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Source: *Waterbirds*, 31(1) : 15-23

Published By: The Waterbird Society

URL: [https://doi.org/10.1675/1524-4695\(2008\)31\[15:SOAHGL\]2.0.CO;2](https://doi.org/10.1675/1524-4695(2008)31[15:SOAHGL]2.0.CO;2)

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Survival of Adult Herring Gulls (*Larus argentatus*) from a Lake Ontario Colony Over Two Decades of Environmental Change

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Abstract.—We estimated annual survival probabilities of color-banded adult Herring Gulls (*Larus argentatus*) breeding in a colony on the Canadian shore of Lake Ontario, between 1981 and 1984, and between 1993 and 1998. During the study, major changes occurred in prey fish availability, fish-eating bird abundance, and organochlorine contaminant exposure. We used capture-mark-recapture models and an information-theoretic approach to model selection to explore the importance of indices for these environmental variables and the role of year and sex as exploratory effects. In our models, year but not sex acquired strong support suggesting important annual variation in our apparent survival (survival confounded by permanent emigration) and encounter probabilities. The arithmetic mean of our model-averaged apparent survival probabilities provided an estimate of 0.91 (0.02 SE), and four out of the seven estimates were high for the species (≥ 0.93). Our model-averaged estimates suggested that apparent survival declined after 1995 to a low of 0.79 ± 0.07 (SE). Of the three environmental covariates fitted to survival, only the abundance of Alewife (*Alosa pseudoharengus*) acquired strong support as an explanatory variable. Annual fluctuations in survival and the decline after 1995 may be a reflection of nutritional stress resulting from the decreased availability of lipid (energy) and high quality protein from this prey species, following the arrival of the exotic Zebra Mussel (*Dreissena polymorpha*). These nutritional impacts may be exacerbated by some of the PCB-associated sublethal health effects documented in these gulls over the period of this study. Our limited success at isolating other important environmental covariates reinforces the difficulty posed in identifying the critical components of the complex of environmental variables encapsulated in 'time'. Received 12 April 2007, accepted 17 August 2007.

Key words.—apparent survival, Herring Gull, *Larus argentatus*, Lake Ontario, adult, capture-mark-recapture, marker effects, organochlorines, PCBs, Alewife abundance, interspecific competition.

Waterbirds 31(1): 15-23, 2008

In North America, the Herring Gull (*Larus argentatus*) breeds coastally from central North Carolina to Baffin Island and across northern arctic Canada to the western coast of Alaska and the Aleutian Islands, and inland along the St. Lawrence River to the Great Lakes and across the northern Canadian prairie provinces (Pierotti and Good 1994). To date, estimates of adult survival of North American Herring Gulls are limited to the marine environment of the Atlantic coast (Paynter 1966; Kadlec 1976; Freeman and Morgan 1992) and Southampton Island in the Canadian arctic (Allard *et al.* 2006).

Adult Herring Gulls in the Great Lakes form what is essentially an isolated and closed population of year-round residents (Gilman *et al.* 1977; Weseloh 1984; Yauk and Quinn 1999). As an opportunistic piscivore (Fox *et al.*

1990; Ewins *et al.* 1994), the Herring Gull is potentially an excellent integrator and indicator with which to assess health and nutritional impacts associated with changes in the aquatic habitat of the five Great Lakes which contain 18% of our planet's fresh surface water.

In the 1950s and 1960s, Lake Ontario experienced numerous stresses including overfishing, exotic species introductions, eutrophication, and toxic chemical contamination which degraded water and habitat quality resulting in the decline of fish communities (Mills *et al.* 2003). By the 1970s, Lake Ontario's once diverse fish communities were dominated by non-native fish, including the Alewife (*Alosa pseudoharengus*), Rainbow Smelt (*Osmerus mordax*) and Pacific Salmon (*Oncorhynchus* spp.). The arrival of Dreissenid mussels in 1989 resulted in increased water

clarity, reduced plankton populations and a redirection of the nearshore energy production to benthic habitats (Mills *et al.* 2003). In 1990, approximately 2% of the 87,000 pairs of Herring Gulls breeding in the Great Lakes nested on Lake Ontario (Morris *et al.* 2003).

In 1970, a number of Herring Gull nests were found on Pigeon Island in eastern Lake Ontario in which the eggs were dented or collapsed (Edwards 1970). Follow-up studies of other colonies in eastern Lake Ontario from 1972-1976 revealed extremely poor breeding success (Gilbertson 1974; Gilbertson and Hale 1974a, b; Teeple 1973; Gilman *et al.* 1977). Their poor hatching success was associated with extrinsic (behavioral) and intrinsic (toxic) factors (Fox *et al.* 1978; Peakall and Fox 1987). Their eggs contained very high concentrations of organochlorine contaminants including polychlorinated biphenyls (PCBs), dichlorodiphenyldichloroethane (DDE), hexachlorobenzene, and 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD) and their eggshells were 9-16% thinner than normal (Gilbertson 1974; Norstrom and Hebert 2006). Although their breeding success improved in 1978 and subsequently, their thyroids remained enlarged and their biomarker responses impaired over the period of this study (Ellenton *et al.* 1985; Moccia *et al.* 1986; Fox *et al.* 1998, 2007).

These observations led to the adoption of the Herring Gull as an indicator of Great Lakes water quality in 1974, and the establishment of an egg contaminants monitoring program throughout the Great Lakes Basin (Hebert *et al.* 1999). A biological research program was initiated in 1978 to support the monitoring program which included an intensive study of a colony in Presqu'ile Provincial Park in eastern Lake Ontario. Our primary goals were to better understand the behavior and physiology of this integrator species in the Great Lakes, monitor pollutant dynamics and effects on marked individuals, and obtain estimates of survival and reproductive success. The colony was studied intensively from 1978-1984 and less intensively from 1993-1998 when the focus was primarily on adult survival and reproductive success. Here we report our findings for adult (ap-

parent) survival and encounter probabilities over two intervals 1981-1984, and 1993-1998, based on a mark-recapture analysis of a subset of our data, and examine the contribution made by changes in forage fish availability, interspecific competition for fish, and exposure to organochlorine contaminants.

STUDY AREA AND METHODS

Study Area

The study was conducted on Gull Island in Presqu'ile Provincial Park, on the north shore of Lake Ontario (48.9817°N-77.7433°W), immediately south of Brighton, Ontario, Canada. Gull Island is an 800 m long shingle and gravel bar 725 ha in area with a protected bay and small interior pond. It is 300-500 m from the mainland depending on water conditions. The number of Herring Gulls nesting there ranged from 82 to 166 pairs during the period of this study.

Sampling

Capture-Mark-Resight/Recapture (CMR) data were collected on Gull Island over two periods: 1981-84, 1993-98. Adult gulls were routinely trapped on their nests using self-triggered drop traps (Mills and Ryder 1979). Body measurements were used to determine sex by a discriminant function (Fox *et al.* 1981).

Marking, Resighting and Recapturing

All birds considered here received a unique combination of five colored plastic bands and a size 6 USGS aluminum band (hereafter, "metal band"). Butt-end style metal bands were used from 1981-1984, and lock-on bands from 1993 to 1998; badly worn or illegible bands were replaced on recaptured birds. Plastic bands were ten mm tall by eleven mm (internal diameter) coils manufactured by J.E. Warner (Durham City, UK) from Darvic® rigid PVC plastic (ICI, UK) in the colors red, orange, yellow, dark green, blue, black, white, and light green. The left leg was given a combination of three color bands, the right leg the metal band, over a yellow "investigator identifier" band and another color. The metal and yellow bands were always applied in the same position, and no combinations were deployed with two adjacent bands of the same color. Light green bands were included in all combinations applied from 1993-1998 as a unique mark for this group.

In each year, color-banded gulls were resighted during regular checks of marked nests and loafing areas. Median dates of the first and last colony visits were 24 April and 20 June, and 11 May and 7 June, for the 1981-84 and 1993-98 periods, respectively. The number of resighting days spent in the colony ranged from twelve to 27 (mean = 19) for the 1979-1984 period and from three to twelve (mean = 8) for the 1993-1998 period, a decrease of 58%, reflecting the greatly scaled back objectives of the study (Table 1).

Capture-Mark-Resight/Recapture Analysis

Data Set. The analysis was based on 274 individuals color-banded from 1981-84 (N = 129) and 1993-97 (N =

Table 1. Temporal variation in resighting effort, number of nesting pairs, and the three environmental covariates that were fitted to apparent survival probabilities for adult herring gulls marked with colored bands on Gull Island in Presqu'île Provincial Park, Lake Ontario, Canada.¹ Environmental covariates are shown without transformation (see text for details).

Year	Resighting days	Nesting pairs studied	Alewife abundance (relative CPUE)	PCB Burden ($\mu\text{g/g}$)	Competition index (competitors/HERGs)
1981	20	95	0.86	39.4	269
1982	17	94	0.96	28.3	324
1983 ¹	12	85	0.76	21.2	326
1993	9	141	0.71	10.3	489
1994	6	(153)	0.76	9.2	444
1995	12	166	0.57	6.9	398
1996	9	(165)	0.43	7.1	400
1997 ²	9	164	0.26	8.4	396

¹Data were not available to estimate values in parenthesis so these were estimated by averaging estimates from one year before and after.

²Survival in year $i+1$ was fitted to the environmental index in year i . Therefore, survival in the winter periods 1983/84 and 1997/98 were fitted to indices measured in 1983 and 1997, respectively.

145). Encounters comprised 906 resightings and 50 recaptures; eleven individuals were encountered dead.

Encounter Histories, Global Model, and Modeling Strategy. CMR data were imported into program MARK as live-reencounter, unknown fate, individual encounter histories (EH). EHs were ten years or occasions long representing 1981-1984 and 1993-1998 (White and Burnham 1999). All of the effects in the model were fitted as fixed effects: sex (female or male); year; and three annual covariates (see below). Recoveries were treated as removals upon capture (White and Burnham 1999). The global or most parameterized model was: $\phi_{\text{Sex} \times \text{Year}}^i p_{\text{Sex} \times \text{Year}}^j$ where ϕ was apparent survival (i.e., survival confounded by permanent emigration [White and Burnham 1999]), and p was a combined resight/recapture probability (hereafter, "survival" and "encounter" probability respectively). Survival was measured over the non-sampling or non-breeding period (September-April); encounter probabilities were measured over the sampling or breeding period (May-August). Addition operators (+) identify additive contributions of main effects; data were too sparse to estimate interaction effects. Variation in encounter probabilities was considered while maintaining sex and year fitted to survival. Subsequently, encounter effects with some support were retained and combined in models incorporating all possible combinations of year and sex fitted to survival (Lebreton *et al.* 1992). Assuming that the year effect fitted to survival acquired some support, three annual environmental covariates (see below) were fitted to survival in place of the non-explanatory year effects.

Goodness-of-fit. To minimize heterogeneity, an important source of poor model fit in capture-recapture data (Anderson *et al.* 1994; White 2002), data were carefully screened prior to analysis. Subsequently, sparse resighting data for the 1981-84 cohorts during the 1985-1998 period were detected as an important source of poor model fit. These data and issues were addressed by reducing the encounter histories from 21 (1977-1998) to only ten occasions (as above); replacing data for the early cohort in the 1993-98 period with 0s (not seen); and fixing associated encounter and survival probabilities to zero.

The overdispersion parameter \hat{c} was estimated using the global model and the parametric bootstrap option in

program MARK: global model deviance/mean deviance from n bootstraps = \hat{c} (White and Burnham 1999); and then applying to all models as a bias adjustment term in formulae for estimating Akaike's Information Criterion (AIC) and variances (White and Burnham 1999).

Model Selection and Inference. To assess support for competing models, Akaike's Information Criterion (AIC; Akaike 1973) adjusted for overdispersion and small sample size (QAIC_c) was calculated for all models in the set (Burnham and Anderson 2002). Based on model selection criteria offered by Burnham and Anderson (2002, 2004) top models in the results were conservatively defined as those within about seven to eight QAIC_c units (ΔQAIC_c) from the best model (lowest QAIC_c value); all others had no support. Evidence ratios were calculated to judge the relative support in favor of model i relative to model j as w_i/w_j where w represents the QAIC_c weight of models i and j . Inference concerning the importance of effects contained in top models was determined by assessing the size and precision of model effect sizes (presented here on the logit scale) in top models. Model-averaging was used to estimate encounter and survival probabilities and their variances from the full model set (Burnham and Anderson 2002).

Hypotheses and Environmental Covariates. Sex and year were identified as two categorical variables potentially important in the processes measured. Given that most birds were resighted near, or recaptured on, their nests, any variation in nest attendance and territoriality between sexes was accounted for by fitting a sex effect to the encounter probabilities. Year was fitted to account for annual variation in resighting/recapture effort. In the survival model set, year was considered as a purely exploratory effect, recognizing that many factors might cause survival to vary annually. Three environmental covariates were identified that varied substantially in Lake Ontario, 1978-1998 and between the two sampling periods (1982-84, 1993-98), and which may have affected Herring Gull survival: organochlorine contaminant exposure (measured as PCBs), prey fish (Alewife) abundance, and interspecific competition for fish in the vicinity of the colony.

Contaminant exposure was assessed on the basis of polychlorinated biphenyl (PCB) residue concentrations in eggs (Table 1) from the colony on Snake Island

(44.1°N-76.55°W), 100 km to the east of Gull Island (Bishop *et al.* 1992; Pekarik *et al.* 1998). PCBs are the family of organochlorine contaminants that are present in the highest concentrations in Lake Ontario Herring Gull eggs and tissues. Data were means for seven to 13 eggs, each representing a different clutch. Analytical methods are described in Norstrom *et al.* (1988) and Norstrom and Hebert (2006). PCB concentrations in year *i* were fitted to survival in the following non-breeding period.

Gulls and other piscivorous birds can be useful indicators of the availability and quality of forage fish (Montevocchi 1993; Hebert *et al.* 1999; Wanless *et al.* 2005). The availability of the Alewife, the principle forage fish consumed by Herring Gulls and other avian piscivores in Lake Ontario (Fox *et al.* 1990; Johnson *et al.* 2002), was estimated using the indices of relative abundance of yearling and older Alewives caught in trawl tows of ten-min duration in late April and early May reported in Figure 5 of Mills *et al.* (2003). The index is a three-year running mean standardized to the maximum catch-per-unit-effort (CPUE). These yearly indices of relative abundance of alewives (Table 1) were fitted to survival in the subsequent breeding period.

Competition for forage fish in the vicinity of Gull Island is probably greatest when adults are provisioning chicks, leading to speculation that survival in the non-breeding period might be negatively affected by competition in the preceding breeding period. The number of pairs of six other species of piscivorous birds nesting on Gull Island and nearby High Bluff Islands in each year based on the observations in this study and Appendix 2 in Weseloh *et al.* (2003) was summed and divided by the number of Herring Gull pairs to obtain a Competition Index. Prior to analysis, the PCB and competitor indices were standardized using the transformation $(x - \bar{x})/SD$ (White and Burnham 1999).

RESULTS

Goodness-of-Fit

The estimate of \hat{c} was 1.36 from 1,000 parametric bootstraps. The estimate suggests some overdispersion in the data but no ma-

ajor structural deficiencies in the global model (Anderson *et al.* 1994).

CMR Analysis

Models 7-10 in Table 2 were built to assess support for effects fitted to encounter probabilities. The no effects and sex only models (9 and 10, respectively) acquired no support relative to models $\phi_{sex+year}, p_{year}$ (7) and $\phi_{sex+year}, p_{sex+year}$ (8; Table 2). Although the sex effect in model $\phi_{sex+year}, p_{sex+year}$ was negative, suggesting that males were more readily detected than females, the effect size was small ($\hat{\beta} = -0.21$) and its estimated 95% confidence interval (CI) widely bounded zero (-0.76, 0.34). “Widely bounded zero” refers to the high uncertainty in the estimate reflected in the CI. Although year effects typically bounded zero, effect sizes and their confidence intervals strongly favored negative effects relative to the reference year, 1994 (e.g., 1996: $\hat{\beta} = -2.13$ [-4.54, 0.29]). The only models that did not acquire support in this subset were those not including year effects suggesting important annual variation in encounter probabilities. Year effects were retained and all possible survival models built and fitted to sex and year. Models 2, 4, and 6-7 were used to assess the importance of sex and year on survival; all of these models acquired some support ($\Delta QAIC_c < 8$). In models 6-7, the sex effect was very close to zero and widely bounded zero ($\hat{\beta} = -0.80$ [-0.80, 0.76]; from model 6, Table 2). Although year effects typically

Table 2. Model structure and reference number from text, information criteria, model deviance, and number of parameters (#Par) arranged ascending by QAIC_c ($\hat{c} = 1.36$; top model QAIC_c = 694.93).

#	Model structure ¹	$\Delta QAIC_c$	Akaike weight	#Par ²	Model deviance
1	$\phi_{alewife}, p_{year}$	0.00	0.796	11	90.727
2	$\phi_{(.)}, p_{year}$	5.32	0.056	10	98.110
3	ϕ_{pchl}, p_{year}	5.37	0.054	11	96.101
4	ϕ_{year}, p_{year}	6.27	0.035	15	88.673
5	$\phi_{competitors}, p_{year}$	7.31	0.021	11	98.042
6	ϕ_{sex}, p_{year}	7.38	0.020	11	98.107
7	$\phi_{sex+year}, p_{year}$	8.34	0.012	16	88.643
8 ³	$\phi_{sex+year}, p_{sex+year}$	9.77	0.006	17	87.970
9	$\phi_{sex+year}, p_{(.)}$	15.72	0.000	11	106.447
10	$\phi_{sex+year}, p_{sex}$	17.34	0.000	12	105.995

¹Model parameters: ϕ = apparent survival, p = encounter; (.) = no effects fitted to parameter.

²Parameter counts include \hat{c} .

³Global or most parameterized model in the set.

bounded zero, effect sizes and their confidence intervals strongly favored negative effects relative to the reference year, 1994 (e.g., 1996: $\hat{\beta} = -1.92$ [-4.49, 0.65]).

Given the strong support for annual variation in survival, the indices of Alewife abundance, PCB burden, and competition were fitted in place of the exploratory year effects. Models 1, 3, and 5 (Table 2) were built to assess these environmental covariates fitted to survival compared to unconstrained models in the previous subset. Although all of the models in this subset acquired some QAIC_c support, PCB and competitor effect sizes widely bounded zero (PCB, $\hat{\beta} = 0.44$ [-0.21, 1.08]; competitors, $\hat{\beta} = -0.07$ [-0.61, 0.47]). In addition, evidence ratios (14-40:1) provided nearly exclusive support for the Alewife model over all other top models in the set (Table 2). The Alewife effect in the top model (1; Table 2) was strongly positive and its 95% CI did not bound zero ($\hat{\beta} = 3.83$ [0.80, 6.88]). From the Alewife model (1; Table 2), the highest estimate of survival, 0.97 ± 0.02 , and CPUE, 0.96, were associated with the interval 1982-83 (Table 1). Using the logit function $1/1 + e^{-\hat{\beta}_i}$ and coefficients, $\hat{\beta}_o = -0.29$ and $\hat{\beta}_{alewife} = -3.83$, from the Alewife model, a 10%, 25%, 50%, and 75% reduction in CPUE reported in 1982 leads to decline in survival by 0.014, 0.045, 0.142, and 0.314 units, respectively, from the 1982 level (0.97). A plot of survival estimates from model ϕ_{year}, p_{year} (provides annual estimates of survival that are unconstrained by covariates) with the index of Alewife abundance also suggests a strong correspondence between these two variables (Fig. 1).

Model-averaged Estimates

Model-averaged encounter probabilities tended to be lower in the early (1981-84) relative to the late period (1993-98); and apparently declined from 1994-1997 (Table 3). Model-averaged (apparent) survival probabilities (Table 4) were high for the species (Allard *et al.* 2006) in four out of the seven years (1998 could not be estimated; see notes, Table 3). Although survival in 1997 appeared to be much lower than the other

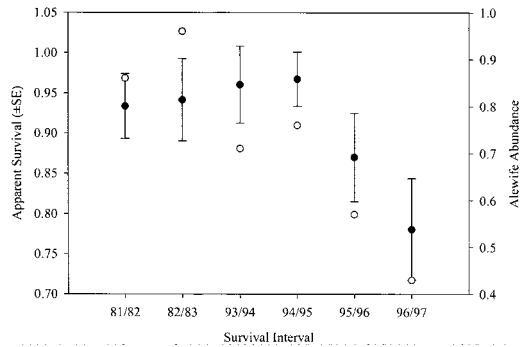


Figure 1. Apparent survival estimates (closed circles \pm SE) for Adult Herring Gulls from Gull Island, Lake Ontario from model ϕ_{year}, p_{year} plotted with an index of Alewife abundance (open circles). Estimates from model ϕ_{year}, p_{year} were selected for this plot because these are not constrained by covariates and model ϕ_{year}, p_{year} had top model support (see text for details). Survival was measured within the survival interval (fall to spring of the following year); Alewife abundance was measured in the summer before the survival interval.

years, this estimate (being at the end of the study) was moderately imprecise and uncertain ($\hat{\phi}_{1997}$ [0.664, 0.903]). Respecting the uncertainty in the 1997 survival estimate, survival may have declined from 1995-1997.

DISCUSSION

Estimates of model-averaged encounter probabilities and apparent survival were biologically reasonable and consistent with the literature and our expectations. Our mean survival estimate for color-banded Herring Gulls from Lake Ontario, 0.91 (0.02 SE), is essentially identical to 0.92 estimated for Herring Gulls from N.E. England, 1979-1985 (Coulson and Butterfield 1986), and our estimates for two of seven years exceeded the range of estimates (0.88-0.94, N = 6) reported for France and the United Kingdom summarized by Allard *et al.* (2006) for this species. Our mean survival estimate exceeded the 0.82 ± 0.07 reported for Herring Gulls on Kent Island, New Brunswick, 1934-1939 (Freeman and Morgan 1992) and 0.87 ± 0.03 reported for Southampton Island, Nunavut, 1998-2004 (Allard *et al.* 2006). Our mean survival estimate for Lake Ontario Herring Gulls is very similar to contemporary estimates for California Gulls (*L. californicus*) on Bamforth Lake, Wyoming (0.92), but much

Table 3. Model-averaged estimates of encounter probabilities (\hat{p}) for adult herring gulls marked with colored bands (1981-1984, 1993-1998) on Gull Island in Presqu'île Provincial Park, Lake Ontario, Canada.¹

Sampling event	\hat{p}	SE	95% $\hat{C}I$	
			Lower	Upper
1982	0.740	0.060	0.607	0.840
1983	0.636	0.063	0.506	0.749
1984	Not estimated ²			
1994	0.955	0.052	0.671	0.995
1995	0.926	0.041	0.793	0.976
1996	0.717	0.058	0.591	0.817
1997	0.794	0.067	0.632	0.897
1998	Not estimated ²			

¹Male and female estimates were identical to three decimal places; females are shown here.

²1984 and 1998 survival and encounter probabilities were confounded in models that included year; therefore, model-averaged estimate of these parameters were not available.

higher than that for Glaucous-winged Gulls (*L. glaucescens*) (0.85) from Protection Island, Washington, and Western Gulls (*L. occidentalis*) (0.82) from Southeast Farallon Island, California (Spear *et al.* 1987; Reid 1988; Pugsek *et al.* 1995).

Adult survival probability is the parameter to which population growth is most sensitive in long-lived species (Lebreton and Colbert 1991). We assume survival in non-harvested species is largely a function of age and disease. Disease has been defined as "any impairment that interferes with, or modifies the performance of normal functions; including responses to environmental factors such as nutrition, toxicants and climate; infectious agents, inherent or congenital effects, or combinations of these factors"

(Wobeser 1997). Three of these potential contributing factors that varied significantly in Lake Ontario from 1978-1998 were fitted to survival: forage fish (Alewife) availability, organochlorine contaminant (PCB) exposure, and interspecific competition for fish in the vicinity of the colony.

Only Alewife abundance appeared to contribute important variation to our survival probabilities. Alewife abundance increased from 1978-1984, stayed high until about 1993, and then steadily declined from 1994-1998. Over the period 1978-1990, the energy density and size of Lake Ontario Alewife declined (Rand *et al.* 1994). Hebert *et al.* (1997) found that the consumption of Alewives by Herring Gulls—as influenced by gull metabolic demands, Alewife abun-

Table 4. Model-averaged estimates of apparent survival probabilities ($\hat{\phi}$) for adult herring gulls marked with colored bands on Gull Island in Presqu'île Provincial Park, Lake Ontario, Canada.¹

Sampling event	$\hat{\phi}$	SE	95% $\hat{C}I$	
			Lower	Upper
1982	0.948	0.027	0.861	0.982
1983	0.959	0.030	0.841	0.990
1984	Not estimated ²			
1994	0.919	0.028	0.844	0.960
1995	0.930	0.028	0.850	0.969
1996	0.874	0.029	0.804	0.922
1997	0.811	0.061	0.664	0.903
1998	Not estimated ²			

¹Male and female estimates were identical to three decimal places; females are shown here.

²1984 and 1998 survival and encounter probabilities were confounded in models that included year; therefore, model-averaged estimate of these parameters were not available.

dance/condition, and Alewife over-winter mortality—was reflected in PCB concentrations and $\delta^{15}\text{N}$ values in their eggs. More recently, using $\delta^{15}\text{N}$ measurements and fatty acid ratios in eggs, Hebert and Weseloh (2006) detected a significant decrease in $\delta^{15}\text{N}$ values in eggs from both the Snake Island and Toronto Harbour colonies, 1974-2003, that reflected decreased fish consumption due to changes in fish availability. These changes in fish availability were manifested through food web changes, in part brought about by the introduction of Dreissenid mussels (Hebert *et al.* 2006). Therefore, the decrease in gull survival from 1995 through 1997 found in this study may be a reflection of nutritional stress resulting from the decreased availability of lipid (energy) and high quality protein in the form of fish and may explain the 12% decrease reported in the Lake Ontario Herring Gull population in the 1990s (Morris *et al.* 2003).

Although no support was found for a role by PCB in our models, PCB burden was 3.8 times higher, on average, in 1978-84 than in the 1990s and the declines in these and other organochlorines were most rapid during the 1978-84 period. PCBs are the family of organochlorine contaminants that are present in the highest concentrations in Lake Ontario Herring Gull tissues and have been associated with effects on metabolism, metabolic synthesis, mineral dynamics, kidney damage, thyroid and immune function, and the stress response in these gulls over the period of this study (Moccia *et al.* 1986; Grasman *et al.* 1996; Lorenzen *et al.* 1999; Fox *et al.* 2007). It is quite probable that some of these sublethal effects exacerbated the nutritional impacts of low Alewife availability and contributed to mortality. Elsewhere, recent investigations of Glaucous Gulls (*Larus hyperboreus*) in the Norwegian Arctic have found associations between PCBs and other organochlorines and adult survival probabilities (Bustnes *et al.* 2003, 2005). In the Great Lakes, Mora *et al.* (1993) found that the proportion of adult Caspian Terns (*Sterna caspia*) trapped on nests in the same region where they were raised was negatively correlated with mean plasma PCB concentrations.

Increased populations of piscivorous birds harvest greater amounts of fish thus potentially increasing competition for this food resource. However, our results provided no support for a role by competitor abundance in the survival process experienced by adult Herring Gulls nesting on Gull Island. The increased competition for fish may have been offset through scavenging, kleptoparasitism, and predation on the abundant eggs and chicks of the competitors.

No evidence was found of an affect of sex on survival which is evident in our model-averaged survival estimates—males and female estimates were identical to three decimal places (Table 4). This is consistent with other studies of Herring Gulls (Pons and Migot 1995; Allard *et al.* 2006), and of California Gulls (Pugesek *et al.* 1995). However, Wanless *et al.* (1996) found a nonsignificant 5% higher survival rate in female Herring Gulls on the Isle of May, and that survival was lower in males than females in three of four years. Spear *et al.* (1987) found that survival of adult male Western Gulls was higher than that of females in six of eight years and suggested that females are disadvantaged during times of food shortage.

Our exploration of the information captured by three (simple) indices of environmental factors which may be important in modulating adult survival provided strong evidence that variation in Alewife abundance may have played a detrimental role in this Lake Ontario Herring Gull colony over the study period. Our limited success at isolating other important environmental covariates reinforces the difficulty posed in identifying the critical components of the complex of environmental variables encapsulated in 'time'.

ACKNOWLEDGMENTS

D. B. Peakall (deceased) and J. A. Keith encouraged and supported the study 1978-1984, and K. Lloyd, K. W. Marshall, and D. Stewart encouraged and supported its resurrection, 1993-1998. Most of the essential work of trapping, banding, resighting, and field record keeping was done by GAF, C. R. Cooper (deceased), J. Struger, R. Tait, D. Jeffrey, M. Rush, J. Ellenton, D. Boersma, and L. Maus, (1978-84), assisted by L. Armour, L. Chalmers, JWC, C. Flynn, A. Gilman, A. Harfinest, E. Hayakawa,

S. Hodgins, G. Lambert, J. Learning, M. MacPherson, D. Noble, D. Peakall, T. Power, P. Fox, P. Stanley, and J. Tremblay; and by D. Jeffrey and J. Newman (1993-98) assisted by A. Ali, J. Bruce, J. Scott, A. Smith, and C. Yauk. We thank Ontario Ministry of Natural Resources (Napane District) for initial permission to work in Presqu'île Provincial Park and the various supervisors and staff for their cooperation, logistic support, and efforts to insure our safety. Tania Havelka, of Canadian Wildlife Service Ontario Region, provided egg contaminant and population data. This project received funding from the Great Lakes Action Plan. We thank Greg Robertson for constructive comments on an earlier version of this manuscript.

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