

GEOGRAPHICAL DISTRIBUTION OF ORGANOCHLORINE CONTAMINANTS AND REPRODUCTIVE PARAMETERS IN HERRING GULLS ON LAKE SUPERIOR IN 1983

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(Received: February 1993; revised: September 1993)

Abstract. As part of the Great Lakes International Surveillance Plan, 1978–83, egg contaminant levels and reproductive output were determined for Herring Gull colonies on Lake Superior in 1983. Since 1974, the Herring Gull has been widely used in the Great Lakes as a spatial and temporal monitor of organochlorine (OC) contaminant levels and associated biological effects. Most eggs contained a wide range of OCs, the main compounds being DDE, polychlorinated biphenyls (PCBs), dieldrin, heptachlor epoxide, oxychlordane, hexachlorobenzene and mirex. Levels of an additional ten OCs and five polychlorinated dibenzo-*p*-dioxin (PCDD) congeners were also determined for some sites. Overall, levels varied significantly among colonies, but there was no obvious relationship to spatial distribution of contaminants in sediments or fish species. OC levels in eggs had declined by up to 84% since 1974. Eggshells were only 8% thinner than before the introduction of DDT, and shell thinning was not a cause of breeding failure. Average reproductive output varied from 0.15 to 1.57 young per apparently occupied nest in 1983: at 56% of colonies the value was below that thought necessary to maintain stable populations. The main causes of failure were egg disappearance and cannibalism of chicks. Despite this, the population appeared to have been increasing at about 4% per annum. Reduced availability of forage fish during the early 1980s was the most likely reason for the poor reproductive output in 1983.

1. Introduction

The Herring Gull (*Larus argentatus*), has been a valuable and successful indicator of toxic chemicals in the Great Lakes ecosystem since the late 1960s and early 1970s (Keith, 1966; Ludwig and Tomoff, 1966; Gilbertson, 1974; Weseloh *et al.*, 1979; Mineau *et al.*, 1984; Government of Canada, 1991). This is due mainly to the relatively widespread breeding distribution and resident status of adults on the Great Lakes (Moore, 1976), as well as the predominantly piscivorous diet and hence capacity to bioaccumulate lipophilic organochlorine contaminants (Braune *et al.*, 1989; Fox *et al.*, 1990; Ewins *et al.*, 1993). By using the Herring Gull as an indicator species, spatial and temporal trends in the concentrations of a variety of chemical contaminants have been identified, and a wide range of biological effects of these toxicants on Herring Gulls have been documented (Gilbertson and

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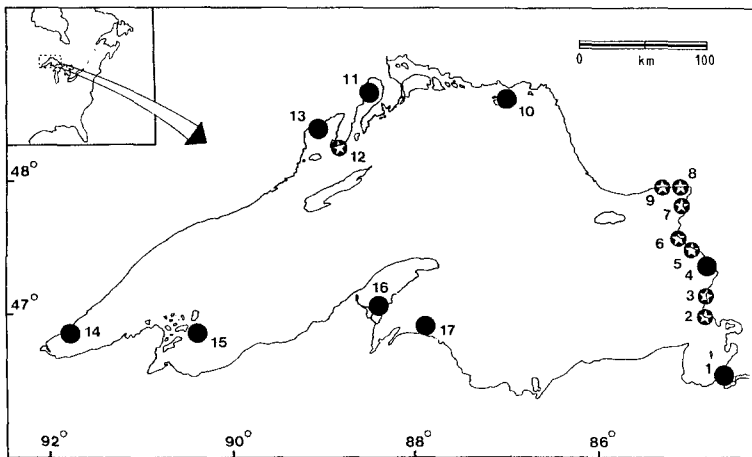


Fig. 1. The location of Herring Gull colonies studied in Lake Superior. Filled circles represent colonies for which egg contaminant data exist, stars indicate colonies having reproductive data only. 1 = Chêne I., 2 = Sawpit Bay, 3 = Cozen's Cove Is., 4 = Agawa Rk., 5 = Telegraph Rk., 6 = Devil's Warehouse I., 7 = Entrance I., 8 = S. Dore Bay, 9 = I. E of False Dog Harbour, 10 = Leadman I., 11 = Granite I., 12 = Silver Islet, 13 = Papoose I., 14 = Knife I., 15 = Gull I. (Apostle Islands), 16 = Lake Linden, 17 = Huron Is.

Fox, 1977; Gilman *et al.*, 1977; Fox *et al.*, 1978, 1988, 1991; Mineau *et al.*, 1984; Moccia *et al.*, 1986).

In this paper we report the results of investigations of contaminants and reproductive biology in Herring Gulls on Lake Superior between 1974 and 1984. Contaminant data are given for Herring Gulls at eight colonies in 1983, as well as trends in two areas since 1974. These results are discussed in relation to similar published data for fish and sediments. Reproductive success, clutch size and eggshell thickness measurements are provided for up to 17 colonies throughout Lake Superior. These results represent the most extensive sampling for chemical contaminants and reproductive parameters in any bird population on Lake Superior, of which we are aware. This was the last lake to be so covered on an interdisciplinary basis under the Great Lakes International Surveillance Plan (GLISP), 1978–83. Data have already been published for the GLISP studies of Herring Gulls on Lake Erie (Weseloh *et al.*, 1990) and Lake Huron (Ewins *et al.*, 1992).

2. Study Areas

Data were collected from a total of 17 Herring Gull colonies on Lake Superior (Figure 1). Colonies were all located on islands, supporting between 30 and 485 pairs of breeding Herring Gulls (Table V). Most islands were granitic bedrock, with some shrub or tree cover, but Chêne Island was a long, sandy bar, only 1–2 m above the water line. The Lake Linden colony was situated on an island which

had been created from old mine tailings and industrial spoil. Rubble from former mining operations was also present on Silver Islet. Names of colonies follow those given by Scharf *et al.* (1978) and Blokpoel *et al.* (1980).

3. Methods

3.1. EGG COLLECTIONS AND ANALYSES

Since 1974 Herring Gull eggs have been collected annually for contaminant analyses from two Lake Superior colonies, as part of the network of Annual Monitoring Colonies (AMCs) on the Great Lakes (Gilman *et al.*, 1977; Mineau *et al.*, 1984). The AMCs have been on Granite Island, in western Lake Superior, since 1973, and at Mamainse Harbour or on Agawa Rock, in eastern Lake Superior, since 1974 (Figure 1). Egg collections could not be made at Granite I. in 1975, 1977 or 1983, but in 1975 and 1977 eggs from Silver Islet, 50 km to the southwest, were used as a substitute. In 1974–75 and 1977 eggs were collected from a colony at Mamainse Harbour. In 1978 the Mamainse Harbour colony was judged too small for further biological work, so Agawa Rock, 36 km to the north, was substituted. In 1983 eggs were collected at seven other colonies distributed throughout the lake (Table I, Figure 1). Reproductive and egg parameters were obtained from these colonies as well as those from a group of islands in eastern Lake Superior (Figure 1).

Between 10 and 13 eggs were collected in early-mid May, usually within the first 10 days of incubation. One egg was taken, at random, from each completed (three-egg) clutch, and nests were selected at irregular intervals across the entire colony. In this way we attempted to reduce any biases which may have arisen because of differential ages of birds at the centre or edge of the colonies (e.g. Coulson, 1968; Aebischer and Coulson, 1990). Eggs were transported in padded containers and stored at 4°C within 48 hours of collection.

Two to three weeks later eggs were cut open around the equator with hexane-rinsed dissecting scissors, and their contents frozen at –20°C in hexane-rinsed glass jars. Up to 10 months later the jar contents were thawed and homogenized and aliquots of each egg were analysed chemically, either individually (AMCs), or as egg pools by colony (Lake Superior GLISP samples). All the chemical analyses of individual eggs were performed, using gas chromatography, by the Ontario Research Foundation, Mississauga, Ontario, according to methods described by Reynolds and Cooper (1978), and Norstrom *et al.* (1980 a,b). Up to 28 organochlorine (OC) contaminants were analysed for, but not all OCs were screened in every year and/or every sample.

The egg pools from 1983 were analysed by the Canadian Wildlife Service (CWS) at the National Wildlife Research Centre (NWRC), Hull, Quebec, for DDE, dieldrin, hexachlorobenzene (HCB), mirex, polychlorinated biphenyls (PCBs), oxychlordane, and heptachlor epoxide (HE), according to methods detailed by Peakall *et al.* (1986). The remaining egg contents were stored in the CWS National Specimen Bank, at the NWRC (Elliott, 1984). Portions of the stored samples from

1981–83 Agawa Rock eggs were analysed subsequently for five polychlorinated dibenzo-*p*-dioxin (PCDD) compounds at the NWRC laboratories, using techniques described by Stalling *et al.* (1985).

Contaminant levels were expressed in $\mu\text{g/g}$, on a wet weight basis, for all compounds except PCDDs (pg/g , wet weight). Minimum detection limits were generally $0.001 \mu\text{g/g}$ for chlorinated benzenes, and $0.01 \mu\text{g/g}$ for other OC pesticides and PCBs. For PCDDs minimum detection limits varied with the compound and the sample cleanliness, but were usually $1\text{--}2 \text{ pg/g}$ for tetrasubstituted dioxins (TCDD), $3\text{--}4 \text{ pg/g}$ for hexasubstituted dioxins ($\text{H}\times\text{Cdd}$), 6 pg/g for heptasubstituted dioxins (HpCDD), and $8\text{--}10 \text{ pg/g}$ for octasubstituted dioxins (OCDD). Eggs for which residues were below the detection limit were assigned a value of one half the minimum detection limit (O.E.C.D. 1981). PCB levels are quoted relative to a 1 : 1 mix of Aroclor 1254 : 1260 as the reference standard. DDE levels quoted are for the *p*, *p'*-homolog.

After opening, the eggshells were rinsed with water and air-dried at room temperature for about three months. The thickness of shells (including membranes), hereafter referred to as 'eggshell thickness', was measured at three points around the equator of the egg, using a Starrett dial-gauge micrometer accurate to 0.001 mm , and the mean thickness then calculated. The length (*L*) and breadth (*B*) of eggs collected in 1983 from Gull I. and Knife I. was measured with Vernier calipers (accurate to 0.1 mm), and a volume index calculated ($L \times B^2$).

3.2. POPULATIONS AND REPRODUCTION

The number of Herring Gull nests was often estimated during the 1970s and early 1980s using techniques such as aerial estimation of nest/territory number, or dividing the number of adults seen by two (Scharf *et al.*, 1978; Blokpoel *et al.*, 1980). We consider the margins of survey error to be unacceptably large for these techniques, and so we considered only ground counts of Herring Gull nests as reliable indicators of colony size. Colonies were visited during the late incubation period (15 May–6 June were the extreme dates) and the number of 'Apparently Occupied Nests' (AONs), and their contents, determined. Old nest-structures which had not been clearly modified during the current year were not scored as AONs, whereas freshly built-up and/or cupped nests were counted, irrespective of their contents. Reproductive output was determined by two subsequent visits to colonies when chicks would have reached a median age of about 21 d, according to methods described by Mineau and Weseloh (1981). It proved unnecessary to fence groups of nests since colonies were sufficiently discrete to permit estimation of chick production using the mark-recapture technique.

3.3. DATA ANALYSES

To facilitate comparisons with other published accounts of contaminants in Herring Gull eggs, as well as fish species, the residue levels for AMCs are presented as arithmetic means. However, for some compounds these individual egg residues were

significantly skewed to the right in their frequency distributions [Kolmogorov-Smirnov One-Sample Test, using Lilliefors 2-Tailed Probability Test (Wilkinson, 1989)]. Therefore, parametric statistical tests were carried out on $\log_{10}$ -transformed data, thereby meeting more closely the assumptions of those tests. As a result of this skewness, geometric mean levels for most contaminants were up to 30% lower than the quoted arithmetic means, but the difference was even greater for mirex (57%) and tetrachlorobenzenes (70%).

Unless stated otherwise, arithmetic means are quoted $\pm$ one standard deviation. The term 'significant' is used in its statistical sense only, statistical significance being accepted at the $P = 0.05$ level. The number of degrees of freedom is given as a subscript to the statistic quoted. Statistical methods and conventions follow those of Sokal and Rohlf (1981) and Rohlf and Sokal (1981). Computation utilised the Systat software packages (Wilkinson, 1989).

4. Results

4.1. CONTAMINANT LEVELS IN 1983

In 1983 the following seven main contaminants were detected at each of the eight colonies sampled: DDE, dieldrin, HE, HCB, mirex, oxychlordane and PCBs (Table I). Levels of these compounds varied significantly amongst colonies (Friedman's test for randomised blocks, $\chi^2 = 45.27$, 6 d.f., $P = 0.0001$). There was a significant inverse correlation ($P < 0.05$) between levels of mirex and both oxychlordane and HE. There were significant positive correlations ($P < 0.05$) for the following pairs of contaminants: DDE-HE, DDE-oxychlordane; HCB-PCBs, HCB-dieldrin; and dieldrin-PCBs. Lipid levels in egg pools ranged between 7.2% and 8.5% of wet weight. Contaminant levels were not significantly correlated with the percentage lipid in eggs.

To obtain an overall index of egg contamination for each colony in 1983 (Figure 2), we used levels of seven main compounds: DDE, dieldrin, oxychlordane, HE, HCB, mirex, and PCBs. For each colony and compound, the level was expressed as a proportion of the highest level in any egg pool for that contaminant. These proportions were then summed for each colony, to provide a composite contaminant index (CCI) for each colony of levels of these seven contaminants, and these indices were then presented as pie diagrams. For example, Gull I. had the highest levels for six of these compounds, scoring 1.0 for each, but mirex levels were 0.43 of the maximum recorded in 1983 (from Leadman I.). This resulted in a total score for Gull I. of $(6 \times 1.0) + (1 \times 0.43) = 6.43$. Since this was the highest colony index it was shown as a filled circle, and indices for other colonies were shown as proportions of this value.

Eggs from Gull Island and Leadman Island exhibited the highest levels for most OCs, and Chêne Island the lowest, but the overall range of variation was not great (Table I; Figure 2). Levels of most contaminants varied amongst the colonies sampled, by a factor of only about two: DDE (1.7), dieldrin (2.1), HCB (2.0),

TABLE I

Levels of selected organochlorine contaminants in fresh Herring Gull eggs on Lake Superior, from eight colonies in 1983, and Granite Island in 1982. Residue levels are given as $\mu\text{g/g}$, on a wet weight basis.

# ^a	Colony Name	# of eggs		% lipid	DDE	HCB	Mirex	PCBs	Oxy-chlordane	HE	dieldrin
1	Chêne I.	10		8.49	2.74	0.053	0.126	14.5	0.134	0.133	0.247
10	Leadman I.	10		8.48	4.00	0.070	0.244	20.0	0.228	0.159	0.460
13	Papoose I.	10		7.76	3.46	0.052	0.051	15.7	0.258	0.183	0.376
14	Knife I.	10		8.33	3.84	0.052	0.032	16.7	0.288	0.173	0.224
15	Gull I.	10		8.40	4.65	0.081	0.106	28.4	0.321	0.252	0.471
16	Lake Linden	12		7.16	3.78	0.041	0.093	15.1	0.228	0.167	0.301
17	Huron I.	12		7.38	3.48	0.049	0.146	15.7	0.194	0.155	0.320
4	Agawa Rk.	12 ^b	$\bar{x}$	8.93	3.24	0.055	0.152	21.1	0.268	0.115	0.329
			sd	0.69	1.08	0.059	0.051	18.7	0.114	0.056	0.125
11	Granite I.	10 ^b	$\bar{x}$	8.45	6.75	0.074	0.220	39.7	0.335	0.201	0.435
			sd	1.19	4.75	0.015	0.298	24.5	0.153	0.090	0.231

^a Colony numbers refer to locations shown in Figure 1.
^b Eggs from Agawa Rk. and Granite I. Annual Monitoring Colonies were analysed individually (arithmetic means levels $\pm$ 1 s.d. are quoted) whereas other samples were analysed as pools.

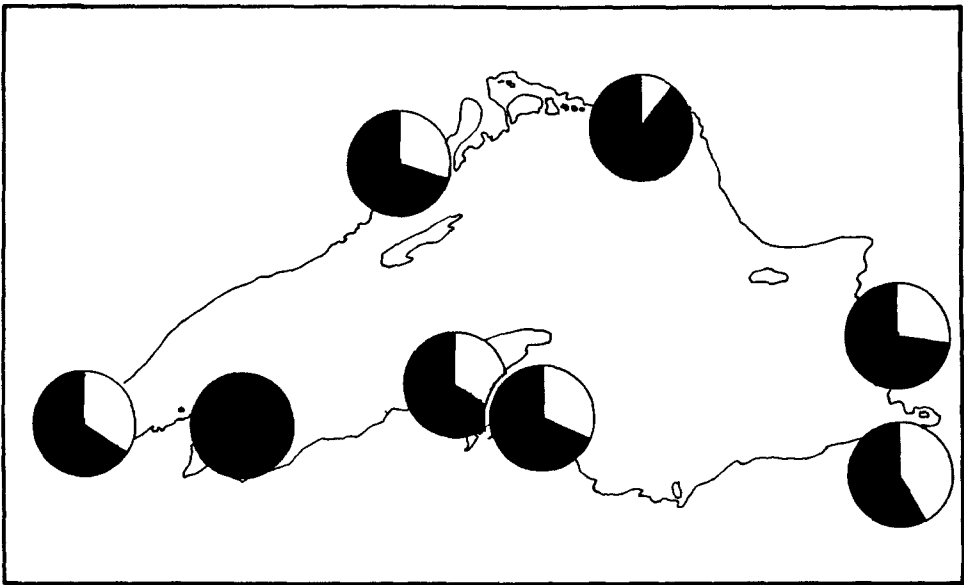


Fig. 2. Geographical variation in egg contaminant levels at eight Herring Gull colonies in 1983. The filled section of each pie is proportional to a composite index of seven main contaminants expressed as a percentage of the most contaminated colony (Gull I.). See text for further explanation of index calculation.

TABLE II

Levels of additional organochlorine contaminants in freshly-laid Herring Gull eggs from Lake Superior Annual Monitoring Colonies, 1981–83. Residue levels are expressed as arithmetic means (± 1 s.d.), in $\mu\text{g/g}$ on a wet weight basis, (except for PCDDs which are in pg/g , for egg pools). Samples were 10–13 eggs, except where indicated in square brackets.

Compound	Granite I.		Agawa Rk.		
	1981	1982	1981	1982	1983
α -chlordane	0.22 ± 0.06	0.02 ± 0.01 [9]	0.20 ± 0.06	0.02 ± 0.01 [7]	0.01 ± 0.01
γ -chlordane	0.07 ± 0.05	0.01 ± 0.002	0.04 [1]	0.01 ± 0.002	0.004 ± 0.001
1,2,3,4 TeCB ^a	0.02 ± 0.07	0.004 ± 0.004	0.01 ± 0.01	0.01 ± 0.004	0.002 ± 0.002
1,2,3,5/1,2,4,5 TeCB	0.01 ± 0.02	0.01 ± 0.002	0.01 ± 0.01	0.01 ± 0.004	0.002 ± 0.002
Pentachlorobenzene	0.01 ± 0.02	0.01 ± 0.01	0.01 ± 0.01	0.01 ± 0.003	0.004 ± 0.003
α -HCH ^b	na	0.003 ± 0.001	na	na	0.007 ± 0.004
β -HCH	0.02 ± 0.01	0.02 ± 0.01	0.02 ± 0.02	0.02 ± 0.01	0.01 ± 0.01
Trans-nonachlor	na	0.09 ± 0.10	na	0.16 ± 0.08	0.03 ± 0.01
2,3,7,8 TCDD ^c	na	na	79	51	13
1,2,3,7,8 PeCDD ^d	na	na	21	18	8
1,2,3,6,7,8 HxCDD ^e	na	na	30	24	8
1,2,3,4,6,7,8 HpCDD ^f	na	na	8	8	8
Octachlorodibenzo- <i>p</i> -dioxin	na	na	14	30	10

^a hexachlorocyclohexane; ^b hexachlorocyclohexane; ^c tetrachlorodibenzo-*p*-dioxin; ^d pentachlorodibenzo-*p*-dioxin;

^e hexachlorodibenzo-*p*-dioxin; ^f heptachlorodibenzo-*p*-dioxin, na = not analysed.

PCBs (2.0), oxychlordane (2.4), HE (2.2), but mirex levels ranged by a factor of 7.6. Mirex levels were lowest in eggs from the western half of Lake Superior (west of 88° W) than in the eastern colonies, but not quite significantly so (medians for five western colonies = 0.093 vs. 0.149 for four eastern colonies, Mann–Whitney *U*-test, $Z = 1.715$, $P = 0.086$). Although eggs were not collected from Granite Island in 1983, levels of all eight main contaminants in 1982 eggs there were higher than at most of the colonies sampled in 1983 (Table I). The individual egg analyses revealed relatively high variation within a sample: coefficients of variation were highest for mirex (0.34–1.35), HCB (0.20–1.07) and PCBs (0.62–0.89) (Table I).

Lower levels of an additional eight organochlorine pesticide residues were also detected in eggs from the Agawa Rock AMC in 1983 and 1982, as well as Granite Island AMC in 1982 (Table II). For eggs in 1982 there was no significant difference in levels between these two AMCs for the OC compounds detected (paired *t*-tests, $P = 0.9$). PCDD levels at Agawa Rock were highest for 2,3,7,8 TCDD, but detectable concentrations of higher chlorinated isomers were present in all samples analysed (Table II).

4.2. TEMPORAL TRENDS IN CONTAMINANTS, 1974–83

At Granite Island and Agawa Rock, the Lake Superior AMCs, Herring Gull eggs were collected and analysed annually during this 10-year period (excepting Agawa Rock in 1976 and Granite I. in 1983). Most OCs listed in Tables I and II were analysed for in these egg samples, and detectable concentrations were found in

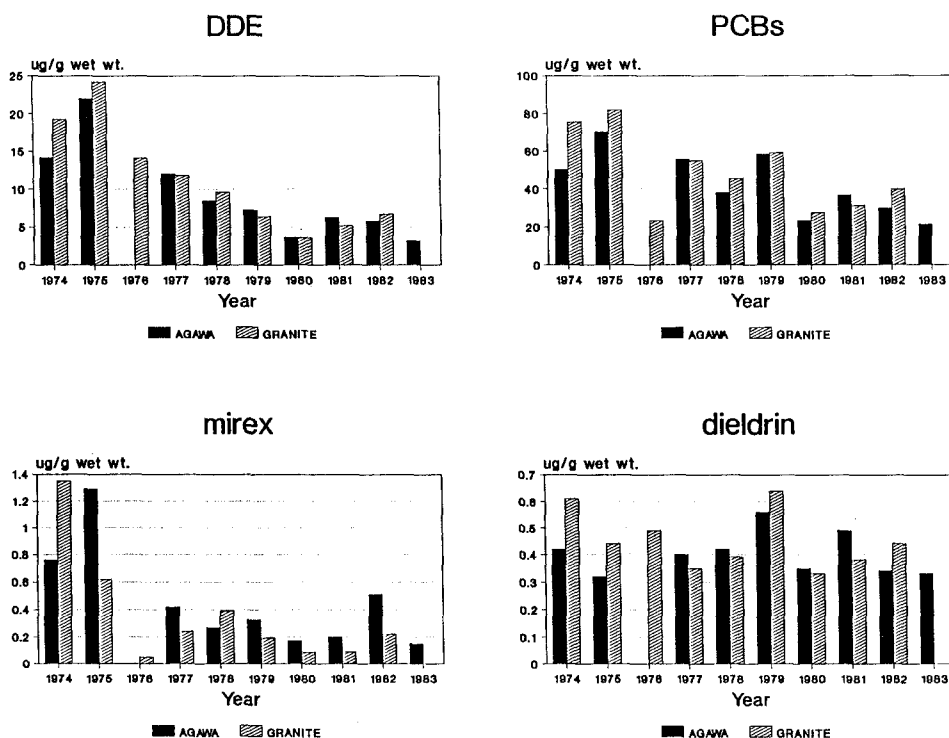


Fig. 3. Trends in levels of DDE, PCBs, mirex and dieldrin in Herring Gulls eggs from Agawa Rk. and Granite I. Annual Monitoring Colonies, 1974–83.

almost all eggs in each year. Compounds which were not detected in 100% of eggs in most years at each colony were: γ -chlordane (found in 55–100% of eggs in any given colony-year); 1,2,3,4 tetrachlorobenzene (TeCB) (30–100%), 1,2,3,5/1,2,4,5 TeCB (30–100%), alpha-hexachlorocyclohexane (α -HCH, 30–73%), and β -HCH (70–100%). The TeCBs and β -HCH occurred less frequently in eggs after 1980.

Lipid levels in eggs (expressed as percentage of wet weight) showed no significant variation amongst years, during the 1974–83 period, at either Agawa Rock ($F_{9,80} = 0.4$; overall mean $8.8 \pm 1.2\%$, range for individual eggs = 5.0–11.7%) or Granite Island ($F_{8,72} = 0.3$; overall mean $8.7 \pm 1.3\%$, range 5.1–12.7%). However, the mean lipid content of eggs at AMCs (pooled for sites and years) during the 1970s was higher than during the 1980s, but not significantly so ($8.9 \pm 0.8\%$, vs. $8.3 \pm 0.7\%$, respectively, $t_{16} = 1.6$, $P = 0.1$).

Levels of most contaminants declined in Herring Gull eggs at both AMCs over the 10-year period, the steepest declines occurring prior to 1980 (e.g. Figure 3). Linear regression analysis of log-transformed egg contaminant levels revealed significant declines ($P < 0.002$) over this period for the following four compounds at both sites (the greatest percentage decline at any one site between 1974 and 1982

TABLE III

Average population half-lives (in years) of organochlorine contaminants in Herring Gull eggs from Lake Superior Annual Monitoring Colonies which showed a significant decline over the 1974–83 period. ns = non-significant decline. Compounds shown in Tables I and II but not mentioned here did not show significant declines during the period.

Compound	Agawa Rk.	Granite I.
hexachlorobenzene (HCB)	4.3	8.6
DDE	3.6	3.4
β hexachlorocyclohexane	4.7	ns
mirex	4.4	2.2
PCBs (1254/1260 1 : 1 mix)	5.8	5.7

or 1983 is given in parentheses): DDE (77%), mirex (84%), HCB (81%), and PCBs (58%). Overall, the declines were slightly greater at Agawa Rock than at Granite Island, β -HCH showed significant decreases at Agawa Rk., but not at Granite I. No significant trend was detected for dieldrin, HE or oxychlordane at either colony, or α -chlordane at Agawa Rk. Levels of α -chlordane increased significantly at Granite I. ($P < 0.001$). Levels of PCDDs in eggs from Agawa Rock declined generally from 1981 to 1983, especially for the most toxic isomer, 2,3,7,8 TCDD (Table II). Levels of 1,2,3,4,6,7,8 heptachlorodibenzo-*p*-dioxin were close to the detection limit and showed no apparent change over this period. Seventeen of the 23 contaminants detected in samples from this period reached their lowest recorded level in eggs in 1983 (Agawa Rock) or 1982 (Granite Island).

Average population half-lives of the contaminants (Armstrong and Sloan, 1980), which measure the length of time necessary for the concentration of a compound to decrease by 50% in population samples of this kind, were calculated for each of the five compounds exhibiting a significant decreasing trend in levels at the AMCs (Table III). These half-lives, which represent the integration of a complex set of biological and chemical processes in the Herring Gull and its prey (Mineau *et al.*, 1984), ranged from a low of 2.2 years for mirex, to 8.6 years for HCB (both at Granite Island).

4.3. EGGSHELL THICKNESS

Mean eggshell thickness ranged between 0.338 mm and 0.363 mm at eight study colonies in 1983, with an overall unweighted mean thickness of 0.345 mm (Table IV). There was no significant difference in eggshell thickness amongst colonies (One-Factor ANOVA, $F_{9,70} = 1.84$, $P = 0.076$), but Gull Island eggshells averaged 4–7% thicker than those in other colonies. On average, eggshells were 8% thin-

TABLE IV

Shell thickness and percentage thinning of Herring Gull eggs from selected colonies on Lake Superior in 1983. Colonies are arranged by degree of shell thinning, and locations are shown in Figure 1.

Colony		N	Eggshell thickness (mm)		% thinning ^a
#	Name		mean $\pm$ s.d.		
15	Gull I.	10	0.363 $\pm$ 0.025		3.1
10	Leadman I.	10	0.349 $\pm$ 0.026		7.0
4	Agawa Rk.	13	0.348 $\pm$ 0.022		7.3
13	Papoose I.	10	0.345 $\pm$ 0.025		8.1
16	Lake Linden	12	0.341 $\pm$ 0.027		9.0
17	Huron Is.	12	0.340 $\pm$ 0.030		9.4
1	Chene I.	10	0.339 $\pm$ 0.024		9.7
14	Knife I.	10	0.338 $\pm$ 0.022		9.8
Mean		87	0.345 $\pm$ 0.025		8.0

^a Compared to pre-1947 mean value of 0.375 mm (Anderson and Hickey, 1972).

ner (range 3–10%) than in pre-1947 (i.e. pre-DDT) samples measured in museum collections (Anderson and Hickey, 1972). For this sample of eight colonies, mean eggshell thickness was positively correlated with contaminant residues (in egg pools), significantly so for HCB ($P = 0.008$), dieldrin ($P = 0.007$) and PCBs ($P = 0.0003$), but not quite significantly so for DDE ($P = 0.06$) or HE ($P = 0.08$). Eggshell thickness was not correlated significantly with breeding output. Egg biometrics taken at Gull I. and Knife I. in 1983, revealed that both larger eggs ($r^2 = 0.254$, $F_{1,18} = 6.11$, $P = 0.02$) and broader eggs ($r^2 = 0.356$, $F_{1,18} = 9.96$, $P = 0.006$) had significantly thicker shells.

4.4. REPRODUCTION

Average reproductive output in 1983 varied between 0.15 and 1.57 young per apparently occupied nest (Y/AON) at nine colonies in Lake Superior (Table V). At 56% of the colonies studied in 1983, productivity was below the 0.8–1.40 Y/AON level, regarded to be necessary to maintain a stable population (Kadlec and Drury, 1968; Kadlec, 1976). For six colonies contaminant levels in eggs (pools) were also determined, and rank correlation analyses performed with reproductive data. All correlations were in a negative direction (colonies with higher levels of contaminants in eggs having lower reproductive output). The relationship was significant only for mirex (Kendall's $T = -0.73$, $P = 0.039$), but that for PCBs approached significance ($T = -0.69$, $P = 0.052$).

Reproductive output at Lake Superior Herring Gull colonies was measured at a total of 17 colonies in the period 1975–84 (Table V). These data indicated a general

TABLE V

Reproductive output of Herring Gulls at selected Lake Superior colonies between 1975 and 1984. Values given are mean numbers of young reaching 21 d of age per Apparently Occupied Nest (AON). Colony locations are shown in Figure 1.

#	Colony	# of AONs ^a	1975 ^b	1976 ^b	1978	1979	1980	1981	1982	1983	1984
	name										
1.	Chêne I.	195	—	—	—	—	—	—	—	1.57	—
2.	Sawpit Bay	38	—	—	—	—	—	0.89	—	—	—
3.	Cozen's Cove	326	—	—	—	—	—	0.77	0.74	1.19	—
4.	Agawa Rk.	232	—	—	1.66	0.88	0.40	0.37	0.14	0.37	0.85
5.	Telegraph Rk.	49	—	—	—	—	—	0.15	0.07	0.46	—
6.	S. Devil's W'h. I. ^c	40	—	—	—	—	—	0.48	—	—	—
7.	Entrance I.	322	—	—	—	—	—	1.04	—	—	—
8.	S. Dore Bay	191	—	—	—	—	—	0.52	—	—	—
9.	E. Dog Harbour	30	—	—	—	—	—	0.85	—	—	—
9.	Leadman I.	134	—	—	—	—	—	—	—	0.15	—
11.	Granite I.	150	1.32	1.55	1.12	1.77	1.40	0.46	—	—	—
12.	Silver Islet	45	—	—	—	—	—	—	1.39	1.36	1.39
13.	Papoose I.	168	—	—	—	—	—	—	1.82	0.94	—
15.	Gull I.	189	—	—	—	—	—	—	—	—	1.43
16.	Lake Linden	198	—	—	—	—	—	—	—	0.61	—
17.	Huron Is.	485	—	—	—	—	—	—	—	0.47	—

^a the most recent ground census for each site: colonies 1,4,5,6,7,12 = 1983 (this study);

2,3,8,9,10,13 = 1981 (CWS, unpubl. data); 11 = 1978 (CWS, unpubl. data); 14–17 =

1976 (Scharf *et al.*, 1978).

^b data from Ryder and Carroll (1978). ^c South Devil's Warehouse Island.

decline in gull productivity over this period, although a few colonies (such as Silver Islet and Papoose Island) appeared to have maintained reasonably high levels of chick production into the 1980s. Over this 10-year period annual production of young at both AMCs declined, but not significantly so (linear regressions, $P > 0.2$). When this period was divided into two five-year periods, and productivity figures for AMCs analysed together, mean reproductive output was found to be significantly lower in the 1980–84 period (0.57 ± 0.42 Y/AON) than in the 1975–79 period (1.38 ± 0.34 Y/AON; $t_{11} = 3.77$, $P = 0.003$), an average decline of 59%. Egg lipid levels were significantly correlated with reproductive output at Agawa Rk. ($r^2 = 0.66$, $F_{1,4} = 7.63$, $P = 0.05$) but not at Granite I. ($F_{1,4} = 0.04$), nor when all AMC data were pooled ($F_{1,10} = 0.07$).

The size of completed clutches at both AMCs and four other colonies was determined in some years between 1975 and 1984. Means varied between 2.99, at Granite I. in 1975, and 2.16 at Island east of False Dog Harbour in 1981 (Table VI). Correlation analysis of mean clutch size against date revealed a significant decline in clutch size over the period at the two AMCs ($r^2 = 0.61$, slope = -0.062 ± 0.018 s.e., $t_8 = 3.59$; $F_{1,8} = 12.54$, $P = 0.008$), and also for all six colonies combined ($r^2 = 0.36$, slope = -0.053 ± 0.02 s.e., $t_{12} = 2.62$; $F_{1,12} = 6.87$, $P = 0.022$). Similarly, the average clutch size for these colonies in the five-year period 1975–79 was significantly greater than during the 1980–84 period (2.66 ± 0.21 vs.

TABLE VI

Herring Gull clutch sizes at Lake Superior colonies, 1975–84. Data for 1975 and 1976 are from Ryder and Carroll (1978).

# ^a	Colony Name	Year	Number of nests with clutch of:					Mean clutch $\pm$ s.e.
			0	1	2	3	4	
11	Granite I.	1975	0	0	2	97	1	2.99 $\pm$ 0.02
		1976	0	0	2	98	0	2.98 $\pm$ 0.01
		1978	0	7	30	92	0	2.66 $\pm$ 0.05
		1979	8	8	14	103	0	2.59 $\pm$ 0.07
		1980	4	1	14	91	0	2.75 $\pm$ 0.06
		1981	14	4	19	72	1	2.38 $\pm$ 0.10
4	Agawa Rk.	1978	2	3	28	93	0	2.68 $\pm$ 0.05
		1979	14	4	27	84	0	2.40 $\pm$ 0.09
		1982	4	10	53	51	0	2.28 $\pm$ 0.07
		1983	6	9	35	86	0	2.48 $\pm$ 0.07
		1984	10	6	35	80	0	2.41 $\pm$ 0.08
8	S. Dore Bay	1981	2	2	15	10	1	2.20 $\pm$ 0.16
9	E Dog Harb.	1981	17	2	27	54	0	2.16 $\pm$ 0.11
5	Telegraph Rk.	1983	3	2	18	26	0	2.37 $\pm$ 0.12
12	Silver Islet	1984	3	3	9	46	0	2.61 $\pm$ 0.10

^a Colony reference numbers refer to locations shown in Figure 1.

2.40 $\pm$ 0.19 respectively, $t_{12} = 2.36$, $P = 0.036$). However, analysing the most recent data for six colonies in the 1981–84 period (given in Table VI), revealed a significant difference in clutch size amongst colonies ($\chi^2 = 26.91$, 10 d.f., $P = 0.003$), suggesting that local factors may have been important in reproductive performance.

4.5. POPULATION TRENDS

Reliable, accurate surveys of Herring Gull nests were only available for a few sites in eastern Lake Superior. For colonies surveyed by systematic ground coverage during the late incubation period in eastern Lake Superior (between Chêne Island and Leadman Island), overall numbers of AONs increased by an average of 4.1% per annum between 1978 and 1983/84, with average rates of change at individual colonies varying from -5.9% to $+4.4\%$ per annum (Table VII). Since Blokpoel *et al.* (1980) estimated 6410 nests of Herring Gulls in the Canadian portion of Lake

TABLE VII

Numbers of apparently occupied Herring Gull nests (AONs) and mean rates of population change at selected colonies in eastern Lake Superior, 1978–84.

Colony name ^a	Number of AONs						Mean annual % change ^b
	1978	1979	1980	1981	1983	1984	
Chêne I.	189	—	—	—	195	—	+0.6
I. W of Chêne I.	253	—	—	—	313	—	+4.4
Is. S of E end of Batchawana I.	161	—	—	—	185	—	+2.8
S tip of S Sandy I.	61	—	—	—	45	—	–5.9
N of N Sandy I.	96	—	—	—	107	—	+2.2
Barrett I.	128	—	—	104	157	—	+4.1
Agawa Rk.	247	—	—	235	232	—	–1.2
Vrooman I.	175	—	—	211	201	—	+2.8
Pukaskwa National Park ^c	721	784	921	754	—	1045	+4.1
Total ^c	2031			2480		1045	+4.1

— no count made. ^a Colony names follow those given in Blokpoel *et al.* (1980).

^b Taken as the difference between the most recent count (RC) and the 1978 count (IC), and calculated as $[(n\sqrt{RC/IC} - 1) \times 100]$

^c 1978 data from Blokpoel *et al.* (1980); other years = Pukaskwa National Park (unpubl. data).

For the purposes of 1978 and 1983 pooled totals, the 1984 count was regarded as 1983 data.

Superior in 1978, our sample of 2031 nests in 1978 was a good representation (32%) of this population.

5. Discussion

5.1. TEMPORAL TRENDS IN CONTAMINANTS

A total of 23 xenobiotic polyhalogenated aromatic hydrocarbons was detected in Herring Gull eggs collected from Lake Superior colonies in 1983. When analysed for, these OCs have been detected in virtually all eggs from the two colonies/areas monitored annually since the program began in 1974. Although the frequency of occurrence of these contaminants in eggs had not diminished by 1983, mean levels of the main compounds (DDE, mirex, HCB and PCBs) declined significantly over the 10-year period, by up to 97%. We have no contaminant data for Lake Superior eggs prior to 1970s, but it is likely that DDE residues were even higher then, since DDE levels in large Lake Trout (*Salvelinus namaycush*) declined by a factor of two between 1965–68 and 1972 in western Lake Superior (Veith *et al.*, 1977). In northern Lake Michigan, Herring Gull eggs had very high mean levels of DDE in 1963 (97 $\mu\text{g/g}$, wet weight) and 1964 (202 $\mu\text{g/g}$) (Keith, 1966).

The only comparable data for OC trends in other biota in Lake Superior during the 1974–84 period are for fish. Mean PCB levels in young-of-the-year Spottail Shiners (*Notropis hudsonius*), at two sites along the northern shoreline, increased at least four-fold between 1979 and 1983, whereas Σ DDT and Σ chlordanes concentrations changed little (Suns *et al.*, 1991). In Lake Trout from the Apostle Islands (the archipelago near colony 15 in Figure 1), Σ DDT residues declined significantly between 1977 and 1982 (by 66% on average, from 1.2 to 0.4 $\mu\text{g/g}$), but PCBs, dieldrin and oxychlordanes did not (DeVault *et al.*, 1986). Similarly, total PCB residues in Lake Trout from Lake Superior aged 4+ years changed little between 1980 and 1983 (or 1985), remaining below 1 $\mu\text{g/g}$ (Baumann and Whittle, 1988). DDE residues in Rainbow Smelt (*Osmerus mordax*), an important food item of Herring Gulls in Lake Superior and other parts of the Great Lakes (Fox *et al.*, 1990), changed little from 1981 to 1983 (0.6 to 0.5 $\mu\text{g/g}$), but total PCB levels increased from 0.06 to 0.23 $\mu\text{g/g}$ (International Joint Commission, 1985). In summary, from the 1970s to the early 1980s, Σ DDT residues in Lake Superior fish appeared to decline dramatically, but other contaminants fluctuated around relatively low levels, and some species showed increases in total PCB residues.

It is now generally accepted that atmospheric deposition represents the main input of OC pesticides and PCBs into Lake Superior, and probably accounted for at least 90% of the annual input during the 1970s (Swain, 1978; Eisenreich *et al.*, 1979, 1981b, 1983; Bahnick and Markee, 1985). Relative to the other North American Great Lakes, Lake Superior covers a high proportion (64%) of its total drainage basin area (Eisenreich *et al.*, 1981a), and so direct atmospheric deposition of contaminants onto the lake's surface will be of greater importance than for other lakes. Further, the hydraulic residence time is long – 180 years, as opposed to three years for Lake Erie, for example (Eisenreich *et al.*, 1980). These physical factors, along with resuspension of sediments by severe weather conditions, contribute to the relatively long residence time of trace organic contaminants in the water column (for example, 5.4 years for PCBs) (Eisenreich *et al.*, 1980). The rates of decline of contaminant concentrations in fish (DeVault *et al.*, 1986; Suns *et al.*, 1991) and Herring Gull eggs (Bishop *et al.*, 1992), following reductions in environmental discharges, have been lower in Lake Superior than further south in the Great Lakes basin. Despite major reductions in PCB production and usage during the 1970s, atmospheric input of contaminants to Lake Superior remained elevated into the 1980s, and showed no signs of reduction (Eisenreich *et al.*, 1983). This was broadly reflected in the trends in OC levels outlined above for fish and Herring Gulls. Since 1980, contaminant levels in Herring Gull eggs from Lake Superior appear to have declined relatively little. In an overall Great Lakes context in 1983, Herring Gull eggs from these Lake Superior colonies contained relatively low levels of PCBs, HCB, and 2,3,7,8-TCDD, but similar levels of DDE and mirex to other lakes (excepting mirex on Lake Ontario), and higher dieldrin levels than in all other lakes except Lake Michigan (Table VIII).

TABLE VIII

Range of arithmetic mean colony OC residues in fresh Herring Gull eggs in different parts of the Great Lakes in 1983. Data for lakes other than Superior extracted from Bishop *et al.*, 1992.

Lake	No. of colonies	$\mu\text{g/g}$, wet weight				pg/g	
		DDE	PCBs	Mirex	Dieldrin	HCB	2,3,7,8 TCDD
Superior	8	3–5	15–28	0.03–0.24	0.22–0.47	0.04–0.08	13 ($n = 1$)
Michigan	2	6–7	28–33	0.04–0.06	0.53–0.68	0.03–0.07	10–18 ^a
Huron	3	2–6	17–49	0.07–0.16	0.16–0.25	0.05–0.14	15–141
Erie	3	2–3	37–65	0.04–0.29	0.17–0.24	0.05–0.56	9–32 ^a
Ontario	3	4–5	34–46	0.33–1.50	0.14–0.22	0.07–0.09	19–90

^a Data for 1984.

5.2. SPATIAL TRENDS IN CONTAMINANTS

The spatial variation in egg residues we did detect in 1983 was probably due in part to the influence of factors such as seasonal fluctuation in sediment resuspension, caused by wind and current action (Capel and Eisenreich, 1985), and variation in the patterns of precipitation and water movement in different parts of Lake Superior. Eisenreich *et al.* (1980) and Frank *et al.* (1980) found marked geographical variation in concentrations of total PCBs and DDE in the top 0.5 cm of surficial sediments. DDE concentrations were highest in depositional basins (as opposed to non-depositional zones) in northern parts of the lake, near Isle Royale and Thunder Bay, whereas PCBs were most concentrated in the western arm (Duluth Basin), as well as southwest of Isle Royale. However, some levels differed greatly even over very short distances, and could not be explained by point sources of municipal and industrial effluents from the major harbours (e.g. Glass and Poldoski, 1974).

In general, egg residues differed relatively little among colonies on Lake Superior, compared to similar analyses for Lake Erie (Weseloh *et al.*, 1990), Lake Huron (Ewins *et al.*, 1992), or the Detroit and Niagara Rivers and Saginaw Bay (Struger *et al.*, 1985), where there were some major point sources of contaminants. Thus, the spatial distribution of PCBs and some other organochlorine contaminants in Herring Gull eggs during the 1970s and early 1980s did not correspond well to their distribution in sediments in Lake Superior.

In 1983, levels of the most toxic PCDD congener, 2,3,7,8, TCDD, in Herring Gull eggs were declining in Lake Superior, and concentrations were similar to, or slightly lower than, those in eggs from Lakes Michigan, Huron and Erie, but considerably lower than those in Lake Ontario or Saginaw Bay, Lake Huron (Mineau

et al., 1984; Stalling *et al.*, 1985). Similarly, Baumann and Whittle (1988) found that total PCDD and polychlorinated dibenzofuran (PCDF) concentrations in whole fish from the central and western parts of Lake Superior in the early 1980s were amongst the lowest recorded on the Great Lakes.

Further interpretation of spatial differences in Herring Gull egg contaminants on Lake Superior is difficult without a better understanding of the wintering areas of adults breeding on different parts of the lake. Gilman *et al.* (1977), in an analysis of band recoveries up to 1973, found that about 20% of Lake Superior birds which were recovered as adults outside the breeding season, were recovered on Lake Michigan, about 40% on Lake Superior itself (though not in mid winter), and the remaining 20% scattered around other Great Lakes and inland in North America. It is likely that most Lake Superior adults winter on Lake Michigan or the Lake Erie area, rather than Lakes Ontario or Huron (Moore, 1976). Unfortunately, there are no subsequent, or more detailed, analyses of movements of Lake Superior Herring Gulls.

Although mirex residues occur in biota throughout the Great Lakes, the only point sources are on Lake Ontario and the Niagara River (Weseloh *et al.*, 1979; Government of Canada, 1991). Our finding of higher mirex levels in eggs from eastern, as opposed to western, Lake Superior colonies would be consistent with a longitudinal difference in wintering areas. The highly variable mirex levels in the Granite I. egg sample may reflect wintering by a small number of those breeders in Lake Ontario or along the Niagara River. Similarly, one would expect birds staging or wintering in Green Bay, or other more heavily contaminated parts of Lake Michigan, to have relatively high egg residues for compounds other than mirex (e.g. Heinz *et al.*, 1985; Government of Canada, 1991).

Another confounding factor in interpreting egg contaminant patterns in different areas of Lake Superior is the occurrence of local dumps of entrails from large fish discarded by commercial fishermen (Kozie and Anderson, 1991). We do not have information on the distribution of these dumps, but consumption of livers and other tissues of large fish by gulls during the egg-formation period would clearly represent a major source of contaminants which may enter the egg.

5.3. POPULATION TRENDS AND REPRODUCTION

Using the number of nests as an indication of the size of a breeding population, the Herring Gull population in eastern Lake Superior appeared to be increasing from 1978 to 1984 at an average rate of 4% per annum. Ground censuses of Minnesota colonies, in the western arm of Lake Superior, between 1977 and 1979 recorded increases of 6–15% per annum (Goodermote, 1980). Although no other reliable census data are available from other parts of the lake, the indications are that Herring Gull numbers were increasing throughout much of Lake Superior between the mid-late 1970s and the mid 1980s.

Since mean DDE levels in Herring Gull eggs were as high as 15–25 $\mu\text{g/g}$ in 1974–75 in Lake Superior, and likely even higher prior to that (see earlier

discussion), it is possible that reproductive output at Lake Superior colonies was low during the 1960s, mainly due to egg breakage and embryonic mortality caused by DDE, but perhaps also due to poorly understood effects of other contaminants. This may have led to population declines, assuming that recruitment was not bolstered by immigrants from less contaminated areas. Thus, it is possible that the declines in DDE residues during the 1970s, which resulted in thicker eggshells and improved reproductive output, led to higher rates of recruitment and overall population increase. Unlike many other piscivorous birds and raptors, Herring Gulls appear to be fairly insensitive to the effects of DDE, and egg concentrations of about 160 $\mu\text{g/g}$ are reported to be required before one finds 'critical' eggshell thinning of 15–20%, and elevated risks of egg breakage during incubation (Keith and Gruchy, 1972). At Lake Michigan colonies in 1964, eggshells averaged 31% thinner than pre-DDT values, associated with average DDE residues of 202 $\mu\text{g/g}$ wet weight (Keith, 1966; Anderson and Hickey, 1972). In 1976, reproductive output at Granite Island was relatively high, at 1.55 chicks fledged per nest, associated with DDE residues in eggs of $14.1 \pm 5.2 \mu\text{g/g}$ (CWS Ontario Region, unpubl. data). In 1983, DDE residues of 2.7–4.7 $\mu\text{g/g}$ were associated with shell thinning of 3–10% overall, and we found no evidence of reproductive failure due to broken, flaked or dented eggs. Since levels of different contaminants are often highly correlated in eggs of fish-eating birds (e.g. Gilman *et al.*, 1977; Weseloh *et al.*, 1990; Bishop *et al.*, 1992), shell thickness may appear to be related to levels of various toxic chemicals although it is DDE which exerts the direct influence on shell structure (Cooke, 1973).

Despite reductions in organochlorine contaminant levels in Herring Gull eggs on Lake Superior during the 1974–84 period, reproductive output declined. If we assume that other demographic parameters were similar on Lake Superior and on the U.S. Atlantic coast, then the calculations of Kadlec and Drury (1968) and Kadlec (1976) indicate that Lake Superior birds were producing insufficient young during the early 1980s to balance adult mortality. Similar low rates of chick production were seen during the early 1970s in Lake Ontario colonies, associated with markedly higher levels of most contaminants, and a range of adverse physiological and behavioural effects were discovered in eggs, chicks and adults (e.g. Gilbertson, 1974; Gilbertson and Hale, 1974; Gilbertson and Fox, 1977; Fox *et al.*, 1978). However, little evidence for contaminant-related biological effects of this kind has been found on Lake Superior (Ryder and Carroll, 1978; CSS unpubl. data; this study). Further, numbers of breeding birds appeared to be increasing throughout the lake during this period.

5.4. THE ROLE OF FOOD AVAILABILITY

Detailed studies in eastern Lake Superior since 1978 (CWS Ontario Region, unpubl. data; McNicholl *et al.*, 1983) found that egg loss (not breakage due to thin shells) was an important cause of breeding failure at colonies in northeastern parts of the lake, whereas chick disappearance prior to fledging was common at all colonies studied.

These causes of failure were recorded, but at much lower levels, on Granite Island in 1975 and 1976 (Ryder and Carroll, 1978; Gilman *et al.*, 1977). Our analysis of clutch sizes in 1983 probably reflects some egg losses prior to the date of census as well as differences in food availability and perhaps age structure of the breeding population. However, reductions in mean completed clutch size have been found in years of food scarcity, during detailed studies of Western Gulls (*Larus occidentalis*) in California (Ainley and Boekelheide 1990).

Observations on Agawa Rock in 1982 and 1983 revealed that chicks were frequently cannibalised by neighbouring breeding adults (McNicholl *et al.*, 1983). Many pairs were absent from the territory for unusually long periods during the nestling stage, presumably due to difficulties in finding sufficient food. Cannibalism appears to have been a profitable foraging strategy for some specialist adults which remained at the colony. When fed, chicks were presented with mostly non-fish items such as invertebrates, passerine birds and garbage, which almost certainly provided much lower energy and nutritive returns than local forage fish such as Rainbow Smelt, Alewife (*Alosa pseudoharengus*), and Yellow Perch (*Perca flavescens*) (Allan, 1977; Pierotti and Annett, 1987; Fox *et al.*, 1990). Chick growth rates and fledging success have been found to correlate positively with the proportion of energy-rich fish in the diet of Western Gulls (Hunt and Hunt, 1976; Ainley and Boekelheide, 1990), and in Herring Gulls overall reproductive output was lowest for those birds specialising on garbage items (Pierotti and Annett, 1987).

Thus, our results suggest that forage fish may have become less readily available to Herring Gulls after the 1970s, at least in eastern Lake Superior. At Granite Island fresh eggs contained 13% less lipid during 1980–83 ($7.9 \pm 1.3\%$) than in the 1970s ($9.1 \pm 1.1\%$) ($t_{87} = 4.5$, $P = 0.0001$), indicating that lipid-rich foods may have been in short supply for these birds in the early 1980s, at least during the egg formation period. At Agawa Rock/Mamainse Harbour, however, there was an average decline of only 1% in egg lipid content over the same period ($8.75 \pm 1.0\%$ and $8.84 \pm 1.4\%$, respectively, $t_{90} = 0.4$). Since eastern and northeastern regions of Lake Superior appear to have much lower primary production, phytoplankton biomass and chlorophyll A concentrations than southern and western parts (El-Shaarawi and Manuwar, 1978), one might expect to see the effects of food stress first amongst terminal predators in such areas.

Similar reproductive phenomena have been observed in gull populations elsewhere. A declining population of Lesser Black-backed Gulls (*L. fuscus*) in the Gulf of Finland appeared to have difficulty finding adequate food supplies, and 16% of eggs were eaten by the territory occupants, and, on average, 86% of chicks died before fledging, mainly due to starvation, or predation by Herring Gulls (Hario, 1990). Studies in Scotland (Parsons, 1971), and the U.S. (Hunt, 1972), have also found high incidences of cannibalism and low breeding output of Herring Gulls, especially when chicks were left unattended whilst both parents searched for scarce food resources.

Fisheries data for Lake Superior lend support to the hypothesis that Herring Gulls could have experienced food shortages there after the 1970s. The Rainbow Smelt has been the main forage fish species in Lake Superior since the 1940s/50s, when Lake Trout and Lake Herring (*Coregonus artedii*) stocks declined dramatically (Heist and Swenson, 1983; Hartman, 1988), largely due to over-exploitation (Selgeby, 1982). The multiple introductions of exotic salmonid species as sport fish (e.g. Bagdovitz *et al.*, 1986), and successful measures to control the parasitic Sea Lamprey (*Petromyzon marinus*) since the 1950s (Smith and Tibbles, 1980; Hartman, 1988), undoubtedly had a major impact on the structure of the fish community in Lake Superior. Although predation on Rainbow Smelt by salmonids increased dramatically during the 1960s and 1970s, and commercial yields of Rainbow Smelt had declined by 1981 to less than 20% of the 1976 catch, the reasons for this crash in smelt numbers are poorly understood (Hartman, 1988). In southern parts of the lake, Rainbow Smelt stocks declined slightly in 1973–78, but sharply after 1978 (J.H. Selgeby, *in litt.*).

The timing of these major declines in Rainbow Smelt populations matched closely the reductions in reproductive output of Herring Gulls in Lake Superior. It seems likely that reductions in contaminant concentrations, particularly DDT and its metabolites, in Herring Gulls during the 1970s, permitted some recovery of populations which may have been adversely affected by higher residues during the 1960s. During the early 1980s though, food shortages probably exerted a greater proximate effect than contaminant burdens on breeding productivity of Herring Gulls in Lake Superior.

Acknowledgements

We are grateful to Larry Benner, Jim Brown, Tom Clark, John Elliott, Jim Ludwig, Jan McDonald, Martin McNicholl, Yves Ouelette, and Stan Teeple, for assistance with the fieldwork. Additional logistical support was provided by the Technical Operations Division of the National Water Research Institute at the Canada Centre for Inland Waters. Lincoln Reynolds, at the Ontario Research Foundation, conducted the chemical analyses of eggs from the Annual Monitoring Colonies. At the National Wildlife Research Centre Mary Simon and Michael Mulvihill conducted the PCDD analyses, and C. McConnell analysed the 1983 egg pools for OCs. Hugh Boyd, Keith Marshall, Iola Price, Richard Turle, and an anonymous referee improved earlier versions of the manuscript.

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