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Great Basin Land Management Planning Using Ecological Modeling

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ABSTRACT / This report describes a land management modeling effort that analyzed potential impacts of proposed actions under an updated Bureau of Land Management Resource Management Plan that will guide management for 20 years on 4.6 million hectares in the Great Basin ecoregion of the United States. State-and-transition models that included vegetation data, fire histories, and many parameters (i.e., rates of succession, fire return intervals, outcomes of management actions, and invasion rates of native and non-native invasive species) were developed through workshops with scientific experts and range management specialists. Alternative restoration scenarios included continuation of *current management*, *full fire suppression*, *wildfire use in designated fire use zones, wildfire use in resilient vegetation types only*, *restoration with a tenfold budget increase*, *no restoration treatments*, and *no livestock grazing*. Under all the scenarios, cover of vegetation states with native perennial understory declined and was replaced by tree-invaded and weed-dominated states. The greatest differences among alternative management scenarios resulted from the use of fire as a tool to maintain native understory. Among restoration scenarios, only the scenario assuming a tenfold budget increase had a more desirable outcome than the current management scenario. Removal of livestock alone had little effect on vegetation resilience. Rather, active restoration was required. The predictive power of the model was limited by current understanding of Great Basin vegetation dynamics and data needs including statistically valid monitoring of restoration treatments, invasiveness and invasibility, and fire histories. The authors suggest that such computer models can be useful tools for systematic analysis of potential impacts in land use planning. However, for a modeling effort to be productive, the management situation must be conducive to open communication among land management agencies and partner entities, including nonprofit organizations.

Since the United States National Environmental Policy Act of 1969, federal land use planners in the United States have been required to formulate alternative management scenarios and conduct analyses of the potential impacts of proposed actions on the landscape under management. The Bureau of Land Management is one of the two major national land management agencies, and its regulations direct planners to "estimate and describe the physical, biological, economic, and social effects of implementing each alternative considered in detail. . .. This analysis should provide adequate information to evaluate the direct, indirect, and cumulative impacts of each alternative in order to determine the best mix of potential planning decisions" (Bureau of Land Management [BLM] Handbook 1601-1). However, evaluating long-term biological impacts of alternative land management scenarios on large landscapes is difficult given the gaps in the scientific understanding of ecological dynamics (Hemstrom and others 2001) and the complexity of interactions between management actions and natural disturbances (Freckleton 2004).

The biological effects of management typically are estimated using professional judgment, which integrates existing data and anecdotal knowledge on landscape condition and the effects of past management.

KEY WORDS: Community dynamics; Federal lands; Grazing management; Great Basin; Prescribed fire; Rangeland; Resilience; Thresholds; Wildfire

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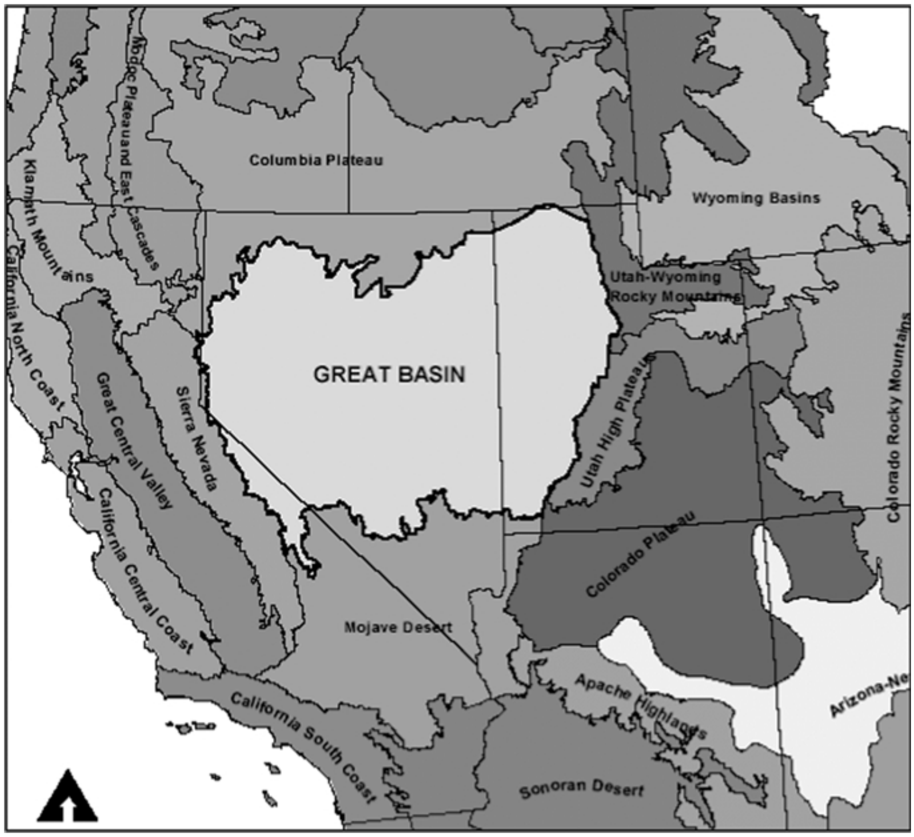

Figure 1. A map of the western United States showing the Great Basin ecoregion in relation to other nearby ecoregions, as defined by Bailey and others (1995) on the basis of vegetation characteristics.

However, this method lacks the repeatability that would allow land managers to evaluate assumptions through the adaptive management process (Walters and Holling 1990; Wilhere 2002). Ecological modeling provides an alternative method for estimating impacts (Hilborn and others 1995) by providing a structured framework for analyzing field data in combination with information derived from panel sessions, expert interviews, or both. The large gaps in existing empirical data make the parameterization of these models difficult, particularly in ecoregions such as the Great Basin that do not have an extensive history of ecological experimentation. Although most assumptions of the resulting models have not been tested empirically, they are documented through the modeling process and can be revised as new data become available.

Loss of Ecological Resilience in the Great Basin

The Great Basin ecoregion, as defined by Bailey (1995), comprises 29.3 million hectares of matrix-forming desert scrub (*Atriplex* spp. and others) and sagebrush (*Artemisia* spp.) communities between the Sierra Nevada Range, the Wasatch Range, the sagebrush steppe of the Columbia Plateau, and the creosote bush (*Larrea tridentata*)-dominated system of the Mojave Desert (Nachlinger and others 2001) in the western United States (Figure 1). The Great Basin is among the top five U.S. ecoregions for number of endemic species, and second for number of imperiled species (Nachlinger and others 2001; Stein and others 2000). Because of a relatively small and mostly urbanized human population, much of the region still is relatively unfragmented. However, over the past 120 years, the Great Basin has been experiencing rapid ecologic change (Ricketts and others 1999) resulting in decreased ecological resilience, or the ability of an ecosystem to recover from disturbance.

Three related factors have lead to decreased resilience in the Great Basin: loss of herbaceous species from shrub understories; invasion of nonnative species, especially cheatgrass (*Bromus tectorum*); and encroachment of native pinyon (primarily *Pinus monophylla*) and juniper (primarily *Juniperus osteosperma*) trees. Historic overgrazing and fire suppression in sagebrush ecosystems have resulted in the loss of native perennial grasses and forb understories (Blackburn and Tueller 1970; Miller and Tausch 2001), leaving an ecological niche open to these native and nonnative invaders (Chambers and others 1999; Tausch 1999; Tausch and Nowak 1999; Young and Sparks 2002; Young and others 1987).

Over the next 30 years, approximately 19% of Great Basin sagebrush systems and 51% of Great Basin salt desert scrub systems are thought to be at high risk for displacement by cheatgrass (*Bromus tectorum*; Wisdom and others 2003), a nonnative annual grass with a life history that emphasizes reproduction and population maintenance through a short-interval fire cycle (Young and others 1987; Whisenant 1990). Perennial species growing with cheatgrass experience decreased soil water and plant water (Melgoza and others 1990), resulting in an inverse relationship between cheatgrass cover and perennial grass cover (West and Yorks 2002).

During the same period, approximately 35% of the Great Basin's sagebrush ecosystem is thought to be at high risk for encroachment by trees (Wisdom and others 2003). During presettlement times in the Great Basin, pinyon (primarily *Pinus monophylla*)–juniper (primarily *Juniperus utahensis*) (PJ) woodlands were located on shallow, rocky soils. However, because of anthropogenic processes that likely include incompatible historic grazing, fire suppression, and increases in temperature and atmospheric carbon dioxide, the two species have moved onto deeper soils previously dominated by sagebrush and other shrub species (Blackburn and Tueller 1970; Miller and Rose 1999; Tausch and Nowak 1999). The encroachment of PJ into shrublands decreases shrub and herbaceous species cover (Barney and Frischknecht 1974; Tausch and Nowak 1999), species diversity, and seed bank density (Koniak and Everett 1982). Additionally, PJ expansion can cause stand-replacing fire to move into old-growth woodlands (Taush 1999), some of which have trees older than 1,000 years (Miller and Rose 1999). These hot fires can lead to erosion, decreased watershed function, and increased management costs.

Restoration Framework

Most of the Great Basin (78%) is under federal management, with the Bureau of Land Management (BLM) responsible for the largest land area (Nachlinger and others 2001). In response to the loss of ecological resilience and productivity of Great Basin ecosystems, the BLM created the Great Basin Restoration Initiative, with a mission to maintain resilient plant communities, restore degraded plant communities, reduce invasive species cover, and sustain long-term multiple use. The Ely, Nevada BLM district (the largest district in the lower 48 states, with 4.6 million hectares under management) was the first district to begin implementation of the initiative, in partnership with the local nonprofit Eastern Nevada Landscape Coalition (ENLC) and The Nature Conservancy of Nevada. The preparation of a Resource Management Plan/ Environmental Impact Statement with a 20-year scope for the Ely District created a unique opportunity to bring together managers and scientists to collect and structure expert knowledge on ecosystem dynamics and anticipated effects of management.

Resource Management Planning Using Computer Modeling

The state-and-transition model concept is increasingly used by land managers to prioritize and understand rangeland ecosystems (Bestelmeyer and others 2004; Westoby and others 1989; U.S. Department of Agriculture (USDA) Natural Resources Conservation Service 1997; see Figure 2 for an example). The accompanying concepts of nonlinear responses, irreversibility, and stable states are particularly applicable to the arid Great Basin (Tausch 1999). Turning conceptual state-and-transition models into computer models with quantifiable parameters can allow for the projection of management outcomes over large scales and long periods (Hann and Bunnell 2001). Such modeling also allows for the inclusion of stochastic drivers such as climate and its effects on fire return intervals. In this study, we used a nonspatial, user-driven computer program to build state-and-transition models based on an annual time step. These models incorporate hypotheses about state stability and transition probabilities, which can be tested as data become available. This process should lead to improved predications about vegetation's response to management actions. In this case, we used these models to compare alternative management scenarios for Great Basin vegetation relevant to Ely BLM's Resource Management Plan/Environmental Impact Statement.

Methods

Study Area Description

The Ely BLM District covers 4.6 million hectares in eastern Nevada. The topography is basin and range, with valley floors ranging in elevation from 2,500 to 3,100 m, and the highest mountains reaching almost 4,000 m. Vegetation is described in detail later (Table 1), but consists of primarily shrublands dominated by

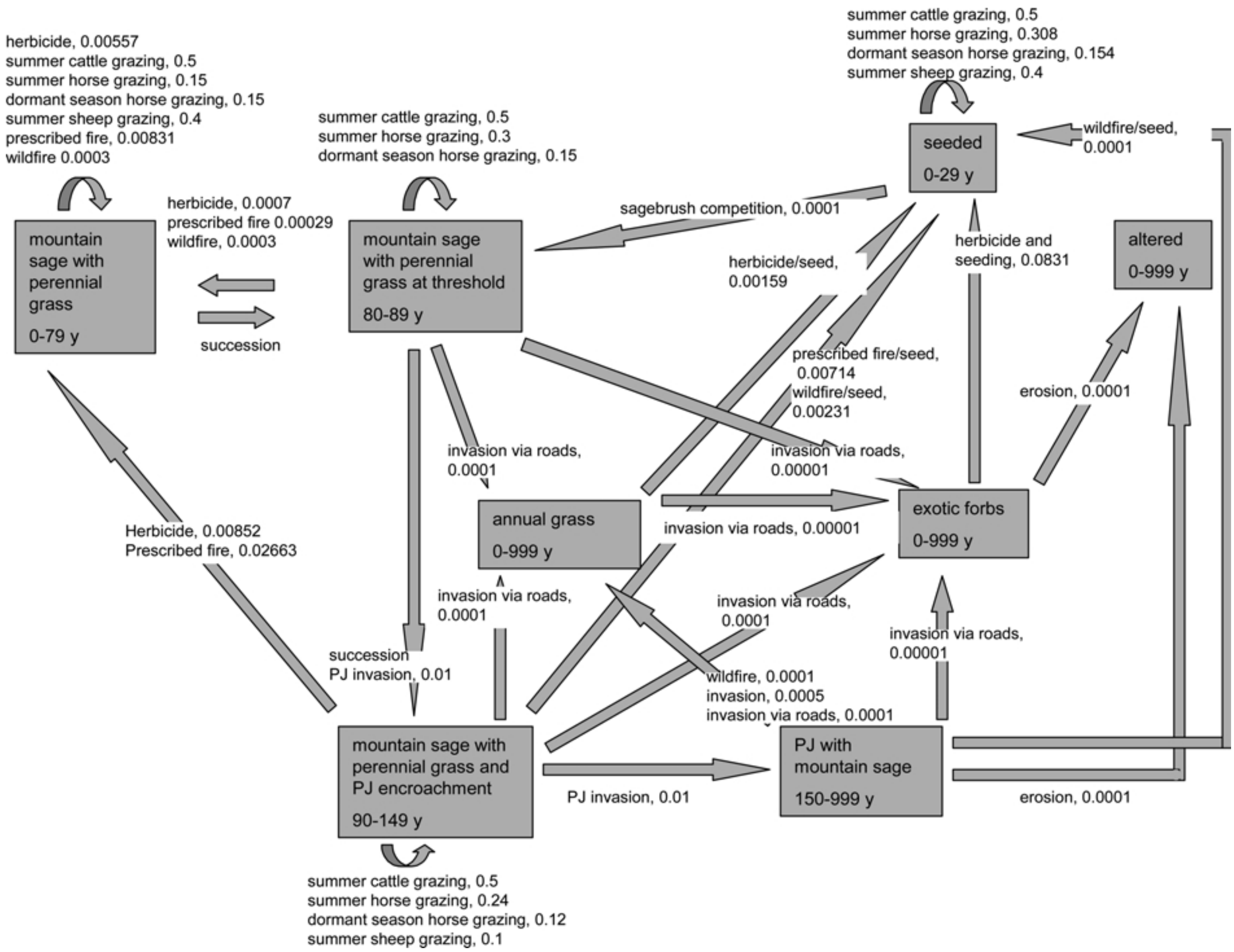

Figure 2. Mountain sagebrush state-and-transition model showing all transitions and probabilities. Each box represents a state of the vegetation, and the arrows represent transitions, including those attributable to natural disturbances, management, and succession. The numbers in each box represent the years that a pixel can remain in that state before undergoing succession to the indicated state.

sagebrush (*Artemisia* spp.) and salt desert species with woodlands of PJ and white fir (*Abies concolor*) as well as limber pine (*Pinus flexilis*) at higher elevations. This study focused on lower- and middle-elevation vegetation types, because the sites at these elevations are managed by BLM, whereas the upper elevations are managed primarily by the U.S. Forest Service.

Computer Model

The Vegetation Dynamics Development Tool (VDDT; http://www.essa.com/downloads/vddt/index.htm) is a user-driven computer program that builds state-and-transition models to examine the role of various disturbance agents and management actions in vegetation change based on discrete time steps. The VDDT is appropriate for modeling stochastic processes, but it also has some deterministic components that drive the primary successional pathways (Barrett 2001). Used previously in large-scale management planning for the Interior Columbia Basin Ecosystem Management Project (Hemstrom and others 2001), the VDDT is a nonspatial model. It therefore is most appropriate for large-scale planning processes rather than project-scale planning in which detailed spatial information might be available (Merzenich and others 2003).

It also is possible to include economic and administrative variables as activity limits in the VDDT. For example, in this analysis, we used available restoration budgets to limit the level of treatment under each scenario.

Comparison of Alternative Management Scenarios

Restoration scenarios were designed to analyze the major options for landscape scale restoration. The

Table 1. Potential vegetation type, successional time (years) in the earliest vegetation state (prethreshold for most vegetation types), percentage of fire rehabilitation success (which determined whether a burned state transitioned to an early successional state or to a cheatgrass-dominated state), percentage of the restoration budget spent on each vegetation type, and dominant fire return interval (FRI)

Vegetation type	Succession	Fire rehab success (%)	% Budget	FRI (years)
Black sage	100	50	50	80–380
Mountain mahogany	10	70	0	200
Mountain sage	80	70	4	20–60
PJ woodland	333	70	10	300
Shadscale	50	30	3	100
Winterfat	150	30	4	No fire
Wyoming sage	70	50	29	30–250

primary differences between restoration scenarios are the available restoration budget ($0, $500,000, or $50,000,000 per year), the level of livestock grazing (*current* or *none*), and the level of fire suppression/fire use (*current suppression, full suppression, no suppression in designated fire use zones, and no suppression of resilient vegetation types*). Although the available restoration budget is not expected to be so great or so variable, it was varied by multiple orders of magnitude because pilot model runs showed small or no differences between more moderate restoration scenarios. The scenarios modeled in this study do not exactly mirror the alternatives specified in Ely BLM's Resource Management Plan (RMP). This results from the dynamic nature of the RMP alternatives, which have changed several times over the past two years in response to comments from BLM's cooperators and the public.

The primary restoration scenarios modeled are shown in Table 2. Restoration treatments include mechanical treatments, herbicide, and prescribed fire, which are combined with seeding of native or native–nonnative seed mixes in vegetation states with no seed source for herbaceous species. Restoration activities cause the vegetation to transition to a restored state, with at least some native herbaceous understory for most potential vegetation types (PVTs; PJ woodland is an exception).

Restoration activities were planned using the budget available under each scenario. The percentage of the budget allocated to each vegetation type was constant across all scenarios (Table 1), and the appropriate treatments for each state of each vegetation type gave highest priority to the treatment of prethreshold states (i.e., mechanical treatments were focused on states with perennial grass). The budget was allocated among vegetation types by giving high priority to types in need of treatment (black and Wyoming big sagebrush: *Artemisia nova* and *Artemisia tridentata* ssp. *wyomingensis*) and deemphasizing types for which treatment methods are poorly understood (winterfat and shadscale: *Krashennenikovia lanata* and *Atriplex canescens*), types that are hard to reach or small and patchy (mountain mahogany: *Cercocarpus ledifolius*), and types thought to respond well to wildfires (mountain big sagebrush: *Artemisia tridentata* ssp. *vaseyana*).

In all cases, the goal of restoration programs was to maintain or increase the area of resilient vegetation, that is, vegetation able to recover from disturbance by recruiting native species and maintaining some level of ecosystem function. The goal was also to reduce the area of vegetation that had crossed thresholds from resilient states to tree-dominated shrubland states, weed-dominated states, or depleted states, which are not weed-dominated but which lack native understory. We refer to a threshold as a state beyond which there is an abrupt change in the properties of the vegetation.

Inputs to the Model

Current condition of the vegetation and succession. The vegetation of the Ely District was stratified into seven PVTs (Figure 3A). Through a collaborative process involving agency resource specialists and other scientific experts, states of each PVT and pathways between all states were defined. The PVTs were based on soil survey data, in combination with precipitation, elevation, and aspect. States within each PVT reflect the potential condition of the vegetation as influenced by site characteristics and natural and anthropogenic disturbances. Therefore, some of the states currently account for no or little land area.

For each PVT, the proportion of land in each state was calculated on the basis of all available data from the seven vegetation types, including BLM data collected in 1993 and 1994 using the Ecological Site Inventory method (BLM 1980) at 1,953 sampling stations in three watersheds (North Spring and Antelope Valleys and Clover Creek South) and more recent (2003)

Table 2. Restoration scenarios modeled as they varied by the available restoration budget, livestock grazing scenario, and wildfire scenario, and the acres treated with each restoration method under each alternative

Scenario	Acres treated						
	Herb	Herb + seed	Mech	Mech + seed	Pf	Pf + seed	Wildfire
Current management	47.2 ± 6.3	31.2 ± 4.7	12.7 ± 3.1	58.5 ± 7.4	68.0 ± 8.1	17.0 ± 3.9	1809.1 ± 324.5
Full fire suppression	48.0 ± 6.4	32.6 ± 5.2	12.0 ± 3.4	58.7 ± 6.6	69.9 ± 6.8	17.3 ± 3.3	3.6 ± 0.9
Wildfire use in designated zones	46.0 ± 5.5	32.5 ± 5.0	13.3 ± 3.4	59.1 ± 7.1	65.3 ± 7.7	16.3 ± 3.7	6915.1 ± 866.1
Wildfire use in resilient types only	46.0 ± 6.8	32.6 ± 5.2	12.3 ± 3.2	58.6 ± 7.8	66.7 ± 7.3	15.6 ± 3.1	1657.2 ± 504.1
Restoration with a 10-fold budget increase	463.1 ± 24.6	323.6 ± 16.4	128.2 ± 13.1	572.6 ± 25.0	675.8 ± 34.0	164.0 ± 11.4	1830.8 ± 421.7
No restoration treatments	0 ± 0	0 ± 0	0 ± 0	0 ± 0	0 ± 0	0 ± 0	1815.6 ± 324.9
No livestock grazing	64.1 ± 7.3		17.1 ± 4.0	57.4 ± 7.6	91.6 ± 8.7	18.2 ± 4.1	1842.3 ± 339.4

[a] The restoration budget was $500,000 under all scenarios except *restoration with a tenfold budget increase*, under which it was $50,000,000, and *no restoration treatments*, under which it was $0.
Livestock grazing was set at current rates under all scenarios except *no livestock grazing*, in which grazing by livestock did not occur. Wildfire was managed at current suppression levels under all scenarios except *full suppression*, under which all fires were suppressed, *fire use*, in which wildfires were unsuppressed when they occurred in designated fire use zones, and *wildfire use in resilient types only*, in which fires were assumed to be successfully suppressed in nonresilient vegetation types.
Herb, herbicide; Herb+seed, herbicide followed by seeding; Mech, mechanical treatment; Mech+seed, mechanical treatment followed by seeding; Pf, prescribed fire; Pf+seed, prescribed fire + seeding. Numbers represent the average number of acres treated per year over 100 years ± the standard deviation over ten Monte Carlo simulations.

vegetation percentage cover data collected using the point intercept method (Elzinga and others 1998) at 152 sampling stations by ENLC in three other watersheds (South Steptoe Valley, and Gleason and Smith Creeks). These watersheds comprise approximately 10% of the area of the district. The proportions of vegetation states in the samples were extrapolated to the 4.27 million hectares of the Ely District that lie within the Great Basin ecoregion. This extrapolation assumes that the sampled sites are representative of the entire area, but this is unlikely to be the case because the watersheds sampled are concentrated in the northern part of the District. However, initial conditions had little effect on ending conditions in the model runs, so these best estimates were assumed to be sufficient. The assumed current condition of the vegetation over all PVTs is shown in Figure 3B.

Estimated successional durations (the amount of time required for a simulation unit to pass from one vegetation state to the next through succession) were assigned to all states of all vegetation types. The time required for each vegetation type to progress through its earliest state to a threshold state is shown in Table 1. Successional durations were assigned primarily by members of the Eastern Nevada Landscape Coalition science committee, facilitated by The Nature Conservancy, and represent expert opinion.

Natural disturbances.

Wildfire and interannual variation. Parameters for modeling wildfire were taken from a variety of sources. First, historic natural fire return intervals (FRIs) were estimated on the basis of information from the Fire Effects Information System (http://www.fs.fed.us/database/feis/) and VDDT models developed for southern Utah (personal communication, J. Merzenich; USDA Forest Service 1997). These natural FRIs then were reviewed by an expert (personal communication, R. Tausch, USDA Forest Service 1997) and changed where appropriate.

To account for current levels of fire suppression in the Ely District, the fire history from 1986 to 2000 for the portion of the District that has a completed soil survey (2,198,125 ha) was used to calculate wildfire probabilities that take current suppression levels into account for each vegetation type. Because of the short available fire history, we used a conservative combination of natural FRIs and fire history. Where these current probabilities were lower than but within one order of magnitude of the estimated natural FRIs, it was assumed that they represented the effects of fire suppression, and the historic probabilities were used.

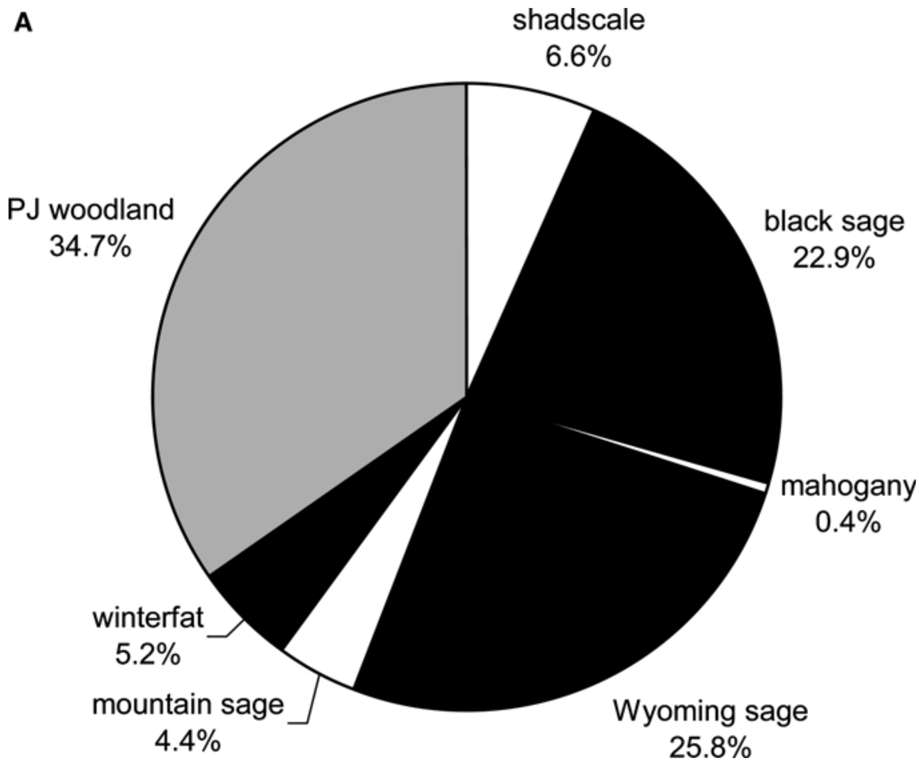

Figure 3. (**A**) The percentage area of each vegetation type in the modeled area. (**B**) The current condition of the vegetation across all potential vegetation types in the modeled area.

Where the current probabilities were orders of magnitude higher, they were assumed to be an artifact of the relatively short fire history available, and the natural FRIs were used. It is unlikely that these higher probabilities are the results of a cheatgrass-caused increase in FRI. Less than 5% of the modeled area is currently dominated by cheatgrass.

The model assumed no fire suppression activities under the fire use scenario and for resilient states (the earliest and threshold states of all vegetation types) per the two entitled resilient types only and times 10. In the current no livestock and no treatment scenarios, fire suppression continued at current rates, as described earlier. Dominant FRIs are shown in Table 1.

When wildfire occurs in vegetation states that have lost native seed banks, the model assumes that fire rehabilitation activities will be performed. The assumed success of these treatments depends on vegetation type. When fire rehabilitation is successful, the vegetation transitions to a restored state. When rehabilitation is unsuccessful, the vegetation transitions to a cheatgrass-dominated state. The assumed success of fire rehabilitation treatments shown in Table 1 is based on expert opinion (personal communication, M. Pellant, BLM).

The VDDT is a probabilistic model. At each time step (year), the occurrence of disturbances is based on a random draw for each simulation unit in the model (Merzenich and others 1999). The number of simulation units affected by a disturbance in any one time step follows the Poisson distribution, and the return interval for disturbances follows the Weibull distribution. The mean for the Poisson distribution of disturbance probability by class will be equal to the probability of disturbance defined for each disturbance for each class. The shape parameters of the Weibull will depend on how the probability of disturbance varies as a pixel ages. In the simplest case, when the probability of disturbance is constant with age, the *P* value will be equal to the constant probability of disturbance, and the other two shape parameters both will be equal to one. However, the stochastic nature of selected disturbances also can be influenced by patterns in climate or other factors. The interannual probability distribution for each disturbance type is determined by user-defined temporal multiplier sequences applied against the base probability of the disturbance. Therefore, multipliers were developed to vary interannual wildfire and insect outbreak probabilities systematically to simulate the El Nino/La Nina cycle. Multipliers, differing

Table 3. Year sequence groups used by the VDDT to generate interannual variation in natural disturbance probabilities attributable to climatic variation[a]

Disturbance	Very low	Low	Normal	High	Severe
Wildfire	35	45	10	5	5
Drought/insect	95	0	0	5	0
Wet/insect	95	5	0	0	0

[a]Numbers refer to the percentage of years that fall into each disturbance category.

for each of ten Monte Carlo simulations, were generated in VDDT. The guidelines used to generate the multipliers can be found in Table 3. As implemented in this study, the VDDT model does not account for the spatial distribution of disturbances.

Invasion by nonnative and native species and Other natural disturbances. Little information is available on invasion rates of particular species in particular vegetation types in the Great Basin, so parameters describing this process were based on nonnative invasive species separated into two categories: annual grass (cheatgrass) and exotic forbs (*Halogeton glomerata, Centaurea* spp. and others). The invasion probabilities for exotic weeds differed because of assumptions about species' invasiveness and site invasibility. For example, annual grasses were assumed to be more invasive than exotic forbs, and sagebrush sites with no understory were assumed to be more invasible than sites with perennial grass and forb understories. Invasion probabilities for annual grasses ranged from 0.1% to 1.5% of a state invaded per year, and probabilities for forbs ranged from 0.05% to 1.5% of a state invaded per year. The invasion of PJ into sites that historically would have been shrublands, as identified by soil type, was set at 1% of an invasible vegetation state invaded per year.

Road proliferation and use affects vegetation in the model because of their tendency to increase the spread of nonnative invasive weeds. The model assumes that the presence and use of roads allows cheatgrass invasion above background rates at probabilities of 0.001 to 0.0001 per year and exotic forbs invasion above background rates at probabilities of 0.0001 to 0.00001 per year in invasible states. Higher-elevation vegetation types were assumed to be less invasible than lower-elevation shrub and salt desert scrub communities. The model does not incorporate any measures of road density.

In addition to fire and invasive species, the effects of the following natural disturbances were included in the models: erosion after catastrophic fire, sagebrush competition with herbaceous species, drought, insect outbreak, unusually wet years, and native grazers (such as rabbits). Because the model was relatively insensitive to these disturbances, they are not discussed further.

Cattle, sheep, and wild horse grazing. Parameters that estimate the effects of cattle, sheep, and wild horse grazing were developed with input from the Ely District's range and wild horse specialists. Grazing practices, spatial extent of grazing, and effects of grazing vary widely from site to site. For example, many of the mountain areas are inaccessible to dormant season grazing and are used only for summer grazing, and spring grazing would be expected to affect some vegetation types much more detrimentally than others. Therefore, the parameters used represent averages estimated across the management area. In determining the average area affected by grazing, range specialists took into consideration the extent of a vegetation state with allotments for each class of livestock. The parameters for all classes of grazers are shown in Appendices 3 through 5.

Generally, range specialists and other scientists believe that cattle grazing affects the vegetation by accelerating or sometimes decelerating succession. This is reflected in the model when instead of moving one annual time step per year, a simulation unit moves two or more steps per year, or moves backward (toward herbaceous dominance rather than woody dominance). The successional effect of cattle grazing was predicted to differ with season of use as well as vegetation type and state, and ranged from reversing succession by 1 year per time step to speeding succession up by 3½ years per time step. Range specialists believe that sheep grazing has a smaller successional effect because sheep browse palatable shrubs during the dormant season. Within herd management areas (HMAs), rates for disturbance attributable to wild horse grazing were expected to range from 5% to 77% of a given vegetation state, and successional effects were predicted to range from zero to five steps per year. For all management scenarios, the probabilities for horse grazing in HMAs were multiplied by 0.4 to represent the current situation: 40% of the Ely BLM District lies within HMAs.

Model results were replicated with ten Monte Carlo simulations, and results reflect averages and standard

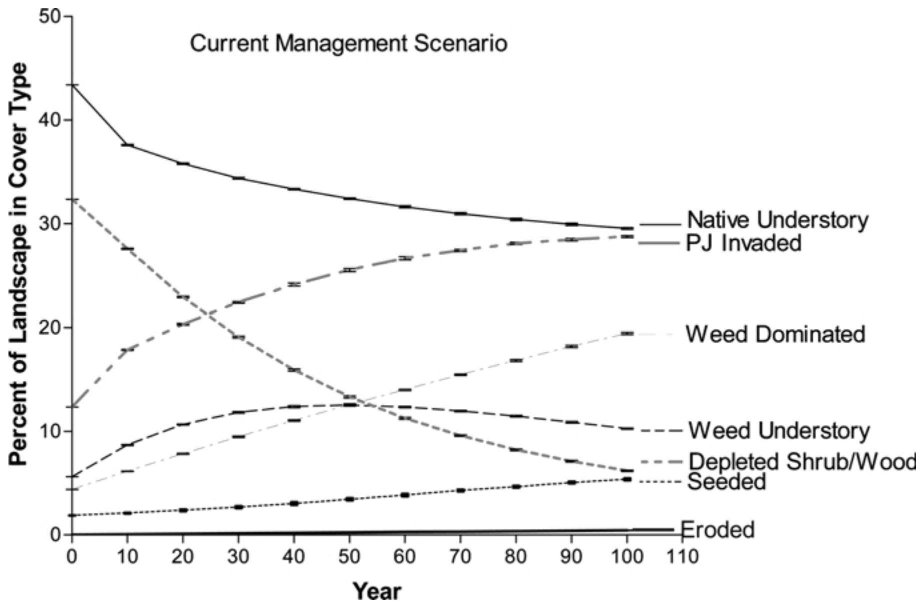

Figure 4. Percentage of the landscape in each cover type over 100 years under the *current management* scenario. Bars represent decadal averages and standard deviations around ten Monte Carlo simulations.

deviations among replicates. Restoration treatments were simulated probabilistically on the basis of the treatment prescriptions developed with managers and limited by budgets per alternative, calculated on a cost per hectare basis. Table 2 shows the acres treated under each scenario.

Results

Overview

Model results track three groups of indicators: (1) resilient vegetation types, which include those with native understory (including threshold states) and seeded states, (2) depleted vegetation types, which include shrubland and woodland sites that have lost their herbaceous understory, and (3) invaded vegetation types, which include sites with a weed understory, sites that are weed dominated, and shrubland sites that have been invaded by PJ.

Under all the scenarios, the percentage of area occupied by vegetation states with native perennial understory declined (Figure 2). The decline was particularly precipitous during the first 10 years under all the scenarios, reflecting dynamics characteristic of Markov and similar models. There were differences between the outcomes of our management scenarios, and although these differences represented small percentages of the landscape, they nevertheless represented substantial land areas due to the enormous size of our modeled landscape.

The results for continuation of current management (Figure 4) indicate that depleted shrub and woodland states will decline the most (by 26%) over the next 100 years, and that states with native understory will decline by about 14% over the next 100 years. These two declining states are respectively replaced with two primary states resulting from invasion of native and nonnative species, with PJ-invaded states projected to increase by 16% and weed-dominated states projected to increase by 14% (Table 4).

Wildfire and Livestock Grazing

The greatest differences among scenarios resulted from alternative fire management scenarios. Among resilient vegetation states, *resilient types only* and *full suppression* scenarios were the most successful at maintaining native understory (−10% and −11%, respectively; Figure 5A, Table 4). The *fire use* scenario resulted in the greatest increase in percentage area of the seeded state (+13%) because emergency fire rehabilitation efforts were highest under this scenario (Figure 5A). The *fire use* scenario also resulted in less increase in PJ-invaded states over time (+7% vs +16% for the *current* scenario). However, much of the depleted and PJ-invaded area transitioned to a weed-dominated landscape under the *fire use* scenario (+22%). Moreover, although the *current* scenario showed an increase in PJ-invaded acreage over time, it showed less increase than the *full-suppression* scenario (+16% vs +22%; Figure 5B, Table 4). There was little

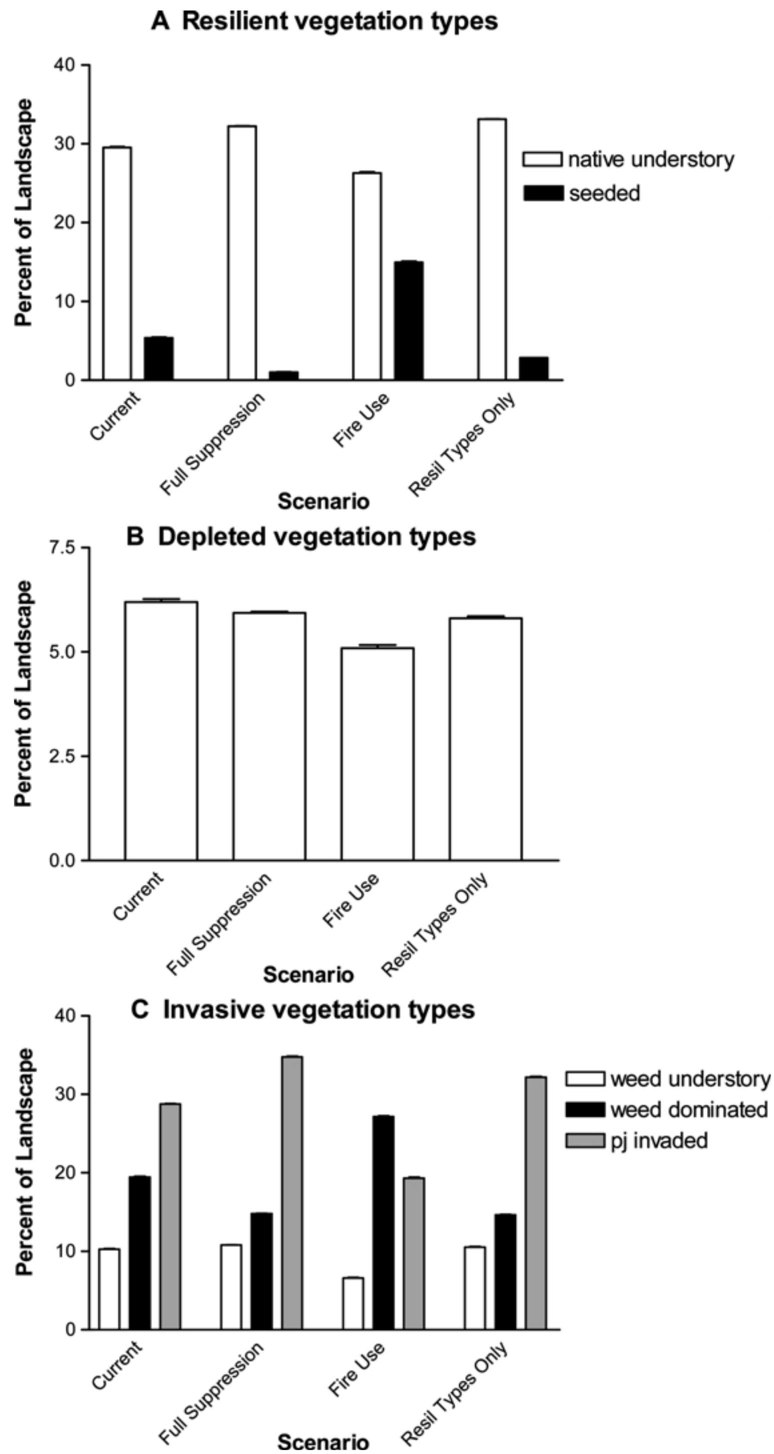

Figure 5. (**A**) Percentage of the landscape in each of two states (native understory and seeded) after 100 years under the alternative fire management scenarios: *current*, *full suppression*, *fire use*, and *resilient types only*. (**B**) Invasive vegetation types (weed understory, weed dominated, and pinyon–juniper invaded) under the same four scenarios. (**C**) Depleted vegetation types (depleted shrubland and woodlands) under the same four management scenarios.

difference among fire alternatives with respect to the amount of depleted vegetation types left after 100 years. The *fire use* scenario left the smallest amount of depleted vegetation (5% remaining; Figure 5C), presumably because this vegetation type burned at a higher rate under this alternative.

Table 4. Percentage of the landscape in each vegetation state grouping after 100 years under each alternative management scenario

Scenario	Native under	Depleted	Weed under	Weed dom	PJ invaded	Eroded	Seeded
Current	−14 ± 0.1	−26 ± 0.1	5 ± 0.1	14 ± 0.10	16 ± 0.1	0 ± 0.0	4 ± 0.1
Full suppression	−11 ± 0.0	−26 ± 0.0	5 ± 0.1	9 ± 0.04	22 ± 0.1	0 ± 0.0	−1 ± 0.0
Fire use	−17 ± 0.1	−27 ± 0.1	1 ± 0.1	22 ± 0.12	7 ± 0.2	1 ± 0.0	13 ± 0.2
Resilient types only	−10 ± 0.0	−27 ± 0.1	5 ± 0.1	9 ± 0.05	20 ± 0.1	0 ± 0.0	1 ± 0.0
No treatment	−14 ± 0.0	−26 ± 0.0	5 ± 0.1	14 ± 0.11	17 ± 0.1	0 ± 0.0	3 ± 0.1
Times 10	−10 ± 0.0	−26 ± 0.0	4 ± 0.1	8 ± 0.05	18 ± 0.1	0 ± 0.0	4 ± 0.1
No livestock	−15 ± 0.1	−26 ± 0.0	5 ± 0.1	14 ± 0.10	18 ± 0.2	1 ± 0.0	3 ± 0.1

Results are combined across separate potential vegetation types and state groups (e.g., native under contains states with native understory and threshold states). Numbers represent the average percentage of the landscape in each cover type ± the standard deviation over ten Monte Carlo simulations. Under = understory; dom = dominant.

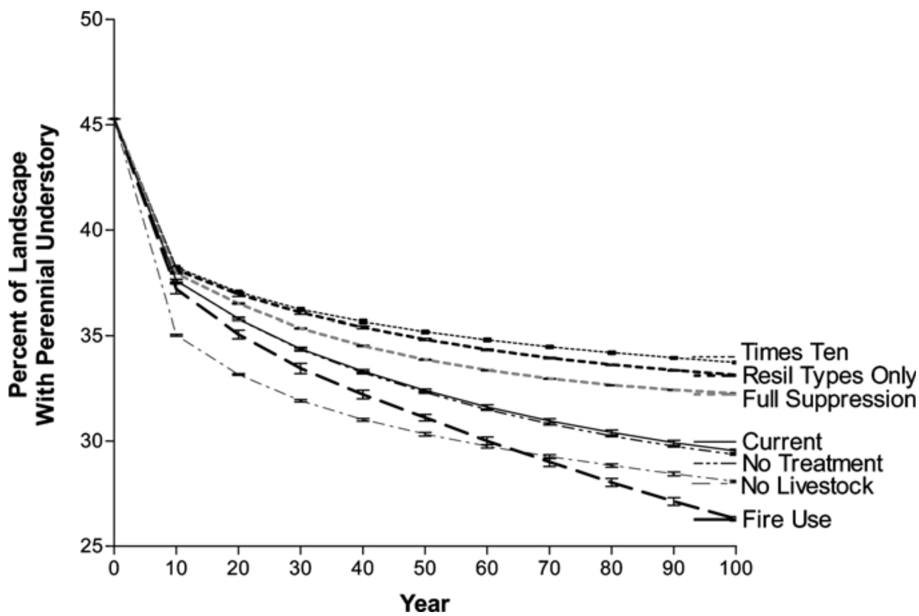

Figure 6. Percentage of the landscape with a perennial understory under each of seven alternative management scenarios over 100 years. Each line represents a different scenario. Bars represent decadal averages and standard deviations around ten Monte Carlo simulations.

The removal of livestock alone had little effect on vegetation resilience. There was less maintenance of native perennial understory (Figure 6) and a 2% increase in the area of PJ-invaded vegetation types, but no appreciable differences in depleted vegetation types (Table 4). More land area was available for grazing under the *fire use* scenario than other scenarios (Figure 7).

Restoration Treatments

The results across all vegetation types showed no substantial differences between the current and no-treatment scenarios with respect to resilient vegetation types, invasive vegetation types, or depleted vegetation types. However, the *times 10* restoration scenario did show progress, at least in terms of maintaining resilient vegetation types. At 100 years, this scenario resulted in 4% less area of perennial understory vegetation types lost as compared with the current scenario. There was little difference in the percentage area of depleted vegetation types between restoration scenarios after 100 years, but there was 6% less increase in weed-dominated areas in the *times 10* restoration as compared with the current scenario.

Some of the more interesting differences among scenarios become apparent when vegetation types are viewed separately. There were a few small differences between the results for the two most successful restoration scenarios (i.e., *wildfire use in resilient types only* and *restoration with a tenfold budget increase*) for the

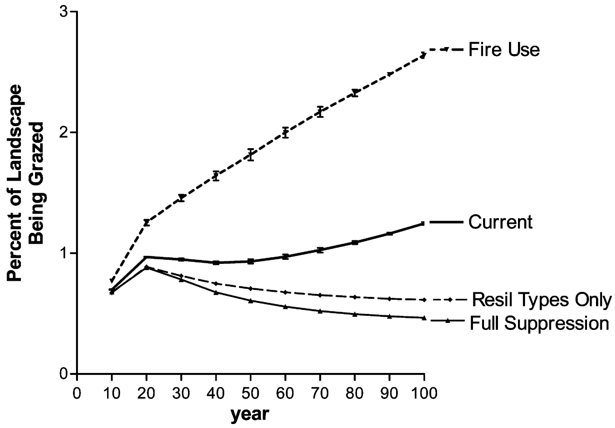

Figure 7. Differences in percentage of the modeled landscape available to be grazed by livestock among alternative fire scenarios: *current, full suppression, fire use, and resilient types only*. Bars represent decadal averages and standard deviations around ten Monte Carlo simulations.

higher-elevation vegetation types (i.e., mountain sage, PJ woodland, and mountain mahogany). However, there were great differences among the lower-elevation vegetation types (i.e., winterfat, shadscale, black sage, and Wyoming sage; Figure 8). Active restoration in the restoration with a tenfold budget increase showed significant progress in the low-elevation vegetation types relative to the *wildfire use in resilient types only*, with beneficial effects of fire seen primarily in the mountain sage community.

Discussion

Management Implications: Passive Versus Active Restoration

This modeling effort resulted in many intuitive outcomes and a few unexpected ones, which might be viewed simply as the logical consequence of interactions among our assumptions about disturbances and successional pathways. Our confidence in the lack of improvement under the livestock removal scenario, the slow spread of weeds through passive means but the quick spread with fire, the precipitous decreases in cover of the depleted shrub state and its replacement by PJ-invaded and weed-dominated states, and the overall lack of large-scale success of restoration scenarios should be limited to our confidence in our initial assumptions. Our confidence in those assumptions is very limited at this time, and parameters that had a large effect on model outcomes (e.g., exotic species' invasion rates) are largely unknown at this time. However, the results do reflect the current understanding of a collaborative group including land managers, academic scientists, users of public lands, and conservation groups and are therefore worthy of consideration in management planning.

Some groups advocate livestock removal as the action crucial to restoration of degraded Western rangeland ecosystems. Although it is likely that past grazing practices have strongly impacted, even devastated, some Great Basin ecosystems (Blackburn and Tueller 1970; Young and Sparks 2002; Young and others 1987), there is little empirical evidence that the removal of livestock grazing alone is an effective restoration technique in most arid ecological systems (Curtain 2002; Jones 2000; Stohlgren and others 1999). The results of this modeling effort support that idea. Across the entire landscape, there was little effect from removal of livestock after 100 years of grazing activity. This result conforms with our assumption that across much of the Great Basin landscape, overgrazing a century or more ago removed both herbaceous perennials and their seed banks, and that seeding would be required to restore much of this area.

Passive restoration via strategic decreases in fire suppression brought about substantial progress in the restoration of high-elevation vegetation types but did little to increase resilience in the lower-elevation sagebrush and salt desert scrub types (Figure 8). Model results suggest that low-elevation sites, along with more highly degraded, higher-elevation systems such as sagebrush lacking understory and PJ-encroached shrublands with high fuel loads, will require active restoration such as the use of mechanical treatments, prescribed fire, or chemical treatments. This result is not unexpected (McIver and Starr 2001), and is a consequence of the higher-precipitation regimes at higher elevations and the greater resilience of these vegetation types in relation to natural fire. Active restoration tools can help to create a mosaic of fuel densities that decreases the probability of catastrophic fire and allows for passive restoration via wildfire use in the future.

Another important aspect of active restoration is the use of native seed or native/nonnative seed mixes in sites lacking native understory. Native seed has been used in restoration with variable success in the Great Basin. Active restoration scenarios modeled in this study showed an increase in acreage of seeded vegetation types. However, climatic variation is predicted to have a strong influence on the success of seeding treatments. Experimental data on seeding treatments are needed to guide management more effectively.

The model results suggest a fairly grim future under all the scenarios, with acreages of PJ-invaded and weed-dominated landscape increasing and acreages of resilient vegetation types decreasing. It seems likely that in the Great Basin, restoration of ecosystems to

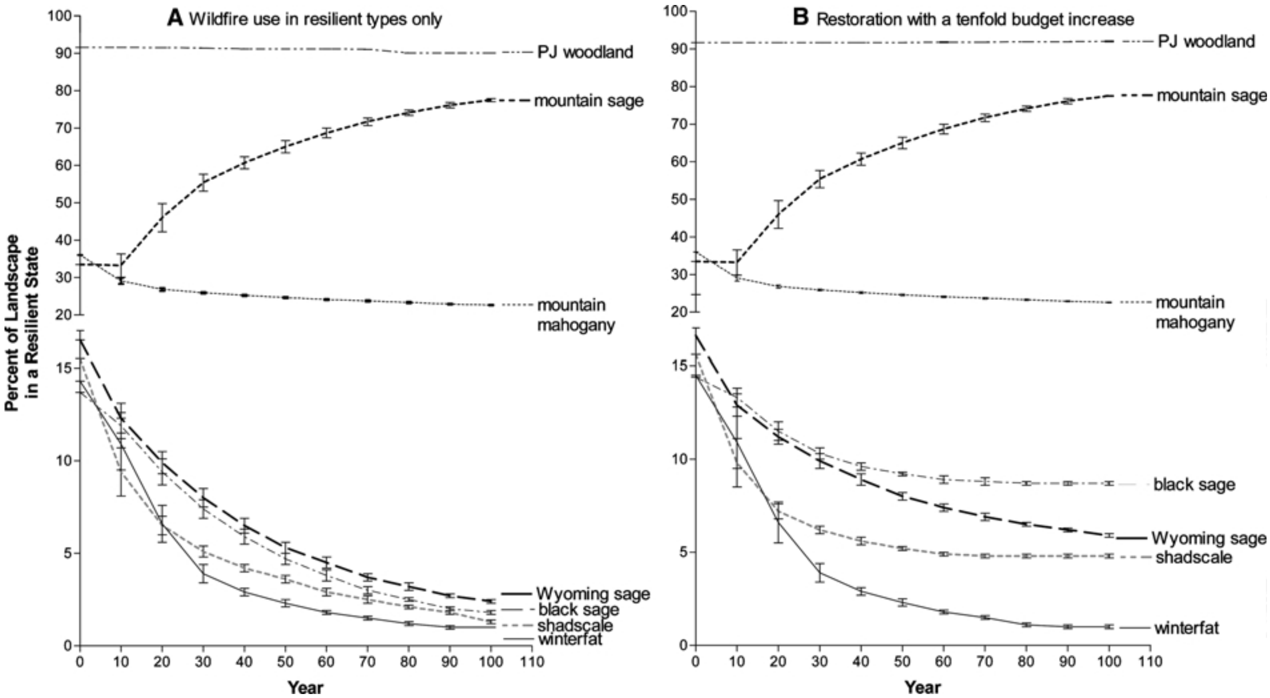

Figure 8. The percentage of the modeled area in resilient vegetation types (including states with perennial grass understory, seeded states, and states of pinyon–juniper woodland without weeds) over the 100-year modeled time frame. (**A**) *Wildfire use in resilient types only* scenario. (**B**) *Restoration with a tenfold budget increase* scenario. Bars represent decadal averages and standard deviations around ten Monte Carlo simulations.

presettlement conditions or restoration of vegetation to native species dominance will be impossible due to the ongoing loss of native species cover and seed banks and the gains made by invasive species (Bates and others 2000; West and Yorks 2002). Although these model results show declines in resilient vegetation types under all scenarios, strategic fire management and increased restoration input scenarios did allow for the maintenance of resilient hectares that would be lost to invasive species under the *current management* scenario. These projected maintenance achievements should be regarded as a significant contribution to ecological resilience and will become increasingly important as the invasion of exotic weeds, changes in fire regimes, increase in carbon dioxide, and other threats to ecosystem resilience continue to increase.

Uncertainty in Models

A few of the parameters used in this model (i.e., some aspects of fire history) were based on empirical data, but most were based on specialist or expert opinion. Therefore, they are part of the adaptive management process and should be changed as our understanding of vegetation dynamics improves, both through the accumulation of empirical data and through reliable observational information. Additionally, there are multiple site-dependent variables within each state that will affect restoration success probabilities (Tausch and others 1995). The effects from many of the disturbances that occur in this landscape are reflected only as averages in this landscape-scale model. These site-specific impacts of the management alternatives should be analyzed outside the context of this model using either professional judgment or spatial modeling. Ultimately, the greatest management need is statistically valid monitoring data on restoration treatments (Romme and others 2003) for determining effects and cost effectiveness. This is particularly important when large-scale management treatments are planned. Small pilot projects often are advisable when treatments will be applied in sites for which there is uncertainty about the ecological response. Additionally, collection of data on ongoing land management projects and analysis or meta-analysis of these data could yield essential information both about effective techniques for maintaining ecological resilience and about lessons from failed land management experiments.

Stochastic Dynamics

Another weakness of this modeling effort is that results largely indicated gradual changes along deterministic trajectories (Figures 4, 5, 7, and 8) despite our inclusion of parameters estimating wildfire and climatic stochasticity (Table 3). In presenting a state-and-transition management approach for the U.S. Southwest, Bestlemeyer and others (2004) suggested that a computer modeling approach such as the one we used in this study has limited value for two primary reasons: (1) modeling ecological change as a probabilistic process does not capture the nonlinear nature of ecological dynamics in the arid West, and (2) climatic variability is a strong driver that is, as yet, impossible to predict. A modeling effort focused on evaluating the sensitivity of models to alternative climate scenarios would be possible, but would be highly speculative. Indeed, processes of ecological recovery are thought to follow different patterns resembling a Markov chain, a random walk, or a more complex trajectory such as a Lorenz attractor (Anand and Desrochers 2004). These concerns are borne out by the anecdotal knowledge that Great Basin landscapes in northern Nevada changed dramatically in 1999 as climate-driven catastrophic wildfires facilitated the conversion of millions of acres to cheatgrass, and that many of the changes which have occurred since settlement are the result of interactions between changing land use patterns and a climate that has changed. State-and-transition models can be created to reflect the nonlinear dynamics in the arid West, but further development of parameters for reasonable description of nonlinear dynamics is necessary.

Appropriate Use of Models in Land Use Planning

Given the aforementioned caveats, we suggest that computer models can be a useful tool for systematically analyzing impacts in land use planning. However, for the modeling effort to be productive, the management situation must be conducive to open communication among partners, as in the current study, in which a community-based stakeholder group (ENLC) partnered with a science-based conservation organization (The Nature Conservancy of Nevada) and a land management agency (BLM). There must be trust among partners because parameters that can be contentious (e.g., livestock effects on vegetation, historic fire frequencies) should be developed cooperatively. There should also be hands-on involvement of managers, at both the senior management and technical specialist levels, so that information relevant to the site can be gleaned and vetted. Another party whose involvement would be beneficial is the land users themselves. In conclusion, it is not so much the results of the models as it is the process of parameterizing and simulating them that is the ultimate benefit of the exercise, and like the restoration treatments proposed in the Ely RMP, the benefits of modeling must be weighed against the economic and human resource costs of creating the models.

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Appendix

Appendix 1. States within each potential vegetation type used in the VDDT models

Potential vegetation type	State abbreviation	Description
BS, WS, MS, MM, PJ, WF, SS	AG	Annual grasses dominant
BS, WS, MS, MM, PJ, WF	Alt	Site altered by catastrophic fire and erosion
BS	BS	Black sagebrush with no understory
BS	BSAG	Black sage with annual grass understory
BS	BSPG	Black sage with perennial grass understory
BS	BSPGThrs	Black sagebrush with perennial grass at threshold
BS, WS, MS, MM, PJ, WF, SS	ExFo	Exotic forbs dominant
PJ	Imm	PJ open woodland with dominant perennial grass understory
PJ	Mat	PJ dominant with sparse perennial grass understory
MM	MM	Mahogany with no understory
MM	MMAG	Mahogany with annual grass understory
MM	MMPG	Mahogany with perennial grass understory
MM	MMPGThrs	Mahogany with perennial grass at threshold
MM	MMPJPG	Mahogany with PJ invading
MS	MS	Mountain sagebrush with no understory
MS	MSPG	Mountain sage with perennial grass understory
MS	MSPGThrs	Mountain sagebrush with perennial grass at threshold
MS	MSPJPG	Mountain sagebrush with PJ invading and perennial grasses
PJ	Over	PJ dominant with no understory
BS	PJAG	PJ with an understory of annual grass
BS	PJBS	Black sage site dominated by PJ
MS	PJMS	PJ dominant with mountain sagebrush present
WS	PJWS	Wyoming sage site dominated by PJ
WS	PJWSAG	Wyoming sage site dominated by PJ and with cheatgrass
PJ	PJExAG	PJ dominant with exotic forbs and annual grasses present
BS, WS, MS, MM, PJ, WF, SS	Seeded	Seeded with nonnative and/or native species
SS	SS	Shadscale with no understory
SS	SSPG	Shadscale with perennial grass understory
SS	SSPGThrs	Shadscale with perennial grass understory at threshold
WF	WF	Winterfat with no understory
WF	WFPG	Winterfat with perennial grass understory
WF	WFPGThrs	Winterfat with perennial grass understory at threshold
WS	WS	Wyoming sagebrush with no understory
WS	WSPG	Wyoming sage with perennial grass understory
WS	WSPGThrs	Wyoming sagebrush with perennial grass at threshold
WS	WSAG	Wyoming sage with annual grass understory

BS = black sagebrush, WS = mountain sagebrush, MM = mountain mahogany, PJ = pinyon-juniper woodland, WF = winterfat; SS, shadscale.

Appendix 2. Parameters used to model natural disturbances[a]

Vegetation type	Disturbance	From state	Probability	To state
BS	Annual grass invasion	BSPGThrs	0.01	BSAG
BS	Annual grass invasion	BSPGThrs	0.01	ExFo
BS	Annual grass invasion	BS	0.015	BSAG
BS	Annual grass invasion	PJBS	0.015	PJAG
BS	Exotic forb invasion	BS	0.001	ExFo
BS	Exotic forb invasion	BSAG	0.001	ExFo
BS	Exotic forb invasion	AG	0.001	ExFo
BS	Exotic forb invasion	PJBS	0.0001	ExFo
BS	Exotic forb invasion	PJAG	0.0001	ExFo
BS	PJ invasion	BSPG	0.01	PJBS
BS	PJ invasion	BSPGThrs	0.01	PJBS
BS	PJ invasion	BS	0.01	PJBS
BS	PJ invasion	BSAG	0.01	PJAG
BS	PJ invasion	Seeded	0.01	PJBS
BS	Wildfire	BSPG	0.00026	BSPG
BS	Wildfire	BSPGThrs	0.00026	BSPG
BS	Wildfire and annual grass invasion	BS	0.00007	AG
BS	Wildfire and annual grass invasion	BSAG	0.00007	AG
BS	Wildfire and annual grass invasion	PJBS	0.0005	AG
BS	Wildfire and annual grass invasion	PJAG	0.0005	AG
BS	Wildfire and seeding	BS	0.00007	Seeded
BS	Wildfire and seeding	BSAG	0.00007	Seeded
BS	Wildfire and seeding	PJBS	0.0005	Seeded
BS	Wildfire and seeding	PJAG	0.0005	Seeded
MM	PJ invasion	MM	0.01	MMPJ
MM	PJ invasion	ExFo	0.005	MMPJ
MM	Wildfire	MMPG	0.0005	MMPG
MM	Wildfire	MMPGThrs	0.0005	MMPG
MM	Wildfire and annual grass invasion	MM	0.00015	AG
MM	Wildfire and annual grass invasion	ExFo	0.00015	AG
MM	Wildfire and annual grass invasion	PJMS	0.00015	AG
MM	Wildfire and seeding	MM	0.00035	Seeded
MM	Wildfire and seeding	ExFo	0.00035	Seeded
MM	Wildfire and seeding	PJMS	0.00035	Seeded
MS	Annual grass invasion	PJMS	0.0005	AG
MS	PJ invasion	MSPGThrs	0.01	MSPG
MS	PJ invasion	MSPG	0.01	PJMS
MS	Wildfire	MSPG	0.003	MSPG
MS	Wildfire	MSPGThrs	0.003	MSPG
MS	Wildfire and annual grass invasion	MSPG	0.0001	AG
MS	Wildfire and annual grass invasion	PJMS	0.0001	AG
MS	Wildfire and seeding	MSPG	0.00231	Seeded
MS	Wildfire and seeding	PJMS	0.00231	Seeded
PJ	Annual grass invasion	Over	0.001	PJExAg
PJ	Exotic forb invasion	Over	0.0001	ExFo
PJ	Exotic forb invasion	AG	0.0001	ExFo
PJ	Exotic forb invasion	Seeded	0.0001	ExFo
PJ	Wildfire	Imm	0.00116	Imm
PJ	Wildfire	Mat	0.00116	Imm
PJ	Wildfire and annual grass invasion	Over	0.000348	AG
PJ	Wildfire and annual grass invasion	PJExAg	0.000348	AG
PJ	Wildfire and seeding	Over	0.000812	Seeded
PJ	Wildfire and seeding	PJExAg	0.000812	Seeded
SS	Annual grass invasion	SSPGThrs	0.005	AG
SS	Annual grass invasion	SS	0.005	AG
SS	Exotic forb invasion	SSPGThrs	0.001	ExFo
SS	Exotic forb invasion	SS	0.001	ExFo

Appendix 2. Continued.

Vegetation type	Disturbance	From state	Probability	To state
SS	Wildfire	SSPG	0.000001	SSPG
SS	Wildfire	SSPGThrs	0.000001	SSPG
SS	Wildfire and annual grass invasion	SS	5E-07	AG
SS	Wildfire and seeding	SS	5E-07	Seeded
WF	Annual grass invasion	WFPGThrs	0.01	AG
WF	Annual grass invasion	WF	0.015	AG
WF	Exotic forb invasion	WFPGThrs	0.001	ExFo
WF	Exotic forb invasion	WF	0.001	ExFo
WF	Exotic forb invasion	AG	0.001	ExFo
WF	Wildfire and annual grass invasion	AG	0	AG
WF	Wildfire and seeding	AG	0	Seeded
WS	Annual grass invasion	WSPGThrs	0.01	WSAG
WS	Annual grass invasion	WS	0.015	WSAG
WS	Annual grass invasion	PJWS	0.015	PJWSAG
WS	Exotic forb invasion	WSPGThrs	0.001	ExFo
WS	Exotic forb invasion	WS	0.001	ExFo
WS	Exotic forb invasion	WSAG	0.001	ExFo
WS	Exotic forb invasion	AG	0.001	ExFo
WS	Exotic forb invasion	PJWS	0.001	ExFo
WS	Exotic forb invasion	PJWSAG	0.001	ExFo
WS	PJ invasion	WSPG	0.01	PJWS
WS	PJ invasion	WSPGThrs	0.01	PJWS
WS	PJ invasion	WS	0.01	PJWS
WS	PJ invasion	WSAG	0.01	PJWSAG
WS	PJ invasion	Seeded	0.01	PJWS
WS	Wildfire	WSPG	0.001	WSPG
WS	Wildfire	WSPGThrs	0.001	WSPG
WS	Wildfire	WSPGThrs	0.001	WSPG
WS	Wildfire and annual grass invasion	WS	0.0005	AG
WS	Wildfire and annual grass invasion	WSAG	0.0005	AG
WS	Wildfire and annual grass invasion	PJWS	0.0005	AG
WS	Wildfire and annual grass invasion	PJWSAG	0.005	AG
WS	Wildfire and seeding	WS	0.0005	Seeded
WS	Wildfire and seeding	WSAG	0.0005	Seeded
WS	Wildfire and seeding	PJWS	0.0005	Seeded
WS	Wildfire and seeding	PJWSAG	0.005	Seeded

BS = black sagebrush; WS = Wyoming sagebrush; MS = mountain sagebrush; MM = mountain mahogany; PJ = pinyon–juniper woodland; WF = winterfat.

[a]State abbreviations are provided in Appendix 1.

Appendix 3. Parameters used to model cattle grazing[a]

		Spring		Summer		Dormant		Year round	
Vegetation type	State	Prob	Effect	Prob	Effect	Prob	Effect	Prob	Effect
BS	BSPG	0.25	3	0.05	2	0.35	1		
BS	BSPGThrs	0.4	3.5	0.05	3	0.3	1		
BS	AG	0.8	0			0.2	0		
BS	Seeded	0.25	3	0.05	2	0.35	1		
WS	WSPG	0.25	3	0.05	2	0.35	1		
WS	WSPGThrs	0.4	3.5	0.05	3	0.3	1		
WS	AG	0.8	0			0.2	0		
WS	Seeded	0.25	3	0.05	2	0.35	1		
MS	MSPG			0.5	1.5				
MS	MSPGThrs			0.5	2				
MS	MSPG			0.5	2				
MS	Seeded			0.5	2				
MM	MMPG			0.5	1.5				
MM	MMPGThrs			0.5	2				
MM	Seeded			0.5	2				
PJ	Seeded			0.5	2				
SS	SSPG	0.4	1			0.7	1		
SS	SSPGThrs	0.4	2			0.7	1		
SS	AG	0.8	0			0.2	0		
SS	SS	0.05	-1						
WF	WFPG	0.4*	3			0.7	0		
WF	WFPGThrs	0.4*	3			0.7	0		
WF	WF	0.4	to altered			0.7	0		
WF	AG	0.7	0			0.2	0		

Prob = Probability; BS = black sagebrush; WS = Wyoming sagebrush; MS = mountain sagebrush; MM = mountain mahogany; PJ = pinyon–juniper woodland; SS = Shadscale; WF = winterfat.

[a]The grazing parameters have two parts: a probability and a successional effect. The probability ranges from 0 to 1 and describes the amount of vegetation in a given state that is affected per year. The successional effect is the number of successional time steps that grazing pushes succession either forward or backward per year.

*Half of the disturbed simulation units go to the altered state; the rest are aged within the state.

Appendix 4. Parameters used to model sheep grazing[a]

		Spring		Summer		Dormant	
Vegetation type	State	Prob	Effect	Prob	Effect	Prob	Effect
BS	BSPG	0.2	2			0.6	−2
BS	BSPGThrs	0.2	2			0.6	−2
BS	BS	0.2	2			0.6	−2
BS	BSAG	0.2	2			0.6	−2
BS	Seeded	0.05	2				
WS	WSPG	0.125	2			0.125	−2
WS	WSPGThrs	0.125	2			0.125	−2
MS	MSPG			0.4	1		
MS	MSPGThrs			0.4	1		
MS	MSPG			0.1	1		
MS	Seeded			0.4	1		
MM	MMPG			0.4	1		
MM	MMPGThrs			0.4	1		
SS	SSPG	0.05	−1			0.3	−2
SS	SSPGThrs	0.05	−1			0.3	−2
SS	SS	0.05	−2			0.3	−2
WF	WFPG					0.2	−1
WF	WFPGThrs					0.2	−1
WF	WF					0.2	−1

Prob = Probability; BS = black sagebrush; WS = Wyoming sagebrush; MS = mountain sagebrush; MM = mountain mahogany; SS = shadscale; WF = winterfat.

[a]Model use of probabilities and effects is described for Appendix 3. State abbreviations are provided in Appendix 1.

Appendix 5. Parameters used to model wild horse grazing[a]

Horses in HMA		Spring		Summer		Dormant		Year round	
Vegetation type	State	Prob	Effect	Prob	Effect	Prob	Effect	Prob	Effect
BS	BS/PG							0.7	5
BS	BS/PG/Thrs							0.7	5
BS	AG							0.5	0
BS	Seeded							0.7	5
WS	WS/PG							0.7	5
WS	WS/PG/Thrs							0.7	5
WS	AG							0.5	0
WS	Seeded							0.7	5
MS	MS/PG			0.75	2.5	0.375	1.5		
MS	MS/PG/Thrs			0.75	3	0.375	2		
MS	Seeded			0.77	3	0.385	2		
MM	MM/PG			0.75	2.5	0.375	1.5		
MM	MM/PG/Thrs			0.75	3	0.375	2		
MM	Seeded			0.77	3	0.385	2		
PJ	Seeded							0.85	5
SS	SS/PG	0.4	1			0.7	1		
SS	SS/PG/Thrs	0.4	2			0.7	1		
SS	AG	0.8	0			0.2	0		
SS	SS	0.05	−1						
WF	WF/PG	0.3	4	0.05	3	0.25	0		
WF	WF/PG/Thrs	0.3	4	0.05	3	0.25	0		
WF	WF	0.03	−3	0.05	−2	0.25	0		
WF	AG	0.7	0			0.2	0		

Prob = Probability; BS = black sagebrush; WS = Wyoming sagebrush; MS = mountain sagebrush; MM = mountain mahogany; SS = shadscale; WF = winterfat.

[a]Model use of probabilities and effects is described for Appendix 3.

Appendix 6. Parameters used to model invasive species movement into various vegetation states due to the effects of road use and road creation

Vegetation type	Invader	From state	Probability	To state
WS	AG	WSPGThrs	0.001	WSAG
WS	ExFo	WSPGThrs	0.0001	ExFo
WS	AG	WS	0.001	WSAG
WS	ExFo	WS	0.0001	ExFo
WS	ExFo	WSAG	0.0001	ExFo
WS	ExFo	AG	0.0001	ExFo
WS	ExFo	PJWS	0.0001	ExFo
WS	AG	PJWS	0.001	PJWSAG
WS	ExFo	PJWSAG	0.0001	ExFo
WF	AG	WFPGThrs	0.001	AG
WF	ExFo	WFPGThrs	0.0001	ExFo
WF	AG	WF	0.001	AG
WF	ExFo	AG	0.0001	ExFo
SS	AG	SSPGThrs	0.001	AG
SS	ExFo	SSPGThrs	0.0001	ExFo
SS	AG	SS	0.001	AG
SS	ExFo	SS	0.0001	ExFo
SS	ExFo	AG	0.0001	ExFo
PJ	AG	Mat	0.0001	PJExAg
PJ	ExFo	Mat	0.00001	PJExAg
PJ	AG	Over	0.0001	PJExAg
PJ	ExFo	Over	0.00001	ExFo
PJ	AG	PJExAg	0.0001	AG
PJ	ExFo	PJExAg	0.00001	ExFo
PJ	ExFo	AG	0.00001	ExFo
MS	AG	MSPGThrs	0.0001	AG
MS	ExFo	MSPGThrs	0.00001	ExFo
MS	ExFo	AG	0.00001	ExFo
MS	AG	MSPG	0.0001	AG
MS	ExFo	MSPG	0.00001	ExFo
MS	AG	PJMS	0.0001	AG
MS	ExFo	PJMS	0.00001	ExFo
MM	AG	MMPGThrs	0.0001	MMAG
MM	ExFo	MMPGThrs	0.00001	ExFo
MM	AG	MM	0.0001	MMAG
MM	ExFo	MM	0.00001	ExFo
MM	ExFo	MMAG	0.00001	ExFo
MM	ExFo	AG	0.00001	ExFo
MM	AG	MMPJ	0.0001	MMAG
MM	ExFo	MMPJ	0.00001	ExFo
BS	AG	BSPGThrs	0.001	BSAG
BS	ExFo	BSPGThrs	0.0001	ExFo
BS	AG	BS	0.001	BSAG
BS	ExFo	BS	0.0001	ExFo
BS	ExFo	BSAG	0.0001	ExFo
BS	ExFo	AG	0.0001	ExFo
BS	AG	PJBS	0.001	AG
BS	ExFo	PJBS	0.0001	ExFo
BS	ExFo	PJAG	0.0001	ExFo

WS = Wyoming sagebrush; WF = winterfat; SS, shadscale; PJ = pinyon-juniper woodland; MS = mountain sagebrush; MM = mountain mahogany; BS = black sagebrush.
[a]State abbreviations are provided in Appendix 1.