheim. 1979. Mercury, DDE, and PCB in the avifauna in Norway 1965-1976. Acta Veterinaria Scandinavica Supp. 70:1-28.

Hyde, K. M., J. B. Graves, A. B. Watts and F. L. Bonner. 1973. Reproductive success of Mallard ducks fed mirex. J. Wildl. Manage. 37:479-484.

IJC, United States and Canada. 1987. Revised Great lakes Water Quality Agreement of 1978 as amended by protocol signed November 18, 1987. Office Consolidation. International Joint Commission, Washington and Ottawa. 84p.

Jacknow, J., J. L. Ludke and N. C. Coon. 1986. Monitoring Fish and Wildlife for Environmental Contaminants: the National Contaminant Biomonitoring Program. U.S. Fish and Wildlife Service, Fish and Wildlife Leaflet 4. 15p.

James Bay and Northern Québec Native Harvesting Research Committee. 1988. Final Report: Research to Establish Present Levels of Native Harvesting for the Inuit of Northern Québec. 1976-1980. James Bay and Northern Québec Native Harvesting Research Committee: Québec, Québec City.

Jehl, J. R., Jr. 1973. Studies of a declining population of Brown Pelicans in Northwestern Baja California. Condor 75:69-79.

Jensen, J., K. Adare and R. Shearer. (eds). 1997. Canadian Arctic Contaminants Assessment Report. Indian and Northern Affairs Canada, Ottawa. 460p.

Johnstone, R. M., G. S. Court, A. C. Fesser, D. M. Bradley, L. W. Oliphant and J. D. MacNeil. 1996. Long-term trends and sources of organochlorine contamination in Canadian Tundra Peregrine Falcons, *Falco peregrinus tundrius*. Environ. Pollut. 93:109-120.

Joshi, S. R. 1991. Radioactivity in the Great Lakes. Sci. Tot. Environ. 100:61-104.

Kearney, J. D. C. Cole and D. Haines. 1995. Draft Report on the Great Lakes Anglers Pilot Exposure Assessment Study. Great Lakes Health Effects Program, Health Canada, Ottawa.

Kim, H. T., K. S. Kim, J. S. Kim and W. B. Stone. 1985. Levels of polychlorinated biphenyls (PCBs), DDE, and Mirex in waterfowl collected in New York State, 1981-1982. Arch. Environ. Toxicol. 14:13-18.

Kim, K. S., M. J. Pastel, J. S. Kim and W. B. Stone. 1984. Levels of poly-chlorinated biphenyls, DDE and Mirex in waterfowl collected in New York State, 1979-1980. Arch. Environ. Contam. Toxicol. 13:373-381.

King, K. A., E. L. Flickinger and H. H. Hildebrand. 1977. The decline of Brown Pelicans on the Louisiana and Texas Gulf Coast. Southwest. Natural. 21:417-431. (Cited in Wiemeyer, 1996)

Kinloch, D., H. Kuhnlein and D. C. G. Muir. 1992. Inuit foods and diet: a preliminary assessment of benefits and risks. Sci. Tot. Environ. 122:247-278.

Klimkiewicz, K. and C. S. Robbins. 1978. Standard abbreviations for common names of birds. North American Bird Bander 3: 16-25.

Kuhnlein, H. V. 1991. Dietary Evaluation of Food, Nutrients and Contaminants in Fort

Good Hope and Colville Lake, Northwest Territories. Final Report to Medical Services Branch and National Health Research and Development Program - North 60°, Health and Welfare Canada. McGill University. 138p.

Legris, A. M. and H. Lévesque. 1991. Migratory Game Birds Harvested in Canada During the 1990 Hunting Season. Canadian Wildlife Service Progress Note 197. 40p.

Lemly, A. D. 1993. Guidelines for evaluating selenium data from aquatic monitoring and assessment studies. Environ. Monit. Assess. 28:83-100.

Lévesque, H. B. Collins and A. M. Legris. 1993. Migratory Game Birds Harvested in Canada During the 1991 Hunting Season. Canadian Wildlife Service Progress Note 204. 42p.

Liska, B. J., A. R. Stemp and W. J. Stadelman. 1967. Effect of method of cooking on chlorinated insecticide residue content in edible chicken tissue. Food Technology, 21:435-438.

Longcore, J. R., F. B. Samson and T. W. Whittendale, Jr. 1971. DDE thins eggshells and lowers reproductive success of captive Black Ducks. Bull. Environ. Contam. Toxicol. 6:485-490.

Macdonald, C. R., L. L. Ewing, B. T. Elkin and A. M. Wiewel. 1996. Regional variation in radionuclide concentrations and radiation dose in Caribou (*Rangifer tarandus*) in the Canadian Arctic; 1992-94. Sci. Tot. Environ. 182:53-73.

McGill, R, J. W. Tukey and W. A. Larsen. 1978. Variations of Box Plots. Amer. Statist. 32:12-16.

Mehrle, P. M., M. T. Finley, J. L. Ludke, F. L. Mayer and T. E. Kaiser. 1979. Bone development in Black Ducks as affected by dietary toxaphene. Pestic. Biochem. Physiol. 10:168-173.

Michigan DNR. 1993. Michigan Wildlife Contaminant Monitoring Program. 1992 Annual Report. Michigan Department of Natural Resources.

Michot, T. C., W. H. Benson and J. M. O'Neil. 1998. Trace element concentrations from Greater Scaup and other diving ducks wintering in Apalachee Bay, Florida, seagrass beds. Scaup Workshop, Jamestown, ND, 9-10 September 1998. (Abstract).

Moisey, J. M. and B. J. Wakeford. 1995. A Combined Method for the Determination of Polychlorinated Dibenzo-p-dioxins, Polychlorinated Dibenzofurans and Non-ortho Substituted Polychlorinated Biphenyls in Wildlife Tissue. Canadian Wildlife Service Technical Report Series 236, Ottawa. 32p.

Mullie, A., R. K. Ross and D. G. Dennis. 1996. Overview of Migrant Waterfowl Use of the Great Lakes Areas of Concern and Other Selected Coastal Sectors in Southern Ontario. Canadian Wildlife Service Technical Report Series No. 261. 144p.

Myklebust, I., S. Nybø, J. A. Kålås and H. C. Pedersen. 1993. Cadmium accumulation in Willow Ptarmigan (*Lagopus L. Lagopus*) and Rock Ptarmigan (*L. Mutus*) in central Norway. Sci. Tot. Environ. Supp:135-139.

Nebeker, A. V., W. L. Griffis, T. W. Stutzman, G. S. Schuytema, L. A. Carey and S. M. Scherer. 1992. Effects of aqueous and dietary exposure of dieldrin on survival, growth and bioconcentration in Mallard ducklings. Environ. Toxicol. Chem. 11:687-699.

Nin.da.waab.jig Research Centre. 1986. Wildmeat consumption survey; Walpole Island Indian Reserve, Walpole Island, Ontario. 13p.

Norstrom, R. J. and M. Simon. 1991. Methods 5: Determination of specific poly-chlorinated dibenzo-*p*-dioxins and dibenzofurans in biological matrices by gel-permeation-carbon chromatography and gas chromatography-mass spectrometry. In: Environmental Carcinogens. Methods of Analysis and Exposure Measurement. Volume 11, Polychlorinated Dioxins and Dibenzofurans. *Edited by:* C. Rappe, H. R. Buser, B. Dodet and I. K. O'Neill. World Health Organization. International Association for Research on Cancer Scientific Publications No. 108, pp 281-297.

Norstrom, R. J., M. Simon, D. C. G. Muir and R. E. Schweinsburg. 1988. Organochlorine contaminants in arctic marine food chains: identification, geographical distribution, and temporal trends in polar bears. Environ. Sci. and Technol. 22:1063-1071.

Norstrom, R. J. and H. T. Won. 1985. Long-term preservation of egg and tissue homogenates for determination of organochlorine compounds: freezing versus freeze-drying. J. Assoc. Off. Anal. Chem. 68:130-135.

NRCC. 1978. Effects of Arsenic in the Canadian Environment. National Research Council of Canada, Pub. No. NRCC 15391. Ottawa. 363p.

Ohlendorf, H. M., R. L. Hothem, C. M. Bunck and K. C. Marois. 1990. Bioaccumulation of selenium in birds at Kesterson Reservoir, California. Arch. Environ. Contam. Toxicol. 19:495-507.

OMOEE. 1997. Guide to Eating Ontario Sport Fish 1997 - 1998. Ontario Ministry of Environment and Energy. Toronto. 188p.

OMOE and OMNR. 1986. Guide to Eating Ontario Sport Fish 1986. Ontario Ministry of the Environment and Ontario Ministry of Natural Resources. Toronto. 280p.

Peakall, D. B. and A. P. Gilman. 1979. Limitations of expressing organochlorine levels in eggs on a lipid-weight basis. Bull. Environ. Contam. Toxicol. 23:287-290.

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

Pearce, P. A. and J. C. Baird. 1970. DDT closes New Brunswick woodcock season. Canadian Field-Naturalist 85:82.

Pearce, P. A., I. M. Price and L. M. Reynolds. 1976. Mercury in waterfowl from eastern Canada. J. of Wildl. Manage. 40:694-703.

Peterson, S. R. and R. S. Ellarson. 1976. Total mercury residues in livers and eggs of Oldsquaws. J. Wildl. Manage. 40:704-709.

P. Lane and Associates. 1986. Risks to Migratory Birds: Industrial Point Sources of Toxic Chemicals in Atlantic Canada. Prepared for Canadian Wildlife Service, Atlantic Region, Sackville NB.

P. Lane and Associates. 1991. Sources and Releases of Toxic Substances in the Atlantic Region, Final Report. Prepared for the Conservation and Protection Service, Environment Canada, Dartmouth, NS.

Platenow, N. S., L. H. Karstad and P. W. Saschenbrecker. 1973. Tissue distribution of polychlorinated biphenyls (Aroclor 1254) in cockerels; relation to the duration of exposure and observations on pathology. Can. J. Comp. Med. 37:90-95.

Porter, P. S., R. C. Ward and H. F. Bell. 1988. The detection limit. Environ. Sci. Technol. 22:856-861.

Potter, C. M., I. L. Brisbin, Jr., S. G. McDowell and F. W. Whicker. 1989. Distribution of ^{137}Cs in the American Coot (*Fulica americana*). J. Environ. Radioactivity 9:105-115.

Prevett, J. P., H. G. Lumsden and F. C. Johnson. 1983. Waterfowl kill by Cree hunters of the Hudson Bay Lowland, Ontario. Arctic 36:185-192.

Prouty, R. M. and C. M. Bunck. 1986. Organochlorine residues in adult Mallard and Black Duck wings. 1981-82. Environ. Monitor. Assess. 6:49-57.

Receveur, O., M. Boulay, C. Mills, W. Carpenter and H. V. Kuhnlein. 1996. Variance in Food Use in Dene/Métis Communities. Centre for Indigenous Peoples' Nutrition and Environment (CINE), McGill University, Montreal. 198 + 38p.

Ritchey, S. J., R. W. Young and E. O. Essary. 1967. The effects of cooking on chlorinated hydrocarbon pesticide residues in chicken tissue. J. Food Sci. 32:238-240.

Ritchey, S. J., R. W. Young and E. O. Essary. 1969. Cooking methods and heating effects on DDT in chicken tissues. J. Food Sci. 34:569-571.

Robbins, C. S., B. Bruun and H. S. Zim. 1983. Birds of North America. Rev. ed. Golden Press, New York. 360p.

Rodrigue, J., J-L. Desgranges and R. Titman. 1992. Use of Peking Ducks (*Anas platyrhynchus*) as a bioindicator of contamination by bioaccumulative substances in the natural environment (St. Lawrence River, Canada). In: Proc. VIth Int. Conf. Bioindicatores Deteriorisation Regionis. *Edited by*: Bohá_, J. Institute of Landscape Ecology CAS, _eské Bud_jovice, Czechoslovakia. pp. 278-285.

Rodrigue, J., J-L. Desgranges and R. Titman. 1994. Utilisation de Canards Domestiques pour la Surveillance des Contaminants dans Le Fleuve Saint-Laurent. Série de rapports techniques No. 192, Service Canadien de la Faune, Région du Québec. ix + 86p.

SAS Institute. 1988. SAS/STAT User's Guide Release 6.03 Edition. SAS Institute, Cary, NC.

Scheuhammer, A. M. and P. J. Blancher. 1994. Potential risk to Common Loons (*Gavia immer*) from methylmercury exposure in acidified lakes. Hydrobiologia 279/280:445-455.

Scheuhammer, A. M., J. A. Perrault, E. Routhier, B. M. Braune and G. D. Campbell.,

1998. Elevated lead concentrations in edible portions of game birds harvested with lead shot. Environ. Pollut. 102:251-257.

Scheuhammer, A. M., A. H. K. Wong and D. Bond. 1998. Mercury and selenium accumulation in Common Loons (*Gavia immer*) and Common Mergansers (*Mergus merganser*) from eastern Canada. Environ. Toxicol. And Chem. 17:197-201.

Schmitt, C. J. and C. M. Bunck. 1995. Persistent environmental contaminants in fish and wildlife. In: Our Living Resources: A Report to the Nation on the Distribution, Abundance and Health of U.S. Plants, Animals and Ecosystems. *Edited by*: E. T. Laroe, G. S. Farris, C. E. Puckett, P. D. Doran and M. J. Mac. U.S. Department of the Interior, National Biological Service, Washington, DC. pp 413-416.

Schwartz, T. R. and D. L. Stalling. 1991. Chemometric comparison of polychlorinated biphenyl residues and toxicologically active polychlorinated biphenyl congeners in the eggs of Forster's Terns (*Sterna fosteri*). Arch. Environ. Contam. Toxicol. 20:183-199.

Sibley, C. G. and B. L. Monroe, Jr. 1990. Distribution and Taxonomy of Birds of the World. Yale University Press, New Haven and London. 1111 p.

Silver, T. M. and T. D. Nudds. 1995. Influence of low-level cadmium and reduced calcium intake on tissue Cd concentrations and behaviour of American Black Ducks. Environ. Pollut. 90:153-161.

Skinner, L. C. 1992. Chemical Contaminants in Wildlife from the Mohawk Nation at Akwesasne and the Vicinity of the General Motors Corporation / Central Foundry Division, Massena, New York Plant. Tech. Rep. 92-4. Division of Fish and Wildlife, New York State Department of Environmental Conservation. Albany, NY. 113p.

Smith, V. E., J. M. Spurr, J. C. Filkins and J. J. Jones. 1985. Organochlorine contaminants of wintering ducks foraging on Detroit River sediments. J. Great Lakes Res. 11:231-246.

Smoczynski, S. and R. Amarowicz. 1986. Effect of cooking on gamma-HCH, DDE, DDD and DDT levels in duck muscle fat. Acta Alimentaria Polonica, 12:21-27.

Sokal, R. R. and F. J. Rohlf. 1981. Biometry: The Principles and Practice of Statistics in Biological Research. 2nd ed. W. H. Freeman and Company, New York, New York.

859p.

Stanley, T. R., J. W. Spann, G. J. Smith and R. Roscoe. 1994. Main and interactive effects of Arsenic and Selenium on Mallard reproduction and duckling growth and survival. Arch. Environ. Contam. Toxicol. 26:444-451.

Taylor, H. W., J. Svoboda, G. H. R. Henry and R. W. Wein. 1988. Post Chernobyl ^{134}Cs and ^{137}Cs levels at some localities in northern Canada. Arctic 41:293-296.

Thomas, D. J., B. Tracey, H. Marshall and R. J. Norstrom. 1992. Arctic terrestrial ecosystem contamination. Sci. Tot. Environ. 122:135-164.

Thompson, D. R. 1996. Mercury in birds and terrestrial mammals. In: Environmental Contaminants in Wildlife: Interpreting Tissue Concentrations. *Edited by*: W.N. Beyer, G.H. Heinz and A.W. Redmon-Norwood. SETAC Special Publication Series, CRC Press Inc, Boca Raton, USA. pp. 341-356.

Tilling, L., J. Kiro and E. Fox. 1994. Sigmaplot Scientific Graphing Software v2.01 Users Manual. Jandel Corp., San Rafael, CA.

Tobias, T. N. and J. J. Kay. 1993. The bush harvest in Pinehouse, Saskatchewan. Arctic 47:207-221.

Turle, R. and B. Collins. 1992. Validation of the use of pooled samples for monitoring of contaminants in wildlife. Chemosphere 25:463-469.

Turle, R. and R. J. Norstrom. 1987. The CWS Guideline to Practical Quality Assurance for Contracted Chemical Analysis. Canadian Wildlife Service Technical Report Series No. 21, Ottawa. 25p.

USEPA. 1995. Great Lakes Water Quality Initiative Criteria Documents for the Protection of Wildlife. EPA-820-B-95-008. US Environmental Protection Agency, Office of Science and Technology, Washington DC.

USFDA. 1979. Polychlorinated biphenyls (PCBs): reduction of tolerances. Fed. Reg. 49(Jun):38330-38340.

USFDA. 1981. FDA Compliance Program Guidance Manual. Section 7304.004 - Pesticides and Industrial Chemicals in Domestic Foods Program (F82). U. S. Food and Drug Administration, Washington D.C.

Usher, P. J., M. Bailie, M. Demmer, D. Nakashima, M. G. Stevenson and M. Stiles. 1995.

Communicating About Contaminants in Country Food: the Experience in Aboriginal Communities. Inuit Tapirisat of Canada. Ottawa. 238p.

Vermeer, K., F. A. J. Armstrong and D. R. M. Hatch. 1973. Mercury in aquatic birds at Clay Lake, western Ontario. J. Wildl. Manage. 37:58-61.

Vermeer, K. and D. B. Peakall. 1979. Trace metals in sea ducks of the Fraser River delta intertidal area, British Columbia. Mar. Pollut. Bull. 10:193-197.

Vermeer, K. and J. A. J. Thompson. 1992. Arsenic and copper residues in waterbirds and their food down inlet from the Island Copper Mill. Bull. Environ. Contam. Toxicol. 48:733-738.

Wakeford, B. and R. Turle. 1997. In-house reference materials as a means to quality assurance: the Canadian Wildlife Service experience. In: Reference Materials for Environmental Analysis. *Edited by*: R. E. Clement, L. H. Keith and K. W. M. Siu. CRC Press , Boca Raton, FL. pp 205-231.

Wein, E. E. and M. M. R. Freeman. 1995. Frequency of traditional food use by three Yukon First Nations living in four communities. Arctic 48(2):161-171.

Wein, E. E., J. Henderson Sabry and F. T. Evers. 1991. Food consumption patterns and use of country foods by Native Canadians near Wood Buffalo National Park, Canada. Arctic 44(3):196-205.

Weseloh, D. V., P. Hamr and C. A. Bishop. 1995. Organochlorine contaminant levels in waterbird species from Hamilton Harbour, Lake Ontario: an IJC area of concern. J. Great Lakes Res. 21:121-137.

Weseloh, D. V., J. Struger and C. Hebert. 1994. White Pekin Ducks (*Anas platyrhynchos)* as monitors of organochlorine and metal contamination in the great lakes. J. Great Lakes Res. 20:277-288.

White, D. H. 1979. Nationwide residues of organochlorine compounds in wings of adult Mallards and Black Ducks, 1976-77. Pestic. Monit. J. 13:12-16.

White, D. H. and M. T. Finley. 1978. Uptake and retention of dietary cadmium in Mallard ducks. Environ. Res. 17:53-59.

White, D. H. and R. G. Heath, 1976. Nationwide residues of organochlorines in wings of adult Mallards and Black Ducks, 1972-73. Pestic. Monit. J. 9:176-185.

White, D. H. and T. E. Kaiser. 1976. Residues of organochlorines and heavy metals in Ruddy Ducks from the Delaware River, 1973. Pestic. Monit. J. 9:155-156.

White, D. H. and J. T. Seginak, 1994. Dioxins and furans linked to reproductive impairment in Wood Ducks. J. Wildl. Manage. 58:100-106.

Whitehead, P. E., J. E. Elliott, R. J. Norstrom and K. Vermeer.1990. PCDD and PCDF contamination of waterfowl in the Strait of Georgia, British Columbia, Canada: 1989-1990. Organohalogen Compounds 1:459-462.

Wiemeyer, S. N. 1996. Other Organochlorine Pesticides in Birds. In: Environmental Contaminants in Wildlife: Interpreting Tissue Concentrations. *Edited by*: W.N. Beyer, G.H. Heinz and A.W. Redmon-Norwood. SETAC Special Publication Series, CRC Press Inc, Boca Raton, USA. pp. 99-115.

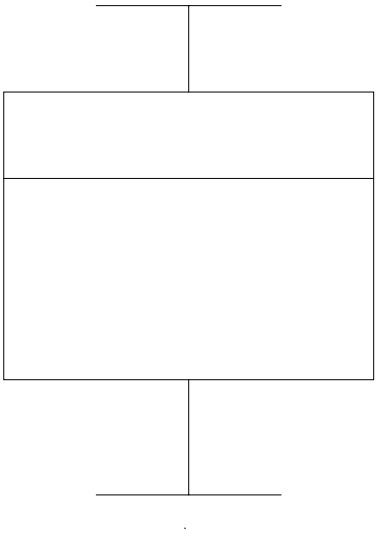
Wong, M. P. 1985. Chemical Residues in Fish and Wildlife Harvested in Northern Canada. Environmental Studies No. 46. Indian and Northern Affairs Canada. Ottawa.

Zhu, J. and R. J. Norstrom. 1993. Identification of polychlorocamphenes (PCCs) in the Polar Bear (*Ursus maritimus*) food chain. Chemosphere 27:1923-1936.

Appendix A - Figures

Notes on Data Presentation and Figure Legends

Box and Whisker Plot Structure



- outlier data points
- 90th percentile
- 75th percentile
- 50th percentile (Median)
- 25th percentile
- 10th percentile
- outlier data point

Values Below Detection Limit

% Quantifiable refers to % of values ≥0.001 mg·kg $^{-1}$.
% Detect refers to % of values ≥0.0001 mg·kg $^{-1}$ and includes trace values.

For purposes of data illustration the actual values were used for trace organochlorine values (<0.001 and >0.0001 mg·kg $^{-1}$). Non-detects (<0.0001 mg·kg $^{-1}$) were set to 0.0.

Metals values were plotted only where there was greater than 65% detection. Values below the detection limits were set to one half the detection limit.

Sample Sizes

The numbers under the box plots (Fig 6-20) refer to sample sizes (i.e - the number of pools) for the given group.

Breeding Season Diets (Fig 2)

This figure is based on information in Ehrlich *et al.*, 1988

Acronyms and Abbreviations Used on Axes

Trophic Groups (Fig 3, 9, 17)

IN - Insectivores BR - Terrestrial Browsers GR - Geese & Swans
DD - Dabbling Ducks BD - Bay Ducks SD - Sea Ducks
ME - Mergansers LN - Loons TOT - all groups

(See Table 2 - Species List for a list of the species in each group)

Upland gamebirds refers to the Insectivore and Terrestrial browser Groups combined, all other groups in the figures are as those in Table 2. Individual Species acronyms are defined in Table 2. OTHER refers to all other species in the relevant group therefore for:

Upland gamebirds - OTHER refers to RUGR, STGR and PTAR
Geese & Swans - OTHER refers to ATBR, BLBR, GWFG and WHSW
Bay Ducks - OTHER refers to CANV & REDH
Mergansers & Loons - OTHER refers to PALO, RTLO & YBLO

Chlordane Residues (Fig 4)

c-chl - *cis*-chlordane o-chl - oxychlordane
t-chl - *trans*-chlordanec-non - *cis*-nonochlor
t-non - *trans*-nonochlor he - heptachlor epoxide

PCB Congener Groups (Fig 5)

tri-cb - Trichlorobiphenyls (PCB 28, 31)
tetra-cb - Tetrachlorbiphenyls (PCB 42, 44, 47, 49, 52, 60, 64, 66/95, 70, 74)
(Although PCB 95 is a pentachlorobiphenyl it co-elutes with and cannot be distinguished from PCB 66)
penta-cb - Pentachlorobiphenyls (PCB 87, 97, 99, 101, 105, 110, 118)
hexa-cb - Hexachlorobiphenyls (PCB 128, 129, 137, 138, 141, 146, 149, 151, 153, 58)
hepta -cb - Heptachlorobiphenyls (PCB 170/190, 171, 172, 174, 180, 182/187, 183, 185)
octa-cb - Octachlorobiphenyls (PCB 194, 195, 200, 201, 203)
nona-cb - Nonachlorobiphenyls (PCB 206)

Geographical Comparisons (Fig 6 - 8)

Eastern Canada (Fig 6) - Newfoundland and Labrador, Nova Scotia, New Brunswick, Prince Edward Island, Québec, Ontario and that part of the Northwest Territories east of

95°W. (Regions 1-7 & 10 from Table 1)

Western Canada (Fig 6) - Manitoba, Saskatchewan, Alberta, British Columbia, Yukon and that part of the Northwest Territories west of 95°W. (Regions 8, 9 & 11 from Table 1)

Northern Canada (Fig 7) - The Northwest Territories, that part of Québec and Labrador north of 55°N, and the Yukon except for the Liard River watershed and the Pacific watershed (but including the Porcupine River watershed). (Regions 10 & 11 from Table 1)

Southern Canada (Fig 7) - Newfoundland, Nova Scotia, New Brunswick, Prince Edward Island, Ontario, Manitoba, Saskatchewan, Alberta, British Columbia, that part of theYukon in the Liard River and Pacific watersheds (except for the Porcupine River watershed) and that part of Québec and Labrador south of 55°N. (Regions 1- 9 from Table 1).

Northeastern Canada (Fig 8) - The Northwest Territories east of 95°W and that part of Québec and Labrador north of 55°N. (Region 10 from Table 1) .

Northwestern Canada (Fig 8) - The Northwest Territories west of 95°W and the Yukon except for the Liard River watershed and the Pacific watershed (but including the Porcupine River watershed). (Region 11 from Table 1).

Southeastern Canada (Fig 8) - Newfoundland, Nova Scotia, New Brunswick, Prince Edward Island, Ontario and that part of Québec and Labrador south of 55°N. (Regions 1-7 from Table 1).

Southwestern Canada (Fig 8) - Manitoba, Saskatchewan, Alberta, British Columbia, that part of theYukon in the Liard River and Pacific watersheds (except for the Porcupine River watershed). (Regions 8 & 9 from Table 1).

Arctic (Fig 16) - For this graph Arctic refers to the Northwest Territories, all of the Yukon and that part of Québec and Labrador north of 55°N. (Regions 10, 11 and those parts of the Yukon in regions 8 & 9 from Table 1).

Species Acronyms (Fig 10 - 16, 18 - 20)

See Table 2 - Species List for a list of all the species and their acronyms.

Appendix B - Tables

Notes on Tables, Symbol Explanations and Table Footnotes

Formulas and Units for Statistical Summary Tables (Tables 3, 13)

POOLS: # of pools in all regions

BIRDS: Sum of all individuals in all pools

DETECTS: Count of all values ≥0.0001 $mg \cdot kg^{-1}$ (> Sample Detection Limit for Metals)

% DETECTS: (# DETECTS/(# DETECTS + # NON-DETECTS))*100

NON-DETECTS (nd): # of values < 0.0001 $mg \cdot kg^{-1}$ (< Sample Detection Limit for Metals)

TRACE VALUES (Organochlorines) (tr): # of detected values that were not confidently quantifiable ($0.0001 \leq tr < 0.001$ $mg \cdot kg^{-1}$)

MISSING VALUES (Metals): # POOLS - # DETECTS - # NON-DETECTS

MIN ($mg \cdot kg^{-1}$ [ppm] wet weight): Minimum value ("tr" if less than 0.001 $mg \cdot kg^{-1}$, "nd" if less than 0.0001 $mg \cdot kg^{-1}$ for organochlorines or if < Sample Detection Limit for Metals)

MAX ($mg \cdot kg^{-1}$ [ppm] wet weight): Maximum value ("tr" if less than 0.001 $mg \cdot kg^{-1}$, "nd" if less than 0.0001 $mg \cdot kg^{-1}$ for organochlorines or if < Sample Detection Limit for Metals)

MEDIAN ($mg \cdot kg^{-1}$ [ppm] wet weight): Median of pooled values ("na" if less than 0.001 $mg \cdot kg^{-1}$ for organochlorines or if < Sample Detection Limit for Metals). Non-detects set to 1/2 detection limit for this calculation.

Data Table Heading and Symbol Explanations (Table 4, 5, 8 - 12, 14 - 23)

Age: I - Immature; A - Adult; U - Unknown

Sex: F - Female; M - Male; U - Unknown

Year: The year code consists of the year and a season code:
a -Jan-Mar; b - Apr-Jun; c - Jul-Aug; d - Sep-Dec

Mapref: See Table 1 and Figure 1 for map location references

Prov: Province of collection

S_CHL: Sum of oxy-, *trans*- & *cis*-chlordane, *trans*- & *cis*-nonachlor and heptachlor epoxide

S_PCB: Sum of PCB congeners 28, 31, 44, 52, 60, 66/95, 87, 97, 99, 101, 105, 110, 118, 138, 141, 146, 153, 170/190, 171, 172, 174, 180, 182/187, 183, 194, 195, 201, 203, 206 (This is standardized to these 32 congeners, [even if more or fewer congeners have been analysed for] to allow comparability through time as the number of congeners analysed changes.

OCS: Octochlorostyrene

S_Mirex: Sum of Photo-mirex and Mirex

S_DDT: Sum of *pp*'-DDE, *pp*'-DDD and *pp*'-DDT

1,2,4,5-T4CB: 1,2,3,5-tetrachlorobenzene

1,2,3,4-T4CB: 1,2,3,4-tetrachlorobenzene

QCB: Pentachlorobenzene

HCB: Hexachlorobenzene

S_CBz: Sum of 1,2,4,5- & 1,2,3,4-Tetrachlorobenzene, Pentachlorobenzene and Hexachlorobenzene

a-HCH: α-hexachlorocyclohexane

b-HCH: β-hexachlorocyclohexane

g-HCH: γ-hexachlorocyclohexane

S_HCH: Sum of α-, β-, and γ- hexachlorocyclohexanes

2378 TCDD: 2,3,7,8-tetrachlorodibenzo-p-dioxin

12378 PnCDD: 1,2,3,7,8-pentachlorodibenzo-p-dioxin

123478 HxCDD: 1,2,3,4,7,8-hexachlorodibenzo-p-dioxin

123678 HxCDD: 1,2,3,6,7,8-hexachlorodibenzo-p-dioxin

123789 HxCDD: 1,2,3,7,8,9-hexachlorodibenzo-p-dioxin

1234678 HpCDD: 1,2,3,4,6,7,8-heptachlorodibenzo-p-dioxin

OCDD: octachlorodibenzo-p-dioxin

2378 TCDF: 2,3,7,8-tetrachlorodibenzofuran

12378 PnCDF: 1,2,3,7,8-pentachlorodibenzofuran

23478 PnCDF: 2,3,4,7,8-pentachlorodibenzofuran

123478 HxCDF: 1,2,3,4,7,8-hexachlorodibenzofuran

123678 HxCDF: 1,2,3,6,7,8-hexachlorodibenzofuran

123789 HxCDF: 1,2,3,7,8,9-hexachlorodibenzofuran

234678 HxCDF: 2,3,4,6,7,8-hexachlorodibenzofuran

1234678 HpCDF: 1,2,3,4,6,7,8-heptachlorodibenzofuran

1234789 HpCDF: 1,2,3,4,7,8,9-heptachlorodibenzofuran

OCDF: octachlorodibenzofuran

MQC: Minimal Quantifiable Concentration for Dioxins (Signal/Noise ratio < 3)

As: Arsenic; Se: Selenium; Pb: Lead; Hg: Total Mercury; Cd: Cadmium

^{137}Cs: Cesium-137 activity (decay-corrected to collection date) measured in Bequerel/kg. The limit of detection, equivalent to a signal three times the standard deviation in the spectrometric background, is given in place of imprecise determinations of low

activities.

Uncertainty (Bq/kg): Uncertainty in the specific activity of ^{137}C, based on the standard counting error, or one standard deviation.

bm: Pectoral (breast) muscle

wb: Whole body

Trigger: Level at which consumption advisories are issued

Guideline: Maximum allowable level but without regulatory force

MAC: Maximum Acceptable Concentration (Regulatory Agency)

NOAEL: No observable effect Level

LOAEL: Lowest observable effect Level

Critical level: Scientific assessment of risk

Blank values indicate not analysed for that parameter

nd (non-detect): < 0.0001 $mg \cdot kg^{-1}$ for organochlorines and PCBs
< Sample Detection Limit for metals
signal/noise < 2 for dioxins, furans and non-ortho PCBs

tr (trace): the compound was detected but was not quantifiable
0.0001 <= "tr" <0.001 $mg \cdot kg^{-1}$ for organochlorines and PCBs
3 > Signal/Noise ratio >2 for dioxins, furans and non-ortho PCBs

Data Table Footnotes (Tables 4, 5, 8 - 12, 14 - 17)

These are the explanations for the flags after the species names

a - average of two replicates: where there was an “nd” value and a positive value, the positive value was used instead of an average

b - S_PCB is underestimated for this sample because fewer than the 32 designated congeners were analysed for

c - Average of 2 replicates (dioxins/furans); where there was a <MQC value and a positive value, the positive value was used instead of an average. Where the MQC's were different the higher MQC was used.

d - Average of two replicates (metals); where there was an "nd" value and a positive value, the positive value was used instead of an average.

e - Pb value is an average of 3 replicates

f - Average of 3 replicates (metals); where there was an "nd" value and a positive value, the positive value was used instead of an average.

g - 43180 & 43257 (Mallard) were pooled into 90-2931 (Health Canada, Radiation Protection Branch, sample ID number), 43279 & 43280 (Common Eider) were pooled into 90-2933 and 49332 & 49333 (Lesser Snow Goose) were pooled into 92-0927 for ^{137}Cesium analysis. No CWS USOX numbers exist for these pools.

h - metals pool values are average of individual analyses for these samples:
the data on individual birds have been previously published in:
Braune, B. M., Wong, M. P., Belles-Isles, J.-C. and Marshall, K. 1991. Chemical Residues in Canad

i - metals data for pectoral muscle also exists for this liver pool, under the USOX # in the BM USOX column