

# A Revised Protocol for the Adaptive Harvest Management of Eastern Mallards

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**Abstract:** The set of population models used for the adaptive harvest management (AHM) of eastern mallards has been reviewed and revised. The revised set of six models: (1) rely solely on federal and state waterfowl surveys rather than the Breeding Bird Survey to index breeding-population size; (2) allow for the possibility of a positive bias in estimates of survival and reproductive rates; and (3) incorporate competing hypotheses of strongly and weakly density-dependent reproduction. A retrospective updating of model weights since 1995 suggests that the bias-corrected models were better predictors than the uncorrected models by a ratio of 3:1, and that the models assuming strongly density-dependent reproduction outperformed those assuming weakly density-dependent reproduction by a ratio of 2:1. Based on the revised models and most recently available model weights (i.e., those from 2001), eastern mallards appear to have considerable potential to tolerate sport harvest without adverse impact. However, we believe caution is warranted in promulgating hunting regulations in the Atlantic Flyway based solely on the status of eastern mallards because of potential adverse impacts to duck species with lower harvest potential.

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## Background

The biology of eastern mallards appears to differ from that of midcontinent mallards in several important ways (Fig. 1). The size of the midcontinent breeding population has been fairly stable over time, and numerically is much larger than the eastern population. The eastern population appears to be more productive than the midcontinent population, however, and based on the Breeding Bird Survey (BBS) (<http://www.mp2-pwrc.usgs.gov/bbs//index.html>) has been growing at least since the mid-1960's. Accordingly, waterfowl managers suspect there may be a significant difference between midcontinent and eastern mallards in sustainable rates of harvest. Moreover, environmental conditions affecting the annual status of the two populations likely are weakly correlated because there is minimal overlap in the birds' breeding, migration, and wintering areas. Because 75% of the eastern mallards harvested in the U.S. occurs in the Atlantic Flyway, the Atlantic Flyway Council has had a long-standing interest in developing harvest strategies that explicitly recognize the differences in these two mallard populations.

When AHM was first implemented in 1995, a common regulatory alternative (i.e., restrictive, moderate, or liberal season) was prescribed for all four Flyways based on the population dynamics and status of midcontinent mallards. By early 2000, however, the AHM Working Group had determined how the existing adaptive harvest management (AHM) protocol might be modified to explicitly recognize the differences between midcontinent and eastern mallards (*Johnson, F., J. Dubovsky, D. Eggeman, and M. Moore, 2000, Adaptive Harvest Management for Eastern Mallards: Progress Report, U.S. Fish and Wildlife Service, 17pp;*

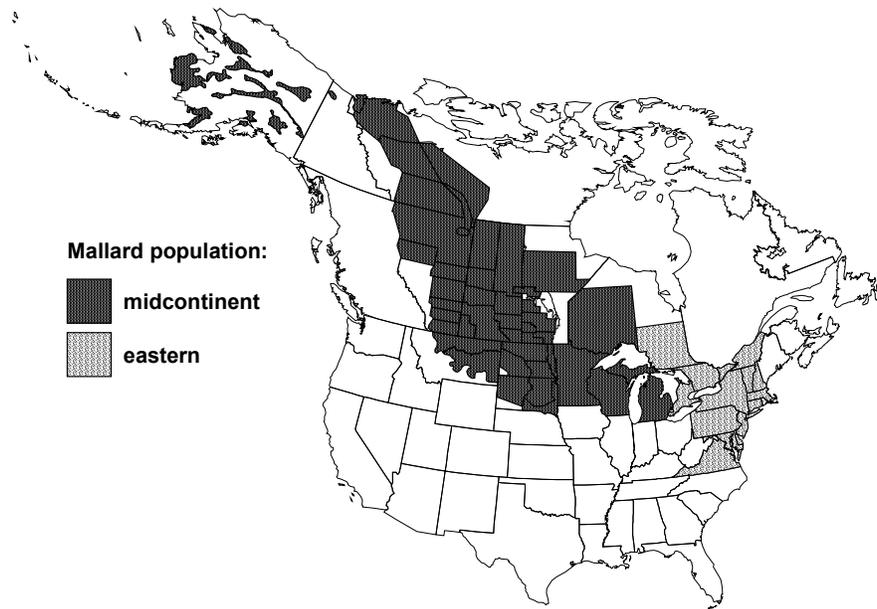


Fig. 1. Survey areas currently assigned to the midcontinent and eastern populations of mallards for purposes of adaptive harvest management.

available online at <http://migratorybirds.fws.gov/mgmt/ahm/ahm-intro.htm>). This involved an explicit recognition of the annual status of both populations, as well as how mallards distributed themselves among the Flyways during the hunting season. Originally, the idea was to allow Flyway-specific regulatory strategies, which for each Flyway represented an average of the optimal regulatory strategies for each mallard population, weighted by the relative contribution of each population to the Flyway’s fall flight. This “joint optimization” thus involved:

- (1) revision of the objective function for midcontinent mallards to account for harvest-management goals for eastern mallards;
- (2) inclusion of models of eastern-mallard population dynamics to predict the effects of regulations; and
- (3) modification of the existing decision rules to allow Flyway-specific regulatory choices.

Investigation of the joint-optimization approach suggested, however, that the optimal regulation within each Flyway was affected principally by a single population of mallards. In other words, the status of midcontinent mallards appeared to have no discernable effect on regulations in the Atlantic Flyway, nor did the status of eastern mallards seem to have a significant effect on regulations in the other Flyways. In retrospect, these results are consistent with the high degree of spatial separation between the two populations during the hunting season. Therefore, the U.S. Fish and Wildlife Service (USFWS) decided to implement separate AHM protocols for midcontinent and eastern mallards, whereby the Atlantic Flyway regulation would be based solely on eastern mallards, and the regulation in the other Flyways would be based solely on midcontinent mallards. This approach was first implemented in the 2000-01 hunting season, and continues to be considered provisional until its implications, particularly for species other than mallards, are better understood.

The AHM Working Group is continuing to investigate spatial differences in the ecology of mallards and how those differences might be recognized explicitly in the AHM process. Recently, the AHM Working Group determined that major revisions to the existing set of population models for eastern mallards may be warranted

prior to the 2002-03 hunting season. There are several reasons why revisions may be appropriate. First, a reduction in the number of models may be possible because current differences in model-specific regulatory strategies are relatively minor. Another motivation concerns the tendency for empirical estimates of survival and reproductive rates of several duck species (including mallards) to imply annual growth rates that are higher than those observed in surveys of population size. Finally, the Breeding Bird Survey (BBS) index currently used to predict reproductive success of eastern mallards may be biased low in years of high spring precipitation.

This purpose of this report is to describe the AHM Working Group's efforts to address these modeling issues. Final decisions regarding modification of the model set for eastern mallards will be made after the USFWS has discussed resulting management implications with the Flyway Councils, States, and the general public (*Federal Register 67:12506*, available online at <http://policy.fws.gov/frsystem>).

## Data Sources

*Survival rates.*—We used banding and recovery records of normal, wild mallards banded prior to the hunting season in reference areas 8, 15, and 16 (Anderson and Henny 1972), which correspond approximately to the geographic bounds used in AHM for the eastern mallard population. We conducted two analyses, in both cases by pooling banding data across reference areas. In the first, we simply estimated annual survival rates by age and sex for the period 1960-98 using the standard Brownie et al. (1985) model that allows for age, sex, and year dependency in band-recovery and survival rates. In the second, we partitioned annual survival rates into hunting and non-hunting components as:

$$S_t^{i,j} = \theta_t^j \cdot \left( 1 - \frac{\left( \frac{f_t^{i,j}}{\lambda_t^j} \right)}{(1-c)} \right)$$

where:

$S$  = annual survival,

$\theta$  = survival from natural causes,

$f$  = band-recovery rate,

$\lambda$  = band-reporting rate,

$c$  = rate of crippling (unretrieved harvest),

$i$  = age (adult or young),

$j$  = sex,

$t$  = year, and

$\left( \frac{f_t^{i,j}}{\lambda_t^j} \right)$  = harvest rate.

The assumptions of this model are that  $\theta$  and  $\lambda$  vary by sex but not by age, and that crippling-loss rate is fixed and known. We relied on estimates of band-reporting rates provided by Nichols et al. (1995). We extended the proportional difference in male and female band-reporting rates observed in midcontinent North America eastward because Nichols et al. (1995) were unable to estimate band-reporting rates for females in eastern North America. We assumed that the crippling-loss rate was 0.2 (Anderson and Burnham 1976). This analysis was limited to the 1979-95 period because of uncertainty about band-reporting rates before and after this period.

*Reproductive rates.*--To estimate the age ratio of eastern mallards at the beginning of the hunting season, we first estimated the age ratio of the harvest from those Atlantic Flyway states that derived most of their mallard harvest from the eastern population (Munro and Kimball 1982). Those states were Maine, New Hampshire, Vermont, Massachusetts, Connecticut, Rhode Island, New York, Pennsylvania, and New Jersey. Harvest age ratios were estimated using wing samples from the Waterfowl Parts Collection Survey (Martin and Carney 1977). Each wing was weighted by the amount of harvest it represented and then weighted samples were pooled over all nine states. To estimate the fall population age ratio of mallards, we corrected annual harvest age ratios for the relative vulnerability of young and adults to harvest. To do this we relied on preseason banding of mallards in reference areas 8, 15, and 16, and calculated year-specific ratios of young:adult direct band-recovery rates from those states included in the wing sample. Estimates of fall age ratios were available for the 1961-99 period.

We initially considered four different sets of estimates of fall age ratios: (1) male age ratios, assuming annual variation in differential vulnerability; (2) male age ratios, assuming constant differential vulnerability; (3) female age ratios, assuming annual variation in differential vulnerability; and (4) female age ratios, assuming constant differential vulnerability.

*Final data sets.*--To determine which set of survival and reproductive rate estimates to use, we first estimated finite growth rates of eastern mallards from successive years in the breeding-population survey (Fig. 1). We then calculated the correlation coefficient between this set of growth rates and the various sets of survival and reproductive rates. Correlation coefficients were much higher for the partitioned survival rates, and for the age ratios that assumed constant differential vulnerability; therefore, only these sets were retained for further analyses. The correlation coefficient was marginally higher for female age ratios than for males, but we chose to use male age ratios in subsequent analyses because we were able to explain (model) significantly more of their annual variation than that for females ( $R^2 = 0.18$ ,  $P = 0.12$  for males;  $R^2 = 0.14$ ,  $P = 0.28$  for females).

### Synthetic Population Model

We relied on a standard balance (or difference) equation to predict changes in breeding-population size as a function of annual survival and reproductive rates. The balance equation had the form:

$$N_{t+1} = N_t \cdot \left( \left( p \cdot S_t^{am} \right) + \left( (1-p) \cdot S_t^{af} \right) + \left( p \cdot \left( A_t^m / d \right) \cdot S_t^{ym} \right) + \left( p \cdot \left( A_t^m / d \right) \cdot \psi \cdot S_t^{yf} \right) \right)$$

where:

$N$  = breeding-population size,

$p$  = proportion of males in the breeding population,

$S^{am}$ ,  $S^{af}$ ,  $S^{ym}$ , and  $S^{yf}$  = survival rates of adult males, adult females, young males, and young females, respectively,

$A^m$  = ratio of young males to adult males in the harvest,

$d$  = ratio of young male to adult male direct recovery rates,

$\psi$  = the ratio of male to female summer survival, and

$t$  = year,

In this model, we assume that  $p$ ,  $d$ , and  $\psi$  are fixed and known. The parameter  $\psi$  is necessary to account for the difference in anniversary date between the breeding-population survey (May) and the survival and reproductive rate estimates (August). This model also assumes that the sex ratio of fledged young is 1:1, hence the reason that  $A^m/d$  appears twice in the above equation. We estimated  $d = 1.043$  as the median ratio of young:adult male band-recovery rates in those states from which wing receipts were obtained. We estimated  $\psi = 1.216$  by regressing through the origin  $\theta_t^m$  against  $\theta_t^f$ , assuming that differences in natural mortality between males and

females occur principally in summer (Blohm et al. 1987). To estimate  $p$ , we used a population projection matrix of the form:

$$\begin{bmatrix} M_{t+1} \\ F_{t+1} \end{bmatrix} = \begin{bmatrix} M_t \\ F_t \end{bmatrix} \cdot \begin{bmatrix} S^{am} + (A^m/d) \cdot S^{ym} & 0 \\ (A^m/d) \cdot \psi \cdot S^{yf} & S^{af} \end{bmatrix}$$

where  $M$  and  $F$  are the relative number of males and females in the breeding populations, respectively. To parameterize the projection matrix we used average annual survival rates and age ratios, and estimates of  $d$  and  $\psi$  provided above. The right eigenvector of the projection matrix is the stable proportion of males and females the breeding population eventually would attain in the face of constant survival and reproductive rates. This eigenvector yielded an estimate of  $p = 0.544$ .

We next attempted to determine whether estimates of survival and reproductive rates were unbiased. We relied on the balance equation provided previously, except that we included new parameters to correct for any bias that might exist. Because we were unsure of the source(s) of potential bias, we alternatively assumed that any bias resided solely in survival rates:

$$N_{t+1} = N_t \cdot \Omega \cdot \left( \left( p \cdot S_t^{am} \right) + \left( (1-p) \cdot S_t^{af} \right) + \left( p \cdot (A_t^m/d) \cdot S_t^{ym} \right) + \left( p \cdot (A_t^m/d) \cdot \psi \cdot S_t^{yf} \right) \right)$$

(where  $\Omega$  is the bias-correction factor for survival rates), or solely in reproductive rates:

$$N_{t+1} = N_t \cdot \left( \left( p \cdot S_t^{am} \right) + \left( (1-p) \cdot S_t^{af} \right) + \left( p \cdot \alpha \cdot (A_t^m/d) \cdot S_t^{ym} \right) + \left( p \cdot \alpha \cdot (A_t^m/d) \cdot \psi \cdot S_t^{yf} \right) \right)$$

(where  $\alpha$  is the bias-correction factor for reproductive rates). We estimated  $\Omega$  and  $\alpha$  by determining the values of these parameters that minimized the sum of squared differences between observed and predicted population sizes. Based on this analysis,  $\Omega = 0.836$  and  $\alpha = 0.701$ , suggesting a positive bias in survival or reproductive rates.

### Survival and Reproduction Sub-models

*Survival.*—For purposes of AHM, annual survival rates must be predicted based on the specification of a regulatory alternative (and perhaps on other uncontrolled factors). We used a slightly modified version of the survival model presented previously, where annual survival for each age-sex class  $i,j$  under a given regulatory alternative is:

$$S_t^{i,j} = \bar{\theta}^j \cdot \left( 1 - \frac{(h_t^{am} \cdot v^{i,j})}{(1-c)} \right)$$

where:

$S$  = annual survival,

$\bar{\theta}^j$  = mean survival from natural causes,

$h^{am}$  = harvest rate of adult males, and

$v$  = harvest vulnerability relative to adult males,

$c$  = rate of crippling (unretrieved harvest).

In addition to the assumptions of the previous survival model, this model assumes that annual variation in survival is due solely to variation in harvest rates, that relative harvest vulnerability of the different age-sex classes is fixed and known, and that survival from natural causes is fixed at its sample mean. We estimated  $\bar{\theta}^j = 0.7307$  and  $0.5950$  for males and females, respectively (Fig. 2). We estimated  $v^{ij}$  by regressing through the origin estimated kill rates (i.e., harvest rates adjusted for crippling loss) of the different age-sex classes against those of adult males for the period 1979-95 (Fig. 3). We estimated  $v^{ij} = 1.153, 1.509,$  and  $1.331$  for adult females, young females, and young males respectively.

Finally, we related  $h^{am}$  to the current regulatory alternatives by assuming that for each alternative:

$$h_t^{am} \sim \text{Normal}(\mu, \sigma^2)$$

Based on previous analyses (U.S. Fish and Wildlife Service, *Adaptive Harvest Management: 2001 duck Hunting Season*, U.S. Dept. Interior, Washington, D.C., 47pp.; available online at <http://migratorybirds.fws.gov/mgmt/ahm/ahm-intro.htm>), we assumed that  $\mu = 0.080, 0.121, 0.135, 0.162,$  and  $0.177$  for the closed, very restrictive, restrictive, moderate, and liberal alternatives in the Atlantic Flyway, respectively. Typically, annual variability in harvest rates under the same regulatory alternative is approximately  $\sigma = \mu \cdot 0.2$  (Johnson et al. 1997). We assumed  $\sigma = \mu \cdot 0.3$  to account for additional variation in harvest rates due to uncontrolled changes in regulatory alternatives in the Mississippi, Central, and Pacific Flyways.

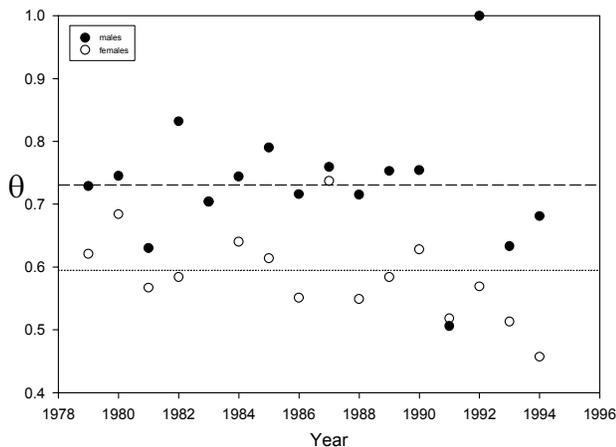


Fig. 2. Estimated survival from natural causes for male and female eastern mallards.

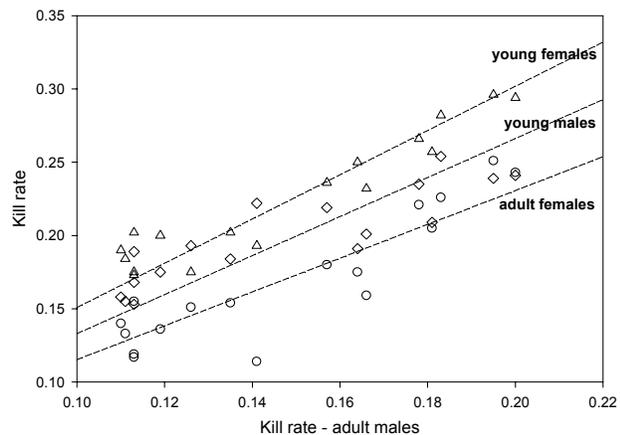


Fig. 3. Relationship of kill rates of adult males to those of the other age-sex classes in eastern mallards.

*Reproduction.*—As with survival, annual reproductive rates must be predicted in advance of regulations setting. We relied on the apparent relationship between breeding-population size and reproductive rates (Fig. 4). We used these data to fit the model:

$$R_t = a \cdot \exp(b \cdot N_t)$$

where  $R_t$  is the reproductive rate (i.e.,  $A_t^m/d$ ),  $N_t$  is breeding-population size in millions, and  $a$  and  $b$  are model parameters. The least-squares parameter estimates were  $a = 2.508$  and  $b = -0.875$  ( $P = 0.12, R^2 = 0.27$ ). Because of both the importance and uncertainty of the relationship between population size and reproduction, we specified two alternative models in which the slope ( $b$ ) was fixed at the least-squares estimate  $\pm$  one standard error, and in which the intercepts ( $a$ ) were subsequently re-estimated. This provided alternative hypotheses of strongly density- (or abundance-) dependent ( $a = 4.154, b = -1.377$ ) and weakly density-

dependent reproduction ( $a = 1.518, b = -0.373$ ).

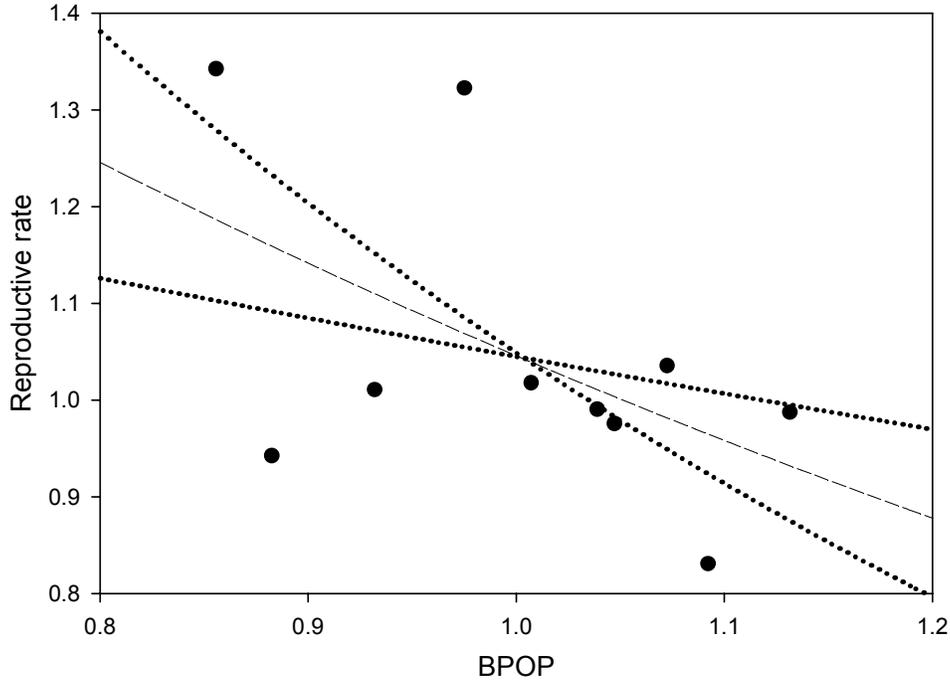


Fig. 4. Apparent relationship between reproductive rate (fall age ratio of males) and breeding-population size (BPOP, in millions) in eastern mallards. The dashed line represents the best fitting exponential model, and the dotted lines represent alternative models that were derived by fixing the slope at the least-squares estimate  $\pm$  one standard error, and then re-estimating the intercepts.

### Variance of Prediction Errors

Using the balance equations and sub-models provided above, predictions of breeding-population size in year  $t+1$  depend only on the specification of a regulatory alternative and on an estimate of population size in year  $t$ . For the period in which comparisons were possible (1991-96), we were interested in how well these predictions corresponded with observed population sizes. In making these comparisons, we were primarily concerned with how well the bias-corrected balance equations and reproductive and survival sub-models performed. Therefore, we relied on estimates of realized harvest rates of adult males rather than predictions as model inputs (also necessary because regulatory alternatives and associated predictions of harvest rates were not available for some years in the comparison).

We estimated a prediction-error variance by assuming that the differences between observed and bias-corrected predicted population sizes ( $e$ ) were distributed normally on a natural-log scale with mean  $\mu = 0$ , and variance  $\sigma^2$ :

$$e_t \sim \text{Lognormal}(0, \sigma^2)$$

The specification of a log-normal distribution for errors assumes errors are proportional to predicted population size. We then estimated the variance as:

$$\sigma^2 = \sum_t (\log(p_t) - \log(o_t))^2 / n$$

where  $p$  and  $o$  are predicted and observed population sizes in millions, respectively, and  $n = 6$ .

Variance estimates were similar regardless of whether we assumed that the bias was in reproductive rates or in survival, or whether we assumed that reproduction was strongly or weakly density-dependent. Thus, we averaged variance estimates to provide a final estimate of  $\sigma^2 = 0.006$ , which is equivalent to a coefficient of variation (*CV*) of 8.0%. We were concerned, however, about the small number of years available for estimating this variance. Therefore, we estimated an 80% confidence interval for  $\sigma^2$  as:

$$\frac{(n-1)\hat{\sigma}^2}{X_{\alpha}^2} < \sigma^2 < \frac{(n-1)\hat{\sigma}^2}{X_{1-\alpha}^2}$$

where  $\hat{\sigma}^2 = 0.006$ ,  $X^2$  is a chi-squared value for  $n-1$  degrees of freedom, and  $\alpha = 0.1$ . For the purposes of deriving optimal harvest strategies and for updating model weights, we used the upper 80% confidence limit for  $\sigma^2 = 0.018$  (i.e., *CV* = 14.5%) to express the additional uncertainty about the magnitude of prediction errors attributable to potentially important environmental effects not expressed by the models (Fig. 5).

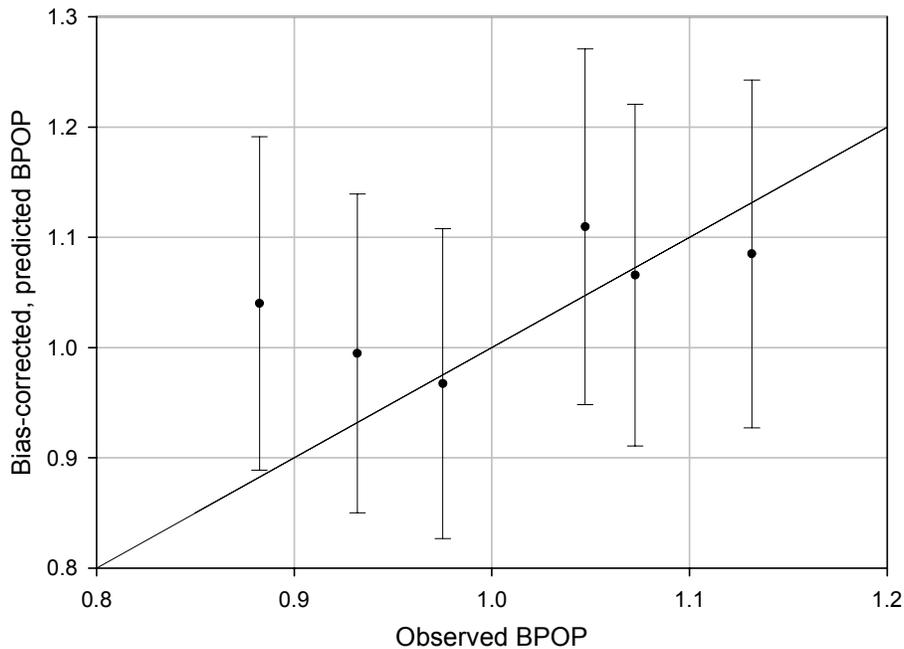


Fig. 5. Comparison of observed and bias-corrected predictions of breeding-population size in eastern mallards (BPOP, in millions) during 1991-96. Predictions were based on the balance equation and survival and reproductive rate sub-models explained in the text. In this particular case, the parameters of the reproductive sub-model are the best fitting, least-squares estimates. Error bars represent one standard deviation on either side of the prediction.

### Final Model Set and Management Implications

*Alternative models and associated harvest strategies.*--Based on the analyses described above, we specified a revised set of models for eastern mallards that includes six alternatives:

*Model 1:* no bias in survival or reproductive rates, and weakly density-dependent reproduction;

*Model 2:* no bias in survival or reproductive rates, and strongly density-dependent reproduction;

*Model 3:* bias-corrected survival, and weakly density-dependent reproduction;

*Model 4:* bias-corrected survival, and strongly density-dependent reproduction;

*Model 5:* bias-corrected reproductive rates, and weakly density-dependent reproduction; and

*Model 6:* bias-corrected reproductive rates, and strongly density-dependent reproduction.

We retained models with no bias-correction in the final model set because the time-series available for comparing observed and predicted population sizes was relatively short. However, these models, like all others, will contribute to the optimal harvest strategy only to the extent they are supported by future observations (see the section on empirical model weights below).

We derived optimal harvest strategies for each of the six models using stochastic dynamic programming (Lubow 1995), and by conditioning on the current set of regulatory alternatives and a management objective to maximize long-term cumulative harvest (*U.S. Fish and Wildlife Service, Adaptive Harvest Management: 2001 duck Hunting Season, U.S. Dept. Interior, Washington, D.C., 47pp.*; available online at <http://migratorybirds.fws.gov/mgmt/ahm/ahm-intro.htm>; Table 1). The variance of the prediction error was explicitly recognized in these optimizations.

Table 1. Model-specific and population-dependent (BPOP, in millions) optimal regulations for eastern mallards, conditioned on the current set of regulatory alternatives and an objective to maximize long-term cumulative harvest.

| <b>BPOP</b> | <b>Model 1</b> | <b>Model 2</b> | <b>Model 3</b> | <b>Model 4</b> | <b>Model 5</b> | <b>Model 6</b> |
|-------------|----------------|----------------|----------------|----------------|----------------|----------------|
| 0.1         | C              | C              | C              | C              | C              | C              |
| 0.2         | C              | L              | C              | L              | C              | VR             |
| 0.3         | C              | L              | C              | L              | C              | L              |
| 0.4         | C              | L              | C              | L              | C              | L              |
| 0.5         | C              | L              | C              | L              | C              | L              |
| 0.6         | C              | L              | C              | L              | C              | L              |
| 0.7         | C              | L              | C              | L              | C              | L              |
| 0.8         | C              | L              | C              | L              | C              | L              |
| 0.9         | C              | L              | C              | L              | C              | L              |
| 1.0         | C              | L              | VR             | L              | C              | L              |
| 1.1         | C              | L              | M              | L              | VR             | L              |
| 1.2         | R              | L              | L              | L              | R              | L              |
| 1.3         | M              | L              | L              | L              | L              | L              |
| ≥1.4        | L              | L              | L              | L              | L              | L              |

Comparison of model-specific strategies is complicated by differences among models in equilibrium population size. Each strategy is designed to hold population size as close as possible to the model-specific equilibrium point where growth rates and, thus, harvest are maximized. Nonetheless, it is apparent that model-specific strategies based on the hypothesis of weakly density-dependent reproduction appear to be considerably more conservative than those based on the hypothesis of strongly density-dependent reproduction. All harvest strategies are “knife-edged,” meaning that large differences in the optimal regulatory alternative can be precipitated by only small changes in breeding-population size. This result is largely due to the small

differences in predicted harvest rates among the current regulatory alternatives.

We determined the expected performance of each of six model-specific harvest strategies using Monte Carlo simulations (Table 2). As suspected, the bias-corrected models lead to considerably more conservative regulations than the bias-uncorrected models.

Table 2. Expected performance characteristics of optimal, model-specific harvest strategies for eastern mallards. The six alternative models are described in the body of the report. K represents expected average population size in the absence of harvest and  $N_e$  represents expected average population size (in millions) expected under the optimal harvest strategy.

| Model | K    | $N_e$ | Expected frequency of regulations: |      |      |      |       |
|-------|------|-------|------------------------------------|------|------|------|-------|
|       |      |       | C                                  | VR   | R    | M    | L     |
| 1     | 3.45 | 1.46  | 0.2                                | 0.0  | 10.6 | 17.8 | 71.4  |
| 2     | 1.67 | 1.11  | 0.0                                | 0.0  | 0.0  | 0.0  | 100.0 |
| 3     | 2.18 | 1.00  | 34.6                               | 34.6 | 0.0  | 21.0 | 9.8   |
| 4     | 1.33 | 0.85  | 0.0                                | 0.0  | 0.0  | 0.0  | 100.0 |
| 5     | 2.50 | 1.10  | 32.0                               | 39.2 | 19.8 | 0.0  | 9.0   |
| 6     | 1.41 | 0.84  | 0.0                                | 0.0  | 0.0  | 0.0  | 100.0 |

*Empirical model weights.*—Model weights were calculated as Bayesian probabilities, which reflect the cumulative (over time) ability of the individual alternative models to predict observed changes in population size. The Bayesian probability for each model is a function of the model’s previous (or *prior*) weight and the likelihood of the observed population size under that model. We used the procedure described by Hilborn and Walters (1992:503-504) and Williams et al. (1996) to calculate model weights from a comparison of predicted and observed population sizes for the years 1997-2001, inclusive, by starting with equal model weights in 1996. Predictions of breeding-population size in year  $t+1$  were based only on the expected harvest rate and the observed breeding population size in year  $t$ . The estimated likelihoods suggest that the model(s) best predicting observed population size varied among years (Table 3). Accordingly, there is no single model that is clearly favored over the others at the end of the time frame. When model weights from 2001 are combined in such a way as to reflect the probability of alternative hypotheses, the evidence for bias in survival or reproductive rates exceeds that for no bias by a ratio of 3:1. Current model weights also suggest that the evidence for strongly-density dependent reproduction exceeds that for weakly density-dependent reproduction by a ratio of 2:1. Of course, model weights may change with additional years of comparisons. Moreover, model weights provided in this report rely on model-based predictions of harvest rates because of uncertainty about current band-reporting rates. Once the planned reward-band study has been implemented, model weights can be updated based on estimates of realized harvest rates.

The optimal harvest strategy associated with the set of six models and their weights from 2001 suggests that the liberal regulatory alternative would be optimal for population sizes  $\geq 0.3$  million, with closed seasons otherwise. Under this scenario, population size would be expected to average about 0.9 million (SD = 0.16 million).

Table 3. Model-specific predictions of breeding-population size (BPOP-hat, in millions), the observed population size (BPOP), and the resulting likelihoods and Bayesian weights for each model of eastern mallard population dynamics. Model weights were assumed to be equal in 1996.

| Year | Parameter  | Model 1 | Model 2 | Model 3 | Model 4 | Model 5 | Model 6 | BPOP   |
|------|------------|---------|---------|---------|---------|---------|---------|--------|
| 1997 | BPOP-hat   | 1.2577  | 1.1791  | 1.0511  | 0.9854  | 1.0625  | 1.0074  | 1.0072 |
|      | Likelihood | 0.7702  | 1.4985  | 2.7974  | 2.9012  | 2.7198  | 2.9392  |        |
|      | Weight     | 0.0565  | 0.1100  | 0.2053  | 0.2129  | 0.1996  | 0.2157  |        |
|      |            |         |         |         |         |         |         |        |
| 1998 | BPOP-hat   | 1.0889  | 1.0867  | 0.9100  | 0.9082  | 0.9177  | 0.9162  | 1.0389 |
|      | Likelihood | 2.7684  | 2.7819  | 1.8255  | 1.7998  | 1.9359  | 1.9147  |        |
|      | Weight     | 0.0775  | 0.1515  | 0.1855  | 0.1897  | 0.1913  | 0.2045  |        |
|      |            |         |         |         |         |         |         |        |
| 1999 | BPOP-hat   | 1.1162  | 1.0958  | 0.9328  | 0.9158  | 0.9417  | 0.9275  | 1.0922 |
|      | Likelihood | 2.9018  | 2.9384  | 1.4964  | 1.2664  | 1.6189  | 1.4227  |        |
|      | Weight     | 0.1257  | 0.2489  | 0.1552  | 0.1344  | 0.1732  | 0.1627  |        |
|      |            |         |         |         |         |         |         |        |
| 2000 | BPOP-hat   | 1.1614  | 1.1102  | 0.9706  | 0.9278  | 0.9816  | 0.9457  | 0.8900 |
|      | Likelihood | 0.4299  | 0.7802  | 2.3969  | 2.8043  | 2.2656  | 2.6596  |        |
|      | Weight     | 0.0297  | 0.1066  | 0.2042  | 0.2068  | 0.2153  | 0.2374  |        |
|      |            |         |         |         |         |         |         |        |
| 2001 | BPOP-hat   | 0.9848  | 1.0485  | 0.8230  | 0.8762  | 0.8268  | 0.8714  | 1.0077 |
|      | Likelihood | 2.8973  | 2.8163  | 0.9662  | 1.7296  | 1.0153  | 1.6572  |        |
|      | Weight     | 0.0553  | 0.1932  | 0.1270  | 0.2303  | 0.1408  | 0.2533  |        |

## Future Work

*Estimating survival rates.*—Fitting standard waterfowl band-recovery models (Brownie et al. 1985) for eastern mallards has always been problematic (e.g., Chu and Hestbeck 1989). Even general models that allow for age, sex, and year dependencies in mortality rates typically provide poor fits to the data. For example, in our efforts, a model that allowed kill rates and survival to vary by banding-reference area, year, age, and sex did not adequately describe the structure of the data ( $P < 0.001$ , variance inflation factor [VIF] = 9.1; see Burnham and Anderson [1998] for an explanation of VIF). Simpler models for estimating survival and band-recovery rates (i.e., those do not rely on knowledge of band-reporting and crippling rates), performed somewhat better, however (e.g., VIF  $\approx$  2). We believe the poor fit could be due to poor specification of banding reference areas. Banding reference areas 8, 15, and 16 are relatively large, and were defined at a time when the amount of banding in the northeastern U.S. and eastern Canada was limited. We suggest a re-examination of the spatial patterns of banding and recoveries to define new reference areas for eastern mallards. J. Kelley (USFWS, pers commun.) has conducted some analyses that might be helpful in this endeavor.

*Sub-model development.*—The use of environmental covariates to predict annual variation in survival or

reproductive rates merits further investigation. In particular, we found evidence that several abnormally low age ratios during the 1990's were associated with widespread, long-lasting droughts in the northeastern United States. Failure to account for (i.e., model) these extreme events can produce relatively large shifts in model weights. In these cases, the shifts in model weights are not necessarily indicative of changes in the evidence for the alternative biological hypotheses codified in the models. We believe that such was the case when model weights shifted significantly between the years 1999 and 2000, probably due to a failure of the models to account for (i.e., predict) the effects of drought and other extreme weather events in the Northeast in 1999 (<http://www.nrcc.cornell.edu/top9.html>).

*Regulating harvest rates.*—Current predictions of regulation-specific harvest rates are based on observed harvest rates during 1979-84, adjusted for differences in season lengths and bag limits and for contemporary numbers of hunters. An implicit assumption in these predictions is that the historic relationship between hunting regulations in Canada and the U.S. will persist in the future. While hunting regulations for mallards in eastern Canada have been relatively stable over time, there is the potential that they could change independently of those prescribed in the AHM process for eastern mallards. Given that approximately 47% of the harvest of eastern mallards occurs in Canada, we believe that it is highly desirable to approach the Canadian Wildlife Service and Provincial governments about the development of a cooperative AHM strategy for eastern mallards that explicitly recognizes the regulatory processes in the two countries. We believe this idea has additional merit because of the biological and regulatory interaction of mallards and black ducks and the international effort to develop a black-duck AHM protocol (see below).

*Consideration of other species.*—The current AHM protocol for the Atlantic Flyway prescribes regulatory alternatives based solely on an objective to maximize long-term cumulative harvest of eastern mallards, yet other mallards (i.e., midcontinent) and other duck species are exposed to the same hunting regulations. Eastern mallards may have a higher harvest potential than at least some of these duck stocks, and vital rates of these stocks may not respond to annual changes in environmental conditions in the same way as eastern mallards. Therefore, we believe that it may be prudent to identify those stocks at risk and constrain hunting regulations when and where appropriate. As a first step, we recommend estimating the finite growth rates of other duck stocks in the Flyway and determining whether those rates appear to be affected by regulations (e.g., season length). If so, the objective to maximize eastern mallard harvest might be constrained by the desire to maintain growth rates  $\geq 1$  for those stocks. Of course, this approach would require agreement on the relative value of the competing objectives to maximize eastern mallard harvest and maintain population levels of other stocks.

The future of the AHM protocol for the Atlantic Flyway also will be affected by ongoing efforts to determine appropriate harvest strategies for black ducks (<http://fisher.forestry.uga.edu/blackduck/>) and wood ducks. Hunting regulations for eastern mallards and black ducks cannot be considered independently because of evidence that black duck reproductive rates are reduced in the presence of mallards. Thus, regulatory decisions for mallards can affect the status, and therefore ultimately the harvest, of black ducks. There also is an inherent regulatory dependency because both black ducks and mallards are exposed at least in part to a common hunting season. We are just beginning to consider the possible relationship between black duck and mallard AHM, but several difficulties are already apparent. These difficulties include differences between species in the spatial and temporal coverages of relevant data, and the need to develop a system for coordinating regulations between the U.S. and Canada. With respect to wood ducks, both the Atlantic and Mississippi Flyways are anxious to explore alternatives to conservative bag limits, at least in some situations. The effort is currently focused on biological modeling, but ultimately the two Flyways will have to decide whether they wish to set separate hunting seasons for wood ducks or else consider a joint optimization of regulations based on the status of both wood ducks and eastern mallards. Here again, the latter approach would require a decision about the relative value of mallard and wood duck harvests.

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