

AN ABSTRACT OF THE DISSERTATION OF

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Abstract approved:

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Wildlife managers are on the front lines of the effort to conserve wildlife and are required to do so cost-effectively. This dissertation consists of three manuscripts that integrate economics and ecology to inform cost-effective wildlife management. The first and second manuscripts focus on identifying cost-effective wildlife management plans. The third manuscript considers wildlife management under uncertainty.

The first manuscript integrates economic and ecological principles to identify cost-effective management plans. Bio-physical simulation and regression analysis are paired to approximate response functions for an important duck species, the mallard (*Anas platyrhynchos*). Response functions are then included in an economic optimization model to estimate management cost functions. Approximated response functions indicate that mallard response is non-linear due to diminishing marginal productivity and interdependence of management activities. This results in non-linear cost functions, which imply that the standard approach of treating ecological production and economic costs independently may result in inefficient management.

The second manuscript extends the first by incorporating landscape heterogeneity. The same modeling approach is replicated for three landscapes that differ in their ecological and economic productivity. This approach demonstrates that taking advantage of landscape heterogeneity can generate cost savings if managers target multiple landscapes simultaneously. Additionally, management activities that do not interfere with agriculture are found to be highly cost-effective, suggesting that common ground exists between conservationist and private landowners.

The first and second manuscripts assume that managers can predict wildlife response with certainty. The third manuscript illustrates the tradeoff between the risk and return to management when response is uncertain. Financial portfolio theory is adapted to account for diminishing marginal productivity and interdependence of management activities. An analytical model is used to determine how these properties alter the standard derivation of mean-variance efficient portfolios. This has implications for addressing uncertainty in many renewable resource contexts. Simulated data on mallards are used to apply the portfolio model to wildlife management. Results indicate that portfolio theory provides practical insights about managing wildlife under uncertainty.

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Essays on the Economics of Wildlife Management

by
Benjamin S. Rashford

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I understand that my dissertation will become part of the permanent collection of Oregon State University libraries. My signature below authorizes release of my theses to any reader upon request.

Benjamin S. Rashford, Author

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- Dr. Richard M. Adams assisted in the design and writing of manuscripts 1 and 2. Dr. Bruce Dugger assisted in the design and writing of manuscript 1. Dr. Andrew J. Plantinga assisted in the design, analysis and writing of manuscript 3.

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ESSAYS ON THE ECONOMICS OF WILDLIFE CONSERVATION

CHAPTER 1

GENERAL INTRODUCTION

Benjamin S. Rashford

Wildlife managers are on the front lines of a global effort to conserve wildlife species. The need to augment wildlife populations through management has changed the typical job description of a wildlife manager from one of setting and enforcing harvest restrictions, to one of actively producing species directly through population management and indirectly through habitat manipulation. The need to use limited wildlife management funds in a cost-effective manner has also increased. Economic considerations, as a result, more regularly influence wildlife management decisions. As Nobe (1971, p24) states, “It appears that economic input is moving rapidly from the role of ‘window dressing’ to an integral part of policy formulation and administration of game and fish resources.”

In contrast to the production-oriented aspect of modern wildlife management, economic analyses have largely emphasized demand for wildlife resources and non-market valuation of these resources (Matulich and Adams 1987). These demand-side economic analyses are more relevant to policymakers than to wildlife managers working on the ground to affect specific wildlife populations. Wildlife managers would benefit from additional production-oriented economic analyses. Specifically, a need exists for analyses that examine the joint role of biological response relationships and economic costs in determining the least-cost approach for achieving management objectives. Not only would production analyses improve wildlife managers’ ability to manage wildlife in a cost-effective manner, they would also, in combination with demand-side analyses, allow a complete analysis of the benefits and costs of alternative wildlife management projects and more efficient allocation of limited conservation resources. This dissertation

consists of three manuscripts that integrate economics and ecology to inform wildlife management decisions. Each manuscript is summarized next.

The first manuscript (Chapter 2), entitled *The Economics of Wildlife Production*, demonstrates the joint manner in which ecology and economics interact to determine cost-effective wildlife management. The manuscript's intended audience is the wildlife biology and management community. The methods used in this paper consist of three steps. First, a biological simulation model is used to simulate the effect of eight waterfowl management activities on population response of mallards (*Anas platyrhynchos*), a waterfowl species with large recreational value. Data from the simulation model are then used to approximate a continuous response function, which predicts mallard response to varying levels and combinations of management activities. Lastly, the response function is embedded within an economic optimization model to identify least-cost management strategies for a range of population objectives.

The approximated response function indicates that mallard response exhibits diminishing marginal productivity for several management activities, and that some management activities are technically competitive. This non-linearity of response, when embedded in the economic optimization model, generates non-linear mallard management cost functions. Results from the analysis demonstrate three points that have important implications for wildlife management: 1) biological response and economic cost jointly determine least-cost management plans, 2) non-linearity of the biological response function should be modeled explicitly to identify cost-effective management plans, and 3) the least-cost management plan depends on the chosen population objective.

The second manuscript (Chapter 3), entitled *Improving the Cost-Effectiveness of Ecosystem Management: An Application to Waterfowl Production*, extends the first manuscript in the following ways: 1) it considers the effect of landscape heterogeneity on the solution to the least-cost management problem, and 2) it examines and discusses policy implications of the relative cost-effectiveness of primary land-use conservation activities versus direct wildlife conservation activities in a landscape dominated by agricultural production. Mallard response to management is approximated using the method described in the first manuscript; however, in this manuscript, it is approximated for three alternative landscapes, which differ in their ecological and agricultural productivity, and for a single composite landscape. Biological response and economic costs vary across the landscapes, as a result of their heterogeneity.

Results indicate that heterogeneity in ecological and agricultural production across landscapes influences the cost of waterfowl management profoundly. Specifically, the simultaneous application of management activities to multiple landscapes, rather than to a single landscape, reduces the cost of achieving population objectives. Results also indicate that direct wildlife conservation activities are relatively more cost-effective than primary land-use conservation activities. This suggests that species conservation plans that only consider primary land-use activities may not be cost-effective. It also suggests that, because many direct conservation activities require relatively little land, wildlife and agricultural production do not necessarily preclude one another.

The third manuscript (Chapter 4), entitled *Towards a Portfolio Theory for Renewable Resource Management*, contributes to the growing literature on renewable resource management under uncertainty. It demonstrates that standard financial portfolio

theory, in some cases, cannot be used to analyze risk-return tradeoffs in a renewable resource management context without violating some of the model's assumptions. The standard portfolio model's assumptions imply that the rate of return on a management activity is constant regardless of the level at which it is applied, and regardless of the intensity levels at which other management activities are simultaneously applied. Physical response of a renewable resource to management activities, in contrast, may exhibit diminishing or increasing marginal productivity to increasing management activity levels, as well as interdependence between multiple activities.

We adjust the standard portfolio model to account for diminishing or increasing marginal productivity and activity interdependence, and demonstrate that the resulting set of mean-variance efficient portfolios may differ in important ways from those identified under the assumptions of the standard model. The renewable resource portfolio model, in fact, generates results that are not possible in the standard portfolio model. The application of portfolio theory to renewable resources still provides useful insights about risk-return tradeoffs; however, the application cannot, in some cases, be applied directly as previous studies have suggested.

CHAPTER 2**THE ECONOMICS OF WILDLIFE PRODUCTION**

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ABSTRACT

Faced with limited budgets and pressure to expend public resources cost-effectively, a wildlife manager must determine the set of management activities that achieves their management objective at least cost. This requires the use of both ecological and economic principles to make management decisions. We use data from a biological simulation model of breeding waterfowl to estimate a continuous, non-linear response function, which is then included in an economic optimization model. Non-linearity of the response function is attributable to density-dependence and interactions between jointly applied management activities. The bioeconomic model is then used to solve a waterfowl manager's least-cost management problem. Model results demonstrate the following: 1) both biological response and economic cost determine the least-cost management plan, 2) non-linearity of the biological response function should be modeled explicitly to identify cost-effective management plans, and 3) the least-cost management plan depends on the chosen population objective. In summary, it can be challenging to identify cost-effective management plans when biological response to management is non-linear. There is no simple rule-of-thumb; no single management activity is cost-effective across all population objectives.

INTRODUCTION

The wildlife manager's historical job of setting and enforcing harvest restrictions has evolved, and now includes an emphasis on the maintenance of species populations through direct population management and indirect habitat manipulation. The job of the wildlife manager, as a result, has become more complex. Faced with limited budgets and pressure to expend public resources cost-effectively, managers must determine the set of management activities that generates the desired outcome at least cost. This requires that managers understand both the ecological and economic tradeoffs of management alternatives and, therefore, integrate both ecological and economic principles into management decisions.

Wildlife managers striving to achieve cost-effective management need two decision aids: 1) a model that accurately predicts wildlife population response to various management activities (Hurley 1986), and 2) a model that relates management activities and the resulting population response to economic costs. Ecologists have expended considerable effort developing population response models (Verner, Morrison and Ralph 1986), and economists have well developed theories and models of the costs of production. Unfortunately, while the need to integrate ecological and economic wildlife research has long been recognized (Pearse and Bowden 1968; Matulich and Adams 1987; O'Neill, Kahn and Russell 1998; Hughey, Cullen and Moran 2003); joint research efforts that produce practical insights for applied wildlife management remain scarce.

Two key economic principles can help managers evaluate the economic and ecological tradeoffs of applying alternative management activities. First, the cost-effectiveness of a management activity is largely determined by the activities marginal

product, which measures the increase in output that can be expected from a marginal increase in the level (e.g. number of hectares) of the activity. Many activities, due to density dependence, are likely to exhibit diminishing marginal productivity; that is, the output produced from incrementally increasing the activity level increases at a decreasing rate. It is important to identify activities that exhibit diminishing marginal productivity because the cost of output (e.g. change in population) produced using these activities will increase at an increasing rate. Thus, a manager choosing among several activities may severely overestimate potential output and underestimate costs if diminishing marginal productivity is not accounted for.

The marginal product of a management activity describes how output changes when the level of a single activity is adjusted. Managers, however, typically apply multiple management activities simultaneously; therefore, they must also understand how productivity and costs change when activities are combined. The economic concept of technical interdependence can be used for this purpose. Technical interdependence describes how the marginal productivity of one activity changes when a second activity is implemented simultaneously. There are three types of activity interdependence: independent, complementary and competitive. Activities are independent if combining them has no effect on either's marginal productivity. When activities are complementary, combining them increases the marginal productivity of each. Thus, complementary activities are more productive in combination than they are alone and, therefore, combining them is more likely to be cost-effective. Lastly, when activities are competitive, combining them reduces the marginal productivity of each and, similarly, may reduce cost-effectiveness.

Although these concepts are rarely discussed in the wildlife management literature, ecologists have not ignored economic costs entirely. Many studies on wildlife response include estimates of economic costs (Lokemoen, et al. 1982; Waer, Stribling and Causey 1997; Stephens, et al. 1998; Nugent and Choquenot 2004). Their focus, however, is typically on response to a single management activity, often applied at a single level of intensity (e.g. Zenner, LaGrange and Hancock 1992; Cowardin, et al. 1998; Garrettson and Rohwer 2001). More realistically, managers must understand response to (and the cost of) various intensity levels of a single management activity, and to combinations of management activities. No single study of response, due to the prohibitive cost and time-scale needed, provides a sufficiently complete data set to fulfill the manager's needs in this regard.

Studies that do focus on the cost-effectiveness of alternative management plans (see e.g. Lokemoen 1984; U.S. Fish and Wildlife Service 1996) must therefore rely on, and extrapolate from, multiple disparate data sources. This approach requires the researcher to make many simplifying assumptions about biological response, including linearity in response (and cost) across varying activity levels, and independence of jointly applied activities. These assumptions, however, are unlikely to hold for biological response due to ecological factors, such as density dependence. As Kenneth Boulding wrote (Rodes and Odell 1998, p. 61):

“One problem that is an ecological upsetter
Is that if anything is good, then more of it is better,
And this misunderstanding sets us very, very wrong,
For no relation in the world is linear for long.”

Integrating either of these assumptions into an economic optimization model can lead the researcher to identify a set of management activities that appear cost-effective, but in reality are not (Matulich and Adams 1987; Cowardin, Shaffer and Kraft 1995). Cowardin, Shaffer and Kraft (1995), using simulated response data on mallards, demonstrate that two management activities, cropland retirement and fenced cover, exhibit diminishing marginal productivity and compete with one another (i.e. are interdependent). As a consequence of the resulting non-linear response, the authors note that using average response measures (e.g. output/ha) scaled up to larger application levels can greatly overestimate the benefits of management. Though Cowardin, Shaffer and Kraft (1995) discuss economic implications using benefit-cost ratios, they conclude their paper by calling for “more cooperative work with economists to give a sound economic basis for management decisions” (p. 54). This paper is an attempt to answer their call.

Specifically, we develop an integrated bioeconomic model that extends the previous literature in two ways: 1) by examining the properties of response across a large suite of management activities, which allows for more complex response behavior and, 2) by explicitly integrating response with economic optimization to highlight the joint nature of the ecological and economic consequences of alternative management plans. We use results from the model to demonstrate the following points, which have important implications for applied wildlife management: 1) both biological response and economic cost determine the least-cost management plan, 2) simplifying assumptions about the biological response function’s properties can lead to management plans that are not cost-effective, and 3) the least-cost management plan depends on the chosen production target.

METHODS

Simulating Mallard Response Data

We simulated mallard response data using the Mallard Productivity Model (version 171.027beta). The MM is a stochastic simulation model designed to predict the impact of management activities on the breeding success of mallards using known relationships between landscape characteristics and waterfowl ecology (Johnson, Sparling and Cowardin 1987). The modeler first inputs details about the landscape they wish to model, including wetland conditions and land use and land cover information. The MM then uses wetland type and abundance to predict the number of breeding pairs expected to nest in the landscape (Cowardin, et al. 1983). The females¹ of each breeding pair are then ushered through stages of the breeding cycle, such as nest site selection and brood rearing. The fate of each virtual nest is tracked through the breeding season, and population characteristics are calculated (see Johnson, Sparling and Cowardin (1987) for a full model description).

We ran simulations for an 809-hectare landscape that is characteristic of agricultural land in the Prairie Pothole Region (Table 2.1)². We simulated mallard response to eight management activities that are representative of those available for breeding waterfowl management³: (1) cropland retirement (CR), (2) conservation tillage (CT), (3) delayed haying (DH), (4) planted cover (PC), (5) planted cover fenced, (6) artificial nest structures (NS), (7) predator control (PRED) and (8) restoration of semi-permanent wetlands (WR). Implementation of management activities was simulated by adjusting model parameters and manipulating landscape characteristics. For example,

cropland retirement is simulated by shifting acreage from the cropland habitat category in the MM, to the Cropland Retirement program habitat category. Activities were simulated at multiple levels (e.g. 100, 200, ..., 1300 acres of CR), in isolation and in combination, to capture substitution possibilities and interdependence between activities. A total of 376 unique combinations of management activities were simulated. Response, which is determined for each individual simulation run, is measured by the number of recruits.

Table 2.1. Land Use Characteristics of the Simulated Landscape

	Hectares (<i>% of total area</i>)
Grain Crops	784 (69.3)
Grassland	77 (9.5)
Hayland	54 (6.7)
CRP	0 (0)
Seasonal Wetland	60 (7.4)
Semi-Permanent Wetland	40 (4.9)
Temporary Wetland	4 (0.5)
Permanent Wetland	1 (0.1)
Other	13 (1.6)

The MM is stochastic; therefore, the recruits produced from an individual simulation run represent one observation of a random process. One could measure the effect of a particular management plan using the result of a single run, but then response could be misrepresented by a “bad” draw. We instead measured response using the average number of recruits produced from multiple runs. We determined the appropriate number of simulations by iteratively simulating response and calculating a 95%

confidence interval around the estimated mean at each iteration (see Ross 2002). We stopped iterating when the confidence interval converged (i.e. became stable). We determined that 300 replications of each unique management plan were sufficient; the resulting 95% confidence interval is +/- 1.5 recruits.

Approximating Continuous Mallard Response Functions

The simulations generate a large set of response data for various levels and combinations of management activities. Next, we used regression analysis to estimate from this data set a continuous response surface. The estimation serves two purposes. It allows us to statistically examine the physical properties of response⁴ (e.g. statistically test for diminishing marginal productivity and interdependence). It also generates a single continuous function that can be used to approximate response for any possible management plan, which serves as the constraint in the cost-minimization problem that determines cost-effective management plans. In lieu of approximating a continuous response function, cost-effective management plans could be identified by simulating every possible combination of management activities and calculating the total cost for each. The number of possible management plans is exhaustively large; therefore, this direct approach is impractical (see Wu and Babcock 1999).

We use a second-order quadratic approximation to estimate the response function because it allows us to capture diminishing marginal productivity and activity interdependence. The estimable equation takes the form:

$$(2) \quad y = \alpha + \sum_i^k (\beta_{1i}x_i + \beta_{2i}x_i^2) + \sum_i^k \sum_j^k \delta_{ij}x_ix_j + \varepsilon ,$$

where y is the simulated mean number of recruits, x_i is the level of management activity i , ε is a random disturbance term, which captures determinants of recruits that are not observed, and α , β_{1i} , β_{2i} , and δ_{ij} are response parameters to be estimated. The unit of measurement for all activities is hectares, except for NS, which is measured as the number of nesting structures installed. Parameters β_{2i} and δ_{ij} capture diminishing marginal productivity and factor interdependence, respectively. Management activity i exhibits diminishing marginal productivity if $\beta_{2i} < 0$. Management activities i and j are independent, complementary or competitive if $\delta_{ij} = 0$, $\delta_{ij} > 0$, or $\delta_{ij} < 0$, respectively.

While it is possible to estimate second-order and interaction terms for every variable, it is not advisable due to the potential for inefficient parameter estimates (see Greene 2000). We consulted the literature and management professionals to identify management activities that are expected, theoretically, to exhibit diminishing marginal productivity and interdependence. Research suggests diminishing marginal productivity for management activities that directly increase mallard nesting habitat (Cowardin, Shaffer and Kraft 1995). This is expected because, for a relatively fixed number of breeding pairs in the management area, recruits produced from nesting habitat is expected to increase at a decreasing rate as the quantity of nesting habitat exceeds the needs of the available breeding pairs. We therefore used second order terms for the following activities, all of which create nesting habitat directly: CR, DH, NT, PC, PCF and NS.

Certain management activities are also expected to exhibit competitive and complementary interdependence. First, habitats that are very similar in their attractiveness to breeding waterfowl are likely to compete for the limited number of

breeding pairs. Cropland retirement, planted cover and planted cover fenced, for example, create nesting cover and therefore may be competitive. This implies that increasing one of these activities may decrease the marginal response of competing activities by attracting breeding pairs away from the activity. We therefore included interaction terms between these activities. Second, nest depredation research suggests that large blocks of intact, dense nesting cover reduce nest densities, and thereby reduce the probability that predators locate and depredate nests (Sovada, et al. 2000). Predator control is therefore expected to compete with activities that create extensive nesting cover. As a result, we included interaction terms between predator control and the following activities: CR, NT, DH, and PC. Lastly, wetland abundance affects the number of breeding pairs on a landscape; therefore, wetland restoration is expected to complement other activities by increasing the number of breeding pairs present to utilize them. Interaction terms between wetland restoration and all other activities were therefore included. We used ordinary least squares to estimate model parameters on the resulting data set; the software package Limdep NLOGIT 3.0 was used (Greene 2003).

Regression with the full variable set discussed above produced insignificant coefficients estimates for CR^2 , DH^2 , NT^2 and an F-test for their joint significance failed to reject the null hypothesis that they are all equal to zero. Additionally, the interaction effects between PC and PCF, and all interaction effects including WR were not significant. The insignificance of these coefficients likely results from the size of the study area used for the simulations. In retrospect, the extensive activities, CR, DH and NT, are not likely to exhibit diminishing marginal productivity unless applied at a scale larger than our simulated area⁵. Similarly, PC and PCF must be applied simultaneously

at high levels to interact. Due to the relatively small size of our simulated area, the data set does not include sufficient observation of these activities in combination at high intensity levels. Similarly, WR must be applied at high levels to generate sufficient increases in the number of breeding pairs to interact with other management activities. Again, the size of our simulated landscape is too restrictive to generate the expected interactions. Since the insignificant variables above are found to be irrelevant at the scale of our analysis, we omit them from the final regression model.

Optimization: Estimating Mallard Management Cost Functions

The final step was to embed the approximated mallard response function within an economic optimization model to determine mallard management cost functions. This requires estimates of management costs (table 2.2), which we collected from the management literature and through correspondence with waterfowl managers. The optimization model, which minimizes the cost of achieving a target level of recruits (y), has the following form:

$$(1) \quad \min_{x \geq 0} \sum_{i=1}^n w_i x_i$$

subject to

$$f(x_1, \dots, x_n) \geq y,$$

where $i = 1, \dots, n$ indexes the available management activities, x_i is the level of management activity i (e.g. hectares), and w_i is the per unit cost of activity i . Thus, for a chosen target level of recruits, the cost minimization problems identifies the set of management activities that can achieve the target level of recruits as least cost.

Two restrictions are imposed on the optimization model to ensure consistency with the ecological literature and the simulated data. First, ecological studies indicate that predator re-distribution and dispersal limit the effectiveness of lethal control on small blocks (Garrettson and Rohwer 2001). This suggests that PRED should not be treated as a continuous variable, thus the PRED management activity is treated as a discrete activity (i.e. apply or do not apply predator control to the entire management area). Second, nesting structures are constrained to one structure per acre of wetland to be consistent with densities observed in the literature (Bishop and Barrat 1970; Marcy 1986).

Table 2.2. Annualized Management Activity Costs⁶

Management Activity	Units	Cost/unit (\$2004)
Cropland Retirement (CR)	Hectares	81.54
Conservation Tillage (CT)	Hectares	37.07
Delayed Hay (DH)	Hectares	61.78
Nesting Structures (NS)	Structures	22.90
Planted Cover (PC)	Hectares	43.24
Planted Cover Fenced (PCF)	Hectares	272.31
Predator Control (PRED)	Hectares	4.94
Wetland Restoration (WR)	Hectares	93.90

In (1), $\sum_{i=1}^n w_i x_i$ is the management expenditure associated with a combination of management activities. The response function approximated from the simulated data is denoted $f(x_1, \dots, x_n)$. The solution to (1) is the set of management activities (x_1^*, \dots, x_n^*) that achieves y at least cost. This set is the cost-effective management

prescription, because it achieves y at less total cost, denoted $\sum_{i=1}^n w_i x_i^*$, than any other combination of management activities.

The total cost function, $C(w,y)$, which reports the minimum total cost of achieving any recruit target, is formed by solving the CMP for a range of output levels. From $C(w,y)$, we derive the marginal cost function $\left(MC = \frac{\partial C(w,y)}{\partial y} \right)$ and the average cost function $\left(AC = \frac{C(w,y)}{y} \right)$. The MC function describes the change in the cost function given a small change in the recruit target y . In fact, because it is derived from the cost function (which reports least-cost management plans), the MC function represents the minimum amount by which cost must increase to produce an additional recruit. The average cost function describes, for every point on $C(w,y)$, the cost per recruit.

RESULTS

Continuous Mallard Response Functions

Parameter estimates for the approximated response surface are reported in table 2.3. The simulated landscape with no management activities produces approximately 18 recruits on average, as indicated by the constant term. The NS activity has the largest coefficient, which implies that one NS produces more recruits than one unit of any other management activity. CT has the smallest coefficient followed by CR. Coefficients on NS^2 , PC^2 , and PCF^2 are negative and significant, which indicates these activities exhibit diminishing marginal productivity. Parameter estimates for interaction terms (e.g.

PRED·CR) are significant and negative for many terms, indicating that these activities are competitive.

Table 2.3. Parameter Estimates of Mallard Response Function

Variable	Estimate
Constant	17.82 (1.56)***
CR	0.042 (0.007)***
DH	0.314 (0.062)***
CT	0.025 (0.007)***
NS	3.69 (0.227)***
PC	0.203 (0.032)***
PCF	1.1 (0.059)***
PRED	0.16 (0.002)***
WR	0.341 (0.04)***
NS ²	-0.046 (0.006)***
PC ²	-0.0004 (0.0001)***
PCF ²	-0.0043 (0.0004)***
PRED·CR	-0.0001 (0.00001)***
PRED·DH	-0.0012 (0.0002)***
PRED·CT	-0.0002 (0.00006)***
PRED·PC	-0.0004 (0.00006)***
CR·PC	-0.00003 (0.0001)
CR·PCF	-0.0012 (0.0002)***
F[n,k]	304.77
R ²	0.935
Adj. R ²	0.932

Key: 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

Note: *** indicates significance at the 1% levels. Standard errors are in parentheses.

Optimization: Mallard Management Cost Functions

Application of the eight management strategies on our virtual landscape can produce a maximum of 277 recruits, i.e. with all activities applied at their maximum levels subject to land-use constraints. Figure 2.1 presents the least-cost management activities levels for every possible level of recruits. The cost-effective mix of management activities derived in the model can be visualized by examining the activity level graphs vertically. For example, 250 recruits are produced cost-effectively by employing 40 NS, 50 ha of WR, 30 ha of PCF and 800 ha of PRED. At low recruit targets (up to 65 recruits) it is cost-effective to only apply increasing numbers of NS. WR and DH are implemented next. Once the recruit target is 84 recruits it becomes cost-effective to initiate PRED. At high recruit targets (195 or greater) PCF is initiated. The other management activities (CR, CT and PC) are never initiated in the cost-effective management plan for any recruit level.

The cost functions behave as expected (figure 2.2); marginal cost increases at an increasing rate, due to the diminishing marginal productivity of many management activities (figure 2.2b). The first recruit produced using only NS has a marginal cost of \$6.23, while the marginal cost of the last recruit (277th) produced with a combination of NS, WR, PCF and PRED (see figure 2.2) has a marginal cost of \$819.

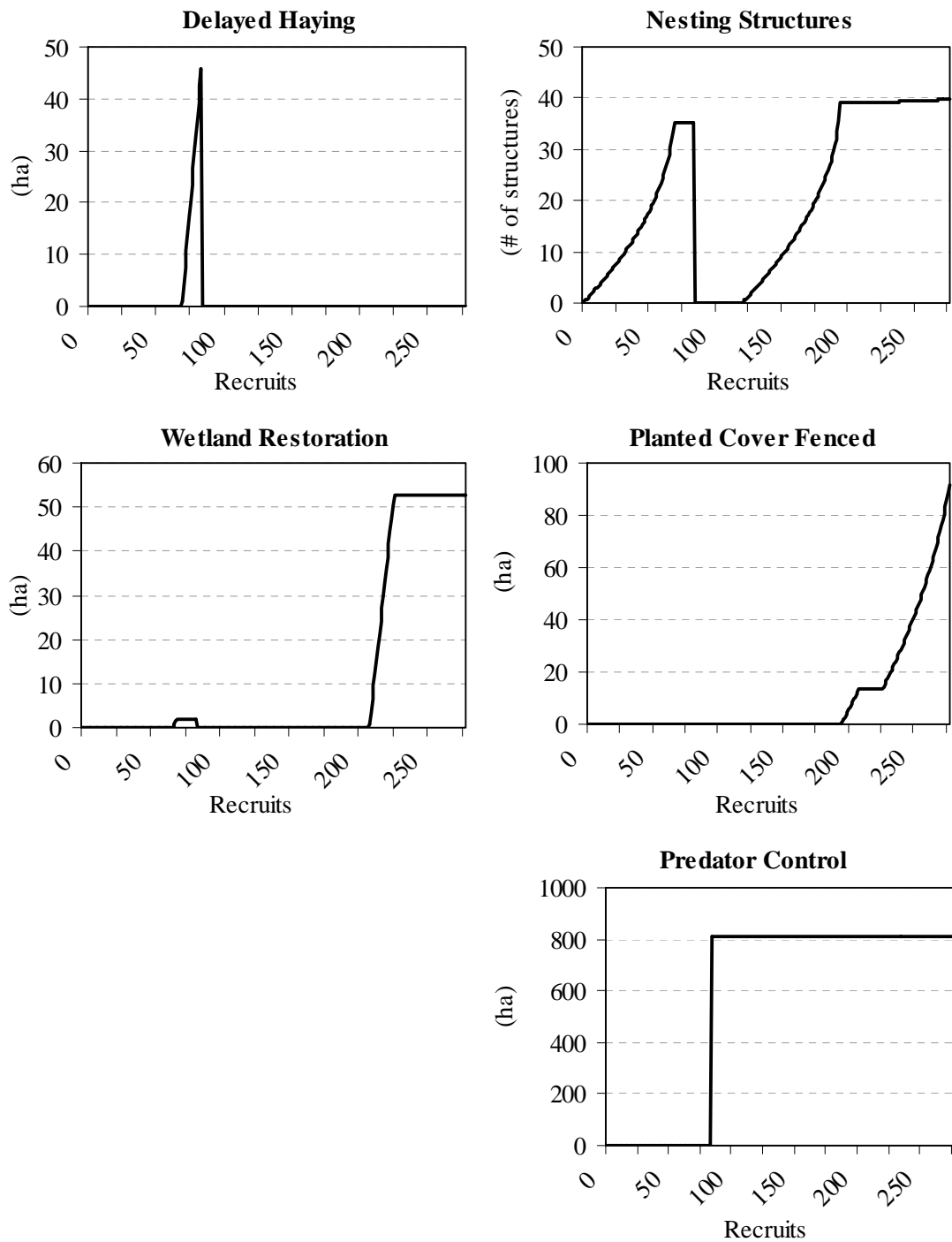


Figure 2.1. Management activity levels applied under cost-effective management. E.g. To produce 200 recruits at least-cost, use 30 nest structures and 2000 acres of predator control.

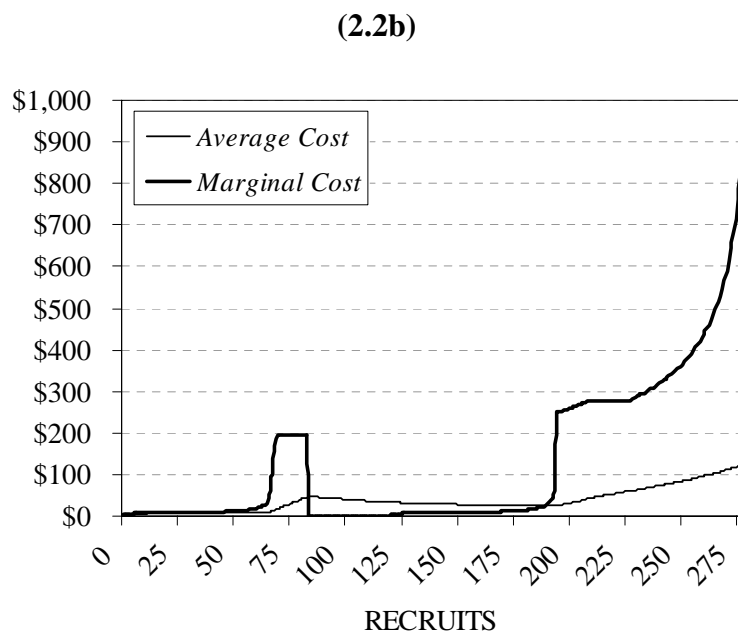
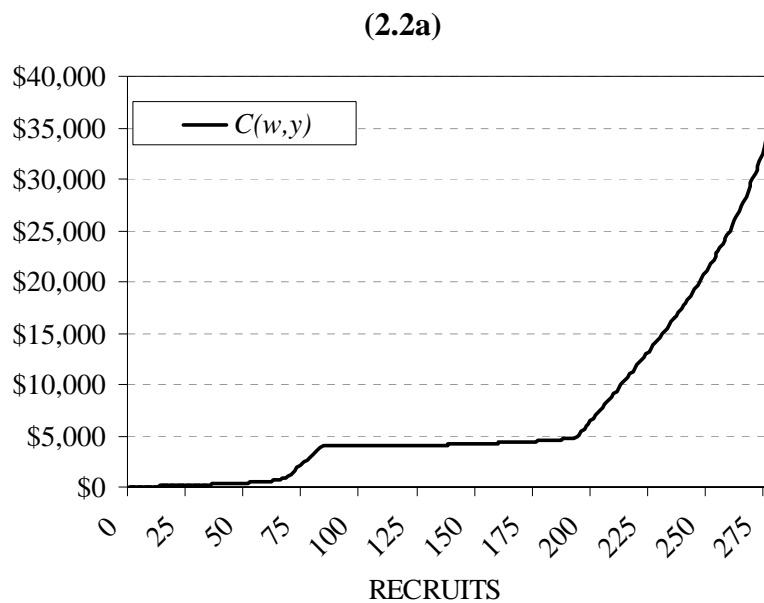


Figure 2.2. Cost function for mallard production. (2.2a) Total cost function ($C(w,y)$). (2.2b) Average cost function and marginal cost function.

DISCUSSION

The integrated bioeconomic model indicates that ecological response and costs jointly determine cost-effective management plans. That is, to identify cost-effective activities managers cannot simply observe productivity on one hand, and then consider cost on the other; it is the interplay between relative marginal productivities of alternative activities and their relative prices that determines relative cost-effectiveness. For example, the mallard response function indicates that planted cover fenced is a highly productive management activity. This activity, however, is not implemented in the cost-effective solution until the production target exceeds 194 recruits (70% of the total production possible on the landscape) due to the relatively high per-unit cost of planted cover fenced. Conversely, conservation tillage is one of the least expensive activities examined, but never enters the cost-effective solution due to its relatively low marginal productivity.

To illustrate the interplay between productivity and cost in determining cost-effective strategies, consider the isoquant-isocost mapping for PCF and CT in figure 2.3. Isoquants are level curves of the approximated response function, i.e. an isoquant depicts all the combinations of two activities that can produce a fixed level of recruits. Similarly, isocost lines map the combinations of the two activities that can be initiated for a fixed level of total cost. Cost-effective combinations of the two activities correspond to tangencies (interior or corner) between the isoquant for the targeted recruit level and the lowest isocost level attainable. Thus, in figure 2.3, a target of 10 recruits can be achieved using approximately 400 ha of CT for a total cost of \$15,000 (point A). However, since PCF is relatively more productive, the target of 10 recruits can be produced using

approximately 10 ha of PCF for a total cost of \$2,646. Thus, despite PCF being more than seven times as costly/ha than CT, it is cost-effective to implement PCF because it is highly productive relative to CT.

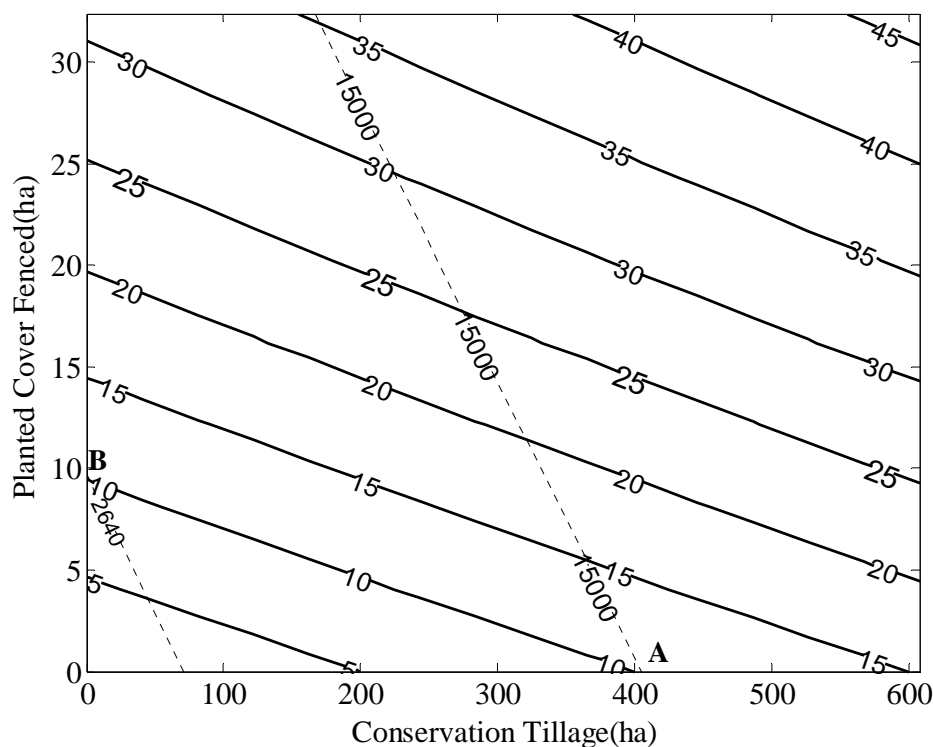


Figure 2.3. Response isoquants and isocost lines between planted cover fenced and conservation tillage. Solid lines are isoquants with markers indicating the recruit level. Dashed lines are isocost lines with markers indicating the total cost level.

These examples highlight a key result of the integrated modeling approach.

Managers, when searching for cost-effective alternatives, should not overlook management activities just because they are highly costly to apply, nor should they quickly apply activities that are inexpensive. The true cost of an activity in the production of wildlife outputs depends on both the activities cost and its productivity.

This emphasizes the importance of detailed response models in determining cost-

effective management plans. Response models that oversimplify the response process can lead to cost-ineffective management plans by providing misleading estimates of the incremental cost and production of management activities.

The approximated response function indicates that mallard response is non-linear and, as a result, the mallard management cost functions are also nonlinear. Thus, our results confirm those of Cowardin, Shaffer and Kraft (1995) who warned that expressing the results of management as recruits/ha then scaling up by multiplying recruits/ha by the number of hectares to be managed may greatly overestimate the number of recruits to be produced. To this we add that overestimating the production from management activities will lead to underestimating the cost of achieving a given production level and, therefore, lead to the cost-inefficient application of management. This point is best illustrated by examining the marginal cost (MC) and average cost (AC) functions derived for mallard management. Figure 2.4 compares the MC and AC functions for producing 1 – 68 recruits derived from the true model with non-linear response, to the MC and AC functions derived assuming linear response. To derive the latter, we resolve the cost-minimization model with all second order and interaction terms in the response function set equal to zero.

In the absence of diminishing marginal productivity, the linear model results in a constant MC of \$6.21, and, due to the imposed linearity, this is equal to the AC. In contrast, when the non-linearity inherent to response is modeled, the MC increases at an increasing rate. This specifically occurs due to the diminishing marginal productivity of NS, which is the first activity applied. Because of diminishing marginal productivity, each additional NS produces fewer recruits than the last, and therefore, each additional

recruit costs incrementally more to produce. Furthermore, the AC is less than the MC because the AC is “averaged” over all the recruits produced. As a result, when MC increases rapidly, the AC only increases slightly because it averages the high cost of the last recruit produced over all the recruits produced to that point.

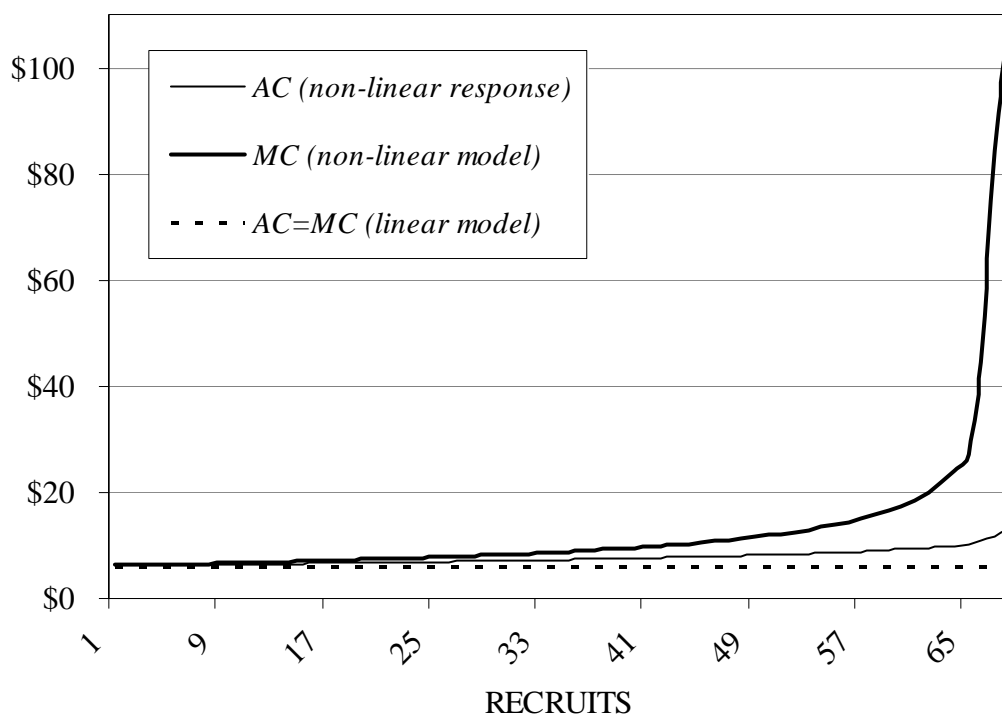


Figure 2.4. Comparison between marginal cost and average cost functions derived with the true non-linear response model and a model assuming linear response.

Studies which assume linear response and report AC estimates for observed production levels may, therefore, send the wrong message to managers about the cost-effectiveness of management alternatives. For example, the linear model predicts that 68 recruits can be produced using 28 NS with an AC of \$6.21. When diminishing marginal productivity is considered, producing 68 recruits actually requires 34 NS with an AC of

\$13.26 and the MC of the 68th recruit is \$108. Thus, the manager would tend to overestimate the productivity of nesting structures and, as a result, underestimate the true cost (the MC) of increasing the number of recruits.

Furthermore studies of individual management activities that assume linear response, and therefore a constant $MC = AC$, leave managers to assume that the observed production level can be scaled up or down, and the AC will remain the same. If this scaling is done around a point of large diminishing returns, the resulting AC may severely underestimate the true cost of production. For example, a manager might assume that the 34 recruits produced in the linear model with 11 NS with an AC of \$6.21 could simply be doubled to produce 68 recruits without impacting the AC. However, the production from 22 NS, accounting for diminishing returns, is only 57 recruits. Producing 68 recruits actually requires 34 NS and the true MC of the last recruit is \$108. These examples emphasize the difference between MC and AC. MC is the appropriate measure of the incremental cost of production. While it may be easy to justify, for example, spending \$13.26 on average to produce 68 recruits, it would be more difficult to justify spending \$108 to produce one more recruit, as is actually the case as a result of diminishing marginal productivity.

Interdependence between management activities can have similar impacts on cost-effectiveness as diminishing marginal productivity. While we identified several activities that were competitive, such interdependence did not play a large role in determining cost-effective management plans for mallards. CR, which is competitive with PRED, PC and PCF, was not cost-effective to apply due to its relatively low marginal productivity and relatively high cost per acre. Similarly, PC and CT, which

compete with PRED, were never cost effective to apply. Thus, the competitive interdependence between these activities was not a factor. DH, which interacts with PRED, is applied at relatively low production levels and is never applied in conjunction with PRED. This result, however, is due to the discreteness of PRED, rather than the competitive interdependence between the two. Nonetheless, in other applications, such as for a different starting landscape or different targeted species, factor interdependence may be important to consider. When ongoing research reveals more about the relationship between landscape characteristics, wetland abundance and waterfowl breeding densities, for example, we may discover complementary interdependence between activities, such as WR and CR. These interdependencies could have significant implications for the design of cost-effective management plans and should be kept in mind when designing experiments and collecting biological data.

The identification of cost-effective management plans also depends on the level of production targeted. Activities that are cost-effective for low levels of production may not be cost effective for high levels of production. This is particularly true if some management activities are discrete, such as PRED in our model. The discrete, all or nothing, nature of PRED causes a corresponding discrete change in the cost-effective management plan. It is not cost-effective to conduct predator control across the entire landscape to achieve low production targets (below 83 recruits) because combinations of other activities (NS, DH and WR) are relatively more cost-effective. However, if the objective were to produce greater than 83 recruits, it becomes cost-effective to initiate PRED and “turn off” all other activities (from 84 – 120 recruits PRED is applied alone, see figure 2.1). This implies that, for some activities like PRED, adding them to a

landscape that already contains other management activities may constitute redundant management and an inefficient use of management resources.

Thus, the recruit level targeted largely determines the cost-effective strategy since the relative productivity and relative cost of activities depends on the level at which they are applied. Managers rarely target a given production level, however, due to the difficulty of measuring output in the field, broad objectives, such as “increasing the population”, may be too general and therefore lead to inefficient resource use. Broad objectives are not specific enough to account for the incremental costs associated with “increasing populations”, nor the discrete nature in which cost-effective strategies should change to “increase the population”. It may in many cases, therefore, be more effective to target lower production levels cost-effectively, and use the cost savings in other management areas or to pursue other objectives. Even if redistributing monies is not practical, such as when a manager of a refuge has a fixed budget that must be spent, the cost-function indicates the maximum production that can be achieved for that production level. The manager should, therefore, consider the full range of affordable management plans, *a priori*, instead of implementing individual management activities incrementally, which may lead to redundant management practices.

Finally, the cost-function approach we used does not directly determine which production level should be targeted. The “optimal” level of production, from a social perspective, depends on the “value” society places on mallards. Socially efficient mallard management would require that the marginal cost of mallard production be equal to the marginal value society places on mallards. Determining this level requires estimates of these non-market values, which can be difficult to obtain and controversial.

The cost function, however, can provide significant insight for managers and policy makers on this matter. The marginal cost function depicts the minimum incremental expenditure necessary to produce each incremental recruit. For any production level chosen, therefore, the associated marginal cost indicates the perceived value to the last recruit produced. If available, managers and policy-makers can use this cost to reach an informed judgment on the best use of public resources.

MANAGEMENT AND RESEARCH IMPLICATIONS

Federal and private agencies invest billions of dollars annually in wildlife management programs. Managers must decide how to allocate these funds, but they lack sufficient guidance on the relative cost-effectiveness of management alternatives. Providing this guidance requires detailed biological response models and the cooperative effort of ecologists and economists. When biological response to management is non-linear, we have shown that identifying cost-effective management plans can be complex. There are no easy answers; there is no single management activity that is “best.” Cost-effective management plans depend on the nature of response, and the level of production the manager hopes to achieve. Unless response is reasonably approximated by a linear function, the production and cost observed in studies of individual management activities applied at a single level will not provide the manager with sufficient information to evaluate the application of multiple management activities, applied at alternative intensity levels.

Unfortunately, collecting response data capable of assessing the full range of tradeoffs across multiple management activities may be prohibitively expensive.

Simulation models, like the one used in this paper, offer an imperfect alternative. To be effective, these models should be designed through cooperative efforts of ecologists and economists. Furthermore, field experiments designed to test for response properties, such as diminishing marginal productivity and interdependence, must be conducted in order to validate the simulation models. While these studies are potentially expensive, the gains in more efficient resource use and ultimately more wildlife production should justify the expense.

ENDNOTES

¹The model tracks female mallards because males outnumber females and therefore do not limit reproduction (Johnson, et al., 1987).

² As noted, biological response is likely to depend upon the landscape characteristics of the management area. See (2006) for a description of the impact of landscape heterogeneity on the design of cost-effective mallard management plans.

³ See Rashford (2006), Appendix A1, for a complete description of management activities and their simulation methods.

⁴ Note that statistical hypothesis testing in this setting is only a test of the underlying properties of the simulation model. Conclusions are, therefore, only as valid as the MM is realistic.

⁵ Cowardin, Shaffer and Kraft (1995) found diminishing marginal productivity for cropland retirement, but their simulated study area was 51.8 km², more than seven times as large as the landscape simulated in this study.

⁶ Costs are annualized using a 4% discount to make management activities with different “useful life” spans comparable. Thus, the annualized cost represents what a manager would need to spend annually per-unit to apply the management activity. See (2006) for full discussion of, and references for, cost data.

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CHAPTER 3

**IMPROVING THE COST-EFFECTIVENESS
OF ECOSYSTEM MANAGEMENT:
AN APPLICATION TO WATERFOWL PRODUCTION**

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ABSTRACT

Wildlife and their habitat provide many ecosystem services. As a result, 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. Three issues complicate this choice. First, knowledge about biological response to alternative activities and the associated economic costs is required. Second, biological response and economic costs vary across the landscape with the quality of existing habitat and alternative land uses. Third, biological response to individual conservation activities depends on the entire portfolio of activities implemented, since activities can compete with or complement each other. Though previous literature has addressed these issues individually, they have not been addressed simultaneously. This paper determines cost-effective species conservation strategies using a bio-economic model that accounts for 1) a broad range of land-use and direct wildlife conservation activities, 2) the effect of landscape heterogeneity on agricultural and biological production, and 3) interactions between conservation activities. The model is applied to waterfowl production in the Prairie Pothole Region of the northern Great Plains. Results indicate that accounting for the three factors listed above improves the cost-effectiveness of species conservation on agricultural land.

INTRODUCTION

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 (Montgomery, Brown and Adams 1994). The ability to achieve species conservation objectives in a cost-effective⁷ manner ultimately allows higher levels of conservation or other social objectives to be achieved (Wu, Adams and Boggess 2000).

To design cost-effective species conservation programs, a resource manager must choose a portfolio of activities from a potentially large suite of conservation activities and conservation sites. Two issues complicate this decision: 1) biological response and economic costs vary across the landscape depending on the quality of existing habitat and alternative land uses, and 2) biological response to individual conservation activities often 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 tradeoffs 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 bio-economic model that accounts for 1) a broad range of land-use and direct wildlife conservation activities, 2) interactions between conservation activities, and 3) the effect of landscape heterogeneity on market (e.g. agricultural crops) and biological production. Although 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 effect of management activities on a species' population response. Data from the simulation model are then used to approximate a continuous response function, which predicts species response to varying levels and combinations of management activities. Lastly, the response function is embedded within an economic optimization model to identify least-cost management strategies for achieving a range of population objectives. The empirical focus of this analysis is waterfowl production in the north-central U.S; 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 lead to more cost-effective strategies for achieving a species conservation objective, and thus more efficient allocation of conservation resources.

LITERATURE REVIEW

There is a growing body of literature that examines cost-effective species conservation. Many of these studies consider conservation activities related to the land's primary use. Examples include the use of alternative crop management practices, such as conservation tillage and hedgerow management, to produce biodiversity on crop farms (Olenick, Wilkins and Conner 2004; Pacini, et al. 2004; van Wenum, Wossink and Renkema 2004), and the use of alternative timber harvest strategies to conserve forest-dwelling species (Calkin, et al. 2002; Lichtenstein and Montgomery 2003; Nalle, et al. 2004). Many of these studies 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, direct habitat manipulation and breeding programs. 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 wildlife conservation activities have generally considered single activities, such as harvest (Boman, Bostedt and Persson 2003; Finnoff and Tschirhart 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, included land price heterogeneity and demonstrated that species conservation costs can be reduced by incorporating heterogeneous land values into the reserve site decision (Ando, et al. 1998; Polasky, Camm and Garber-Yonts 2001). While this literature captures the effect of economic landscape heterogeneity, it typically operates at too large a scale to capture the effects of landscape heterogeneity on biological production parameters. Additionally, since biological outputs are usually not represented in the market place, heterogeneous land values rarely capture the relationship between landscape characteristics and species production.

The ability of a landscape to support species is inherently linked to the landscape's characteristics. Previous studies use two approaches to incorporate the impact of landscape heterogeneity on species production. One approach uses general landscape measures, such as habitat clustering, continuity and compactness, to relate

landscape characteristics to biological processes (Hof and Raphael 1997; Nalle, et al. 2002; 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 (Boman, Bostedt and Persson 2003; Nalle, et al. 2004; van Wenum, Wossink and Renkema 2004).

Conservation activities that alter landscape characteristics are therefore directly linked to species production, allowing for inclusion of a broader range of 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 protecting the endangered Kirtland's warbler are applied simultaneously. The authors use interactions terms in a regression model that relate the cost of achieving species population to key biological parameters. Although their results suggest the potential for interactions between conservation activities for the warbler, interactions are modeled hypothetically, no actual direct conservation activities are identified, and no direct conservation costs are considered. 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 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 bio-economic model developed in this study is applied to the conservation of waterfowl⁸ in the Prairie Pothole Region (PPR) of the Northern Great Plains. Waterfowl are an important recreational resource, generating revenues of \$2.3 billion from hunting activities alone in 2001 (Henderson 2005), which does not include non-consumptive revenues (waterfowl viewing and photography) nor does it capture non-use values. The United States, Canada and Mexico have invested over \$3.2 billion to protect and enhance waterfowl habitats through the North American Waterfowl Management Plan (U. S. Fish and Wildlife Service 2004). Waterfowl are an ideal subject for this analysis for the following reasons: 1) they benefit from both altering primary land-use activities and direct wildlife conservation activities, 2) 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 3) a detailed

biological simulation model exists for one waterfowl species, the mallard duck (*Anas platyrhynchos*), which allows for the simulation of multiple conservation activities applied simultaneously and landscape heterogeneity. Though this study is applied to mallard ducks, 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 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:

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

subject to

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

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 model presented above could be made dynamic, such that the manager must meet time specific population objectives by choosing activity, location and time triplets. However, time-specific species population targets may not be meaningful since populations often respond slowly to conservation activities (Conrad and Salas 1993), and are subject to unpredictable exogenous factors (e.g. catastrophic weather events). As a result, many species conservation programs define species population targets in terms of long-run average population levels (U. S. Fish and Wildlife Service 2004). 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. 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 mallard ducks 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. Each procedure is described and motivated below, followed by a description of conservation costs.

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 relationships between landscape characteristics and waterfowl biology. Detailed information about the landscape being modeled is required, including wetland conditions and land use/land cover information. Landscapes are defined using 26 habitat types included in the MM; example habitat types include fall-plowed grains, semi-permanent wetlands, and land enrolled in the Cropland Reserve Program (see Mack (1991) for a full description of habitat types). 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 (Cowardin, et al. 1983). The females⁹ of each breeding pair are then ushered through stages in the breeding cycle, such as nest site selection and brood rearing (Johnson, Sparling and Cowardin 1987). 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.

Cropland Retirement (CRP): 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%) (Klett 1988; Greenwood, et al. 1995). Cropland retirement programs, such as the Conservation Reserve Program (CRP), reduce this nest mortality and increase waterfowl production (Kantrud 1993; 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 (Kirsch, Duebbert and Kruse 1978). 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. The delayed hay category is similar to hayland except for the harvest date of July 15.

Planted Cover (PC): Low waterfowl production in the PPR is, in part, the result of low average nest success, which results from a lack of dense nesting cover. 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. Predator fences greatly increase localized nest success in the PPR (Cowardin, et al. 1998; Jimenez, Conover and Messmer 2001). 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-topped 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 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). Restoration of prairie wetlands is believed critical to increasing waterfowl production. The MM includes habitats for four wetland types: seasonal, temporary, semi-permanent and permanent. 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 this category.

To capture the role of land 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 2000 acres (approximately the average farm size in the PPR) and is differentiated from the others by its relative productivity for waterfowl versus agriculture in the absence of conservation activities. Land use and land cover characteristics for the three landscapes are presented in table 3.1.

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.

Table 3.1. Initial Habitat Configuration of Simulated Landscapes

	Landscape		
	Poor Acres (%)	Average Acres (%)	Good 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)

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 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. Additionally, since wetland conditions affect the relative productivity of conservation activities, each conservation

strategy is replicated three times with the MM wetland parameter set to wet, average and dry. Waterfowl production is measured in the number of ducks added to the fall population, termed recruits.

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 calculating a confidence interval around the estimated mean, stopping when the confidence interval converges (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 manager’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. Given that diminishing marginal productivity and interaction effects exist for some waterfowl conservation activities, we adopt a quadratic form for the waterfowl response functions. The response functions take the following general form:

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

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 NB, 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 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; Mack 1991)) and, therefore, there is not significant variation in the number of breeding pairs to capture interactions. It is possible that WR would complement other conservation activities on different landscapes, but this study was not designed to explore that possibility¹⁰.

Since there is theoretical justification for the insignificant variables and since our objective is to approximate the simulated data, we exclude the insignificant variables for brevity noting that the regression results change insignificantly. 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 3.2.

Table 3.2. Parameter Estimates for the Waterfowl Response Function

Variable	Landscape		
	Poor	Average	Good
Constant	7.98 (0.843)**	17.82 (1.56)**	62.32 (1.78)**
CRP	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

Key: CRP = 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

Note: *, ** indicate significance at the 5% and 1% levels, respectively. Standard errors are in parentheses.

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 four-percent discount rate. Table 3.3 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.

Table 3.3. 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

The average and poor landscapes have similar cropland acreage (1710 acres versus 1851 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. Land use costs discussed above apply to all conservation activities except nesting structures, which occupy no land area.

The cost estimates for direct conservation activities are obtained directly from the wildlife management literature (see Appendix A1). 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 schemes¹¹. We abstract away from this complexity; therefore, our land use costs should be viewed as conservative 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 (Smolik and Dobbs 1991; Zentner, et al. 1996; Kurkolova, Kling 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 (Norton, et al. 1997) and the lost revenues from foregoing multiple harvests (Norton, et al. 1997; Sopuck 2004). 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 vs. 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.

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

subject to

$$(5) \quad \sum_l f_l(x_{1l}, \dots, x_{nl}) \geq Y^*$$

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

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

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

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

Here, w_{il} denotes the per unit cost of management activity i on landscape l ; x_{ij} denotes the level of management activity i on landscape l ; $f_l(x_{1l} \dots x_{nl})$ 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 recommendation 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 non-linear 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 vs. 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 3.1 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.

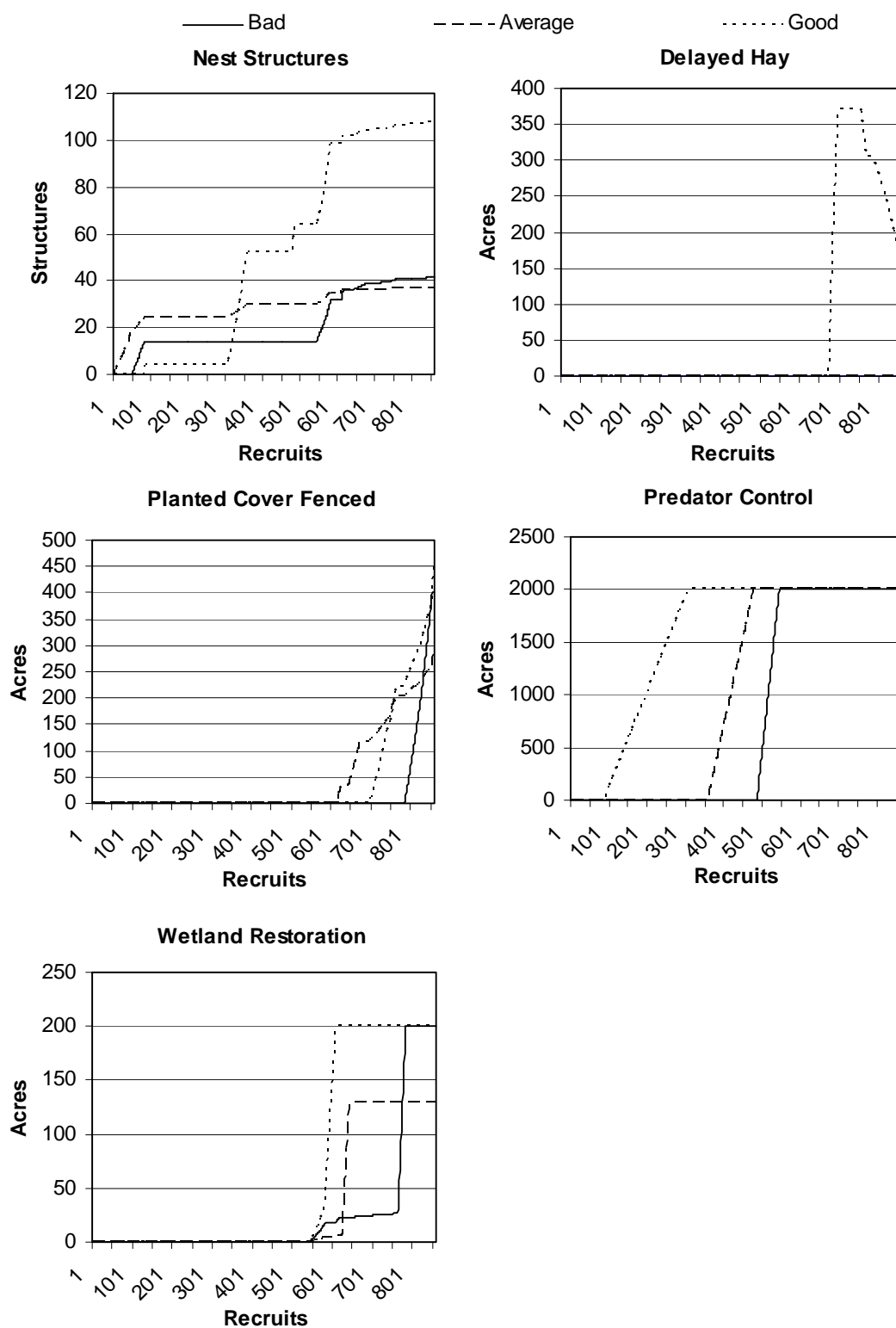


Figure 3.1. Cost-effective management activity levels derived in the full model

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 (Szentandrasei, et al. 1995). Because these other ecosystem services are not included, the full cost of primary land-use activities is apportioned to mallard production alone. If only the relevant portion of costs were assigned to mallard production, it is possible 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 mallard production, they could generate more mallard recruits by spending funds on direct wildlife 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 3.2. 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.

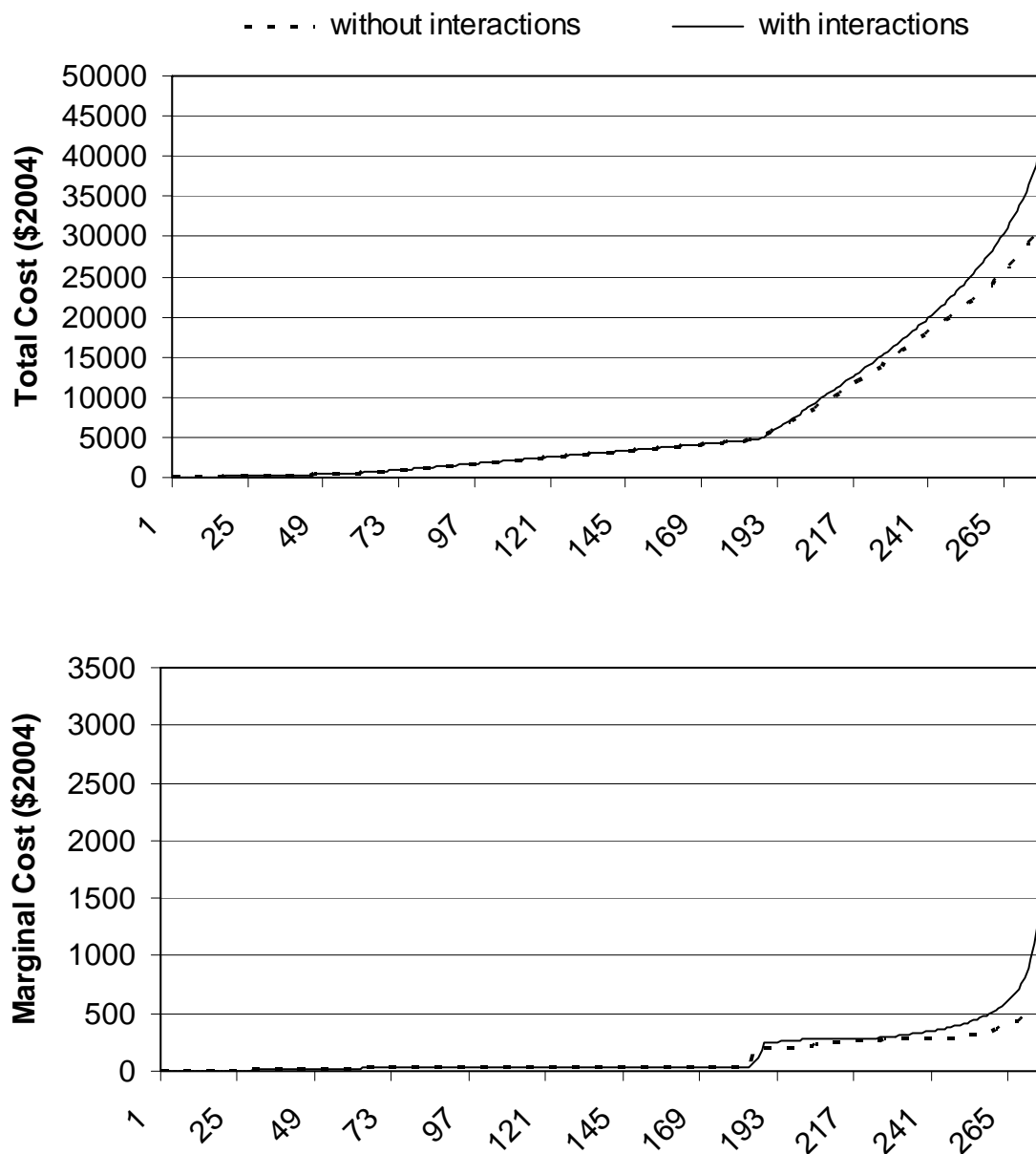


Figure 3.2. Comparison of conservation costs on the average landscape with and without interactions between conservation activities

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 3.3 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

then results in a prescription of conservation activities that are efficient for one landscape type, yet inefficient for another. Additionally, when multiple landscape types are available, differences in marginal costs can be exploited to improve cost-effectiveness.

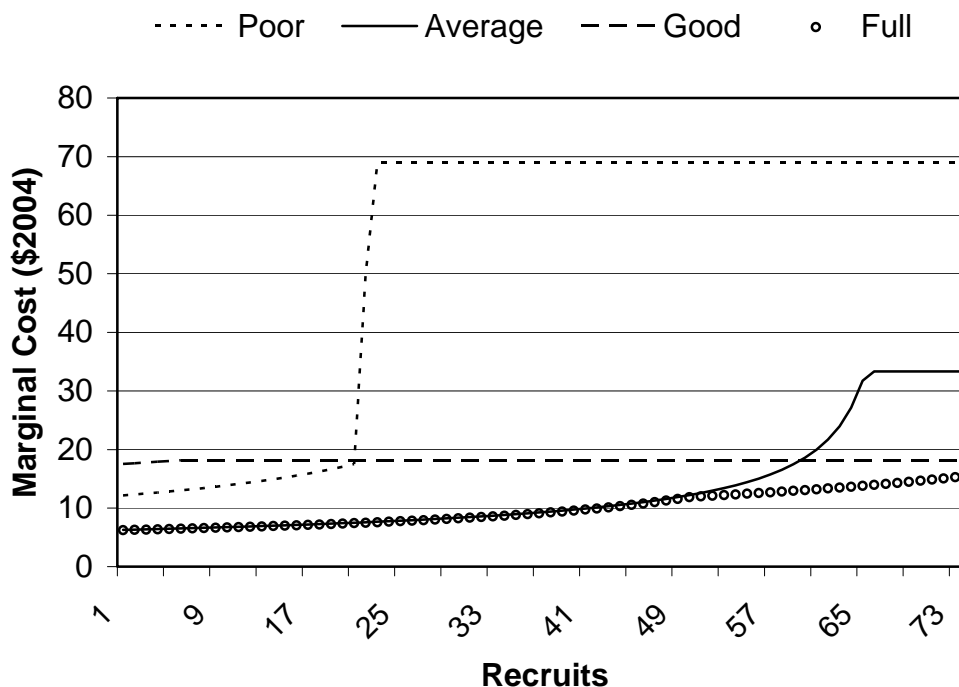


Figure 3.3. Comparison of marginal cost curves across the poor, average and good landscapes

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: 1) the availability of a broad range of land-use and direct wildlife conservation activities, 2) interactions between conservation activities, and 3) 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 profoundly 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 production 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.

In addition to improved collaboration between economists and ecologists, there are several specific directions in which this research should be extended. First, although the bioeconomic model developed in this manuscript considered landscape heterogeneity, the heterogeneity was not spatially explicit. The Mallard Model only uses the relative quantities of different habitat types to determine production, but does not consider the relative locations of different habitat types. In reality, the relative location of habitats that meet different needs of the targeted species can be important. The growing availability of GIS data and progress on spatial analysis methods should make spatially explicit modeling possible in the near future. Incorporating space into bioeconomic models will ultimately lead to more realistic and policy relevant conclusion.

The second important extension concerns the scale of the bioeconomic model. This analysis considered a small scale (2000 acres), relative the entire region over which management decisions are often made. The small scale may be relevant for individual managers, such as the manager of a wildlife refuge, however, broad policy decisions,

such as the disbursement management budgets, are often determined at a larger scale. Bioeconomic models of cost-effectiveness at this larger scale must also consider interactions across different management areas; for example, management activities applied on one area may compete with, or complement, activities applied at other management areas in the region. Additionally, examining management plans at various scales may reveal threshold effects (i.e. regions over which a certain threshold of habitat must be created prior to producing any benefits), which can complicate the identification of cost-effective management plans. This added complexity would complicate the bioeconomic model and increase the data requirements, however, the result would again be more realistic and policy relevant conclusions.

Finally, 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 objects, 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 once again be a marked improvement in the cost-effective conservation of ecosystem services.

ENDNOTES

⁷ 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 non-market valuation of species-specific benefits.

⁸ “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).

⁹The model tracks female mallards because males outnumber females and therefore do not limit reproduction (Johnson, et al., 1987).

¹⁰ 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 re-examining 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.

¹¹ We thank an anonymous referee for making this point.

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CHAPTER 4

**TOWARDS A PORTFOLIO THEORY OF
RENEWABLE RESOURCE MANAGEMENT**

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ABSTRACT

Portfolio theory provides a simple and intuitive model with which renewable resource managers can consider the tradeoff between risk and return. The standard portfolio model's underlying assumptions imply, however, that the rate of return on a management activity is constant regardless of the level at which it is applied, and regardless of the intensity levels at which other management activities are simultaneously applied. Physical response of a renewable resource to management activities, in contrast, may exhibit diminishing or increasing marginal productivity to management activity levels, as well as interdependence between multiple activities. The standard portfolio model's assumptions and results no longer hold for such a resource. In this paper, we modify the standard portfolio model to account for diminishing or increasing marginal productivity and activity interdependence, and demonstrate that the resulting set of mean-variance efficient portfolios may differ in important ways from those identified under the assumptions of the standard model. Specifically, 1) with diminishing marginal productivity to management activities or complementary interdependence there can exist a diversified portfolio that both decreases risk and increases expected return, and 2) with increasing marginal productivity to management activities or competitive interdependence between management activities there can exist a diversified portfolio that both increases risk and decreases expected return. Neither result is possible in the standard portfolio model. The application of portfolio theory to renewable resources still provides useful insights about risk-return tradeoffs; however, the application cannot be applied as directly as previous studies have suggested.

INTRODUCTION

Financial portfolio theory has been used in recent studies to examine the risk-return tradeoff for renewable resource management strategies. The standard portfolio model, developed by Markowitz (1952), is intuitively appealing in this setting; however, its underlying assumptions may not hold for all renewable resources. Specifically, the standard model assumes the following: the expected rate of return on an asset is constant regardless of the amount invested in that asset, and the expected rate of return on an asset is independent of the amount invested in other assets. This implies, in the resource management context, that the rate of return on a management activity is constant regardless of the level at which it is applied, and regardless of the intensity levels at which other management activities are simultaneously applied.

Physical response of a renewable resource to management activities (e.g. wildlife population response to habitat conservation), in contrast to the above assumptions, may exhibit diminishing or increasing marginal productivity to management activity levels, as well as interdependence between multiple activities. The marginal productivity, and hence, the rate of return of such a management activity is not constant; therefore, the assumptions of the standard portfolio model no longer hold. In this paper, we modify the standard portfolio model to account for diminishing and increasing marginal productivity and activity interdependence, and demonstrate that the resulting set of mean-variance efficient portfolios may differ in important ways from those identified under the assumptions of the standard model.

Markowitz (1952; 1959) developed portfolio theory, or mean-variance (EV) analysis, as a criterion for financial portfolio selection when asset returns are uncertain. Markowitz recognized that a tradeoff exists between return and risk, and that the tradeoff depends on the correlation between assets. He therefore proposed that investors should select assets into a portfolio based on each asset's effect on the expected return and variance of the entire portfolio, not solely on the assets expected return. The portfolio model proposes that an investment manager should choose the portfolio share (i.e. the proportion of total amount invested in each asset) to minimize risk (i.e. variance of returns) subject to a target level of expected returns. A portfolio of assets is mean-variance (EV) efficient if either of the following are true: 1) no other portfolio exists that has a higher expected return for a given variance of return, or 2) no other portfolio exists that has a lower variance of return for a given level of expected return. The standard model demonstrates that diversification can reduce risk; the degree of risk reduction depends on the correlation between asset returns. As correlation decreases, the variances of portfolios that combine them decrease. Assets with relatively low expected returns may, therefore, be attractive to an investor for their ability to reduce portfolio risk.

The renewable resource manager's problem is similar to the portfolio investment decision examined by Markowitz. Resource managers choose from a suite of management activities to generate "returns" to the resource, where returns to most activities are uncertain due to environmental variability. For example, a wildlife refuge manager chooses how to allocate a limited budget between alternative habitat management activities to increase the population of the target species, where population

response cannot be predicted with certainty. Returns to alternative activities are correlated to varying degrees because they are subject to the same environmental variability. Resource managers could, therefore, use the portfolio approach to minimize the variance of return subject to a target level of expected return. An important application would be to minimize the variance of a species population to avoid having the species listed as threatened or endangered (Hilborn, et al. 2001).

A range of approaches, other than portfolio theory, have been used to model uncertainty in renewable resource management. The most common approach treats population growth rates as stochastic and uses stochastic differential equations or stochastic dynamic programming to examine the implication of alternative management strategies. This approach has been used to examine the risk of population collapse under alternative harvest policies for fisheries (Spencer 1997; Braumann 1999; Enberg 2005), several terrestrial species (Jensen 2002), and in controlled laboratory experiments for ciliate (Fryxell, Smith and Lynn 2005). A second common approach defines the resource management problem in probabilistic terms. One example is the selection of sites into a conservation reserve network to achieve an “expected coverage” level or “occurrence probability threshold” (see e.g. Arthur, et al. 2002). Similarly, Haight and Travis (1997) develop a programming model for the gray wolf that minimizes the cost of meeting a population viability constraint (by choosing the size of habitat areas).

These alternative approaches each provide a framework for analyzing the risks associated with resource management alternatives. In applied settings, however, these approaches are often limited to a relatively small set of discrete management alternatives,

such as the number of reserve sites, to assure that a solution to the problem exists. For many renewable resources, such as terrestrial species, managers choose from a large set of management alternatives that can be applied at multiple levels. The portfolio model, which has been used in financial markets to assess the risk-return tradeoff between thousands of financial assets, may be an advantageous approach for such resources.

Several authors have applied portfolio theory to renewable resource management under uncertainty. Forestry applications include the selection of timber management regimes to reduce income risk (Reeves and Haight 2000), and the selection of EV-efficient portfolios of timber and financial assets (Thomson 1991; Reeves and Haight 2000). There have also been several applications of portfolio theory to determine optimal harvest regimes in multi-species fisheries (Edwards, Link and Rountree 2004; Perruso, Weldon and Larkin 2005; Sanchirico, Smith and Lipton 2005). The portfolio problem in these applications is to assign a proportion of total landings to each fish species to reduce income risk. These applications maintain the standard assumption that the expected rate of return is constant regardless of the activity's level (i.e. returns are linear) or the presence of other activities (i.e. returns are independent). Returns to alternative activities are therefore assumed to be related through their covariance only.

The standard assumptions of linear and independent returns are a simplification of the actual return-generating process for many renewable resources. These assumptions imply, in the forestry applications, that the rate of return per-acre for a given management regime (e.g. 20 year rotation) is constant and independent of the management regimes on neighboring parcels. These implications are reasonable if the timber market is

competitive (i.e. no single forest manager can produce enough product to change the market price), and the choice of timber management regimes does not significantly affect the spread of pests and disease between neighboring forest parcels.

The assumptions of linear and independent returns, in the fisheries applications, seem less reasonable. Consider a fisherman's portfolio problem of choosing the proportion of landings to assign to each fish species (see e.g. Perruso, Weldon and Larkin 2005; Sanchirico, Smith and Lipton 2005). The proportion of total landings assigned to a particular species corresponds to a level of effort exerted to target that species (e.g. number of hours spent targeting that species). The assumption of a constant expected rate of return to each fish species implies that the marginal productivity of effort is constant. Fisheries yield functions are generally assumed, to the contrary, to exhibit diminishing marginal productivity with respect to effort (Clark 1990; Conrad 1999). Additionally, because fish species are linked through predator-prey interactions, the landings of one species is likely to impact (in a dynamic setting) the expected landings of other competitive or complementary species. The rate of return on a given species is, therefore, a function of the portfolio share invested in that species. The rate of return on the portfolio, as a result, is no longer a simple weighted average of the rates of returns on individual activities.

Papers that use the weighted average to calculate returns on the portfolio, despite violations of the standard model's assumptions, are likely to mis-identify the EV-efficient frontier. Sanchirico, Smith and Lipton (2005) recognize that the standard model imposes simplifying assumptions about biological response, but argue that the interactions

between species can be captured through the covariance of returns. This assumes away any diminishing marginal productivity or interdependence, which impact expected returns. Given the difficulty of developing ecosystem level response models (see e.g. Finnoff and Tschirhart 2003), the standard portfolio model applied to multi-species management portfolios can provide valuable insights, despite simplifying assumptions. For other applications, such as management of a single terrestrial species, additional insights can be gained by relaxing the assumptions of linear and independent returns.

In addition to the assumption of linear and independent returns, the standard portfolio model assumes that uncertainty originates from stochastic asset prices, which cause the rates of return on assets to be uncertain. Many renewable resources are not traded in a market and therefore have no associated price. The lack of output prices implies that physical output must be used as an alternative measure of returns. This is problematic, however, because physical output generated with different expenditure levels are not comparable. Additionally, the source of uncertainty for non-market resources is clearly not price uncertainty, but production uncertainty and, therefore, resource response to management activities must be defined stochastically.

In this paper we move towards a general portfolio theory for renewable resources by modifying the standard portfolio model to account for the lack of resource prices and the variable marginal productivity of management activities. We develop an analytical model in the next section to examine the effects of a more complex response function on the EV-efficient set. The response function accounts for variable marginal productivity of management activities and interactions between management activities, and is

represented as a second-order quadratic response function. In the absence of resource prices, “returns” are defined as physical output. However, only output levels that can be produced with the same budget are considered. In the final section of the paper, we use simulated data on waterfowl response to three common management activities to construct portfolios and demonstrate results of the analytical model.

Both the analytical model and the application to waterfowl management indicate that EV-efficient portfolios generated by a more complex response function can differ significantly from those derived with the assumptions of the standard model. When management activities exhibit diminishing marginal productivity or complementary interdependence, we find that diversification can both reduce risk and increase expected return. Alternatively, with increasing marginal returns to management activities or competitive interdependence, diversification can increase risk and decrease expected return. Neither of these results are possible in the standard model.

CONCEPTUAL MODEL

In this section we adjust the standard portfolio model for the case of renewable resource management, such that it accounts for variable marginal productivity. We begin by reviewing the results for a standard portfolio of financial assets to facilitate later comparison to resource management portfolios.

The Standard Portfolio Model Applied to Financial Assets

An EV-efficient portfolio is derived in the standard model by solving the following quadratic programming problem for a specific expected return level (\tilde{M}):

$$\begin{aligned}
& \min_{\alpha} \alpha' \Omega \alpha \\
& \text{s.t. } \alpha' \mu \geq \tilde{M} \\
(1) \quad & \sum_{i=1}^n \alpha_i = 1, \quad \forall i = 1, \dots, n \\
& 0 \leq \alpha_i \leq 1, \quad \forall i = 1, \dots, n
\end{aligned}$$

where α is a $(n \times 1)$ vector of proportional investments made in each of the n assets, μ is the $(n \times 1)$ vector of expected returns per dollar invested in each asset, Ω is the $(n \times n)$ covariance matrix of returns, and \tilde{M} is the target level of expected returns. The EV-efficient frontier is derived by parametrically solving the QP problem for all possible target levels of expected returns. A particular investor's utility maximizing portfolio is then determined by their risk preferences¹².

The portfolio problem in (1) is generalized by defining a production process of the general form $y_i = f_i(x)$, where y_i is the physical units of output from asset i , and x is a $(1 \times n)$ vector of activity levels for each asset (i.e. x_i is the number of stocks of asset i). We explicitly model the production process here to facilitate comparison to resource portfolios presented below. The standard model assumes that

$$(2) \quad y_i = f_i(x) = x_i \quad \forall i = 1, \dots, n.$$

Next, define p_i as per unit selling price of asset i , and w_i as the per unit cost of purchasing asset i . Then the net return on an investment in asset i (NR_i) is given by

$$(3) \quad NR_i = p_i x_i - w_i x_i,$$

and the rate of return (RR_i) is

$$(4) \quad RR_i = \frac{p_i x_i - w_i x_i}{w_i x_i} = \frac{p_i - w_i}{w_i}.$$

Definition (4) establishes that the rate of return on an asset is independent of the number of stocks (x_i) an individual investor chooses to hold. This is akin to a perfect competition assumption, i.e. no single investor can purchase a quantity that is sufficiently large to impact the market rate of return on an asset¹³. A portfolio of assets defined by a unique vector $\{x\}$ is formed by combining assets subject to the total investment and non-negativity constraints:

$$(5) \quad \sum_{i=1}^n x_i w_i = W_p, \text{ and}$$

$$(6) \quad x_i \geq 0 \quad \forall i = 1, \dots, n,$$

where W_p is the total dollar amount invested in a portfolio. The total investment constraint (5) implies that an investor must be fully invested (since cash can always be considered an asset, this constraint is not restrictive) and (6) disallows short-sales. Now, the proportion of the portfolio invested in asset i (α_i) can be explicitly defined from (5) as

$$(7) \quad \alpha_i = \frac{w_i x_i}{W_p},$$

which, combined with (6), implies that $0 \leq \alpha_i \leq 1, \forall i = 1, \dots, n$. The rate of return on a portfolio is, therefore, given by

$$(8) \quad RR_p = \frac{\sum_{i=1}^n (p_i x_i - w_i x_i)}{\sum_{i=1}^n w_i x_i} = \sum_{i=1}^n \alpha_i RR_i \text{ (see Appendix A2).}$$

Uncertainty enters the standard model because the future per-unit selling price (p_i) is unknown at the time an asset is purchased. Assume p_i is a random variable with finite mean and variance. Then it is straightforward to show that

$$(9) \quad RR_p = \sum_{i=1}^n \alpha_i RR_i \sim \left(\sum_{i=1}^n \alpha_i \mu_i, \sum_{i=1}^n \sum_{j=1}^n \alpha_i \alpha_j \sigma_{ij} \right),$$

where $\mu_i = E[RR_i]$, and $\sigma_{ij} = E[(RR_i - \mu_i)(RR_j - \mu_j)]$.

While the rate of return on an individual asset in the standard model is independent of the asset level, the rate of the return on a portfolio is dependent on the chosen asset levels via α_i . Choosing a particular portfolio share, α_i , is equivalent to choosing an asset level and vice versa. Note that α_i is homogeneous of degree zero in x and, therefore, RR_p is homogeneous of degree zero in x_i , i.e. if you double the quantity of each asset held, the rate of return on the portfolio does not change. This result is only assured when individual assets exhibit a constant rate of return.

Now we consider the properties of EV-efficient frontiers for $n = 2$. We limit the analytical derivation to the case of $n = 2$ assets for simplicity. When $n = 2$, and the expected rate and variance of return differ between the two assets, EV-efficient portfolios can be calculated directly without solving the portfolio minimization quadratic programming problem (1). This is because every combination of the assets has a unique mean and variance and is therefore a point on the EV-efficient frontier. Note also that, since any two portfolios containing any number of assets can themselves be combined in a weighted average to construct new portfolios, the $n = 2$ case generalizes to $n > 2$.

Recall that the EV-efficient frontier consists of portfolios that either have the lowest variance for a given level of return, or the highest return for a given level of variance. Denoting the two assets by 1 and 2 respectively, assume that $\mu_2 > \mu_1$, and $\sigma_2^2 > \sigma_1^2$; thus asset 2 is more risky than asset 1, but generates a higher expected return. We consider three degrees of correlation between assets, which encompass all others: (1) perfect negative correlation ($\rho = -1$), (2) no correlation ($\rho = 0$), and (3) perfect positive correlation ($\rho = 1$). EV-efficient portfolios for these three cases are depicted in figure 4.1, with σ_p denoting the standard deviation of R_p .

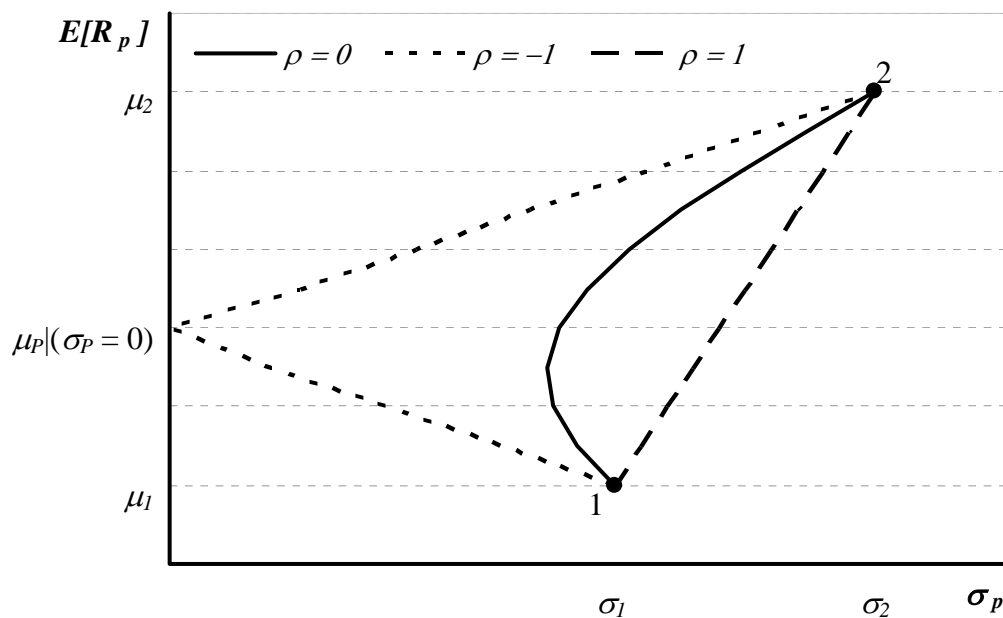


Figure 4.1. EV-efficient frontiers for $\rho = 0$, $\rho = -1$, and $\rho = 1$ under assumptions of the standard portfolio model

In the standard model with $n = 2$, the maximum return portfolio (E_{max}) always occurs when $\alpha_1 = 0$ and $\alpha_2 = 1$, regardless of the degree of correlation. That is, an investor who is not concerned about risk should invest all of their funds in the asset that has the highest expected rate of return. Because the rate of return on the portfolio is a linear combination of the individual assets' rates of return, it is not possible to exceed the rate of return on a portfolio consisting only of asset 2. The minimum variance portfolio can likewise be identified, but it depends on the value of ρ . When $\rho = 1$, all portfolios in EV-space lie on a line connecting the EV points or $\alpha_1 = 1$ and $\alpha_2 = 1$ (see Appendix A2). The minimum variance portfolio, therefore, consists entirely of the low risk asset. For any $\rho < 1$, the portfolio variance will be less than weighted average of the assets' variances for all portfolios containing positive quantities of both assets. Thus, if assets are not perfectly positively correlated, diversification reduces risk. The EV-efficient frontier when $\rho = -1$ is composed of two line segments; one connects the minimum variance portfolio and the maximum return portfolio, and the other connects the minimum variance portfolio and the minimum return portfolio (see Appendix A3). Risk can be eliminated completely through diversification when $\rho = -1$. A portfolio with zero variance exists ($\mu_P | (\sigma_P = 0)$ in figure 2), which is given by $\arg \min_{\{\alpha_1, \alpha_2\}} \left(\sum_{i=1}^n \sum_{j=1}^n \alpha_i \alpha_j \sigma_{ij} \right)$.

The above results can be summarized as follows, which assumes RR_i constant across α_i : (1) expected rate of return is maximized by fully investing in the asset with the highest expected rate of return, (2) expected rate of return is minimized by fully investing in the asset with the lowest expected rate of return, and (3) the variance of any possible

portfolio is bounded above by the variance of the asset with the highest risk, and bounded below by zero (the minimum variance possible when assets are perfectly negatively correlated). These results, combined with the equations for the EV-efficient frontier provided in Appendix A2 imply that all EV-efficient portfolios regardless of ρ are contained in a triangle in EV-space, whose sides are defined by the EV-efficient frontiers for $\rho = 1$ and $\rho = -1$. Next we relax the assumptions that RR_i is constant across α_i and independent of α_i , and examine the effect on the EV-efficient frontiers.

Renewable Resource Management Portfolios

In this section we modify the standard portfolio model to account for the lack of resource prices and the variable marginal productivity of management activities. Recall that the lack of prices causes difficulty because different levels of physical output produced for different expenditure levels are not comparable. In the standard model the rate of return is defined over net returns and therefore incorporates different expenditure levels. Furthermore, since the rate of return is homogeneous of degree zero in activity levels, changing the total amount invested (i.e. budget) will not affect the EV-efficient portfolio shares. As a result, two alternative portfolios involving different total investment levels are comparable. Alternative resource portfolios, in contrast, must strictly satisfy a budget constraint to be comparable. In this case, alternative physical output levels are directly comparable and the tradeoff between risk and return of various output levels is clear (i.e. to reduce risk, the manager must be willing to accept less units of output). This approach is also intuitively appealing since renewable resource managers often face budget constraints.

Next, we must define a “return” generating process that is consistent with variable marginal productivity. Variable marginal productivity generally results from increasing or decreasing marginal productivity to individual activities and technical interdependence between management activities. We capture these properties by specifying a resource production process with the following general form

$$(10) \quad y_i = f_i(x; \nu, \varepsilon_i),$$

where y_i is the physical units of output from management activity i , x is a $(1 \times n)$ vector of management activity levels, ν is a vector of non-stochastic parameters and ε_i is a random disturbance term with finite mean and variance. The assumptions on (10) that capture variable production are summarized by:

i)

$$\frac{\partial^2 y_i}{\partial x_i^2} = \frac{\partial^2 f_i(x, \nu, \varepsilon_i)}{\partial x_i^2} \left\{ \begin{array}{l} > 0 \text{ if } i \text{ exhibits increasing marginal productivity} \\ = 0 \text{ if } i \text{ exhibits constant marginal productivity} \\ < 0 \text{ if } i \text{ exhibits decreasing marginal productivity} \end{array} \right\}, \text{ and}$$

$$\text{ii) } \quad \frac{\partial^2 y_i}{\partial x_i \partial x_j} = \frac{\partial^2 f_i(x, \nu, \varepsilon_i)}{\partial x_i \partial x_j} \left\{ \begin{array}{l} > 0 \text{ if } i \text{ and } j \text{ are complementary} \\ = 0 \text{ if } i \text{ and } j \text{ are independent} \\ < 0 \text{ if } i \text{ and } j \text{ are competitive} \end{array} \right\}.$$

Resource production defined by (10) and various combinations of assumptions (i) and (ii) captures a variety of resource production processes.

No general conclusions about the properties of the EV-efficient frontier can be reached without explicitly specifying the production process. We, therefore, specify the production process as follows:

$$(11) \quad y_i(x_i) = a_i x_i + b_i x_i^2 + \sum_{j=1}^n c_{ij} x_i x_j + x_i \varepsilon_i,$$

where a_i , b_i and c_{ij} are non-stochastic parameters, and $\varepsilon_i \sim (0, \sigma_i^2)$. The multiplicative error structure implies that variance increases as the activity level increases. This error structure is commonly applied to natural production processes, such as agriculture (Hardaker, Huirne and Anderson 2002)¹⁴. From (11) we define the total production (Y_p) from the use of multiple management activities as

$$(12) \quad Y_p(x) = \sum_{i=1}^n y_i = \sum_{i=1}^n a_i x_i + b_i x_i^2 + \sum_{i=1}^n \sum_{j>i}^n \delta_{ij} x_i x_j + \sum_{i=1}^n x_i \varepsilon_i,$$

where $\delta_{ij} = (c_{ij} + c_{ji})$. The output from a portfolio of management activities subject to (5) and (6) can be defined by solving (7) for x_i and substituting the result into (12), which generates:

$$(13) \quad Y_p(\alpha) = \sum_{i=1}^n (\tilde{w}_i a_i \alpha_i + \tilde{w}_i^2 b_i \alpha_i^2) + \sum_{i=1}^n \sum_{j>i}^n \tilde{w}_i \tilde{w}_j \delta_{ij} \alpha_i \alpha_j + \sum_{i=1}^n \alpha_i \tilde{w}_i \varepsilon_i,$$

where $\tilde{w}_i = \frac{W_p}{w_i}$. Since inputs prices are strictly positive, assumptions regarding variable marginal productivity and factor interdependence made on $Y_p(x)$ follow directly to $Y_p(\alpha)$.

The resource manager's problem, therefore, is to select the portfolio of management activities $\{\alpha\}$ that minimizes variance of output, subject to an output target (M) and an implicit budget constraint defined in terms of α . Denoting the variance of output as $\sigma_p^2(\alpha)$, the manager's problem is given by¹⁵

$$(14) \quad \min_{\{\alpha\}} \sigma_p^2(\alpha) = \sum_{i=1}^n \sum_{j>i}^n \tilde{w}_i \tilde{w}_j \alpha_i \alpha_j \sigma_{ij}$$

subject to

$$(15) \quad E[Y_p(\alpha)] = \sum_{i=1}^n \tilde{w}_i a_i \alpha_i + \tilde{w}_i b_i \alpha_i^2 + \sum_{i=1}^n \sum_{j>i}^n \tilde{w}_i \tilde{w}_j \delta_{ij} \alpha_i \alpha_j \geq M \quad ,$$

$$(16) \quad \begin{aligned} \sum_{i=1}^n \alpha_i &= 1, \quad \forall i = 1, \dots, n \\ 0 &\leq \alpha_i \leq 1, \quad \forall i = 1, \dots, n \end{aligned}$$

We now characterize solutions to the manager's problem for $n = 2$ activities. We assume that activity 2 is relatively more productive and risky than activity 1, i.e. $E[Y_p(\alpha_1 = 0)] > E[Y_p(\alpha_1 = 1)]$, and $\sigma_p^2(\alpha_1 = 0) > \sigma_p^2(\alpha_1 = 1)$. Consider first the bounds on portfolio variance for three degrees of correlation ($\rho = 1$, $\rho = 0$, and $\rho = -1$). With portfolio variance given in (14), we derive the following results for the bounds (see Appendix A5):

- 1) Portfolio variance, $\sigma_p^2(\alpha)$, is bounded above by the portfolio variance attained when fully invested in the high risk activity ($\sigma_p^2(\alpha_1 = 0) = \tilde{w}_2^2 \sigma_2^2$), regardless of the degree of correlation;
- 2) When $\rho = 0$, there exists a portfolio with positive investments in both activities that has a lower variance than is attained by being fully invested in the less risky activity;
- 3) When $\rho = -1$, there exists a portfolio with positive investments in both activities that generates zero variance.

Thus, variance of resource portfolios behaves very similarly to variance of portfolios in the standard model.

The expected “return” of resource portfolios, in contrast, behave much differently than the expected returns of portfolios in the standard model. The expected return, or expected output, for resource portfolios, substituting $\alpha_2 = (1 - \alpha_1)$ into the left hand side of (15), is given by

$$(17) \quad E[Y_p(\alpha_1)] = \tilde{w}_1 a_1 \alpha_1 + \tilde{w}_1 b_1 \alpha_1^2 + \tilde{w}_2 a_2 (1 - \alpha_1) + \tilde{w}_2 b_2 (1 - \alpha_1)^2 + \tilde{w}_2 \tilde{w}_2 \delta_{12} \alpha_1 (1 - \alpha_1).$$

To examine the bounds on expected output, first set $\frac{\partial E[Y_p(\alpha_1)]}{\partial \alpha_1} = 0$ and solve for α_1 to

get

$$(19) \quad \alpha_1^* = \frac{a_2 \tilde{w}_2 - a_1 \tilde{w}_1 + 2b_2 \tilde{w}_2 - \delta_{12} \tilde{w}_1 \tilde{w}_2}{2b_1 \tilde{w}_1 + 2b_2 \tilde{w}_2 - 2\delta_{12} \tilde{w}_1 \tilde{w}_2}.$$

This interior portfolio, is a candidate solution to either $\max_{\alpha_1} E[Y_p(\alpha_1)]$, or $\min_{\alpha_1} E[Y_p(\alpha_1)]$.

The second order derivative for both problems is

$$(20) \quad 2b_1 \tilde{w}_1 + 2b_2 \tilde{w}_2 - 2\delta_{12} \tilde{w}_1 \tilde{w}_2.$$

If α_1^* is a maximum, (20) implies that $2b_1 \tilde{w}_1 + 2b_2 \tilde{w}_2 - 2\delta_{12} \tilde{w}_1 \tilde{w}_2 < 0$. There are a wide range of parameter values for which this second order condition will hold; note that diminishing marginal productivity ($b_i < 0$) and complementary interdependence ($\delta_{ij} > 0$) increase the likelihood of α_1^* being a maximum. Similarly, if α_1^* is a minimum, (20) implies that $2b_1 \tilde{w}_1 + 2b_2 \tilde{w}_2 - 2\delta_{12} \tilde{w}_1 \tilde{w}_2 > 0$. In this case, increasing marginal productivity ($b_i > 0$) and competitive interdependence ($\delta_{ij} < 0$) increase the likelihood that α_1^* is a minimum.

To determine the conditions under which the interior solution is a maximum and a minimum, we need to evaluate the relative values of $E[Y_p(\alpha_1^*)]$, $E[Y_p(\alpha_1 = 1)]$, and $E[Y_p(\alpha_1 = 0)]$. However, because this involves the relative values of multiple parameters, general analytical solutions can only be attained by making excessively restrictive assumptions. In lieu of deriving such analytical solutions, we assume various parameter values and examine some key numerical results. The objective is to determine whether conditions (i.e. parameter values) exist such that a portfolio, with positive investments in both activities, can generate an expected output greater (less) than the expected return when fully invested in the single activity with the higher (lower) output.

Case 1: Decreasing and Increasing Marginal Productivity

Consider first the case of $\delta_{l2} = 0$ with no restrictions on b_i . The portfolio's expected return is given by

$$E[Y_p(\alpha_1)] = \tilde{w}_1 a_1 \alpha_1 + \tilde{w}_1 b_1 \alpha_1^2 + \tilde{w}_2 a_2 (1 - \alpha_1) + \tilde{w}_2 b_2 (1 - \alpha_1)^2,$$

which can be written as

$$(21) \quad E[Y_p(\alpha_1)] = E[Y_1(\alpha_1)] + E[Y_2(1 - \alpha_1)].$$

Recall that the maximum output attained when fully invested in a single activity is

$$(22) \quad E[Y_p(\alpha_1 = 0)] = \tilde{w}_2 a_2 + \tilde{w}_2 b_2,$$

and that the minimum expected output attained when fully invested in a single activity is

$$(23) \quad E[Y_p(\alpha_1 = 1)] = \tilde{w}_1 a_1 + \tilde{w}_1 b_1.$$

Suppose the interior solution, (19), is the maximum expected output. Then

because $E[Y_p(\alpha_1)]$ is quadratic in α , it must be that $\lim_{\alpha_1 \rightarrow 0} \frac{\partial E[Y_p(\alpha_1)]}{\partial \alpha_1} > 0$, i.e., the expected

output function approaches (22) from above. This implies that

$$(24) \quad \tilde{w}_1 a_1 > \tilde{w}_2 a_2 + 2\tilde{w}_2 b_2,$$

i.e. an interior maximum exists if the gains from a marginal increase in α_1 (from $\alpha_1 = 0$) exceed the losses. Note that diminishing marginal productivity reduces the losses of an increase in α_1 (i.e. a decrease in α_2). If activity 2 exhibits sufficient diminishing marginal productivity, then the loss from a marginal decrease in α_2 will be small relative to the gains of a marginal increase in α_1 . This implies that when activities exhibit diminishing marginal productivity, an interior portfolio that generates expected output greater than that which can be achieved by fully investing in any one activity may exist. Figure 4.2 displays the EV-frontiers for this case.

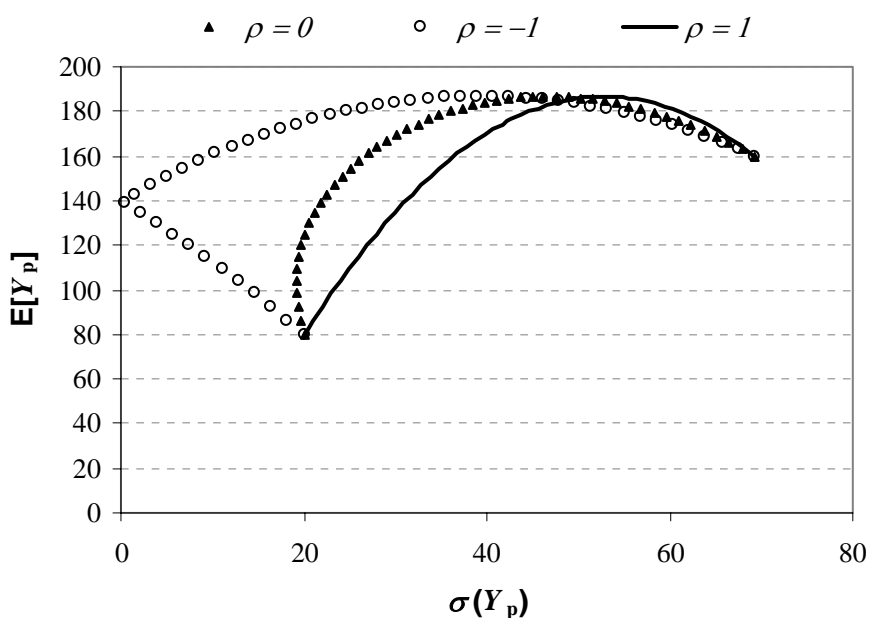


Figure 4.2. EV-frontiers for $\rho = 0$, $\rho = -1$, and $\rho = 1$ when diminishing marginal productivity is present. Derived with $W_P = 20$, $w_1 = 1$, $w_2 = 1$, $a_1 = 4$, $a_2 = 20$, $b_1 = 0$, $b_2 = -0.6$, $\delta_{12} = 0$.

The existence of an interior portfolio that generates expected output greater than full investment in activity 2 and has zero risk is even possible in this resource model. This occurs when the difference between expected outputs at full investment between the two activities is relatively small, and the activities are perfectly negatively correlated (figure 4.3). In this case, diversification can increase expected returns and eliminate risk.

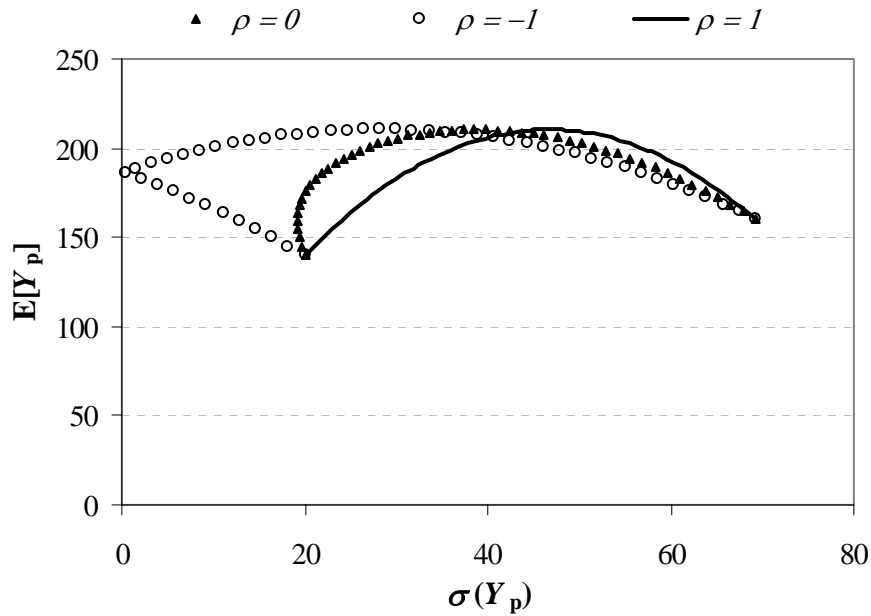


Figure 4.3. EV-frontiers with diminishing marginal productivity assuming that activity 1 is highly productive relative to activity 2. Derived with $W_P = 20$, $w_1 = 1$, $w_2 = 1$, $a_1 = 7$, $a_2 = 20$, $b_1 = 0$, $b_2 = -0.6$, $\delta_{12} = 0$.

Suppose the interior solution, (19), is instead the minimum expected output; then

it must be that $\lim_{\alpha_1 \rightarrow 1} \frac{\partial E[Y_p(\alpha_1)]}{\partial \alpha_1} > 0$, i.e. the expected output function approaches (23) from

below. This implies that

$$(25) \quad \tilde{w}_1 a_1 + 2\tilde{w}_1 b_1 > \tilde{w}_2 a_2.$$

The intuition is similar to that presented for a maximum. The interior solution is a minimum if the losses from a marginal decrease in α_1 (from $\alpha_1 = 1$) exceed the gains.

Note that increasing marginal productivity increases the losses of a decrease in α_1 ;

therefore, it increases the likelihood that an interior minimum exists. If activity 1 exhibits sufficient increasing marginal productivity, then the loss from a marginal decrease in α_1

(from $\alpha_1 = 1$) will be large relative to the gains from a marginal increase in α_2 (from $\alpha_2 = 0$), and an interior minimum will exist. This implies that when activities exhibit increasing marginal productivity an interior portfolio that generates expected output below that which can be achieved by fully investing in the low output activity may exist (figure 4.4).

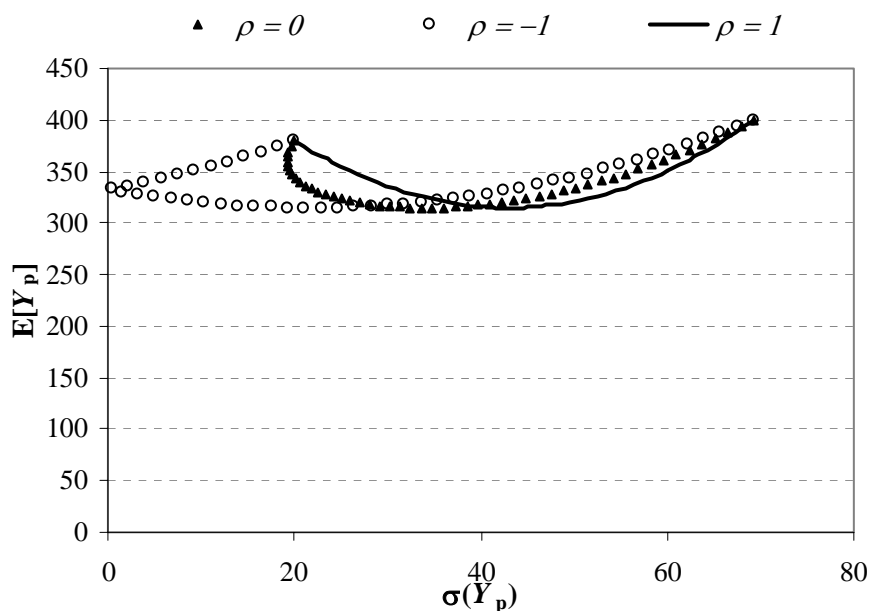


Figure 4.4. EV-frontiers for $\rho = 0$, $\rho = -1$, and $\rho = 1$ when increasing marginal productivity is present. Derived with $W_P = 20$, $w_1 = 1$, $w_2 = 1$, $a_1 = 4$, $a_2 = 20$, $b_1 = 0.75$, $b_2 = 0$, $\delta_{12} = 0$.

Case 2: Complementary and Competitive Interdependence

We have examined some extreme cases for diminishing and increasing marginal productivity; we now turn our attention to activity interdependence. For simplicity, we restrict $b_1 = b_2 = 0$. Portfolio expected output is now

$$(26) \quad E[Y_p(\alpha_1)] = \tilde{w}_1 a_1 \alpha_1 + \tilde{w}_2 a_2 (1 - \alpha_1) + \delta_{12} \tilde{w}_1 \tilde{w}_2 \alpha_1 (1 - \alpha_1),$$

with first derivative

$$(27) \quad \frac{\partial E[Y_p(\alpha_1)]}{\partial \alpha_1} = \tilde{w}_1 a_1 - \tilde{w}_2 a_2 + \delta_{12} \tilde{w}_1 \tilde{w}_2 - 2\delta_{12} \tilde{w}_1 \tilde{w}_2 \alpha_1.$$

The maximum and minimum returns at full investment are, respectively,

$$(27) \quad E[Y_p(\alpha_1 = 0)] = \tilde{w}_2 a_2, \text{ and}$$

$$(28) \quad E[Y_p(\alpha_1 = 1)] = \tilde{w}_1 a_1.$$

Suppose that the interior solution (19) is a maximum, then it must be the case that

$$\lim_{\alpha_1 \rightarrow 0} \frac{\partial E[Y_p(\alpha_1)]}{\partial \alpha_1} > 0, \text{ i.e., the expected output function approaches (27) from above. This}$$

implies

$$(29) \quad \tilde{w}_1 a_1 + \delta_{12} \tilde{w}_1 \tilde{w}_2 > \tilde{w}_2 a_2.$$

Again, the interior solution is a maximum if the gains from a marginal increase in α_1 (from $\alpha_1 = 0$) exceed the losses of a marginal decrease in α_2 (from $\alpha_2 = 1$). Given that $\tilde{w}_2 a_2 > \tilde{w}_1 a_1$ by assumption, the interior maximum can only occur if δ_{12} is positive and sufficiently large. Intuitively, if the activities are technical complements and the gains from their combination are sufficiently large, the maximum expected output will occur with positive investments in both activities. The implications for the EV-frontier are similar to those of diminishing marginal productivity (figure 4.5)

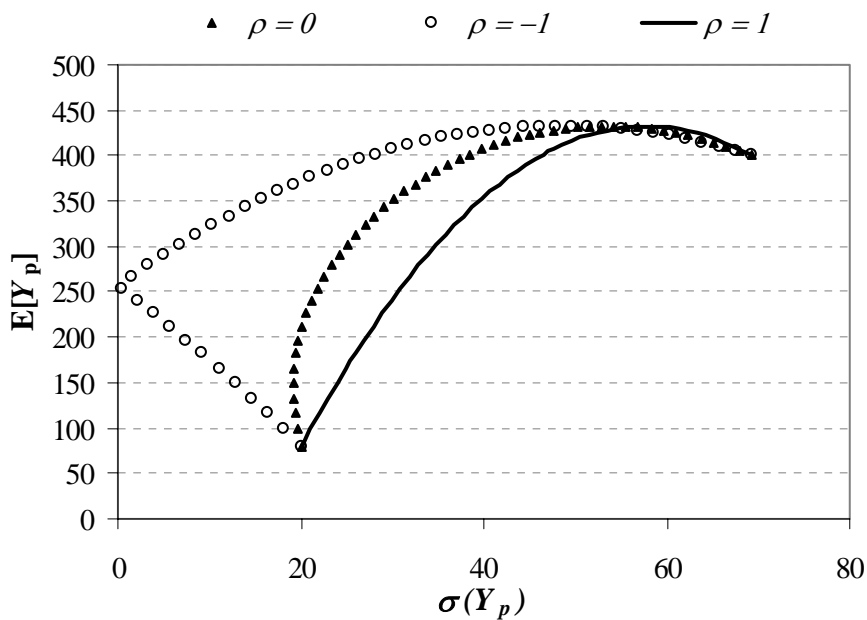


Figure 4.5. EV-frontiers for $\rho = 0$, $\rho = -1$, and $\rho = 1$ when complementary interdependence is present. Derived with $W_p = 20$, $w_1 = 1$, $w_2 = 1$, $a_1 = 4$, $a_2 = 20$, $b_1 = 0$, $b_2 = 0$, $\delta_{12} = 1.5$.

Suppose the interior solution is instead a minimum, then $\lim_{\alpha_1 \rightarrow 1} \frac{\partial E[Y_p(\alpha_1)]}{\partial \alpha_1} > 0$, i.e.,

the expected output function approaches (28) from below. This implies

$$(30) \quad \tilde{w}_1 a_1 > \tilde{w}_2 a_2 + \delta_{12} \tilde{w}_1 \tilde{w}_2.$$

The interior solution is a minimum if the loss in expected output from marginal decrease in α_1 (from $\alpha_1 = 1$) exceeds the gains of a marginal increase in α_2 (from $\alpha_2 = 0$). This can only occur if δ_{12} is negative and sufficiently large. Intuitively, if the activities are technically competitive then combining them can decrease total expected output. The effects on the EV-frontier are similar to those of increasing marginal returns (figure 4.6).

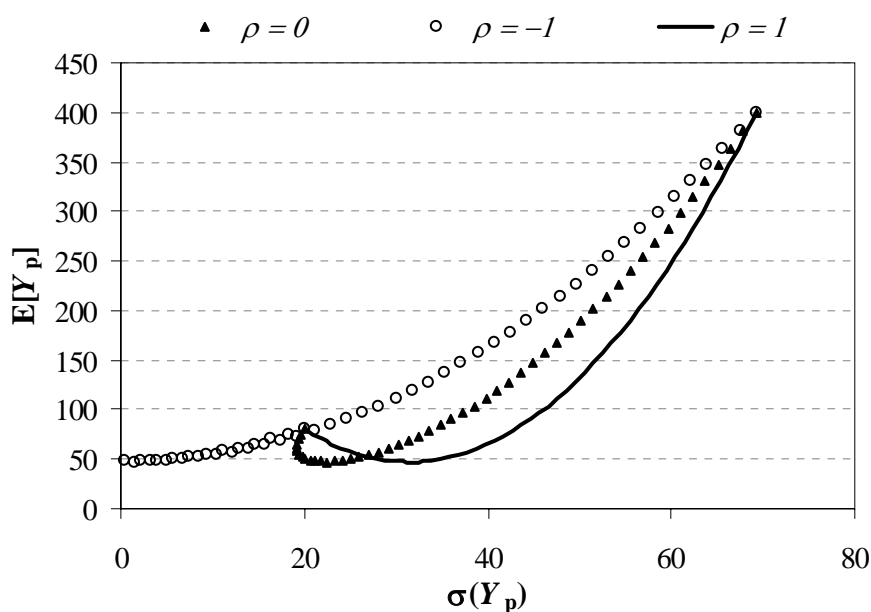


Figure 4.6. EV-frontiers for $\rho = 0$, $\rho = -1$, and $\rho = 1$ when competitive interdependence is present. Derived with $W_P = 20$, $w_1 = 1$, $w_2 = 1$, $a_1 = 4$, $a_2 = 20$, $b_1 = 0$, $b_2 = 0$, $\delta_{12} = -1.5$.

The cases examined above reflect the extremes; however, it is clear that portfolios containing activities which exhibit increasing or decreasing marginal productivity, or activity interdependence, can generate EV-frontiers that differ significantly from those in the standard portfolio model. Specifically, we have the following results for the resource production process defined by (13):

- 1) Portfolio variance is bounded above by the variance of the portfolio with full investment in the high risk activity, and bounded below by zero, if the activities are perfectly negatively correlated.

- 2) When activities exhibit diminishing marginal productivity or competitive interdependence, an interior portfolio that reduces risk and increases expected output can exist.
- 3) When activities exhibit increasing marginal productivity or complementary interdependence, an interior portfolio that both increases risk and decreases output can exist.

These results provide important insights about the nature of risk and return in the renewable resource manager's problem. In the next section we demonstrate the renewable resource portfolio model derived above in an application to waterfowl management.

APPLICATION OF RESOURCE PORTFOLIOS TO WATERFOWL MANAGEMENT

Waterfowl managers use a wide array of direct population management activities and indirect habitat management activities to increase waterfowl populations. Rashford (2006), using simulated response data for mallards (*Anas platyrhynchos*), found that mallard response exhibits diminishing marginal productivity to many management activities and some activities are competitively interdependent. To demonstrate the effect of these properties on the EV-efficient portfolio set, we use the same simulation model to generate response data for three management activities, 1) the construction of artificial nest structures (NS), the construction of fenced planted cover (PCF) and the retirement of cropland (CR). NS and PCF exhibit diminishing marginal productivity while PCF and CR are technically competitive.

Mallard response data is simulated using the Mallard Productivity Model (MM). 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 relationships between landscape characteristics and waterfowl biology. Landscapes are defined using 26 habitat types included in the MM; example habitat types include fall-plowed grains, semi-permanent wetlands, and land enrolled in the Cropland Reserve Program (see Mack (1991) for a full description of habitat types). Management activities are simulated by manipulating habitat categories to mimic the effect of management. 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. Since the source of uncertainty in the MM is a function of habitat conditions, active management can alter the distribution of mallards produced on the landscape.

We simulate the application of NS, CR and PCF on the landscape described in table 4.1. Simulations are performed for 46 unique combinations of the three activities that satisfy a budget of \$3000 (see Appendix A1) using management activity cost data from Rashford (2006), which indicates activity costs of \$23/structure, \$33/acre and \$110/acre for NS, CR and PCF, respectively. Response is measured using mallard recruits, which indicates the number of new individuals produced during the nesting

season. The mean and variance for each simulation is calculated using point estimates from 300 simulations of each portfolio.

Table 4.1. Habitat Characteristics of the Simulated Landscape (*total area is 2000 acres*)

	Acres (<i>% of total area</i>)
Grain Crops	1385 (69.3)
Grassland	191 (9.5)
Hayland	134 (6.7)
CRP	0 (0)
Seasonal Wetland	147 (7.4)
Semi-Permanent	
Wetland	99 (4.9)
Temporary Wetland	10 (0.5)
Permanent Wetland	3 (0.1)
Other	31 (1.6)

Simulated Mallard Management Portfolios

The simulated mallard management portfolios have some of the properties derived with the analytical model (figure 4.7). At full investment (i.e. the entire \$3000 budget invested in one activity) NS are the most productive and risky activity with expected output of 98 recruits and a standard deviation of 42 (see table Appendix A.6). PCF produces 49 recruits with a standard deviation of 16 at full investment, while CR, the least productive and least risky, produces 35 recruits with a standard deviation of 11. Consistent with the results of financial portfolios, diversifying reduces risk. That is, combining NS with PCF or CR can reduce the standard deviation of recruits produced. If

a waterfowl manager were concerned with minimizing population fluctuations, to reduce costly adjustments to harvest regulations for example, they might consider using a diversity of management activities. Investing half the budget in NS and CR, for example, produces 92 recruits on average with a standard deviation of 35.

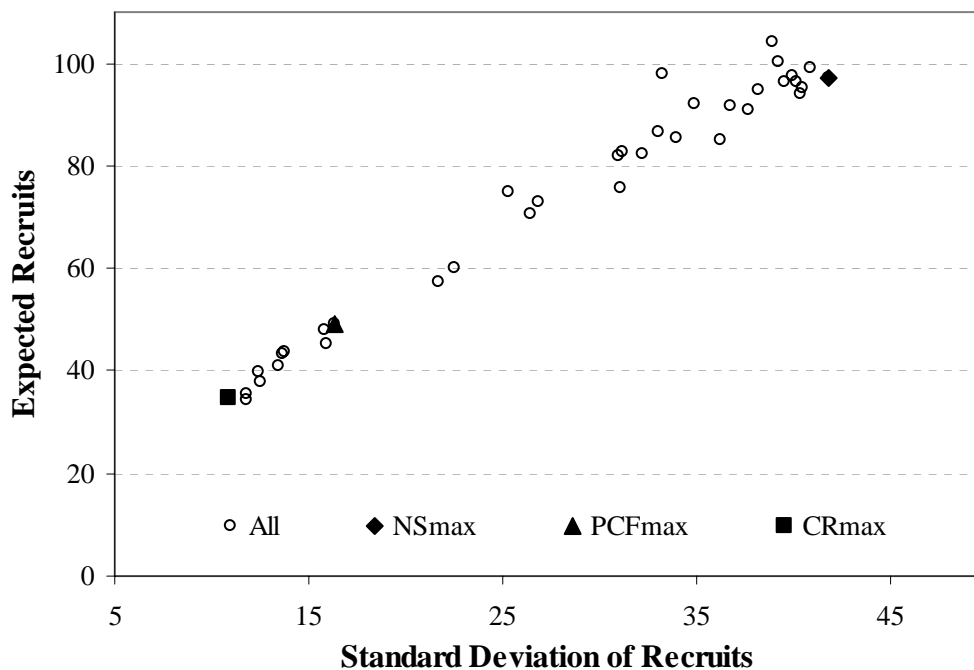


Figure 4.7. Simulated mallard management portfolios. NSmax, PCFmax and CRmax represent the portfolios with the full \$3000 budget invested in NS, PCF and CR, respectively.

The 50/50 investment in NS and CR also highlights the difference between mallard management portfolios and standard portfolios of financial asset. In the standard model with the assumption of constant and independent returns, the return on a 50/50 investment in NS and CR should only produce approximately 67 recruits. The fact this combination produce 92 recruits suggests that either, NS and CR are highly

complementary, one or both of the activities exhibit diminishing marginal returns, or both. Rashford (2006) did not identify complementary interdependence between NS and CR but did find that NS exhibited significant diminishing marginal productivity. Due to the diminishing marginal productivity of NS, it must be the case that the gains in recruits from increasing CR outweigh the loss in recruits from decreasing NS, as derived analytically in the previous section.

The simulated portfolios display some of the critical departures from the standard model derived analytically: interior portfolios that both increase output and decrease risk. This occurs for portfolios with slightly less than full investment in NS, with the balance applied to the other activities. Once again, this is most likely the result of diminishing marginal productivity to NS. The maximum output portfolio occurs when 90% of the budget is invested in NS and 10% in CR. This portfolio generates expected output of 104 recruits, five more recruits than full investment in NS, with a standard deviation of 39, three recruits less than full investment in NS. Although the maximum output portfolio is only slightly more productive and less risky than full investment in NS, this result suggests that at a larger scale (i.e. larger budgets), or with a larger set of management activities, interior portfolios could differ significantly from the full investment portfolios. Note that the maximum output combination is also cost-effective, since it maximizes expected output for the given budget. A highly diversified portfolio with 25% in CR, 50% in NS and 25% in PCF generates approximately the expected output (98) as full investment in NS but reduces risk significantly with a standard deviation of 33.

CONCLUSION

Portfolio theory provides a simple and intuitive model with which renewable resource managers can consider the tradeoff between risk and return. Prior applications of portfolio theory to renewable resource management have demonstrated the utility of the approach; however, they have all maintained the standard portfolio model assumptions of constant marginal productivity, and have all been applied to resources for which market prices exist. In this paper we move towards a general portfolio theory for renewable resources that allows these assumptions to be relaxed. We specifically modify the standard portfolio model to account for the variable marginal productivity of resource management activities and the lack of resource prices.

We use a simple quadratic response function to analytically demonstrate that the set of EV-efficient resource portfolios can differ substantially from those derived under the assumptions of the standard portfolio model. Specifically, 1) with diminishing marginal productivity to management activities or complementary interdependence there can exist a diversified portfolio that both reduces risk and increases expected return, and 2) with increasing marginal productivity to management activities or competitive interdependence between management activities there can exist a diversified portfolio that both increases risk and decreases expected return. These results are confirmed in an application to waterfowl management, which support the relevance of this approach for applied analyses.

This first step towards a more general portfolio theory of renewable resources suggests the following research and policy implication: research that observes total

returns from a set of resource portfolios and assumes that the portfolios can be combined to generate expected returns consistent with a linear combination, as in the standard portfolio model, may prescribe faulty management recommendations. At best, they will underestimate the portfolios expected returns by not accounting for diminishing marginal productivity or complementary interdependence. At worst, they will overestimate the portfolios expected returns and underestimate risk, which could, in the case of species management programs, increase the risk of having to list the species as threatened or endangered.

Future research could make progress in several areas. Analytical research could examine the effects of alternative response function specifications, such as density-dependent logistic growth, on the properties of the EV-efficient frontier. Additional complexity of the response function will, at some point, exhaust the intuitive appeal of the portfolio approach, but more research is needed to identify the model's limits. Empirically, the utility of the modified portfolio model needs to be tested using actual, rather than simulated, resource management data. A meta-analysis of existing applications may also be useful to identify any settings in which deviations from the standard model assumptions exist, and to examine whether those deviations have important management implications.

ENDNOTES

¹² Meyer (1987) shows that the EV and expected utility rankings are consistent under relatively unrestrictive assumptions on the distribution of returns.

¹³ Note that if the cost per unit were not constant, as is the case if transactions costs are dependent on trade volume, then the rate of return will be dependent on x_i and the assumption of constant rate of return would be violated.

¹⁴ If the error term was additive, every interior portfolio would have identical variance and there would be no potential for diversification to reduce risk. There are, however, many alternative error structures that interact the error term with the choice variable. We chose the error structure given in (11) to make the analytical results more tractable. The general results derived in this article also hold for more general multiplicative error structures, such as $y_i = (a_i x_i + b_i x_i^2) \varepsilon_i$.

¹⁵ See Appendix A4 for the derivation of the expected output in (15) and variance of output in (14).

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ESSAYS ON THE ECONOMICS OF WILDLIFE CONSERVATION

CHAPTER 5

GENERAL CONCLUSION

Benjamin S. Rashford

A need exists for analyses that examine the joint role of biological response and economic costs in determining the least-cost approach for achieving wildlife management objectives. Such analyses would improve the cost-effectiveness of wildlife management, and in combination with demand-side analyses, allow a complete analysis of the benefits and costs of alternative wildlife management projects, leading to more efficient allocation of conservation resources. This dissertation consists of three manuscripts that integrate economics and ecology to inform wildlife management decisions.

The first manuscript (Chapter 2), entitled *The Economics of Wildlife Production*, demonstrates the joint manner in which ecology and economics interact to determine cost-effective wildlife management. Results from the analysis demonstrate three points that have important implications for wildlife management: 1) biological response and economic cost jointly determine least-cost management plans, 2) non-linearity of the biological response function should be modeled explicitly to identify cost-effective management plans, and 3) the least-cost management plan depends on the chosen population objective.

The second manuscript (Chapter 3), entitled *Improving the Cost-Effectiveness of Ecosystem Management: An Application to Waterfowl Production*, extends the first manuscript in the following ways: 1) it considers the effect of landscape heterogeneity on the solution to the least-cost management problem, and 2) it examines and discusses policy implications of the relative cost-effectiveness of primary land-use conservation activities versus direct wildlife conservation activities in a landscape dominated by agricultural production. Results indicate that the simultaneous application of

management activities to multiple landscapes, rather than to a single landscape, reduces the cost of achieving population objectives. Results also indicate that direct wildlife conservation activities are relatively more cost-effective than primary land-use conservation activities, which suggest that there is common ground between conservation interest and agricultural landowners.

The third manuscript (Chapter 4), entitled *Towards a Portfolio Theory for Renewable Resource Management*, contributes to the growing literature on renewable resource management under uncertainty. It demonstrates that standard financial portfolio theory, in some cases, cannot be used to analyze risk-return tradeoffs in a renewable resource management context without violating some of the model's assumptions. This manuscript modifies the standard portfolio model to account for diminishing or increasing marginal productivity and activity interdependence, and demonstrates that the resulting set of mean-variance efficient portfolios display properties that are not possible in the standard portfolio model.

A common theme emerges from the three manuscripts. The non-linearity of biological response has important economic implications in a world of certainty or uncertainty. Each manuscript elucidates the effects of a more thorough treatment of non-linear biological response on the solutions to wildlife management problems. They show that a response function that exhibits diminishing marginal productivity and technical interdependence increases the difficulty of identifying least-cost combinations of management activities, and mean-variance efficient management portfolios. Simple rules-of-thumb do not exist, and no single management plan is cost-effective for all

population objectives. Landscape heterogeneity, which is introduced in the second manuscript, adds an additional layer of complexity to the management problem; however, it also provides cost-reducing opportunities. The third manuscript demonstrates that a more complex biological response function can, in fact, generate results that are not possible with a simple response function. For example, diversifying a portfolio of management activities, in some cases, reduces risk and increases returns; while, in other cases, it increases risk and reduces returns. The potential for these outcomes to occur would not be revealed if a linear response function were assumed. More thorough treatment of complex response functions may not be necessary in all future work. However, the results of this dissertation provide insight about the potential consequences of using a linear biological response function when the true function is non-linear.

This research could be expanded in a variety of ways. First, sensitivity analyses could be conducted to determine the conditions that necessitate using a non-linear response function, rather than a simplified linear function. Second, the biological simulation model on which this dissertation relies affords economists an opportunity to ask data-intensive questions without the need for customized biological experiments. Simulation models for other wildlife species could be used to identify least-cost or mean-variance efficient management plans for those species. Economists and ecologists are also increasingly interested in management questions that involve multiple species, as well as other types of ecosystem services. Multi-species, or ecosystem-level simulation models might be useful in addressing these questions. Finally, the third manuscript only scratches the surface of potential research in the area of renewable resource management

portfolios. Much additional work is needed to fully develop a portfolio theory for renewable resources. Immediate research needs include determining whether the results reported here hold for other response function specifications and error structures.

This research also highlights a need for continued cooperation between ecologists and economists at all stages of research. Specifically, a need exists to solicit feedback from ecologists about the questions economists are addressing, as compared to the questions that ecologists would like to have addressed. Economists then need to provide feedback to ecologists regarding the types of data and biological models required to answer those questions. Research results from one discipline can also inspire work in the other. This research, for example, suggests a need for biological experiments that identify, more precisely, the level at which various management activities display diminishing marginal productivity and technical interdependence. Such results could be useful to wildlife managers, particularly those who face shrinking management budgets.

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APPENDICES

A.1. Determination of Management Costs

The per-unit cost (input price) of each management activity is needed to determine optimal management strategies. This section documents the per-unit cost of each management activity, as used in this study (see table 8 for a summary of the data). All prices are adjusted to US\$2004 using the CPI.

Cropland Retirement (CR)

The cropland retirement (CR) management strategy converts active cropland to idle farmland, much like the U.S Farm Bill's Cropland Retirement Program (CRP). The opportunity cost of retiring farmland is, largely, foregone rents to the landowner. One potential measure of the cost of cropland retirement, therefore, is the rental rate of farmland. Rental rates in the central prairies differ by land use (cropland, hayland and pastureland) and location. Rental rates for North and South Dakota are presented in Table A1.

Table A1. Rental rates for non-irrigated farmland

	South Dakota (\$2004/acre)		North Dakota (\$2004/acre)		
	Cropland	Pasture	Cropland	Pasture	Hayland
Mean	43.29	20.30	33.47	11.98	24.59
Median	37.07	20.63	31.10	11.45	21.8
Max	105.42	42.21	73.50	24.20	50.6
Min	12.08	5.88	20.90	7.30	15.6

*source: National Agricultural Statistics Service (2004)

Alternatively, per-acre payments to CRP lands in the PPR could be used to approximate per-unit cost of management activity CR. CRP payments are presented in Table A2.

Table A2. CRP payments in the PPR (\$2004/acre)

	North Dakota	South Dakota	Iowa	Minnesota	Montana
Mean	33.15	51.54	100.85	74.30	32.80
Median	31.24	48.63	103.74	78.44	32.48
Max	56.72	88.49	127.32	106.79	44.19
Min	21.81	23.39	69.13	39.40	24.97

*source: Farm Services Agency (2003)

CRP payments are similar to the rental rate of cropland in North and South Dakota, suggesting that the approaches are similar proxies for the opportunity cost of cropland retirement. Rental rates are readily available for all three farmland uses (cropland, hayland, and pasture), while CRP payment data does not distinguish between land uses; therefore, we use rental rates to approximate the cost of CR.

Rental rates have significant in-state variation, which is largely attributable to heterogeneous land quality across rented parcels. Land quality heterogeneity is incorporated in the model by assigning different land use costs to the three simulated habitats. The three simulated habitats, which have different levels of productivity for agriculture and waterfowl, are defined as follows:

- 1) “Bad Habitat” – Intensive agriculture dominates the landscape; few wetlands remain and baseline waterfowl productivity is low as a result.
- 2) “Average Habitat” – Agriculture dominates the landscape, but many productive wetlands remain and baseline waterfowl production is significantly higher as a result, as compared to the Bad Habitat.
- 3) “Good Habitat” – Conservation dominates the landscape; agricultural activities are limited to hay and pasture. Wetlands are abundant, and baseline waterfowl productivity is high as a result, as compared to the other landscapes.

We assume that landscapes dominated by agriculture have the highest quality land for agricultural production, and thus the highest opportunity cost of cropland retirement. The landscape, “Poor Habitat,” is therefore assumed to have the highest opportunity cost, and is assigned approximately the maximum cropland rental rate for North Dakota. The landscape, “Average Habitat,” is assumed to have a moderate opportunity cost, and is assigned the mean cropland rental rate for North Dakota. The landscape, “Good Habitat,” is assumed to have the lowest opportunity cost. Good Habitat contains only hayland and pasture, so it is assigned the mean hayland rental payment for North Dakota. The per-acre costs of management activity CR are presented in Table A3.

Some might argue that the presence of cropland implies a particular land quality and therefore cropland on each landscape should be assigned the same opportunity cost. Differences in opportunity cost can still be justified by assuming that a landscape with more wetlands has lower returns to cropland due to nuisance costs associated with the avoidance of wetlands during cultivation. This additional cost reduces the opportunity cost of removing land from agricultural production.

Table A3. The assigned per-acre cost of management activity CR.

Landscape	Cost per Acre (\$2004)
Poor Habitat	70.00
Average Habitat	33.00
Good Habitat	25.00

No-Till Cropping (NT)

No-till cropping refers to the practice of seeding grain crops into the stubble of last year's crop. NT affects crop yield and input costs, and thus returns to land. These effects differ significantly by region, soil type, weather, crops produced, input and output prices, risk preferences and management skill (Fox et al., 1991, Orlick, C. C. et al., 1995). The opportunity cost of NT can be measured as the difference in net revenue of crops grown under conventional versus no-till practices (Manley et.al, 2005). Several studies have analyzed the cost of no-till cropping in the PPR. Results from representative studies are summarized in Table A4. Average net revenue loss, across studies, is approximately \$15.00 per acre. We therefore assume that NT has an opportunity cost of \$15 per acre.

Table A4. Cost of no-till cropping in the PPR (\$2004)

Study	Region	Net revenue/acre lost due to NT
Smolik, J. D & T. L. Dobbs (1991)	South Dakota	\$9 – 12
Zentner R. P. et al. (1991)	Saskatchewan	\$16 – 19
Zentner R. P. et al. (1996)	Saskatchewan	\$9 – 13
Smith E. G. et al. (1996)	Alberta	\$28
Kurkolova L. et al. (2003)	Iowa	\$3 – 7
Manley J. et al. (2005)	Corn Belt/Prairies	\$20 – 21

Delayed Hay (DH)

The delayed hay management activity postpones the hay harvest until July 15th to protect nesting ducks. Hay harvest dates vary in the central prairies, but generally begin in mid to late June; two harvests typically occur during the summer. Delaying the hay harvest reduces the hay's nutritional value (Norton, N. A. et al., 1997) and may reduce

the number of harvests. The opportunity cost of DH is, therefore, the foregone value of hay harvested at its optimal nutritional value and of the maximum number of harvests. Unfortunately, few studies estimate the cost of delayed hay. Managers of the Prairie Pothole Joint Venture at Devils Lake, North Dakota assigned a land-use cost of \$10 per acre to DH (USFWS, 1996). They included an additional cost of about \$2 per acre for contracting and management; it is not clear whether they included the opportunity cost of foregone rents.

A simple estimate of the opportunity cost of DH is the average returns to hayland. The five-year average market prices for alfalfa and other hay (1999-2003) are \$53.40 and \$37.10 per ton, respectively (NDSU Extension Service, 2005). With average total yields in a two-cut system of approximately 3.5 tons per acre, and production costs of approximately \$60 per acre (NDSU Extension Service, 1994), the returns to hay should fall in the range of \$70 to \$117 per acre. A single harvest is expected if harvest is postponed until mid-July, and the average yield falls to 2.6 tons per acre (NDSU Extension Service, 1994). Foregone net revenue due to DH, without accounting for quality losses, is therefore \$24 to \$28 per acre. These estimates are crude, but capture the major components of a landowner's opportunity cost of delaying the hay harvest. Therefore, we assign DH an annual cost of \$25 per acre.

Nesting Structures (NS)

The management activity NS constructs over-water nesting structures to reduce predation on waterfowl nests. Nesting structures have many forms, including open-

topped cones, cylinders (hen houses), or concrete culverts, which are the most commonly used form. The costs of alternative forms of nest structure (table A5) differ greatly.

Table A5. Cost of nest structures (\$2004)

Source	NS Type	Const- ruct (\$/ac)	Main- tain (\$/ac)	Useful Life
Doty H. A. et al. (1975)	Open Cone	65.14 ¹	11.49	20
Lokemoen J.T. (1979)	Open Cone	94.71	6.87	20
Marcy L. E. (1986)	Cylinder	49.09	5.45	7 – 20
USFWS, HAPET (1996)	Culvert	248.00	37.00	20
Dakota Nesting Structures (2005)	Cylinder	59.00	NA	NA
Fisher, J. (2005)	Cylinder	40.00	12-20	10 – 20

¹*Includes materials cost, but not labor cost.*

Cylinder nesting structures are consistent with those simulated in the Mallard Model, so we adopt this form. Fisher (2005) provides cost estimates for cylinder nesting structures in the PPR. We therefore assign construction and installation costs of \$40 and a maintenance cost of \$20, per nesting structure; we assume a 20-year useful life. The maintenance cost may seem high relative to the construction cost; however, proper maintenance involves visiting each structure prior to the breeding season, typically during winter conditions, to reset it, if necessary, and replace old straw. Proper maintenance ensures that nesting structures function well each breeding season. The required maintenance materials are inexpensive, but the required time and labor are relatively expensive (Fisher, J. 2005). The annualized cost of nesting structures adopted in this study is \$22.90 per structure.

Planted Cover (PC)

The planted cover management activity plants a mixture of grasses and legumes to provide wildlife cover. Costs include the cost to establish the cover (seeds, equipment, labor, etc.), and the opportunity cost of land. Cost estimates from the literature are presented in Table 6.

Table A6. Cost of planted cover (\$2004)

Source	Establish (\$/acre)	Land Use (\$/acre)	Useful Life	Annual Cost (\$/acre)
Lokemoen J.T. (1979)	29.00	1.64 ¹	10	17.24
USFWS, HAPET (1996)	48.00	36.00 ²	10	53.90

¹*The yearly maintenance cost; no land use cost was estimated in this study.*
²*Planted cover was established on cropland in this study, not grassland.*

Cost to establish PC is assumed to be \$45 per acre with a 10-year useful life. Unlike the studies in table 6, we assume that PC is established on pastureland (i.e. grassland) at a land-use cost of \$12 per acre, which is consistent with the average rental rate of pastureland in North Dakota (table 1). The cost of PC, annualized over the life of the activity, is \$17.50 per acre.

Planted Cover Fenced (PCF)

The management activity “planted cover fenced” is the same as the activity PC, except that the planted cover is surrounded by an electric fence to reduce predators’ access to nests located in the fenced area. Land use cost and the cost to establish cover are the same as those assumed for PC. The cost of constructing predator-exclusion

fences must be included. Table A7 presents the cost of planted cover fenced as reported in the literature.

Table A7. Cost of planted cover fenced (\$2004)

Source	Establish (\$/acre)	Maintain (\$/acre/yr)	Land Use (\$/acre)	Useful Life	Annual Cost
Lokemoen J.T. (1979)	106.33	15.53	NA	20	23.40
Lokemoen J.T. et al., (1982)	155.35	NA	NA	20	11.40
Grenwood R. J. et al. (1990)	197.50 ¹	NA	NA	NA	14.50
USFWS, HAPET (1996)	1239.50	4.65	24.00 ²	10	119.90

¹*Includes materials cost only; labor cost is excluded.*

²*Seventy-five percent established on cropland.*

It is difficult to compare fencing costs across studies. Fence design and technology has changed over time, as our understanding of the biological effectiveness of alternative fence designs has improved. Studies do not construct fenced plots of equivalent size (plots range from 21 to 80 acres), which is problematic because there are diseconomies of scale in the cost of fence construction. Finally, many studies exclude maintenance and land use costs from their cost estimates.

The Prairie Pothole Joint Venture project (USFWS, HAPET 1996) provides the most up-to-date fence design. Therefore we assume a fence construction cost of \$1200 per acre, an annual maintenance cost of \$5.00 per acre, and a 20-year useful life. We assume that PCF is established on pastureland, so a yearly land use cost of \$12 per acre is assumed. A cost of \$67 per acre to establish planted cover is also assumed. Note that the useful life of PCF is 20 years, while the useful life of PC is only 10 years. This implies that cover must be established twice in PCF, once in year 1 of the project, and again in year 11. We therefore assume a cost of \$40 per acre for the initial establishment, and a

cost of \$27 for the second establishment (which is the discounted value of \$40 per acre over a ten-year period). The total annualized cost of PCF is \$110.20 per acre.

Wetland Restoration (WR)

The wetland restoration management activity restores semi-permanent wetlands that were previously drained for use as cropland. Wetland restoration projects vary in their scope and complexity, from expensive engineering and construction projects to simple projects that can be completed with farm equipment. The type of restoration project assumed for this study is described in USFWS (1996); they also provide a cost estimate of \$66.00 per acre (\$2004) for the restoration of a 10-acre semi-permanent wetland. This cost includes \$57.00 per acre to survey and design the wetland, and an additional \$1.57 per acre per year to maintain the wetland. A 30-year useful life is assumed. We assume that wetlands are created in cropland, so a land use cost of \$25, \$33, or \$70 per acre is added (depending on the landscape—see the description of CR for more information). The annualized cost of WR for each landscape is \$30, \$38 and \$75 per acre, per year, respectively.

Predator Control (PRED)

The predator control management activity removes waterfowl nest predators by lethal or live traps. Low waterfowl breeding success in the PPR has been attributed, largely, to nest depredation, which decreases nest-success rates (Sovada, M. A. et al. 2001). The most direct method for reducing depredation is to reduce the population of nest predators. Predator trapping has been shown to dramatically increase nest success in

the PPR (Hoff, M. J. 1997, Garrettson, P. R. et al. 1996, Duebbert, H. F. and J. T. Lokemoen 1980).

Little information on the cost of predator control exists. Garrettson (personal communication) paid professional trappers \$19,000 to trap 10,250 acres (4150 ha) for 5 months, resulting in a per acre cost of \$1.85. Hoff (1996) paid \$19,000 to trap 24,527 acres (9930 ha) for 3.5 months, resulting in a per acre cost of \$0.77. Brice (2005) paid professional trappers, on average, \$21,500 to trap 23,040 acres for 4 months, resulting in a per acre cost of \$0.93. These projects differ by the size of the areas trapped and the duration of trapping, but they apply similar trapping methods and report similar increases in nest success. This suggests that the cost of PRED falls in the range of \$0.75 to \$2.00 per acre. We assume a conservative cost for predator control of \$2.00 per acre on all simulated landscapes.

Table A8. A summary of management activity costs used in this study.

Management Activity	Cost by Landscape (\$2004/acre)		
	Poor	Average	Good
Cropland Retirement (CR)	70.00	33.00	25.00
No-till Cropping (NT)	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
Wetland Restoration (WR)	75.00	38.00	30.00
Predator Control (PRED)	2.00	2.00	2.00

A2. Derivation of the portfolio distribution in the standard model

First, to derive expression (8), note that

$$RR_p = \frac{\sum_{i=1}^n (p_i x_i - w_i x_i)}{\sum_{i=1}^n w_i x_i} = \frac{\sum_{i=1}^n x_i (p_i - w_i)}{W_p}.$$

Next, multiply each term in the sum of the numerator by the associated $\frac{w_i}{w_i}$ to get

$$RR_p = \frac{\sum_{i=1}^n \left(\frac{w_i}{w_i} \right) x_i (p_i - w_i)}{W_p} = \sum_{i=1}^n \frac{w_i x_i}{W_p} \frac{(p_i - w_i)}{w_i} = \sum_{i=1}^n \alpha_i RR_i.$$

A3. Derivation of EV-efficient frontiers for $n = 2$ in the standard model¹

We consider the three cases of $\rho = 0$, $\rho = 1$, and $\rho = -1$. First, when $\rho = 0$, the variance of R_p , substituting $\alpha_2 = (1 - \alpha_1)$ and $\rho = 0$, simplifies to

$$(a.1) \quad \sigma_p^2 = \alpha_1^2 \sigma_1^2 + (1 - \alpha_1)^2 \sigma_2^2.$$

Minimizing σ_p^2 and solving for α_1 results in an interior solution, where

$$(a.2) \quad \alpha_1 = \frac{\sigma_2^2}{\sigma_2^2 + \sigma_1^2}.$$

The interior solution always involves positive investments in both assets because $\sigma_i^2 > 0 \forall i$ combined with (a.2) implies that $0 < \alpha_1 < 1$, and subsequently $0 < \alpha_2 < 1$. Checking this interior solution against the corner solutions, $\alpha_1 = 0$ and $\alpha_1 = 1$, verifies that interior solution minimizes portfolio variance. To see this, substitute (a.2) into (a.1) to

¹ The derivations in this section largely follow Elton and Gruber (1984). The interested reader should consult their text for more detail and numerical examples.

get $\sigma_p^2 = \frac{\sigma_1^2 \sigma_2^2}{(\sigma_1^2 + \sigma_2^2)}$, which is always less than the smaller of σ_1^2 and σ_2^2 . When $\rho = 0$

diversification, therefore, reduces risk.

When $\rho = 1$, the variance of R_p simplifies to

$$(a.3) \quad \sigma_p^2 = \alpha_1^2 \sigma_1^2 + (1 - \alpha_1)^2 \sigma_2^2 + 2\alpha_1(1 - \alpha_1)\sigma_1\sigma_2,$$

by substituting $\sigma_{12} = \rho\sigma_1\sigma_2$. This is a quadratic form so we can write

$$\sigma_p^2 = [\alpha_1\sigma_1 + (1 - \alpha_1)\sigma_2]^2, \text{ and}$$

$$(a.4) \quad \sigma_p = [\alpha_1\sigma_1 + (1 - \alpha_1)\sigma_2].$$

The portfolio expected return is given by

$$(a.5) \quad \mu_p = \alpha_1\mu_1 + (1 - \alpha_1)\mu_2.$$

Thus, the expected return and variance of the portfolio are both linear combinations of the expected return and variance of each asset. Solving for α_1 in (a.4) and substituting the result into (a.5) results in

$$\mu_p = \left(\frac{\sigma_p - \sigma_2}{\sigma_1 - \sigma_2} \right) \mu_1 + \left(1 - \frac{\sigma_p - \sigma_2}{\sigma_1 - \sigma_2} \right) \mu_2.$$

Simplifying this expression results in

$$(a.6) \quad \mu_p = \left(\mu_2 - \frac{\mu_1 - \mu_2}{\sigma_1 - \sigma_2} \right) \sigma_2 + \left(\frac{\mu_1 - \mu_2}{\sigma_1 - \sigma_2} \right) \sigma_p,$$

which is the equation of a straight line in EV-space. Since the EV-efficient frontier is a straight line, σ_p^2 is never less than the smaller of σ_1^2 and σ_2^2 . The minimum variance portfolio is, therefore, the portfolio that fully invests in the asset with minimum variance.

When $\rho = -1$, the portfolio variance simplifies to

$$(a.7) \quad \sigma_p^2 = \alpha_1^2 \sigma_1^2 + (1 - \alpha_1)^2 \sigma_2^2 - 2\alpha_1(1 - \alpha_1)\sigma_1\sigma_2,$$

which is equivalent to either

$$(a.8) \quad \sigma_p = [\alpha_1\sigma_1 - (1 - \alpha_1)\sigma_2],$$

or

$$(a.9) \quad \sigma_p = [-\alpha_1\sigma_1 + (1 - \alpha_1)\sigma_2].$$

We took the square root of (a.7) to obtain (a.8) and (a.9), which is a real number only if the right hands side of (a.8) and (a.9) are positive. Since the right hand side of (a.8) is always positive if (a.9) is negative and vice versa, there is a unique solution for the

expected return and variance of any portfolio. To find the minimum variance portfolio,

minimize (a.7) and solve for α_1 . This produces the interior solution $\alpha_1 = \frac{\sigma_2}{\sigma_1 + \sigma_2}$. The

interior solution will always include positive investment in each asset because σ_1 and σ_2

are strictly positive and $(\sigma_1 + \sigma_2) > \sigma_2$. The interior solution in this case achieves a

minimum variance portfolio that has in zero risk. To see this, substitute α_1 into (a.7); the

result is $\sigma_p^2 = 0$. Using (a.8) and (a.9) and following the same procedure as above, one

can readily show that the EV-efficient frontier is comprised of the following two line

segments in EV space:

$$\mu_p = \left(\mu_2 + \frac{\mu_1 - \mu_2}{\sigma_1 + \sigma_2} \right) \sigma_2 + \left(\frac{\mu_2 - \mu_1}{\sigma_1 + \sigma_2} \right) \sigma_p, \text{ and}$$

$$\mu_p = \left(\mu_2 + \frac{\mu_1 - \mu_2}{\sigma_1 + \sigma_2} \right) \sigma_2 - \left(\frac{\mu_2 - \mu_1}{\sigma_1 + \sigma_2} \right) \sigma_p.$$

A4. Derivation of the Distribution of Renewable Resource Portfolios

The expected output on resource portfolios is attained directly by taking $E[Y_p(\alpha)]$ and noting that $E[\varepsilon_i] = 0, \forall i = 1, \dots, n$ by assumption. The variance of portfolio output is derived as follows:

$$\begin{aligned}
 \text{var}(Y_p(\alpha)) &= \text{var} \left(\sum_{i=1}^n (\tilde{w}_i a_i \alpha_i + \tilde{w}_i^2 b_i \alpha_i^2) + \sum_{i=1}^n \sum_{j>i}^n \tilde{w}_i \tilde{w}_j c_{ij} \alpha_i \alpha_j + \sum_{i=1}^n \alpha_i \tilde{w}_i \varepsilon_i \right) \\
 &= \text{var} \left(\sum_{i=1}^n \alpha_i \tilde{w}_i \varepsilon_i \right) \\
 &= \sum_{i=1}^n (\alpha_i \tilde{w}_i)^2 \text{var}(\varepsilon_i) + 2 \sum_{i=1}^n \sum_{j>i}^n \alpha_i \alpha_j \tilde{w}_i \tilde{w}_j \text{cov}(\varepsilon_i, \varepsilon_j) \\
 &= \sum_{i=1}^n \sum_{j>i}^n \tilde{w}_i \tilde{w}_j \alpha_i \alpha_j \sigma_{ij}
 \end{aligned}$$

where σ_{ij} is the ij^{th} element of the variance-covariance matrix of ε_i .

A5. Derivation of the Bounds on the Portfolio Variance in the Renewable Resource Model

When $\rho = 1$ the portfolio variance is given by

$$(a.10) \quad \sigma_p^2(\alpha_1) = (\alpha_1 \tilde{w}_1 \sigma_1 + (1 - \alpha_1) \tilde{w}_2 \sigma_2)^2,$$

and the standard deviation by

$$(a.11) \quad \sigma_p(\alpha_1) = \alpha_1 \tilde{w}_1 \sigma_1 + (1 - \alpha_1) \tilde{w}_2 \sigma_2.$$

From (16), note that $\tilde{w}_1 \sigma_1$ is the standard deviation of the portfolio when $\alpha_1 = 1$, and

$(1 - \alpha_1) \tilde{w}_2 \sigma_2$ is the portfolio standard deviation $\alpha_2 = 1$. The standard deviation of a

portfolio when $0 < \alpha_1 < 1$ and $0 < \alpha_2 < 1$ is, therefore, a simple linear combination of these

extremes. Furthermore, since the $\arg \min(\sigma_p^2) = \arg \min(|\sigma_p|)$, the minimum variance portfolio occurs when $\alpha_1 = 1$ and $\alpha_2 = 0$, and similarly, the maximum portfolio variance occurs when $\alpha_1 = 0$ and $\alpha_2 = 1$. The portfolio variance in the renewable resource model when $\rho = 1$ is, therefore, bounded just as the standard model is bounded with $\rho = 1$.

When $\rho = 0$ the portfolio variance is given by,

$$(a.12) \quad \sigma_p^2 = \alpha_1^2 \tilde{w}_1^2 \sigma_1^2 + (1 - \alpha_1^2) \tilde{w}_2^2 \sigma_2^2.$$

Setting the derivative of (a.12) with respect to α_1 equal to zero and solving for α_1 produces

$$(a.13) \quad \alpha_1^* = \frac{\tilde{w}_2^2 \sigma_2^2}{\tilde{w}_1^2 \sigma_1^2 + \tilde{w}_2^2 \sigma_2^2},$$

which is a candidate for the $\arg \min(\sigma_p^2)$ because $\frac{\partial^2(\sigma_p^2)}{\partial \alpha_1^2} = 2\tilde{w}_1^2 \sigma_1^2 + 2\tilde{w}_2^2 \sigma_2^2$ is strictly

positive. Next, compare the interior solution, $\sigma_p^2(\alpha_1^*)$, to $\sigma_p^2(\alpha_1 = 1)$ and $\sigma_p^2(\alpha_1 = 0)$ (the portfolio variances that can be achieved by fully investing in either activity).

Substituting α_1^* into (a.12) and simplifying produces

$$(a.14) \quad \sigma_p^2(\alpha_1^*) = \frac{(\tilde{w}_1^2 \sigma_1^2)(\tilde{w}_2^2 \sigma_2^2)}{\tilde{w}_1^2 \sigma_1^2 + \tilde{w}_2^2 \sigma_2^2}.$$

This implies that: $0 < \sigma_p^2(\alpha_1^*) < \sigma_p^2(\alpha_1 = 1) < \sigma_p^2(\alpha_1 = 0)$, because

$\sigma_p^2(\alpha_1 = 1) < \sigma_p^2(\alpha_1 = 0)$ by assumption and $\tilde{w}_i^2, \sigma_i^2 > 0 \forall i$. The portfolio variance in the renewable resource model when $\rho = 0$ is therefore bounded above by the variance when

fully invested in the riskier activity, and below by some strictly positive variance that is strictly less than the variance when fully invested in the less risky activity (i.e. $\sigma_p^2(\alpha_1^*)$)

Lastly, when $\rho = -1$, the portfolio variance is given by

$$(a.15) \quad \sigma_p^2 = \alpha_1^2 \tilde{w}_1^2 \sigma_1^2 + (1 - \alpha_1^2) \tilde{w}_2^2 \sigma_2^2 - 2\alpha_1(1 - \alpha_1) \tilde{w}_1 \tilde{w}_2 \sigma_1 \sigma_2.$$

Setting the derivative of (a.15) with respect to α_1 and solving for α_1 produces

$$(a.16) \quad \alpha_1^* = \frac{\tilde{w}_2^2 \sigma_2^2}{\tilde{w}_1^2 \sigma_1^2 + \tilde{w}_2^2 \sigma_2^2},$$

which is a candidate for the $\arg \min(\sigma_p^2)$ because $\frac{\partial^2(\sigma_p^2)}{\partial \alpha_1^2} = 2\tilde{w}_1^2 \sigma_1^2 + 2\tilde{w}_2^2 \sigma_2^2 + 4\tilde{w}_1 \tilde{w}_2 \sigma_1 \sigma_2$

is strictly positive. Next, compare the interior solution,

$\sigma_p^2(\alpha_1^*)$, to $\sigma_p^2(\alpha_1 = 1)$ and $\sigma_p^2(\alpha_1 = 0)$ (the portfolio variances that can be achieved by fully investing in either activity). Substituting α_1^* into (a.15) and simplifying produces the desired result: $\sigma_p^2(\alpha_1^*) = 0$. Thus, when $\rho = -1$, there exists a portfolio with $0 < \alpha_1 < 1$ and $0 < \alpha_2 < 1$ that produces zero risk. The portfolio variance in this case is, therefore, bounded below by zero and above by $\sigma_p^2(\alpha_1 = 0) = \tilde{w}_2^2 \sigma_2^2$.

A.6 *Management Activity Levels Simulated to Generate Portfolio Data*

Portfolio Share			Activity Levels			Distribution of Recruits		
$\alpha(\text{CR})$	$\alpha(\text{NS})$	$\alpha(\text{PCF})$	CR	NS	PCF	Mean	Variance	St. Dev
0.1	0.9	0	9.09	117.39	0.00	104.12	1515.13	38.92
0.2	0.8	0	18.18	104.35	0.00	94.99	1641.81	40.52
0.3	0.7	0	27.27	91.30	0.00	93.86	1631.07	40.39
0.4	0.6	0	36.36	78.26	0.00	91.78	1352.21	36.77
0.5	0.5	0	45.45	65.22	0.00	92.09	1219.09	34.92
0.6	0.4	0	54.55	52.17	0.00	86.42	1096.81	33.12
0.7	0.3	0	63.64	39.13	0.00	82.82	972.25	31.18
0.8	0.2	0	72.73	26.09	0.00	73.03	723.64	26.90
0.9	0.1	0	81.82	13.04	0.00	57.37	472.58	21.74
0.1	0	0.9	9.09	0.00	24.55	47.81	251.76	15.87
0.2	0	0.8	18.18	0.00	21.82	45.17	252.91	15.90
0.3	0	0.7	27.27	0.00	19.09	43.51	188.33	13.72
0.4	0	0.6	36.36	0.00	16.36	43.19	186.03	13.64
0.5	0	0.5	45.45	0.00	13.64	41.08	180.61	13.44
0.6	0	0.4	54.55	0.00	10.91	39.61	154.53	12.43
0.7	0	0.3	63.64	0.00	8.18	37.80	157.07	12.53
0.8	0	0.2	72.73	0.00	5.45	35.55	139.51	11.81
0.9	0	0.1	81.82	0.00	2.73	34.46	138.56	11.77
0	0.9	0.1	0.00	117.39	2.73	98.94	1675.40	40.93
0	0.8	0.2	0.00	104.35	5.45	100.19	1544.12	39.30
0	0.7	0.3	0.00	91.30	8.18	96.43	1615.36	40.19
0	0.6	0.4	0.00	78.26	10.91	96.26	1562.27	39.53
0	0.5	0.5	0.00	65.22	13.64	94.70	1464.15	38.26
0	0.4	0.6	0.00	52.17	16.36	90.78	1423.07	37.72
0	0.3	0.7	0.00	39.13	19.09	85.21	1315.99	36.28
0	0.2	0.8	0.00	26.09	21.82	81.80	963.79	31.05
0	0.1	0.9	0.00	13.04	24.55	74.86	641.68	25.33
0.25	0.25	0.5	22.73	32.61	13.64	82.25	1041.99	32.28
0.5	0.25	0.25	45.45	32.61	6.82	75.84	965.83	31.08
0.25	0.5	0.25	22.73	65.22	6.82	98.09	1106.19	33.26
0.8	0.1	0.1	72.73	13.04	2.73	60.14	508.95	22.56
0.1	0.8	0.1	9.09	104.35	2.73	97.51	1601.98	40.02
0.1	0.1	0.8	9.09	13.04	21.82	70.48	701.95	26.49
0.33	0.34	0.33	30.00	44.35	9.00	85.40	1154.72	33.98
0	1	0	0.00	130.43	0.00	97.11	1755.39	41.90
0	0	1	0.00	0.00	27.27	48.97	266.20	16.32
1	0	0	90.91	0.00	0.00	34.55	118.96	10.91