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Abstract

The Great Lakes of North America are one of the most heavily contaminated freshwater ecosystems in the world, yet they support at least one-third of North America's breeding Caspian Terns *Sterna caspia*. To assess the health of this breeding population in relation to organochlorine contaminant (OC) levels, we studied 16 colonies (6800 nests) throughout its Great Lakes range in 1991. A wide range of OCs was detected in eggs from each colony, but at markedly lower concentrations than recorded in previous studies. Eggs from Saginaw Bay (Lake Huron) and Green Bay (Lake Michigan), both industrialized areas, were the most heavily contaminated. Eggshells were as thick as pre-DDT. Clutch size, hatching success, and overall reproductive output were high relative to other studies, and we found little evidence for adverse biological effects, at the population level, of current contaminant loads. Wasting syndrome was identified in chicks at Saginaw Bay, associated with the highest TCDD toxic equivalent (TEQ = 1.6 $\mu g \cdot kg^{-1}$ wet weight) in egg pools from 10 Great Lakes colonies. Diet differed among lakes, although alewife *Alosa pseudoharengus* and Centrarchids dominated overall. Since the 1960s, Centrarchids have replaced alewife as the main prey of Caspian Terns in Lake Huron and at some other sites, broadly reflecting major changes in Great Lakes fish communities. The Great Lakes breeding population of Caspian Terns has at least doubled since the late 1970s, with average annual increases higher on Lake Ontario (23%) than on Lake Michigan (6%) or Lake Huron (0.3%). Contaminants now seem to exert a relatively small influence on this population.

Résumé

Les Grands Lacs sont au nombre des écosystèmes d'eau douce les plus pollués du monde et pourtant, on y retrouve au moins le tiers de la population nicheuse de Sternes caspiennes *Sterna caspia* de l'Amérique du Nord. En 1991, pour évaluer l'état de santé de cette population reproductrice en fonction de la concentration des polluants organochlorés (OC), on a étudié 16 colonies (6 800 nids) réparties dans l'ensemble de l'aire de distribution de cette espèce dans la région des Grands Lacs. On a détecté une grande variété d'OC dans les oeufs de chaque colonie, mais à des concentrations nettement plus faibles que les valeurs mesurées dans les études antérieures. Les oeufs de la baie Saginaw (lac Huron) et de la baie Green (lac Michigan), deux endroits situés en région industrialisée, étaient les plus contaminés. L'épaisseur des coquilles était comparable à celle des oeufs produits avant la contamination au DDT. La taille des couvées, le taux d'éclosion et le taux de reproduction global étaient élevés, par comparaison aux valeurs observées dans d'autres études; en outre, à l'échelle de la population, on n'a guère trouvé de signes dénotant des effets biologiques délétères attribuables aux charges actuelles de contaminants. On a observé le syndrome cachectique chez des petits de la baie Saginaw, en association avec l'équivalent toxique de TCDD le plus élevé (ÉT = 1,6 $\mu g \cdot kg^{-1}$ en poids humide) dans des oeufs éclos provenant de 10 colonies des Grands Lacs. L'alimentation des sternes différait d'un lac à l'autre, mais, globalement, le gaspareau *Alosa pseudoharengus* et les Centrarchidae dominaient. Depuis les années 1960, le gaspareau, auparavant la principale proie de la Sterne caspienne, a été remplacé par les Centrarchidae dans le lac Huron et ailleurs, phénomène où l'on voit une indication générale des grands changements survenus dans les communautés ichtyennes des Grands Lacs. La population nicheuse de Sternes caspiennes a au moins doublé dans les Grands Lacs depuis la fin des années 1970; on a observé des augmentations annuelles moyennes plus élevées dans le lac Ontario (23 %) que dans le lac Michigan (6 %) ou dans le lac Huron (0,3 %). Il semble que maintenant, les polluants n'aient qu'une influence relativement faible sur cette population.

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1. Introduction

The Great Lakes of North America were among the most heavily contaminated of the world's freshwater ecosystems during the 1960s and 1970s (Nriagu and Simmons 1984; Government of Canada 1991). The widespread manufacture and use of numerous synthetic chemicals in the Great Lakes drainage basin since the 1940s resulted in toxic contamination of the aquatic environment. Impaired reproduction, severe thinning of eggshells, congenital and physiological abnormalities, acute toxicity, and population declines were noted in many aquatic organisms, especially among predators such as piscivorous birds (Faber and Hickey 1973; Gilbertson and Fox 1977; Gilman et al. 1977; Mineau et al. 1984; Gilbertson 1988; Fox et al. 1991; Government of Canada 1991).

The introduction of regulatory controls on the use and discharges of these toxic compounds since the 1970s has resulted in declining contaminant levels in Great Lakes fish (DeVault et al. 1986; Baumann and Whittle 1988; Borgmann and Whittle 1991; Suns et al. 1991) and aquatic birds (Weseloh et al. 1979, 1989, 1994; Mineau et al. 1984; Heinz et al. 1985; Government of Canada 1991; Bishop et al. 1992a, 1992b; Ewins et al. 1992; Hebert et al. 1994; Pettit et al. 1994a, 1994b). Although the largely piscivorous Herring Gull *Larus argentatus* (Fox et al. 1990; Ewins et al. 1993) has been used to monitor temporal and spatial trends in organochlorine contaminant (OC) levels on the Great Lakes since 1974 (Mineau et al. 1984; Bishop et al. 1992a, 1992b; Pettit et al. 1994a, 1994b), it appears to be relatively insensitive to the toxic effects of environmental contaminants at current residue levels (Keith and Gruchy 1972; Fox et al. 1991).

The cosmopolitan Caspian Tern *Sterna caspia* is one of a small number of entirely piscivorous bird species nesting on the Great Lakes. The North American population is disjunct but is by far the largest of all worldwide populations (Croxall et al. 1984; Cramp 1985). The Great Lakes are a stronghold of this species, accounting for at least one-third of the North American population (Martin 1978; Ludwig 1979; Blokpoel 1983; Kress et al. 1983). Caspian Terns breed on small islands in the Great Lakes from May to August and then move to wintering areas in the Caribbean, as far south as Colombia and Venezuela (Ludwig 1942, 1965; L'Arrivée and Blokpoel 1988).

Most studies of Caspian Terns on the Great Lakes have concentrated on their reproductive biology (Ludwig 1965; Haymes and Blokpoel 1978; Shugart et al. 1979; Quinn 1980; Fetterolf and Blokpoel 1983; Quinn and Morris 1986), diet (H. Blokpoel, J. Gregorich, and L. Still, unpubl. data), or population dynamics and movements (Ludwig 1942, 1965, 1968; Cuthbert 1981, 1985; L'Arrivée and Blokpoel 1988). Three studies reported reproductive or population parameters in relation to OC levels in eggs (Struger and Weseloh 1985; Ludwig et al. 1993; Yamashita et al. 1993). In addition, Mora et al. (1993) recently detected a significant negative correlation between levels of polychlorinated biphenyl compounds (PCBs) in adult Caspian Tern blood serum and the degree of natal philopatry.

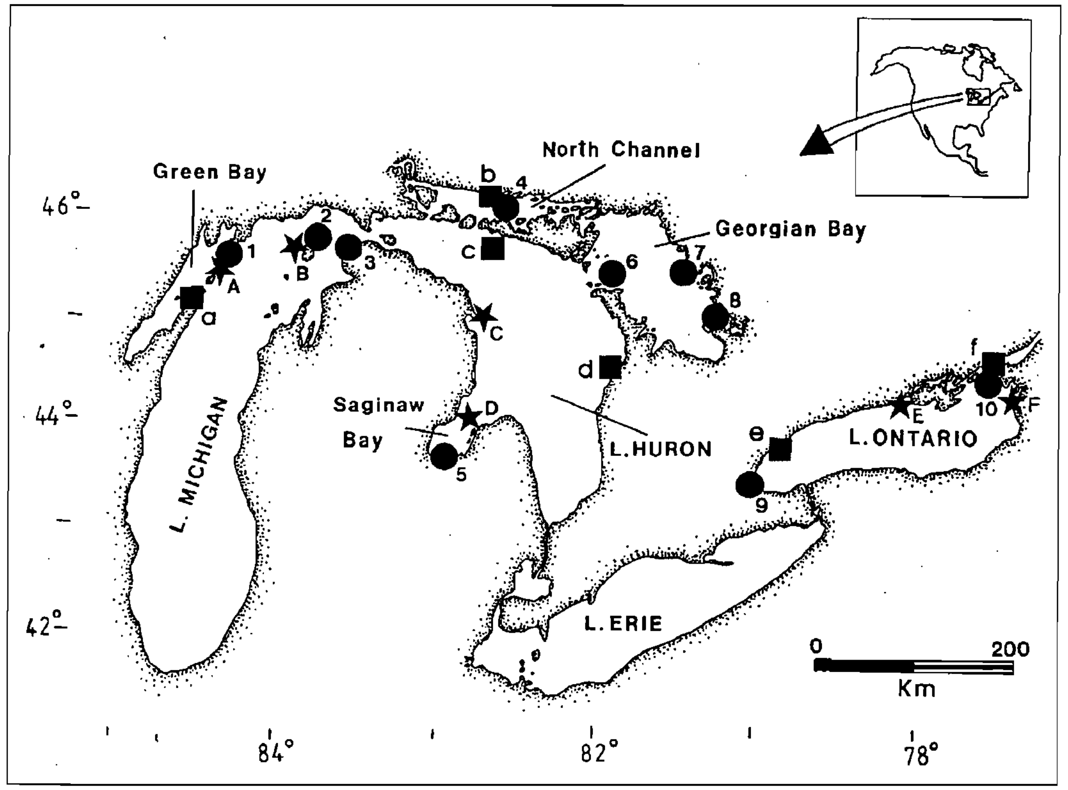
The limited contaminant data available indicate substantial declines in OC residues in Caspian Tern eggs over the past two decades, at the same time as reproductive output seemed to be relatively high (Struger and Weseloh 1985), and the Great Lakes Caspian Tern population appears to have increased steadily since the 1960s (Ludwig 1979; Shugart and Scharf 1983; Weseloh et al. 1986; Blokpoel and Scharf 1991; Blokpoel and Tessier 1991). However, Yamashita et al. (1993) carried out a detailed study in 1988, mostly at colonies on the U.S. Great Lakes, and found that organochlorine contamination in Caspian Terns influenced embryonic abnormalities and egg viability. Severe reproductive impacts were noted among Caspian Terns breeding in Saginaw Bay, Lake Huron, following disturbance of highly contaminated sediments by a 100-year flood event in 1986 (Ludwig et al. 1993). Similar ecotoxicological studies were completed in the early 1980s in California (Ohlendorf et al. 1985, 1988), but few contaminant data exist for this species in other parts of its range.

We present here the results of a study of OCs in Caspian Terns at 10 colonies across the entire Great Lakes breeding range in 1991, a decade after a similar extensive study (Struger and Weseloh 1985). We consider OC residues in relation to reproductive output, diet, and population trends at individual colonies and examine the suitability of this species as a sensitive indicator of the toxic effects of anthropogenic chemicals.

2. Study areas

Most of the 16 Caspian Tern colonies studied in 1991 were on small rocky islands or shingle bars, up to 15 km from the nearest mainland (Fig. 1). The exception was the mainland colony at Hamilton Harbour, Lake Ontario, situated on waste ground adjacent to partially developed docking facilities. At most of these sites, Caspian Terns bred in close proximity to Double-crested Cormorants *Phalacrocorax auritus*, Herring Gulls, and Ring-billed Gulls *Larus delawarensis*. Recent population data were examined for two colonies in Lake Ontario, at Little Galloo Island (43°53′N, 76°24′W) and Gull Island at Presqu'Ile Provincial Park (43°59′N, 77°45′W). In Saginaw Bay (Lake Huron), Channel and Shelter islands were formerly separated but have now been amalgamated in a Confined Disposal Facility (CDF) for dredged material. The new island is now known as Channel/Shelter Island, New Island, or the Saginaw Bay CDF (Fig. 1). Comparative contaminant data for Herring Gull eggs were obtained in 1991 at six additional island colonies and at Hamilton Harbour (Fig. 1).

Figure 1
The location of study colonies in 1991. Circles represent Caspian Tern colonies from which eggs and other data were collected: 1, Gravelly Island; 2, Hat Island; 3, Ile aux Galets; 4, Cousins Island; 5, Channel/Shelter Island; 6, Halfmoon Island; 7, South Limestone Island; 8, South Watcher Island; 9, Hamilton Harbour; 10, Pigeon Island. Stars represent Caspian Tern study colonies from which eggs were not collected: A, Gull Island (Lake Michigan); B, High Island; C, Thunder Bay Island; D, Charity Island Reef; E, Gull Island (Lake Ontario); F, Little Galloo Island. Squares represent Herring Gull colonies at which comparative contaminant data were collected: a, Big Sister Island; b, Double Island; c, Manitoba Reef; d, Chantry Island; e, Leslie Spit; f, Snake Island.



3. Methods

3.1 Egg collection and measurements

One fresh egg was collected at random from between nine and 13 freshly laid three-egg clutches along a broad transect walked through each colony. In a few cases, two-egg clutches were sampled on account of insufficient three-egg clutches. Eggs were collected within 7–10 days of laying, as determined by previous knowledge of breeding phenology and flotation tests. Caspian Terns frequent Great Lakes colonies from mid-April onwards, and most clutches are initiated between 5 and 21 May (Ludwig 1965; Quinn 1980; Cuthbert 1981; Mora et al. 1993; Canadian Wildlife Service [CWS], unpubl. data). Eggs analyzed in 1991 were collected between 8 and 18 May (i.e., within about four weeks of adults arriving on the Great Lakes).

The eggs were labelled and stored at 2–4°C in cardboard egg boxes within 48 hours of collection. Two months later, at the National Wildlife Research Centre in Hull, Quebec, the maximum length (L) and breadth (B) were measured, accurate to 0.01 cm, with dial-gauge vernier calipers. A volume index (VI) was calculated using the formula $VI = LB^2$ (Ratcliffe 1967). Eggs were opened around their equator using a surgical scalpel, and the contents were emptied into hexane-rinsed glass jars and frozen until analyzed chemically. All pieces of eggshell were then rinsed thoroughly in water and air-dried for at least three months. The total weight of each eggshell (including membranes) was measured on an electronic balance, accurate to 0.1 mg. The Ratcliffe index of eggshell thickness (Ratcliffe Thickness Index, or RTI) was calculated using the formula RTI = shell weight (mg) / [L (mm) × B (mm)] (Ratcliffe 1967). Eggshell thickness was measured accurate to 0.001 mm with a Starrett dial micrometer at five points around the equator, and a mean was calculated. For eggs in the museum collections, total shell weight was measured similarly with an electronic balance (accurate to 0.1 mg), but a correction was made for shell missing from the blowhole (some holes were up to 8 mm in diameter). This was done by adding an equivalent weight of shell plus membrane, determined as 0.5 $mg{\cdot}mm^{-2}$ for eggs collected in 1991.

Similar collection and storage methods were used for freshly laid Herring Gull eggs sampled for contaminant analyses (Mineau et al. 1984; Bishop et al. 1992a, 1992b; Pettit et al. 1994a, 1994b) and for Caspian Tern eggs collected in the 1980–81 study (Struger and Weseloh 1985).

3.2 Contaminant analyses

Egg pools were made by mixing equal-weight aliquots of homogenates of 10 whole eggs from three-egg clutches (and a few two-egg clutches) in each colony. Three-egg nests were targeted, thereby randomizing the differences in contaminant levels among the successively laid eggs in each clutch. Moderate intraclutch variation in contaminant levels has been noted in several species of colonial fish-eating birds (Mineau 1982; Nisbet 1983; Custer et al. 1990).

Organochlorine compounds were analyzed according to a protocol very similar to that described in Peakall et al. (1986). Egg pools have been found to produce results comparable to arithmetic mean contaminant levels determined for individual egg analyses, and it is much cheaper to use egg pools than individual eggs (Turle and Collins 1992). Briefly, pooled egg homogenate (1.5 g) was ground with 20 g of sodium sulphate and extracted with dichloromethane:hexane (1:1) in a column. One-third was evaporated to dryness for gravimetric lipid determination. The remaining 1.0-g equivalent sample was cleaned up and divided into three fractions by chromatographing on 1.2% water-deactivated Florisil. The fractions were analyzed with a Hewlett-Packard 5890 gas chromatograph/electron capture detector (GC/ECD) by splitless injection, using a 0.25 mm internal diameter × 60 m long, 0.1-µm DB5 column (J and W Scientific). Fraction 1 was analyzed for the presence of chlorobenzenes, octachlorostyrene, trans-nonachlor, p,p′-dichlorodiphenyldichloroethylene (DDE), photomirex, mirex, and 42 PCB congeners, including the two mono-ortho substituted (mo-PCB) congeners used in the toxic equivalency calculations, PCB-118 and PCB-105 (IUPAC numbering system; Ballschmiter and Zell 1980). Fraction 2 was analyzed for residues of 18 organochlorine pesticides and metabolites, mainly chlordane-related. Fraction 3 was analyzed for heptachlor epoxide and dieldrin. The minimum detectable concentration for most organochlorines and individual PCB congeners was 0.001–0.002 $\mu g{\cdot}g^{-1}$ wet weight. For chlorinated benzenes and hexachlorocyclohexane (HCH), it was 0.005 $\mu g{\cdot}g^{-1}$ wet weight. Samples for which a chemical occurred at a concentration below the detection limit were assigned a value of one-half of the minimum detection limit for the purposes of statistical analysis.

Polychlorinated dibenzo-p-dioxin (PCDD) and dibenzofuran (PCDF) levels were determined in 25-g pooled homogenates by gas chromatography/low-resolution mass spectrometry (GC/LRMS) according to the methods described by Norstrom and Simon (1991) and Norstrom et al. (1990). In this method, the homogenate was extracted in a column, as for the OC analyses, after the addition of $^{13}C_{12}$-labelled tetrachloro- (TCDD) to octachloro- (OCDD) dibenzo-p-dioxin internal standards. Bulk lipid was removed by gel permeation chromatography, and the PCDDs and PCDFs were selectively adsorbed on a column of AX-21 carbon/glass fibre. PCDDs and PCDFs were back-eluted from the carbon with toluene, and the sample was chromatographed on Florisil and basic alumina columns. A performance internal standard, $^{37}Cl_4$-2378-TCDD, was added just prior to analysis. Analysis was performed on a Hewlett-Packard 5987 quadrupole gas chromatograph/mass spectrometer (GC/MS) using conditions similar to those described for GC/ECD analysis above, except that a 30-m DB5 column was used. Average internal standard recoveries ranged from 80 to 100%. The minimum detectable concentration under these conditions ranged from 1 $ng{\cdot}kg^{-1}$ for 2378-TCDD to 8 $ng{\cdot}kg^{-1}$ for 12346789-OCDD.

Non-ortho substituted PCBs (no-PCBs) were determined by a method similar to that used for PCDDs and PCDFs. The procedure is described in detail in Ford et al. (1993). The samples (5 g) were spiked with $^{13}C_{12}$-PCB-77, $^{13}C_{12}$-PCB-126, and $^{13}C_{12}$-PCB-169 internal standards and ground with sodium sulphate. The sample plus sodium sulphate was layered on top of a column consisting of 10 g each of silica gel, KOH:silica gel (33%), silica gel, and sulphuric acid:silica gel (40%). The sample was extracted and the bulk lipid removed by elution of the column with 600 mL of 1:1 dichloromethane:hexane. After evaporation, the samples were loaded onto a 5-cm-long AX-21 carbon/glass fibre column in hexane, and the organochlorines and PCBs, except for no-PCBs, were eluted with 60 mL of hexane and 90 mL of dichloromethane. The no-PCBs were back-eluted with 120 mL of toluene and cleaned up by chromatography on 8 g of Florisil (1.2% water), eluting with 40 mL of 5% dichloromethane:hexane. The no-PCBs were determined in the same way as PCDDs and PCDFs, using a Hewlett-Packard 5971A gas chromatograph/mass-selective detector (GC/MSD) instrument. Average internal standard recoveries were 78 ± 12% for PCB-77, 84 ± 13% for PCB-126, and 85 ± 13% for PCB-169.

3.3 Reproduction in 1991

Counts of nests and their contents were made by two people walking slowly across the colony and marking each nest counted with a scratch in the sand or shingle or a small spot of paint nearby. These counts were made during the mid- to late incubation period. The census unit was an active nest, being a well-defined scrape depression containing one or more eggs. Empty nest scrapes were not included in the calculations of reproductive output but were noted separately. We worked in colonies for up to 40 minutes maximum and avoided extreme weather conditions that might have adversely affected reproductive output.

Reproductive success was determined at six colonies, with estimates relating to productivity for the peak of nesting. At the three Canadian colonies (Cousins Island, South Watcher Island, Hamilton Harbour), this was done by erecting a fine-mesh (6 mm), high-quality wire fence 1 m high around groups of occupied nests (28, 41, and 31, respectively, in the aforementioned colonies) during the late incubation period. The use of a fine-mesh fence (hardware cloth) eliminated any problems with chicks damaging their bills or necks in the wire. In each colony, the enclosure extended from near the edge towards the centre, so minimizing any biases that may have resulted from sampling birds of differing age or breeding experience (as found in other larids; e.g., Coulson 1968; Coulson and Thomas 1985). Enclosures were revisited in late June – early July, at a median chick age of 21–28 days, and the numbers of chicks and eggs were noted (cf. Mineau and Weseloh 1981). Previous studies have indicated that the bulk of nestling mortality occurs prior to 21–28 days (Quinn 1980; Quinn and Morris 1986), so the number of chicks reaching this age has been taken as a measure of reproductive output (Shugart et al. 1979; Struger and Weseloh 1985). Eggs were inspected at this stage for viability by flotation, searches for shell fractures, and gentle shaking to see if the contents were firm or sloppy (most remaining eggs were rotten by this stage). Hatching success was then estimated for each quadrat by adding the number of live and dead chicks, then dividing the total number by the number of eggs counted in the quadrat on the incubation stage census. This method assumed that no eggs or chicks had been removed from the quadrat (e.g., by predation). Fences were then dismantled.

An index of relative chick growth rates among some colonies was obtained by regressing chick mass against wing length. Only data for chicks with wing lengths between 75 and 250 mm were used, as mass did not appear to increase linearly with wing length (i.e., chick age) outside this range.

At the three U.S. colonies (Green Bay — Gravelly Island and nearby Gull Island, combined; High Island; and Channel/Shelter Island), reproductive output was estimated without using enclosures, following procedures used by Ludwig (1965, 1968). Nests and eggs were censused in well-defined, relatively well synchronized subsections of each colony late in incubation; 54–59 nests were selected for intensive monitoring. Seven to 10 days post-hatch for the majority of nests in these marked areas, all chicks were banded with standard U.S. Fish and Wildlife Service metal bands. All dead, damaged, predated, pipping, and missing eggs, and any dead or injured chicks, were also noted. At the estimated mean age of 21–28 days, each study area was revisited and all live chicks were recaptured or counted. As older chicks often run out of colonies ahead of investigators, all live chicks were counted from a distance of 50–60 m with 10× binoculars before investigators entered the marked plots. This procedure was essential at High Island (which had little vegetation near the colony), as more than half the large chicks (>21 days) vacated their nest territories when approached. Other colonies had sufficient vegetation to hide most chicks. In these U.S. colonies, about 6–10% of marked chicks disappeared between the nestling period visits, probably the result of gull predation.

Although this technique was more likely to underestimate overall productivity than that used at the Canadian study colonies, at present we have no means of quantifying any biases inherent in either technique. Chicks exhibiting the so-called "wasting syndrome" were diagnosed tentatively on the nestling stage visits, particularly in the U.S. colonies. They appeared lethargic and emaciated, especially relative to other chicks at least 10 days old. Further explanations and descriptions of this syndrome are presented by Peterson et al. (1984), Kubiak et al. (1989), and Ludwig et al. (1993).

3.4 Diet analyses

At most Canadian study colonies and two U.S. colonies, up to 45 freshly regurgitated pellets were collected from the immediate vicinity of nests. These were stored individually and their contents analyzed using a reference collection of fish scales, otoliths, and skeletal parts, as well as published identification guides (Lagler 1947; Scott and Crossman 1973) and other reference material. We relied mostly on fish scales for identifications, as other skeletal parts were often broken or worn. Diet composition was expressed as the percentage of pellets from each collection in which each diet item occurred.

At three U.S. study colonies, greater numbers of pellets were collected, but these were stored as a combined pool for each site/date. A measure of diet composition was obtained by estimating the minimum number of individuals (MNI) of each prey type and expressing this as a percentage of the total MNI for each pool of pellets. For fish species, MNIs were estimated by pairing otoliths, pectoral spines, pharyngeal arches, and various distinctive head bones, and by grouping scales and vertebrae of different sizes. Owing to difficulties in distinguishing scales of alewife *Alosa pseudoharengus* and gizzard shad *Dorosoma cepedianum*, these species were grouped together. Sunfishes and some bass species (Centrarchidae) were treated as one prey type.

3.5 Population changes

In 1991, and in most previous survey years, numbers of active (and occasionally empty) nests were determined during the mid- to late incubation period. Census methods were as described in Section 3.3. Average rates of population change (GR) were calculated from a recent count (RC) and an initial count (IC), n years previously, using the formula, adapted from Ricklefs (1980): $GR = [\ln(RC) - \ln(IC)] / n$.

3.6 Statistics

The term significant is used in its statistical sense only. Statistical significance is accepted at the $P = 0.05$ level. The number of degrees of freedom (d.f.) is often given as a subscript to a statistic. Means quoted are arithmetic, ±1 standard deviation (SD), unless stated otherwise. Egg contaminant data were $\log_{10}$-transformed to reduce heteroscedasticity and to meet distributional requirements for parametric analyses more closely. When transformations failed to normalize most distributions and equalize variances, nonparametric tests were used. Statistical methods follow Sokal and Rohlf (1981).

4. Results

4.1 Contaminants

Of the 21 organochlorine compounds screened in the egg pools, the following 11 were detected in egg pools from all colonies: hexachlorobenzene (HCB), DDE, mirex, photomirex, oxychlordane, cis-chlordane, cis-nonachlor, trans-nonachlor, heptachlor epoxide (HE), dieldrin, and PCBs. Four compounds were detected at only some of the 10 colonies: dichlorodiphenyltri-chloroethane (DDT) (at eight), dichlorodiphenyldichloroethane (DDD) (four), octachlorostyrene (five), and trans-chlordane (two). The following six compounds were below detectable limits at all colonies: 1245- and 1234-tetrachlorobenzene, pentachlorobenzene, and α-, β-, and γ-isomers of HCH.

Concentrations of the main contaminants in fresh eggs are presented in Table 1 and Figure 2. PCBs, DDE, Σchlordane compounds and their metabolites (the summed concentrations of cis- and trans-chlordane, oxychlordane, cis- and trans-nonachlor, and HE), dieldrin, and mirex occurred in the highest concentrations in eggs. To facilitate direct comparisons with other published accounts, PCB levels are given both as the sum of the 42 most abundant congeners (ΣPCBs) and as total residues relative to a reference standard 1:1 mix of Aroclor 1254:1260. Conversion of pre-1991 PCB concentrations in Caspian Tern eggs to ΣPCB equivalents was based on the following ratios, derived from the 1991 chemical analyses for these study colonies:

$$\Sigma\text{PCBs:PCB 1:1 mix of Aroclor 1254:1260} = 0.539 \pm 0.026$$
$$\text{PCBs:PCB Aroclor 1260} = 1.048 \pm 0.120$$

Subsequent analyses utilize only the actual or converted ΣPCB values.

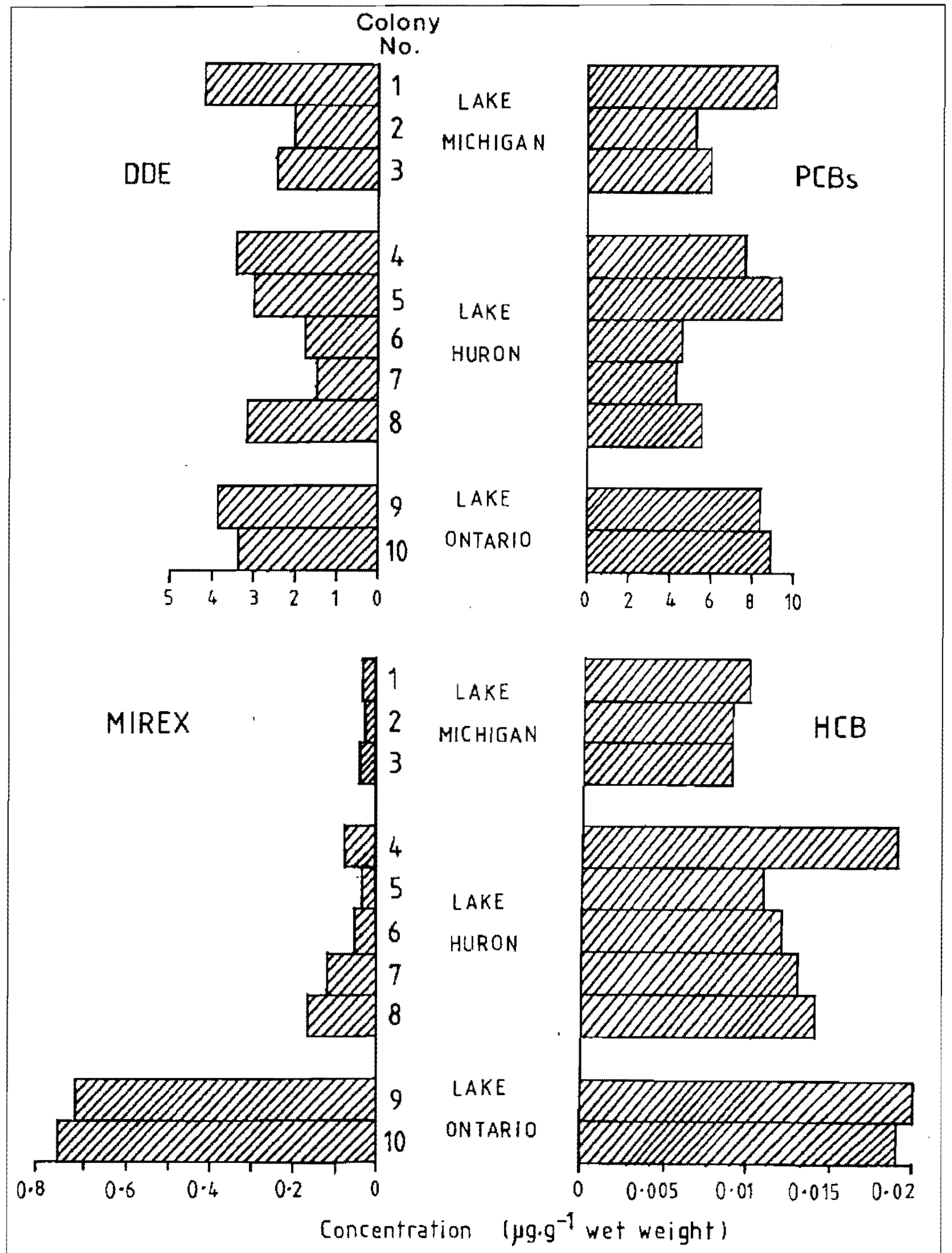
For most compounds, levels varied relatively little among colonies, by factors ranging from 1.6 (for Σchlordane compounds) to 2.9 (for DDE). The range of variation was much greater for mirex and photomirex

Table 1
Levels of six organochlorine compounds in Caspian Tern eggs from 10 colonies on the Great Lakes in 1991[a]

Colony[b]	% lipid	Concentration ($\mu g \cdot g^{-1}$ wet weight) DDE	Dieldrin	ΣCHL[c]	HCB[d]	Mirex	ΣPCBs[e]	PCBs (1:1)[f]
Lake Michigan								
1. Gravelly I.	8.2	4.23	0.087	0.281	0.010	0.032	9.165	15.820
2. Hat I.	7.8	2.06	0.093	0.222	0.009	0.024	5.201	10.298
3. I. aux Galets	6.9	2.45	0.093	0.219	0.009	0.036	6.000	11.358
Mean:	*7.6*	*2.91*	*0.091*	*0.241*	*0.009*	*0.031*	*6.789*	*12.492*
Lake Huron								
4. Cousins I.	8.8	3.41	0.076	0.300	0.019	0.074	7.681	14.552
5. Channel/Shelter I.	8.4	2.97	0.042	0.205	0.011	0.033	9.393	16.556
6. Halfmoon I.	8.9	1.77	0.093	0.204	0.012	0.053	4.596	8.563
7. South Limestone I.	9.2	1.47	0.076	0.191	0.013	0.118	4.289	7.483
8. South Watcher I.	9.1	3.12	0.077	0.220	0.014	0.167	5.452	10.206
Mean:	*8.9*	*2.55*	*0.073*	*0.224*	*0.014*	*0.089*	*6.282*	*11.472*
Lake Ontario								
9. Hamilton Hbr.	8.6	3.82	0.080	0.249	0.020	0.719	8.543	16.302
10. Pigeon I.	9.3	3.34	0.061	0.184	0.019	0.767	8.967	17.500
Mean:	*9.0*	*3.58*	*0.071*	*0.217*	*0.020*	*0.743*	*8.755*	*16.901*

[a] Values were determined for pools of 9–13 eggs for each colony.
[b] Colony numbers refer to locations shown in Figure 1.
[c] Sum of chlordane-related compounds (oxychlordane, cis- and trans-chlordane, cis- and trans-nonachlor, heptachlor epoxide).
[d] Hexachlorobenzene.
[e] Sum of 42 PCB congeners.
[f] Level determined against a reference 1:1 mix of Aroclor 1254:1260.

Figure 2
Concentrations of DDE, total PCBs, mirex, and HCB in pools of fresh eggs from 10 Caspian Tern colonies in 1991. Colony numbers refer to those shown in Figure 1 and Table 1.



(factors of 32.0 and 24.7 [data not shown], respectively). Among the six main contaminant classes (Table 1), the only significant correlations across colony means were between DDE and ΣPCBs ($r_{18} = 0.84$, $P = 0.002$) and between HCB and mirex ($r_{18} = 0.78$, $P = 0.008$).

There was no significant correlation between percent lipid and OC levels in the egg pools across the 10 colonies, but the lipid content of eggs did vary significantly among the three lakes (One-Way Analysis of Variance [ANOVA], $F_{2,7} = 7.78$, $P = 0.017$). Lake Michigan eggs had significantly lower percent lipid content (7.6 ± 0.7) than those from Lake Huron (8.9 ± 0.3) or Lake Ontario (9.0 ± 0.5) (Tukey's HSD [Honestly Significant Difference] tests, $P < 0.05$) (Table 1).

Levels of OCs varied significantly among colonies for the 11 compounds detected in all egg pools (Friedman Two-Way ANOVA, $\chi^2 = 89.0$, 10 d.f., $P < 0.001$) and also for the six main compound classes in Table 1 ($\chi^2 = 47.0$, 5 d.f., $P < 0.0001$). When $\log_{10}$-transformed residues were analyzed by lake (using One-Way ANOVAs), significant variation among lakes was found for four of the 11 OCs: HCB ($F_{2,7} = 10.2$, $P = 0.008$), mirex ($F_{2,7} = 201$, $P < 0.0005$), photomirex ($F_{2,7} = 164$, $P < 0.0005$), and HE ($F_{2,7} = 7.67$, $P = 0.017$). Mean HCB levels were significantly

Table 2
Comparison of 1991 egg contaminant levels with previous data[a] from Lake Huron and Lake Ontario[b]

Colony	Year	N	% lipid	Concentration (µg·g⁻¹ wet weight) DDE	Dieldrin	Mirex	ΣPCBs[c]
Lake Huron							
South Limestone I.	1972	pool	9.8	15.8	0.08	n.a.[d]	33.3
	1976	10	n.a.	7.8 ± 3.2	0.14	0.51 ± 0.21	22.6 ± 2.2
	1980	10	7.1	3.7 ± 0.4	n.a.	0.31 ± 0.05	22.7 ± 3.1
	1991	pool	9.2	1.5	0.08	0.12	7.5
1980–91: overall change:				−60%	–	−61%	−69%
1980–91: mean per year:				−8.2%	–	−8.6%	−10.7%
Lake Ontario							
Pigeon I.	1972	4	n.a.	13.8 ± 2.5	0.20	n.a.	85.1
	1981	8	8.3	5.2 ± 0.6	n.a.	1.57 ± 0.19	21.2 ± 3.0
	1991	pool	9.3	3.3	0.06	0.77	9.0
1981–91: overall change:				−36%	–	−51%	−58%
1981–91: mean per year:				−4.6%	–	−7.1%	−8.6%

[a] Previous data are from Gilbertson and Reynolds (1974) (1972 data), Martin (1978) (1976 data), Struger and Weseloh (1985) (1980–81 data), and Bishop et al. (1992a, 1992b) (1991 data).
[b] All values other than pooled samples are given as arithmetic means ± SE.
[c] Sum of 42 PCB congeners, pre-1991 values converted to ΣPCBs using ratios given in text.
[d] n.a. = not analyzed.

higher in Lake Ontario eggs (0.020 ± 0.0007) than in Lake Michigan eggs (0.009 ± 0.001), whereas the reverse was true for HE levels (0.027 ± 0.002 in Lake Ontario eggs and 0.048 ± 0.008 in Lake Michigan eggs) (Tukey's HSD tests, $P < 0.05$). Mean levels of mirex and photomirex were significantly higher in Lake Ontario eggs (0.743 ± 0.034 and 0.275 ± 0.029, respectively) than in either Lake Huron eggs (0.089 ± 0.054; 0.037 ± 0.017) or Lake Michigan eggs (0.031 ± 0.006; 0.015 ± 0.002) (Tukey's HSD tests, $P < 0.05$). Within Lake Huron, residue levels were generally lower at colonies in Georgian Bay (numbers 6, 7, and 8 in Figure 1) than elsewhere (Table 1; Figs. 1 and 2), but levels of mirex, photomirex, and dieldrin were markedly higher on average in Georgian Bay (but, again, not significantly so). Concentrations of PCBs were highest at colonies in Lake Ontario, Saginaw Bay (colony number 5), the North Channel (number 4), and outer Green Bay, Lake Michigan (number 1). DDE concentrations were also highest at these colonies and at South Watcher Island (colony number 8). Lake-wide mean levels of DDE and PCBs were highest in Lake Ontario, followed by Lake Michigan, then Lake Huron (Table 1). Levels of dieldrin and Σchlordane compounds were generally higher in eggs from Lake Michigan than elsewhere, but not significantly so (Table 1).

Temporal changes in OC levels over the past 20 years were assessed using egg data from Lake Huron (South Limestone Island) and Lake Ontario (Pigeon Island) (Table 2). These trends could not be assessed statistically because of the use of pooled samples for the 1991 analyses. Levels of DDE, mirex, and ΣPCBs had all decreased markedly. The declines since 1980–81 were greater for all compounds at South Limestone Island (60–69%, mean rate 8–11% per annum) than at Pigeon Island (36–58%, mean rate 5–9% per annum) (Table 2). Since 1972, egg DDE levels had declined by 91% and ΣPCB levels by 80% at South Limestone Island, and by 76% and 89%, respectively, at Pigeon Island. Unlike other compounds, dieldrin levels had not declined noticeably at South Limestone Island. At the three colonies in Lake Michigan and at the Cousins Island and Halfmoon Island colonies in Lake Huron, egg contaminant levels could be compared with those in 1980 (Struger and Weseloh 1985). At most of these colonies, egg levels of ΣPCBs, mirex, and DDE had declined at rates similar to those in Lake Ontario and at South Limestone Island — i.e., a reduction of approximately 50% between 1980 and 1991. At Cousins Island, North Channel, levels of DDE had declined by only 27% (an average decline of 3% per annum), whereas mirex levels had increased by 7% over the 11-year period.

Levels of PCDDs and PCDFs in eggs are presented in Table 3. Three PCDD and two PCDF congeners were found routinely above the detection limit. OCDD was identified in all but one sample, but levels were generally below the limit of reliable quantitation. Traces of 12378-pentachlorodibenzo-p-dioxin (PnCDF), 123478-hexachlorodibenzo-p-dioxin (HxCDF), and 123678-HxCDF below 5 ng·kg⁻¹ were detected in the Cousins Island egg pool. Eggs from the two Lake Ontario colonies contained 3–5 ng·kg⁻¹ of the two HxCDF congeners. Distribution of TCDD levels among colonies within a lake was quite uniform and increased in the order Lake Michigan < Lake Huron < Lake Ontario. $\log_{10}$-transformed TCDD levels varied significantly among lakes (One-Way ANOVA, $F_{2,7} = 10.7$, $P = 0.007$), with Lake Michigan concentrations averaging significantly lower than those on Lake Huron ($P = 0.022$) or Lake Ontario ($P = 0.008$). Levels of the other congeners were more evenly distributed among lakes, with occasional deviations, but there were no significant overall differences among lakes (One-Way ANOVAs, $P > 0.3$). The highest levels of OCDD, HxCDD, and 2378-TCDF, but not of other congeners, were usually found at Gravelly Island in Lake Michigan's Green Bay and at Cousins Island in Lake Huron's North Channel.

Concentrations of the five most important no-PCB and mo-PCB congeners are presented in Table 4. TCDD toxic equivalents (TEQs) based on the toxic equivalency factors (TEFs) given by Safe (1990) for PCDDs/PCDFs and no- and mo-PCBs are also shown in Table 4. The no-/mo-PCB patterns differed significantly among colonies (Friedman's Two-Way ANOVA on levels of PCB

Table 3
Levels of PCDD and PCDF congeners in Caspian Tern eggs from Great Lakes colonies in 1991

Colony	Concentration (ng·kg^{-1} wet weight) 2378-TCDD[a]	12378-PnCDD[b]	123678-HxCDD[c]	OCDD[d]	2378-TCDF[e]	23478-PnCDF[f]
Lake Michigan						
1. Gravelly I.	8	6	10	28	11	4
2. Hat I.	7	4	6	(4)	6	4
3. I. aux Galets	6	5	5	(3)	6	3
Mean:	*7*	*5*	*7*	–	*8*	*4*
Lake Huron						
4. Cousins I.	22	15	16	(3)	23	9
5. Channel/Shelter I.	22	14	6	(6)	7	4
6. Halfmoon I.	8	4	4	(3)	4	<1
7. South Limestone I.	26	4	5	(6)	4	3
8. South Watcher I.	18	9	8	(4)	9	5
Mean:	*19*	*9*	*8*	–	*10*	*4*
Lake Ontario						
9. Hamilton Hbr.	31	4	4	(<2)	5	5
10. Pigeon I.	27	5	5	(7)	4	3
Mean:	*29*	*5*	*5*	–	*5*	*4*

[a] Tetrachlorodibenzo-p-dioxin.
[b] Pentachlorodibenzo-p-dioxin.
[c] Hexachlorodibenzo-p-dioxin.
[d] Octachlorodibenzo-p-dioxin; levels in parentheses are positive identifications but semiquantitative levels (signal noise between 2 and 3 ng·kg^{-1}.
[e] Tetrachlorodibenzofuran.
[f] Pentachlorodibenzofuran.

Table 4
Levels of non-ortho (no-) and mono-ortho (mo-) PCBs, and their respective 2378-TCDD toxic equivalents (TEQs), in Caspian Tern egg pools from Great Lakes colonies in 1991

Colony	Concentration (µg·kg^{-1} wet weight) PCB-77 (no-)	PCB-126 (no-)	PCB-169 (no-)	PCB-118 (mo-)	PCB-105 (mo-)	TEQ (µg·kg^{-1} wet weight) PCDDs/PCDFs[a]	PCBs	Total
Lake Michigan								
1. Gravelly I.	6.11	3.31	0.213	831	191	0.015	1.43	1.44
2. Hat I.	2.73	2.59	0.208	483	102	0.012	0.88	0.89
3. I. aux Galets	2.72	2.21	0.186	520	105	0.011	0.88	0.89
Mean:	*3.85*	*2.70*	*0.202*	*611*	*133*	*0.013*	*1.06*	*1.08*
Lake Huron								
4. Cousins I.	3.18	3.30	0.463	642	127	0.037	1.15	1.19
5. Channel/Shelter I.	2.54	2.63	0.184	1060	216	0.032	1.57	1.61
6. Halfmoon I.	1.48	1.77	0.147	374	86	0.011	0.66	0.67
7. South Limestone I.	2.39	1.08	0.122	323	66	0.030	0.53	0.56
8. South Watcher I.	2.53	1.62	0.201	446	88	0.027	0.73	0.76
Mean:	*2.42*	*2.08*	*0.223*	*569*	*117*	*0.027*	*0.93*	*0.96*
Lake Ontario								
9. Hamilton Hbr.	1.30	1.67	0.134	553	135	0.036	0.88	0.91
10. Pigeon I.	1.40	1.88	0.147	668	151	0.032	1.03	1.06
Mean:	*1.35*	*1.78*	*0.141*	*610*	*143*	*0.034*	*0.95*	*0.99*

[a] TEQs for PCDDs and PCDFs are calculated from Table 3. Toxic equivalency factors (TEFs) for PCDDs, PCDFs, and PCBs are from Safe (1990).

congeners 77, 126, 169, 118, and 105 at 10 colonies, χ^2 = 38.0, 4 d.f., P < 0.001). Comparisons of coplanar PCB congener levels ($\log_{10}$-transformed) among lakes revealed significant variation only for PCB-77 ($F_{2,7}$ = 5.3, P = 0.04). The concentration of PCB-77 in eggs from Gravelly Island was twice as high as that at the next highest colony, Cousins Island. PCB-126 levels were similar in these two colonies. Lake-wide mean levels of PCB-77 and PCB-126 increased in the order Lake Ontario < Lake Huron < Lake Michigan. Lake-wide means of the other coplanar PCB congeners did not vary as much.

In all cases, PCDDs and PCDFs accounted for <5% of the total calculated 2378-TCDD TEQs. Lake-wide mean TEQs were similar in all three lakes (P = 0.8). The highest total TEQ value was at Channel/Shelter Island (1.61 µg·kg^{-1} wet weight), but it was only 1.9 times higher than the lowest value (South Limestone Island). For egg pools from these 10 study colonies in 1991, there was a significant correlation between TEQs and ΣPCB levels (r = +0.73, $F_{1,8}$ = 21.8, P < 0.002) and between concentrations of the two main mo-PCBs (numbers 118 and 105) (r = +0.98, $F_{1,8}$ = 204.1, P < 0.0001). Because TEFs have not yet been validated for Caspian Terns, the calculated TEQs in Table 4 must be considered a semi- quantitative indication of relative risk among colonies to dioxin-like toxic effects. The calculation is not robust to variation in TEF values because of the changes in congener patterns among colonies. For example, the general system of TEFs

Table 5
Mean shell thickness, volume index, and Ratcliffe Thickness Index (RTI) of Caspian Tern eggs on the Great Lakes in 1991

Colony	No. of eggs	Eggshell thickness (mm)		Volume[a] index (cm^3)	RTI (mg·mm^{-2})		RTI change[b] since pre-1945 (%)
Lake Michigan							
1. Gravelly I.	13	0.335	AB[c]	129.96	1.572	ACD	+ 2.3
2. Hat I.	13	0.343	A	124.39	1.623	AC	+ 5.6
3. I. aux Galets	11	0.356	A	131.30	1.655	AD	+ 7.7*
Mean:		*0.345*		*128.78*	*1.616*		*+ 5.1*
Lake Huron							
4. Cousins I.	10	0.316	B	123.65	1.503	BCD	– 2.2
5. Channel/Shelter I.	12	0.335	AB	128.68	1.574	ACD	+ 2.4
6. Halfmoon I.	10	0.327	B	126.08	1.561	ACD	+ 1.6
7. South Limestone I.	10	0.329	B	124.41	1.535	ACD	– 0.1
8. South Watcher I.	9	0.315	B	123.44	1.469	BD	– 4.4
Mean:		*0.325*		*125.42*	*1.532*		*– 0.3*
Lake Ontario							
9. Hamilton Hbr.	10	0.339	AB	126.08	1.585	ACD	+ 3.1
10. Pigeon I.	10	0.325	B	128.08	1.497	BD	– 2.6
Mean:		*0.332*		*127.08*	*1.541*		*+ 0.3*
ANOVA, P		<0.0005		n.s.[d]	<0.0005		
Total (mean)	108	0.332		126.61	1.557		+ 1.3

[a] Length × breadth^2.
[b] Pre-1945 value taken from 18 eggs in Royal Ontario Museum (mean = 1.537 ± 0.134 mm). * indicates P = 0.01 (t-test), other colonies nonsignificant.
[c] Means sharing the same letter are not significantly different from each other (Tukey's HSD tests, P < 0.05).
[d] n.s. = not significant.

proposed by Safe (1990) gives considerably more weight to PCB-118 (TEF = 0.001) than did a similar study of Great Lakes Caspian Terns (PCB-118, TEF = 0.000006; Yamashita et al. 1993). Application of the Yamashita et al. (1993) TEFs to the results for Channel/Shelter Island and Gravelly Island gives TEQs about 40% and 20% lower, respectively, than those given in our Table 4. The differences among calculations and colonies are almost entirely due to the larger influence of PCB-118 in our calculations and the relative levels of PCB-118 at Channel/Shelter Island.

4.2 Egg biometrics

Mean values for shell thickness, volume index, and RTI of eggs in 1991 study colonies are presented in Table 5. Egg volume indices in 1991 did not vary significantly among colonies ($F_{9,98}$ = 1.2) or among lakes ($F_{2,105}$ = 1.6). For six colonies at which eggs were collected in both 1991 and 1980–81, the mean volume index was 0–6% lower in 1991, but none of the differences between years was significant (t-tests).

There was a significant positive correlation between shell thickness (at the equator region) and egg volume index [thickness (mm) = 10 × VI (mm^3) + 0.254; r = 0.26, $F_{1,106}$ = 7.54, P = 0.007]. Shell thickness varied significantly among colonies (One-Way ANOVA, $F_{9,98}$ = 5.5, P < 0.0005) and among lakes ($F_{2,105}$ = 11.9, P < 0.0005). Eggshells from Lake Michigan (0.345 ± 0.020 mm) were on average significantly thicker than those from Lake Huron (0.325 ± 0.019 mm) or Lake Ontario (0.332 ± 0.018 mm) (Tukey's HSD test, P < 0.05). Similarly, RTI varied significantly among colonies ($F_{9,98}$ = 4.43, P < 0.0005) and among lakes ($F_{2,105}$ = 9.09, P < 0.0005). Lake Michigan RTIs (1.616 ± 0.095 mg·mm^{-2}) were on average significantly higher than those for Lake Huron (1.532 ± 0.095 mg·mm^{-2}) or Lake Ontario (1.541 ± 0.098 mg·mm^{-2}) (Tukey's HSD test, P < 0.01). Thus, Lake Michigan eggs in 1991 were significantly thicker than those from other colonies and had significantly higher RTIs. On average, eggs from Lake Huron colonies were the smallest and had the lightest and thinnest shells.

It is now well established that DDE is the compound largely responsible for thinning of avian eggshells (Cooke 1973). However, there was no significant relationship between pooled $\log_{10}$-transformed DDE levels and average shell thickness for the 10 colonies sampled in 1991 (r^2 = 0.04, $F_{1,8}$ = 0.29, P = 0.6). A stepwise multiple regression analysis of mean shell thickness for the 10 colonies, on 11 OC residues, identified four compounds (none being DDE) that together explained 90% of the overall variance in thickness (cumulative percentage of variance explained is given in parentheses): HCB (28%), mirex (51%), cis-nonachlor (81%), and cis-chlordane (90%). A similar analysis for RTIs identified only HCB, explaining only 35% of the variance in RTIs among colonies.

Biometrics of 18 Royal Ontario Museum eggs that had been collected prior to 1945 (i.e., before the introduction of DDT) were compared with those of 13 eggs from the 1945–78 period, the so-called "DDT era." The respective means were (<1945 vs. ≥1945): for volume index, 118.09 ± 13.35 and 129.81 ± 6.84 cm^3 (t_{29} = 2.89, P = 0.007): for shell weight, 4.189 ± 0.576 and 4.170 ± 0.524 g (t_{29} = 0.1, P = 0.9); and for RTI, 1.537 ± 0.134 and 1.438 ± 0.143 mg·mm^{-2} (t_{29} = 1.963, P = 0.059). Thus, on the basis of this sample, Caspian Tern eggs were significantly smaller before the introduction of DDT. The 6.4% reduction in the mean RTI after 1945, although not quite significant, could have been largely due to an increase in

Table 6
The distribution of clutch size in Caspian Terns during the late incubation period in 1991

Colony	No. of eggs in nest 1	2	3	4	No. of nests with eggs	No. of empty[a] nests	Clutch size[b] (mean ± SD)
Lake Michigan							
1. Gravelly I.	5	69	11	0	85	0	2.07 ± 0.43
Gull I.	1	44	15	0	60	0	2.23 ± 0.47
2. Hat I.	8	48	7	0	63	0	1.98 ± 0.49
High I.	9	60	8	0	77	1	1.99 ± 0.47
3. I. aux Galets	14	63	9	1	87	25	1.97 ± 0.56
Lake Huron							
4. Cousins I.	60	336	56	0	452	15	1.99 ± 0.51
5. Channel/Shelter I.	3	139	46	1	189	10	2.24 ± 0.47
6. Halfmoon I.	22	138	14	0	174	10	1.95 ± 0.45
7. South Limestone I.	86	239	5	0	330	5	1.75 ± 0.46
8. South Watcher I.	80	548	44	0	672	21	1.95 ± 0.43
Lake Ontario							
9. Hamilton Hbr.	28	114	75	3	220	n.c.[c]	2.24 ± 0.68
10. Pigeon I.	43	343	113	4	503	n.c.	2.16 ± 0.56

[a] Well-formed scrapes containing no eggs. Nests containing small chicks and/or hatched shells were excluded.
[b] Zero-egg clutches not included in these calculations.
[c] n.c. = not counted.

egg volume, as the total shell weight declined by only 0.5% on average.

Using this pre-DDT value for RTI of 1.537 mg·mm^{-2}, RTIs of eggshells in 1991 averaged 1.3% higher than before the introduction of DDT (Table 5). Colony averages ranged between 4.4% lower (South Watcher Island) and 7.7% higher (Isle aux Galets), the latter value being the only significant change (t_{29} = 2.71, P = 0.011). Compared with shell thickness in 1980–81 (overall unweighted mean from seven colonies was 0.323 ± 0.013 mm; Struger and Weseloh 1985), eggs in 1991 averaged nearly 3% thicker (Table 5). Those from South Watcher Island averaged 2.5% thinner and those from Cousins Island averaged 2.2% thinner, whereas eggs from the other eight colonies became thicker by 0.6–10.2%. RTI values increased in a similar manner, by 3.5% overall, from an unweighted mean of 1.505 ± 0.089 mg·mm^{-2} in 1980–81 (Struger and Weseloh 1985), and individual colonies ranged from a 2.1% decline (Pigeon Island) to a 9.6% increase (South Limestone Island).

4.3 Reproduction in 1991

Clutch size, as determined during the late incubation period, varied between one and four eggs in the 12 colonies studied (Table 6). At colonies in Lake Michigan and Lake Huron, well-formed nest scrapes that were empty were also recorded. These comprised up to 5% of all discernible scrapes at most colonies, but 22% at Isle aux Galets. Mean clutch size varied significantly among different sites within a lake. In Lake Michigan, colonies in Green Bay had significantly larger mean clutches than those in the main lake (2.14 vs. 1.98, respectively, G_2 = 12.2, P < 0.01). In Lake Huron, clutches were significantly larger at colonies outside (numbers 4 and 5) than within (numbers 6–8) Georgian Bay (2.06 vs. 1.89, respectively, G_2 = 62.2, P < 0.001). In Lake Ontario, Hamilton Harbour clutches were significantly larger than those on Pigeon Island (G_2 = 17.4, P < 0.001) (Table 6). At Hamilton Harbour, there were two separate colonies in 1991. One laid two weeks later than the other. Mean clutch size at the "early" colony was significantly higher than at the "late" colony (2.33 vs. 1.94, respectively, G_2 = 12.2, P < 0.01). Overall, clutch size did not differ significantly among lakes (G-tests), but the mean was higher in Lake Ontario (2.18 ± 0.60) and Lake Michigan (2.04 ± 0.49) than in Lake Huron (1.95 ± 0.48). There was no significant relationship between mean clutch size and mean egg volume among the 10 main study colonies (correlation analysis, r^2 = 0.001).

Reproductive success varied considerably among the six colonies studied in the three lakes. In the sample subsections or enclosures, average hatching success ranged between 47 and 85%, and the average number of young fledged per active nest (Y/AN) ranged from 0.70 to 1.61 (Table 7). The low hatching success at Cousins Island and South Watcher Island was due mostly to egg disappearance, as we recovered few addled, unhatched, or broken eggs from these nests or enclosures. Only at Channel/Shelter Island were we able to estimate reproductive performance of repeat (or late) clutches (initiated after 1 June): hatching success was slightly lower than in earlier clutches (60% vs. 75%, respectively), but very few young fledged from these late nests (0.11 Y/AN on average). A late visit to this colony, on 3 August, indicated that considerable mortality of chicks over 21 days old had occurred, so that the actual production of flying young from these late clutches was even lower than 0.11 Y/AN. Overall, in 1991, reproduction was more successful on Lake Ontario (Hamilton Harbour at least) and Lake Michigan than on Lake Huron.

Although our colony visits were too infrequent to provide detailed figures for causes of egg loss and breeding failure, a variety of mortality factors was noted. Predation of eggs by Herring Gulls and, more commonly, Ring-billed Gulls was witnessed on several occasions, and predated or pierced eggs were found in small numbers at all colonies. Unhatched eggs (i.e., cold or rolled well out of nest scrapes) were examined at four Lake Huron colonies and at Pigeon Island. Of 14 Lake Huron eggs for which the contents were not rotten, seven (50%) contained a dead embryo; at Pigeon Island, the corresponding figure

Table 7
Hatching success and reproductive output at Great Lakes Caspian Tern colonies in 1991[a]

Colony	No. of active nests[b]	Total no. of eggs	Hatching success (%)	No. of young fledged[c]	No. fledged per active nest	Method[d]
Lake Michigan						
1. Green Bay[e]	59	129	79	63	1.07	A
2. High I.	56	118	85	63	1.13	A
Lake Huron						
4. Cousins I.	28	58	47	22	0.79	B
5. Channel/Shelter I.	54	103	75	39	0.70	A
Channel/Shelter I. (re-lays)	30	58	60	–	0.11[f]	A
8. South Watcher I.	41	79	52	34	0.83	B
Lake Ontario						
9. Hamilton Hbr.	31	69	75	50	1.61	B

[a] All data refer to first clutches unless indicated otherwise.
[b] Number of nests containing at least one egg in colony subsection sampled.
[c] Based on number of chicks of median age of about 21 days, after which prefledging mortality is usually very low.
[d] A = observations in unfenced subsection of colony; B = observations in fenced enclosure of part of colony.
[e] Data combined for Gravelly Island and Gull Island.
[f] Calculated from entire colony, not the sample quadrat.

Table 8
Regression statistics for Caspian Tern chick growth at three colonies in 1991

Colony	No. of chicks	Mass:wing regression Slope ± SE	r^2	Predicted mean mass (g) at 200-mm wing
4. Cousins I.	22	1.62 ± 0.17	0.82	469
8. South Watcher I.	34	1.58 ± 0.29	0.48	491
9. Hamilton Hbr.	50	1.33 ± 0.16	0.62	605

was only 27% (three of 11 eggs). No embryonic deformities were detected. The remaining eggs either were infertile or had died at a very early stage of development.

Limited predation of chicks (both small and large) by Herring Gulls was recorded, even involving a healthy fledgling Caspian Tern at Cousins Island. Dead chicks of all ages were found in small numbers within most study plots. No deformities or abnormalities of external features could be detected on any of the approximately 500 chicks we inspected in 1991.

However, at Channel/Shelter Island in Saginaw Bay, more than half the chicks known to have died exhibited signs of the so-called "wasting syndrome" (Peterson et al. 1984): 18 (53%) of 34 among early nesters, and 49 (60%) of 81 among late nesters. These chicks appeared to be lethargic and emaciated, yet apparently food was not scarce, as some dying chicks had a full gullet and were attended by adults. Ludwig et al. (1993) provided detailed observations on this phenomenon at a Saginaw Bay colony (Channel/Shelter Island) in the year prior to (no wasting) and five years following (19–77% incidence of wasting) a 100-year flood event, which resuspended large volumes of contaminated sediments. Chicks exhibiting wasting syndrome were found in each of the five years after the flood and averaged 51% of those assessed for causes of death. Furthermore, the highest rates of embryo abnormalities at any Great Lakes Caspian Tern colony were found at this site (Ludwig et al. 1993). These reproductive abnormalities have been linked to egg concentrations of the no- and mo-PCBs at this site (Ludwig et al. 1993; Yamashita et al. 1993). One chick at Cousins Island also exhibited signs of wasting syndrome. A small chick at South Limestone Island had choked to death on a large alewife. Only two adults were found dead, both in the South Watcher Island colony, but the cause of death was not obvious.

An index of relative chick growth rates was obtained at three Ontario colonies by regressing chick mass on wing length for chicks in the enclosures. The variance about the regression line was much greater in Georgian Bay than in the North Channel or in Lake Ontario (Table 8). Chicks with a wing chord of 200 mm (representing a large chick about three-quarters grown) were on average 23–29% heavier (using the regression equations) at Hamilton Harbour than at either of the Lake Huron colonies (Table 8).

4.4 Diet

A wide range of items was found in pellets analyzed individually (Table 9) and as pools (Table 10). Overall, fish were the predominant prey, particularly Centrarchids, alewife, and yellow perch *Perca flavescens*. With the exception of Channel/Shelter Island in Saginaw Bay, alewife almost certainly predominated in the alewife/gizzard shad category. Bird down or feathers were found in up to 65% of pellets at a colony, but in only a few cases were more than 10 feathers present. However, some pellets comprised almost entirely larid eggshell, down, or feathers, probably of either Caspian Tern or Ring-billed Gull, indicating that chicks and/or eggs had been eaten, either scavenged or taken live. Remains of beetles, dipterans, and other insects were found in up to 20% of pellets, but never in large quantities within a given pellet. Vegetation and grit occurred in many pellets, often (but not always) adhering to the outer parts of the pellet, suggesting that they may not have been ingested by the bird. Short lengths of monofilament line and plastic fragments (up to 1 cm long) were recorded in pellets from three colonies in Lake Huron.

Temporal variation in diet composition was investigated at four colonies in Lake Huron. Groupings were usually Centrarchids, alewife/gizzard shad, other fish, and bird remains. There was no significant variation from May through July at South Watcher Island (P = 0.2) or between May and June diets at South Limestone Island

Table 9
Diet composition of Caspian Terns at Great Lakes colonies in 1991

	% frequency of occurrence in sample of pellets for each date											
	Lake Ontario		Lake Huron								Lake Michigan	
Colony[a]:	P.I.	H.H.	South Watcher I.			South Limestone I.		Halfmoon I.		C.I.	H.I.	G.I.
Date:	10 May	13 May	6 Jun	3 Jun	2 Jul	6 May	6 Jun	14 May	5 Jun	9 Jul	4 Jun	3 Jun
Prey type / No. of pellets:	31	25	20	23	26	22	17	24	26	45	20	31
Centrarchidae	74	–	90	96	77	77	88	71	65	73	20	3
Alewife *Alosa pseudoharengus*/ gizzard shad *Dorosoma cepedianum*	29	96	25	44	12	32	18	63	54	18	90	100
Yellow perch *Perca flavescens*	39	–	20	35	39	5	6	–	–	22	25	–
Rainbow smelt *Osmerus mordax*	3	–	5	–	4	5	6	4	–	7	15	10
Pike (*Esox* spp.)	–	–	–	–	–	5	6	17	–	7	–	–
Salmonidae	–	–	–	–	–	5	–	–	–	–	–	–
Cyprinidae	–	–	–	–	4	5	–	–	–	–	–	–
Common carp *Cyprinus carpio*	–	–	–	–	–	–	–	4	–	–	–	–
Shiner (*Notropis* spp.)	3	–	–	–	–	–	–	–	–	–	–	–
American eel *Anguilla rostrata*	–	–	–	–	–	–	6	–	–	2	–	–
Trout-perch *Percopsis omiscomaycus*	–	–	–	4	–	–	6	–	–	–	–	–
White bass *Morone chrysops*	3	–	–	–	–	–	–	–	–	–	–	–
White perch *Morone americana*	7	–	–	–	–	–	–	–	–	–	–	–
Unidentified fish	–	8	25	22	4	23	–	21	27	18	–	7
Small mammal	–	–	–	–	–	9	–	–	–	–	–	–
Bird feathers/down	7	20	35	65	46	27	24	38	8	42	60	13
Bird eggshell	3	–	–	–	4	–	–	4	–	–	–	–
Crayfish spp.	–	–	–	4	–	–	–	–	19	–	–	–
Invertebrate spp.	7	–	20	9	–	14	–	–	–	–	–	–
Insect spp.	13	8	–	4	12	–	–	13	–	16	5	–
Beetle	–	–	5	4	–	–	12	8	–	4	–	–
Mollusc	–	–	–	4	–	–	–	–	–	–	–	–
Vegetation	–	4	40	57	4	73	24	92	58	42	10	–
Monofilament/plastic	3	–	–	–	–	9	–	–	–	–	–	–
Grit	–	8	90	96	77	5	12	13	19	89	5	–

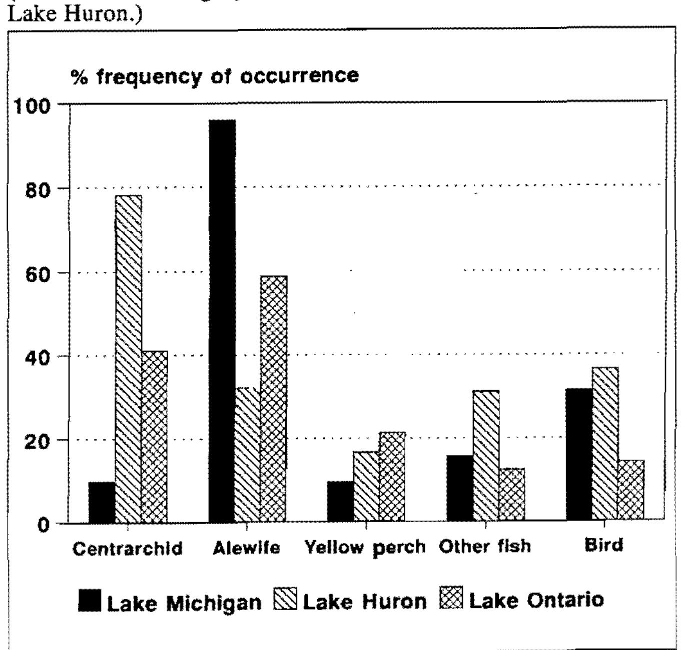
[a] Colony locations are shown in Figure 1. P.I. = Pigeon Island; H.H. = Hamilton Harbour; C.I. = Cousins Island; H.I. = High Island; G.I. = Gull Island.

Table 10
Diet composition of Caspian Terns in 1991 at Lake Huron and Lake Michigan colonies for which pellets were not stored individually

	% of total minimum no. of individual prey items in pooled sample of pellets						
	L. Huron						L. Michigan
Colony[a]:	Channel/Shelter I.					C.I.R.	H.I.
Date:	8 May	23 May	29 May	11 Jun	28 Jun	3 Jun	11 Jul
Prey type[b] / No. of pellets:	56	59	136	60	46	27	16
Centrarchidae	24	7	0.3	6	18	4	31
Alewife/gizzard shad	3	70	90	62	27	82	31
Yellow perch	21	3	0.3	4	34	2	13
Rainbow smelt	3	1	–	–	2	2	–
Pike spp.	3	–	–	–	–	–	–
Sucker (*Catostomus* spp.)	6	–	0.3	2	5	–	–
Cyprinidae	–	–	–	–	2	–	–
Common carp	3	–	–	–	–	–	–
Shiner spp.	–	1	–	–	–	–	–
Bullhead/catfish (*Ictalurus* spp.)	6	3	3	6	–	–	–
Freshwater drum *Aplodinotus grunniens*	9	4	2	–	–	–	–
Unidentified fish	3	–	–	–	–	2	6
Bird feathers/down	3	1	0.3	2	–	2	–
Bird eggshell	–	1	–	2	–	–	–
Mud puppy *Necturus maculosus*	–	–	–	–	2	–	–
Crayfish spp.	–	–	–	–	–	–	13
Insect spp.	3	1	0.3	7	2	2	6
Mullusc	12	7	4	11	7	2	–
Vegetation[c]	√	√	√	√	–	–	√
Monofilament/plastic[c]	–	–	√	–	–	√	–
Grit[c]	√	√	√	√	√	√	√

[a] Colony locations are shown in Figure 1. C.I.R. = Charity Island Reef; H.I. = High Island.
[b] For scientific names not shown here, see Table 9.
[c] Types of pellet contents merely scored as present (√) or absent (–).

Figure 3
Caspian Tern diet composition in 1991, by frequency of occurrence of prey types in samples of 51 pellets from Lake Michigan, 203 from Lake Huron, and 56 from Lake Ontario. (Note: alewife category includes some gizzard shad, mostly in Lake Huron.)



or Halfmoon Island (P > 0.4). At Channel/Shelter Island, alewife/gizzard shad predominated in May pellets (77% of the MNIs), whereas a significantly wider range of prey types was found in June (χ^2= 55.1, 3 d.f., P < 0.001) (Table 9). As temporal variation was significant at only one of these four colonies, data from different dates were pooled for subsequent analyses.

Within Lake Huron, there was no significant variation in diet composition among colonies (P > 0.07), but there was for Lake Ontario's two colonies in mid-May (χ^2 = 38.9, 1 d.f., P < 0.001) and in Lake Michigan in early June (χ^2 = 9.4, 2 d.f., P = 0.009) (Table 9). Comparing overall diets by lake, there were highly significant differences (χ^2 = 90.4, 8 d.f., P < 0.001). The main prey type in pellets from Lake Michigan was alewife, whereas alewife, yellow perch, and Centrarchids predominated in Lake Ontario (gizzard shad are not found in Lake Ontario), and Centrarchids plus a wider range of other prey items were found in Lake Huron pellets (Fig. 3).

4.5 Population changes

Numbers of Caspian Tern nests have increased dramatically at many colonies on the Great Lakes since the late 1970s, with increases over the last 15 years of 6% on Lake Michigan and 23% on Lake Ontario, whereas numbers have remained stable on Lake Huron (Table 11). Decreases have been seen at four colonies, averaging as much as 14% per annum at Channel/Shelter Island in Saginaw Bay. The steepest increases (30% per annum, on average) have occurred at Hamilton Harbour, where the colony was founded in 1986, probably by breeders from the now-abandoned colony on the Toronto Islands, 50 km to the east (Dobos et al. 1988; Blokpoel and Tessier 1991).

Table 11
Recent changes in the number of Caspian Tern nests at selected colonies on the Great Lakes

Colony	1991	No. of nests Previous[a]	(year)	Source	Average % change per annum[b]
Lake Michigan					
1. Gravelly I.	738	580	(1978)	A	+ 1.9
Gull I.	313	22	(1977)	A	+ 19.0
2. Hat I.	650	685	(1977)	A	− 0.4
High I.	1233	315	(1978)	A	+ 10.5
3. I. aux Galets	146	280	(1978)	A	− 5.0
Total	*3080*				*+ 6.3*[c]
Lake Huron					
4. Cousins I.	467	395	(1980)	B	+ 1.5
Charity I. Reef	78	0	(1975–78)	A	+
5. Channel/Shelter I.	199	305	(1988)	C	− 14.2
6. Halfmoon I.	196	259	(1980)	B	− 2.5
7. South Limestone I.	363	334	(1980)	B	+ 0.8
8. South Watcher I.	793	523	(1980)	B	+ 3.8
Thunder Bay I.	45[d]	0	(1975–78)	A	+
Total	*2141*				*+ 0.3*[c]
Lake Ontario					
9. Hamilton Hbr.	220	48	(1986)	D	+ 30.4
10. Pigeon I.	503	40	(1976)	E	+ 16.9
Gull I.	294[e]	87	(1987)	F	+ 20.3
Little Galloo I.	555[f]	112	(1986)	G	+ 26.7
Total	*1572*				*+ 22.9*[c]

Sources: A = Shugart et al. 1979; B = Weseloh et al. 1986; C = Ludwig et al. 1993; D = Dobos et al. 1988; E = Blokpoel and Tessier 1991; F = Richards and McRae 1988; G = Weseloh and Blokpoel 1993.

[a] Reliable counts only.
[b] Growth rate (GR) calculated from initial count (IC) and recent count (RC) n years later, according to formula adapted from Ricklefs (1980): GR = [ln (RC) − ln (IC)] / n.
[c] Mean % change per annum, weighted by 1991 colony size.
[d] Estimated number of nests.
[e] 1993 count (CWS, unpubl. data).
[f] 1992 count (CWS, unpubl. data).

5. Discussion

5.1 Contaminant levels

Although a wide range of OCs was detected in Great Lakes Caspian Tern eggs from each colony studied in 1991, their concentrations were markedly lower than in 1980–81 (Struger and Weseloh 1985) or during the 1970s (Gilbertson and Reynolds 1974; Martin 1978; Bishop et al. 1992a, 1992b). The 7% increase in mirex levels in Cousins Island eggs over the past 11 years was unexpected, because mirex levels in Herring Gull eggs from nearby Double Island decreased by 31% over the same period (Bishop et al. 1992a, 1992b; CWS, unpubl. data). As the main sources of mirex on the Great Lakes are in Lake Ontario (Oswego, New York) and the Niagara River (Kaiser 1978), this increase could reflect a greater amount of time spent by North Channel Caspian Terns on Lake Ontario during migration than was formerly the case. Mirex levels at the two Lake Ontario colonies were much higher than levels found elsewhere in 1991, although moderately high residue levels at South Limestone Island and South Watcher Island also suggest that birds breeding there spent some time on Lake Ontario. Adult Caspian Terns breeding on the Great Lakes appear to use two different migration routes to and from the wintering areas — the U.S. Atlantic coast, and the Mississippi valley flyway (Ludwig 1965; L'Arrivée and Blokpoel 1988). It is likely that most birds breeding in northern and western parts of the Great Lakes use the Mississippi route and so are not exposed to as high mirex levels as birds breeding in Georgian Bay and Lake Ontario, which feed on Lake Ontario fish at some stage each year.

Spatial variation in OC levels in Caspian Tern eggs in 1991 broadly matched that seen in recent studies of eggs of other fish-eating bird species (Government of Canada 1991; Tillitt et al. 1991; Bishop et al. 1992a, 1992b; Yamashita et al. 1993; Pettit et al. 1994a, 1994b). Eggs from Saginaw Bay (Lake Huron) and Green Bay (Lake Michigan) — both historically polluted, heavily industrialized areas — still have the highest levels of most contaminants. Concentrations of OCs in plasma of incubating adult Caspian Terns showed geographic variation very similar to that seen in eggs from Lake Michigan and Lake Huron colonies (Mora et al. 1993; this study), suggesting that egg OC levels are indicative of contaminant levels in fish on the breeding grounds.

Table 12
Mean levels of DDE, mirex, and PCBs in eggs of Caspian Terns and Herring Gulls from three Great Lakes in 1980–81 and 1991[a]

	Lake Michigan[b] Caspian Tern	Lake Michigan[b] Herring Gull[d]	Lake Huron[c] Caspian Tern	Lake Huron[c] Herring Gull[e]	Lake Ontario[c] Caspian Tern	Lake Ontario[c] Herring Gull[f]
	Concentration ($\mu g \cdot g^{-1}$ wet weight ± SD)					
No. of colonies:	3	1–2	5	2–3	2	1–3
DDE						
1980–81	6.4 ± 3.7	12.2 ± 4.9	3.9 ± 1.6	3.0 ± 1.2	5.2 ± 1.6	11.8 ± 4.5
1991	2.9 ± 1.2	8.0	2.6 ± 0.9	3.0 ± 1.8	3.6 ± 0.3	3.3 ± 0.7
Mirex						
1980–81	0.05 ± 0.02	0.11 ± 0.10	0.16 ± 0.14	0.12 ± 0.11	1.57 ± 0.53	2.8 ± 1.6
1991	0.03 ± 0.01	0.05	0.09 ± 0.05	0.09 ± 0.03	0.74 ± 0.03	0.56 ± 0.11
PCBs[g]						
1980–81	30.2 ± 9.3	57.8 ± 17.9	25.4 ± 9.9	17.5 ± 6.9	39.3 ± 17.8	85.6 ± 40.7
1991	12.5 ± 2.9	24.9	11.5 ± 3.9	11.9 ± 3.2	16.9 ± 0.9	16.8 ± 2.8
DDE:PCB ratio						
1980–81	0.21	0.21	0.15	0.17	0.13	0.14
1991	0.23	0.32	0.22	0.24	0.21	0.20

[a] 1980–81 data from Struger and Weseloh (1985); Bishop et al. (1992a, 1992b); 1991 Herring Gull data from Pettit et al. (1994a, 1994b). Values for the 1991 Herring Gull eggs were calculated as the mean of egg pools for each colony, whereas 1980–81 data are means of individual egg analyses (colony locations are shown in Figure 1).
[b] Collections made in 1980.
[c] Collections made in 1981.
[d] Big Sister Island, Gull Island (1980); Big Sister Island (1991).
[e] Double Island, Castle Rock Island (1980); Chantry Island, Double Island, Manitoba Reef (1991).
[f] Snake Island (1981); Snake Island, Leslie Spit, Hamilton Harbour (1991).
[g] Expressed as 1:1 ratio of Aroclor 1254:1260 (for direct comparison with data in Struger and Weseloh 1985).

In Saginaw Bay and at colonies in northeastern Lake Michigan, levels of DDE, ΣPCBs, and individual PCB congeners — including the three major contributors to dioxin-like toxicity, PCB-126, PCB-118, and PCB-105 — did not change dramatically from the levels in eggs collected during late incubation in 1988 from the same colonies (Yamashita et al. 1993). However, concentrations of PCB-77 appeared to have decreased by 64–87% in these areas by 1991. No discernible pattern emerges with PCDDs and PCDFs. Some levels appear to have increased, others to have decreased. It is probable that these changes are within the natural range of variation in the system. A TEQ of 1.60 $\mu g \cdot kg^{-1}$ was recalculated by Ludwig et al. (1993) from the data for fresh Caspian Tern eggs from Channel/Shelter Island reported in Yamashita et al. (1993). Ludwig et al. (1993) used the TEF values of Safe (1990), so their results can be compared directly with our almost identical value of 1.61 $\mu g \cdot kg^{-1}$ in 1991.

Mean concentrations of DDE, mirex, and PCBs in eggs of Caspian Terns and Herring Gulls in 1991 were very similar in both Lake Huron and Lake Ontario, but residue levels in Lake Michigan Caspian Tern eggs averaged about 50% lower than those in Herring Gull eggs (Table 12). In the 1980–81 study, a similar result was found for Lake Michigan, but average Caspian Tern egg residue levels for Lake Huron were 30–40% lower than Herring Gull egg residue levels, and in Lake Ontario they were about 50% lower (Struger and Weseloh 1985). Herring Gulls are resident as adults on the Great Lakes (Moore 1976; Gilman et al. 1977), whereas Caspian Terns are migratory (Ludwig 1965; L'Arrivée and Blokpoel 1988). As the diet of both species comprises predominantly fish of broadly similar sizes and species (Fox et al. 1990; Ewins et al. 1993; P.J. Ewins, pers. obs.), one might expect higher OC levels in Herring Gull eggs, as contaminant levels in aquatic habitats are not known to be as high in Caspian Tern wintering areas (southeastern U.S. coastal states and the Caribbean mostly) as on the Great Lakes (Nriagu and Simmons 1984). In 1991, this was true only for Lake Michigan eggs.

A number of different factors must be considered when interpreting differences in contaminant concentrations between these two species. Although diets are broadly similar, the size and proportion of different fish species consumed may be important, as fish OC levels vary among species (Government of Canada 1991) and with age (Borgmann and Whittle 1991). We cannot rule out the possibility that some Herring Gulls now consume substantial amounts of nonfish, terrestrial items, particularly garbage, during the winter and early spring (Ewins et al. 1993) and the breeding season (Allan 1977; Fox et al. 1990; CWS, unpubl. data) in Lakes Huron and Ontario, resulting in unexpectedly low egg OC concentrations (Table 12). Also, Caspian Terns are smaller than Herring Gulls (about 40% lighter; Cramp and Simmons 1983; Cramp 1985) and have a higher metabolic rate (King and Farner 1961), and so they would be expected to bioaccumulate lipophilic OCs at a faster rate, given the same diet, feeding, and climatic conditions.

Another notable difference between Caspian Terns and Herring Gulls is seen with egg levels of 2378-TCDD. Since 1984, Herring Gull eggs from Saginaw Bay (Channel/Shelter Island) and Lake Ontario have consistently had 2–4 times higher concentrations of 2378-TCDD than at any other Great Lakes colonies (Hebert et al. 1994). No such spatial pattern is seen for Caspian Terns (Table 3) or Double-crested Cormorants (D.V. Weseloh, P. Hamr, G.A. Fox, P.J. Ewins, and R.J. Norstrom, unpubl. data). Based on the similar biomagnification factors of 2378-TCDD and HCB (32 and 31, respectively) from alewife to Herring Gulls (Braune and Norstrom 1989), the half-lives are expected to be similar. The half-life for HCB in a wild adult female gull has been estimated to be 107 days (Clark et al. 1987). This suggests that

2378-TCDD is only a moderately persistent compound in Herring Gulls compared with compounds like mirex, which have a half-life of around 300 days (Clark et al. 1987). If the same is true for terns, a good portion of the observed egg 2378-TCDD residues may not have been accumulated locally at the breeding areas, unless dietary levels were very high there during the egg formation period. Lake Michigan breeders may migrate along the Mississippi River watersheds (Ludwig 1965; L'Arrivée and Blokpoel 1988) and so would not be exposed to the elevated levels of compounds such as 2378-TCDD or mirex found in Lake Ontario fish. The unexpectedly high 2378-TCDF (and 2378-TCDD) levels at Cousins Island in the North Channel presumably reflect elevated levels in local fish, but we have no comparable data from other biota with which to verify this.

In summary, we suspect that the relatively high egg contaminant concentrations in (migratory) Caspian Terns compared with (resident) Herring Gulls on the Great Lakes reflects primarily a higher rate of intake of lipophilic contaminants by the terns, rather than any major difference in chemodynamics or hepatic detoxification mechanisms between the species.

The average DDE:PCB ratios for Caspian Tern eggs were much more consistent among lakes and colonies in 1991 than in the 1980–81 study (Table 12), indicating similar routes of contaminant uptake. The Lakes Huron and Ontario ratios have increased, reflecting mostly the relatively small reductions in DDE levels when compared with Lake Michigan. Significant increases in DDE:PCB ratios in Herring Gull eggs since 1979 have been noted for all the Great Lakes except Lake Erie (K.D. Hughes, D.V. Weseloh, and B.M. Braune, unpubl. data). The 1991 ratios were very similar for Caspian Terns and Herring Gulls in both Lake Huron and Lake Ontario, but for Lake Michigan Herring Gulls the ratio was 50% higher than elsewhere, suggesting different contaminant exposure. In general, spatial variation in contaminant exposure within the Great Lakes has decreased markedly as point sources have been identified and discharges regulated. Contaminant return from sediments and atmospheric input is now a dominant source throughout much of the Great Lakes (e.g., Eisenreich et al. 1983; Bahnick and Markee 1985). In migratory birds that nest on the Great Lakes, contaminant sources in wintering and staging areas now exert a much greater influence than 10–30 years ago.

5.2 Reproduction and contaminant effects

Reproductive failure has been associated with DDE-induced eggshell thinning in many species of birds (Ratcliffe 1970; Wiemeyer and Porter 1970; Cooke 1973; Blus 1982). Our results indicate that Caspian Tern eggs throughout the Great Lakes in 1991 showed no signs of shell thinning at mean DDE residue levels of up to 4 $\mu g \cdot g^{-1}$ wet weight. The pre-DDT RTI value of 1.537 $mg \cdot mm^{-2}$, which we calculated, was 1.7% higher than the 1.511 value quoted by Martin (1978) and Struger and Weseloh (1985). Unlike the other studies, our figures include an adjustment for the shell removed for blowing out the contents, but that increased the RTI only by 0.2% on average. These were different samples of pre-DDT eggs and were measured by two different individuals, which may account for the slight discrepancy.

As pre-DDT eggs were significantly smaller than eggs collected during the DDT era on the Great Lakes, the mean RTI would have been significantly lower after the introduction of DDT if shell thickness had not changed. Therefore, interpretation of changes in RTI values must also take full account of egg volume changes. This has seldom been done in avian studies. Lake Michigan eggs had significantly thicker shells and RTIs in 1991 than those from other lakes, but this appeared to be due mainly to their being significantly larger.

The average clutch sizes recorded in 1991 were within the range of values found in other studies on the Great Lakes: in Lake Michigan, 2.81 (early clutches only; Ludwig 1965), 2.0–2.5 (Shugart et al. 1979), and 2.12–2.26 (Mora et al. 1993); in Lake Ontario, 2.00–2.30 (Haymes and Blokpoel 1978; Fetterolf and Blokpoel 1983); in Saginaw Bay, 2.04–2.22 (Mora et al. 1993; J.P. Ludwig, unpubl. data); in Georgian Bay, 1.87–2.15 (Quinn 1980; Mora et al. 1993; P.J. Ewins, A. Snedden, and D.V. Weseloh, unpubl. data); and in the North Channel, 2.19–2.21 (Mora et al. 1993). But why were Georgian Bay clutches in 1991 significantly smaller than elsewhere? Egg volumes in southern Georgian Bay were also among the lowest recorded in 1991. Clutch size and egg mass (or volume) have been found to vary with food availability in many species of birds, including terns and other larids (Klomp 1970; Lemmetyinen 1973; Gaston and Nettleship 1982; Pierotti and Bellrose 1986; Martin 1987; Hiom et al. 1991), although such relationships have not always been detected (Morris 1986; Safina et al. 1988; Bolton et al. 1992). Caspian Terns in the Baltic Sea laid smaller clutches when preferred fish prey were less readily available (Soikelli 1973). Although fisheries data are scarce for forage fish taken by terns, other piscivorous bird species breeding in Georgian Bay in the early 1990s (e.g., Ospreys *Pandion haliaetus*) appeared to experience food shortages (P.J. Ewins, unpubl. data). Thus, we feel that the quality and/or quantity of available prey may have limited egg and clutch size of Caspian Terns in Georgian Bay, and perhaps other parts of Lake Huron, in 1991.

In 1991, the lower productivity figures from Lake Huron Caspian Tern colonies were accompanied by chicks being up to 29% lighter than in Lake Ontario, suggesting that they were receiving inadequate food. Although we have no pellets from the Hamilton Harbour colony during the nestling period, in May these birds were feeding mainly on alewife, which have a high lipid content and high energy equivalence (1.64 $kcal \cdot g^{-1}$ wet weight; Rottiers and Tucker 1982). In contrast, at the two Lake Huron colonies at which chick growth rates were investigated, the pellets collected during the nestling period (and also in May, in the case of South Watcher Island) contained mostly Centrarchids and yellow perch, species that have lower lipid levels (Dugal 1962) and lower energy content per gram (0.79–1.41 $kcal \cdot g^{-1}$ wet weight; Niimi 1972; Watt and Merrill 1975; Craig 1977) than alewife. Variation in local feeding conditions and forage fish availability (as opposed to persistent contaminants) probably accounted for much of the observed variation in reproductive success of Caspian Terns among Great Lakes colonies in 1991.

Ludwig (1965) estimated the annual adult (>3 years old) mortality rate of Great Lakes Caspian Terns to be 11%; combining this with estimates of prebreeding mortality generated from banding data, he calculated that an average reproductive output of 0.6 young per pair per year could maintain population stability. Hickey (1952) calculated the annual mortality rate after the second year to be 18%, based on analyses of band recovery data for Caspian Terns elsewhere in North America. However, adult survival rates can vary substantially within the Great Lakes (Ludwig 1979), and other studies of larids have found large annual differences in adult survival (e.g., Aebischer and Coulson 1990). Furthermore, it is now generally accepted that age-specific survival rates cannot be calculated reliably from recovery data of birds banded only as chicks (Lakhani and Newton 1983; Anderson et al. 1985). Therefore, previous calculations of Caspian Tern survival rates and break-even productivity values should be taken only as a rough indication.

Thus, in the absence of better data, an annual production of 0.6 young per pair might represent a rough minimum "break-even" point for Caspian Terns on the Great Lakes. The 1991 figures from initial clutches were all above this value, but only slightly so in Lake Huron colonies (Table 7). On this basis, there was no evidence for any overall adverse effect of OCs on reproductive output in 1991. Weather conditions were very favourable for breeding in 1991 for many colonial birds, as there were few periods of prolonged cold or strong winds during May or June. Reproductive output was also well above this break-even threshold in 1990 in Lake Michigan (0.9–1.08) and in Saginaw Bay (1.06) (Mora et al. 1993).

Similar studies in California found few adverse reproductive effects in Caspian Terns, with higher egg DDE residue levels (geometric mean 9–10 $\mu g \cdot g^{-1}$, range 3–56 $\mu g \cdot g^{-1}$), but much lower egg PCB residue levels (mean 2 $\mu g \cdot g^{-1}$, maximum 8 $\mu g \cdot g^{-1}$) (Ohlendorf et al. 1985). However, 5% of chicks died at hatching, which was thought to be a relatively high frequency. Shell thickness index (but not actual shell thickness) was significantly correlated (negatively) with DDE levels but averaged only 5% thinner in 1981 than pre-DDT values (Ohlendorf et al. 1985). In Texas, similar relatively low levels of DDE and PCBs were found in Caspian Tern eggs in 1984, and residues of OCs and metal contaminants in eggs of Forster's Terns *Sterna forsteri* and Black Skimmers *Rynchops niger* from the same area were below embryotoxic and observable effect levels (King et al. 1991). At South Limestone Island in 1972, Caspian Tern eggs had much higher concentrations of DDE (16 $\mu g \cdot g^{-1}$ wet weight) and total PCBs (33 $\mu g \cdot g^{-1}$ wet weight) (Table 2), yet hatching success was reasonably high (71%), and production was at least 0.4 young per active nest (P.J. Ewins, A. Snedden, and D.V. Weseloh, unpubl. data), indicating that this species was much less sensitive to contaminants than were Double-crested Cormorants nesting on the Great Lakes at that time (Weseloh et al. 1983).

Hatching success of Common Terns *Sterna hirundo* in Germany was only slightly lower at a contaminated site (69%) than at a relatively uncontaminated colony (73%), but predators accounted for much of the egg loss. However, levels of total PCBs (and particularly those of the higher-chlorinated congeners) were significantly higher in unhatched (failed) eggs than in freshly laid eggs, suggesting some intrinsic embryotoxic effect (Becker et al. 1993). In another recent ecotoxicological study of Common Terns at Dutch colonies exposed to differing levels of industrial contaminants, field results indicated no impairment of reproductive success attributable to organochlorine pollutants — flooding, adverse weather, and predation were the most important factors (Rossaert et al. 1993). Although some biochemical changes in chicks studied in the laboratory were associated with higher levels of PCBs and PCDDs/PCDFs in the Dutch study, no effects on chick morphology or growth were detected (Murk et al. 1993). At higher levels of "dioxin"-like halogenated polyaromatic compounds, significant adverse changes in liver weights and growth have been found in Forster's Tern chicks in Green Bay, Lake Michigan (Hoffman et al. 1987; Kubiak et al. 1989).

At most Great Lakes colonies in 1991, hatching success of Caspian Terns was high relative to published values (Quinn 1980; Fetterolf and Blokpoel 1983; Ohlendorf et al. 1985; Mitchell and Custer 1986; Mora et al. 1993). The poor hatching success (for first clutches) at Cousins Island and South Watcher Island (47–52%) was probably due to egg predation by Ring-billed Gulls or other Caspian Terns. Any food shortages at these colonies are likely to have caused additional stress, and even reduced nest attentiveness, among adults during the incubation period.

The wasting syndrome seen in many chicks at the Saginaw Bay colony has been described in animals under laboratory conditions (Peterson et al. 1984) and in wild Forster's Terns in Lake Michigan's Green Bay (Kubiak et al. 1989). Most often, chick wasting results from exposure to 2378-TCDD or PCBs with TCDD-like toxicity — i.e., the 3,4,3′,4′-chlorine-substituted no- and mo-PCBs (Rifkind et al. 1984; Brunström and Andersson 1988). Ludwig et al. (1993) suggested that the levels of TCDD TEQs, mainly due to PCBs, were high enough in the diet of Caspian Terns in the Saginaw Bay colony to account for the wasting syndrome in chicks. Because their chemical analysis was confined to eggs, rather than the diet or wasted chicks, cause–effect and dose–response correlations cannot be established from that study. Nevertheless, it is reasonable to assume that TEQ levels in eggs are an indirect index of exposure of chicks to TCDD-like toxicities at a colony. Ankley et al. (1993) showed that the no- and mo-PCB levels were similar in eggs and 5- to 27-day-old chicks from the same nest of Forster's and Common terns. Our present data tend to support this argument. The TEQs calculated in Caspian Tern eggs in the present study and those in the 1988 study by Ludwig et al. (1993) were identical using the same TEF system (Safe 1990), and the incidence of wasting syndrome at Channel/Shelter Island was similar in the two years — 52% of dead chicks assessed in 1988 (Ludwig et al. 1993), and 63% of dead chicks in 1991 (this study).

Despite the high incidence of wasting syndrome in 1991 chicks in Saginaw Bay, average reproductive output of first clutches was still higher than the supposed break-even level of 0.6 young per nest. In 1990, average production there was 1.06 young fledged per nest, but elevated concentrations of PCBs and other organic contaminants were associated, in repeat nesting attempts, with reduced hatching success and higher mortality of chicks, compared with less contaminated colonies (Mora et al. 1993). Acute episodes of wasting syndrome were also noted at Channel/Shelter Island in 1987 and 1988 (Ludwig et al. 1993). In general, Caspian Terns seem to reproduce well on the Great Lakes despite relatively high body and egg OC burdens, and they appear to be less sensitive than Forster's Terns, at least at the population level, to the effects of PCBs (Kubiak et al. 1989; Mora et al. 1993).

5.3 Diet

Temporal shifts in diet composition can be assessed from adult and chick dietary data collected over the past 30 years (Allan 1977; Quinn 1980; H. Blokpoel, J. Gregorich, L. Still, and J.P. Ludwig, unpubl. data). In Lake Michigan, the diet appears to have changed little, remaining dominated by alewife and rainbow smelt *Osmerus mordax*. The diet at Lake Huron colonies has consistently been more diverse than in other lakes, but a steady decline in the proportion of alewife and rainbow smelt in the diet since the 1960s has continued through 1991. Centrarchids have assumed much greater importance in the diet at the same time. The Lake Ontario diet remains dominated, in general, by alewife, but rainbow smelt is now scarcely taken by Caspian Terns. At Pigeon Island, Centrarchids formed only 2–14% of chick and adult diets (by item numbers) in 1977, and only 7% in 1987, but by 1991 they were found in 74% of adult pellets. There appears to have been a steady shift in the diet over the past 20 years, at least in eastern Lake Ontario, towards fewer alewife and smelt and greater numbers of rock bass *Ambloplites rupestris* and other Centrarchids.

Marked changes have occurred over the last 40–50 years in fish populations in each of the Great Lakes (Christie 1974; Hartman 1988). We suspect that Caspian Tern diet shifts are a response by this opportunistic piscivore to changing availability of preferred prey. Non-fish items appear to have been taken more frequently in 1991 than has been found in previous studies. The surprisingly high incidence of invertebrates and larid eggs or remains of small chicks may be indicative of fish scarcity, especially at some Lake Huron colonies. Vermeer (1973) also found insects (6%), bird eggshell (3%), and bird bones (0.3%) in pellets at Lake Winnipeg colonies.

5.4 Population changes

Our sample of colonies in 1991 accounted for nearly 6800 active nests, including all colonies in Lake Ontario and most in Lakes Huron and Michigan. Caspian Terns do not nest on Lake Erie or Lake Superior. Although we do not have a complete census from 1991 on the Great Lakes, the population comprised about 5700 pairs in 1987 (Blokpoel and Scharf 1991), so our study in 1991 almost certainly covered over 80% of the breeding population. The Great Lakes Caspian Tern population has undoubtedly grown greatly (probably by at least 90%) since the 1978 count of 3600 pairs; the 1963 count was only 1995 pairs (Ludwig 1979). Similar increases have occurred in the Baltic population (Bergman 1980). The Great Lakes clearly remain a very important stronghold for this cosmopolitan species. The steepest increases have occurred on Lake Ontario and in parts of Lake Michigan, whereas Lake Huron numbers have remained relatively stable over the past decade. Blokpoel and Scharf (1991) noted the following average rates of annual increase up to 1987: 4% since 1976 on Lake Michigan, 3% since 1980 on Lake Huron, and 29% since 1976 on Lake Ontario. Since the mid-1970s, breeding Herring Gulls have also increased markedly on Lake Ontario, by 8% per annum on average (Blokpoel and Tessier 1991), but have declined somewhat on Lake Huron, by about 3% per annum between 1977 and 1989 (CWS, unpubl. data). For both species, these population trends are likely a response both to decreasing contaminant loads and to major changes in fish availability, resulting from major changes in the structure of fish communities (Christie 1974; Christie et al. 1987; Hartman 1988).

Between 1972 and 1982, numbers of Caspian Terns nesting at Canadian colonies increased by about 50% (Blokpoel 1983), whereas numbers were relatively stable over that period at the U.S. colonies, reflecting a shortage of suitable nesting islands when existing colonies became flooded (Shugart and Scharf 1983). High water levels have, in the past, removed suitable nesting habitat in Saginaw Bay (Ludwig et al. 1993; J.P. Ludwig, pers. obs.) and in northeastern Lake Michigan (Cuthbert 1985), and adults have moved to nearby colonies when possible. However, water levels in 1991 were relatively low, and we were not aware of any intraseasonal shifts that may have confused the nest counts we made.

5.5 General

In summary, Caspian Terns on the Great Lakes experienced relatively good reproductive success in 1991, and the only indication of effects likely to have been due to elevated OC levels was the high incidence of chicks exhibiting the wasting syndrome on Channel/Shelter Island in Saginaw Bay. Populations whose eggs contained much higher levels of DDE, other organochlorine pesticides, and PCBs appeared to reproduce successfully, and breeding numbers were usually increasing (Ohlendorf et al. 1985; Struger and Weseloh 1985; P.J. Ewins, A. Snedden, and D.V. Weseloh, unpubl. data). Levels of most known contaminants in Caspian Tern eggs (and other biota) have declined markedly over the past 20 years throughout much of the Great Lakes, and we found no evidence in 1991 for adverse effects at the population level.

However, Mora et al. (1993) reported a significant negative association between PCB concentrations in plasma of adult Caspian Terns in early June and the degree of fidelity to the natal region within Lakes Michigan and Huron. The lowest rates of return to the natal region were for Saginaw Bay and Green Bay, which could have been a reflection of reduced survival or reduced natal philopatry. Our productivity and population trend data are consistent with the hypothesis that Saginaw Bay offers relatively poor conditions for Caspian Terns, but our findings for Green Bay suggest that overall conditions there are relatively good for this species. Possible effects of contaminants (particularly no- and mo-PCB congeners) on parameters such as site fidelity, adult and prebreeding survival, and recruitment age should be investigated further before drawing any firm conclusion that OCs at current levels have no adverse effects on Caspian Tern populations. We suspect that factors such as the availability of forage fish, weather conditions, water levels, human disturbance, nest predation by gulls, and interspecific competition for nesting space currently exert a much greater influence on Great Lakes Caspian Terns, relative to OCs, than was the case during the 1950s to 1970s. Therefore, the use of Caspian Terns as sensitive indicators of contaminant-related biological effects, at least on the Great Lakes, seems now to be complicated by the relatively large influence of other environmental stressors on various population parameters.

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