

Conservation

LOW CLUTCH VIABILITY OF AMERICAN ALLIGATORS ON LAKE APOPKA

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ABSTRACT—Clutch viability of American alligators was evaluated on lakes Apopka, Griffin, Jessup, and Okeechobee, Florida, during 1983-86 to examine its association with alligator population trends. Clutch viability was lower ($P < 0.05$) on Lake Apopka and higher ($P < 0.05$) on Lake Griffin than on any other lake. Annual clutch viability rates declined ($P < 0.05$) on Lake Apopka during the study, but no trends in viability rate were detected on other lakes. Juvenile alligator density was relatively stable during 1980-87 on lakes Griffin and Jessup, but plunged to 10% of the 1980 level on Lake Apopka ($P = 0.002$), coincident with falling clutch viability. Viability rates were not related to clutch size but did increase with clutch weight ($P = 0.013$). Egg banding rates declined on lakes Apopka and Jessup and increased on lakes Griffin and Okeechobee. Unexplained mortality of large alligators was commonly observed on Lake Apopka. We discuss several possible causes of low clutch viability including pesticide contamination, shifts in the age structure of the breeding population, and density-related stress of the adult population, and we recommend a course for further investigation.

THE Florida population of the American alligator (*Alligator mississippiensis*) increased after federal protection in the early 1970s (Hines, 1979; Wood et al., 1985) and continued to grow on most wetlands through the 1980s (Woodward and Moore, 1989). However, Jennings and co-workers (1988) reported a rapid, severe (approx. 90%) decline of the juvenile alligator population on Lake Apopka during 1981-86 and proposed reproductive failure as a primary cause.

Prior to the 1940s, Lake Apopka was a clear, open lake (12,960 ha) with an adjacent marsh (8,000 ha). However, events in the 1940s drastically altered the water quality, turning Lake Apopka into one of Florida's most polluted (nutrient and pesticide) lakes (U.S. Environ. Prot. Agency, 1979). During the mid-late 1940s, 7285 ha of marsh on the north side of Lake Apopka was converted to vegetable farming, and much of the uplands surrounding the lake was planted with citrus (Fig. 1). Concurrently, there was an increase in effluent discharges into the southeast part of the lake from a citrus processing plant and the sewage treatment facility of the city of Winter Garden. The ecological integrity of Lake Apopka was further damaged in 1947 when a hurricane uprooted most of the aquatic vegetation in the main lake, reducing the nutrient cycling capacity of the wetland (Conrow et al., 1989).

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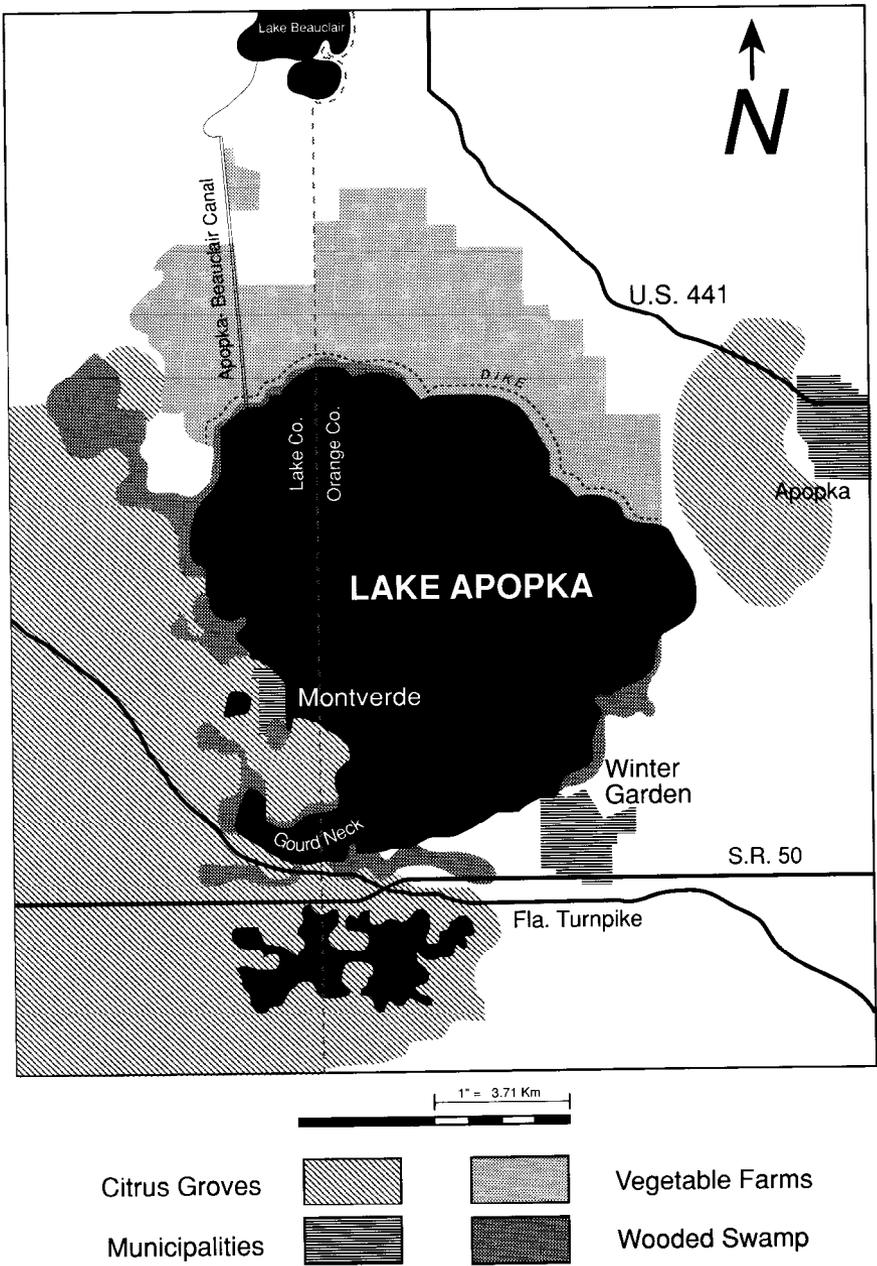


FIG. 1. Lake Apopka, Florida and surrounding agricultural operations during the early 1980s.

Agricultural operations have introduced substantial pesticide loads into Lake Apopka since the early 1940s through direct pumping or through seepage into the Apopka drainage (Huffstutler et al., 1965; Florida Dept. Environ. Reg., 1979). Vegetable farms must continually remove water from fields during the wet season, and most operations flood and drain fields every 2 years for nematode control. Pesticides commonly used during the late 1970s included toxaphene, parathion, and chlorobenzilate, all of which have been restricted by the U.S. Environmental Protection Agency (U.S. EPA) because of chronic or acute effects on wildlife or fish (U.S. EPA, 1990). In 1980, an extensive spill of Kelthane, an EPA-approved pesticide primarily composed of dicofol, was reported (U.S. EPA, unpubl. rep.) at the Tower Chemical Company located between State Road 50 and Gourd Neck of Lake Apopka (Fig. 1).

Few accounts exist of fish and wildlife populations on Lake Apopka and how they may have been affected by the above events. Increased build-up of unconsolidated bottom sediment and subsequent accelerated eutrophication contributed to a large fish kill in 1963 (Clugston, 1963; Huffstutler et al., 1965). Shotts and co-workers (1972) reported a die-off of fish, alligators, and Florida softshell turtles (*Apalone ferox*) during 1971 and attributed it to stress-induced bacterial (*Aeromonas spp.*) infections exacerbated by low water level. Johnson and Jenkins (1984) reported reproductive failure of largemouth bass (*Micropterus salmoides*) on Lake Apopka in 1982. Heinz and co-workers (1991) found Lake Apopka alligator eggs collected during 1984-85 to have elevated levels of several organochloride compounds relative to eggs from lakes Griffin and Okeechobee, but they found no direct association with clutch viability. Although alligator populations normally respond positively to eutrophication (Wood et al., 1985), the alligator population crash reported by Jennings and co-workers (1988) indicated that conditions in Lake Apopka were unfavorable for alligator population growth during the early 1980s.

Artificially incubated alligator clutches from the Jennings and co-workers (1988) study provided an opportunity for us to compare clutch viability rates among 4 Florida lakes, estimate clutch viability trends, and develop hypotheses about the possible relation of clutch viability to the alligator population status of Lake Apopka.

MATERIALS AND METHODS—Entire clutches of alligator eggs were systematically collected (see Woodward et al., 1989 for procedures) from accessible nests on lakes Apopka, Griffin (5742 ha), Jessup (4862 ha), and Okeechobee (191,223 ha) during 1983-86 as part of experimental egg and hatching harvests (Jennings et al., 1988). Two subareas of Okeechobee were sampled; Indian Prairie Marsh on the northwest shore and Observation Shoal on the southwest portion of the lake. Clutches were transported to incubators at commercial alligator farms, incubated in either natural nest material or *Paspalum notatum* (see Woodward and co-workers [1989]), and maintained at 30-33 C and >95% relative humidity until hatching. Clutch allocations to the 1-3 participating farms in any year were determined primarily on logistical convenience. Prior to incubation, eggs were inspected for presence of an opaque band indicating fertility (Ferguson, 1985; Webb et al., 1987) and for early embryo viability (Woodward et al., 1989). Nonviable eggs were opened to verify status and stage of embryo mortality and then discarded. From each clutch, one egg that represented the most advanced stage of development was sacrificed to determine embryo age. Only clutches collected in 1986 were weighed (WT).

Clutch size (C) was the total of all shelled and unshelled eggs found in a nest. Banding rate (B) was the number of banded eggs divided by C . Clutch viability rate (V) was the number of hatching alligators surviving ≥ 1 day divided by $C-1$ (C less the sacrificed egg). Because we were mainly interested in the inherent viability of eggs, clutches from flooded or disturbed (by predators, turtles, humans, or other alligators) nests were excluded from viability analyses. However, only clutches from disturbed nests were excluded from analyses of C and B .

The clutch was the experimental unit in analyses of V , C , and B . None of the variables was distributed normally or with constant variance. We squared C to normalize the distribution then analyzed C^2 via ordinary least squares regression methods. The rate variables V and B included many values representing either complete viability failure ($V = 0.0$, 19% of clutches) or complete banding success ($B = 1.0$, 21%). No single transformation could normalize the truncated distributions of V and B . We considered the nonparametric alternative of ranking the data and performing regression analyses of the ranks (Conover and Iman, 1981), but we were concerned that the preponderance of tied values resulting from this approach would bias tests of significance. Instead, we divided the rate data into 2 components, each of which could be suitably transformed for analysis.

Into the first component, we assigned all rate values not representing complete successes or failures, i.e., $0 < V < 1$ and $0 < B < 1$. We converted rates to odds ratios ($V/[1-V]$ and $B/[1-B]$) and found values of λ such that $(V/[1-V])^\lambda$ and $(B/[1-B])^\lambda$ were approximately normally distributed (Box and Cox, 1964). The value $\lambda = 0.25$ was satisfactory for both variables, and we confirmed the transformation through inspection of normal probability plots. We then conducted ordinary least squares regression analyses on $V^* = (V/[1-V])^{0.25}$ and $B^* = (B/[1-B])^{0.25}$ for V and B values between 0 and 1.

We assigned all the data into the second component, but rates were replaced by binary indicator values (0 and 1). For $V > 0$, we reassigned V the indicator value 1. Thus, each clutch was designated as either completely nonviable or partially viable. For $B < 1$, we reassigned B the value 0 to distinguish clutches with some unbanded eggs from clutches with all banded eggs. We assumed the indicator values were binomially distributed and used logit analysis (Agresti, 1990) to estimate relative probabilities of complete/partial viability failure and banding success.

Although data assigned to both components comprised transformed values of V and B , we present median and untransformed means for reference and comparison purposes. Means presented for C represent back-transformed means of C^2 .

In all analyses, our study design was a 4-year by 5-area fixed effects factorial, but we tested for differences between Okeechobee subareas for possible pooling as 1 area. Woodward and co-workers (1989) found no differences in hatch rate among clutches incubated at 3 facilities in 1985, thus we assumed that facility effects were negligible and could be ignored. We assumed that accessible clutches constituted a completely random sample of clutches. An effect for a rate variable was considered significant ($P < 0.05$) if analysis of data in either or both so indicated. If we detected no area-year interaction, we estimated linear, quadratic, and cubic contrasts for the year effect means and compared all possible pairs of area means. When areas and years did interact, we compared area means if the area effect was significant. Because analysis of trends among areas was a principal study objective, we tested whether a significant area-year interaction was due to differences in trend among areas. In this circumstance, we estimated linear contrasts of year within areas and compared all possible pairs of contrast estimates. Mean or trend pairs were declared different when $P < 0.05/k$ (where k was the number of comparisons to be made), to guarantee that the probability of falsely declaring significant 1 of the k comparisons was no greater than 5% (Bonferroni adjustment). F -tests or G -tests were used to determine significance of effects in the factorial design for the regression and logit analyses. V^* was regressed on area, year, C , and WT to determine the relationship of selected clutch characteristics to clutch viability.

Counts of juvenile (30–121 cm total length [TL]), ≥ 122 cm TL, and adult (≥ 183 cm TL) alligators were obtained from night-light surveys conducted during 1980–87 on Apopka, Griffin, and Jessup and provided estimates of observed densities (alligators/shoreline km). Density data were not available for Okeechobee. We regressed log-transformed density on year to estimate population trends. A water level covariate was included in the regression models to statistically remove its effect from alligator observability. Unlike Jennings and co-workers (1988), our trend analysis included alligators 30–60 cm in the juvenile size class and added 1987 data to reflect the population response to 1986 events.

The observation rate of dead alligators was the number of dead alligators ≥ 122 cm observed during night-light and tagging activities divided by the number of activity nights.

RESULTS—Clutch Viability—In Okeechobee, mean V^* did not vary by subarea or subarea-year combination. Complete viability failures did not occur in 2 years on Indian Prairie Marsh; thus, we could not investigate the subarea-year interaction in the logit analysis. The odds of sampling a partially nonviable clutch rather than a completely nonviable clutch was 7 times greater on Indian Prairie Marsh (28:1) than on Observation Shoal (4:1) ($P = 0.002$). Recognizing that complete viability failure probabilities differed within Okeechobee, we nevertheless ignored subarea distinc-

TABLE 1. Sample size (n), mean (\bar{x}) and median (m) clutch viability, and percentage observations of zero alligator clutch viability rates (%0) for alligator clutches collected on 4 Florida lakes 1983-86. Data consist of only non-flooded and non-disturbed clutches.

Study area		Year				
		1983	1984	1985	1986	All years
Lake Apopka	n	11	8	28	22	69
	\bar{x}	0.54	0.22	0.18	0.13	0.21
	m	0.45	0.05	0.01	0.00	0.03
	%0	18.2	37.5	50.0	59.1	46.4
Lake Griffin	n	12	27	67	61	167
	\bar{x}	0.53	0.65	0.59	0.60	0.60
	m	0.74	0.75	0.73	0.72	0.73
	%0	25.0	3.7	19.4	13.1	15.0
Lake Jessup	n	18	32	37	40	127
	\bar{x}	0.36	0.54	0.34	0.63	0.48
	m	0.26	0.65	0.30	0.77	0.54
	%0	16.7	6.2	32.4	7.5	15.7
Lake Okeechobee	n	15	35	42	24	116
	\bar{x}	0.47	0.40	0.40	0.59	0.45
	m	0.46	0.44	0.44	0.71	0.46
	%0	6.7	14.3	16.7	12.5	13.8
All areas	n	56	102	174	147	479
	\bar{x}	0.44	0.50	0.42	0.54	0.48
	m	0.47	0.58	0.46	0.63	0.56
	%0	16.1	10.8	26.4	18.4	19.4

tion and combined Indian Prairie Marsh and Observation Shoal for comparisons among main study areas.

Mean and median V for 479 clutches from all 4 lakes, 1983-86, were 0.48 and 0.56 (Table 1). Mean viability varied ($P < 0.001$) among areas in both the regression and logit analyses. Griffin mean V was greater than for any other area, and Apopka mean V was less than for Jessup. No other mean V pair was different. Complete viability failures were more likely to be encountered on Apopka than on the other areas, but Griffin, Jessup, and Okeechobee were not different (Table 1). Across areas, no yearly effect was detected in the regression analysis, but a cubic trend ($P = 0.025$) was discovered in the logit analysis; mean probability of complete viability failure was smallest in 1984, greatest in 1985, and intermediate in 1983 and 1986. Interaction between area and year was nearly significant ($P = 0.066$) in the regression analysis of V , particularly in the component that measured linear trend variability among areas ($P = 0.051$). However, we failed to find differences between any trend pairs even though V decreased ($P = 0.026$) with time on Apopka. No interaction was detected in the logit analysis.

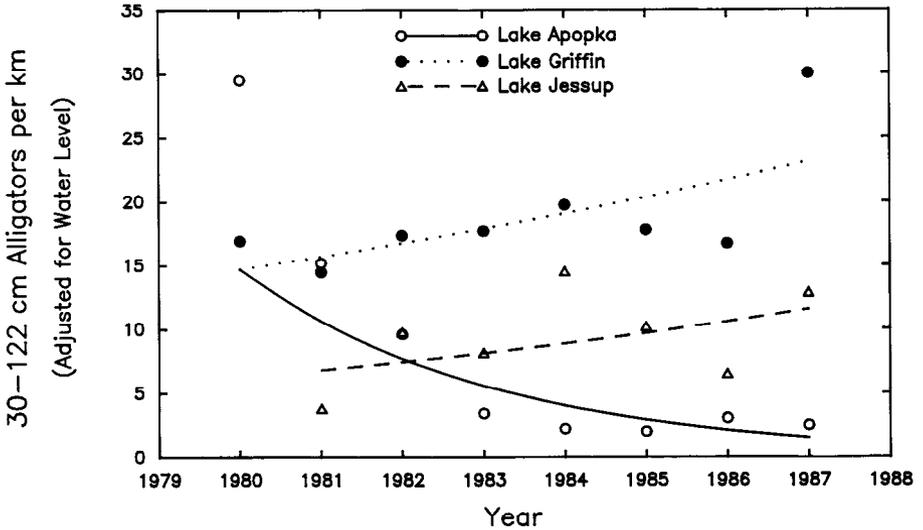


FIG. 2. Estimated trends of juvenile (30-122 cm) alligator populations on lakes Apopka, Griffin, and Jessup during 1980-87. Estimates were derived from log-transformed night-light counts, adjusted for water level, and presented on an untransformed scale.

Clutch Size and Weight—Overall mean C for all areas was 45.5 eggs/clutch (Table 2). Although mean C was greater on Indian Prairie Marsh (45.4 eggs/clutch) than on Observation Shoal (41.6 eggs/clutch), we pooled data from the Okeechobee subareas as 1 area. C^2 differed ($P < 0.001$) among areas, and the pooled Okeechobee mean was less than either the Griffin or Jessup means (Table 2). Linear trends in C^2 means also varied ($P = 0.026$) by area; clutch size declined on Okeechobee and increased on Griffin. We detected no effect of clutch size on V , but V increased ($P = 0.001$) with clutch weight. A weak ($P = 0.067$) negative quadratic relationship was detected, suggesting that clutch viability peaked at intermediate clutch weights, then declined for the heaviest clutch weights.

TABLE 2. Back-transformed mean (\bar{x}_b) and untransformed mean (\bar{x}) clutch sizes and mean (\bar{x}) and median (m) banding rates for undisturbed alligator clutches collected from 4 Florida lakes, 1983-86.

Study area	Clutch size			Banding rate		
	n	\bar{x}_b	\bar{x}	n	\bar{x}	m
Lake Apopka	81	45.8	45.2	80	0.70	0.89
Lake Griffin	211	46.1	45.8	209	0.86	0.94
Lake Jessup	161	47.2	46.5	159	0.89	0.94
Lake Okeechobee	229	43.6	42.4	229	0.85	0.95
All areas	682	45.5	44.8	677	0.84	0.94

Banding Rates—Overall median and mean banding rates were 0.94 and 0.84 (Table 2). We found no differences between Okeechobee subarea mean banding measures in either the regression or logit analysis; thus, we pooled the data. Mean *B'* did not vary among areas or among years. However, we detected an area-year interaction ($P = 0.014$), especially in the cubic component ($P = 0.005$) which indicated that banding oscillated but was asynchronous among study areas. Mean probability of complete banding success varied ($P = 0.021$) among years and among levels of the area-year interaction ($P = 0.028$), but not among areas. Complete banding success probability declined on Apopka and Jessup but increased on Griffin and Okeechobee. The increasing Okeechobee trend was also greater than the increasing Griffin trend, but we could detect no difference between the Jessup and Apopka trends.

Population Densities—Juvenile alligator densities were similar on Apopka and Griffin during 1980-82, but declined on Apopka during 1980-87 by 90% (28%/year, $b = -0.323$, 10 df, $P = 0.002$; Fig. 2). No trends were detected in densities of larger size classes on Apopka. Densities of juvenile alligators increased on Griffin (6.7%/year, $b = 0.065$, 9 df, $P = 0.029$; Fig. 2) but not on Jessup. Densities of ≥ 122 cm TL and adult alligators increased on both Griffin ($\geq 8.3\%$ /year, $b = 0.080$, 9 df, $P < 0.01$) and Jessup ($\geq 11\%$ /year, $b = 0.100$, 9 df, $P < 0.05$). Ranges of estimated adult alligator densities during 1982-86 were 0.1-3.2 alligators/km on Apopka, 1.4-3.2 on Griffin, and 1.5-3.1 on Jessup.

Observed Mortality—Dead alligators were more frequently observed on Apopka (0.414 alligators/night, $n = 29$ nights) than on Griffin (0.032, $n = 93$) and Jessup (0.038, $n = 26$). Fish kills were common on Apopka, and, on one occasion in 1983, we observed 2 dead Florida softshell turtles, a dead, unidentified water snake (approx. 1.2 m TL), and 2 dead alligators (≥ 1.8 m). Superficial examinations of carcasses revealed no evidence of trauma caused by humans or other animals.

DISCUSSION—The alligator population decline on Lake Apopka reported by Jennings and co-workers (1988) and reconfirmed in our analysis coincided with rapidly declining clutch viability rates during 1983-86. Unfortunately, clutch viability rates prior to 1983 were not available for comparison with post-decline rates. However, juvenile alligator population densities on Lake Apopka during 1980-82 were similar to those observed on Lake Griffin, suggesting that clutch viability had been greater before 1983. Although harvests of eggs and hatchlings were concurrent during this study, we submit, as did Jennings and co-workers (1988), that a decline in juveniles would have been evident on all lakes had over-harvest been a factor. On the contrary, juvenile population densities were relatively stable on Lake Jessup and increased on Lake Griffin. The coincident juvenile alligator population decline and low clutch viability level on Lake Apopka suggest that poor reproductive success contributed to the alligator population decline.

Median viability rates among lakes in our study were extremely variable and ranged from 0.73 on Lake Griffin to 0.03 on Lake Apopka. Clutch viability was higher on Lake Griffin and lower on Lake Apopka than on other lakes and did not differ

between lakes Jessup and Okeechobee. Clutch viability also differed within Lake Okeechobee, being lower on Observation Shoal than on Indian Prairie Marsh. We did not detect clutch viability trends on lakes Griffin, Jessup, and Okeechobee, but we found strong evidence of annual fluctuations.

Viability rates of clutches in this study were much lower than viability rates reported for artificially incubated wild produced eggs from Rockefeller Refuge, Louisiana ($V = 0.86$; Joanen and McNease, 1987) or for eggs collected from Texas coastal marshes ($V = 0.83$; Johnson et al., 1989). No other published viability data are available for artificially incubated wild alligator eggs. We note that our rates were likely biased low, but by no more than 0.03, because we sacrificed the best-developed egg in each clutch for age determination and excluded it from analyses.

The causes of low and decreasing clutch viability on Lake Apopka are unclear. Woodward and co-workers (1989) found no embryo age-specific differences in hatch rates of alligator clutches when collected and handled under careful procedures. They also presented evidence that substantial mortality of alligator embryos from lakes Griffin, Jessup, and Okeechobee occurred prior to collections and continued through incubation, independent of collection time. We found a decreasing trend in banding rate on Lake Apopka, and an average of 30% of each clutch was unbanded (Table 2), indicating substantial infertility or pre-oviposition embryonic death. Thus, we believe that major factors contributing to low egg viability are manifest prior to embryo attachment to the shell membrane.

Some factors that may influence egg viability are female age (Ferguson, 1985), density-related stress (Joaanen and McNease, 1989; Elsey et al., 1990); adult nutritional status (Joaanen and McNease, 1989), toxicosis through exposure to environmental contaminants (Clark, 1990), nest flooding (Joaanen et al., 1977), and extremes in pre-collection clutch cavity temperatures (Joaanen and McNease, 1987; Webb and Cooper-Preston, 1989). We were able to eliminate flooding and disturbance as sources of variation by selecting unflooded and undisturbed nests. However, we could not sample and control early nest temperature nor obtain dietary data for assessment of adult nutritional status. Population densities of adult alligators were not greater on Lake Apopka than on lakes Griffin and Jessup during 1982-86. Superficially, this does not support the hypothesis that density-related stress caused low clutch viability on Lake Apopka.

Ferguson (1985) and Joanen and McNease (1989) reported that the youngest and oldest females produce the least viable clutches, and Wilkinson (1983) and Ferguson (1985) found an association between number of eggs per clutch and female body size. Clutch size means did not differ among lakes Apopka, Jessup, and Griffin, but were noticeably greater than means reported for most other alligator populations (Metzen, 1977; Goodwin and Marion, 1978; Deitz and Hines, 1980; Ruckel and Steele, 1984; Carbonneau, 1987; Joanen and McNease, 1989; Kushlan and Jacobsen, 1990). Mean clutch size for South Carolina coastal impoundments was comparable to our findings (Wilkinson, 1983). These comparisons imply that nesting females in our study areas were larger and possibly older than those of other alligator populations, although site-specific factors (e.g., nutrition and genetics) may affect the

female size/clutch size relationship.

Clutch weight rather than clutch size may be more closely associated with female size (Hall, 1990). Clutch weight was less on Lake Apopka than the other lakes. The positive linear and negative quadratic relationship between viability and clutch weight implies that greater clutch weight enhances clutch viability, but that the rate of enhancement decreases for the greatest clutch weights (oldest females). Therefore, lower clutch viability on Lake Apopka may have resulted from a predominance of both younger and older, less productive females. Our clutch weight sample size was small and limited to 1 year (1986), but these findings identify an area that deserves more investigation.

Although we detected no decline in population densities of alligators ≥ 122 cm on Lake Apopka, population growth apparently stagnated while population densities on lakes Griffin and Jessup increased sharply. The frequent occurrence of dead alligators may further reflect chronic underlying problems in the lake. Mortality of alligators, softshell turtles, and fish on Lake Apopka during 1971 and 1972 was attributed to bacterial (*Aeromonas* spp.) infections induced by drought-related stress (Shotts et al., 1972). *Aeromonas* are common organisms in Florida wetlands and usually are non-pathogenic. However, when conditions sufficiently stress alligators, *Aeromonas* can proliferate and cause mortality (Shotts et al., 1972). Although Lake Apopka water levels were relatively stable during our study, fish kills and alligator mortality were common. Therefore, low water levels probably did not cause the decline.

Surrounding agricultural operations have introduced considerable amounts of pesticides into Lake Apopka since the 1940s (Huffstutler et al., 1965; U.S. EPA, 1979). Heinz and co-workers (1991) found no association between selected pesticide levels and clutch viability, but emphasized that toxic chemicals could adversely affect adult alligator reproductive endocrinology in ways that may not be measured in the eggs. If such a phenomenon was at work, we would have expected chronic low reproductive success rather than an apparent acute problem.

The strong association between the 1980 Kelthane spill and subsequent declines in the juvenile alligator population and egg viability provides the best clue to the cause of acute reproductive failure. Although the full impact of the 1980 Kelthane spill is not yet known, the maximum effect of the organochloride pesticide and its breakdown metabolites, DDD and DDE (Clark, 1990), would have likely occurred during the early 1980s. Fish kills resulting from the advanced eutrophic state of Lake Apopka may have been amplified by periodic release of toxicants from bottom sediments. Alligators commonly fed on dead fish and, as a consequence, may have been exposed to occasional high concentrations of toxicants. In high concentrations, dicofol can be lethal to vertebrates, and in lower doses it can cause reproductive impairment (Clark, 1990). Thus, the high incidence of dead alligators and low clutch viability could have been caused by the Kelthane spill.

The implications of low alligator clutch viability are important from ecological, aesthetic, and economic perspectives. Alligators are top carnivores (Delany and Abercrombie, 1986; Delany, 1990) in the aquatic food web on Florida lakes and may

play an important role in maintaining a balance in prey species populations. Furthermore, most Floridians value alligators for aesthetic reasons (Hines and Scheaffer, 1977; Delany et al., 1986). Lower alligator densities on Lake Apopka will reduce opportunities for the public to observe alligators. Depressed populations and productivity also will have a negative effect on sustained-yield alligator hunting and egg harvests. We estimate that the harvest potential of alligators in Lake Apopka has been reduced by 150 large alligators (@ \$400 each) and 1200 hatchlings per year (@ \$15 each) for an estimated loss in wholesale value of \$78,000 per year.

The effects of lake degradation on fish and wildlife may be expressed through acute mortality or more subtly through depressed reproduction, poor growth, and low survival. Factors contributing to low alligator clutch viability on Lake Apopka may have contributed to mortality of fish and turtles (Shotts et al., 1972), poor reproductive success in largemouth bass (Johnson and Jenkins, 1984), and may have negatively affected other, less conspicuous vertebrates that depend on wetlands for food.

Adverse effects of environmental degradation may not be limited to Lake Apopka. Low alligator clutch viability rates observed on lakes Griffin, Jessup, and Okeechobee relative to rates reported for Louisiana and Texas may indicate chronic reproductive problems on those wetlands as well.

Our clutch viability comparisons among areas and trend estimations over years may have suffered from the variation sources of uncontrollable error, sampling bias, and inadequate sample size. Therefore, we may not have been able to detect differences that actually occurred among areas and years. We recommend continuing clutch viability investigations and making special efforts to reduce sources of variation and sampling bias by carefully controlling collection and handling procedures, incubating eggs under standardized conditions at a common facility, and increasing clutch sample sizes to increase statistical power. Further studies should address relationships of clutch viability to adult nutritional status, population density, nest characteristics, nest temperatures, female age, and toxic contaminant levels. We suggest broadening the sample of wetlands to gain a better idea of the range in clutch viability rates throughout Florida.

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FIRST RECORD OF THE EASTERN BIG-EARED BAT (*PLECOTUS RAFINESQUII*) IN SOUTHERN FLORIDA —

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ABSTRACT: *The first specimen of the eastern big-eared bat, *Plecotus rafinesquii*, taken in southern Florida, was recorded roosting in an old cabin in northeastern Collier County, a short distance south of the Seminole Indian Reservation. It was an adult female, and this occurrence extends the known range of the eastern big-eared bat significantly southward from central Florida to the Big Cypress Swamp region of south-western Florida.*

BROWN (1974) first reported the presence of *Plecotus rafinesquii* as far south as central Florida. The overall status of the species was later summarized by Brown (1978). A subsequent update on rare and endangered biota in Florida, by Humphrey (1992) failed to include the known records for *Plecotus rafinesquii* from central Florida. Brown (1993) corrects this oversight, and summarizes current life history and distribution information for the species in Florida. This bat tends to live in small colonies often associated with pine flatwood forests and abandoned cabins or hollow trees in the forest.