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**Modeling Economic and Ecological Benefits of Post-Fire Revegetation
in the Great Basin: an Application of Markov Processes***

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Abstract

This study employs a Markov chain model of vegetation dynamics to examine the economic and ecological benefits of post-fire revegetation in the Great Basin sagebrush steppe. The analysis is important because synergies between wildland fire and invasive weeds in this ecosystem are likely to result in the loss of native biodiversity, less predictable forage availability for livestock and wildlife, reduced watershed stability and water quality, and increased costs and risk associated with firefighting.

The analysis is based on a parameterized state-and-transition model of vegetation change for Wyoming big sagebrush community in the Great Basin sagebrush steppe. This conceptual model was formulated into a quantitative, predictive model by implementing it as a Markov chain process that links vegetation change, management, and costs. Simulation results were used to develop cost curves for achieving ecological goals and to evaluate uncertainty in future vegetation conditions.

The Markov chain model shows that post-fire revegetation using either a native seed mix or crested wheatgrass was more effective than no revegetation for achieving ecosystem objectives. Further, post-fire revegetation with either seed mix cost less than no revegetation because of resulting reductions in fire suppression costs. Consequently, post-fire revegetation makes both ecological and economic sense, and the choice of seed mix should depend on the prioritization of management objectives.

Identifying the economic and ecological tradeoffs of different management strategies should enable improved management of the sagebrush-steppe, and Markov processes provide a straight-forward method for identifying these trade-offs.

Introduction

Synergies between natural disturbance and human-related environmental changes have resulted in rapid, widespread ecological change in a number of systems (Schlesinger et al. 1990; D'antonio and Vitousek 1992). Consequences include changes in community composition, new alternative vegetation states, and alteration of natural disturbance patterns. In turn, these changes can have extensive, often irreversible, ecological and economic impacts. Evaluating potential management strategies for these rapidly changing systems can be extremely difficult because of limited knowledge about the systems, nonlinear ecosystem dynamics, and difficult to predict costs. However, understanding the risks, tradeoffs, and possible outcomes of different management scenarios can lead to improvement in both ecological and economic efficiency.

Land managers must evaluate and prioritize a variety of economic and ecological criteria when making planning decisions. Thus, tools that link ecological outputs directly to economic inputs will likely facilitate efficient management planning. This paper applies Markov chain processes to operationalize a conceptual model of vegetation dynamics into a quantitative, predictive model that can be used to inform ecosystem management at regional scales. By linking vegetation change, management, and costs, the Markov model can be used to predict ecosystem change under different scenarios and to develop cost curves for ecological outputs. In addition, sensitivity analysis can be used to evaluate the effects of parameter uncertainty on model predictions, a feature that should aid land managers who often are constrained by data availability and large parameter uncertainty.

This methodology is applied to the Great Basin sagebrush steppe ecosystem, which is incurring rapid loss of its native vegetation, invasion of annual grasslands, and dramatic

increases in fire frequency. This ecosystem presents major conservation and management challenges because of the rapid rate of current changes, the large levels of uncertainty in the system, and the high costs associated with both fire suppression and restoration efforts.

The accepted theoretical model for describing current sagebrush steppe vegetation dynamics is the state-and-transition model (Laycock 1991; West 1999a, 1999b; West and Young 2000; Perryman and Swanson 2003). This conceptual model defines sets of discrete vegetation states and transition pathways and describes factors that affect the rate and probability of transition between states, such as livestock grazing, weather, and fire. By quantifying transition rates and probabilities between vegetation states, the state-and-transition model is easily formulated as a Markov process.

This Markov model of sagebrush steppe vegetation dynamics is employed to evaluate several appropriate restoration strategies, including post-fire revegetation with a native seed mix versus crested wheatgrass. Model simulations are used to evaluate the efficacy of post-fire revegetation for reducing management costs (the sum of revegetation and fire suppression costs), maintaining native vegetation on the landscape, and reducing the extent of annual grass monoculture. To evaluate these ecological and economic trade-offs, simulation results are used to develop cost curves for each ecological goal, examine and interpret uncertainty in model results, and identify the ratio of costs for which revegetation pays for itself through reduced fire suppression costs.

The paper is laid out as follows: Relevant background on the ecology, history, and management of the focal system are overviewed first. Details of the conceptual and Markov model, including sensitivity analysis, are described next, followed by the

formulation of the conceptual and Markov models for the sagebrush steppe. The next section describes solution methods, including model simulation, cost curve development, and calculations of the break-even cost ratio. Next the results and discussion of model applications are presented. Finally, the paper concludes with a discussion of the applicability of this modeling approach to other management strategies and systems.

Study System: The Great Basin Sagebrush Steppe

The Great Basin sagebrush steppe, which covers more than 206,000 square kilometers of the Great Basin, is experiencing rapid, large-scale ecosystem changes associated with alterations of the historic disturbance regimes and the introduction of non-native species, including: (i) loss of its native shrublands, (ii) the expansion of invasive, annual grasslands, and (iii) increases in fire frequency.

The pre-European sagebrush steppe community of the Great Basin was characterized by co-dominant sagebrush and perennial bunchgrasses. Periodic fires occurred on average every 40 to 100 years (Wright and Bailey 1982; Winward 2000) and maintained a mosaic of successional states on the landscape. The introduction of livestock in the 1860s, followed by the accidental introduction of a Eurasian annual grass, cheatgrass (*Bromus tectorum*), in the early 1900s, have resulted in the decline of native bunchgrasses and rapid spread of cheatgrass. Cheatgrass is highly flammable, and its invasion has greatly reduced fire intervals, with many areas burning on intervals shorter than five years (Whisenant 1990). Cheatgrass is well adapted to fire and successfully out-competes native species following fire, particularly in areas that burn frequently. Together, fire frequency and cheatgrass presence have resulted in an increasingly rapid conversion of

sagebrush steppe to cheatgrass monoculture (Trimble 1989; Grayson 1993).

These changes in vegetation and increased fire frequency are occurring at large scales and are having significant ecological and economic ramifications. Impacts include population declines for a number of species that depend on the sagebrush steppe, reduced value from recreation, decreased forage availability for livestock and wildlife, increased erosion and watershed deterioration, and increased risks and costs associated with fire fighting (Roberts 1991; BLM 1999).

Currently, the most common and effective method for controlling cheatgrass expansion and further range deterioration is to revegetate with perennial bunchgrasses following fire. The nonnative perennial grass, crested wheatgrass (*Agropyron* sp.), historically has been used because it provides quality forage for livestock, is well adapted to the climate of the Great Basin, and can prevent the invasion of cheat grass (Trimble 1989). However, large-scale revegetation with crested wheat grass often results in monocultures that are beneficial for livestock but do not provide suitable habitat for many native species. On the other hand, seedings using native grasses and forbs were largely unsuccessful in the past because the native grasses rarely grew early and rapidly enough to outcompete cheatgrass (West 1999a, 1999b). As of late, the success rate of native seedings has increased, but the high costs and limited availability of native seed still limit widespread use. However the movement toward using native seeds for revegetation is underway.

Model Development

State-and-transition models is the formal name given to a class of conceptual models used

to describe vegetation dynamics for a variety of rangeland systems in Australia, South Africa, and North America, including the Great Basin sagebrush steppe (Westoby, Walker, and Noy-Meir 1989; Laycock 1991). However, state-and-transition models are representative of a general class of vegetation models that describe discrete vegetation states and all potential transition pathways between those states that occur because of natural succession or disturbance events. Such models are useful for providing a conceptual description of vegetation dynamics, but they are less useful for evaluating management or future states of a system without some quantitative knowledge of transition rates or probabilities. To formulate any conceptual model of vegetation dynamics as a Markov model simply requires definition of vegetation categories and pathways between categories and some quantitative knowledge of transition rates or probabilities.

For application of Markov chains to vegetation modeling, each of K discrete vegetation states can be represented by a chain state x_i in the Markov process, where X is the 1 by K vector of chain states. The model is applied at the landscape level by allowing X_t to represent the predicted allocation of the landscape across vegetation types (i.e. percent of landscape in each vegetation type) at time t , such that the vector sums to 100% of the landscape.

P is a K by K matrix of transition probabilities p_{ij} , where p_{ij} is the probability of transitioning from vegetation type i to vegetation type j in each time period. These transition probabilities are functions of (i) natural rates of succession, (ii) probabilities of succession-altering disturbance events such as fire, and (iii) implementation and success of management. Defining the average time it takes to transition in one step from vegetation

state i to state j as the holding time m_{ij} , the transition probability between the two states in the absence of stochastic disturbance is

$$p_{ij} = \frac{1}{m_{ij}} \quad i \neq j \quad \text{Eq. 1}$$

and the probability of remaining in the same state is

$$p_{ii} = 1 - \frac{1}{m_i} \quad i = 1, 2, \dots, K \quad \text{Eq. 2}$$

as per Logofet and Lesnaya (2000). To incorporate disturbance events, such as fire, into the model, the probabilities of transitions are adjusted based on the probability of relevant disturbances and their impact. Management is incorporated in a similar manner, by adjusting transition probabilities based on whether management is implemented and the probability of its success. Simulating the model across time as $X_{t+1} = X_t P$ yields the predicted the proportion of the landscape in each vegetation state.

Sensitivity analysis can be performed by changing the values of transition probability matrix P and re-predicting X_t with the recursive relationship. Since in ecological models parameters are rarely known with certainty, a probability distribution for each parameter can be specified in advance. Parameter values randomly assigned from the specified distributions can be used to calculate P , and by rerunning the model with these different transition probability matrices, boot-strapped estimates of predicted outcomes can be obtained.

Model Formulation for the Sagebrush Steppe

Vegetation states and transitions for this application were adapted from state-and-transition models described by Laycock (1991), West (1999a, 1999b), West and Young (2000), Perryman and Swanson (2003), and Brackley (personal communication) and are presented in Figure 1. This model defines eleven states, representing each of the vegetation types and successional stages that occur in this system. The vegetation types include three successional stages of semi-pristine sagebrush step (states 1-3), dense sagebrush with sparse bunchgrass and cheatgrass understory (state 4), three successional states of cheatgrass dominated community with sagebrush potential (states 5-7), cheatgrass monoculture (state 8), and three successional states characterized by crested wheatgrass (states 9-11). Transitions between states result from fire events, natural succession, and revegetation management. Each of these vegetation categories and transitions were reviewed and approved by sagebrush steppe and rangeland experts (as per Logofet and Lesneya 2000). For a detailed description of this model, see Niell (2003).

The generalized matrix of transition probabilities for this system is presented in Table 1. In this matrix p_{ij} is probability of transitioning from vegetation type i to j in the absence of disturbance. This probability is adjusted based on the probability of disturbance by fire and the implementation of revegetation. The probability of a fire event in state i is denoted as $pfire_i$. Revegetation of areas post-fire influences transitions into either vegetation type 1 or 9, depending on the type of seeds used for revegetation. The probability of these transitions, V_i , is equal to the product of the proportion of area burned that is reseeded, s , and the probability that the reseeded is successful, $preveg$, such that $V_i = s * preveg$.

Estimates of transition times, fire frequencies, probabilities of successful revegetation post-fire, and costs were based on expert opinion and information in published literature as per Logofet and Lesneya (2000). Since there is no widespread agreement on the exact values of these parameters, each was treated as a unique uniform random variable with the lower bound given by the lowest estimate across sources and the upper bound by the highest estimate. For tables of all model parameter ranges, see Niell (2003). Parameter estimates for revegetation costs and success probabilities were most data-limited. It is generally agreed that, although success of revegetation with native seed is increasing, it is still lower than success of revegetation with crested wheatgrass. However, by selecting the high and the low estimates of revegetation success that were provided by experts, identical parameter ranges were identified for native seed and for crested wheatgrass. Thus, improved estimates of revegetation success rates might improve model predictions.

Solution Methods

In the following methods descriptions, “native vegetation” is the total percentage of the landscape in states 1- 3 combined, and “total area burned” is the total percentage of the landscape burned summed across time periods.

Estimates of transition times, fire frequencies, and revegetation success probabilities are defined as uniform random variables with upper and lower bounds. Therefore, to incorporate the random variables into the model, the model was simulated repeatedly using randomly assigned values to calculate a unique transition probability matrix, P , for each run. The same probability matrix was used across all time periods and

management strategies within a given run, and new parameter values were assigned for each successive run. This method enabled boot-strapped estimation of predicted vegetation conditions and costs.

Model simulations were designed to compare the ecological and economic trade-offs of a variety of post-fire revegetation strategies across a fifty year time horizon. Two methods were employed for examining the relative costs and efficacy of the different revegetation strategies. These methods are:

- developing (i) total cost curves for the reduction of cheatgrass monoculture and (ii) total, average, and marginal cost curves for the maintenance of native vegetation,
- calculating the break-even cost ratio between revegetation costs and fire suppression costs.

In addition, to examine the variance in predicted values that stems from uncertainty in model parameters, simulation results were graphed as cumulative probability distributions.

Cost Curves

Cost curves can provide information to managers regarding the cost of achieving different levels of each management objective and the most cost-efficient strategy for achieving those levels. To create the cost curves for maintenance of native vegetation and reduction of cheatgrass monoculture, changes in vegetation were simulated under 21 different revegetation strategies. These strategies included post-fire revegetation rates of 25%, 50%, 75%, and 100% of areas in states 4-6 that burned in each time period, and for each revegetation rate 5 different ratios of area seeded with a native seed mix versus crested wheatgrass were

compared. These ratios were 100:0, 75:25, 0:50, 25:75, and 0:100. In effect, this represents a 4 x 5 factorial design of reseeding rates and ratios. In addition to the 20 different revegetation strategies, a no revegetation alternative was included, resulting in a total of 21 strategies examined. Each model simulation was performed over a 50-year time horizon, and each particular management strategy was simulated 500 times.

Costs were calculated as the net present value of management costs across fifty years, assuming a 3 percent discount rate. Three percent represents a low rate of discounting that is reasonable for discounting of costs of social benefits accrued long-term (Loomis 2002).

Management costs were calculated as the sum of revegetation costs and fire suppression costs -- the two major variable costs in these management scenarios. Currently, fire suppression is applied to all fires in the Great Basin sagebrush steppe unless a fire extinguishes itself prior to the arrival of fire crews. Thus, fire suppression costs were calculated as the product of the discounted per acre fire suppression cost and the area burned each year, summed across 50 years. Likewise, revegetation costs were calculated as the product of the area revegetated each year and the discounted per acre cost of seeding, summed across 50 years.

Total cost curves for (i) the reduction of cheatgrass monoculture and (ii) the maintenance of native vegetation, were developed by plotting the net present value of management costs (averaged across 500 runs) against the average predicted area of (i) cheatgrass monoculture and (ii) native vegetation, in year 50, for each of the 21 management strategies. Based on these results, the total minimum cost curve, average costs curve, and marginal cost of maintaining native vegetation were calculated.

Break-even Cost Ratio

Post-fire revegetation is expected to reduce fire frequency by converting high fire frequency vegetation types characterized by cheatgrass into vegetation types with lower fire frequency. Thus, money spent on revegetation is expected to reduce the amount of money that would be spent on fire suppression. Given this relationship, there is a point where the cost of revegetation is mitigated by cost savings from unneeded fire suppression activities. More specifically, the break-even cost ratio for revegetation is the ratio of per acre revegetation costs to per acre fire suppression costs at which the cost of revegetation plus cost of fire suppression exactly equals the cost of fire suppression if no revegetation were implemented. The break-even cost ratio is, therefore, a function of the percent of the landscape revegetated and the number of years over which costs are averaged. The break-even cost ratio was calculated for a 50-year time period for 500 runs assuming 100% post-fire revegetation with native seed and a 3% discount rate. Calculations were based on the data from the cost curve simulations described above.

The break-even ratio of revegetation costs to fire suppression costs equals the difference between the area burned in the absence of revegetation and area burned with revegetation, divided by the total area revegetated, with each of these values discounted at 3%.

$$\text{break even cost ratio} = \frac{(\text{areaburned}_{\text{noreveg}} - \text{areaburned}_{\text{wreveg}})}{\text{areareveged}} \quad \text{Eq. 5}$$

Results and Discussion

Figures 2 and 3 show total cost curves for reducing the area of cheatgrass

monoculture and increasing the area of native vegetation on the landscape. The points plotted represent the predicted net present value of management costs per acre summed across 50 years (averaged across 500 runs, assuming a 3% discount rate) and response variable values in year 50 (averaged across 500 runs). The five lines in each figure represent different ratios of area reseeded with crested wheatgrass versus reseeded with a native seed mix: 100:0, 75:25, 50:50, 25:75, and 0:100, and the points along each curve represent strategies of revegetating none, 25%, 50%, 75%, and 100% of the areas in vegetation types 4-6 that burn. Total management costs are the sum of fire suppression costs and revegetation costs.

From figure 2 it can be seen that the present value of management costs during the 50-year time period declines as the percent area of post-fire revegetation increases. Additionally, predicted costs decline as higher proportions of the area are seeded with crested wheatgrass rather than a native seed mix. In addition, revegetation with crested wheatgrass and revegetation with native seed were equally effective at reducing the predicted area of cheatgrass monoculture on the landscape and were increasingly effective as greater proportions of burned areas were revegetated. This shows that the least cost strategy for reducing the amount of cheatgrass on the landscape is 100% post-fire revegetation with crested wheatgrass.

In contrast, the cost curves for maintaining native vegetation show an alternative least cost strategy (Figure 3). As expected, reseeded with a native seed mix was the most effective strategy for increasing predicted native vegetation on the landscape, while seeding with crested wheatgrass was as ineffective as no revegetation. Thus, the least cost strategy for increasing the area of native vegetation on the landscape is to increase the proportion of

revegetation done with a native seed mix while implementing 100% post-fire revegetation.

These results are important because they show that even though revegetation costs are high, doing revegetation not only has ecological benefits, but also costs less than doing no revegetation at all. This information should help land managers to effectively prioritize their budgets and to justify costs associated with revegetation.

The minimum total cost curve for maintaining native vegetation was estimated as linear and quadratic functions of the percent area of native vegetation on the landscape. Both functions explain 99.9% of the variance in the total minimum cost, and all parameters in each model are significant ($p < 0.001$). Although the quadratic term is significant, it contributes negligibly to cost estimates. Therefore, the net present value of expected costs (\$ per 100 acres across 50 years) for achieving different percentages of native vegetation on the landscape is well estimated by the linear function:

$$E(\text{cost}) = 15926.39 + 65.96 * \text{native} \quad \text{Eq. 3}$$

$$10\% \leq \text{native} \leq 42\%$$

Although maintaining higher levels of native vegetation on the landscape has higher costs, the cost per unit of native vegetation (average costs) decreased as increasing amounts of native vegetation were maintained (Figure 4). In other words, there are increasing returns to scale for increasing the area of native vegetation on the landscape using post-fire revegetation. The marginal cost of a percent increase in the area of native vegetation on the landscape was found by taking the derivative of total cost curve with respect to the percent area of native vegetation. This shows that the cost for each additional percent area of native vegetation is well approximated by a constant marginal cost (summed across 50 years) of

\$65.96 per 100 acres, based on the average of 500 simulations. To achieve greater than 42% native vegetation on the landscape, however, additional or alternative strategies to post-fire revegetation must be implemented, and the cost curves cannot be extrapolated beyond the predicted 42% native vegetation.

The cost predictions discussed thus far are dependent on the average per acre revegetation and fire suppression cost estimates provided by experts. However, the actual per acre cost of revegetation varies dramatically across years and regions and depends on a variety of factors including the specific seed mix and seeding technologies used. Likewise, fire suppression costs per acre are highly variable and are affected by factors such as fire size, location, and proximity to urban areas. Break-even cost ratios, however, enable us to compare the costs of revegetation to no revegetation, independent of actual per acre costs, by comparing the relative costs of revegetation and fire suppression. Figure 5 is a histogram of 500 predicted 50-year break-even cost ratios for revegetation costs and fire suppression costs for post-fire revegetation with native seed. The break-even ratios across 500 runs ranged from 0.66 to 6.5. The average of all 500 runs predicted that revegetation costs per acre can equal up to 1.9 times the per acre fire suppression costs and still break-even over 50 years.

This break-even cost ratio provides some important perspectives for viewing the costs of revegetation. First, post-fire revegetation becomes increasingly cost-effective as per acre fire suppression costs increase. For example, revegetation near urban areas is likely to be cost effective even at high per acre revegetation costs because per acre fire suppression costs are very high in those areas. Also, the costs of native seeds are expected to decline as demand for native seed increases. Thus, as the amount of revegetation done with native seed increases, it

is likely to become increasingly cost effective.

Cumulative probability distributions are presented for each management strategy and objective to compare the full range of predicted values from the 500 50-year simulations (Figures 6-8). These graphs show the same results as for the cost curves but also illustrate the variability in predicted values that results from uncertainty in the model parameters. Figure 6 shows the range of predicted values for native vegetation across all runs and shows that revegetation with native seed is the dominant strategy for maintaining native vegetation. However, for reducing the area of cheatgrass monoculture, differences between post-fire revegetation with crested wheatgrass and with native seed were small, and both revegetation strategies were dominant to no revegetation (Figure 7). For cost minimization, post-fire revegetation with crested wheatgrass was dominant to both no revegetation and revegetation with native seed across the full range of management costs (Figure 8). At low management costs, no revegetation cost less than revegetation with native seed, but in the 66% of runs that had the highest costs, native revegetation cost less than no revegetation. In fact, cost savings from revegetation with either seed mix (as compared to no revegetation) increased with increasing predicted management costs. Thus, revegetation may reduce both expected costs and risk. Displaying sensitivity analysis results as cumulative probability distributions allows managers to visually examine the uncertainty surrounding predicted costs and ecological outcomes and to make better informed decisions in the context of risk and uncertainty.

Past trends, current expectations, and model predictions all support that, without immediate intervention, sagebrush steppe vegetation will undergo continued rapid degradation, bringing about numerous ecological and economic ramifications. In the absence

of revegetation, cheatgrass monoculture and associated fire frequency are predicted to increase further, and the area of native vegetation will continue to decline.

Post-fire revegetation is predicted to curb some of the increase in cheatgrass monoculture and fire frequency, and post-fire revegetation with native seed is predicted to increase the amount of native vegetation on the landscape. In addition, despite the high costs of revegetation management, post-fire revegetation is predicted to decrease overall management costs by reducing the need for costly fire suppression activities. Therefore, post-fire revegetation should contribute significantly to achieving ecological and economic goals for management of the sagebrush steppe, and a strategy of 100% post-fire revegetation with native seed or crested wheatgrass is the optimal management strategy considered.

The appropriate choice between native seed and crested wheatgrass for revegetation depends on the prioritization of management objectives. Revegetation with native seed is critical for the maintenance of native vegetation on the landscape, but currently costs more than reseeding with crested wheatgrass, which is equally effective as native seed for achieving some other ecosystem objectives. If maintenance of native vegetation were the highest priority management objective, then 100% post-fire revegetation with a native seed mix would be the best strategy for achieving the management goals. However, if cost minimization were the highest priority objective, the optimal strategy would be 100% revegetation with crested wheatgrass. Prioritization of these objectives depends on how society values different goods, including livestock forage, biodiversity, and ecosystem function, and the “best” management strategy depends on the decided prioritization. The model used in this paper identifies the on-the-ground trade-offs of different strategies so that managers can make better informed

decisions in the context of the decided priorities and society values.

Despite the variety of predicted benefits gained relative to no revegetation, post-fire revegetation is probably insufficient for maintaining long-term sustainability of the sagebrush steppe. Even with post-fire revegetation, the area of cheatgrass monoculture on the landscape is predicted to increase well beyond current estimated levels of 25% (Figures 2 and 7). If society desires to maintain or improve the current state of the sagebrush steppe, other management strategies will need to be explored.

Conclusions

In this study a predictive model that links management, vegetation dynamics, and economics was developed and used to predict potential future states of the sagebrush steppe and to evaluate the ecological and economic trade-offs of different management strategies. This study has shown that formulating a state-and-transition model as a Markov chain process is one means for moving between a conceptual description of vegetation dynamics and a predictive model that can be used to test hypotheses and inform management.

Land planning and management requires consideration of both economic efficiency and other planning criteria. Determining an optimal or preferred management strategy requires knowledge of both the ecosystem's response to management and clear linkages between management and costs. In this paper the Markov model is applied to the sagebrush steppe ecosystem whose dynamics are nonlinear, thus making costs and vegetation changes difficult to predict. Despite the complexity of the system and limited knowledge of the parameters guiding it, model results justify the costs associated with post-fire revegetation while

providing warning that other management strategies should also be explored to meet ecological objectives long-term.

In this paper, repeated model simulations based on random variables was used to look at the effects of parameter uncertainty on model predictions. However, parameter-specific sensitivity analysis could be used to prioritize areas of research based on factors that will most improve management effectiveness, such as revegetation success or native seed availability.

A variety of management strategies can be examined using the methods presented in this paper. For example, strategies such as pre-fire revegetation of degraded sagebrush sites, revegetation of cheatgrass monoculture sites, and revegetation of failed revegetation efforts could be included in this model by adding the appropriate transition pathways. Other forms of management, such as grazing, increased fire suppression efforts, and controlled burns, could be modeled by adjusting transition parameters.

The same model also could be applied to examine management in other sagebrush steppe communities by using a set of model parameters appropriate to the system. Furthermore, the Markov model can be formulated for any system for which discrete vegetation states, transition pathways, and transition times or probabilities can be estimated.

The application of Markov models to examining vegetation dynamics and management costs and outcomes, makes the possibility of achieving ecological goals much more realistic, because it provides a means for exploring potential management options and directly ties economics to the future states and health of ecosystems.

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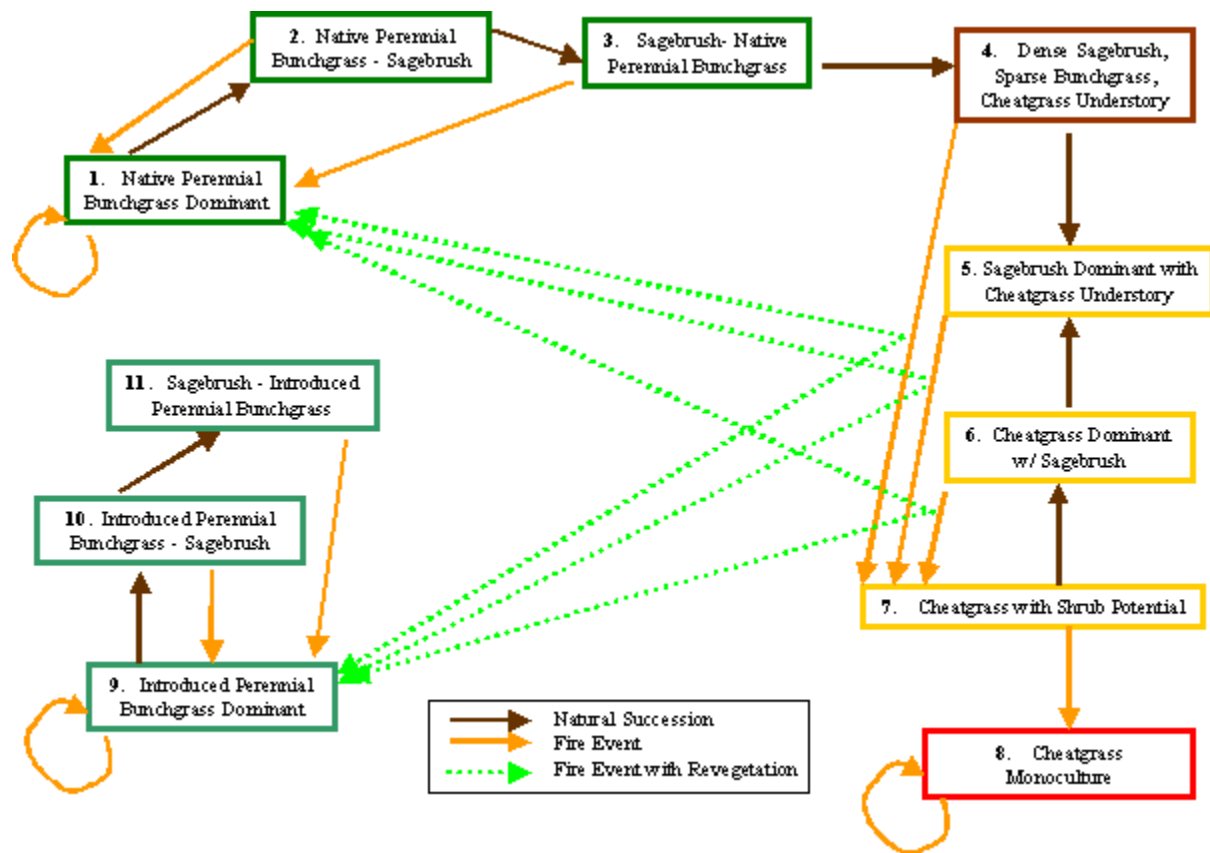


Figure 1. State-and-transition model of current Great Basin sagebrush steppe vegetation dynamics.

Table 1. Generalized Transition Probability Matrix, P , for the Great Basin Sagebrush Steppe.

Vegetation Type	1	2	3	4	5	6	7	8	9	10	11
1	$1-p_{1,2}^*$ ($1-p_{\text{fire}_1}$)	$p_{1,2}^*(1-p_{\text{fire}_1})$	0	0	0	0	0	0	0	0	0
2	p_{fire_2}	$(1-p_{2,3})^*$ ($1-p_{\text{fire}_2}$)	$p_{2,3}^*(1-p_{\text{fire}_2})$	0	0	0	0	0	0	0	0
3	p_{fire_3}	0	$(1-p_{3,4})^*$ ($1-p_{\text{fire}_3}$)	$p_{3,4}^*(1-p_{\text{fire}_3})$	0	0	0	0	0	0	0
4	$p_{\text{fire}_4}^*$ V_{nat}	0	0	$(1-p_{4,5})^*$ ($1-p_{\text{fire}_4}$)	$(p_{4,5})^*(1-p_{\text{fire}_4})$	0	$p_{\text{fire}_4}^*(1-V_{\text{cwg}}-V_{\text{nat}})$	0	$p_{\text{fire}_4}^*$ V_{cwg}	0	0
5	$p_{\text{fire}_5}^*$ V_{nat}	0	0	0	$1-p_{\text{fire}_5}$	0	$p_{\text{fire}_5}^*(1-V_{\text{cwg}}-V_{\text{nat}})$	0	$p_{\text{fire}_5}^*$ V_{cwg}	0	0
6	$p_{\text{fire}_6}^*$ V_{nat}	0	0	0	$p_{6,5}^*(1-p_{\text{fire}_6})$	$(1-p_{6,5})^*$ ($1-p_{\text{fire}_6}$)	$p_{\text{fire}_6}^*(1-V_{\text{cwg}}-V_{\text{nat}})$	0	$p_{\text{fire}_6}^*$ V_{cwg}	0	0
7	0	0	0	0	0	$p_{7,6}^*(1-p_{\text{fire}_7})$	$(1-p_{7,6})^*$ ($1-p_{\text{fire}_7}$)	p_{fire_7}	0	0	0
8	0	0	0	0	0	0	0	1	0	0	0
9	0	0	0	0	0	0	0	0	$1-p_{9,10}^*$ ($1-p_{\text{fire}_9}$)	$p_{9,10}^*(1-p_{\text{fire}_9})$	0
10	0	0	0	0	0	0	0	0	$p_{\text{fire}_{10}}$	$(1-p_{10,11})^*$ ($1-p_{\text{fire}_{10}}$)	$(p_{10,11})^*$ ($1-p_{\text{fire}_{10}}$)
11	0	0	0	0	0	0	0	0	$p_{\text{fire}_{11}}$	0	$1-p_{\text{fire}_{11}}$

$p_{i,j}$ is the probability of transitioning to state j from state i if fire were absent from the system.

p_{fire_i} is the probability of fire in vegetation type i .

V_{nat} and V_{cwg} are the proportion of burns that are successfully revegetated with a native seed mix and crested wheatgrass.

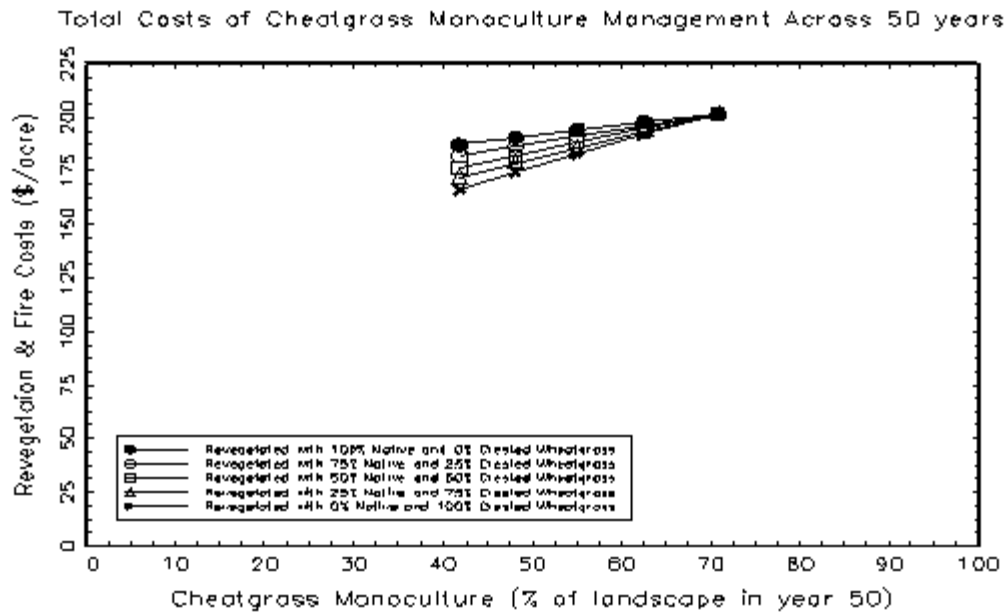


Figure 2. Total cost curves for reducing cheatgrass monoculture.

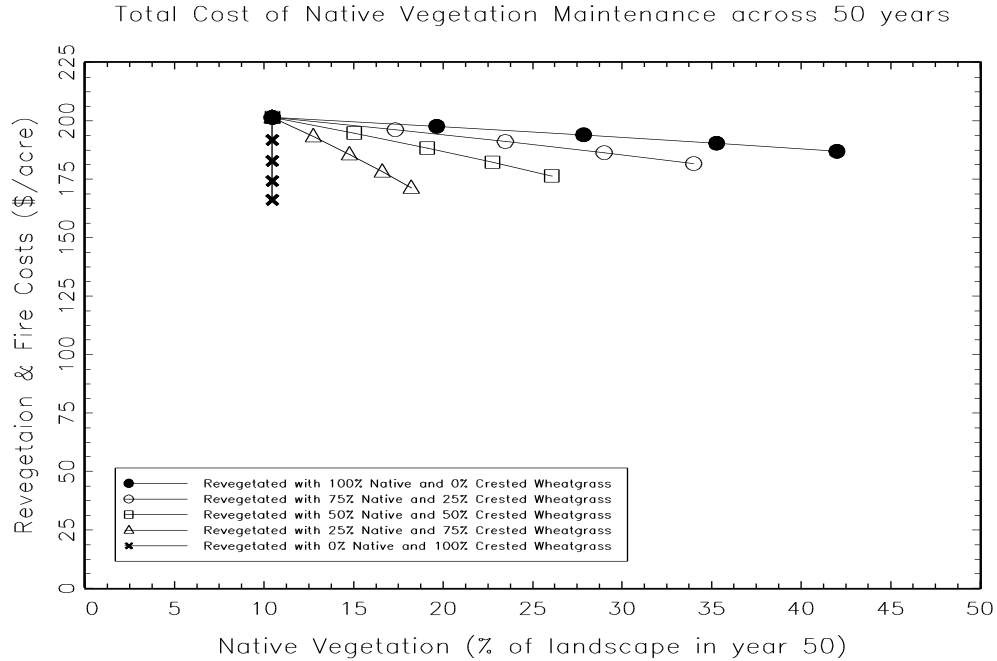


Figure 3. Total cost curves for maintenance of native vegetation.

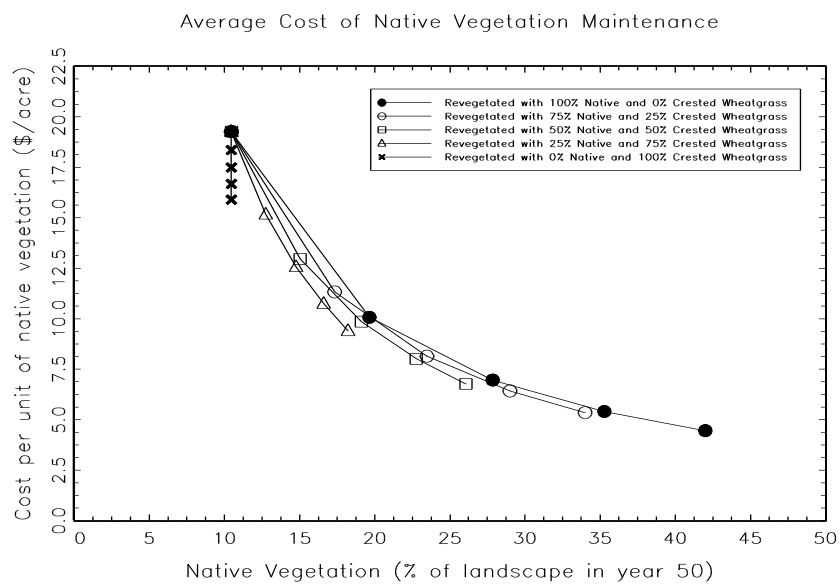


Figure 4. Average cost curve for maintenance of native vegetation.

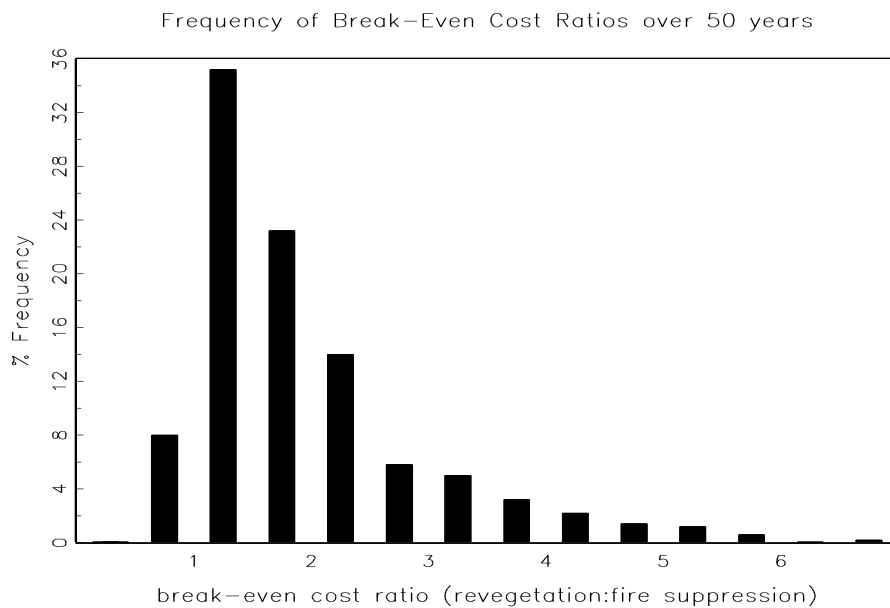


Figure 5. Histogram of 500 50-year break-even cost ratios.

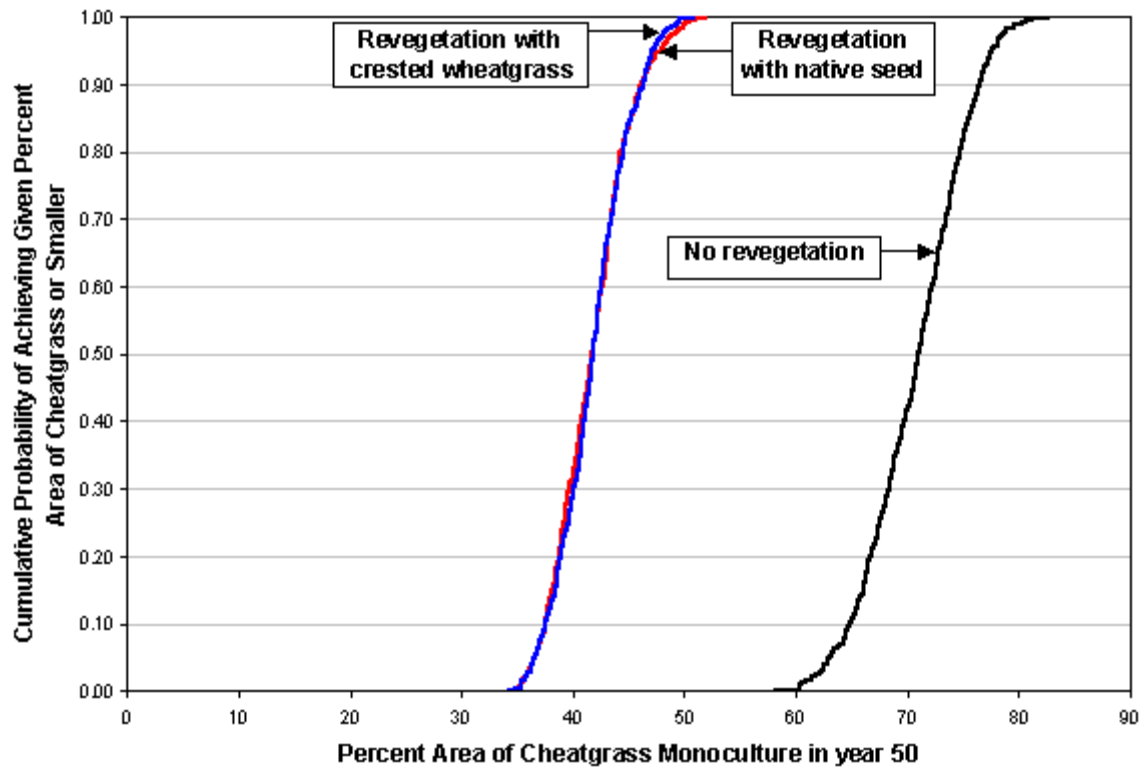


Figure 6. Cumulative distribution of percent area of native vegetation in year 50.

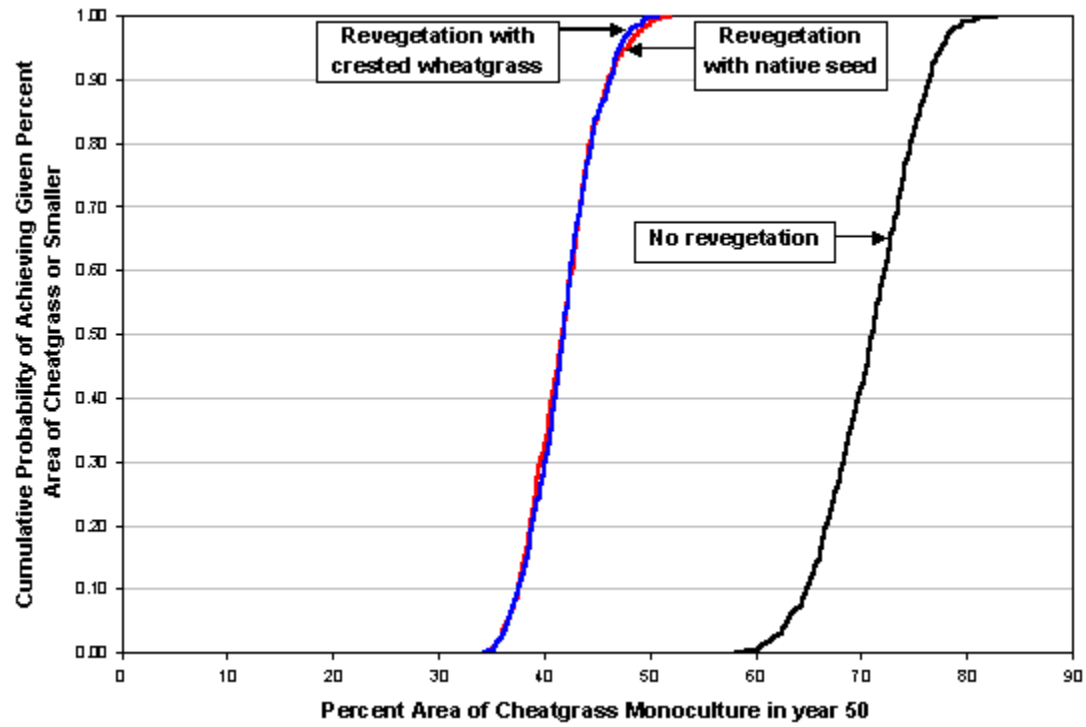


Figure 7. Cumulative distribution of percent area of cheatgrass monoculture in year 50.

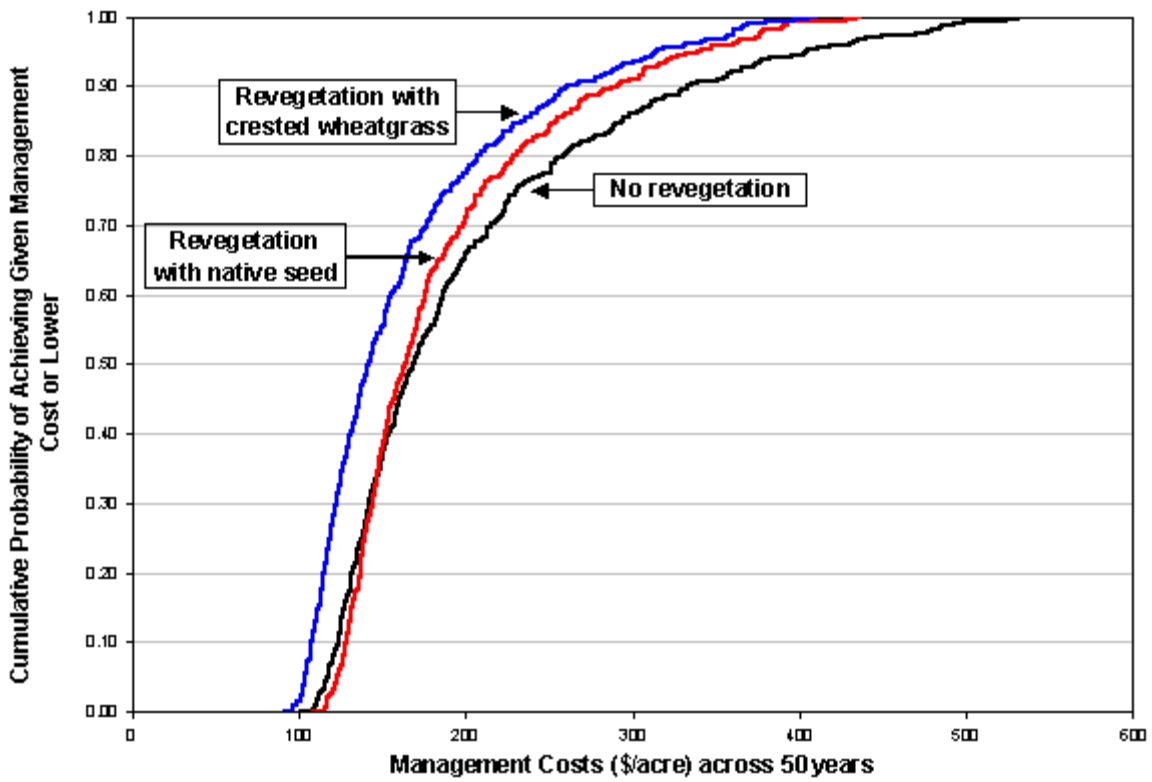


Figure 8. Cumulative distribution of management costs (net present values) across 50 years.