Variation in Population Growth Rates of Mottled Ducks in Texas and Louisiana

By Fred A. Johnson

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Summary

1. Sex-specific matrix models of mottled ducks in Texas and Louisiana were used to estimate finite growth rates (λ), to determine whether and how λ varied over time, and to assess which demographic rates were most responsible for variability in λ.
2. Estimated survival rates of mottled ducks derived from band-recovery data were low compared to those of most dabbling ducks, as well as to those of mottled ducks in Florida. Estimates of reproductive rates derived from hunter-collected wings also were low compared to closely-related species. Both asymptotic and stochastic growth rates of mottled ducks were substantially <1 for both sexes. However, in absolute terms, male and female growth rates did not always agree and this might be attributed to sampling error, which was not modeled explicitly.
3. Reproductive rates varied more than survival rates, but variation in annual survivorship contributed >60% of the variation in λ. The growth rates of males responded as expected to varying environmental moisture regimes (i.e., increasing growth rates with increasing moisture), but female growth rates varied little among dry, normal, and wet periods. This was primarily due to the large negative contributions of female survival rates during the wet period, which may reflect an ecological cost associated with the high level of nesting and re-nesting expected in wet years. Large, negative contributions of survival rates were not observed among males in the wet periods, lending credence to this hypothesis.
4. My analyses suggest a rapidly declining mottled duck population on the western Gulf Coast during 1994-2005. This finding is in general agreement with a variety of population surveys, but the severity of the decline was matched only by survey data from national wildlife refuges in coastal Texas. Although results from this survey have been criticized for their limited geographic scope, the survey design is likely the least biased of any of the surveys examined.
5. Mottled ducks are a popular game bird, but I was unable to examine the effects of hunting on population growth rates because hunting regulations were relatively stable (and liberal) over the period of record. Because of the sensitivity of growth rates to survivorship, experiments or other studies to understand the effects of hunting seem warranted.

Key-words: demography, growth rates, matrix models, mottled ducks, population projections

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Introduction

The mottled duck (*Anas fulvigula*) is a non-migratory resident of the southeastern U.S. and northeastern Mexico. A member of the mallardine clad, the mottled duck is more closely related to the black duck (*Anas rubripes*), both genetically and phenotypically, than to the mallard (*Anas platyrhynchos*) (McCracken and others, 2001). Mottled ducks occur in two distinct populations (McCracken and others, 2001) - one in peninsular Florida and one along the western coast of the Gulf of Mexico (hereafter, the WGC population) (Fig. 1). The size of the Florida breeding population is estimated at roughly 40,000 (D. Eggeman, Florida Fish and Wildlife Conservation Commission, pers. comm.), but the WGC population may exceed 600,000 birds (B. Wilson, U.S. Fish and Wildlife Service, pers. comm.). The mottled duck has always been of conservation concern due to its limited range and small population size, as well as its popularity as a game bird. Threats to the species include loss and degradation of wetlands, hybridization with feral mallards, and the possibility of excessive sport harvest (Moorman and Gray, 1994).

The Florida population appears to be relatively stable at present (D. Eggeman, pers. comm.), but the status of the WGC population is less clear. In 2006, the U.S. Fish and Wildlife Service sponsored a workshop to assemble all available information on mottled duck demography, but participants failed to reach consensus about the status of the WGC population. Nonetheless, this population has become a source of increasing conservation concern because of breeding-season surveys on national wildlife refuges in coastal Texas that suggest a precipitous decline since the surveys began in 1985 (D. Haukos, U.S. Fish and Wildlife Service, pers. comm.). Other possible sources of information for deriving population trends include the Midwinter Waterfowl Inventory (MWI, *http://mbdcapps.fws.gov/*), the Christmas Bird Count (CBC, *http://www.audubon.org/bird/cbc/history.html*), and the Breeding Bird Survey (BBS, *http://www.pwrc.usgs.gov/BBS/about/*), but none of these surveys is considered particularly reliable for mottled ducks, and population trends derived from them vary depending on the time period chosen.

I was thus interested in other sources of information that might shed light on the dynamics and status of the WGC population. Using data from Texas and Louisiana during 1994-2006, this study focused on:

a) the calculation of population growth rates derived from estimates of survivorship and reproduction;
b) whether growth rates have varied over time and, if so, which demographic rates were most responsible;
c) whether growth rates have varied with environmental moisture regimes; and
d) the extent to which growth rates estimated in this study help corroborate or refute those derived from population surveys.

Methods

Survivorship and reproductive success were estimated using a combination of band-recovery data and samples of wings submitted by hunters. Mottled duck banding has been sporadic throughout the species' range, but there has been a fairly consistent, post-breeding (June-August) banding effort in Texas and Louisiana since 1994. Bandings have been well distributed throughout this major portion of the mottled duck's range (Fig. 2), although how well these banded samples represent the population at large is unknown. I acquired 44,147 records of banded birds of known

age and sex for the years 1994-2006 from the USGS Bird Banding Laboratory. Bandings from the summer of 2007 were not included because recoveries of these bands had not yet been compiled and error-checked by the Bird Banding Laboratory. At the time of banding, birds were classified as local, hatch-year, and after-hatch-year. Hatch-year birds were fledged, while locals were not. To maximize sample size, local and hatch-year birds were pooled and classified as juveniles. Of the sample of banded birds, 5,476 recoveries were reported during the hunting season, which runs from late September to late January. Banding and recovery data were compiled by year and cohort (i.e., adult female, juvenile female, adult male, juvenile female), and I used the program MARK (White and Burnham, 1999) to calculate estimates of recovery and survival rate. I examined 16 different models, in which recovery or survival rate could either be constant, vary by year, or vary by cohort. I then used AIC (Burnham and Anderson, 1998) to identify the most appropriate model from which to derive estimates.

Reproductive success was estimated using a sample of hunter-collected wings, which can be identified as to age (juvenile, adult) and sex. The U.S. Fish and Wildlife Service provided records of 3,220 known-age and sex wings that were collected during the 1994-2006 hunting seasons. This sample, however, does not accurately reflect the structure of the post-breeding population because of cohort-specific differences in harvest rates; males and young typically have higher harvest rates than females and adults. I corrected for this differential sampling probability by dividing the ratio of young to adults by the ratio of their recovery rates (Martin and others, 1979):

$$R_t^{\bullet} = \frac{w_t^{J\bullet} / w_t^{A\bullet}}{f_t^{J\bullet} / f_t^{A\bullet}}$$

where R = the ratio of young to adults in the post-breeding population, w = number of wings, f = recovery rate, J = juvenile, A = adult, t = year, and $\bullet$ is a place-holder for sex. In spite of nearly even sex-ratios of most young waterfowl (Johnson and others, 1992), post-breeding age ratios are usually higher among females than males, presumably due to the relatively high mortality suffered by breeding females (Blohm and others, 1987, Johnson and others, 1992). The estimation of R in this manner implicitly assumes that reproductive success of 1-year-old birds is the same as that of older birds. While there is reason to question the validity of this assumption (Johnson and others, 1992), I was forced to accept it because the nature of the wing sample would not allow otherwise.

I initially constructed a 4-stage class, post-breeding model consisting of juvenile females, juvenile males, adult females, and adult males. However, the resulting projection matrix is reducible, meaning that there is not at least one path from every node to every other node in the life-cycle diagram (Fig. 3) (Caswell, 2001). A reducible projection matrix renders traditional methods of matrix model analysis (which rely on ergodic properties) useless except in very limiting cases (Caswell, 2001, p. 88-92). In this case, the dominant eigenvalue (i.e., the finite rate of population growth, λ) of the matrix would be equal to the eigenvalue of the sex that appears to be growing at the fastest rate, and the right (stable-stage distribution) and left (reproductive values) eigenvectors may not be strictly positive. Caswell (2001) demonstrated a way to overcome these problems by using a 2-sex model, in which the equilibrium dynamics (if they exist) are calculated directly from the life-cycle diagram. However, this solution is possible only if reproductive success, as a function of the population sex ratio, is known or assumed. Waterfowl have complex and variable mating systems (Oring and Sayler, 1992), and I was hesitant to posit a fertility function for mottled ducks. A more traditional solution to this dilemma has been to focus solely on

females, assuming that there will always be enough males to fertilize all the females. I also found this approach untenable because I was also interested in the demography of males. Therefore, I constructed sex-specific projection matrices, in which the first and second rows consisted of fecundities and survival rates, respectively:

$$\begin{bmatrix} S^{JF}R^{F} & S^{AF}R^{F} \\ S^{JF} & S^{AF} \end{bmatrix} \quad \text{and} \quad \begin{bmatrix} S^{JM}R^{M} & S^{AM}R^{M} \\ S^{JM} & S^{AM} \end{bmatrix}$$

where R, J, A, are as before, and S = survival rate, F = female, and M = male. Projection matrices for each sex were analyzed independently. Of primary interest was an understanding of patterns in sex-specific growth rates, but I was also interested in the extent to which growth rates from female-only and male-only projection matrices agreed. Assembling a matrix for each sex and year for which survivorship and reproductive success were available yielded a total of 24 matrices for analysis (Appendix 1). Each matrix represents the dynamics of mottled ducks from September of year t to August of year t+1.

Asymptotic analyses of projection matrices involved calculation of growth rates, and a random-design life table response experiment (LTRE, Caswell, 2001, p. 269-270) to estimate the variation in λ. Using the LTRE this variance was decomposed into contributions from the variances and covariances of the matrix entries, as well as from the lower-level parameters (i.e., the vital rates).

Also, a 1-way fixed-design LTRE (Caswell, 2001, p. 260-262) was used to investigate the effects of environmental moisture regimes on mottled duck growth rates. To characterize periods of varying moisture, I used the Palmer Drought Severity Index (PDSI, *http://www.drought.noaa.gov/palmer.html*), which is a monthly value indicating the balance between moisture supply and demand (i.e., evapotranspiration). PDSI values were acquired from the National Oceanic and Atmospheric Administration (*http://www1.ncdc.noaa.gov/pub/data/cirs/*) for the period September 1994 to August 2006 for four coastal regions of Texas and Louisiana. These included three (regions 7, 8, and 9) that covered all of coastal Louisiana and one (region 8) that covered most of the Texas coast. Temporal patterns of the PDSI were generally consistent among regions, so monthly values from the four regions were averaged to help discern range-wide moisture patterns (Fig. 4). Based on annual averages, 1998, 1999, and 2005 were classified as dry years (PDSI < -1); 1994, 1995, 1997, 2000, 2001, and 2004 as normal years (-1 ≤ PDSI ≤ +1); and 1996, 2002, and 2003 (PDSI > +1) as wet years (remember that for my purposes a "year" was defined as September through the following August). I used the sex-specific mean of all annual projection matrices as the reference matrix for this LTRE analysis.

Finally, to avoid relying on the asymptotic properties of the projection matrices, a simple population projection (Caswell, 2001, p. 11) was used to calculate the average growth rate for each sex over the period of record. These projections required an initial (1994) age structure for each sex, which was estimated in a manner analogous to that of the post-breeding age structure. From these stochastic projections, the geometric mean and standard error of the growth rates were calculated by using a log-transformation of λ:

$$\hat{r}_t = \log\left(\hat{\lambda}_t\right)$$

$$\hat{\bar{r}} = \frac{1}{T}\sum_{t=1}^{T}\hat{r}_t$$

$$\mathrm{se}\left(\hat{\bar{r}}\right) = \sqrt{\frac{\mathrm{var}\left(\hat{r}_t\right)}{T}}$$

Results were then compared with growth rates for comparable periods derived from the BBS, the CBC, the MWI in Texas and Louisiana, and the survey of national wildlife refuges in Texas.

Results

Among the 16 models examined to describe survivorship, the best model (as judged by AIC) was the one in which both recovery and survival rates varied by year and cohort (Table 1). The second-best model was similar in structure, except that survival rates were constant over time; however, the delta-AIC of this model was 39.4, clearly making it not worthy of further consideration. Using the best-fitting model, survival rates averaged 0.37 (se = 0.03), 0.47 (se = 0.04), 0.48 (se = 0.03), and 0.58 (se = 0.04) for juvenile females, adult females, juvenile males, and adult males, respectively (Table 2).

With respect to reproductive success, juvenile to adult ratios of the harvest tended to be higher for females (mean = 1.68, se = 0.21) than for males (mean = 1.01, se = 0.09) (Table 3). Although the data were quite variable, these discrepancies persisted after harvest age ratios were corrected for the differential vulnerability of young and adults (females: mean = 0.95, se = 0.15; males: mean = 0.61, se = 0.07).

Asymptotic growth rates for all 24 matrices varied between 0.49 and 1.16, with a geometric mean of 0.82. For males, the geometric mean λ was 0.85 (range: 0.65–1.15), and for females the mean λ was 0.79 (range: 0.49–1.16). Although the sex-specific values of the discrete growth rate (λ -1) for males and females tended to vary together (Fig. 5), the correlation coefficient was relatively low (0.31).

For females, the randomized-design LTRE predicted a var(λ) = 0.0405, which is similar to the observed var(λ) = 0.0433. This suggests that the first-order approximation of the variance described by Caswell (2001, p. 269) was sufficient. Variances of the matrix entries were large relative to the covariances, with the exception of the positive covariances of the fecundities because they shared a common reproductive rate. The variances of female matrix entries and their contributions to var(λ), respectively, were:

$$\begin{bmatrix} 0.0372 & 0.0361 \\ 0.0130 & 0.0202 \end{bmatrix}, \quad \begin{bmatrix} 0.0066 & 0.0080 \\ 0.0028 & 0.0068 \end{bmatrix}$$

Variation in fecundity was higher than that of survival, but all matrix entries except juvenile survival made important contributions to var(λ). A clearer picture of the determinants of var(λ) was obtained by examining the variances and contributions of the lower-level parameters. The variance-covariance matrix for the lower-level parameters was:

$$
\begin{array}{c|cccc}
 & S^{JF} & S^{AF} & R^{JF} & R^{AF} \\
\hline
S^{JF} & +0.0103 & +0.0101 & -0.0117 & -0.0117 \\
S^{AF} & +0.0101 & +0.0202 & -0.0386 & -0.0386 \\
R^{JF} & -0.0117 & -0.0386 & +0.2631 & +0.2631 \\
R^{AF} & -0.0117 & -0.0386 & +0.2631 & +0.2631
\end{array}
$$

The last two rows and columns are identical because of the assumption that reproductive rates are the same for birds of all ages. Reproductive success varied much more than survival, survival of juveniles and adults was positively related, and reproduction and survival were negatively related. The standardized contributions of the lower-level parameters to var(λ) were 0.1776, 0.4168, 0.1360, and 0.2696 for juvenile survival, adult survival, reproduction by 1-year-olds, and reproduction by older adults, respectively. In spite of the high variability in reproductive success among females, variation in juvenile and adult survival contributed 60% of the total variation in λ. Finally, it's worth noting that reproduction by birds older than one year appeared to contribute more to var(λ) than that by 1-year-olds, even though I had to assume that reproductive rates don't vary by age. This is so because the calculation of contributions involves the different sensitivities of juvenile and adult fecundity.

The LTRE predicted a var(λ) = 0.0252 for males, which is similar to the observed var(λ) = 0.0237. As with females, variances of the male matrix entries were large relative to the covariances, with the exception of the positive covariances of the fecundities. The variances of matrix entries and their contributions to var(λ), respectively, were:

$$
\begin{bmatrix} 0.0106 & 0.0131 \\ 0.0115 & 0.0153 \end{bmatrix}, \quad \begin{bmatrix} 0.0011 & 0.0041 \\ 0.0018 & 0.0069 \end{bmatrix}
$$

These results suggest a similar degree of variability in survival and fecundity, but it was adult male survival and fecundity that contributed the most to var(λ). In terms of lower-level parameters, the patterns of variances and covariances were similar to those for females:

$$
\begin{array}{c|cccc}
 & S^{JM} & S^{AM} & R^{JM} & R^{AM} \\
\hline
S^{JM} & +0.0115 & +0.0077 & -0.0114 & -0.0114 \\
S^{AM} & +0.0077 & +0.0153 & -0.0186 & -0.0186 \\
R^{JM} & -0.0114 & -0.0186 & +0.0670 & +0.0670 \\
R^{AM} & -0.0114 & -0.0186 & +0.0670 & +0.0670
\end{array}
$$

Among males, reproductive success varied more than survival, survival of juveniles and adults was positively related, and reproduction and survival were negatively related. The standardized contributions of the lower-level parameters to var(λ) were 0.1437, 0.5477, 0.0588, and 0.2498 for juvenile survival, adult survival, reproduction by 1-year-olds, and reproduction by older adults, respectively. Variation in juvenile and adult survival contributed almost 70% of the total variation in male λ.

The 1-way, fixed-design LTRE for the effects of dry, normal, and wet periods predicted growth rates extremely well using the approximation for treatment effects provided by Caswell (2001, p. 261) (Table 4). For males, the pattern of λ's were consistent with my *a priori* hypothesis (i.e., increasing growth rates with increasingly wet conditions), but λ was <1 even in wet periods. Adult fecundity and survival contributed most to the relatively low growth rate during the dry period and to the higher growth rate during the wet period (Fig. 6). For females, λ was similar and <1 for all three periods. During the dry period, adult fecundity and survival made large, negative contributions to λ. During the wet period, the contributions of female fecundity were positive, but were overwhelmed by the negative contributions of juvenile and adult survival.

For the stochastic projections, I estimated the following initial age structures:

$$n_{1994}^{M} = \begin{bmatrix} 0.42 \\ 0.58 \end{bmatrix} \qquad n_{1994}^{F} = \begin{bmatrix} 0.41 \\ 0.59 \end{bmatrix}$$

The mean $r = \log(\lambda)$ for the entire time period was -0.17 (se = 0.05) and -0.23 (se = 0.09) for males and females, respectively. The approximate 95% confidence intervals for these rates do not include zero, in spite of the fact that the standard errors include both sampling and process error. The growth rates from the stochastic projection are nearly identical to the asymptotic rates derived from the mean matrices (males: -0.15; females: -0.22). This was not surprising given the models' limited age structure, which implies that variation in the initial population structure will not produce significant transient dynamics.

Next, mean *r*'s were compared with those derived from other population surveys (Fig. 7). Although an attempt was made to align the years used from other surveys with those of this study, this could not be done exactly because the population surveys have different anniversary dates than those of my models. For example, my "year" 1994 extended from September 1994 to August 1995. I chose to align this year with the 1995 CBC (winter of 1994-95), the 1995 MWI in Texas and Louisiana (January 1995), and the 1995 BBS (June 1995). Thus, my model-based growth rates apply to the period September 1994 to August 2006, while those from the other surveys cover the period from early to mid-1995 to the same time in 2007. Note also that BBS data were available only through 2006, and that *r* from the BBS was calculated on-line (*http://www.mbr-pwrc.usgs.gov/bbs/*) due to the complexity of the analyses (*http://www.mbr-pwrc.usgs.gov/bbs/trendin.html*). Although all of the mean *r*'s from these population surveys were negative, none were significantly different from zero. The MWI had the highest rate (*r* = -0.01, se = 0.10), followed by the CBC (*r* = -0.03, se = 0.12), the BBS (*r* = -0.07), and the survey of Texas national wildlife refuges (*r* = -0.15, se = 0.15).

Discussion

Estimated survival rates of mottled ducks of the WGC population were low compared to most dabbling ducks (*Anatinae*) (Krementz and others, 1997, Johnson and others, 1992), as well to mottled ducks in Florida (Johnson and others, 1995). Estimates of reproductive rates derived from hunter-collected wings and adjusted for the differential vulnerability of juveniles and adults also were low relative to those for closely-related species, which often are ≥1 (S. Boomer, U.S. Fish & Wildlife Service, pers. comm.). Both asymptotic and stochastic growth rates of WGC mottled ducks were substantially <1 for both sexes. However, in absolute terms, male and female growth rates did not always agree and this could be attributed to un-modeled sampling error. Generally, sample sizes of band-recovery data were larger for males than females, and coefficients of variation

on annual survival rates were about twice as high for females than males. The number of wing samples was similar for males and females (1662 males vs. 1558 females), but annual samples were relatively small (averaging about 130 each year for each sex).

This study focused on variation in growth rates, but it is important to recognize that growth rates were derived from vital-rate estimates with varying degrees of sampling error. Unfortunately, standard procedures for dealing with sampling error are of limited utility when dealing with non-linear functions of estimates in which the underlying distributions of the functions are unknown (e.g., we have no reason to expect *a priori* that the distribution of λ will be normal; in fact, it probably is not). Caswell (2001) provides some analytical approximations and simulation tools to quantify the uncertainty in sample-based estimates of λ, but a full likelihood-based approach for analyzing matrix models does not yet exist. Hoekman and others (2002), following a suggestion of Caswell (2001), used a simulation approach to isolate process variance from total variance, but the reliability of the approach depends on good estimates of the process variance for, and covariances among, each demographic parameter, as well as an assumed distribution for each demographic parameter. I have not yet applied these procedures to mottled ducks, so the reader must consider the possibility that patterns in λ documented in this study may be influenced by sampling error; this is especially true of the LTRE analyses. On a more positive note, stochastic growth rates in which total variances (process + sampling error) were included were very similar to those derived from asymptotic analyses of deterministic matrices. Finally, patterns in growth rates and matrix entries were similar to those for other avian species (Stahl and Oli, 2006), and generally consistent with mottled duck ecology as we understand it.

Among both sexes, fall age ratios varied much more than survivorship, but variation in survival contributed more than half of the total variance in λ. This result is consistent with many other avian species (Stahl and Oli, 2006), although notably different than mid-continent mallards where reproductive rates played a major role in var(λ) (Hoekman and others, 2002). Population growth rates of males responded as expected to environmental moisture regimes, with increasing growth rates associated with increasingly wet conditions. Although male λ was <1 for all periods, it was close to one during the wet period. On the other hand, female growth rates varied little with moisture conditions and were well <1 for all periods. The large negative contributions of female survival during the wet period may reflect an ecological cost associated with the high level of nesting and re-nesting expected in wet years. That large, negative contributions of survival were not observed among males in the wet period lends credence to this hypothesis because males do not help with incubation or brood-rearing.

These results suggest that mottled ducks declined dramatically during the period of record, but the severity of the predicted decline does not agree with that observed in three of four population surveys (the survey of national wildlife refuges on the Texas coast being the exception). There could be a number of reasons for these discrepancies. First, one would not expect mottled duck vital rates to be homogenous across their range, and this variation could manifest itself as varying levels of abundance depending on the specific period and area sampled. However, estimates of growth rates derived from the population surveys may also be suspect: the MWI often experiences unaccounted-for changes in methodology (Eggeman and Johnson, 1989; D. Morrison, Texas Parks and Wildlife Dept., pers. comm.); the BBS and CBC surveys rely solely on what can be observed from roads (which may not be representative habitat); and none of these surveys account for the imperfect probability of detecting mottled ducks, given that they are present. Of all the population surveys, the survey of Texas national wildlife refuges is probably the most reliable because of its rigorous sampling design, its explicit correction for imperfect detection (see Smith,

1995), and its consistent methodology over time. Nonetheless, it can be criticized for sampling only a (possibly unrepresentative) portion of the range of WGC mottled ducks. Likewise, my estimates of survivorship and reproductive success could be biased for a variety of reasons, including unrepresentative banding locations, or the non-random sample of hunter-killed wings. Also, the extent to which my results depend on the assumption of equal productivity among 1-year-old and older birds is unknown.

On the whole, however, most lines of evidence suggest that the abundance of mottled ducks in the WGC population has declined - if not in the short term, then almost certainly over the long term (see Stutzenbaker, 1998). Loss of estuarine and palustrine wetlands along the coasts of Louisiana and Texas in the last half-century has exceeded a million acres due to subsidence, sea-level rise, and conversion to agriculture (B. Wilson, unpub. data). Salt-water intrusion has also increased with sea-level rise and anthropogenic development (Chabreck and Linscombe, 1982), and Moorman and others (1991) found that salinities >9 ppt had negative effects on mottled duck brood survival. Unfortunately, I was unable to investigate the potential role of sport hunting on growth rates because hunting regulations were relatively constant throughout the period of this study. However, hunting regulations during the last 10 years have been the most liberal since the early 1950's, and the mottled duck is highly prized by hunters. Because of the sensitivity of growth rates to adult survival, a well-designed, experimental manipulation of hunting regulations could help shed light on how varying levels of harvest might affect mottled duck growth rates. Such experiments have been repeatedly called for in other duck species (e.g., Anderson and others, 1987).

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Table 1. Comparison of band-recovery models for post-breeding mottled ducks in Texas and Louisiana, 1994-2006. S = survival rate, f = recovery rate, g = cohort effect, t = year effect, "*" denotes interaction effects, and "." denotes a constant rate.

Model	Parameters	AIC	Delta AIC
{S(g*t) f(g*t)	100	44582.7	0.0
{S(g) f(g*t)	56	44622.1	39.4
{S(t) f(g*t)	64	44743.4	160.8
{S(.) f(g*t)	53	44802.1	219.4
{S(g*t) f(g)	52	44819.2	236.5
{S(g) f(g)	8	44878.6	295.9
{S(g*t) f(t)	61	44886.4	303.8
{S(g) f(t)	17	44929.3	346.6
{S(t) f(g)	16	44943.9	361.2
{S(.) f(g)	5	45039.4	456.7
{S(g*t) f(.)	49	45118.4	535.8
{S(t) f(t)	25	45163.0	580.3
{S(g) f(.)	5	45174.3	591.6
{S(.) f(t)	14	45251.5	668.8
{S(t) f(.)	13	45394.2	811.6
{S(.) f(.)	2	45494.7	912.0

Table 2. Estimates of recovery (f) and survival (S) rates of post-breeding mottled ducks banded in Texas and Louisiana, 1994-2006.

		Juvenile				Adult			
Sex	**Year**	**f**	**se(f)**	**S**	**se(S)**	**f**	**se(f)**	**S**	**se(S)**
Females	1994	0.0238	0.0054	0.4756	0.1054	0.0267	0.0132	0.8171	0.3814
	1995	0.0386	0.0064	0.3530	0.0658	0.0130	0.0035	0.4245	0.1169
	1996	0.0529	0.0060	0.3704	0.0496	0.0297	0.0075	0.3965	0.1079
	1997	0.0858	0.0076	0.4290	0.0807	0.0475	0.0081	0.5717	0.1324
	1998	0.0708	0.0126	0.5206	0.1059	0.0347	0.0066	0.5710	0.1382
	1999	0.0804	0.0094	0.3631	0.0575	0.0403	0.0076	0.3624	0.0828
	2000	0.1153	0.0101	0.2474	0.0440	0.0514	0.0062	0.3267	0.0744
	2001	0.0716	0.0081	0.4592	0.0755	0.0548	0.0110	0.5686	0.1590
	2002	0.0811	0.0101	0.4580	0.0838	0.0382	0.0085	0.4761	0.1259
	2003	0.0754	0.0085	0.1931	0.0359	0.0566	0.0097	0.4019	0.1159
	2004	0.1205	0.0102	0.2611	0.0508	0.0511	0.0105	0.3940	0.1270
	2005	0.0794	0.0080	0.3425	0.0676	0.0317	0.0058	0.3224	0.0793
	2006	0.0737	0.0071			0.0647	0.0071		
	mean	0.0746		0.3728		0.0416		0.4694	
	se	0.0073		0.0292		0.0040		0.0411	
Males	1994	0.0462	0.0074	0.5758	0.0887	0.0196	0.0112	0.6595	0.1977
	1995	0.0617	0.0084	0.3719	0.0496	0.0372	0.0048	0.4177	0.0514
	1996	0.1050	0.0082	0.5606	0.0529	0.0665	0.0077	0.6955	0.0883
	1997	0.0943	0.0073	0.5058	0.0682	0.0560	0.0064	0.6743	0.0835
	1998	0.0944	0.0124	0.5989	0.0861	0.0531	0.0058	0.6426	0.0770
	1999	0.0926	0.0087	0.5294	0.0574	0.0553	0.0057	0.5319	0.0547
	2000	0.1225	0.0093	0.3824	0.0423	0.0792	0.0053	0.5839	0.0806
	2001	0.1110	0.0093	0.5999	0.0722	0.0571	0.0078	0.5628	0.0905
	2002	0.0926	0.0093	0.4823	0.0628	0.0549	0.0065	0.5697	0.0777
	2003	0.1088	0.0096	0.3136	0.0406	0.0665	0.0069	0.6416	0.0852
	2004	0.1361	0.0101	0.5011	0.0740	0.0659	0.0064	0.6839	0.1119
	2005	0.0705	0.0073	0.3062	0.0496	0.0389	0.0051	0.2763	0.0487
	2006	0.1018	0.0077			0.0826	0.0071		
	mean	0.0952		0.4773		0.0564		0.5783	
	se	0.0068		0.0310		0.0048		0.0357	

Table 3. Estimates of the ratio of juvenile to adults in the harvest (harv), the ratio of young to adult band-recovery rates (f), and the ratio of young to adults in the post-breeding population (pop) of mottled ducks on the western coast of the Gulf of Mexico, 1994-2005.

	Females			Males		
Year	**J:A (harv)**	**J:A (f)**	**J:A (pop)**	**J:A (harv)**	**J:A (f)**	**J:A (pop)**
1994	0.9231	2.9774	0.3100	0.8077	2.9774	0.4865
1995	1.7143	1.7817	0.9622	1.0755	1.7817	0.68102
1996	1.7931	1.8078	0.9919	1.3774	1.8078	0.81724
1997	2.4783	2.0371	1.2166	0.9565	2.0371	0.53767
1998	1.1071	1.9960	0.5547	0.5455	1.9960	0.3258
1999	1.0606	2.2417	0.4731	0.4565	2.2417	0.29517
2000	1.6923	1.3054	1.2964	0.9216	1.3054	0.47388
2001	0.8909	2.1219	0.4199	0.9773	2.1219	0.58001
2002	1.9783	1.3324	1.4847	1.3958	1.3324	0.85326
2003	1.0769	2.3583	0.4567	0.9423	2.3583	0.45628
2004	3.2000	2.5019	1.2790	1.1600	2.5019	0.63932
2005	2.2174	1.1403	1.9446	1.5172	1.14031	1.2312
mean	1.6777	1.9668	0.9491	1.0111	1.9668	0.6148
se	0.2062	0.1545	0.1481	0.0932	0.1545	0.0747

Table 4. Results of a 1-way fixed-design LTRE, in which the effects of environmental moisture regimes on population growth rates (λ) were examined for mottled ducks in Texas and Louisiana, 1994-2005. Delta(λ) represents the difference between the period-specific λ and λ from the mean of the matrices for all periods. C represents the contribution matrix; a good model fit is suggested when the sum of its entries are similar to Delta(λ).

Sex	Period	λ (observed)	λ (predicted)	Delta(λ)	∑(C)
Male	Dry	0.7197	0.7200	-0.1392	-0.1389
	Normal	0.8719	0.8719	0.0130	0.0130
	Wet	0.9612	0.9611	0.1023	0.1022
Female	Dry	0.7888	0.7888	-0.0140	-0.0140
	Normal	0.8216	0.8216	0.0189	0.0189
	Wet	0.7841	0.7814	-0.0186	-0.0186

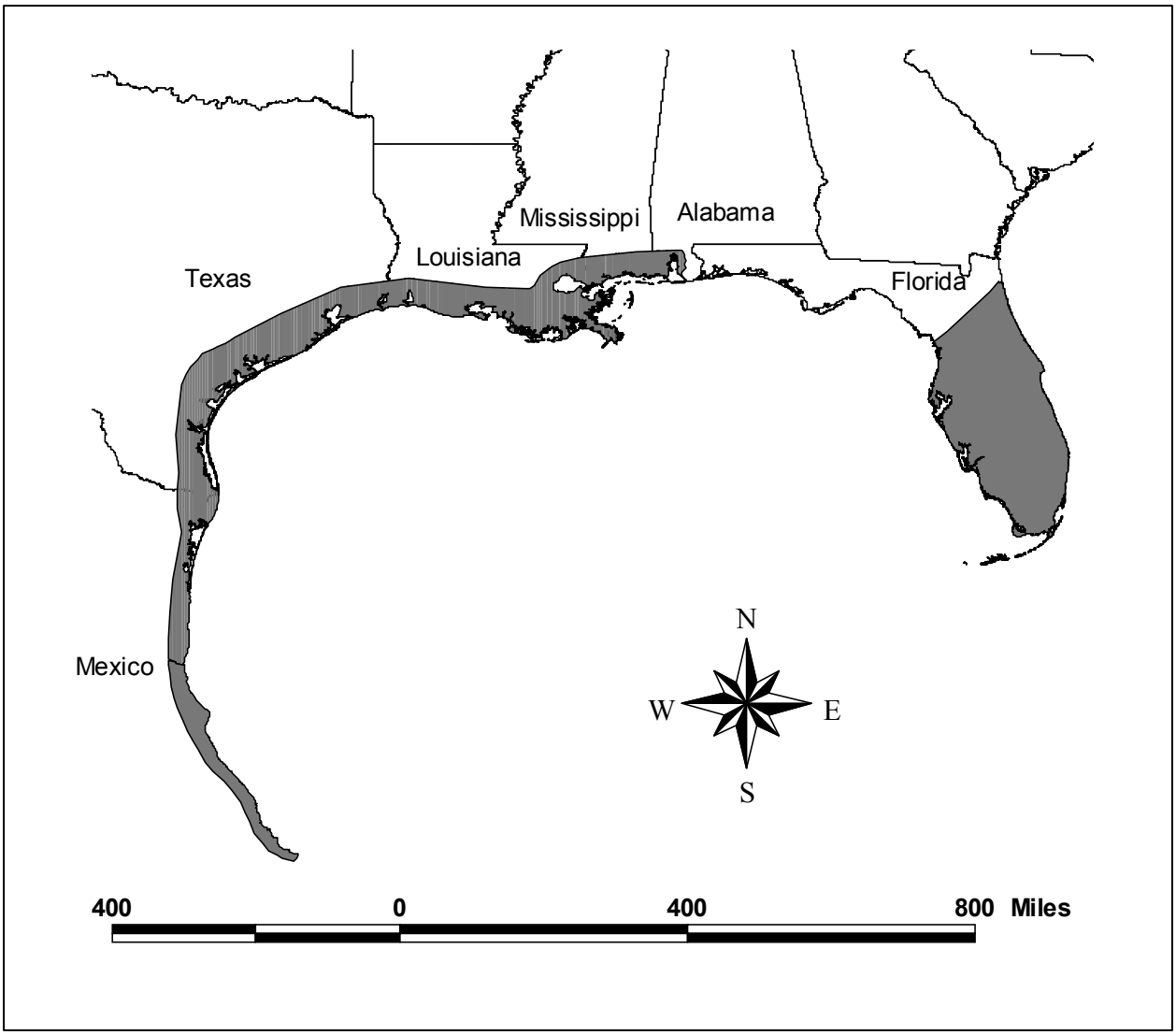

Figure 1. Range of the mottled duck (Ridgely and others, 2003).

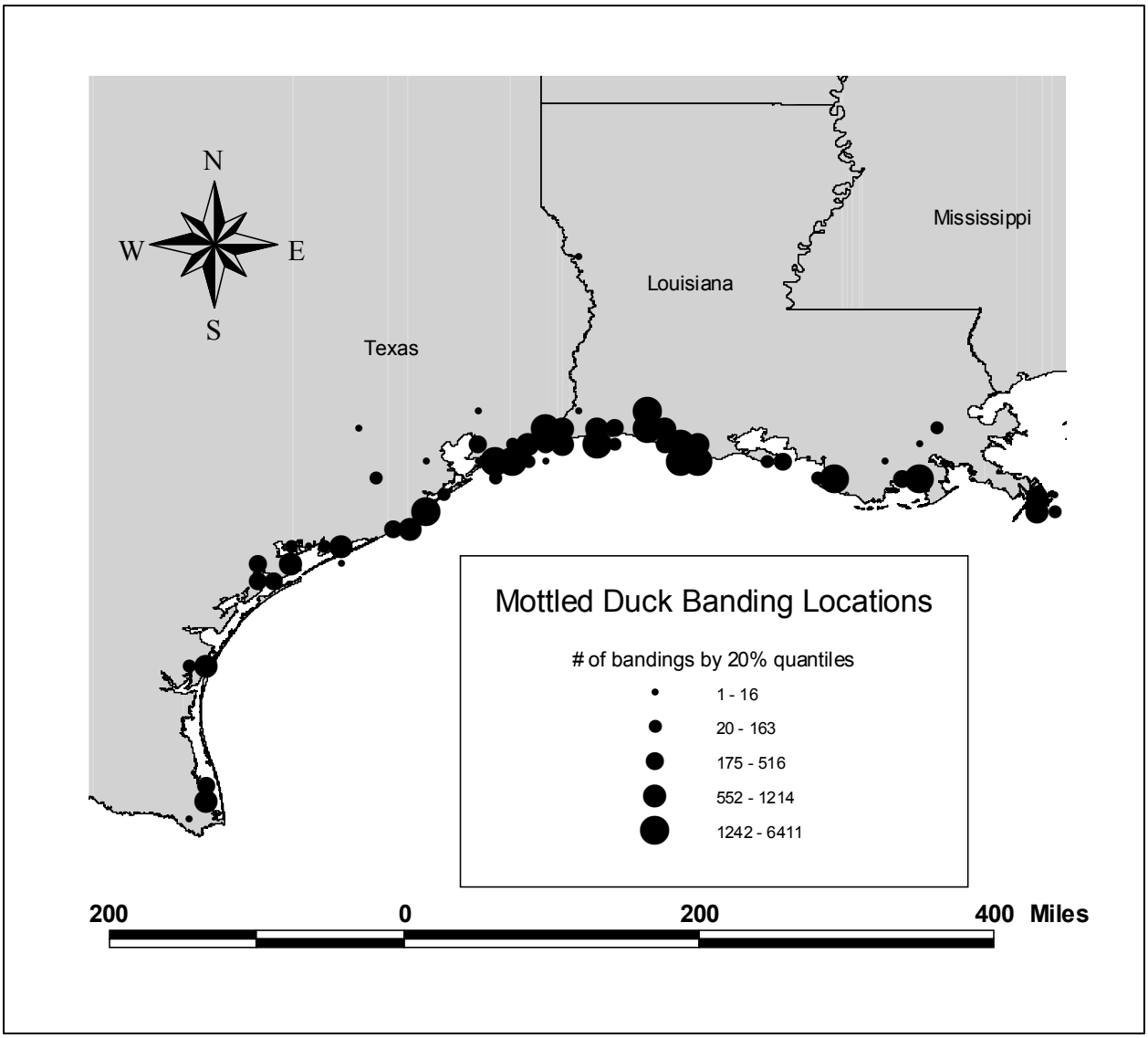

Figure 2. Distribution of bandings of post-breeding mottled ducks in Texas and Louisiana, 1994-2006.

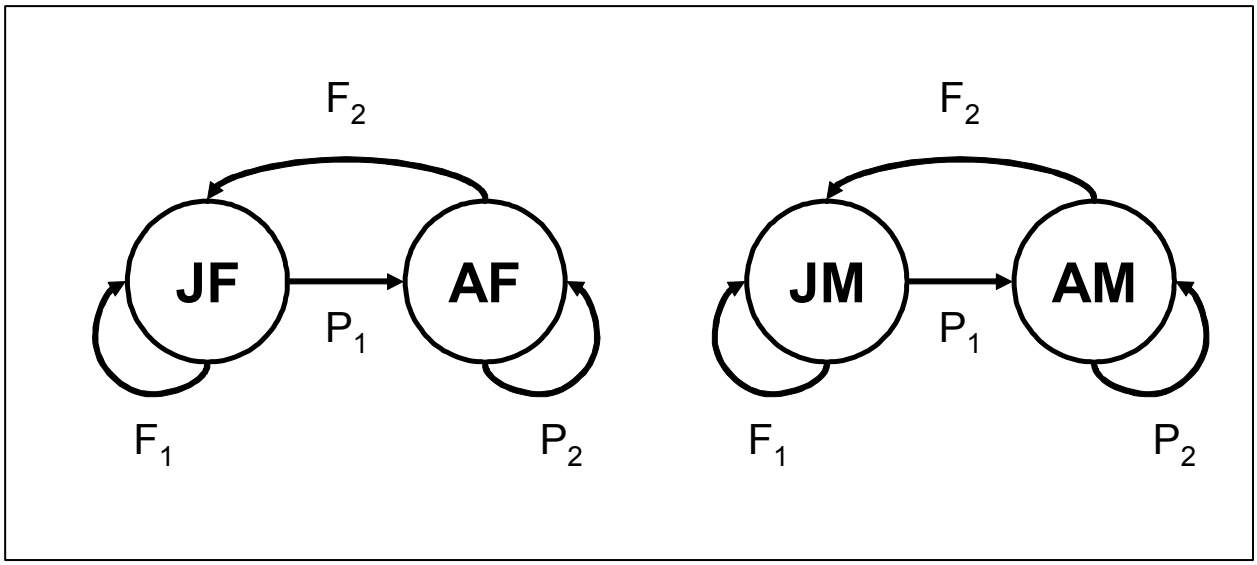

Figure 3. Life-cycle diagram used for modeling the post-breeding mottled duck population in Texas and Louisiana, 1994-2005.

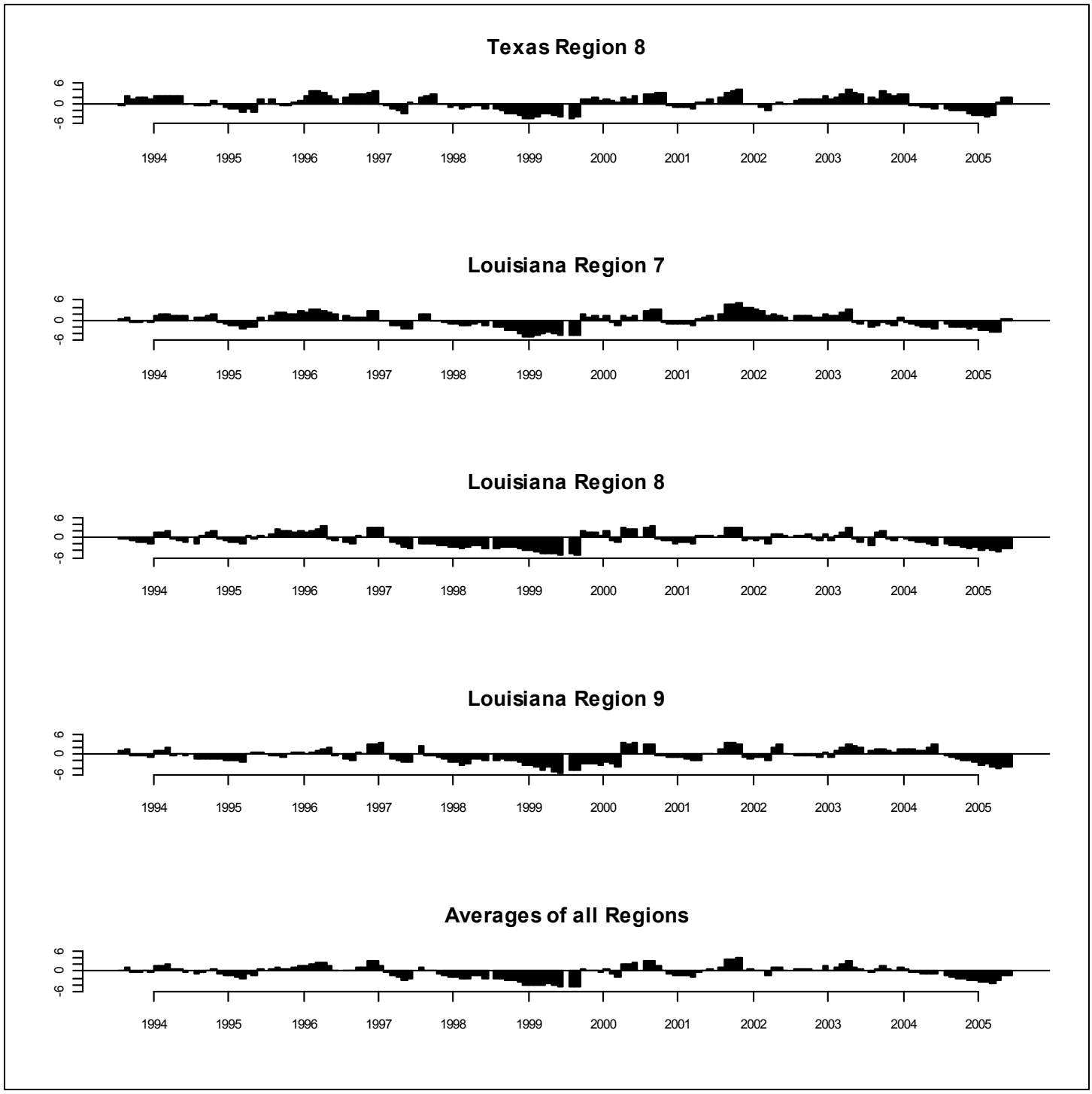

Figure 4. Monthly Palmer Drought Severity Indices for September 1995 to August 2006.

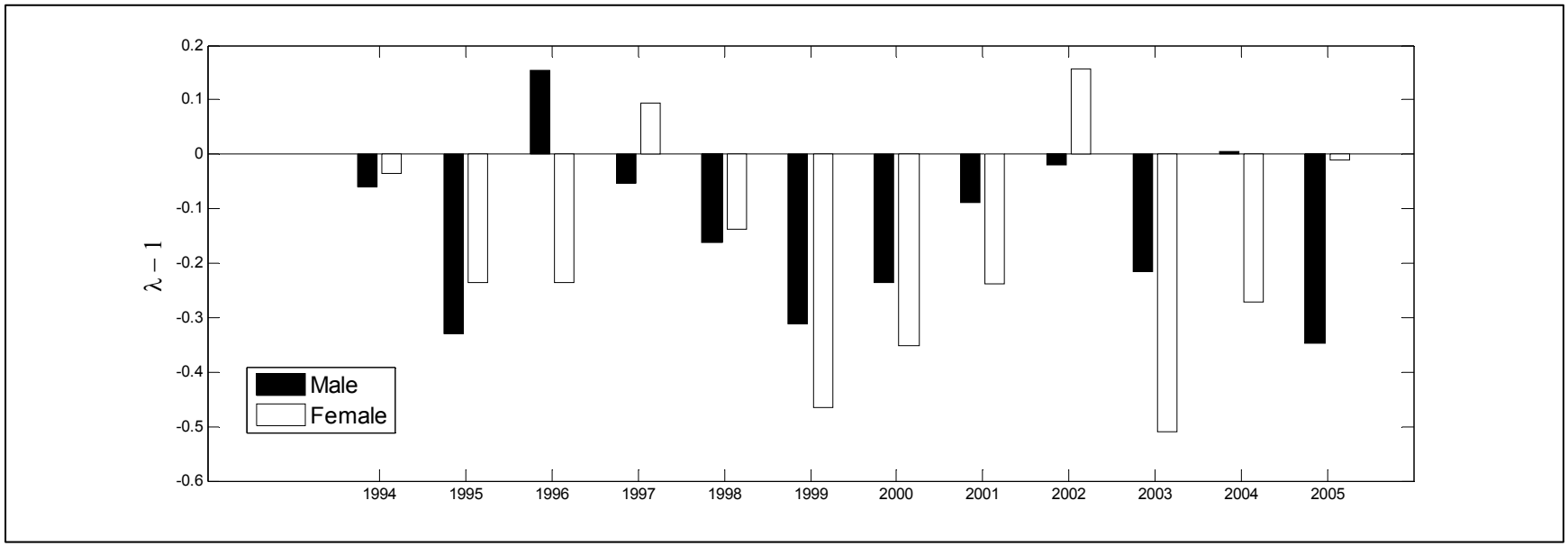

Figure 5. Discrete population growth rates, λ - 1, for mottled ducks in Texas and Louisiana, 1994-2005, as based on estimates of survival and fecundity.

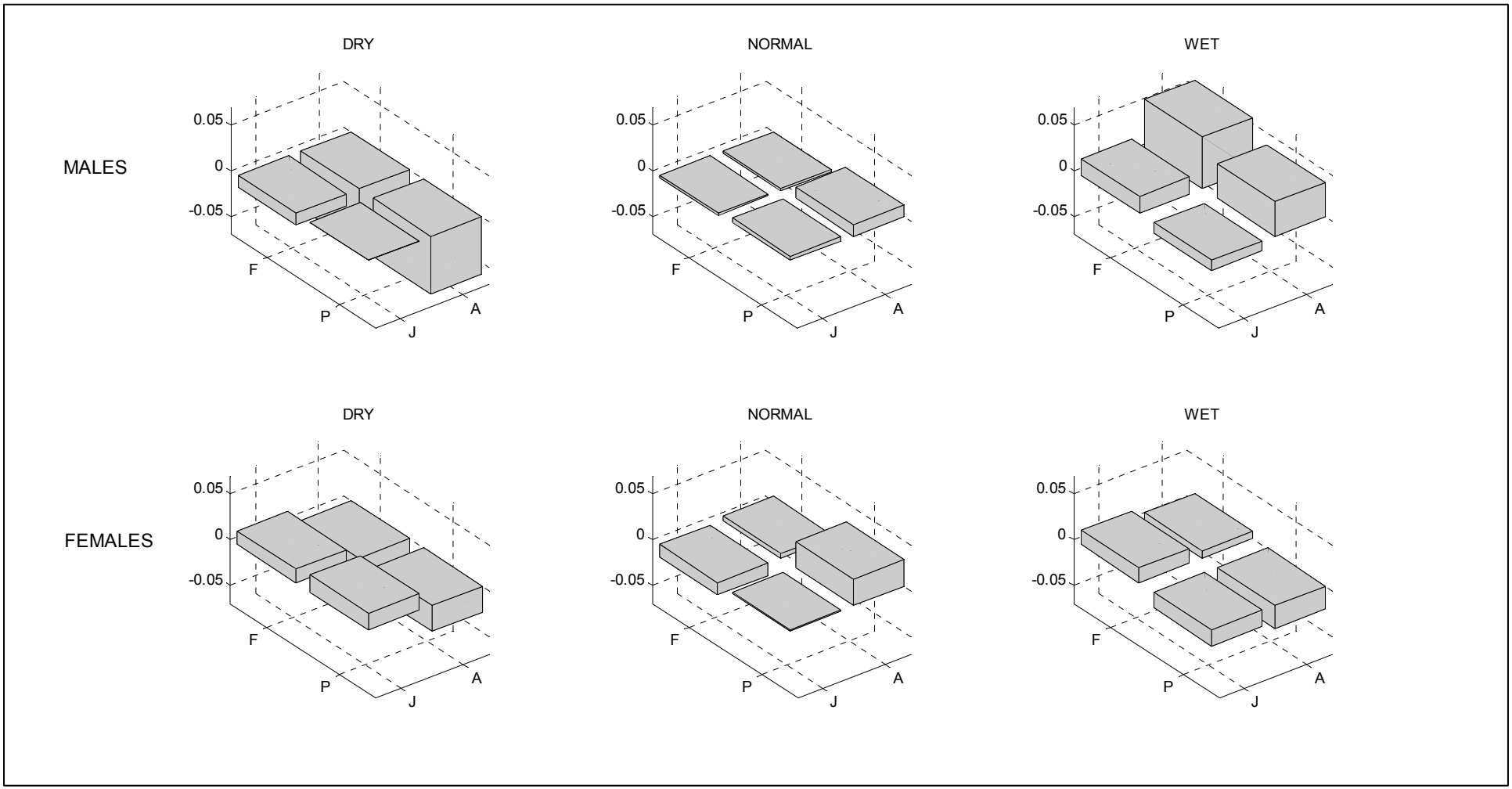

Figure 6. Contributions of projection-matrix entries for mottled ducks during dry, normal, and wet periods along the coast of Texas and Louisiana, 1994-2005. F = fecundity, P = survivorship, J = juvenile, and A = Adult.

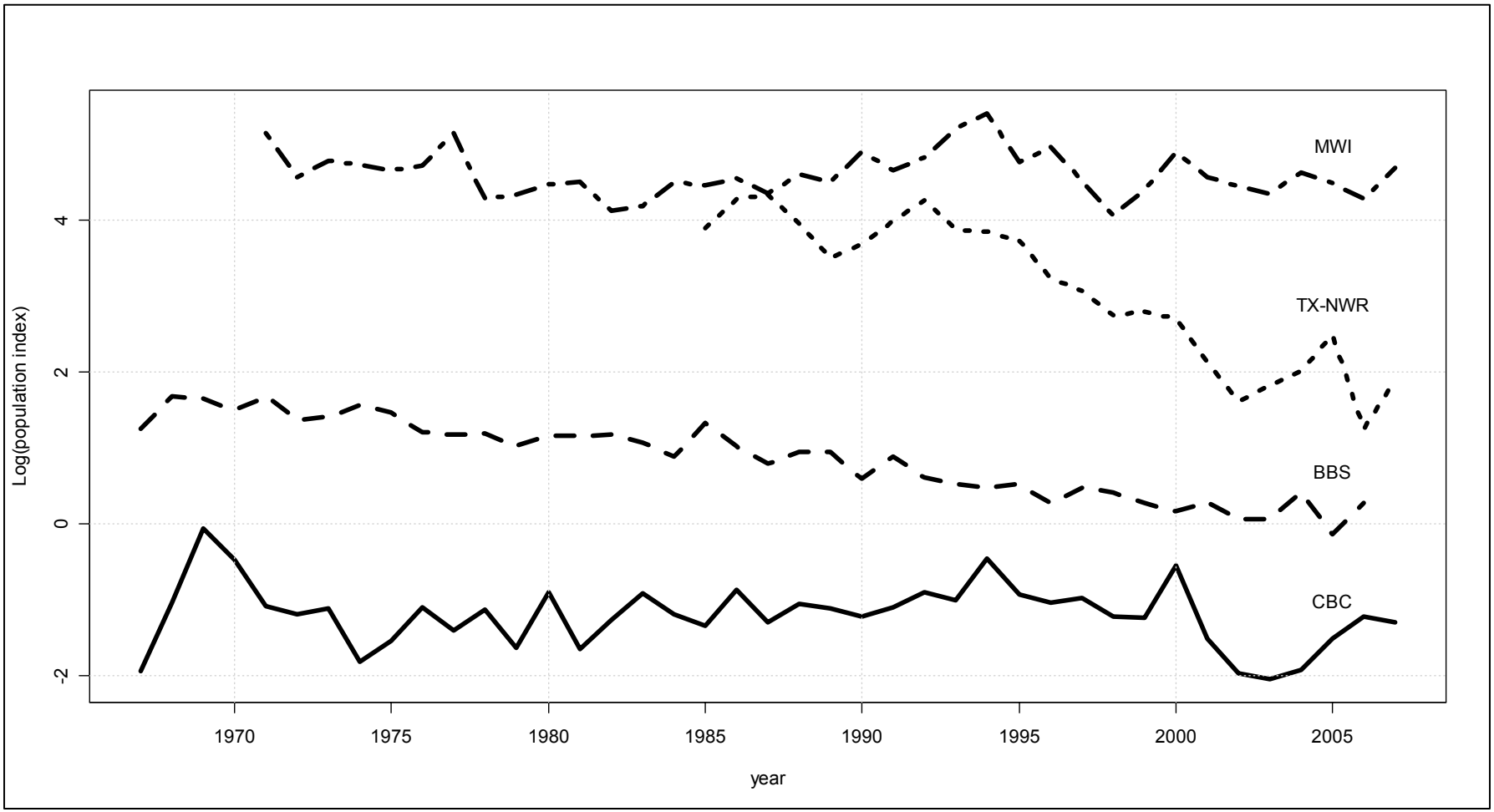

Figure 7. Results of population surveys of mottled ducks on the western coast of the Gulf of Mexico. MWI = Midwinter Inventory, TX-NWR = survey of national wildlife refuges in coastal Texas, BBS = Breeding Bird Survey, and CBC = Christmas Bird Count.

Appendix 1 Two-age class projection matrices for male and female mottled ducks in Texas and Louisiana, 1994-2005.

	Males	Females
1994	$\begin{bmatrix} 0.2801 & 0.3208 \\ 0.5758 & 0.6595 \end{bmatrix}$	$\begin{bmatrix} 0.1474 & 0.2533 \\ 0.4756 & 0.8171 \end{bmatrix}$
1995	$\begin{bmatrix} 0.2533 & 0.2845 \\ 0.3719 & 0.4177 \end{bmatrix}$	$\begin{bmatrix} 0.3397 & 0.4085 \\ 0.3530 & 0.4245 \end{bmatrix}$
1996	$\begin{bmatrix} 0.4581 & 0.5684 \\ 0.5606 & 0.6955 \end{bmatrix}$	$\begin{bmatrix} 0.3674 & 0.3933 \\ 0.3704 & 0.3965 \end{bmatrix}$
1997	$\begin{bmatrix} 0.2720 & 0.3626 \\ 0.5058 & 0.6743 \end{bmatrix}$	$\begin{bmatrix} 0.5219 & 0.6955 \\ 0.4290 & 0.5717 \end{bmatrix}$
1998	$\begin{bmatrix} 0.1951 & 0.2094 \\ 0.5989 & 0.6426 \end{bmatrix}$	$\begin{bmatrix} 0.2918 & 0.3167 \\ 0.5260 & 0.5710 \end{bmatrix}$
1999	$\begin{bmatrix} 0.1563 & 0.1570 \\ 0.5294 & 0.5319 \end{bmatrix}$	$\begin{bmatrix} 0.1718 & 0.1715 \\ 0.3631 & 0.3264 \end{bmatrix}$
2000	$\begin{bmatrix} 0.1812 & 0.2767 \\ 0.3824 & 0.5839 \end{bmatrix}$	$\begin{bmatrix} 0.3207 & 0.4235 \\ 0.2474 & 0.3267 \end{bmatrix}$
2001	$\begin{bmatrix} 0.3479 & 0.3264 \\ 0.5999 & 0.5628 \end{bmatrix}$	$\begin{bmatrix} 0.1928 & 0.2388 \\ 0.4592 & 0.5686 \end{bmatrix}$
2002	$\begin{bmatrix} 0.4115 & 0.4861 \\ 0.4823 & 0.5697 \end{bmatrix}$	$\begin{bmatrix} 0.6800 & 0.7069 \\ 0.4580 & 0.4761 \end{bmatrix}$
2003	$\begin{bmatrix} 0.1431 & 0.2928 \\ 0.3136 & 0.6416 \end{bmatrix}$	$\begin{bmatrix} 0.0882 & 0.1835 \\ 0.1931 & 0.4019 \end{bmatrix}$
2004	$\begin{bmatrix} 0.3204 & 0.4372 \\ 0.5011 & 0.6839 \end{bmatrix}$	$\begin{bmatrix} 0.3339 & 0.5039 \\ 0.2611 & 0.3940 \end{bmatrix}$
2005	$\begin{bmatrix} 0.3770 & 0.3402 \\ 0.3062 & 0.2763 \end{bmatrix}$	$\begin{bmatrix} 0.6660 & 0.6269 \\ 0.3425 & 0.3224 \end{bmatrix}$
Mean	$\begin{bmatrix} 0.2830 & 0.3385 \\ 0.4773 & 0.5783 \end{bmatrix}$	$\begin{bmatrix} 0.3435 & 0.4102 \\ 0.3732 & 0.4694 \end{bmatrix}$