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Comparing alternative management strategies of fire, grazing, and weed control using spatial modeling

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ABSTRACT

Modeling can be used to resolve controversies generated by differing opinions about the effects of livestock grazing, fire management, and herbicide application on western public lands. We used spatial simulations of 10 potential vegetation types to compare 6 management scenarios over 20 years in a 141,853 ha landscape in eastern Nevada. Scenarios were compared by incrementally varying one factor at a time and were based on the Bureau of Land Management's (BLM's) potential restoration plans. The following factors were varied: managed fire, livestock grazing, mechanical and chemical treatment of vegetation, and restoration budgets. After 20 years the differences in vegetative composition between scenarios were small. BLM's level of funding was too low to improve ecological condition because the landscape was too degraded, however, current funding could maintain communities that retained native perennial understories. In general, the effects of livestock grazing were minor and undesirable compared to benefits gained from the use of mechanical and chemical methods followed by seeding. Mechanical methods and herbicide application in addition to current fire management had more desirable effects than without fire management.

1. Introduction

Livestock grazing, fuels management, and herbicide application on western public lands are controversial topics often strongly opposed or supported by environmental advocacy groups, local communities, the livestock industry, conservation organizations, Native American tribes, and other groups (Fleischner, 1994; Brown and McDonald, 1995; Brussard et al., 1994; Wuerthner and Matteson, 2002; Freilich et al., 2003). Stakeholders support or challenge the actions of public land managers because they share different values about land uses and/or because there is historic distrust of public land management. Disagreements about public land management often increase with the size of a landscape and its ecological complexity (Walters and Holling, 1990), and the paucity of ecological knowledge on key features of the ecosystem (e.g., Baker and Shinneman, 2004).

Controversies related to range management are common because over the last 150 years western rangelands have undergone unprecedented change (Blackburn and Tueller, 1970; National Research Council, 1994; McPherson and Weltzin, 2000; Young and Sparks, 2002). Prior to settlement, the grasslands and shrublands of the arid West were structured primarily by fire, precipitation cycles, and insects with grazing

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ungulates playing a role whose importance varied regionally. However, these roles have changed; domestic livestock now graze the large majority of both private and public lands in western North America, and wildfire occurs at times, frequencies, and intensities that are outside of pre-settlement ranges (Blackburn and Tueller, 1970; Brown and McDonald, 1995; Schmidt et al., 2002). Longer fire-free intervals, the long-term historic consumption of fine fuels by livestock, and aggressive policies of fire-suppression starting in the 1920s (Pyne, 2004) have favored the expansion of woody species throughout grasslands and steppes that historically supported few trees, even in areas that have had livestock use removed for decades (Miller and Rose, 1999; Tausch and Nowak, 1999; Curtin and Brown, 2001; Pyne, 2004).

While longer fire-free intervals have favored woody species, the regional-scale invasion of cheatgrass (*Bromus tectorum* L.) has shortened fire-free intervals. Cheatgrass, a non-native annual, increased dramatically after historic livestock use reduced native bunchgrasses and forbs (Young et al., 1987; Young and Sparks, 2002). Because native plant species do not survive the frequent fires facilitated by cheatgrass (Young et al., 1987), or do not compete successfully against cheatgrass for soil moisture (Melgoza et al., 1990), and some do not disperse as effectively, the system moves toward a cheatgrass monoculture nearly devoid of biodiversity, habitat, and economic values. Cheatgrass control, even for the purpose of restoring native species, is resisted by the public because it is best achieved by the application of herbicides.

Adaptive management theory proposes that stakeholders may reduce the uncertainty of management dilemmas by comparing the effects of alternative, sometime novel management actions on whole ecosystems using simple, yet robust experimental design procedures (Walters and Holling, 1990; Wilhere, 2002). Because the space, investment, and time frame required to carry out an experiment can be large, modeling of alternative management actions is often recommended prior to experimentation, if only to discard ineffective actions and document beliefs about system function (Hilborn et al., 1995; Hardesty et al., 2000; Forbis et al., 2006). Managers also may not have the time or funding to wait several years for experimental results, therefore, modeling provides more immediate recommendations while field data are being collected and interpreted.

State-and-transition models (Horn, 1975; Westoby et al., 1989; McIver and Starr, 2001; Bestelmeyer et al., 2004) are increasingly popular in natural resource management because their discrete representations of vegetation dynamics simplify ecological complexity and can be developed in cooperation with specialists and lay-people. It is also useful that public domain software exists to easily develop state-and-transition models from scratch and rapidly view simulated results (e.g., Beukema et al., 2003b; Forbis et al., 2006).

State-and-transition modeling is largely a-spatial (e.g., Westoby et al., 1989; Miller and Tausch, 2001; Stringham et al., 2003; Bestelmeyer et al., 2004). A-spatial models are far easier to understand and quantify than spatial ones. There are, however, compelling circumstances in which the spatial component cannot be ignored because the spatial interactions among vegetation types and states change ecological processes and management outcomes (Schroeder et al., 1999; Hemstrom et al., 2001; Keane et al., 2002). Spatial modeling might also appeal to managers if the model is applied to the digital version of a real landscape where they can test alternative scenarios and view simulation results on maps of relevant landscapes (e.g., Hemstrom et al., 2001; Hardesty et al., 2000; Keane et al., 2002).

We spatially simulated the effects of six different scenarios of livestock, fire, and non-native species management on the composition of vegetation for a 141,853 ha public lands landscape. A central goal of our spatial modeling effort was to integrate expert knowledge to best estimate the effects of controversial management strategies for public lands. We chose

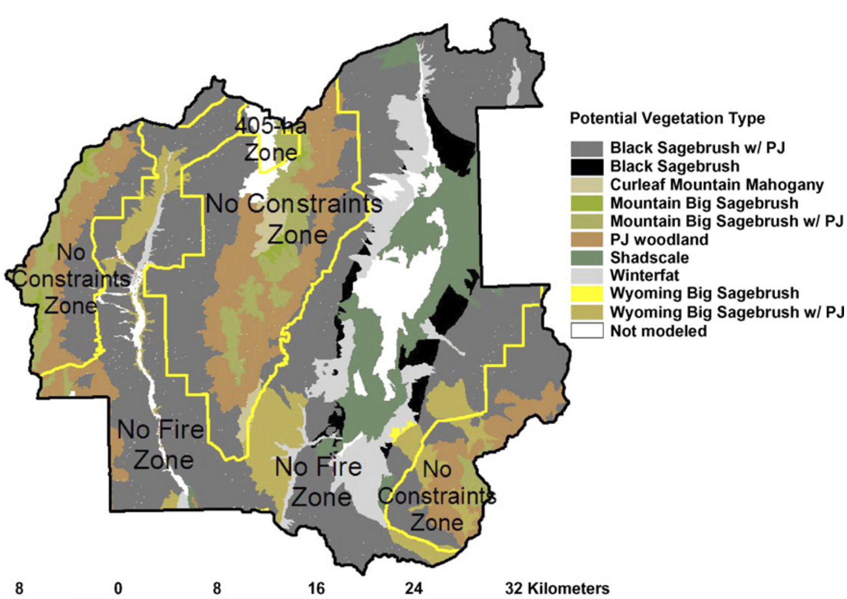

Fig. 1 – Potential vegetation types and fire suppression zones for the Antelope and North Spring valleys, Eastern Nevada. The black lines delineate the fire suppression zones; no constraints, 405-ha (1000-acres) fire, and no fire.

20-year simulations because that is the average lifespan of a BLM Resource Management Plan. The modeled landscape was Antelope and North Spring Valleys (ANSV; Fig. 1), which consists of two adjacent watersheds with a central mountain range near the Nevada–Utah border. It exhibits classic basin and range zonal vegetation with strong elevation gradients extending from saline valley bottoms at ~1524 m to mountain tops at ~3364 m.

Management scenarios were proposed by BLM managers and stakeholder groups. Specifically, we wanted answers to the following Questions. (1) What is the effect of current fire management (fire suppression and prescribed burning) compared to managing the land with unmanaged fire regimes only? (2) What is the effect of livestock management compared to managing the land with spatially constrained wildfire and prescribed fire, but no livestock? (3) What is the effect of vegetation management with mechanical and chemical methods of restoration compared to managing the land only with fire and livestock? (4) Does increasing the size of the Ely BLM's restoration budget cause a proportional improvement in vegetation types (greater percentage of states dominated by perennial grasses and smaller percentage of weeds and tree encroachment states)?

2. Methods

We used TELSA®, the Tool for Exploratory Landscape Scenario Analyses (ESSA Technologies, Ltd.; Kurz et al., 2000; Beukema et al., 2003a), to develop models of alternative management strategies for ANSV. TELSA is a spatially explicit simulator that interfaces with geographic information system software (ArcView® by ESRI) and a relational database (MS Access®) to model ecological succession, vegetation transitions caused by natural (e.g., drought) or anthropogenic (e.g., exotic species invasion and livestock grazing) processes, and management actions. TELSA requires the following data inputs: (1) a polygon map of vegetation cover initial conditions with attributes for potential vegetation type (defined in next paragraph), vegetation state, and age since the last event that removed all vegetation (i.e., severe fire), (2) for each potential vegetation type, a state-and-transition model developed with Vegetation Dynamics Development Tool (VDDT; Barrett, 2001; Beukema et al., 2003b; Forbis et al., 2006), (3) a polygon map of management regions specifying what kinds of management could occur, and its annual limits, (4) size distributions for each natural disturbance, (5) multiplier sequences describing the temporal variability of disturbance probabilities, and (6) management rules including treatment block sizes, annual limits and adjacency constraints.

Potential vegetation types are one type of biophysical classification based on dominant plant species that are indicators of the natural disturbance regime, local climate, and topo-edaphic relationships (Schmidt et al., 2002; Hann, 2004). Biophysical characteristics that to a large extent control fire regimes and the distribution of vegetation are reflected in the distribution of potential vegetation types (Keane et al., 2002). The potential vegetation types represent the vegetation type that would exist under pre-settlement or current natural regimes of ecological processes in the absence of modern human interference (Schmidt et al., 2002; Hann, 2004). Potential vegetation types were the foundation for stratification of reference and current vegetation. In the model, potential vegetation types were represented by static polygons on the landscape. Within potential vegetation type polygons, the total possible set of vegetation states, and the transitions between these states were defined. Thus, each potential vegetation type could be represented by a different state-and-transition model.

A vegetation state in VDDT and TELSA is defined by its successional or structural stage (for example post replacement versus decadent). The definition of "state" in the rangeland literature is operationally the same as used here, but more formal (Bestelmeyer et al., 2004): states are persistent vegetation and soil changes per potential vegetation type that can be represented in a diagram with two or more boxes (phases of the same state). Moreover, different states are separated by "thresholds", which implies that expensive management actions would be required to restore ecosystem structure and function. Relatively reversible changes (e.g., fire, flooding, drought, insect outbreaks, and others), unlike thresholds, operate between phases within a state, but not among states. Therefore, each potential vegetation type is subdivided into polygons representing vegetation states ranging from post-disturbance states (generally dominated by herbaceous vegetation) to late-successional states (generally dominated by woody vegetation) and including states with different levels of invasive species cover. Modeled states within each potential vegetation type are shown in Table 1.

2.1. *Model algorithms*

In VDDT, succession and disturbance are simulated in a semi-Markovian framework. Each vegetation state has one possible deterministic transition based on time in the state (succession) and several possible probabilistic transitions (natural and management disturbances). Each of these transitions has a new destination state and probability associated with it. Based on the timing of the deterministic transition and the probabilities of the stochastic transitions, at each time step a polygon may remain the same, undergo a deterministic transition based on elapsed time in the current state or undergo a probabilistic transition based on a random draw. TELSA puts this semi-Markovian framework into a spatially explicit context in which polygons interact with each other. For example, in VDDT disturbance events are non spatial and occur independently at the simulation unit level; in TELSA disturbance events initiate at a single polygon and then spread to adjacent polygons and beyond. In TELSA, disturbance events may even spread between potential vegetation types. The VDDT models developed here were later modified for the revision of the Resources Management Plan for the Ely BLM (Forbis et al., 2006); therefore, we refer the reader to Forbis et al. (2006) for the general VDDT methodology and one example of the diagram of a state-and-transition model while we present specific features of VDDT and elaborate more on TELSA development. The TELSA model algorithms are described in detail by Kurz et al. (2000).

Each 20 year simulation was replicated three times. Results show means ± 1S.E. While this may seem like a low number

Table 1 – Names and codes of 10 Eastern Nevada potential vegetation types, states, and phases

Vegetation type	Vegetation type code	State or phase name	State or phase code
Shadscale	SH		
		Perennial grass dominant	SSPG
		Perennial grass dominant at threshold	SSPGThrs
		Shrub dominant	SS
		Exotic forb dominant	ExFo
		Annual grass dominant	AG
Winterfat	WF		
		Perennial grass dominant	WFPG
		Perennial grass dominant at threshold	WFPGThrs
		Shrub dominant	WF
		Exotic forb dominant	EXFO
		Annual grass dominant	AG
		Altered	ALT
Black sagebrush with or without pinyon–juniper	BS, BSPJ		
		Perennial grass dominant	PGBS
		Perennial grass dominant at threshold	BSPGThrs
		Shrub dominant	BS
		Shrub dominant with annual grass understory	BSAG
		Exotic forb dominant	EXFO
		Annual grass dominant	AG
		Altered	ALT
		Seeded	Seeded
	BSPJ (only)	Pinyon–juniper dominant	PJBS
	BSPJ (only)	Pinyon–juniper dominant with annual grass understory	PJAG
Wyoming big sagebrush with or without pinyon–juniper	WS, WSPJ		
		Perennial grass dominant	WSPG
		Perennial grass dominant at threshold	WSPGThrs
		Shrub dominant	WS
		Shrub dominant with annual grass understory	WSAG
		Exotic forb dominant	EXFO
		Annual grass dominant	AG
		Seeded	Seeded
		Altered	ALT
	WSPJ (only)	Pinyon–juniper dominant	PJWS
	WSPJ (only)	Pinyon–juniper dominant with annual grass understory	PJWSAG
Pinyon–juniper woodland	PJ		
		Perennial grass dominant	PJPG
		Tree dominant with an understory of exotic forbs or annual grasses	PJEXAG
		Exotic forb dominant	EXFO
		Annual grass dominant	AG
		Seeded	Seeded
		Altered	ALT
Mountain mahogany woodland	MM		
		Perennial grass dominant	MMPG
		Perennial grass dominant at threshold	MMPGThrs
		Mountain mahogany with pinyon–juniper co-dominance	MMPJ
		Pinyon–juniper dominant with mountain mahogany subdominant	PJMM
		Pinyon–juniper dominant	PJ
Mountain big sagebrush with or without pinyon–juniper	MS, MSPJ		
		Perennial grass dominant	MSPG
		Perennial grass dominant at threshold	MSPGThrs
		Shrub dominant	MS, MSPJPG
		Exotic forb dominant	EXFO
		Annual grass dominat	AG
		Altered	ALT
		Seeded	Seeded

of replications we were limited by the large size of the landscape and the practicalities of processing time and computer memory. Furthermore, while this amount of replication may be inappropriate for describing the range of variability at a site specific level, our indicators for the evaluation of strategies are aggregated across the entire landscape and therefore replicated across thousands of polygons in each potential vegetation type.

2.2. Eastern Nevada state-and-transition models

We stratified our landscape into 10 potential vegetation types (Fig. 1): shadscale (*Atriplex confertifolia*), winterfat (*Krascheninnikovia lanata*), black sagebrush (*Artemisia nova*) with and without trees, Wyoming big sagebrush (*A. tridentata* subsp. *wyomingensis*) with and without trees, and mountain big sagebrush (*A. tridentata* subsp. *vaseyana*) with and without trees, pinyon (*Pinus monophylla*)–juniper (*Juniperus osteosperma*) woodland (as defined by Miller et al., 1999), and curlleaf mountain mahogany (*Cercocarpus ledifolius* var. *intermontanus*). Black, Wyoming big, and mountain big sagebrush models were developed in the "with and without trees" versions to account for the physiological limits of pinyon and juniper to invade lower (<1775 m) and higher (>2700 m) elevations.

We used the Natural Resource Conservation Service (NRCS) order three soil surveys (USDA NRCS, 1997) and U.S. Geological Survey's Digital Elevation Model data to map potential vegetation types by pooling different ecological sites with the same dominant upper-layer species (Fig. 1). Soils take centuries to form as an interaction of climate, geology, and vegetation. Therefore, they can be used to approximate the pre-settlement or current natural, long-term ecological potential for soil–vegetation interactions (Haines-Young, 1991; Franklin, 1995). Given that the pre-settlement period ended approximately 150 years ago in the Great Basin, current soils should be reliable predictors of potential vegetation types unless soil horizons were mechanically removed or severely eroded due to post-settlement land management practices. A few minor potential vegetation types were omitted because their areas were small, they were not fire dependent, or we could not find any information about state-and-transition models for them; they included pygmy sagebrush (*A. pygmaea*), low sagebrush (*A. arbuscula*), alkali sacaton (*Sporobolus airoides*), sickle saltbush (*Atriplex gardneri*), and fourwing saltbush (*A. canescens*). Table 1 represents the different states, phases, and their abbreviations per potential vegetation type.

Wildfire was the primary stochastic disturbance in all vegetation types, except for shadscale and winterfat that are potential vegetation types with no evolutionary history of fire (Young and Sparks, 2002). We assumed that the duration of mean fire return intervals decreased with soil productivity or moisture (Table 2). Wildfire generally resets the successional clock to zero within the reference condition, which is labeled the PERENNIAL GRASS DOMINANT state in most potential vegetation types. The PERENNIAL GRASS DOMINANT state represents a native condition of shrubland with a functioning cover of herbaceous species dominated by perennial cool-season bunch grasses. Wildfire was predicted to cause stand replacement and type conversion for states lacking an understory or invaded by pinyon and juniper for sufficiently long period of time (Tausch et al., 1993; Frelich and Reich, 1998; Tausch, 1999; Anderson and Inouye, 2001). Following BLM policy that allows for rehabilitating or stabilizing wildfire burned areas with a reduced native understory, these states were artificially reseeded with native or a mixture of native and non-native non-invasive species. This reseeding transition (termed WF.SEED), modeled as a natural disturbance, transforms the randomly chosen pixel into a SEEDED state or, for shadscale and winterfat only, to the PERENNIAL GRASS DOMINANT state. Yearly variation in wildfire activity was also built into all models using yearly multipliers developed during a workshop with BLM staff (Fig. 2).

Cattle and sheep grazed many states in every potential vegetation type, although cattle were restricted to the PERENNIAL GRASS DOMINANT, SEEDED, and SHRUB DOMINANT (winterfat only) states where forage was available (Table 3; states and phases defined in Table 1). In addition to these states, sheep

Table 2 – Fire return intervals of potential natural vegetation types

PVT	State	Time of earliest fire (years)	Fire return interval (years)[a]
Shadscale	SH	61	133
Winterfat	WF	200[b]	
Black sagebrush[c]	BSPG, BSPGThrs	50	75
Black sagebrush with trees	PJBS	70	100
Wyoming big sagebrush[c]	WSPG, WSPGThrs	20	33
Wyoming big sagebrush	WS	80	60
Wyoming big sagebrush with trees	PJWS	50	250
Pinyon–juniper	PJPG	125	300
Curlleaf mountain mahogany	MMPG, MMPGThrs, MMPJ, PJMM, PJ	1	200
Mountain big sagebrush[c]	MSPG, MSPGThrs, MS	1	20
Mountain big sagebrush with trees	MSPJ	81	50

State legend: BSPJ, black sagebrush with pinyon or juniper invasion; EXFO, exotic forbs; MM, curlleaf mountain mahogany; MS, mountain sagebrush; MSPJ, mountain sagebrush with pinyon or juniper invasion; PG, perennial grass; PJ, pinyon and juniper; SH, shadscale; Thrs, ecological threshold; WF, winterfat; WS, Wyoming big sagebrush; WSPJ, Wyoming big sagebrush with pinyon or juniper invasion.

[a] Prob/year, which is equal to 1/fire return interval, is used in VDDT.

[b] Wildfire only occurs in later seral stages with a cheatgrass component.

[c] With and without trees.

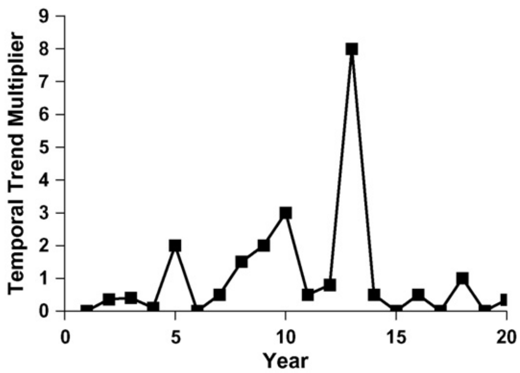

Fig. 2 – Temporal multipliers applied to modify rates for wildfire and sagebrush competition.

could browse shrubs in states without an understory or with a cheatgrass understory. The greatest differences between cattle and sheep grazing were the greater use of a polygon by sheep than cattle and the differential effect on vegetation succession. Sheep operators frequently move flocks to maintain a fresh and palatable supply of forage, therefore sheep grazing was assumed to affect 80% of the polygons in a state. On the other hand, cattle affected 5% of polygons per year. Season of use was modeled explicitly to reflect the use of the elevational gradient and forage types by livestock operators. Mountain big sagebrush, mountain mahogany, and pinyon–juniper were only grazed during the summer, whereas winterfat was only used during the winter. While model time steps were annual, the potential vegetation types where grazing transitions occur depend on season of use. Shadscale was grazed during the spring and winter. Black sagebrush and Wyoming big sagebrush had identical seasons of use with 50% of the utilization occurring year round, and 25% use during both spring and summer.

Cattle primarily grazed herbaceous vegetation; therefore they generally increased the cover of woody vegetation (Table 3), which was equivalent to accelerating succession (West and Yorks, 2002; Beever et al., 2003). There was one exception; spring grazing in winterfat caused a reversal of woody succession because cattle would select winterfat over grass and winterfat is more sensitive than grass to spring grazing. Successional effects varied with season of use (Table 3). Because sheep eat both herbaceous and woody material they have more complex effects on rangelands, which varied with season of use and vegetation types (Table 3; Harniss and Wright, 1982; Bork et al., 1998).

Livestock were not the only important herbivores in this landscape. Wild horses were common, more mobile than cattle, grass specialists, and *de facto* year-round grazers (Berger, 1986; Beever et al., 2003). Native herbivory included rabbit browsing on winterfat, and deer, elk, rodents, and rabbits maintaining the perennial grass dominance by browsing of mountain mahogany seedlings (Arno and Wilson, 1986; Schultz et al., 1996; Ross, 1999).

Pinyon and juniper encroachment of shrublands was a time-dependent process because seedlings require sagebrush as a nurse plant. The rate of pinyon–juniper invasion was 0.001 per year and could start, respectively, as early as 70, 50, 31, and 41 years following a stand replacement event for black, Wyoming big, and mountain big sagebrush, and mountain mahogany.

An important group of anthropogenic disturbances was the invasion of non-native plant species, specifically cheat-

Table 3 – Disturbance rates and successional effects of cattle and sheep grazing per potential vegetation type

PVT	Season of use							
	Spring Pr/yr	Succ. effect	Summer Pr/yr	Succ. effect	Dormant Pr/yr	Succ. effect	Year round effect	Succ. effect
Cattle								
Shadscale	0.025	4			0.025	1		
Winterfat	0.002	−1	0.008	3	0.04	1		
Black sagebrush[a] and Wyoming big sagebrush[a]	0.0125	4	0.0125	3			0.025	5
Pinyon–juniper			0.05	3				
Curlleaf mountain mahogany			0.05	3				
Mountain big sagebrush[a]			0.05	3				
Sheep								
Shadscale	0.4	2			0.4	−2		
Winterfat					0.8	−1		
Black sagebrush[a]	0.2	2	0.2	1			0.4	−1
Wyoming big sagebrush[a]	0.125	2	0.125	1			0.25	−1
Pinyon–juniper			0.8	1				
Curleaf mountain mahogany			0.8	1				
Mountain big sagebrush[a]			0.8	1				

Disturbance rate is expressed as a probability/year (Pr/yr) of grazing. Successional effect (Succ. effect) is the number of years a polygon is moved backward or forward in succession time if selected for 1 year (i.e., reversed woody succession or accelerated woody succession, respectively). *Legend*: Spring, spring grazing; Summer, summer grazing; Dormant, dormant season of grazing; Year round, year round grazing.

[a] With or without trees.

grass (annual grass or AG) and exotic forbs (EXFO) represented mainly by knapweeds (*Centaurea* spp.) and halogeton (*Halogeton glomeratus*). Forbis et al. (2006) describe in detail the process of non-native plant species invasion.

Other natural disturbances included insect outbreaks (stand replacing events), which were associated with high moisture and drought years in shadscale and winterfat, as well as sagebrush competition, and soil erosion. Shadscale is susceptible to multi-year drought that weakens plants, making them susceptible to insect outbreaks. A disturbance rate of every 20 years (0.05 per year) for the combined drought event and insect outbreak was used. The yearly variation for high moisture (flood) and drought related parameters were further modified by setting TELSA temporal multipliers to zero in all years except in year 12 (multiplier of 1) for flooding and year 7 (multiplier of 20) for drought. Sagebrush competition represented a weak and temporally variable process where the growth of a sagebrush plant caused the reduction, but not the elimination of the understory through shading and nutrient competition (Blaisdell, 1949; Pedersen et al., 2003). Erosion was the slow loss (Prob/year = 0.0001) of topsoil when high precipitation events fall on a burned area without native understory.

Names and probabilities, respectively, associated with all natural disturbances are available in Electronic Archives I and II (doi:10.1016/j.ecolmodel.2007.06.030).

2.3. Management activities

Management activities included mechanical treatment, prescribed fire, seeding, and herbicide. There were two distinct aspects to management disturbances: the rates used in VDDT models and the spatial constraints imposed by TELSA. We discuss here the VDDT parameters and address TELSA constraints and scenarios later.

As a rule of thumb, prescribed fire and any operation not followed by seeding were applied to states where the native perennial understory vegetation was present and was assumed to be releasable; the PERENNIAL GRASS DOMINANT state and PERENNIAL GRASS DOMINANT AT THRESHOLD state. This last state is a late-succession phase of the former that is at the brink of permanently losing its ability to recover its native cover from a natural disturbance. In general, the rate for prescribed fire attempts to match the natural (i.e., pre-settlement or current naturally functioning) wildfire rate of the potential vegetation type (Table 2). Both herbicide application and mechanical treatment without seeding for sagebrush were only used during the PERENNIAL GRASS DOMINANT AT THRESHOLD state to reverse the closure of sagebrush cover or to kill low levels of invading cheatgrass and exotic forbs.

In all other states, seeding followed herbicide application, mechanical treatment, or prescribed fire (rarely used in this context) because these states have lost their native understory, and/or the understory was dominated by non-native species. Although there was some variation in rates of application, a prob/year of 0.01 was commonly used to reflect that 1% of the state was treated on average every year. Probabilities associated with all management disturbances are available in Electronic Archive III (doi:10.1016/j.ecolmodel.2007.06.030).

2.4. Spatial constraints and scenarios

2.4.1. Fire management zones

Fire suppression zones from the Ely's BLM Fire Plan (USDI-BLM, 2000) were used as planning zones in TELSA (Fig. 1). In TELSA, full fire suppression (No Fire zone) was achieved by constraining the wildfire rate to 0% of its pre-settlement value for 20 years. This zone approximately overlapped with the area of salt desert shrub and greatest cheatgrass infestation. In the 405-ha (1000-acre English unit was used in Fig. 1 as this is a recognized agency constraint) fire constraint zone, which was small and designated as Greater Sage-grouse habitat, fires were not allowed to exceed a maximum size of 405 ha. We constrained the wildfire activity rate at 10% of its pre-settlement rate. The fire management zone with no constraints indicated that fires were allowed to burn relatively freely assuming adequate fire monitoring and staffing to initiate suppression when necessary. No constraints were applied to the wildfire rate. This zone was mostly at higher elevations where cheatgrass was less common and sagebrush communities still supported the PERENNIAL GRASS DOMINANT state.

2.4.2. Size class distributions of natural disturbances

For each simulation time-step, TELSA determines the total area to be disturbed for each type of disturbance as the sum product of area and probability for each polygon on the landscape. The model then partitions this total area into discrete events based on a predefined size distribution (Kurz et al., 2000). Through a workshop with experts and managers, we identified four size distributions to be used for natural disturbances. We reasoned that weed invasion, pinyon–juniper invasion, wild horse grazing, native herbivore grazing, insect outbreaks, and sagebrush competition were all small scale or localized processes less than 10 ha. Livestock grazing disturbances are assumed to affect patches <1, 1–10, 10–100, and 100–1000 ha in equal proportions. We used a decreasing distribution for wildfire under current management (CURRENT FIRE MANAGEMENT scenario) because we assumed that fire suppression activities were more likely to keep fires small, thus larger fires became increasingly rarer; therefore the spatial distribution was 45% (1 ha), 40% (10 ha), 9% (100 ha), 5% (1000 ha), and 1% (10,000 ha). This is consistent with a Weibull distribution ($p = 0.43$, $c = 0.54$). The size distribution for "unmanaged" wildfire (NATURAL FIRE ONLY scenario) was also Weibull ($p = 0.27$ and $c = 0.7$) with 30% (1 ha), 50% (10 ha), 10% (100 ha), 5% (1000 ha), 4% (10,000 ha), and 1% (100,000 ha) and characterizes lower likelihoods of small fires being put out as suppression is not active in this scenario. Managers argued that replacement fire, which dominates the fire regime of sagebrush shrublands (Miller and Rose, 1999; Young and Sparks, 2002), cannot stop until reaching a natural barrier, a recent burn, or a large change in relative humidity. Therefore, small fires <1 ha burning on large patches of sagebrush should be less common that larger fires. The effect of current fire suppression was to effectively catch fires before they get larger, especially if fire vehicles are driven 60 km to this landscape.

2.4.3. Management scenarios

Management scenarios were developed with BLM staff during a 3-day workshop. Three primary issues were addressed

Table 4 – Six management scenarios for fire management, livestock grazing, vegetation management, and agency funding level

Scenario	Wild fire suppression	Rx fire[a]	Livestock grazing	Wild horse grazing (current AML)	Weed control with herbicide	Mechanical treatment
Natural fire only	0x	0x	0x	1x	0x	0x
Fire management	1x	1x	0x	1x	0x	0x
No vegetation Treatment (normal livestock grazing)	1x	1x	1x	1x	0x	0x
CURRENT COST ($400,000 per year)	1x	1x	1x	1x	1x	1x
2 × COST ($800,000 per year)	1x	2x	1x	1x	2x	2x
8 × COST ($3,200,000 per year)	1x	8x	1x	1x	8x	8x

These scenarios were built in TELSA and are based on possible transitions in VDDT.

[a] Prescribed burning.

with six scenarios (Table 4): fire management, livestock grazing, and level of mechanical and chemical vegetation treatment. The six scenarios were: NATURAL FIRE ONLY, CURRENT FIRE MANAGEMENT, NO VEGETATION TREATMENT (CURRENT LIVESTOCK GRAZING), CURRENT COST, 2 × COST, and 8 × COST. The NATURAL FIRE ONLY scenario represents unmanaged fire regimes and no other management actions applied to the current landscape. The CURRENT FIRE MANAGEMENT scenario is similar to the previous one except that fire is managed within fire management zone constraints and prescribed burning is used. NO VEGETATION TREATMENT (CURRENT LIVESTOCK GRAZING) scenario is simply the CURRENT FIRE MANAGEMENT scenario with the addition of livestock grazing but no mechanical or chemical vegetation treatments. The CURRENT COST scenario was the BLM's $400,000 budget that included fire management, livestock grazing, and mechanical and chemical methods of vegetation management. The CURRENT COST scenario was the baseline scenario with which others were compared (Table 4). We originally simulated 6 additional scenarios to simulate no fire, vegetation management without the use of herbicide, two different seasons of use for livestock grazing (dormant season only and no-year round grazing), aggressive non-native species control, and wild horse herd reduction. These extra scenarios were not presented here to keep the analysis comprehensible and because their effects were small or no new information was gained.

We assigned realistic costs to management activities and then calculated the maximum number of hectares treated based on the partitioning of the BLM's $400,000 annual budget (Table 5). We partitioned the budget by fire suppression zones and by the relative use of the different activities by managers (Table 5). The consequence of the scenario limits is that management disturbances operate at the rate discussed above until the budget is spent, whereas thereafter disturbance rates become null.

2.4.4. Response variables and comparison to baseline

We used the following response variables (aggregations of states within each potential vegetation type) to address these five issues; PERENNIAL GRASS DOMINANT states (including at threshold), SEEDED states, ANNUAL GRASS DOMINANT states, EXOTIC FORB DOMINANT states, and PINYON AND JUNIPER DOMINANT (i.e., invaded) states. The SEEDED state is a polygon covered with artificially seeded species, which may be a mix of native and non-native forage species. Polygons labeled the ANNUAL GRASS DOMINANT and EXOTIC FORB DOMINANT are as described. The PINYON AND JUNIPER DOMINANT state represents a condition of tree encroachment into shrublands, usually >20% canopy cover, that has been sufficiently long to cause the elimination of the native understory. We did not include SHRUB DOMINANT states because most of the landscape was in this condition, therefore the other response variables were better measures of incremental success or lack thereof. The areas of PERENNIAL GRASS DOMINANT states and SEEDED states were especially important to track because all reductions of ANNUAL GRASS DOMINANT states, EXOTIC FORB DOMINANT states, and PINYON AND JUNIPER DOMINANT resulted in increases in SEEDED STATES. This was not the case for the PERENNIAL GRASS DOMINANT states. Because area differences among scenarios for any response variable were small relative to the size of the landscape, we present change as the difference in area between each scenario and the CURRENT COST one, which we consider to be the baseline.

Electronic archives containing detailed information on states within each potential vegetation type, natural disturbance and livestock grazing parameters by state, management parameters by state, and detailed model output (Electronic Archive IV; doi:10.1016/j.ecolmodel.2007.06.030).

3. Results

3.1. *Fire management*

Two scenarios were compared to test increasing fire activity in the absence of other non-fire activities: NATURAL FIRE ONLY and CURRENT FIRE MANAGEMENT (NO LIVESTOCK) (Fig. 3). These two scenarios produced similar results, especially given the size of error bars relative to the mean differences, but the CURRENT FIRE MANAGEMENT (NO LIVESTOCK) scenario had more PERENNIAL GRASS DOMINANT and SEEDED states, and less pinyon–juniper than the NATURAL FIRE ONLY scenario (Fig. 3). The CURRENT FIRE MANAGEMENT (NO LIVESTOCK) and NATURAL FIRE ONLY scenarios, however, contributed some of the largest areas of EXOTIC FORB DOMINANT states of all scenarios compared to the CURRENT COST scenario.

Although cover values were close between the CURRENT FIRE MANAGEMENT (NO LIVESTOCK) and NATURAL FIRE ONLY scenarios, the distribution of wildfire probability (how fre-

Table 5 – Maximum limits of hectares treated with each restoration method per year across all potential natural vegetation types under the CURRENT COST scenario

Fire suppression zone	Herbicide application (ha)[a]	Herbicide application and seeding (ha)	Herbicide application for exotic forbs and seeding (ha)	Mechanical treatment (ha)	Mechanical treatment and seeding (ha)	Prescribed fire (ha)
Few constraints	36.8	4.63	1.15	110.41	7.9	552.07
405-ha fire[b]	0	0	0	0	0	0
Full suppression	3.7	45.79	4.6	27.27	79.0	55.21

Once these limits are reached, the probability of the action becomes zero. BLM managers determined the proportion of effort per management activity allocated to each fire suppression zone. Numbers were obtained by limiting restoration cost per year to \$400,000 and calculating treatment limits by assigning current cost per hectare to each treatment. To obtain the area treated for other scenarios, multiply according to Table 4.

[a] Total area calculated was partitioned according to the proportion of management effort based on the following equations: (a) \$400,000 = \$358.15 × HERBICIDE APPLICATION and SEEDING + \$1790.75 × MECHANICAL TREATMENT and SEEDING + \$1482 × MECHANICAL TREATMENT + \$49.4 × HERBICIDE APPLICATION + \$22.3 × PRESCRIBED FIRE; (b) HERBICIDE APPLICATION and SEEDING = 3 × MECHANICAL TREATMENT and SEEDING; (c) MECHANICAL TREATMENT = 3 × HERBICIDE APPLICATION; (d) PRESCRIBED FIRE = 5 × MECHANICAL TREATMENT; (e) HERBICIDE APPLICATION = 15 × HERBICIDE APPLICATION; (f) PRESCRIBED FIRE = 607.3 ha; (g) HERBICIDE APPLICATION FOR EXOTIC FORBS and SEEDING = HERBICIDE APPLICATION and SEEDING/5.

[b] This fire suppression zone was so small that it was left unmanaged.

quently wildfire burned an area) differed between the two fire scenarios (Fig. 4). With natural fire, wildfire had a tendency to return to the same areas (higher elevations) more frequently (Fig. 4A; darker shades) than for managed fires (Fig. 4B). A greater abundance of low-probability areas was found for managed fires than for natural fires, indicating the role of prescribed fire in capturing areas less likely to burn repeatedly on their own. It is also noteworthy that simulated managed fire activity was more frequent during 20 years (Fig. 4B), although overlapping, than reported fire starts from 1986 to 2003 (Fig. 4C; 17 years).

3.2. Livestock grazing

We compared no livestock grazing (CURRENT FIRE MANAGEMENT [NO LIVESTOCK]) to current livestock grazing management (NO VEGETATION TREATMENT [CURRENT LIVESTOCK GRAZING]) in the absence of vegetation management (except fire) (Fig. 3). Livestock grazing reduced the area of PERENNIAL GRASS DOMINANT, SEEDED, and EXOTIC FORB DOMINANT states, respectively, by 79.5, 50, 8.2 ha, and increased the area of CHEATGRASS DOMINANT and PINYON–JUNIPER DOMINANT states, respectively, by 9.4 and 150 ha.

Fig. 5 shows not only where grazing occurred but its intensity by season of use. The vast majority of summer grazing happened at higher elevations although cattle and sheep were capable of grazing black and Wyoming big sagebrush types at lower elevations during this season. Moreover, intensity of use was high and widespread as indicated by darker shades (Fig. 5). This was somewhat expected given the assumed high utilization from sheep grazing added to that of cattle.

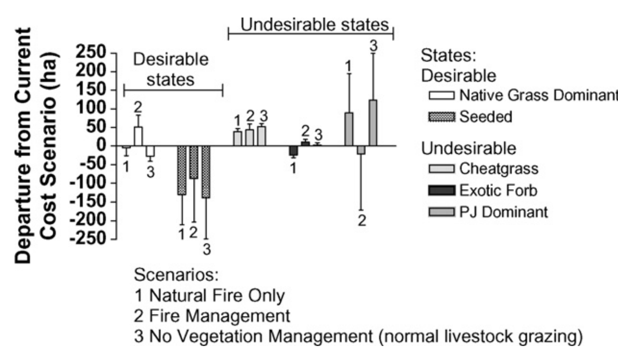

Fig. 3 – Area difference (ha ± 1 S.E., $n = 6$) between each of three TELSA management scenarios (Table 4 and the CURRENT COST scenario, which is represented by the *x*-axis) to test the effects of fire management, livestock grazing, and vegetation treatment on the PERENNIAL GRASS DOMINANT, SEEDED, CHEATGRASS INVADED, EXOTIC FORB INVADED, PINYON–JUNIPER DOMINANT states. Scenarios (codes) are NATURAL FIRE ONLY (1), FIRE MANAGEMENT (2), and NO VEGETATION MANAGEMENT (normal livestock grazing) (3). Desirable states are PERENNIAL GRASS DOMINANT and SEEDED. Undesirable states are CHEATGRASS INVADED, EXOTIC FORB INVADED, AND PINYON–JUNIPER DOMINANT. Error bars are the joint standard errors calculated from the difference between the means of two scenarios (three Monte-Carlo replicates each).

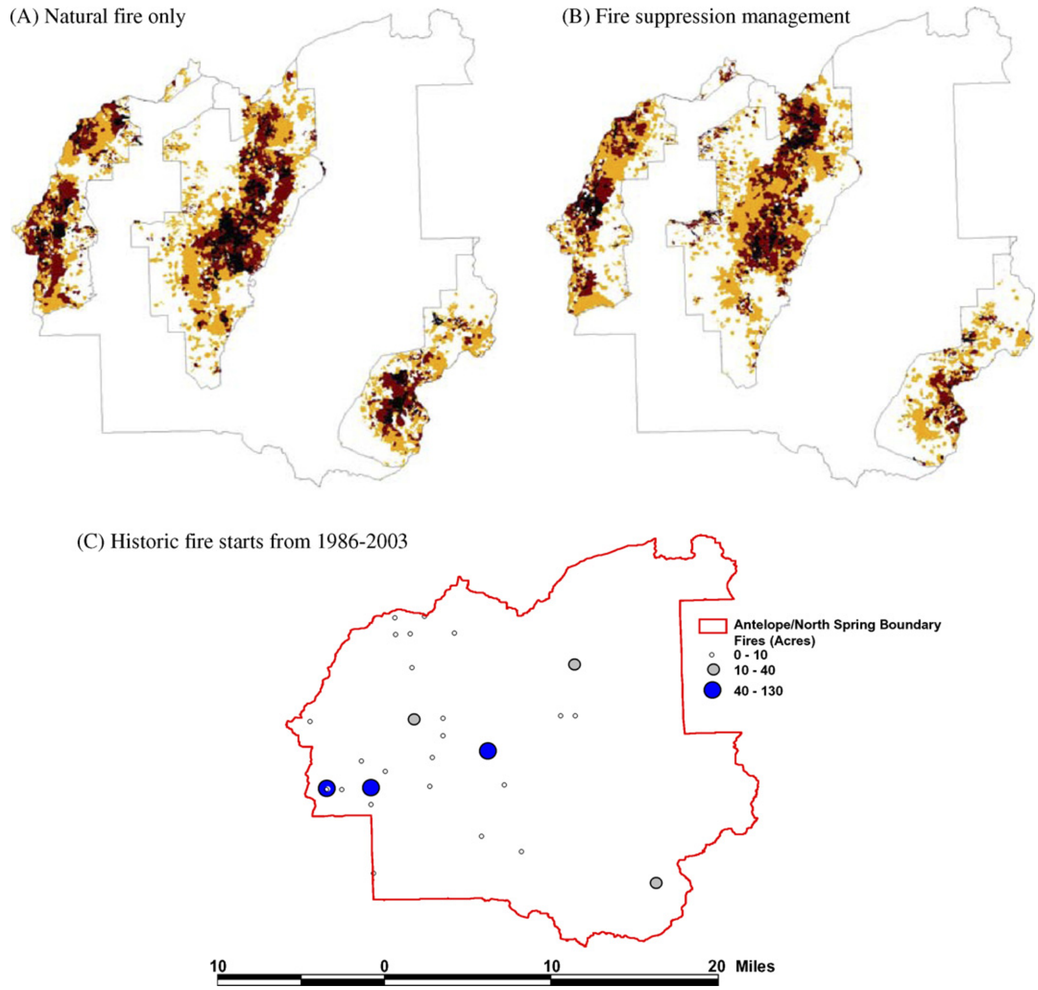

Fig. 4 – Fire activity in the ANSV landscape. Wildfire disturbance frequency for the (A) NATURAL FIRE ONLY SCENARIO and (B) FIRE SUPPRESSION ONLY scenario during a 20-year period, and (C) historic fire starts from 1986 to 2003. Darker areas indicate a greater probability of wildfire activity in (A) and (B) (black represents a probability of 0.8–1.0, whereas white is a probability of 0). Fire suppression zones are easily distinguished.

3.3. *Vegetation management*

We compared the NO VEGETATION TREATMENT (CURRENT LIVESTOCK GRAZING) scenario to the CURRENT COST scenario (baseline value in Fig. 3) to detect the effect of increased mechanical treatment and herbicide application (Fig. 3). The CURRENT COST scenario had the most desirable results compared with the NO VEGETATION TREATMENT (CURRENT LIVESTOCK GRAZING) scenario with 27 ha more area of PERENNIAL GRASS DOMINANT states, 137.6 ha more of SEEDED states, 2.6 ha less area dominated by exotic species, and 123.7 ha less area of pinyon–juniper encroachment. Because removal of pinyon and juniper encroachment was the main contributor to area changed, this implies that mechanical methods also contributed most to restoration activities.

3.4. *Funding level*

We multiplied funding for prescribed fire, herbicide, mechanical operations, and seeding from the CURRENT COST baseline by 2 ($2 \times$ COST scenario) and 8 times ($8 \times$ COST scenario) (Table 4). Higher funding always helped treat more area (Fig. 6). The $4 \times$ increase in funding from $2 \times$ COST to $8 \times$ COST scenarios caused a disproportionately greater change in all response variables for undesirable cover types (Fig. 6B) than the $2 \times$ increase in funding. The area of SEEDED states is a good indicator of budget effects: indeed, we would predict a 300 ha increase of SEEDED area from the $2 \times$ COST to $8 \times$ COST scenarios based on the 75 ha more of SEEDED states with a doubling of the CURRENT COST budget. The observed increase in SEEDED states was 792.4 ha, which was primarily due to an approximately 5 km^2 reduction in PINYON–JUNIPER DOMINANT states. The only improvement that was less than proportional was for the area PERENNIAL GRASS DOMINANT states. Fig. 6A revealed that greater funding caused diminishing returns on PERENNIAL GRASS DOMINANT states because the available area for treatment was exhausted twice in 20 years. In reality only a quadrupling of funding is needed to maximize the area treated over the 20-year period with greater investments early on.

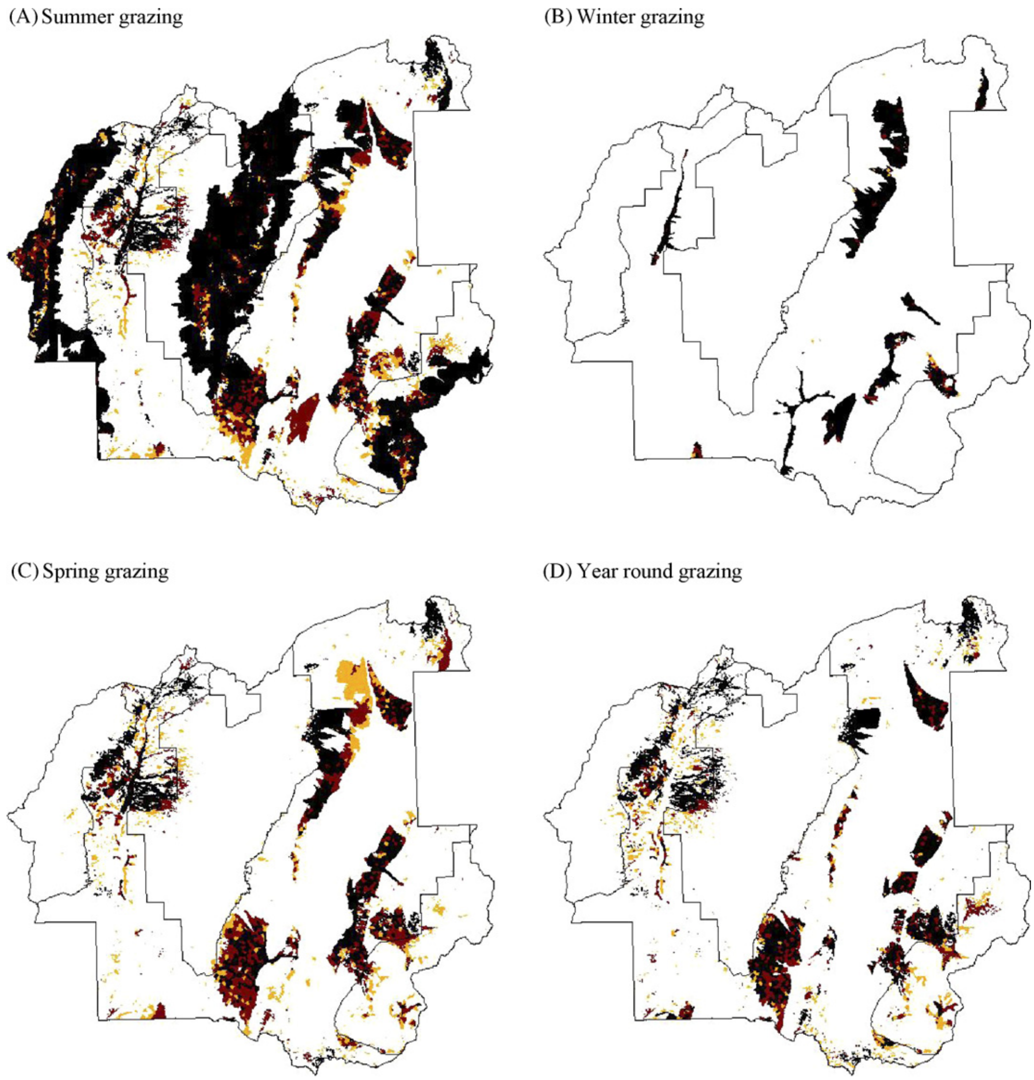

Fig. 5 – Disturbance probabilities for (A) summer, (B) winter, (C) spring (boot stage), and (D) year round grazing by cattle and sheep under the LOW COST scenario during a 20-year period. Darker areas indicate a greater probability of grazing activity (black represents a probability of 0.8–1.0, whereas white is a probability of 0).

4. Discussion

Simulations showed that various treatments and funding levels had at best small effects relative to the spatial extent of degraded cover types in the landscape. In many cases, however, the absolute area changed by management actions was impressive given BLM's recent experience with implementation. At the extremes, a landscape with unmanaged fire regimes resulted in the worst possible outcomes, whereas a budget that afforded a high level of management (8 × normal budget), which involved chemical applications, mechanical thinning, and prescribed burning, produced the most desirable outcomes (Fig. 3).

4.1. *Funding level*

A clear message of these simulations is that the current annual level of restoration funding ($400,000) was so low as to affect no substantial improvement for a period of 20 years in a landscape of 141,853 ha. This conclusion was somewhat expected in light of imposed activity limits from Table 5, the successional duration of many states (often >20 years), and the results of previous, non-spatial models for a larger project area in Eastern Nevada (Forbis et al., 2006). For example, the CURRENT COST scenario allowed for a maximum of 60 ha of herbicide application followed by seeding per year, which amounted to 1200 ha maximum over 20 years. Simulations of three broad management scenarios for the Interior Columbia River Basin revealed similar funding constraints for rangelands (Hemstrom et al., 2001; Wisdom et al., 2002). Keane et al. (1996) showed that only 3% of the modeled landscape improved over a period of 100 years using VDDT models and the coarse-scale (1 km^2) spatial Interior Columbia River Basin Succession Model. They concluded that funding as proposed in the Environmental Impact Statement for the Interior Columbia River Basin, especially for BLM lands, was insufficient to address restoration of landscapes altered by

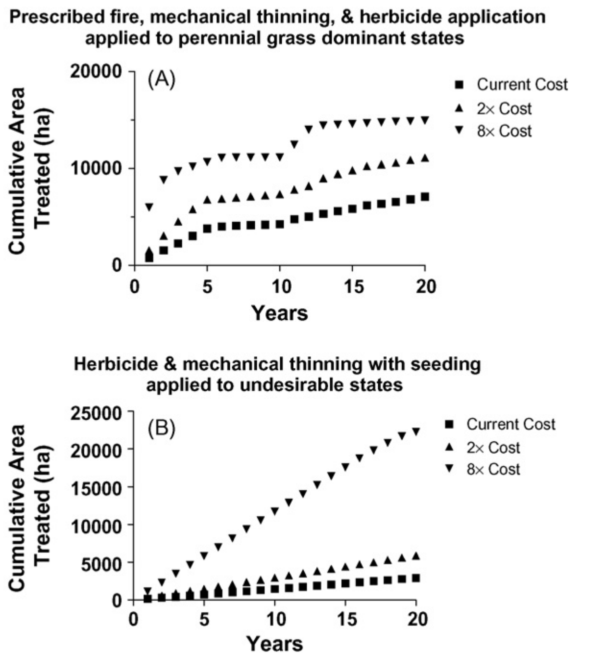

Fig. 6 – Cumulative area of (A) PERENNIAL GRASS DOMINANT states treated (ha) by prescribed fire, mechanical thinning, and herbicide and (B) cumulative area of undesirables cover types treated (ha) by mechanical thinning and herbicide application with seeding for the LOW COST, 2 × COST, AND 8 × COST scenarios. *N* = 3 Monte-Carlo replicates.

exotic invasive species, livestock grazing, and aggressive fire suppression. Greatest improvements were accomplished by concentrating funding on the restoration of whole sub-basins as opposed to a piecemeal approach across many sub-basins (Hemstrom et al., 2001). An important limitation for recovery was the slow dynamics of extensive sub-xeric systems, which were mostly Wyoming big sagebrush steppe (Hemstrom et al., 2001; Wisdom et al., 2002). West and Yorks (2002) noted much slower recovery of Wyoming big sagebrush after fire in the semi-desert of western Utah, which is more similar to eastern Nevada than sagebrush steppe on the Columbia Plateau.

Increasing funding made a difference, but, interestingly, payoffs were disproportionally greater between the 2 × COST and 8 × COST scenarios than between the CURRENT COST and 2 × COST scenarios (Fig. 6). For instance, 2.1 × more area invaded by pinyon–juniper was restored than predicted by quadrupling the 2 × COST budget than by doubling the CURRENT COST budget. The 4 × increase of the budget resulted in a 5 km^2 removal of pinyon–juniper and creation of SEEDED areas, which would be large by current standards. Increased funding would allow larger polygons to be restored. We doubt this result would have been obtained with a-spatial VDDT models alone because it is not based on polygons.

Varying the level of funding revealed a trade-off between the maintenance of resilient states (PERENNIAL GRASS DOMINANT states; Fig. 6A) and chipping away at an abundance of non-resilient states that had crossed at least one ecological threshold (i.e., loss of the perennial grass understory, establishment of cheatgrass, or tree establishment; Fig. 6B). Generally, it is more economical to maintain or restore ecological systems while they are releasable, i.e., have a perennial grass understory that does not require expensive and scarce native plant seeding. Our results support the findings of Tausch (1999) and McIver and Starr (2001). Moreover, sagebrush steppe and semi-desert that maintain a viable and native understory of perennial grass are more likely to resist cheatgrass invasion (Anderson and Inouye, 2001; West and Yorks, 2002), which is an additional benefit to maintaining land in PERENNIAL GRASS DOMINANT states. Simulations showed that the PERENNIAL GRASS DOMINANT states, albeit limited in quantity in the degraded ANSV landscape, can be maintained with prescribed fire, mechanical thinning, and herbicide application drawing from funding not much larger than the CURRENT COST budget (Fig. 6A). More funding captured larger polygons, but the return on the investment decreased. This is in contrast with the restoration of non-resilient states; indeed, there was no shortage of areas to treat and no end to monotonically increasing costs (Fig. 6B).

Given the financial limitation of the Ely BLM's budget and the large need for restoration, identifying the treatments that achieved the most efficient improvements in the degraded ANSV landscape was important. Mechanical and chemical vegetation management and fire management appeared to provide the highest return on the investment, whereas livestock management generally reversed the benefits of either fire management or mechanical and chemical methods (Fig. 3; the x-axis represents the CURRENT COST scenario, which includes mechanical and chemical methods).

4.2. *Vegetation management and seeding*

Although not shown here explicitly, the increase in SEEDED states and reduction of PINYON–JUNIPER DOMINANT states suggested that the mechanical methods accomplished more area treated than herbicide application within the CURRENT COST budget. Mechanical operations followed by seeding were primarily used to restore SHRUB DOMINANT (data not shown) and TREE DOMINANT states, which were the most common states in the ANSV landscape. Mechanical methods, however, also represented the most expensive management action. Therefore, funding most limited this method. Tausch and Tueller (1995) showed that the success of native plant recovery increased and need for non-native plant seeding decreased with decreased encroachment of pinyon and juniper. In central New Mexico pinyon–juniper savannas still maintaining an understory, Broackway et al. (2002) showed that mechanical tree thinning increased native understory biomass by 200%.

4.3. *Fire effects*

It was appropriate to ask whether letting fires follow their natural course over potentially large areas could help advance restoration on the cheap. The overall differences between unmanaged fire regimes (NATURAL FIRE ONLY) and current fire management (CURRENT FIRE MANAGEMENT) scenarios were small (Fig. 3). In terms of most response variables, except the EXOTIC FORB DOMINANT states, the CURRENT FIRE MANAGEMENT had marginally more desirable outcomes.

Combining both wildland fire use and prescribed burning, which are not exclusive, might be the most beneficial approach for fire management, but one should not expect large differences over 20 years. Model results indicate that wildfires simply reburned areas already in the PERENNIAL GRASS DOMINANT states, with new fires seldom venturing into less ignitable states (Fig. 4A). However, prescribed fire spreads out fire activity (Fig. 4B).

4.4. *Livestock grazing*

Overall, livestock grazing had small effects, but these effects consistently counteracted the beneficial effects of fire management and vegetation management (Fig. 3). We expected stronger effects given the heated debate on the effects of livestock grazing on public lands (Fleischner, 1994; Brussard et al., 1994; Wuerthner and Matteson, 2002; Freilich et al., 2003). Increasingly it is recognized that simply removing livestock from degraded xeric rangelands and grasslands will not result in restoration to reference conditions (Curtin, 2002); indeed, our results showed small increases of resilient states and small decreases of non-resilient states with 20 years of livestock removal (Fig. 3). ANSV is a highly degraded, shrub-dominated, exotic species-dominated, and tree-dominated landscape with its best herbaceous forage (PERENNIAL GRASS DOMINANT states) at higher elevations (i.e., fire zone with no constraints; Fig. 1). This herbaceous forage was only available during the summer. Summer grazing is usually less harmful to grasses than spring grazing (Harniss and Wright, 1982). Therefore, livestock grazing at higher elevations would have moderate effects resulting in mildly accelerated woody succession (Table 4). On the other hand, sheep browsing on woody species during the dormant or year-round season of use would partly reverse this trend, which was weak in any case. Thus, either a very small portion of the ANSV experienced cattle grazing or the effects of grazing were moderate when they occurred.

Detrimental effects of grazing will increase with drought, which has been common in recent years. Moreover, livestock will remove the fine fuels needed to carry wildfires early during the dormant season, thus altering fire regimes (Fleischner, 1994; McPherson and Weltzin, 2000). For landscapes with a higher proportion of PERENNIAL GRASS DOMINANT states at lower elevations, conclusions reached here about livestock grazing do not apply because cattle grazing becomes a more prominent component of the total effect that accelerates woody succession. Also, lower elevation ecological communities are less forgiving of disturbances than the higher elevations where greater moisture supports faster recovery (West and Yorks, 2002; Hemstrom et al., 2001). Grazing would be expected to have different effects in riparian systems, which were not included in our models.

4.5. *Limitations of simulations*

Keane et al. (2002) examined factors that affect spatial simulation results in which fire is the dominant process. They determined that the size of a landscape is important; smaller landscapes (e.g., <5000 ha) underestimate fire activity because fires cannot immigrate into the landscape. At 141,853 ha and delineated by surrounding watershed ridges, we believed that we minimized this type of error. Keane et al. (2002) also noted that modeled fire spread that is not influenced by fuel moisture, wind, relative humidity, and topography, but only constrained by the availability of burnable polygons (e.g., TELSA) will simulate too much fire activity. Maps of past fire activity during 17 years for this area (Fig. 4C) indicated less fire than we obtained, however this period of activity also corresponded with aggressive fire suppression everywhere which is not the case since 2001 for all the higher elevations.

Models included assumptions that could change outcomes of a revised version of the simulations. We assumed that the success of reseeding degraded systems with native plant material was perfect (no failure). Although this may be adequate for intensive restoration activities, it did not reflect success rates for wild fire rehabilitation areas or for more xeric potential vegetation types (e.g., black sagebrush and salt desert communities). More realistic success rates, dependent on vegetation type and ranging from 30% in more xeric types to 70% in less xeric types was adopted by Forbis et al. (2006); the remaining proportion of pixels transitioned to CHEATGRASS DOMINANT states under the assumption of failure. This potential source of error was minimized in our simulations because these conditions applied mostly to the full fire suppression zone of the landscape (i.e., no fire zone). Imposing complete suppression to this fire zone was another assumption that could be relaxed; we believe that lower elevation sagebrush and salt desert shrubs with a cheatgrass understory could catastrophically burn under special conditions (i.e., several wet years followed by a dry or average year; Miller and Rose, 1999).

4.6. *Management implications*

In the next decades, the Ely BLM will be examining its 61 watersheds through the Watershed Analysis process. The outcome of this process will be proposed localized restoration actions. Simulations revealed important lessons. BLM's funding was too low to improve the ecological condition of its degraded rangelands and woodlands. The current funding (if applied all to one watershed), however, could maintain ecological PERENNIAL GRASS DOMINANT states where they occur, thus preventing further loss of desirable habitat. Fire, either prescribed or unmanaged, will always be more beneficial than no fire in these communities (results of NO FIRE scenario not shown). For altered rangelands requiring more expensive management, success will be more likely if the BLM (1) concentrates restoration activities to achieve disproportionate increases of desirable cover types, (2) emphasizes the use of mechanical methods and herbicide application followed by seeding, and possibly (3) increases the wildfire use in resilient vegetation types and states. As more expensive management actions are implemented, land managers should ensure that livestock management practices minimize detrimental impacts of stocking or seasons of use (Budd, 1999, 2000) while the species composition of potential vegetation types improves.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at doi:10.1016/j.ecolmodel.2007.06.030.

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