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UNCERTAINTY AND THE MANAGEMENT OF MALLARD HARVESTS

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Abstract: Those charged with regulating waterfowl harvests must cope with random environmental variations, incomplete control over harvest rates, and uncertainty about biological mechanisms operative in the population. Stochastic dynamic programming can be used effectively to account for these uncertainties if the probabilities associated with uncertain outcomes can be estimated. To use this approach managers must have clearly-stated objectives, a set of regulatory options, and a mathematical description of the managed system. We used dynamic programming to derive optimal harvest strategies for mallards (*Anas platyrhynchos*) in which we balanced the competing objectives of maximizing long-term cumulative harvest and achieving a specified population goal. Model-specific harvest strategies, which account for random variation in wetland conditions on the breeding grounds and for uncertainty about the relation between hunting regulations and harvest rates, are provided and compared. We also account for uncertainty in population dynamics with model probabilities, which express the relative confidence that alternative models adequately describe population responses to harvest and environmental conditions. Finally, we demonstrate how the harvest strategy thus derived can "evolve" as model probabilities are updated periodically using comparisons of model predictions and estimates of population size.

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Every decision-making situation involves objectives, alternative decision choices, and consequences of each choice relevant to attainment of the stated objectives. In the context of waterfowl harvest management, objectives typically involve the amount of harvest (or harvest opportunity) and the size of the population. Alternative decision choices consist of a range of hunting regulations from closed to liberal seasons, with the particular choice based on the anticipated effects on harvest and population status. Various survey programs provide the feedback necessary to gauge management performance.

Although this characterization is the basis of sound and informed management, there have been several problems in practice. The use of a large number of regulatory options has hindered assessment of their effects, considerable uncertainty exists about the effects of hunting regulations on harvest and population status, and harvest management objectives have not always been explicit or agreed-upon (Johnson et al. 1993). In this paper we demonstrate how managers can cope with some of these problems.

One of the most difficult aspects of regulating waterfowl harvests is the pervasiveness of uncertainty. Managers must account somehow for random environmental variation, incomplete control over harvest rates, and uncertainty about biological mechanisms operative in the population. Historically, managers have dealt with these uncertainties by developing their best deterministic predictions and then regulating harvest more conservatively than the analyses suggest (U.S. Fish and Wildl. Serv. 1988). Although

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risk-aversive from a resource perspective, this approach sacrifices hunting opportunity. Moreover, the decision-making process seldom has been treated as an opportunity to compare quantitative predictions with observations, resulting in a failure to learn.

These difficulties can be addressed by structuring decision-making as an adaptive process, where uncertainty is recognized explicitly and dealt with strategically (Walters 1986, Williams and Johnson 1995). A useful conceptual framework for sequential decision-making in the face of uncertainty is the theory and application of Markov decision processes, also referred to as stochastic control problems (Williams 1982, Anderson 1985, Puterman 1994). The solution of stochastic control problems involves identifying that set of state-specific decisions (called a strategy) that is optimal with respect to explicit criteria. To derive optimal harvest strategies for waterfowl, the manager must have: (1) an objective function describing the goals of management; (2) a set of regulatory options; and (3) a mathematical description of the managed system, including important stochastic elements (Johnson et al. 1993, Puterman 1994).

A useful solution algorithm for the stochastic control problem is discrete stochastic dynamic programming (Bellman 1957, Anderson 1985, Puterman 1994, Lubow 1995). Dynamic programming involves working backward from some future time step and calculating both the short- and long-term rewards (as defined by the management objectives) associated with each possible decision and system state. Uncertainty is accounted for by calculating the expected rewards associated with uncertain outcomes, based on their specified probabilities of occurrence. An optimal harvest strategy is then that set of state-specific decisions that maximizes the expected sum of rewards over a specified time frame. Dynamic programming is appropriate for linear or nonlinear systems, permits complex objective functions, and can be used to incorporate realistic constraints on the exploitation process (Anderson 1985, Williams 1989).

Anderson (1975*a*) first used dynamic programming to account for environmental variation in computing optimal mallard harvests. We demonstrate the use of dynamic programming for deriving mallard harvest strategies where managers are faced with additional sources of uncertainty, including incomplete control over harvests and uncertainty regarding population dynamics. The management objective, set of regulatory options, models of system dynamics, and other parameters used herein are intended primarily for the purpose of illustration. While we believe our conceptual approach has considerable merit for improving the regulation of mallard harvests, the formulation actually used to guide management decisions may differ from that presented here.

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SYSTEM DYNAMICS

A general formulation allows for a multidimensional resource system, where $\mathbf{X}_t$ represents the state of the system that includes both population size and indicators of habitat conditions. System transitions are given by

$$\mathbf{X}_{t+1} = \mathbf{X}_t + F_i(\mathbf{X}_t, H_t, z_t),$$

where H_t represents time-dependent harvests, z_t represents time-dependent environmental effects, and the index i specifies one of a number of models that is used to represent population and habitat dynamics.

Structural Uncertainty

Structural uncertainty refers to incomplete knowledge of biological mechanisms operative in the population (Williams et al. 1996), which we expressed as a set of alternative models of population dynamics. We let $X1_t$ and $X2_t$ represent the number of adult mallards present in the mid-continent survey area and the number of ponds in Prairie Canada in May of each calendar year t, respectively (Can. Wildl. Serv. and U.S. Fish and Wildl. Serv. 1994). We further defined $y_{i,t,s}$ as the model-specific number of young of sex s in the fall population, $\phi_{i,t,s}$ as the model and sex-specific survival rate of adults from May of year t through April of year t + 1, and $\phi'_{i,t,s}$ as the model and sex-specific survival rate of young from September of year t through

April of year $t + 1$. Thus, the model-specific transition for population size was

$$X1_{t+1,s} = X1_{t,s}\,\phi_{i,t,s} + y_{i,t,s}\,\phi'_{i,t,s}, \quad (1)$$

where

$$y_{i,t,s} = g_i(X1_{t,s}, X2_t), \quad (2)$$

and where g_i was a model-specific recruitment function (eqs 9–11).

Survival.—We considered annual survival of mallards to be the product of survival from hunting and survival from natural mortality factors outside the hunting season. Model and sex-specific survival of adults and young were defined as

$$\phi_{i,t,s} = \alpha_s \beta_{i,t,s} \gamma \quad (3)$$

and

$$\phi'_{i,t,s} = \beta'_{i,t,s} \gamma, \quad (4)$$

respectively, where α_s is a sex-specific summer survival rate, $\beta_{i,t,s}$ and $\beta'_{i,t,s}$ are model and sex-specific hunting-season survival rates for adults and young, respectively, and γ is winter survival rate.

We estimated nonhunting-season survival rates using the methods of Smith and Reynolds (1992) and assuming a crippling loss of $c = 0.2$ (Anderson and Burnham 1976) and a band-reporting rate of $\lambda = 0.32$ (Nichols et al. 1991). We used the same data as Smith and Reynolds (1992), but fit a reduced model in which there was no geographic variation in natural mortality rates (likelihood ratio test $\chi^2 = 21.0$, 16 df, $P = 0.2$). Estimates of annual survival from natural causes were 0.81 (SE = 0.02) for males and 0.64 (SE = 0.01) for females.

These survival rates were partitioned into winter and summer components with results from the Stabilized Regulations Study (Off. Migr. Bird Manage., U.S. Fish and Wildl. Serv., Washington, D.C., unpubl. data) and checked against other literature (Cowardin and Johnson 1979, Reineke et al. 1987, Dugger et al. 1994) to ensure reasonable estimates. For males, $\alpha_s = 0.90$, and for females, $\alpha_s = 0.71$. Winter survival for both sexes was $\gamma = 0.90$. We had no empirical evidence to specify age or year-dependency in rates of survival from natural causes.

One of the most important sources of uncertainty in waterfowl harvest management is the effect of exploitation on survival (Anderson and Burnham 1976, Johnson et al. 1993). To characterize this uncertainty we considered 2 alternative forms of survival during the hunting season. In the case of additive hunting mortality (i = A), sex-specific survival of adults and young during the hunting season declined linearly with increases in harvest rate:

$$\beta_{i,t,s} = 1 - \frac{h_{t,s}}{1 - c} \quad (5)$$

and

$$\beta'_{i,t,s} = 1 - \frac{h'_{t,s}}{1 - c}, \quad (6)$$

where $h_{t,s}$ and $h'_{t,s}$ are sex-specific harvest rates of adults and young, respectively. Inclusion of crippling loss (c) accounts for birds that are killed by hunters but not retrieved.

In the second form (i = C), we allowed complete compensation for hunting mortality up to a threshold kill rate (defined as the annual rate of nonhunting mortality) so that annual survival was constant for kill rates below this threshold. Beyond this threshold, survival declined linearly with increases in harvest rate:

$$\beta_{i,t,s} = \begin{cases} 1.0, & \text{if } \dfrac{h_{t,s}}{1 - c} \le 1 - \alpha_s\gamma; \\[2ex] \dfrac{1 - \dfrac{h_{t,s}}{1 - c}}{\alpha_s\gamma}, & \text{if } \dfrac{h_{t,s}}{1 - c} > 1 - \alpha_s\gamma; \end{cases} \quad (7)$$

and

$$\beta'_{i,t,s} = \begin{cases} 1.0, & \text{if } \dfrac{h'_{t,s}}{1 - c} \le 1 - \alpha_s\gamma; \\[2ex] \dfrac{1 - \dfrac{h'_{t,s}}{1 - c}}{\alpha_s\gamma}, & \text{if } \dfrac{h'_{t,s}}{1 - c} > 1 - \alpha_s\gamma. \end{cases} \quad (8)$$

For both the additive and compensatory models we assumed that natural mortality during the hunting season was negligible (Cowardin and Johnson 1979, Reineke et al. 1987).

Recruitment.—We first estimated annual fall age ratios of female mallards originating from the region of North America surveyed in spring 1961–93 (Can. Wildl. Serv. and U.S. Fish and Wildl. Serv. 1994). We defined A_t as the ratio of young females to adult females in the preharvest population, which is estimated from the age ratio of the harvest corrected for relative harvest vulnerability (young:ad ratio of direct

recovery rates of banded F), in year t (Martin et al. 1979). Age ratios of the harvest were calculated from parts-collection surveys (Martin and Carney 1977) in those portions of the Central Flyway and Mississippi Flyway that derive ≥80% of their harvest from the mallard population of interest (Munro and Kimball 1982). The harvest vulnerability of young relative to adults was estimated for each of 8 banding reference areas (Anderson and Henny 1972) within the breeding range, then averaged for a single estimate of relative vulnerability for each year, with estimates of population size within reference areas as weights.

We next developed models that described A_t as a linear function of mallard population size ($X1_t$) and the number of ponds ($X2_t$). We permitted interaction between $X1$ and $X2$ and allowed the linear relation to vary between 2 unspecified "epochs" within the period 1961–93. We used weighted least squares and inversely weighted A_t by the variance of the annual harvest age ratio, which we believed to be proportional to variability in A_t. We fit all possible regression models induced by interactive combinations of $X1$, $X2$, and epoch, with epoch boundary (i.e., the first year of the second epoch) adjusted to each year between 1965 and 1990 (14 models/epoch partition × 26 epoch partitions + 5 "no epoch" models = 369 models). We selected the model with lowest Akaike Information Criterion (AIC) (Akaike 1974, Burnham and Anderson 1992) and checked model residuals for conformity with least squares regression assumptions (Draper and Smith 1981).

The model with the smallest of the 369 AIC values contained effects $X1$ and $X2$, and no interaction. The model also distinguished an epoch boundary at 1970, where the regression coefficient for $X1$ changed from a pre-1970 value of 0.0874×10^{-6} (SE = 0.0622×10^{-6}) to -0.0547×10^{-6} (SE = 0.0225×10^{-6}) afterward. We took that portion of the model corresponding to the most recent epoch as a weakly density-dependent model (i = W) of recruitment:

$$A_{t,t} = 0.8249 - 0.0547 \times 10^{-6} X1_t + 0.1130 \times 10^{-6} X2_t. \quad (9)$$

To express the uncertainty about the degree of density-dependence in recruitment (i.e., the magnitude of the coefficient for $X1$), we also considered the minimum parameter estimate for the coefficient of $X1$ located on the 95% confidence ellipsoid for all the parameters (Draper and Smith 1981). We chose the minimum, rather than the maximum, estimate as the most reasonable alternative to the mean based on descriptions of the mechanisms likely responsible for density-dependent recruitment (Dzubin 1969). This strongly density-dependent model (i = S) of recruitment was

$$A_{t,t} = 1.1081 - 0.1128 \times 10^{-6} X1_t + 0.1460 \times 10^{-6} X2_t. \quad (10)$$

The number of young females in the fall population ($y_{t,t,\varsigma}$) was the product of the predicted age ratio and the number of adult females in the fall. The number of adult females in the fall was the product of summer survival and the number of females in the spring, which was determined by assuming a constant sex ratio of 1.2 males per female (Anderson 1975*b*). To determine the number of young males we assumed a sex ratio of 1.0 for young birds in the fall (Bellrose et al. 1961, Hestbeck et al. 1989). Thus:

$$y_{t,t,\varsigma} = y_{t,t,\delta} = g_t(X1_{t,s}, X2_t) = A_{t,t}\alpha_\varsigma X1_t/2.2 \quad (11)$$

The combination of the 2 recruitment (R) hypotheses and the 2 survival (S) hypotheses resulted in 4 alternative models of mallard population dynamics: (1) additive hunting mortality and weakly density-dependent recruitment (S_AR_W); (2) additive hunting mortality and strongly density-dependent recruitment (S_AR_S); (3) compensatory hunting mortality and weakly density-dependent recruitment (S_CR_W); and (4) compensatory hunting mortality and strongly density-dependent recruitment (S_CR_S).

Environmental Variation

The number of wetland basins containing surface water (ponds) in the Prairie Pothole Region during the breeding season is an important determinant of mallard production (Pospahala et al. 1974). Since 1961, the number of ponds in Prairie Canada during May has varied from 1.443 million (SE = 0.075 million) in 1981 to 6.390 million (SE = 0.308 million) in 1974 (Can. Wildl. Serv. and U.S. Fish and Wildl. Serv. 1994). Managers facing a harvest management decision cannot know with certainty what pond numbers (and thus what mallard production) to expect in the future. However, it is possible to make probabilistic statements about temporal changes in pond abundance, which in turn allow

managers to assess expected future harvests associated with current regulatory decisions.

We used the estimated number of ponds in Prairie Canada and records of monthly (1 Jun 1974–31 May 1992) precipitation (mm) from 5 weather stations in southern Alberta, Saskatchewan, and Manitoba to construct an autoregressive model of pond abundance:

$$X2_{t+1} = -3835087.53 + 0.45X2_t + 13695.47r_t, \quad (12)$$

where r_t was total precipitation during the 12-month period from time t to $t + 1$ ($F_{2,14} = 5.26$, $P = 0.02$; $X2_t$: $t = 2.60$, $P = 0.02$; r_t: $t = 2.44$, $P = 0.03$; and $R^2 = 0.43$). Pond numbers predicted by this model are nearly identical to those of the model provided by Pospahala et al. (1974), despite our use of different time periods and sources of precipitation data.

We next examined annual (1 Jan–31 Dec) precipitation records for the period 1942–91 from the same 5 weather stations described previously. We could not reject the hypothesis that annual precipitation was distributed normally (Shapiro-Wilk $W = 0.97$, $P = 0.36$, range 304–574 mm, $\bar{x} = 418$ mm, SD = 56 mm). Because these results were virtually identical to those reported by Pospahala et al. (1974), we did not repeat their extensive time-series analyses to determine if long-term patterns or cycles exist in precipitation. However, preliminary analyses with several datasets (some of >100 yr) failed to provide strong evidence of precipitation cycles (J. R. Sauer, Natl. Biol. Serv., pers. commun.). Therefore, we accepted the conclusion of Pospahala et al. (1974) that annual precipitation in Prairie Canada can be described adequately as a normally-distributed, independent random variable.

We made the normal distribution of precipitation discrete by using 5 intervals of equal probability between quantiles corresponding to the 0.001 and 0.999 probability values. We assigned a conditional probability of 0.2 to each value of rainfall that divided an interval into 2 sub-intervals of equal probability. Random draws of the 5 values of r_t provided stochasticity in pond abundance (eq 12).

Partial Management Control

Managers control hunting regulations rather than harvest rates directly, and accounting for uncertainty in the functional relation between the 2 is important. Using preseason band-recovery data, we estimated the harvest rates of mid-continent mallards that might be realized under each of 3 regulatory options. These options, which we characterized as liberal, moderate, and restrictive, were those regulations in effect during 1979–84, 1985–87, and 1988–93, respectively. Each regulatory option contained flyway-specific season lengths and bag limits (Appendix A).

We relied on direct recovery rates of mallards banded before the hunting season in a representative portion of the mid-continent region (banding reference areas 3–5, Anderson and Henny 1972). We first estimated the mean ($f_{AM,p}$) and variance ($s^2_{total}[f_{AM,p}]$) of direct recovery rates for adult male mallards for each of the 3 time periods (p). We focused on adult males because they generally had the largest banded-sample sizes.

The variance of the direct recovery rate is composed of both temporal and sampling components. For the purpose of choosing hunting regulations, we were interested primarily in the temporal component, which is a measure of the variability in recovery rates that could be expected when using the same regulations in different years (i.e., partial controllability). We estimated this temporal variation ($s^2_{temp}[f_{AM,p}]$) using the approach suggested by Burnham et al. (1987:260–269):

$$s^2_{temp}(f_{AM,p}) = s^2_{total}(f_{AM,p}) - \frac{\sum_t^{n_p} s^2(f_{AM,p,t})}{n_p}, \quad (13)$$

where n_p is the number of years in period p and $s^2(f_{AM,p,t})$ is the estimated binomial variance. We then estimated period-specific harvest rates ($h_{AM,p}$) and their temporal variances ($s^2_{temp}[h_{AM,p}]$) using a constant band-reporting rate of $\lambda = 0.32$ (Nichols et al. 1991):

$$h_{AM,p} = \frac{f_{AM,p}}{\lambda}, \quad (14)$$

and

$$s^2_{temp}(h_{AM,p}) = \frac{s^2_{temp}(f_{AM,p})}{\lambda^2}. \quad (15)$$

Based upon this analysis, we assigned mean harvest rates of adult males to 0.090 ($s_{temp}[h_{AM,p}] = 0.016$) for the restrictive option, 0.120 ($s_{temp}[h_{AM,p}] = 0.022$) for the moderate option, and 0.156 ($s_{temp}[h_{AM,p}] = 0.025$) for the liberal option. We also considered explicitly a closed season, in

which the mean harvest rate was assumed to be 0 with no variation.

We next estimated the vulnerability to harvest for each of the other age-sex cohorts (ad F, young M, young F) relative to that of adult males. The mean relative vulnerability $w_{a,p}$ of each cohort a during each period p was calculated by averaging the ratio of annual recovery rates within the specified period:

$$w_{a,p} = \frac{\sum_{t}^{n_p} \frac{f_{a,p,t}}{f_{AM,p,t}}}{n_p}. \quad (16)$$

Mean relative vulnerabilities did not differ among periods (asymptotic normal test of general contrast; Sauer and Williams 1989) for adult females ($\chi^2 = 3.68$, 2 df, $P = 0.16$), immature males ($\chi^2 = 1.65$, 2 df, $P = 0.44$), or immature females ($\chi^2 = 0.40$, 2 df, $P = 0.82$). Thus, we used constant rates of differential vulnerability of 0.480 for adult females, 1.310 for young males, and 0.868 for young females.

Based on the period-specific means and variances from equations 14 and 15, we assigned a 2-parameter gamma distribution of adult-male harvest rates to each regulatory option (Fig. 1). We made the probability distributions discrete in the same manner as that described for rainfall. We obtained distributions of harvest rates for the other age-sex cohorts by multiplying the constant rate of differential vulnerability for each cohort by the set of discretized adult-male harvest rates.

THE OPTIMAL STOCHASTIC CONTROL PROBLEM

The Objective Function

As a result of implementing a regulatory strategy for sport hunting, rewards of mallard harvest are received at each decision point. The goal of the manager is to determine the regulatory strategy that will provide as large a temporal sum of rewards as possible over an extended time-frame. The immediate harvest reward v at time t for state $\mathbf{X}_t$ and regulatory decision d_t is:

$$v(\mathbf{X}_t, d_t) = \sum_{h_t} p(h_t \mid d_t)\left(\sum_{i} l_i u_{i,t} H_{i,t}\right), \quad (17)$$

where

$$u_{i,t} = u(E[X1_{i,t+1}]) = u(\mathbf{X}_t, H_{i,t}), \quad (18)$$

$$H_{i,t} = H_i(\mathbf{X}_t, h_t), \quad (19)$$

and $p(h_t \mid d_t)$ is the probability of a specific harvest rate conditioned on the regulatory decision, and l_i is the probability that model i is appropriate for the population ($\Sigma\, l_i = 1.0$). The model-specific relative value of a unit of harvest ($u_{i,t}$) is a function of expected population size in year $t + 1$, which in turn depends on population size $X1_t$, habitat conditions $X2_t$, and harvest $H_{i,t}$ at time t (eq 18; specific formulation in eq 24). Harvests are a model-specific function of system state $\mathbf{X}_t$ and harvest rate h_t (eq 19).

The expected value $V(R_t \mid \mathbf{X}_t)$ of total harvest over the time frame $\tau = t, t + 1, \ldots, T$ is conditional on system state $\mathbf{X}_t$ at time t, with R_t being a strategy of time-specific and state-specific harvest decisions:

$$\begin{aligned} V(R_t \mid \mathbf{X}_t) &= E\left[\sum_{\tau=t}^{T} u_{i,\tau} H_{i,\tau} \mid \mathbf{X}_t\right] \\ &= E\left[u_{i,t} H_{i,t} \mid \mathbf{X}_t \right. \\ &\qquad \left. + \sum_{\tau=t+1}^{T} u_{i,\tau} H_{i,\tau} \mid \mathbf{X}_t\right] \end{aligned} \quad (20)$$

where the expectation accounts for random harvest rates, stochastic environment effects, and model uncertainty in transitions from $\mathbf{X}_\tau$ to $\mathbf{X}_{\tau+1}$ over the entire time frame. Because $V(R_t \mid \mathbf{X}_t)$ is a sum of the expected immediate harvest value and expected future harvest value, then equation 20 can be written as

$$\begin{aligned} &V(R_t \mid \mathbf{X}_t) \\ &= E(u_{i,t} H_{i,t} \mid \mathbf{X}_t) + E\left[\sum_{\tau=t+1}^{T} u_{i,\tau} H_{i,\tau} \mid \mathbf{X}_t\right] \\ &= \sum_{h_t} p(h_t \mid d_t)\left(\sum_{i} l_i u_{i,t} H_{i,t} \mid \mathbf{X}_t\right) \\ &\quad + E_{h_t \mid d_t}\left[E_{X_{t+1} \mid h_t}\left[\sum_{i} l_i \left[E\left(\sum_{\tau=t+1}^{T} u_{i,\tau} H_{i,\tau} \mid \mathbf{X}_{t+1}\right)\right]\right]\right] \\ &= v(\mathbf{X}_t, d_t) + E_t[V(R_{t+1} \mid \mathbf{X}_{t+1})] \end{aligned} \quad (21)$$

where the last expectation is with respect to random variation in harvest rates at time t and environment and model variation in the transition from t to $t + 1$. Because by definition the optimal strategy satisfies

$$V^*(R^*_t \mid \mathbf{X}_t) = \max_{R_t}\left(E\left[\sum_{\tau=t}^{T} u_{i,\tau} H_{i,\tau} \mid \mathbf{X}_t\right]\right), \quad (22)$$

the appropriate computing algorithm is

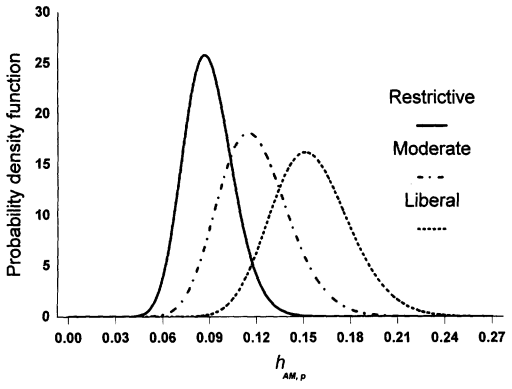

Fig. 1. Probability density function of harvest rate of adult male mallards ($h_{AM,p}$) for 3 regulatory periods.

$$V^*(R^*_t \mid \mathbf{X}_t) = \max_{d_t} (v[\mathbf{X}_t, d_t] + E_t[V^*(R^*_{t+1} \mid \mathbf{X}_{t+1})]). \quad (23)$$

After some finite number of iterations, continued application of equation 23 typically produces the same state-specific decisions, and the decision structure becomes time-independent. This set of state-specific decisions are then applied in an infinite time horizon.

Balancing Competing Objectives

At least part of the difficulty in waterfowl harvest regulation has been the lack of well-focused, unambiguous management objectives. Many managers fail to recognize that an objective to provide "sustainable hunting opportunity" is not in itself sufficient for defining a unique harvest strategy. There are always multiple harvest strategies that will fulfill the basic goal of resource conservation, but that will produce different temporal averages and variances of harvest rates, harvest, population size, and other parameters of management interest.

We here focus on 2 measures of performance typically of interest to managers of waterfowl harvest: long-term cumulative harvest and population size. The North American Waterfowl Management Plan (Plan) (U.S. Dep. Inter. and Environ. Can. 1986) identifies numerical goals for the size of most waterfowl populations. These goals were established to ensure satisfactory levels of hunting opportunity, but also were chosen for ecological and non-consumptive purposes. Although the Plan is intended primarily to guide habitat conservation efforts, the population goals have been endorsed by the federal governments of Canada, the United States, and Mexico.

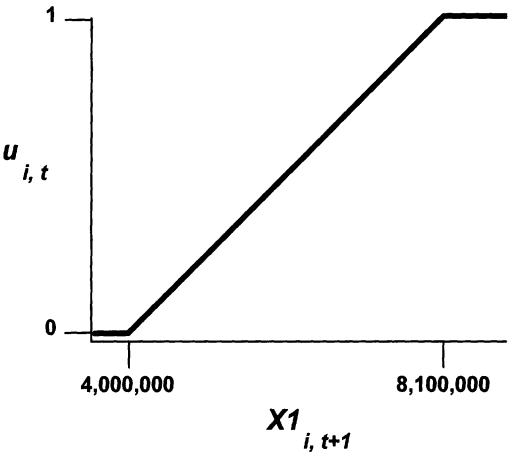

Fig. 2. The relative value of a unit of harvest ($u_{i,t}$) at time t as a function of population size ($X1_{i,t+1}$) expected at time t + 1. Model-specificity is denoted by the subscript i.

We used a harvest-management objective that balances the desire to maximize long-term cumulative harvest with the desire to maintain mallard population levels at or above the Plan goal of 8.1 million. Thus, the relative value of harvest is highest when the Plan goal is met, but declines linearly as the population falls short of the goal (Fig. 2). In developing this management objective, we considered it appropriate to devalue harvest completely if the mallard population in the continental survey area were expected to fall below 4 million. The utility function expressing the model-specific relative value of harvest $u_{i,t}$ had the form (eq 18):

$$u_{i,t} = \begin{cases} 1.0, & \text{if } E(X1_{i,t+1}) \geq 8{,}100{,}000; \\ 0.0, & \text{if } E(X1_{i,t+1}) \leq 4{,}000{,}000; \\ \dfrac{[E(X1_{i,t+1}) - 4{,}000{,}000]}{(8{,}100{,}000 - 4{,}000{,}000)}, & \text{if } 4{,}000{,}000 < E(X1_{i,t+1}) < 8{,}100{,}000. \end{cases} \quad (24)$$

This utility function represents a compromise over the range of population sizes of 4–8.1 million, in that neither the objective to maximize harvest nor the objective to maintain the mallard population at or above the Plan goal would be realized fully.

When the population at time t ($X1_t$) is below the goal of 8.1 million, the capacity of available breeding habitat ($X2_t$) to promote population growth during the interval t to t + 1 is consid-

Table 4. Optimal regulatory choices[a] for mallards, conditioned on the dual objectives of maximum long-term harvest and achieving a population goal of 8.1 million, assuming additive hunting mortality and weakly density-dependent recruitment.

	X2[b]												
X1[c]	1.0	1.5	2.0	2.5	3.0	3.5	4.0	4.5	5.0	5.5	6.0	6.5	7.0
2.0	C	C	C	C	C	C	C	C	C	C	C	C	C
2.5	C	C	C	C	C	C	C	C	C	C	C	C	C
3.0	C	C	C	C	C	C	C	C	C	C	C	C	C
3.5	C	C	C	C	C	C	C	C	C	C	C	C	C
4.0	C	C	C	C	C	C	C	C	C	C	C	C	C
4.5	C	C	C	C	C	C	C	C	C	C	C	C	C
5.0	C	C	C	C	C	C	C	C	C	C	C	C	C
5.5	C	C	C	C	C	C	C	C	C	C	C	C	C
6.0	C	C	C	C	C	C	C	C	C	C	C	R	R
6.5	C	C	C	C	C	C	C	C	R	R	R	R	M
7.0	C	C	C	C	C	C	R	R	R	M	M	M	L
7.5	C	C	C	R	R	R	R	M	M	L	L	L	L
8.0	C	R	R	R	R	M	M	L	L	L	L	L	L
8.5	R	R	R	M	M	L	L	L	L	L	L	L	L
9.0	R	R	M	L	L	L	L	L	L	L	L	L	L
9.5	M	M	L	L	L	L	L	L	L	L	L	L	L
10.0	M	L	L	L	L	L	L	L	L	L	L	L	L
10.5	L	L	L	L	L	L	L	L	L	L	L	L	L
11.0	L	L	L	L	L	L	L	L	L	L	L	L	L
11.5	L	L	L	L	L	L	L	L	L	L	L	L	L
12.0	L	L	L	L	L	L	L	L	L	L	L	L	L

[a] C = closed, R = restrictive, M = moderate, and L = liberal.
[b] *X2* = no. of ponds in May in Prairie Canada, in millions.
[c] *X1* = no. of mallards in the population, in millions.

ered in the determination of the optimal regulatory decision for $\mathbf{X}_t$. In other words, liberal hunting regulations could still be appropriate for a mallard population that is below goal, if current habitat conditions were expected to result in good production of young.

OPTIMALITY ANALYSES

We used the software package SDP (Lubow 1994, 1995) to implement system dynamics (eqs 1–12) and the objective function (eq 21) to derive optimal harvest strategies. We calculated optimal regulatory decisions for population sizes of 2–12 million and for pond numbers of 1–7 million, both in increments of 0.5 million. Successive backward iterations were performed until state-specific decisions appeared to be time-independent (i.e., stationary). Although use of dynamic programming cannot guarantee that a time-independent strategy has been found (Puterman 1994), an optimal strategy that remains unchanged for several iterations is likely to be stationary (Lubow 1994).

In the first analysis, we derived optimal harvest strategies for each of the 4 mallard models described above (Tables 1–4). We accomplished this by assigning a probability of $l_t = 1$ to the model of interest and probabilities of $l_t = 0$ to the remaining models. In all cases, we accounted explicitly for environmental variation (i.e., random variation in rainfall) and for partial control of harvest rates. If the appropriate model were known, a manager would simply choose the model-specific regulatory option corresponding to the estimated size of the mallard population (*X1*) and the number of ponds (*X2*) in year *t*.

As expected, the optimal regulatory choice can be more liberal with higher numbers of mallards and ponds, regardless of the specific model that is used. However, there is considerable difference in the harvest strategies among the 4 models. The model with additive hunting mortality and weakly density-dependent recruitment (S_AR_W) leads to the most conservative harvest strategy, whereas the model with compensatory hunting mortality and strongly density-dependent recruitment leads to the most liberal strategy (S_CR_S). The other 2 models (S_AR_S and S_CR_W) lead to strategies that are intermediate between these extremes.

The existence of 4 different model-specific harvest strategies emphasizes the need to account for uncertainty about which model best describes mallard population dynamics. Thus, a harvest strategy assuming complete uncertainty about the most appropriate model was derived

Table 2. Optimal regulatory choices[a] for mallards, conditioned on the dual objectives of maximum long-term harvest and achieving a population goal of 8.1 million, assuming additive hunting mortality and strongly density-dependent recruitment.

	X2[b]												
X1[c]	1.0	1.5	2.0	2.5	3.0	3.5	4.0	4.5	5.0	5.5	6.0	6.5	7.0
2.0	C	C	C	C	C	C	C	C	C	C	C	C	C
2.5	C	C	C	C	C	C	C	C	C	C	C	C	C
3.0	C	C	C	C	C	C	C	C	C	C	C	C	C
3.5	C	C	C	C	C	C	C	C	C	C	C	C	C
4.0	C	C	C	C	C	C	C	C	C	C	C	C	C
4.5	C	C	C	C	C	C	C	C	C	C	C	C	C
5.0	C	C	C	C	C	C	C	C	C	C	R	R	R
5.5	C	C	C	C	C	C	R	R	R	R	R	R	M
6.0	C	C	C	C	R	R	R	R	R	M	M	M	L
6.5	C	R	R	R	R	R	R	M	M	M	L	L	L
7.0	R	R	R	R	R	M	M	M	L	L	L	L	L
7.5	R	R	R	R	M	M	L	L	L	L	L	L	L
8.0	R	R	M	M	M	L	L	L	L	L	L	L	L
8.5	R	M	M	M	L	L	L	L	L	L	L	L	L
9.0	R	M	M	L	L	L	L	L	L	L	L	L	L
9.5	M	M	L	L	L	L	L	L	L	L	L	L	L
10.0	M	M	L	L	L	L	L	L	L	L	L	L	L
10.5	M	L	L	L	L	L	L	L	L	L	L	L	L
11.0	M	L	L	L	L	L	L	L	L	L	L	L	L
11.5	L	L	L	L	L	L	L	L	L	L	L	L	L
12.0	L	L	L	L	L	L	L	L	L	L	L	L	L

[a] C = closed, R = restrictive, M = moderate, and L = liberal.
[b] *X2* = no. of ponds in May in Prairie Canada, in millions.
[c] *X1* = no. of mallards in the population, in millions.

Table 3. Optimal regulatory choices[a] for mallards, conditioned on the dual objectives of maximum long-term harvest and achieving a population goal of 8.1 million, assuming compensatory hunting mortality and weakly density-dependent recruitment.

	X2[b]												
X1[c]	1.0	1.5	2.0	2.5	3.0	3.5	4.0	4.5	5.0	5.5	6.0	6.5	7.0
2.0	R	R	R	R	R	R	R	R	R	R	R	R	R
2.5	R	R	R	R	R	R	R	R	R	R	R	R	R
3.0	R	R	R	R	R	R	R	R	R	R	R	R	R
3.5	R	R	R	R	R	R	R	R	R	M	M	M	M
4.0	R	R	R	M	M	M	M	M	M	M	M	M	M
4.5	M	M	M	M	M	M	M	M	M	M	M	L	L
5.0	M	M	M	M	M	M	M	L	L	L	L	L	L
5.5	M	M	M	M	L	L	L	L	L	L	L	L	L
6.0	M	L	L	L	L	L	L	L	L	L	L	L	L
6.5	L	L	L	L	L	L	L	L	L	L	L	L	L
7.0	L	L	L	L	L	L	L	L	L	L	L	L	L
7.5	L	L	L	L	L	L	L	L	L	L	L	L	L
8.0	L	L	L	L	L	L	L	L	L	L	L	L	L
8.5	L	L	L	L	L	L	L	L	L	L	L	L	L
9.0	L	L	L	L	L	L	L	L	L	L	L	L	L
9.5	L	L	L	L	L	L	L	L	L	L	L	L	L
10.0	L	L	L	L	L	L	L	L	L	L	L	L	L
10.5	L	L	L	L	L	L	L	L	L	L	L	L	L
11.0	L	L	L	L	L	L	L	L	L	L	L	L	L
11.5	L	L	L	L	L	L	L	L	L	L	L	L	L
12.0	L	L	L	L	L	L	L	L	L	L	L	L	L

[a] R = restrictive, M = moderate, and L = liberal.
[b] *X2* = no. of ponds in May in Prairie Canada, in millions.
[c] *X1* = no. of mallards in the population, in millions.

Table 4. Optimal regulatory choices[a] for mallards, conditioned on the dual objectives of maximum long-term harvest and achieving a population goal of 8.1 million, assuming compensatory hunting mortality and strongly density-dependent recruitment.

	$X2$[b]												
$X1$[c]	1.0	1.5	2.0	2.5	3.0	3.5	4.0	4.5	5.0	5.5	6.0	6.5	7.0
2.0	R	R	R	R	R	R	R	R	R	R	R	R	R
2.5	R	R	R	R	R	R	R	R	R	R	R	R	R
3.0	R	R	R	R	R	R	R	R	M	M	M	M	M
3.5	R	R	R	M	M	M	M	M	M	M	M	L	L
4.0	M	M	M	M	M	M	M	L	L	L	L	L	L
4.5	M	M	M	M	L	L	L	L	L	L	L	L	L
5.0	M	L	L	L	L	L	L	L	L	L	L	L	L
5.5	L	L	L	L	L	L	L	L	L	L	L	L	L
6.0	L	L	L	L	L	L	L	L	L	L	L	L	L
6.5	L	L	L	L	L	L	L	L	L	L	L	L	L
7.0	L	L	L	L	L	L	L	L	L	L	L	L	L
7.5	L	L	L	L	L	L	L	L	L	L	L	L	L
8.0	L	L	L	L	L	L	L	L	L	L	L	L	L
8.5	L	L	L	L	L	L	L	L	L	L	L	L	L
9.0	L	L	L	L	L	L	L	L	L	L	L	L	L
9.5	L	L	L	L	L	L	L	L	L	L	L	L	L
10.0	L	L	L	L	L	L	L	L	L	L	L	L	L
10.5	L	L	L	L	L	L	L	L	L	L	L	L	L
11.0	L	L	L	L	L	L	L	L	L	L	L	L	L
11.5	L	L	L	L	L	L	L	L	L	L	L	L	L
12.0	L	L	L	L	L	L	L	L	L	L	L	L	L

[a] R = restrictive, M = moderate, and L = liberal.
[b] $X2$ = no. of ponds in May in Prairie Canada, in millions.
[c] $X1$ = no. of mallards in the population, in millions.

by assigning equal probabilities of $l_i = 0.25$ to each model. The resulting harvest strategy (Table 5) is neither extremely liberal nor extremely conservative, but represents an "average" of model-specific strategies. Other combinations of model probabilities also provide intermediate strategies, with the similarity to a model-specific strategy strongly associated with the magnitude of its associated model probability. In the face of uncertainty about the most appropriate model, the optimal regulatory choice depends not only on $\mathbf{X}_t$, but on the probabilities $l_{i,t}$ assigned to models at time t.

ADAPTIVE HARVEST MANAGEMENT

In waterfowl harvest management, the iterative process of monitoring, assessment, and setting regulations provides the feedback loop necessary for making good decisions in the face of uncertainty. Once a regulatory option has been chosen, the effect of that decision is reflected in large-scale monitoring programs (Smith et al. 1989), whereby information from various surveys allows managers to gauge performance relative to their objectives and adjust harvest strategies as needed. Thus, waterfowl harvest management as it currently exists in North America can be considered an adaptive process, where changes in management are made in response to changing conditions. However, managers have failed to take full advantage of the capability of the regulations process for providing useful information about population dynamics.

We suggest a more efficient process where harvest strategies "evolve" over time in response to changes in the characterization of structural uncertainty. Given a particular regulatory decision in year t, each model of population dynamics provides a prediction for population size in year $t + 1$. Some models can be expected to perform better than others, and this performance can be assessed by comparing the model-specific prediction of population size with the population estimate derived from the monitoring program. The probability l_i for a particular model then is increased to the extent that the predicted and observed population sizes correspond, and decreased to the extent that they do not. Thus, the approach is a 3-step process:

(1) in year t, an optimal regulatory decision is identified based on model weights $l_{i,t}$ and system state $\mathbf{X}_t$;

(2) the regulatory decision having been made, model-specific predictions for population size in year $t + 1$ ($X1_{i,t+1}$) are determined; and

(3) when monitoring data from year $t + 1$

Table 5. Optimal regulatory choices[a] for mallards, conditioned on the dual objectives of maximum long-term harvest and achieving a population goal of 8.1 million, assuming equal probabilities for 4 alternative models of population dynamics.

	$X2$[b]												
$X1$[a]	1.0	1.5	2.0	2.5	3.0	3.5	4.0	4.5	5.0	5.5	6.0	6.5	7.0
2.0	C	C	C	C	C	C	C	C	C	C	C	C	C
2.5	C	C	C	C	C	C	C	C	C	C	C	C	C
3.0	C	C	C	C	C	C	C	C	C	C	C	C	C
3.5	C	C	C	C	C	C	C	C	C	C	C	C	C
4.0	C	C	C	C	C	C	C	C	C	C	C	C	R
4.5	C	C	C	C	C	C	C	C	R	R	R	R	M
5.0	C	C	C	C	C	R	R	R	R	M	M	M	L
5.5	C	C	R	R	R	R	M	M	M	L	L	L	L
6.0	R	R	R	M	M	M	M	L	L	L	L	L	L
6.5	R	M	M	M	M	L	L	L	L	L	L	L	L
7.0	M	M	M	L	L	L	L	L	L	L	L	L	L
7.5	M	M	L	L	L	L	L	L	L	L	L	L	L
8.0	M	L	L	L	L	L	L	L	L	L	L	L	L
8.5	L	L	L	L	L	L	L	L	L	L	L	L	L
9.0	L	L	L	L	L	L	L	L	L	L	L	L	L
9.5	L	L	L	L	L	L	L	L	L	L	L	L	L
10.0	L	L	L	L	L	L	L	L	L	L	L	L	L
10.5	L	L	L	L	L	L	L	L	L	L	L	L	L
11.0	L	L	L	L	L	L	L	L	L	L	L	L	L
11.5	L	L	L	L	L	L	L	L	L	L	L	L	L
12.0	L	L	L	L	L	L	L	L	L	L	L	L	L

[a] C = closed, R = restrictive, M = moderate, and L = liberal.
[b] $X2$ = no. of ponds in May in Prairie Canada, in millions.
[c] $X1$ = no. of mallards in the population, in millions.

are available, model probabilities are adjusted to reflect the relative performance of the alternative models.

The new model probabilities $l_{i,\,t+1}$ then are used to derive a regulatory strategy for $\mathbf{X}_{t+1}$, to start another iteration of the process. We suggest a cumulative procedure for updating model probabilities that relies on Bayes' theorem:

$$l_{i,t+1} = \frac{l_{i,t}p_i(X1_t,\, X1_{t+1})}{\sum_i l_{i,t}p_i(X1_t,\, X1_{t+1})}, \qquad (25)$$

where $p_i(X1_t,\, X1_{t+1})$ is the probability assigned to model i for observed changes in population size from $X1_t$ to $X1_{t+1}$ (Hilborn and Walters 1992:503–504). Thus, the new probability $l_{i,\,t+1}$ is a function of how well model i predicted $X1_{t+1}$ relative to how well any model predicted $X1_{t+1}$. Procedures for calculating $p_i(X1_t,\, X1_{t+1})$ are beyond the scope of this paper, but a useful example is provided by Williams et al. (1996).

This process can be described as passively adaptive, in the sense that informative changes in model weights occur as an unplanned byproduct of the regulatory process (Hilborn and Walters 1992:489). A major advantage of this process over the traditional approach is an explicit accounting for structural uncertainty by using empirical assessments of model probabilities. The primary disadvantage of passive adaptive management is in a failure to recognize that there are material benefits to reducing key structural uncertainties (Johnson et al. 1993, Williams and Johnson 1995).

The recognition that some regulatory strategies are more informative of system dynamics than others has led to consideration of more actively adaptive approaches (Walters and Holling 1990). Active adaptive management refers to the development of management strategies that are expected to reduce uncertainty about system dynamics so that future decisions can be improved. The development of such strategies involves a tradeoff between short-term management performance and the long-term value of knowing which alternative model of population dynamics is most appropriate (Walters and Holling 1990). The optimal balance between short- and long-term management performance can be determined with a generalization of the stochastic control problem, in which the temporal trajectories of $l_{i,t}$ are a function of regulatory choices (in much the same way that temporal changes in $X1$ are a function of regulations) (Walters 1986:265–269, Williams

1996*a*). The recent availability of computing algorithms, software (Williams 1996*b*), and affordable high-performance workstations should enable managers to overcome many of the past limitations (Walters 1986:277–278) in computing optimal adaptive strategies.

FUTURE APPLICATIONS

System Modeling

Our approach to harvest management is dependent on explicit hypotheses (i.e., models) describing how populations and management benefits respond to alternative actions. Although less formal, trial-and-error approaches to management often are defended, we agree that "models are indispensable, because without them human misunderstanding persists unaware of its errors" (Lee 1993:62). Use of explicit models in wildlife management has been limited, probably because of a belief that models must be complex to be useful or that their use is preempted by the presence of structural uncertainty. In many cases where explicit models have been used, their application has not been accompanied by the critical questioning necessary to make them useful (Conroy 1993).

The challenge in constructing useful models is to identify those key system features and attendant uncertainties that are relevant to the performance of management, as measured by an objective function. As in this paper, the focus typically will be on reproduction and survival and how controlled (e.g., harvest) and uncontrolled (e.g., weather) effects influence these processes. Frequently, reproduction and survival are described as being density independent, such that net growth rates are independent of the ratio of animals to some limiting resource. In this case dynamic programming can be used to derive optimal harvest strategies, but the manager must impose ad hoc constraints unrelated to system dynamics (e.g., min. acceptable population size or max. size of the harvest) (Getz and Haight 1989:19–65). A more realistic scenario is one in which competition for available resources among individuals increases mortality and/or decreases recruitment. In this case optimal harvest strategies are determined by the internal dynamics of the resource system and by a potentially less-constrained objective function. Of course a key difficulty in modeling density dependence and its use to guide management lies with recognizing, measuring, and forecasting the temporal and spatial dynamics of limiting resources (unless an assumption of constancy of these resources is reasonable). Such assessments are notoriously difficult, particularly for highly mobile species like migratory birds. An investigation of the implications for harvest management arising from the nature and forms of density-dependence (Fowler 1981) could help focus and establish priorities for these assessments.

Accounting for Uncertainty

Environmental variation, structural uncertainty, and partial management control are, by-and-large, permanent features of natural resource management. Our approach accounts explicitly for these sources of uncertainty by defining a set of possible management outcomes, rather than a single prediction. Whether management strategies thus identified are better than those arising from deterministic descriptions of system dynamics depends in large part on the ability to recognize and describe key uncertainties. Key uncertainties are those to which management rewards are sensitive, so that alternative system outcomes lead to alternative harvest strategies. Sensitivity analysis and other procedures can be used to examine candidate sources of uncertainty to determine whether an explicit accounting would materially alter the optimal management strategy. These assessments are critical from a practical viewpoint, as necessary computing resources grow exponentially as more stochastic features are included in the system description. For each source of uncertainty, the manager must be able to assign the probability of occurrence to different outcomes. Often this probability distribution can be estimated based on historic experience (e.g., annual precipitation in Prairie Canada). In rare cases, the expectation of the random variable is sufficient (Walters and Hilborn 1978, Puterman 1994:22) for calculating optimal strategies.

Conclusions

We believe our approach will have broad application to wildlife-harvesting problems. The descriptions of system dynamics, objective function, and set of management decisions can be quite complex, and are limited only by the availability of computing resources (Lubow 1994). Moreover, the need for a precise and explicit description of the harvesting problem will force managers to think critically about both the bi-

ological and sociological issues involved in harvest management. Fortunately, the availability of user-friendly software like SDP will allow managers to focus on these aspects of the harvesting problem, rather than on the mathematics and programming necessary to solve stochastic control problems.

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Appendix A. Duck hunting regulations in the United States, 1979–93. Shooting hours were always ½-hour before sunrise to sunset, except in 1988, when they were sunrise to sunset.

		Framework dates			Daily bag limit—mallards				
					Males		Females		
Flyway	Year	Opening	Closing	Season length (days)	Birds	Points	Birds	Points	States with point system
Atlantic	1979	1 Oct	20 Jan	50	5	25	5	70	5
	1980	1 Oct	20 Jan	50	5	25	5	70	4
	1981	1 Oct	20 Jan	50	5	25	5	70	4
	1982	1 Oct	20 Jan	50	5	25	5	70	4
	1983	1 Oct	20 Jan	50	5	25	5	70	4
	1984	1 Oct	20 Jan	50	5	25	5	70	4
	1985	5 Oct	13 Jan	40	5	35	5	100	1
	1986	4 Oct	18 Jan	40	3	35	1	100	1
	1987	1 Oct	18 Jan	40	3	35	1	100	1
	1988	7 Oct	8 Jan	30	3		1		0
	1989	7 Oct	7 Jan	30	3	35	1	100	0
	1990	6 Oct	6 Jan	30	3	35	1	100	0
	1991	5 Oct	5 Jan	30	3	35	1	100	0
	1992	1 Oct	20 Jan	30	3	35	1	100	0
	1993	1 Oct	20 Jan	30	3	35	1	100	0
Mississippi	1979	29 Sep	20 Jan	50	5	25	2	70	13
	1980	4 Oct	20 Jan	50	3	25	2	70	12
	1981	3 Oct	20 Jan	50	3	25	2	70	12
	1982	2 Oct	20 Jan	50	3	25	2	70	12
	1983	1 Oct	20 Jan	50	3	25	2	70	12
	1984	29 Sep	20 Jan	50	3	25	2	70	12
	1985	5 Oct	13 Jan	40	2	35	1	100	11
	1986	4 Oct	18 Jan	40	2	35	1	100	12
	1987	3 Oct	17 Jan	40	2	35	1	100	12
	1988	8 Oct	8 Jan	30	2		1		0

Appendix A. Continued.

Flyway	Year	Framework dates		Season length (days)	Daily bag limit—mallards				States with point system
					Males		Females		
		Opening	Closing		Birds	Points	Birds	Points	
	1989	7 Oct	7 Jan	30	2	50	1	100	0
	1990	6 Oct	6 Jan	30	2	50	1	100	0
	1991	5 Oct	5 Jan	30	2	50	1	100	0
	1992	1 Oct	20 Jan	30	2	50	1	100	0
	1993	1 Oct	20 Jan	30	2	50	1	100	0
Central	1979	29 Sep	20 Jan	60[a]	5	20	1	100	9
	1980	4 Oct	18 Jan	60[a]	5	20	1	100	9
	1981	3 Oct	17 Jan	60[a]	5	20	1	100	9
	1982	2 Oct	23 Jan	60[a]	5	20	1	100	9
	1983	1 Oct	22 Jan	60[a]	5	20	1	100	9
	1984	29 Sep	20 Jan	60[a]	5	20	1	100	9
	1985	5 Oct	13 Jan	50[b]	3	35	1	100	9
	1986	4 Oct	18 Jan	51[b]	4	35	1	100	9
	1987	3 Oct	17 Jan	51[b]	4	35	1	100	9
	1988	8 Oct	8 Jan	39[c]	2		1		
	1989	7 Oct	7 Jan	39[c]	2	50	1	100	5
	1990	6 Oct	6 Jan	39[c]	2	50	1	100	5
	1991	5 Oct	5 Jan	39[c]	2	50	1	100	6
	1992	1 Oct	20 Jan	39[c]	2	50	1	100	6
	1993	1 Oct	20 Jan	39[c]	2	50	1	100	7
Pacific	1979	29 Sep	20 Jan	93[d]	7		7		0
	1980	4 Oct	18 Jan	93[d]	7		7		0
	1981	3 Oct	17 Jan	93[d]	7		7		0
	1982	2 Oct	23 Jan	93[d]	7		7		0
	1983	1 Oct	22 Jan	93[d]	7		7		0
	1984	29 Sep	20 Jan	93[d]	7		7		0
	1985	8 Oct	13 Jan	79[d]	5		1		0
	1986	4 Oct	18 Jan	79[d]	4		1		0
	1987	3 Oct	10 Jan	79[d]	4		1		0
	1988	8 Oct	8 Jan	59[d]	3		1		0
	1989	7 Oct	7 Jan	59[d]	3		1		0
	1990	6 Oct	6 Jan	59[d]	3		1		0
	1991	5 Oct	5 Jan	59[d]	3		1		0
	1992	1 Oct	20 Jan	59[d]	3		1		0
	1993	1 Oct	20 Jan	59[d]	3		1		0

[a] Plus an extra 23 days in the High Plains Mallard Management Unit.
[b] Plus an extra 15–16 days in the High Plains Mallard Management Unit.
[c] Plus an extra 12 days in the High Plains Mallard Management Unit.
[d] Plus an extra 7 days in the Columbia Basin Mallard Management Unit.