Improving the Cost-Effectiveness of Ecosystem Management: An Application to Waterfowl Production

Benjamin S. Rashford and Richard M. Adams

Species conservation is an important global policy issue. The design of cost-effective species conservation programs requires resource managers to choose from a suite of conservation activities and sites. This article determines cost-effective conservation strategies for waterfowl using a bioeconomic modeling framework, which is developed using a biological simulation model for waterfowl and regression analysis. The model accounts for (*a*) a broad range of land-use and direct wildlife conservation activities, (*b*) the effect of landscape heterogeneity, and (*c*) interactions between conservation activities. Results indicate that accounting for the three factors listed above can improve the cost-effectiveness of waterfowl conservation on agricultural land.

Key words: cost minimization, simulation, waterfowl, wildlife management.

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Wildlife and their habitat play a critical role in the provision of many ecosystem services, such as soil protection and pest control, in addition to providing consumptive and passive benefits. In recognition of these values, species conservation is an important global policy issue. A common goal for species conservation programs is to minimize the cost of achieving species population objectives, because cost-effective[1] conservation ultimately allows higher levels of conservation to be achieved (Wu, Adams, and Boggess 2000).

To design cost-effective species conservation programs, a wildlife manager must choose a portfolio of activities from a potentially large suite of conservation activities and conservation sites (U.S. Fish and Wildlife Service 1996). Two issues complicate this decision: (*a*) biological response and economic costs vary across the landscape depending on the quality of existing habitat and alternative land uses, and (*b*) biological response to individual conservation activities depends on the entire portfolio of activities implemented. Accounting for landscape heterogeneity and the interactions between conservation activities requires detailed biological response functions. One of the greatest management challenges is the derivation of detailed response functions that reveal the trade-offs between a range of conservation activities across a heterogeneous landscape (Matulich and Hanson 1986).

In this study, we determine cost-effective species conservation strategies using a bioeconomic model that accounts for (*a*) a broad range of land-use and direct conservation activities, (*b*) interactions between conservation activities, and (*c*) landscape heterogeneity in both market (e.g., agricultural crops) and biological production. Previous literature has addressed these issues individually, the authors are aware of no species conservation study that addresses all three simultaneously. The bioeconomic model is developed in three steps. First, a biological simulation model is used to simulate the affect of management activities on species' response. Data from the simulation model are then used to approximate a continuous response function, which predicts response to varying levels and combinations of management activities. Lastly, the response function is embedded within an economic optimization

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[1] Cost-effective means that conservation goals are achieved at minimum cost. This condition is necessary but not sufficient for full economic efficiency. Economic efficiency, in this context, would imply that the level and mix of species was achieved where marginal social costs equal marginal social benefits. Efficiency would require nonmarket valuation of species-specific benefits.

model to identify least-cost management strategies. The empirical focus of this analysis is waterfowl production in the north-central United States; however, the results have implications for species conservation and ecosystem service provision in other settings. Results indicate that accounting for the three factors listed above can improve the cost-effectiveness of species conservation programs.

Literature Review

There is a growing body of literature that examines cost-effective species conservation. Many of these studies only consider conservation activities related to the land's primary use. Examples include the use of alternative crop management practices, such as conservation tillage, to produce biodiversity on crop farms (Pacini et al. 2004; van Wenum, Wossink, and Renkema 2004), and the use of alternative timber harvest strategies to conserve forest-dwelling species (Lichtenstein and Montgomery 2003; Nalle et al. 2004). Many of these studies explicitly model tradeoffs between species preservation and the production of marketable outputs (Rohweder, McKetta, and Riggs 2000; Lichtenstein and Montgomery 2003; Nalle et al. 2004).

Most studies do not consider conservation activities that directly manipulate the species, such as the control of predators and direct habitat manipulation. Since many direct activities do not infringe on the production of marketable outputs, they may reduce the cost of conservation (Marshall, Homans, and Haight 2000). Prior studies of the cost-effectiveness of direct conservation activities have generally considered single activities, such as harvest (Boman, Bostedt, and Persson 2003) and predator control (Engeman et al. 2002).

Previous studies have also captured the economic effect of landscape heterogeneity on species conservation. The reserve site selection literature has, for example, demonstrated that species conservation costs can be reduced by incorporating heterogeneous land values into the reserve site decision (Polasky, Camm, and Garber-Yonts 2001). However, since biological outputs are usually not represented in the marketplace, heterogeneous land values rarely capture the relationship between landscape characteristics and species production. Two common approaches have been used to incorporate the impact of landscape heterogeneity on species production. One approach uses general landscape measures, such as habitat clustering, to relate landscape characteristics to biological processes (see, e.g., Lichtenstein and Montgomery 2003). Conservation activities are typically defined broadly with this approach, such as the size and location of preserved areas. The second approach uses regression or simulation modeling to derive species or biodiversity production relationships as a function of landscape characteristics (Nalle et al. 2004; van Wenum, Wossink, and Renkema 2004). Conservation activities that alter landscape characteristics are therefore directly linked to species response, allowing for the inclusion of more specific conservation activities.

Few studies consider interactions that occur when many conservation activities are applied simultaneously. Several studies that examine diverse sets of conservation activities have not included interactions in their models (Lokemoen 1984; Matulich and Hanson 1986). Marshall, Homans, and Haight (2000), in contrast, explicitly discuss the potential for interactions when multiple conservation activities for the endangered Kirtland's warbler are applied simultaneously. Interactions are modeled indirectly in a regression analysis that includes interactions terms between key population parameters (e.g., nest density and adult survival). Interactions between conservation activities, therefore, arise if conservation activities impact different population parameters. The authors do not explicitly include such activities or consider their associated costs in the analysis. Biologists also discuss interactions between conservation activities, noting that similar activities applied simultaneously may shift species populations across space without increasing populations (Johnson, Sparling, and Cowardin 1987). The economic consequences of such interactions, however, have not been examined directly.

This study complements and extends the literature on cost-effective species conservation by developing a bioeconomic model that accounts for a broad range of land-use and direct wildlife conservation activities, some of which interact, across a heterogeneous landscape. Expanding the activities to include both primary land-use activities and direct wildlife conservation activities may increase the cost-effectiveness of achieving species goals; in part, because direct conservation activities typically do not preclude market production. Landscape heterogeneity generates opportunities to reduce conservation costs through spatially differentiated strategies. Finally, inclusion of interactions between conservation activities, when they are significant, should improve estimates of species response to conservation activities and lead to more cost-effective allocation of conservation resources.

The bioeconomic model developed in this study is applied to the conservation of waterfowl[2] in the Prairie Pothole Region (PPR) of the Northern Great Plains. Waterfowl are an ideal subject for this analysis for the following reasons: (*a*) a broad range of land-use and direct wildlife conservation activities are available for waterfowl conservation, (*b*) the primary waterfowl breeding ground in the PPR coincides with high-value agricultural land, which allows for an analysis of the substitutability of primary land-use and direct conservation activities, and (*c*) a detailed biological simulation model exists for one key indicator species, the mallard (*Anas platyrhynchos*), which allows for the simulation of multiple conservation activities applied simultaneously, and landscape heterogeneity. Though this study is applied to mallards, the results are expected to apply to other waterfowl species with similar habitat requirements. The insights gleaned also apply, in general, to terrestrial species for which many conservation activities exist.

The basic theoretical framework of the bioeconomic model is presented in the next section, followed by a description of its application to waterfowl conservation, which includes a description of the biological simulation model, methods used to simulate conservation activities, methods to approximate continuous waterfowl response functions, and a description of conservation activity costs. This is followed by a detailed description of the wildlife manager's optimization problem, followed by results and discussion.

Bioeconomic Decision Model

Consider the problem of a wildlife manager, whose goal is to achieve a specific population objective for a particular species and has a portfolio of conservation activities and locations from which to choose. Let x be a vector of conservation activities, w a vector of conservation activity costs, l a vector of landscape characteristics, and y a species population objective. This problem can then be written as:

$$(1) \qquad \min_{\{x\}} C(x, w; l)$$

subject to

$$(2) \qquad f(x; l) \geq y.$$

The cost relationship, $C(\cdot)$, is a function of landscape characteristics, l. This allows the cost of each activity to vary based on landscape characteristics, such as soil quality. The biological response relationship, $f(\cdot)$, maps conservation activities to species population levels and is also landscape-specific, i.e., change in species population from the application of conservation effort also varies by landscape. Thus, application of identical conservation activities on alternative landscapes can result in diverse species response. The population target, y, is interpreted here as an average population level to be achieved over a specified time horizon, such as the length of the current conservation planning period, to be consistent with typical species conservation plans (U. S. Fish and Wildlife Service 2004). Solving the constrained minimization problem yields the set x^*, the vector of landscape-specific conservation activities that minimize the cost of satisfying the species population objective y. Iterating y over a range of population targets traces out the species conservation cost frontier.

Applying the Bioeconomic Decision Model to Waterfowl Conservation

Application of the model to waterfowl conservation requires derivation of the biological response relationship, $f(\cdot)$, and identification of conservation activity costs, w. Biological response relationships are derived in two steps. First, response data for mallards are generated using a biological simulation model, the Mallard Productivity Model (MM), version 171.027beta. Second, the data are used in a regression analysis to estimate continuous response functions.

Simulating Waterfowl Response

The MM is a stochastic simulation model designed to predict the impact of conservation activities on mallard populations (Johnson, Sparling, and Cowardin 1987). The MM simulates the mallard reproductive cycle using

[2] "Waterfowl" refers to a large and diverse group of birds that are ecologically dependent on a watery environment for some stage of their lifecycle. Taxonomically waterfowl are of the order Anseriformes, family Anatidae, and encompass 147 species of ducks, geese, and swans worldwide (Welty and Babtista 1988).

relationships between landscape characteristics and waterfowl ecology. Detailed information about the landscape being modeled is required, including wetland conditions and land use/land cover information. Once a landscape has been defined, the MM uses wetland type and abundance to predict the number of breeding pairs expected to nest in the area. The females[3] of each breeding pair are then ushered through stages in the breeding cycle, such as nest site selection and brood rearing. The fate of each nest is tracked through the breeding season, and population characteristics are calculated (see Johnson, Sparling, and Cowardin [1987] for full model description).

Each run of the MM represents a single breeding season that is treated in isolation of past and future seasons. Since the MM is static, inter-annual dynamics, such as long-term population growth and predator-prey dynamics are not captured. However, dynamic changes within a breeding season are captured. Specifically, the MM simulates habitat changes through the breeding season, including the natural draw down of ponds, growth of cover and crops, and anthropogenic disturbances related to farm operations. Stochasticity enters the MM at several key stages in the breeding cycle. For example, nest initiation is modeled as a random variable since females do not always initiate a nest. The probability that a female initiates a nest depends on her physical condition, the date, and wetland conditions (Johnson, Sparling, and Cowardin 1987). Nest site selection, the fate of individual nests, and daily survival of hens and ducklings are also random variables that depend on habitat conditions.

Implementation of conservation activities is simulated by manipulating habitat categories and adjusting model parameters. The following eight conservation activities are simulated, each of which are widely applied in the PPR. Three activities, cropland retirement, conservation tillage, and delayed haying, are primary land-use activities, while the remaining activities of planted cover, planted cover fenced, nest structures, predator control, and wetland restoration are direct wildlife conservation activities. A brief description of each conservation activity and how it is simulated is presented below.

[3] The model tracks female mallards because males outnumber females and therefore do not limit reproduction (Johnson, Sparling, and Cowardin 1987).

Cropland retirement (CR). Fall-plowed grains (wheat, barley, or oats) present a serious problem for waterfowl that nest in early spring, such as mallards. In the absence of alternative cover, many waterfowl species will initiate nests in grain fields prior to seeding. The seeding process causes nearly total nest destruction (99%) (Greenwood et al. 1995). Cropland retirement programs, such as the Conservation Reserve Program (CRP), reduce this nest mortality and increase waterfowl production (Reynolds et al. 2001). We simulate cropland retirement by transferring acreage from the fall-plowed grain category to the CRP category.

Conservation tillage (CT). No till cropping systems, where grain crops are seeded directly into the stubble from the previous year, are more productive for waterfowl than annually tilled crops (U.S. Fish and Wildlife Service 1996). We simulate the conversion of conventionally tilled to no-till cropland by shifting acreage from the fall-plowed grain category to the no-till cropland category.

Delayed haying (DH). Hay in the PPR (primarily alfalfa) is harvested in mid to late June, when many waterfowl are still occupying nests initiated in hay fields, and results in high nest mortality. Delaying the harvest of hay until mid July, when most nesting activities are complete substantially reduces nest mortality (Higgins et al. 1992). We simulate delayed haying by transferring acreage from the hayland category to the delayed hay category, which has a harvest date of July 15.

Planted cover (PC). Land planted to various mixtures of grasses and legumes provide the cover necessary to increase waterfowl production (Cowardin, Sargeant, and Duebbert 1983). We simulate the planting of cover by converting hayland and grassland acreage to the planted cover category.

Planted cover-fenced (PCF). Nest predation is another cause of reduced waterfowl production in the PPR (Sovada, Anthony, and Batt 2001). One way to reduce nest predation is to surround nesting cover with an electric predator fence. We simulate this management action by transferring acreage from hayland and grassland to the planted cover-fenced habitat category.

Nest structures (NS). Waterfowl nesting structures are also used to reduce nest predation (Marcy 1986). Nest structures take various forms, including open-toped cones, cylinders, and concrete nesting culverts. The MM includes a nest structures category, which has success rates similar to those documented for cones and cylinders. We assume that nesting structures are of the cylinder variety, which are readily available and commonly applied in the PPR. Nest structures are simulated by adjusting the nesting structure category. Nesting structures essentially occupy no land area, so no acreage is removed from other habitat types when nesting structures are employed.

Predator control (PRED). Although nest structures and fenced enclosures reduce nest depredation, research suggests that lethal predator control using professional trappers is an effective way to increase waterfowl production (Garrettson and Rohwer 2001). The MM captures the effects of predation through a predation index parameter, which can be set to high, medium, or low to represent the abundance of waterfowl predators. Two levels of predator control are simulated by changing the predation index from high (no predator control) to medium or low to represent reduced predator abundance.

Wetland restoration (WR). Wetland composition and abundance influence the number of waterfowl attracted to breeding areas, as well as their nesting effort (Johnson, Cowardin, and Sparling 1986). Wetlands in the PPR have been extensively drained to accommodate agricultural land uses (Tiner 1984). Semi-permanent wetlands are the most productive for waterfowl (Cowardin et al. 1983), so we assume that the wetland restoration conservation activity restores previously drained semi-permanent wetlands by shifting acreage from Fall-plowed Grains and Hayland habitats to the semi-permanent wetland category.

To capture the role of landscape heterogeneity in the design of cost-effective conservation, we simulate waterfowl response to the conservation activities discussed above on three alternative landscapes: poor, average, and good. These landscapes were designed to represent the spectrum of land use currently observed in the PPR. Each landscape is 2,000 acres (approximately the average farm size in the PPR) and is differentiated by its relative productivity for waterfowl versus agriculture in the absence of conservation. Land use and land cover characteristics for the three landscapes are presented in table 1.

The poor and average landscapes are both dominated by agricultural activity, but differ in wetland abundance. The poor landscape has the highest quality agricultural land and significant wetland drainage. The average landscape has average quality agricultural land and, consequently, significantly less wetland drainage. Although active farmland dominates much of the PPR, there has been significant conservation and restoration through the CRP and Wetland Reserve Program. The good landscape represents this restoration trend, with largely intact wetlands and a significant portion of land in CRP.

On each landscape we simulate single conservation activities and combinations of activities, with activities applied at multiple levels (e.g., 10, 20, . . . , 100 acres). The resulting set of simulations consists of 352, 376, and 427 conservation strategies on the poor, average, and good landscapes, respectively. Since the

Table 1. Initial Habitat Configuration of Simulated Landscapes

	Poor Landscape Acres (%)	Average Landscape Acres (%)	Good Landscape Acres (%)
Fall-plowed grain	862 (43.1)	784 (39.2)	0 (0)
Stubble grain	366 (18.3)	400 (20)	0 (0)
Summer fallow	184 (9.2)	201 (10.1)	0 (0)
Grassland	317 (15.9)	191 (9.5)	357 (17.9)
Hayland	122 (6.1)	134 (6.7)	371 (18.5)
CRP	0 (0)	0 (0)	706 (35.3)
Seasonal wetland	74 (3.7)	147 (7.4)	230 (11.5)
Semi-permanent wetland	36 (1.8)	99 (4.9)	200 (10)
Temporary wetland	3 (0.15)	10 (0.5)	100 (5)
Permanent wetland	1 (0.05)	3 (0.1)	5 (0.25)
Other	35 (1.7)	31 (1.6)	31 (1.55)

landscapes differ in initial habitat configuration, conservation activities cannot be applied identically on all landscapes. For example, the good landscape contains no fall-plowed grains; therefore, there is no acreage upon which to apply conservation tillage.

Because the MM is stochastic, the result of an individual run represents one observation of a random process. We, therefore, use an estimate of the sample mean of recruits to avoid the influence of "bad" draws. To determine the sufficient number of runs, we iteratively simulate data and calculate a confidence interval around the estimated mean, stopping when the confidence interval converges (see Ross 2002). This iterative process is conducted on the good landscape with wetland parameters set to wet because that scenario has the greatest variability. With 100 runs, the confidence interval around the estimated mean stabilizes to ± 3.5 recruits. One hundred runs are sufficient to assure a confidence interval at least this small for scenarios with less variability. For each conservation strategy 100 observations per wetland setting (dry, average, wet) are simulated, for a total of 300 observations per strategy. We then measure production with the average number of recruits across the 300 observations.

Approximating Continuous Waterfowl Response Functions

The simulations generate mean waterfowl response data for various levels and combinations of conservation activities on three alternative landscapes. We next use regression analysis to approximate a continuous response surface, which we then embedded within the manger's decision model. This "metamodeling" approach allows us to approximate response without having to simulate every possible combination of management activities (Wu and Babcock 1999). Alternatively, the simulation procedure could be embedded directly within the optimization routine. The MM, however, is too complex and simulations are too slow for this approach. The use of such regression models to proxy for detailed biological simulation models is established in the literature (Hof and Raphael 1997; Calkin et al. 2002; Nalle et al. 2004).

Little guidance exists on the appropriate specification of a waterfowl response function. A quadratic response function is often used to model biological response because second-order (or higher) polynomials allow for diminishing marginal productivity (Adams et al. 1993). A quadratic specification also allows us to capture interaction effects using multiplicative terms. The quadratic response functions take the following general form:

$$(3) \qquad y_l = \alpha_l + \sum_i^k \left(\beta_{1il} x_{il} + \beta_{2il} x_{il}^2\right) + \sum_i^k \sum_j^k \delta_{ijl} x_{il} x_{jl} + \varepsilon_l$$

where y_l is mean recruits on landscape l, x_{il} is the level of conservation activity i on landscape l, ε_l is a random disturbance term, and α_l, β_{1il}, β_{2il}, and δ_{ijl} are parameters to be estimated. The unit of measurement for all activities is acres, except for *NS*, which is measured as the number of nesting structures installed.

Diminishing marginal productivity has been observed for activities that directly increase nesting habitat (Cowardin, Shaffer, and Kraft 1995). Given a fixed number of ponds, and therefore a fixed number of breeding pairs, incremental recruits are expected to increase at a decreasing rate as the quantity of nesting habitat increases, because nesting habitat eventually exceeds the needs of the fixed number of breeding pairs. Second-order terms for the activities that directly create nesting habitat are therefore included: *CRP*, *DH*, *NT*, *PC*, *PCF*, and *NS*.

Three groups of conservation activities are expected to generate interaction effects. First, habitats similar in their attractiveness to breeding waterfowl and located within the same landscape will compete for the fixed number of breeding pairs. Cropland retirement, planted cover, and planted cover fenced, which create similar nesting cover, are, therefore, treated as competitive and we include interaction terms between these activities. Second, research on waterfowl nest depredation suggests that large blocks of intact grassland cover reduces nest densities, which reduces the probability that predators locate and depredate a nest (Sovada et al. 2000). Predator control will compete with activities that create extensive nesting cover, and we include interaction terms between predator control and the following activities: *CRP*, *NT*, *DH*, and *PC*. Lastly, the number of breeding pairs on a landscape is affected by wetland abundance. Increasing wetland acres will, therefore, complement other activities by increasing the number of breeding pairs present to utilize them. Interaction terms between wetland restoration and all other activities are thus included.

We estimate model parameters by ordinary least squares using the software package Limdep NLOGIT 3.0 (Greene 2003). Second-order terms for *CRP*, *DH*, *NT*, *PC*, *PCF*, and *NS* were estimated. However, coefficients on CR^2, DH^2, and NT^2 were not significant on any landscape, and an F-test for their joint significance failed to reject the null hypothesis that they are all equal to zero, suggesting no diminishing marginal productivity at the levels applied here. The interaction effects between *PC* and *PCF* were also insignificant on all landscapes. Since both *PC* and *PCF* are simulated by replacing the same land-use categories, it is not possible to simulate both activities at high levels simultaneously. There are, therefore, an insufficient number of observations on these activities jointly to capture their interaction. Finally, though biological principles suggest that *WR* should complement other activities, all interaction effects for *WR* were insignificant. This likely results because increasing wetlands attracts new breeding pairs relatively slowly (15–20 acres must be added to observe a new breeding pair [Cowardin et al. 1983]) and, therefore, there is not significant variation in the number of breeding pairs to capture interactions.[4]

The intuition above suggests that the insignificant second-order and interaction terms from the first regression model are the result of the scale of this analysis. On small landscapes, their insignificance is theoretically justified though this fact was not known a priori; therefore, we omit these variables and reestimate the models. Using the final specification, we estimate four models, one for each landscape, and a pooled model that restricts the coefficients to be constant across landscapes (excluding the intercept term). The pooled model assumes that waterfowl response to management is independent of initial landscape characteristics. To test this hypothesis, we perform a Chow test of the individual landscape models versus the pooled model. The calculated F-statistic is 126.18, with a 99% critical value given by $F[40, \infty] = 1.59$; therefore, we reject the null hypothesis that the estimated coefficients are the same across landscapes, and adopt separate waterfowl response functions for each landscape. Parameter estimates for each landscape are provided in table 2.

Conservation Costs

Conservation activity costs are obtained from the wildlife management literature and through correspondence with waterfowl managers. All costs are expressed in \$2004, and for activities that have a useful life of more than one year, relevant costs are annualized using a 4% discount rate. Table 3 provides the per unit cost estimate for each conservation activity by landscape. Costs include land use, management, and construction costs, when applicable. Land-use costs are based on annual farmland rental rates for North Dakota cropland, pasture and hayland (Knopf 2004). We assume that differences in initial land use reflect differences in cropland quality, such that landscapes with more intense agricultural land use are assigned higher rental rates. Therefore, the rental rate of cropland on the average landscape is set equal to the average rental rate; the rental rate of cropland on the good and poor landscapes is equal to the maximum and minimum observed rental rates, respectively. The average and poor landscapes have similar cropland acreage (1,710 acres versus 1,851 acres), but we assign a lower rental rate to the average landscape. This is because the average landscape has more wetlands (259 acres versus 114 acres), which have a "nuisance cost" to farmers associated with maneuvering large machinery around wetlands during cultural operations (Higgins, Naugle, and Forman 2002) and therefore a lower return to agricultural land use.

The cost estimates for direct conservation activities are obtained directly from the wildlife management literature (see Rashford [2006] for complete description and references). Cost estimates for conservation tillage and delayed hay are less straightforward. For both activities we assume that the landowner is always willing to accept their opportunity cost to undertake these activities. This is a reasonable assumption on public land managed for wildlife, but may not always hold on private land where landowners face multiple incentives and may attempt to "game" conservation schemes.[5] We abstract away from this complexity; therefore, our land-use costs should be viewed as conservative

[4] The insignificant interaction between WR and other activities observed here is noteworthy given that many waterfowl habitat programs, such as the North American Wetlands Conservation Act, emphasize wetland restoration. Current biological research is reexamining the role of wetlands in predicting waterfowl pair abundance (Gorsuch 2004); this information should be integrated into future economic and policy analyses of waterfowl conservation.

[5] We thank an anonymous referee for making this point.

Table 2. Parameter Estimates for the Waterfowl Response Function

	Landscape		
Variable	Poor	Average	Good
Constant	7.98 (0.843)**	17.82 (1.56)**	62.32 (1.78)**
CR	0.008 (0.001)**	0.017 (0.003)**	0.022 (0.006)**
DH	0.065 (0.015)**	0.127 (0.025)**	0.064 (0.016)**
CT	0.004 (0.002)**	0.01 (0.003)**	–
NS	1.92 (0.132)**	3.69 (0.227)**	1.31 (0.155)**
PC	0.043 (0.007)**	0.082 (0.013)**	0.05 (0.018)**
PCF	0.158 (0.006)**	0.446 (0.024)**	0.278 (0.017)**
PRED	0.029 (0.0007)**	0.064 (0.001)**	0.11 (0.001)**
WR	0.114 (0.009)**	0.138 (0.016)**	0.17 (0.017)**
NB^2	−0.023 (0.004)**	−0.046 (0.006)**	−0.006 (0.003)*
PC^2	−0.00003 (0.00001)**	−0.00006 (0.00002)**	−0.00003 (0.00003)
PCF^2	−0.0001 (0.00001)**	−0.0007 (0.00007)**	−0.0002 (0.00004)**
$PRED * CR$	−0.000008 (0.000001)**	−0.00002 (0.000003)**	0.00003 (0.000006)**
$PRED * DH$	−0.00009 (0.00001)**	−0.0002 (0.00003)**	−0.00008 (0.00002)**
$PRED * CT$	−0.00001 (0.000002)**	−0.00003 (0.000003)**	–
$PRED * PC$	−0.00002 (0.000005)**	−0.00006 (0.00001)**	−0.00004 (0.00001)**
$CRP * PC$	−0.000009 (0.00001)	−0.000005 (0.00002)	−0.000001 (0.00006)
$CRP * PCF$	−0.00007 (0.00001)**	−0.0002 (0.00003)**	−0.00003 (0.00007)
$F[n,k]$	223.49	304.77	840.63
R^2	0.919	0.935	0.977
Adj. R^2	0.915	0.932	0.976

Notes: *, ** indicate significance at the 5% and 1% levels, respectively. Standard errors are in parentheses.
CR = cropland retirement program, DH = delayed hay, CT = conservation tillage, NS = nest structures, PC = planted cover, PCF = planted cover fenced, PRED = predator control, WR = wetland restoration.

Table 3. Per Unit Management Costs by Landscape

Management Activity	Landscape		
	Poor	Average	Good
Cropland retirement (*CR*)	$70.00	$33.00	$25.00
Conservation tillage (*CT*)	$15.00	$15.00	$15.00
Delayed hay (*DH*)	$25.00	$25.00	$25.00
Nesting structures (*NS*)	$22.90	$22.90	$22.90
Planted cover (*PC*)	$17.50	$17.50	$17.50
Planted cover fenced (*PCF*)	$110.20	$110.20	$110.20
Predator control (*PRED*)	$2.00	$2.00	$2.00
Wetland restoration (*WR*)	$75.00	$38.00	$30.00

estimates. For conservation tillage, costs are estimated as the difference between net revenues from conventional versus conservation tillage practices. Though some studies indicate that no-till practices increase net revenues, the benefits and costs of conservation tillage practices differ significantly with soil type, weather, crops, input and output prices, risk preferences and management skill (Fox et al. 1991). The reduction in net income from conservation tillage in the PPR ranges from $3 to $28 per acre (Zentner et al. 1996; Kurkolova, Cling, and Zhao 2003). We assign conservation tillage a constant per acre cost equal to the average net income reduction estimated for the PPR.

The opportunity cost of delaying hay harvest until July 15 is the foregone value of harvesting hay at its optimal nutritional value and the lost revenues from foregoing multiple harvests. Published data relating hay nutritional value on July 15 to hay prices could not be found for the PPR. Therefore, we approximate the cost of delayed haying with the value of foregone multiple cuttings. This value is estimated by the difference between net returns from a two cut versus single cut hay harvest using yield and price data from Meyer and Helm (1994).

Estimating Cost-Effective Waterfowl Conservation Strategies

The estimated waterfowl response functions and cost estimates discussed above are used to specify and implement the general equation system in (1) and (2) as follows.

$$(4) \qquad \min_{\{x\}} \sum_i \sum_l w_{il} x_{il}$$

subject to

$$(5) \qquad \sum_l f_l(x_{1l}, \ldots, x_{Il}) \geq Y^*$$

$$(6) \qquad \sum_i x_{il} \leq A_{grain,l} \; \forall l, \quad i = CR, CT, WR$$

$$(7) \qquad \sum_i x_{il} \leq A_{hay,l} + A_{grass,l} \; \forall l, \quad i = DH, PC, PCF$$

$$(8) \qquad x_{il} \leq x_{jl} + B_l \; \forall l, i = NB, j = WR$$

$$(9) \qquad 0 \leq x_{il} \leq X_{il}.$$

Here, w_{il} denotes the per unit cost of management activity i on landscape l; x_{il} denotes the level of management activity i on landscape l; $f_l(x_{1l} \ldots x_{Il})$ denotes the response function for landscape l as given by (3); Y^* denotes the production objective, measured as the incremental change in recruits (i.e., $Y - \alpha$ in (3)); $A_{grain,l}$ is initial acreage in grain crops (fall-plowed grain, stubble grain, and summer fallow) on landscape l; $A_{hay,l}$ is initial acreage in hayland on landscape l; $A_{grass,l}$ is initial acreage in grassland on landscape l; B_l is the initial number of mallard breeding pairs on landscape l; and X_{il} is the maximum amount of activity i that can be applied to landscape l given initial landscape characteristics. Equation (6) limits *CRP*, *NT*, and *WR* to no more than the initial eligible acres. Equation (7) similarly restricts *DH*, *PC*, and *PCF*. Equation (8) restricts the number of nesting structures to no more than the initial number of breeding pairs plus the acres of *WR* to be consistent with recommendations in the waterfowl literature (Marcy 1986; U.S. Fish and Wildlife Service 1996).

We use the software package Matlab 6.5 (The Mathworks 2002) to solve the wildlife manager's nonlinear programming problem, as expressed in (4)–(9). The manager's problem is solved iteratively for alternative production objectives that range from zero to $Y^{\max}$, where $Y^{\max}$ is the maximum number of recruits that can be produced subject to (5)–(9). This iterative process yields the total cost function for the production of waterfowl recruits on one landscape. Average and marginal cost functions are then derived from the total cost function. This process is repeated for each landscape and for a pooled model that optimizes over all landscapes simultaneously (hereafter denoted the full model).

Results and Discussion

Primary Land Use versus Direct Conservation Activities

Activity levels employed in the full model are used to compare the cost-effectiveness of primary land-use conservation activities (*CRP*, *NT*, *DH*) versus direct conservation activities (*NS*, *PC*, *PCF*, *PRED*, *WR*). Figure 1 shows the activity levels employed in the cost minimizing solution for the full range of mallard production objectives. Direct activities are employed at all production targets, indicating that they are cost-effective relative to primary land-use activities Nesting structures are the first conservation activities implemented, followed by predator control and wetland restoration. In contrast, only one primary land-use activity, *DH*, is implemented in the cost-minimizing strategy and only on the good landscape when the production target exceeds 675 recruits. These results indicate that direct conservation strategies are more cost-effective for producing mallards, because direct conservation activities have high marginal productivity per acre and therefore require less land to be removed from agricultural production than primary land-use activities. Thus, mallard production does not compete with the production of marketable outputs when direct conservation activities are used.

It is important to note that although primary land-use activities are less cost-effective for mallard production, they generate other ecosystem services that are not included in this model. For example, *CRP* and *NT* reduce soil erosion, improve water quality, and sequester carbon. They also generate benefits for other waterfowl and grassland bird species. Research suggests that pintails (*Anas acuta)*, for example, benefit significantly more from *NT* than mallards (U.S. Fish and Wildlife Service 1996), and that grassland birds benefit relatively more from *CRP* (Szentandrasi et al. 1995). Because these other ecosystem services are not included, the full cost of primary land-use activities is apportioned to mallard

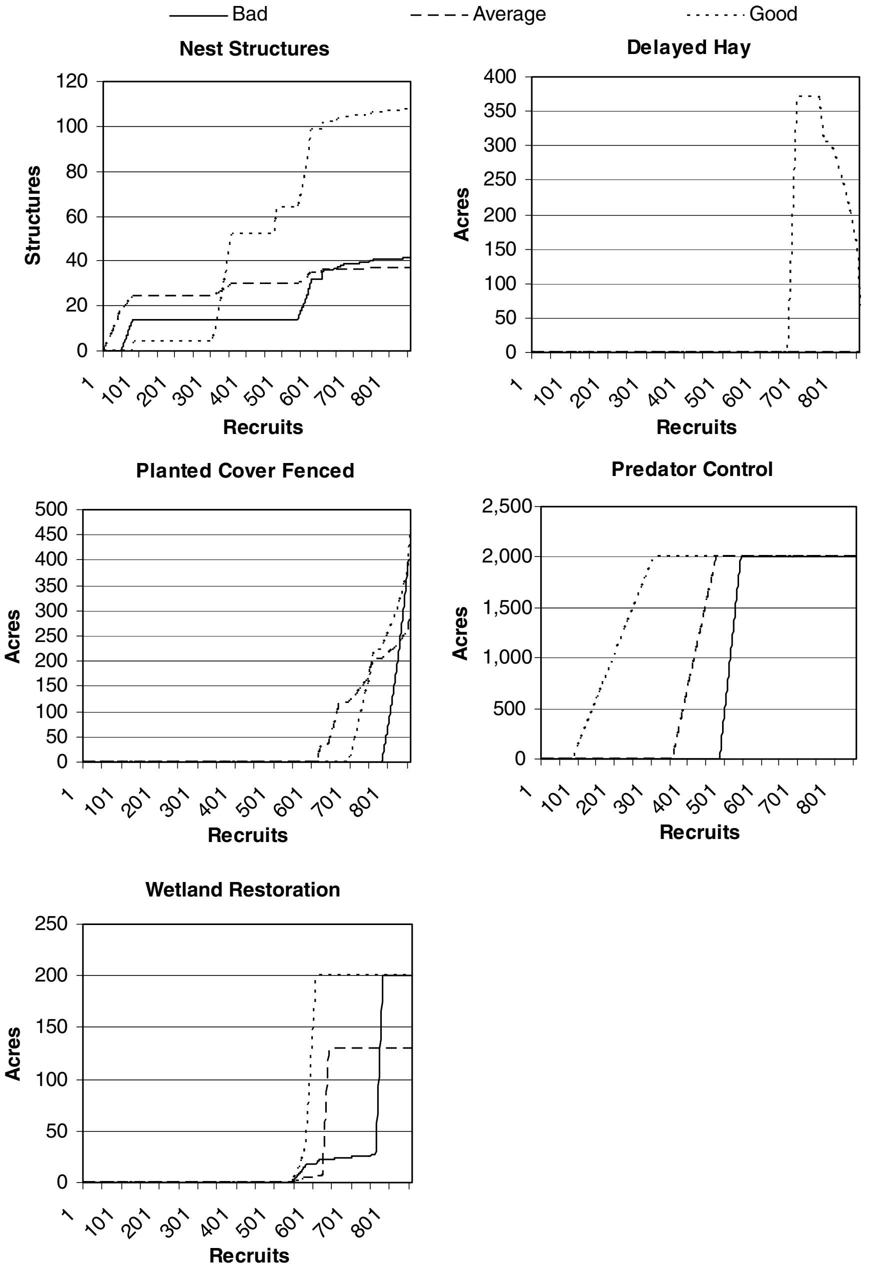

Figure 1. Cost-effective management activity levels derived in the full model

production alone. If only the relevant portion of costs were assigned to mallard production, it is possible that they would be more cost-effective than the results suggest.

We test the plausibility of this hypothesis for *CRP* by altering its cost relative to the other conservation activities. When only 5% of the original *CRP* costs are apportioned to mallard production (i.e., *CRP* costs for waterfowl conservation are \$3.50, \$1.65, and \$1.25 per acre on the poor, average and good landscapes, respectively), *CRP* still fails to be used on any landscape at any production level. If only 1% of *CRP* costs are attributed to mallard production, some *CRP* is implemented (on the average and good landscapes at low production targets). However, *CRP* is not used to achieve higher production targets, due to its inability to produce large numbers of recruits. Thus, if waterfowl managers are interested solely in

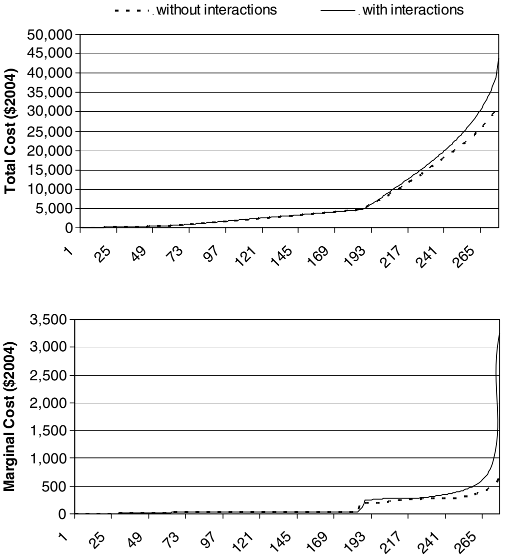

Figure 2. Comparison of conservation costs on the average landscape with and without interactions between conservation activities

mallard production, they could generate more mallard recruits by spending funds on direct conservation activities, instead of investing in broad landscape programs.

Interactions between Conservation Activities

Competition between activities is present on all landscapes, and significantly impacts the design of cost-effective conservation strategies. We confirm the importance of these negative interactions by setting the coefficient on each interaction term in (5) to zero (i.e., eliminating interaction effects), and resolving the model. The total and marginal cost functions for the average landscape, with and without interaction effects, are illustrated in figure 2. The two models are identical at low production targets because only *NS* and *PRED* are implemented, and neither model has an interaction term for these two activities. However, at higher production targets, the omission of interaction terms causes costs to be underestimated. For example, *DH* and *PC*, are implemented only when interactions terms are omitted, because the omission of competitive interactions increases their marginal productivity. This makes it optimal to substitute them in place of *WR* and *PCF*, which are more costly but relatively more productive when interaction effects are included. The implication of failing to include interactions is an underestimation of the cost of achieving a production target, increasing the likelihood that less cost-effective conservation activities are implemented.

Importance of Landscape Heterogeneity

Comparison of cost-effective conservation strategies across landscapes provides perspective on the importance of modeling landscape heterogeneity. The need for such comparisons arises because not all results are intuitively obvious. Intuition might suggest, for example, that conservation activities would be most cost-effective on the good landscape, given its abundant wetlands and breeding pairs, and relatively low opportunity cost of land. However, nesting habitat is also initially abundant on the good landscape, which reduces the marginal productivity of many conservation activities relative to other landscapes, because any introduced activity competes with existing habitat. Unless such activities create habitat with significantly higher productivity than existing habitat, mallards will simply shift between habitats with no appreciable increase in production. Activities are actually most cost-effective (for low production targets) on the average landscape, which has abundant wetlands that attract many breeding pairs, but less initial nesting habitat. Therefore, conservation activities can attract breeding pairs to nesting habitat with significantly higher productivity than the existing habitat. While the poor landscape has little initial nesting habitat, few breeding pairs and relatively high land-use costs, activities are still initiated there prior to the good landscape under cost minimizing strategies. Once again, there is little competing nesting habitat and, therefore, breeding pairs attracted to nesting structures on the poor landscape achieve relatively high success rates. Additionally, since nesting structures occupy no land area, the relatively high land-use costs on the poor landscape are not a factor.

Accounting for market and biological production variability across heterogeneous landscapes provides insights about efficient spatial targeting of conservation efforts. Figure 3 compares the marginal cost curves across landscapes for production objectives ranging from 0 to 75 recruits. Marginal costs differ significantly across landscapes and production targets. Failure to recognize differences in marginal cost across landscapes can result in a prescription of conservation activities that are efficient for one landscape type,

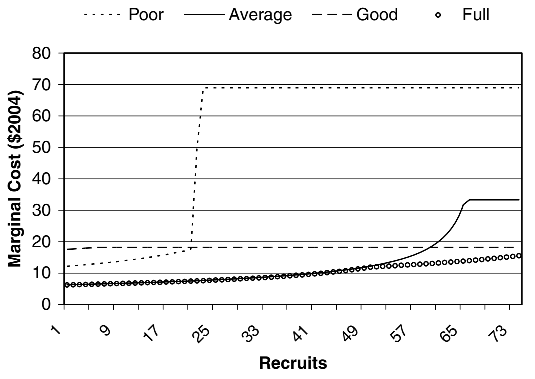

Figure 3. Comparison of marginal cost curves across the poor, average, and good landscapes

yet inefficient for another. Additionally, when multiple landscape types are available, differences in marginal costs can be exploited to improve cost-effectiveness.

The full model's solution generates a multi-landscape marginal cost curve. The marginal cost of the 75th recruit in the full model is $16, $3 less than marginal cost of the 75th recruit on the average landscape (the least expensive of the three single landscapes). The total cost of producing 75 recruits in the full model is $300 less than total cost of producing 75 recruits in the least expensive single landscape model. Optimizing simultaneously over multiple landscapes results in efficiency gains because each additional landscape introduced into the analyses relaxes resource constraints and creates additional low marginal cost activities for the manager to exploit. This is analogous to efficiency gains achieved through tradable pollution permits when marginal abatement costs differ across firms. When wildlife response or costs differ across landscapes, managers can more efficiently reach species population objectives by reallocating management activities to landscapes with the lowest marginal cost.

Conclusion

Species conservation is an important policy issue. This study examines the design of species conservation strategies using a model that reflects the following features: (*a*) the availability of a broad range of land-use and direct wildlife conservation activities, (*b*) interactions between conservation activities, and (*c*) the effect of landscape heterogeneity on market and biological production. Results indicate that failure to consider these factors can lead to an inefficient allocation of conservation resources. Direct wildlife conservation activities are found to be relatively cost-effective, suggesting that species conservation plans that only consider primary land-use activities may not identify globally cost-effective strategies. Furthermore, many direct activities require relatively little land inputs and, therefore, allow for the joint production of wildlife and marketable goods. Interactions between conservation activities play an important role in defining wildlife production and, therefore, cost-effective conservation. Landscape heterogeneity in both market and biological production influences cost-effective species conservation. When both market production (i.e., the opportunity cost of land) and biological production are linked to landscape characteristics, conservation costs can be reduced by targeting multiple landscapes simultaneously.

Although this study was applied to mallards, the insights gleaned are expected to apply to other terrestrial species for which many conservation activities exist. However, generating wildlife response functions that capture the details highlighted here requires extensive biological data. Biological simulation models are an effective tool for summarizing biological data and conducting economic analysis. Simulation models that directly link conservation effort to species population changes and capture landscape heterogeneity, however, remain limited. This raises the oft-made but seldom heeded plea for more communication between biologists and economists to develop the needed data sets and simulation models.

The model presented here examines a single species. While conservation targeted to individual species is an important policy issue (e.g., The Endangered Species Act), there is growing emphasis on managing from an ecosystem perspective. The approach developed here could be adapted in a multi-objective framework to target multiple species and an array of associated ecosystem services. The challenge is to account for interactions between conservation activities and landscape heterogeneity. With multiple objectives, the set of inclusive activities will be large, and interactions between conservation activities will become increasingly important and challenging to model. Additionally, relationships between ecosystem production and landscape heterogeneity will be difficult to estimate and will require extensive biological and land-use data. The benefits of accounting for these complexities, however, will be a marked improvement in the cost-effective conservation of valuable ecosystem services.

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