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Joel M. Diamond  
*Utah State University*

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EFFECTS OF TARGETED GRAZING AND PRESCRIBED BURNING ON FIRE  
BEHAVIOR AND COMMUNITY DYNAMICS OF A CHEATGRASS  
(*BROMUS TECTORUM*)-DOMINATED

LANDSCAPE

by

Joel M. Diamond

A dissertation submitted in partial fulfillment  
of the requirements for the degree

of

DOCTOR OF PHILOSOPHY

in

Ecology

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Logan, Utah

2009

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## ABSTRACT

Effects of Targeted Grazing and Prescribed Burning on Fire Behavior and Community  
Dynamics of a Cheatgrass (*Bromus tectorum*)-Dominated  
Landscape

by

Joel M. Diamond, Doctor of Philosophy

Utah State University, 2009

Major Professor: Christopher A. Call  
Department: Wildland Resources

Studies were conducted to determine the effectiveness of using targeted grazing and prescribed burning as tools to reduce fire hazards and cheatgrass (*Bromus tectorum*) dominance on rangelands in the northern Great Basin. A field study, with four grazing-burning treatments (graze and no-burn, graze and burn, no-graze and burn, and no-graze and no-burn), was conducted on a *B. tectorum*-dominated site near McDermitt, Nevada from 2005-2007. Cattle removed 80-90% of standing biomass in grazed plots in May 2005 and 2006 when *B. tectorum* was in the boot (phenological) stage. Grazed and ungrazed plots were burned in October 2005 and 2006. Targeted grazing in May 2005 reduced *B. tectorum* biomass and cover, which resulted in reductions in flame length and rate of spread when plots were burned in October 2005. When grazing treatments were repeated on the same plots in May 2006, *B. tectorum* biomass and cover were reduced to the point that fires did not carry in grazed plots in October 2006. Fuel characteristics of

the October 2005 burns were used to parameterize dry climate grass models in BehavePlus 3.0, and simulation modeling indicated that grazing in spring (May) would reduce the potential for catastrophic fires during the peak fire season (July-August). The graze-and-burn treatment was more effective than grazing alone (graze and no-burn treatment) and burning alone (no-graze and burn treatment) in reducing *B. tectorum* cover, biomass, plant density, and seed density, and in shifting species composition from a community dominated by *B. tectorum* to one composed of a suite of species [including tumble mustard (*Sisymbrium altissimum*), clasping pepperweed (*Lepidium perfoliatum*), and Sandberg bluegrass (*Poa secunda*)], with *B. tectorum* as a component rather than a dominant.

A simulation study was designed to compare the cost-effectiveness of using cattle grazing and herbicide to create fuel breaks on *B. tectorum*-dominated landscapes in the northern Great Basin. Fuel characteristics from this targeted grazing study and from a Plateau<sup>®</sup> (Imazapic) herbicide study near Kuna, Idaho were used to parameterize fire behavior models and simulate flame lengths and rates of spread for the two fuel reduction treatments under peak fire conditions using BEHAVE Plus. Targeted grazing and Plateau<sup>®</sup> had similar reductions in flame length and rate of spread. Cattle grazing had high fixed costs (primarily fencing), and was more cost-effective than applications of Plateau<sup>®</sup> under five fuel loading scenarios except for three consecutive years of low fuel loads.

## DEDICATION

To my little brother who pursued a different form of education.

June 20, 1979 – September 27, 2005

## ACKNOWLEDGMENTS

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I give special thanks to my wife, Gabrielle F. Diamond, for her unending support, both morally and physically, and her valuable service in the field, greenhouse, and laboratory.

I could not have done this without all of you.

Joel M. Diamond

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## CHAPTER 1

### INTRODUCTION

*Bromus tectorum* (cheatgrass) is an invasive annual grass that has altered the fire behavior and community dynamics of several vegetation types in the Great Basin (Stewart and Young 1939; Billings 1994; Knapp 1996; Knick and Rotenberry 1997). The invasion of this species has set in motion a grass/fire cycle where *B. tectorum* provides the fine fuel necessary for the initiation and propagation of fire (D'Antonio and Vitousek 1992). After burning, *B. tectorum* recovers more rapidly than most native species and often dominates plant communities, promoting more frequent, larger fires (Peters and Bunting 1994; Knick and Rotenberry 1997; Whisenant 1990). Wildfire suppression and rehabilitation costs have increased dramatically with the increasing size and number of fires (Roberts 1994). Fuel reduction treatments are being promoted as a proactive approach for altering the fire disturbance regime and reducing fire suppression costs (Pellant and Hall 1994; Knapp 1996, USDI and others 2001).

### FIRE BEHAVIOR

Fire has long been recognized as one of the dominant disturbances affecting grass and shrubland plant communities (Gleason 1926; Hobbs and Huenneke 1992). Sagebrush-grassland communities are adapted to relatively long fire return intervals (50-100 years), which allow for a mosaic of woody plant-dominated communities interspersed with herbaceous communities dominated by perennial bunchgrasses and

forbs (Harniss and Murray 1973; Yensen 1981; Wright and Bailey 1982). Fire is typically a rarity in salt-desert shrub communities, which will only have sufficient fuel following seasons of abnormally high precipitation events (Wright and Bailey 1982). However, during the last several decades these plant communities have declined precipitously, due primarily to *B. tectorum* invasion and a changing fire regime (Miller et al. 1994; Pellant and Hall 1994; Rice 2005). In 2003, an estimated 22.6 million ha of western rangelands were infested with *B. tectorum* (Rice 2005).

Sagebrush-grasslands in the northern Great Basin not only burn at much lower frequencies, but fires also behave differently than in *B. tectorum*-dominated communities (Fleming et al. 1942; Whisenant 1990; D'Antonio and Vitousek 1992). A longer active growing season for native bunchgrasses, forbs and shrubs can result in higher fuel moisture contents which moderate fire behavior in sagebrush-grasslands during peak fire season (Fleming et al. 1942). In contrast, *B. tectorum* fuel moisture decreases rapidly within 1-2 weeks after active growth ends in June (Platt and Jackman 1946; Billings 1952). *B. tectorum* matures quickly and produces fine stems and pedicels along with soft, low fiber tissues, resulting in more thorough curing, compared to native bunchgrasses (Stewart and Hull 1949). By peak fire season (July-August), *B. tectorum* is fully cured and simply awaits an ignition source (McAdoo et al. 2007).

*B. tectorum* fuel loads can accumulate to the point where they support catastrophic fires within a few years (Pyne et al. 1996). In the northern Great Basin, fuel loading for *B. tectorum* averages 500-600 kg ha<sup>-1</sup>, and can range from 30 kg ha<sup>-1</sup> to >1,500kg ha<sup>-1</sup>, depending on precipitation (Uresk et al. 1979). Given that *B.*

*tectorum*-dominated communities are more likely to burn than native plant communities, such biomass results in high flame lengths and rapid rates of spread for wildland fires (Platt and Jackman 1946; Pyne et al. 1996). These fires are difficult and expensive to control (Stewart and Hull 1949; Knapp 1996), they lead to further invasion of *B. tectorum*, and thus promote more fires of the same character (Bunting et al. 1987). The continued dominance of *B. tectorum* affects not only the fire behavior but also the plant species capable of occupying a site.

## COMMUNITY DYNAMICS

A successional management model can be used as a framework to understand the plant strategies and ecological processes influencing community dynamics on *B. tectorum*-dominated landscapes. One model, proposed by Pickett et al. (1987) and applied to rangeland management by Sheley et al. (1996, 2006), identifies three causes of succession (site availability, species availability and species performance), ecological processes and components primarily responsible for controlling the causes of succession, and the factors that modify those processes and components (Table 1.1). For *B. tectorum* invasion and expansion to occur, safe sites must be present, propagules (seeds) need to occupy those sites, and plants must perform successfully in the new sites (Krueger-Mangold et al. 2006). Knowledge about the causes of succession and the associated ecological processes, components and modifying factors can help in identifying methods for reducing *B. tectorum* dominance and creating more desirable plant communities.



### *Site Availability*

Site availability for *B. tectorum* in sagebrush-grasslands and other communities depends on the size, severity, frequency and patchiness of disturbances, and the predisturbance history of these communities (Table 1.1). The introduction and expansion of *B. tectorum* in the northern Great Basin are closely tied to historical grazing and dryland farming practices and to the development of transportation routes since the late 1800's (Costello 1944; Piemeisel 1951; Miller et al. 1994). Season-long, heavy grazing by livestock suppressed native perennial species, allowing *B. tectorum* to occupy open sites in the understory, and dryland farms replaced native species with grain crops that later provided open areas for *B. tectorum* colonization after they were abandoned (Pickford 1932; Spilsbury and Tisdale 1944; Billings 1952; Miller et al. 1994). Transportation routes, particularly railroad corridors, disturbed large areas that were colonized by a variety of weedy species, including *B. tectorum* (Billings 1952; Miller et al. 1994).

Once occupied by *B. tectorum*, areas become more prone to fire disturbance. Fire size is primarily limited by the continuity of the fuel (Pyne et al. 1996), and *B. tectorum* often provides a continuous fuel bed that allows for large fires (Whisenant 1990). The severity of rangeland fires increases markedly in *B. tectorum*-dominated communities due to fuel loads up to five times higher than in native sagebrush-grasslands (Paysen et al. 2000). The high fuel loads and fuel continuity provided by *B. tectorum* reduce the patchiness of plant communities on rangelands, because fire now readily spreads into communities with even a component of *B. tectorum* (Whisenant 1990). As noted earlier, the disturbance interval for native sagebrush-

bunchgrass communities may be as long as 100 years (Paysen et al. 2000). With the introduction of *B. tectorum*, those return intervals decrease to 2-15 years (Whisenant 1990). This change in historical disturbance patterns can alter the landscape from one containing diverse successional stages of native sagebrush-grassland to near monocultures of *B. tectorum* (Pickford 1932; Peters and Bunting 1994).

Safe sites for the germination and establishment of *B. tectorum* and other species can be altered by livestock grazing and fire disturbances. These safe sites include cracks and depressions and litter beds that moderate temperature and moisture conditions on the soil surface (Evans and Young 1970, 1972; Facelli and Pickett 1991). Grazing removes standing biomass that contributes to litter buildup and hoof action can reduce litter bed depth (Thurow 1991). Fire reduces standing biomass and litter to ash, and exposes bare soil (Wright and Bailey 1982). Without litter accumulation after a fire, the reoccupation of burned sites by *B. tectorum* is slowed (Evans and Young 1984). Annual forbs, such as *Sisymbrium altissimum* (tumble mustard) and *Lepidium perfoliatum* (clasping pepperweed), are capable of germinating on bare soil due to a mucilaginous seed coat (Young et al. 1970). These two species provide the litter bed necessary for safe site maintenance, which allows for *B. tectorum* germination and establishment (Evans and Young 1970; Evans et al. 1975).

### *Species Availability*

Species availability on *B. tectorum*-dominated landscapes is influenced by dispersal mechanisms, landscape features and propagule pool dynamics (Table 1.1).

*B. tectorum* seeds generally disperse from late-May to late-June in the northern Great Basin, depending on moisture conditions (Mack and Pyke 1983). The majority (~90%) of seeds disperse a short distance (<1 m) from the parent plant by dropping to the soil surface (Phase I dispersal) and moving along the surface via wind (Phase II dispersal) (Mack and Pyke 1983; Chambers and MacMahon 1994). Short rigid hairs on the awns and lemmas facilitate long distance dispersal by attaching to animal fur and human clothing (Pyke and Novak 1994; Chambers and MacMahon 1994). Seed caching rodents may also play a role in the long distance dispersal of *B. tectorum* (La Tourette et al. 1971). The landscape features influencing seed dispersal include soil surface characteristics (Chambers 2000) and the height, density, structural attributes and composition of vegetation (Davies and Sheley 2007). Chambers (2000) modified soil surface characteristics in a sagebrush steppe community and observed greater seed entrapment and retention, and subsequent seedling emergence and survival in large depressions (50 cm wide X 10 cm deep) than in smaller depressions. Davies and Sheley (2007) found that wind dispersal of *Taeniatherum caput-medusa* (medusahead) and *Tragopogon dubius* (yellow salsify) seed was reduced as the height of neighboring *Agropyron desertorum* (crested wheatgrass) plants increased from 10-60 cm.

The proportions of desirable and undesirable species in propagule pools are influenced by land use, disturbance interval, and species life history traits (Table 1.1). As season-long, heavy grazing by livestock progressed from the late 1800's to the early 1930's, weedy species, especially annuals like *B. tectorum*, replaced perennial herbaceous species aboveground and in the propagule pool in the soil (Mack 1981;

Mack and Pyke 1984). Since the 1930's, grazing management programs on *B. tectorum*-infested rangelands have relied on deferred-rotation and rest-rotation systems to promote vegetative growth and seed production of remaining desirable perennial herbaceous species; however, when grazing pressure is relaxed in these systems, it also allows *B. tectorum* to produce seeds and maintain its dominance in the propagule pool (Daubenmire 1940; Young and Allen 1997). *B. tectorum* seed input into the propagule pool can be reduced when plants are intensively grazed (Vallentine and Stevens 1994; Mosley and Roselle 2006). Fires occurring before seed shatter can also reduce *B. tectorum* seed input into the propagule pool; however, most wildfires occur later in the growing season after seeds have dispersed into litter and soil seed banks (Rasmussen 1994). *B. tectorum* perpetuates itself in frequently disturbed environments (i.e., 2-15 year fire return intervals) by producing large amounts of seed and forming a type III seed bank with a mix of transient and persistent seeds (Thompson and Grime 1979; Pyke 1994). After summer dispersal, most seeds germinate the following fall through spring; however, seeds can remain viable in the soil for 2 to 5 years (Burgert et al. 1971; Wick et al. 1971; Smith et al. 2008). In a recent study in the West Desert of Utah, the majority of the seed bank (96%) germinated in the first year; by year two only 3.6% of the initial seed bank germinated and by year three 0.4% of the seed bank remained, without any additional input (Smith et al. 2008). Seed bank densities of 2,400-8,300 seeds m<sup>-2</sup> have been reported for over-grazed sagebrush communities (Young and Evans 1975), and 4,800-19,000 seeds m<sup>-2</sup> for unburned, *B. tectorum*-dominated communities in the northern Great Basin (Humphrey and Schupp 2001; Hempy-Mayer and Pyke 2008).

Immediately following fire, *B. tectorum* seed density can be reduced by as much as 97%; however, it can return to pre-burn levels within 2 years (Humphrey and Schupp 2001). Even in a late-seral sagebrush-grassland community, *B. tectorum* made up only 3% of the aboveground current year's growth, yet it made up 46% of the viable seed pool (Hassan and West 1986). Thus, the introduction of desirable species is necessary, often at high seeding rates, to compete with the large numbers of *B. tectorum* and other weed seeds when trying to revegetate burned and unburned sites (Kreuger-Mangold et al. 2006)

### *Species Performance*

The performance of an invasive species, such as *B. tectorum*, depends upon resource supply, ecophysiological and life history traits, stress and interference (Table 1.1). Resource supply is modified by soil topography, climate, site history, and microbes. *B. tectorum* will grow on almost any type of soil, but it does best on deep, loamy or coarse textured soils in sagebrush steppe communities (Klemmedson and Smith 1964; Young 2000). It can grow on calcareous and saline soils, but is not very tolerant of acidic soils (Billings 1952). *B. tectorum* can be competitive on low-fertility soils, including those where the A horizon has been eroded away (Klemmedson and Smith 1964; Young 2000). It often thrives and dominates, however, under conditions of increased nitrogen availability after disturbance (D'Antonio 2000) *B. tectorum* produces more tillers and allocates more nitrogen to shoots and roots when supplied with nitrate compared with nitrite (Monaco et al. 2003). The forms and amounts of nitrogen available for *B. tectorum* are strongly influenced by soil bacteria (Young et

al. 1995). Roots are also colonized by mycorrhizal fungi, which facilitate the uptake of nutrients and water (Goodwin 1992). Topography can play a role in *B. tectorum* performance; it tends to be more invasive on southern and western aspects than on northern aspects (Goodrich 1999), and at elevations below 2000 m (Hull and Pechanec 1947; Rice 2005). Thus, *B. tectorum* is a dominant or codominant in salt desert shrub, sagebrush-grassland and pinyon-juniper communities where mean annual precipitation ranges from 180-430 mm (Mosley et al. 1999). Its broad distribution reflects its ability to perform on a wide variety of sites, usually following some type of disturbance (previously described in the Site Availability section).

Ecophysiological and life history traits include germination requirements, assimilation rates, growth rates, genetic differences, allocation patterns, and phenological development (Table 1.1). *B. tectorum* persistence on rangelands with highly variable environmental conditions is related to its ability to exist as a winter or spring annual (Mack and Pyke 1983, 1984). After dispersal, most seeds remain dormant through the summer and typically germinate in the fall (and sometimes the winter) with increasing precipitation (Allen and Meyer 2002, Roundy et al. 2007). If fall moisture is limiting, seeds germinate the following spring or enter into a secondary dormancy period and carryover into the next fall germination period (Young et al. 1969; Hull and Hansen 1974; Evans and Young 1975; Mack and Pyke 1984; Meyer et al. 1997; Smith et al. 2008). Fall, winter, and early spring germination generally occurs when temperatures are above 0° C and the soil surface is free of snow. Fructan metabolism (Chatterton 1994) permits early, rapid shoot and root growth at relatively low temperatures (Nasri and Doescher 1995). Water-use

efficiency is also high for early season growth when transpiration rates are low (Hulbert 1955). *B. tectorum* exhibits considerable plasticity in its response to variable site conditions; a plant may produce a single culm < 10 cm tall with only one spikelet when growing in a dense monotypic stand or on a dry, infertile site, while a plant on a moist, fertile site may produce 12-15 culms, up to 75 cm tall, bearing hundreds of spikelets (Mosley et al. 1999).

Plant stress is modified by climate, site history, prior occupants, herbivory and natural enemies (Table 1.1). The roles of climate (in terms of germination, plant growth and seed production), site history (in terms of grazing, farming and fire disturbances), and prior occupants (in terms of native perennial species competing with *B. tectorum*, and introduced annuals facilitating *B. tectorum* establishment) have been described in previous sections and will not be covered here. The natural predators and parasites of *B. tectorum* have for the most part been left behind in Eurasia but rodent herbivory and grainivory, and fungi can impact *B. tectorum* survival (Mack and Pyke 1984). Rodents remove germinated seedlings under snow cover and remove aboveground biomass in spring and fall, but do not affect recruitment significantly. The fungus *Ustilago bullata* can infect seeds at germination through the seedling stage, and plants generally become moribund. The overall effect of these organisms is not a significant cause of *B. tectorum* mortality (Mack and Pyke 1984).

Interference refers to the reduction in fitness of plants due to various factors, including competition, allelopathy, herbivory, resource availability and predators. The effects of herbivory, resource availability and predators on *B. tectorum* performance

have been described in previous sections and will not be covered here. The success of *B. tectorum* seedling establishment and its subsequent competitiveness with many native perennial species can be attributed to its rapid germination and growth, and fast developing root system that is structurally efficient in exploiting soil moisture and nutrients (Harris 1967; Mosley et al. 1999; Rice 2005). *B. tectorum* can also outcompete introduced annual forbs, including *Sasola iberica* (Russian thistle), *L. perfoliatum* and *S. altissimum* (Piemeisel 1951), except after fire when these forbs readily occupy exposed soil surfaces (Young et al. 1976). After litter accumulation facilitates *B. tectorum* establishment, the annual forb species can make up 10% (wet year) to 40% (dry year) of the post fire community (Young et al. 1970). The native bunchgrasses *Elymus elymoides* (squirreltail) and *Poa secunda* (Sandberg bluegrass) can flourish after fire in native sagebrush-bunchgrass communities and remain a component of *B. tectorum*-dominated sites (Hironaka and Tisdale 1963; Wright and Klemmedson 1965). *P. secunda* is capable of reestablishing a burned site by tillering from axillary buds (Antos et al. 1983). Many *B. tectorum*-dominated communities in the northern Great Basin are characterized by the presence of *S. altissimum*, *L. perfoliatum* and *P. secunda*. However, as *B. tectorum* density increases, the density of these three species decreases (Young and Evans 1978). On some sites, *B. tectorum* may be replaced by other highly competitive invasive species such as *Centaurea solstitialis* (yellow starthistle), *C. maculosa* (spotted knapweed), *C. diffusa* (diffuse knapweed), *C. squarrosa* (squarrose knapweed), *Chondrilla juncea* (rush skeletonweed), *Euphorbia esula* (leafy spurge) and *Linaria dalmatica* (dalmatian toadflax) (Sheley and Petroff 1999).



## VEGETATION MANIPULATION

The previously described successional weed management model also provides a framework for identifying strategies and methods to shift the community dynamics of *B. tectorum*-dominated landscapes. In order to alter the successional trajectory of a plant community, we can address the causes of succession with designed disturbances, controlled colonization and controlled species performance (Sheley et al. 1996; Sheley and Krueger-Mangold 2003). Designed disturbances increase site availability for desirable species and/or decrease site availability for undesirable species (Sheley et al. 1996). Controlled colonization is the intentional alteration of availability and establishment of various species, while controlled species performance involves manipulating the relative growth and reproduction of species (Sheley et al. 1996). This framework allows for a more process-based approach to address *B. tectorum*-dominated rangeland management.

Mechanical, fire, herbicide, grazing, and revegetation treatments have all been used to reduce the dominance of *B. tectorum*. Most of these treatments, when used individually, can affect site availability as designed disturbances, and often control colonization (species availability) and/or species performance. When used in combination, they can have an impact on all three causes of succession; however, the level of impact also depends on their severity and frequency (Sheley and Krueger-Mangold 2003). Plowing and disking, used for preparing seedbeds in some revegetation operations (Stevens and Monsen 2004), can bury *B. tectorum* seeds to depths (> 6 cm) that limit site availability for germination and seedling establishment

(Hulbert 1955). However, a portion of the seed pool remains at shallower depths in soil depressions and can still germinate and establish (Hulbert 1955). Mowing has been used to control *B. tectorum* performance, but required retreatment every 3 weeks during the growing season (Ponzetti 1997). Depending on the timing, prescribed fire can limit site availability for *B. tectorum* by consuming litter and exposing bare soil, and limit species availability by killing seeds in the seedheads before they disperse or in litter after they disperse (Evans and Young 1984; Rasmussen 1994). However, the effect can be short lived, because *B. tectorum* plants establishing from seed in protected sites (soil, unburned litter) can produce large quantities of seed to build up the seed pool after fire (Pellant and Hall 1994). A variety of herbicides have been used to control *B. tectorum* performance, primarily in preparation for revegetation. Paraquat, a quick acting contact herbicide, kills emerged *B. tectorum* plants, leaves no soil residues, and allows planting of perennial species immediately after application (Evans et al. 1975). Fall application of Atrazine effectively controls emerging *B. tectorum* seedlings during a chemical fallow period; however, at least 1 year must be allowed for dissipation before seeding desirable species (Eckert and Evans 1967). Plateau<sup>®</sup> has recently been used as a preemergence herbicide to control *B. tectorum* performance for fuel break establishment on fire prone sites (Kury et al. 2002). Plateau<sup>®</sup> has been shown to be effective in suppressing another invasive annual grass, *T. caput-medusae* (medusahead), and opening up a window of opportunity for revegetation with desirable species (Monaco et al. 2005). As previously mentioned, intensive livestock grazing can affect site availability for *B. tectorum* by reducing litter depth and changing soil surface microrelief through hoof action (Thurow 1991).

Intensive grazing at the boot stage (inflorescence emerging from the leaf sheath) can also control *B. tectorum* colonization and performance by reducing seed production and subsequent vegetative growth (Vallentine and Stevens 1994; Mosley and Roselle 2006). The success of revegetation efforts, in terms of the ability of desirable species to colonize a site and interfere with *B. tectorum* and other weedy species depends on the impact of the above-mentioned treatments, the revegetation methods and species used, and most importantly, on environmental conditions.

### ECONOMIC ANALYSIS

The 1995 Federal Wildland Fire Management Policy and Program Review report (updated, USDI and others 2001) encourages a more proactive approach to reduce the threat of catastrophic wildfires on western rangelands. The report states that strategic landscape-scale fuel management will require the integration of a variety of treatment methods (including chemical and biological), and recommends research on, and development of, fuel reduction alternatives. Herbicide treatments can reduce fuel build-up in *B. tectorum*-dominated landscapes, but they can be costly, and they may leave residues in soils and impact adjacent water sources (Vallentine 1989; Wester 1990). Livestock grazing, primarily by sheep, has been used for *B. tectorum* fuel reduction in Nevada (Davison 1996; Smith et al. 2000). Cattle have yet to be utilized in *B. tectorum* fuel reduction projects, and their cost effectiveness compared to other treatments is not known.

Cost-benefit analysis has typically been used to evaluate the effectiveness of fuel reduction treatments on public lands (Kline 2004). A cost-benefit analysis is

dependent on the incremental change in all pertinent costs and benefits using a single metric, generally dollars (Johannsen 1993). It requires enumeration of all costs and benefits of a fuel treatment, such as suppression costs, reseeding costs, forage loss or gain, and air quality impacts and property damage associated with any wildfire (Kline 2004). While a cost-benefit analysis provides a high degree of specificity, it does not allow for inference or generalization. An emerging method for analyzing fuel treatment methods is cost-effectiveness analysis.

Cost-effectiveness analysis evaluates treatment alternatives according to their costs and their effects with regard to producing some outcome or set of outcomes which are not measured solely in terms of dollars (Levin and McEwan 2001). It was initially used to evaluate health care interventions but the methodology can be used for other applications as well. Within the last 10 years, cost-effectiveness analysis has been used to compare the effectiveness of different forest harvest and planting practices, threatened animal species recovery protocols, stream restoration approaches, and forest fuel reduction treatments (Rideout et al. 1999; Cullen et al. 2005; Frimpong et al. 2006; Dampier et al. 2006; Van Landingham et al. 2008). In these cases, cost-effectiveness analysis enabled scientists and managers to select those treatments or approaches which provided the maximum effectiveness per level of cost or which required the least cost per level of effectiveness. Thus, cost-effectiveness analysis is well-suited for comparing the effectiveness of fuel reduction treatments, such as targeted cattle grazing and herbicide application, in reducing wildfire flame lengths and rates of spread on *B. tectorum*-dominated rangelands.

## OBJECTIVES

The goal of this research project was to determine the effectiveness of using cattle and prescribed burning as tools to reduce fire hazards and *B. tectorum* dominance on rangelands in the northern Great Basin. The specific objectives were to: 1) evaluate the effects of targeted cattle grazing and prescribed burning on fire behavior, 2) assess the impact of targeted cattle grazing and prescribed fire on the seed dynamics of *B. tectorum* and associated species, 3) determine the effects of targeted cattle grazing and prescribed fire on aboveground community dynamics, and 4) compare the economic effectiveness of using targeted cattle grazing and herbicide to create fuel breaks. I conducted a series of experiments to address these objectives. Chapter 2 describes the effects of targeted cattle grazing and prescribed burning on actual and modeled fire flame lengths and rates of spread. Chapter 3 details the effects of targeted grazing and prescribed burning on seed input and seed bank density, and aboveground biomass, cover, density, and species composition. Chapter 4 describes a cost-effectiveness analysis comparing the use of targeted cattle grazing and Plateau<sup>®</sup> herbicide to create a fuel break between a *B. tectorum*-dominated community and a remnant sagebrush-grassland community. And, Chapter 5 provides a synthesis of my findings.

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**Table 1.1** Causes of succession, contributing processes and components, and modifying factors (from Sheley et al. 1996, 2006).

Causes of succession	Processes and components	Modifying factors
Site availability	Disturbance	Size, severity, time intervals, patchiness, predisturbance history
Species availability	Dispersal	Dispersal mechanisms and landscape features
	Propagules	Land use, disturbance interval, species life history
Species performance	Resources	Soil, topography, climate, site history, microbes, litter retention
	Ecophysiology	Germination requirements, assimilation rates, growth rates, genetic differentiation
	Life history	Allocation, reproduction timing and degree
	Stress	Climate, site history, previous occupants, herbivory, natural enemies
	Interference	Competition, herbivory, allelopathy, resource availability, predators

## CHAPTER 2

EFFECTS OF TARGETED CATTLE GRAZING ON FIRE BEHAVIOR OF  
CHEATGRASS-DOMINATED RANGELAND IN THE  
NORTHERN GREAT BASIN, USA<sup>1</sup>

*Abstract.* We evaluated the effectiveness of using targeted, or prescribed, cattle grazing to reduce the flame length and rate of spread of fires on cheatgrass (*Bromus tectorum*) dominated rangeland in northern Nevada. Cattle removed 80-90% of *B. tectorum* biomass during the boot (phenological) stage in grazed plots in May 2005. Grazed and ungrazed plots were burned in October 2005 to assess fire behavior characteristics. Targeted grazing reduced *B. tectorum* biomass and cover, which resulted in reductions in flame length and rate of spread. When the grazing treatments were repeated on the same plots in May 2006, *B. tectorum* biomass and cover were reduced to the point that fires did not carry in the grazed plots in October 2006. Fuel characteristics of the 2005 burns were used to parameterize dry climate grass models in BehavePlus 3.0, and simulation modeling indicates that targeted grazing in spring (May) will reduce the potential for catastrophic fires during the peak fire season (July – August) in the northern Great Basin.

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<sup>1</sup> Coauthored by J.M. Diamond, C.A. Call, and N. Devoe.

## Introduction

*Bromus tectorum* (cheatgrass) is an invasive annual grass that originated in Eurasia and is now dominant on many rangelands in the western USA (Mack 1981). Invasion has set in motion a grass/fire cycle where *B. tectorum* provides the fine fuel necessary for the initiation and propagation of fire (D'Antonio and Vitousek 1992). *B. tectorum* recovers more rapidly than native species and facilitates an increase of more frequent and larger fires.

The 1995 Federal Wildland Fire Management Policy and Program Review report (updated, USDI and others 2001) encourages a more proactive approach to reduce the threat of catastrophic wildfires on rangelands in the western USA. The report states that strategic landscape-scale fuel management will require the integration of a variety of treatment methods (fire, chemical, and biological), and recommends research and development on fuel reduction alternatives. Using prescribed fire to reduce fuel loads is effective but can be risky in areas that are dominated by flashy fuels like the annual grass *B. tectorum*. In addition, prescribed fire only has an impact on seed input prior to seed shatter when fuel moistures are still fairly high and thus difficult to burn (Rasmussen 1994; Brooks 2002). Herbicide treatments can reduce fuel build-up in *B. tectorum*-dominated landscapes, but they can be costly and have real or perceived effects on environmental quality (Vallentine 1989). Livestock grazing, primarily by sheep and goats, is recognized as an effective tool for fuel reduction in brush communities in Texas and California (Taylor 1994). And, sheep grazing has been used for *B. tectorum* fuel reduction in Nevada (Davison 1996; Smith *et al.* 2000). However, there is little

information available on the use of cattle as fuel reduction agents on rangelands, although they have been used to suppress *B. tectorum* prior to the seeding of desirable perennials (Vallentine and Stevens 1994).

To be effective, fuel reduction must keep pace with fuel accumulation (Pyne *et al.* 1996). In the northern Great Basin, biomass production for *B. tectorum* averages 500-600kg ha<sup>-1</sup>, and can range from 30 to >1,500 kg ha<sup>-1</sup>, depending on precipitation (Uresk *et al.* 1979). Fuel loading has a strong influence on flame length and rate of spread (Pyne *et al.* 1996), and without some type of fuel reduction, loads can get high enough to support fires within a few years.

Targeted, or prescribed, cattle grazing at the most susceptible phenological stage (boot stage) can remove biomass, diminish subsequent regrowth, and reduce seed input of *B. tectorum* (Vallentine and Stevens 1994). The purpose of this project was to evaluate the effectiveness of using targeted cattle grazing to reduce the flame length and rate of spread of fires on *B. tectorum*-dominated rangelands in the northern Great Basin.

## Methods

### *Site description*

The study site is located in northwestern Nevada, 20 km southeast of McDermitt (E. 455618 N. 4641643) within the Quinn River Management Area of the Bureau of Land Management Winnemucca Field Office. It is on a 5% slope with a western aspect at 1400m elevation. Average annual precipitation is 228 mm, most of which falls as snow from November through March. Mean maximum (July) and minimum (January) temperatures are 17 and -1°C, respectively. The site has 50-60% *B. tectorum* cover.

Other species include annual pepperweed (*Lepidium perfoliatum*), tumble mustard (*Sisymbrium altissimum*), Scotch thistle (*Onopordum acanthium*), Sandberg bluegrass (*Poa secunda*), bulbous bluegrass (*Poa bulbosa*) and eightweek fescue (*Vulpia octoflora*). Islands of the native shrub big sagebrush (*Artemisia tridentata* spp. *wyomingensis*) are scattered throughout. The site is part of a 19,830 ha grazing allotment that is divided into 15 pastures (dominated by *Agropyron desertorum*, crested wheatgrass) and grazed in a rest-rotation/deferment system, where pastures are used early (March 1 to May 15), late (May 15 to August 31), deferred (July 1 to August 31), or fall/winter (October 1 to February 28), or receive complete rest in alternating years (USDI-BLM 1998). About 1,500 cow/calf pairs are divided into four distinct herds, each of which is generally grazed in separate pastures throughout the grazing season. Historically, herbaceous forage utilization estimates have ranged between 20-40% for the pastures. The site has burned in 1972, 1985, 1994 and 1996 as the result of wildfires.

Soils are characteristic of the McConnel series (sandy-skeletal, mixed, mesic Xeric Haplocambids). These are deep soils formed with mixed rock particles and components of loess and volcanic ash over lacustrine deposits or gravelly alluvium fans extending into the Quinn River Valley (USDA-NRCS 1997). These soils correspond to Loamy, Claypan, and Droughty Loam ecological sites in the 200-350 mm precipitation zone (USDA-NRCS 1997).

#### *Fuel treatments and experimental design*

Four grazing-burning treatments were arranged in a 2 X 2 factorial design in a block, and replicated 3 times (Fig. 2.1). Treatment plots were 60 X 60 m. Shred lines,

mowed to 4-8 cm high and 10m wide, were placed between treatments to reduce the potential of fire spread. The southern edge of each block had a 35m-wide *B. tectorum* “wick” to carry fires into the treatment plots (Fig. 2.1).

The four treatments included: graze and no-burn (GNB), graze and burn (GB), no-graze and burn (NGB), and a no-graze and no-burn control (NGNB). Although the present paper only addresses fire behaviour, the overall study was designed to also assess the resulting seed bank dynamics and above-ground community composition. Thus, we focus here on the two burn treatments (GB and NGB); the grazing only and control treatments (GNB and NGNB) are presented to provide a clearer understanding of the overall study layout. The GB and GNB treatments were intensively grazed (equivalent of 83 cow/calf pairs ha<sup>-1</sup>) during the boot stage (inflorescence emergence from the sheath) of *B. tectorum* in early May 2005. The plots were grazed to 80-90% removal of aboveground biomass over a 32-40 h period. Cool temperatures and frequent precipitation promoted regrowth and additional germination of *B. tectorum*, so intensive grazing (same duration and stocking density) was repeated on GB and GNB treatments in late May (boot stage) to maintain 80-90% removal of aboveground biomass. The GB and NGB treatments were burned in mid-October 2005 to assess the effects of fuel reduction on flame length and rate of spread. The NGNB control provided an estimate of aboveground biomass and species composition in the absence of grazing and burning. Grazing and burning treatments were repeated, respectively, in May and October 2006 for the seed bank and community composition portion of the overall study. The 2006 grazing treatments also required two periods of grazing (boot stage), in response to cool, moist spring conditions.



The GB and NGB treatments were located at the southern end of all three blocks (Fig. 2.1). The ignition point for all prescribed burns was a 35m-wide *B. tectorum* wick (Fig. 2.1), which allowed fires to reach peak behavior (flame length and rate of spread) before contacting the interface of the two burn treatments.

Fire behavior was recorded using three video cameras. One camera was mobile, and moved with the flame front along the north-south axis of the burn. This camera recorded the rate of spread and flame length of the burn in the wick 10m before reaching the GB and NGB treatments, at the interface of the wick and treatments, and at 5, 15, 35, and 55 m inside the treatment plots. Each of these points was marked with a 2-m Robel pole. This allowed for an accurate estimation of the flame length and rate of flame spread in the wick and treatments. The two additional cameras were placed 20m beyond the plot boundary at the northeast and northwest corners of each plot. This allowed for a different view of flame length and rate of spread. This was necessary because wildfire behavior is dependent on wind speed and direction, which can force smoke across the camera view, thus occluding filming.

The effects of targeted grazing on flame length and rate of spread were analyzed as a two-way factorial (grazing X distance) in a split-plot design with whole plots in blocks. Grazing is the treatment variable (GB or GNB) and distance is the distance between Robel poles. The significance of the relationship between grazing and distance, and flame length and rate of spread was tested using a mixed-model ANOVA ( $P$ -value  $\leq 0.05$ ) (SAS Institute 2005).

Climatic and fuel variables were also recorded to enable an accurate prediction of fire behaviour for the fuel type, fuel loading, fuel moisture, fuel bed depth, and weather

conditions. A portable weather station was on site at the time of the burn, recording air temperature, relative humidity, and wind speed and direction. To estimate fuel loads, we clipped the vegetation in five randomly located, 0.5 x 0.5-m quadrats within each burn plot prior to burning. Vegetation was clipped at the soil surface, separated by species, weighed and then later dried at 60° C for 48 hours. Fuel moisture was measured by comparing the wet weight to dry weight within the pre-burn biomass samples. To evaluate fuel continuity, percent cover measurements were collected (after grazing, before burning) using 10 sample points alternating along three permanent 30-m transects in each treatment plot. Cover (live plant canopy by species, litter, rock and soil surface) was measured in a 0.5 X 0.5-m quadrat at each sampling point. Litter depth was measured to the nearest mm when encountered in the cover survey. Fuel bed depth was measured at each Robel pole by assessing the plant height at which the pole width was obscured by standing biomass. The relationships between treatment and fuel load, fuel bed depth, litter depth and percent cover were evaluated with a two-tailed t-test (p-value  $\leq 0.05$ ).

#### *Fire behavior modeling*

Since treatment plots were burned during post-peak fire season, fire behavior was also analyzed using the BehavePlus 3.0 fire modeling system (Andrews *et al.* 2003). Fuel models were created by substituting fuel parameters (fuel load, surface area/volume ratio, fuel bed depth, heat content, extinction moisture) into the low load, dry climate grass (GR2 dynamic) fuel model for the GB treatment and the high load, dry climate grass (GR7 dynamic) fuel model for the NGB treatment (Scott and Burgan 2005). These

models were selected due to the similarity to actual fuel conditions within each treatment. The models were parameterized with the actual fuel and climatic conditions as the input variables. The models were then assessed based on the similarity of output (flame length, rate of spread) to actual fire behavior. Simulations were then run to determine fire behaviour in the GB and NGB treatments under changing environmental conditions. The simulation conditions were based on actual values (October) for wind speed ( $3 \text{ km h}^{-1}$ ) and fuel moisture (6%) and those found at peak fire season (July-August) in 2005, i.e. fuel moisture (2%) and wind speed ( $20 \text{ km h}^{-1}$ ).

## Results

### *Fuel treatments and prescribed burns*

Targeted grazing in May 2005 led to significant reductions in total biomass ( $P < 0.001$ ) and *B. tectorum* cover ( $P < 0.001$ ) in the GB plots prior to the implementation of prescribed burns in October 2005. When compared to the NGB treatment, the GB treatment had less than half the amount of total biomass, two-thirds the amount of *B. tectorum* cover, similar litter cover, half the litter depth, one-third the fuel bed depth, and twice the amount of soil cover (Table 2.1).

During the prescribed burns in October 2005 air temperature was  $25 \pm 6 \text{ }^\circ\text{C}$  and relative humidity was  $21 \pm 8\%$ , resulting in a fuel moisture of  $6 \pm 2\%$ . Wind speed was  $3 \pm 3 \text{ km h}^{-1}$  and direction was variable. Grazing resulted in significant reductions in flame length between the GB and NGB treatments ( $F=140.39$ ;  $P < 0.001$ ) and across distance within the GB treatment plots ( $F=18.25$ ;  $P < 0.001$ ) (Fig. 2.2; Table A.1). In the wick and at the wick-treatment interface, flame lengths were indistinguishable for the two

treatments ( $F=0.46$ ;  $P=0.801$ ). As the flame front reached the 5-m point inside the treatments, flame lengths in the GB treatment ( $0.5 \pm 0.1$  m) were one-fourth as long as those in the NGB treatment ( $2.3 \pm 0$  m). By the 15-m point, flame lengths in the GB treatment ( $0.25 \pm 0.1$  m) were one-eighth as long as those in the NGB treatment ( $2.3 \pm 0.2$  m). Grazing, however, did not lead to significant changes in rate of spread between the GB and NGB treatments ( $F=3.46$ ;  $P=0.069$ ) (Fig. 2.3; Table A.2). After the fire encountered the wick-treatment interface, the rate of spread did not exceed  $7 \text{ m min}^{-1}$  for either treatment.

A second targeted grazing period in May 2006 led to significant reductions in total fuel biomass ( $P<0.001$ ), *B. tectorum* cover ( $P<0.001$ ) and fuel bed depth ( $P<0.001$ ) in the GB treatment prior to the October 2006 prescribed burns. When compared to the NGB treatment, the GB treatment had half the amount of total biomass, less than one-third the amount of *B. tectorum* cover, similar litter cover and depth, and one-fourth the fuel bed depth (Table 2.1; Tables A.1 and A.2).

During the prescribed burns in October 2006, air temperature was  $22 \pm 10$  °C and relative humidity was  $18 \pm 12\%$ , resulting in a fuel moisture of  $6 \pm 2$  %. Wind speed was  $3 \pm 3$  km h<sup>-1</sup> and direction was variable. Grazing resulted in complete extinction of the prescribed burn. As the flame front reached the wick-treatment interface the fire slowed drastically and only carried up to 5m in several areas of the GB treatment before complete extinction. The NGB treatment in 2006 had flame lengths ( $F=0.54$ ;  $P=0.751$ ) and rates of spread ( $F=0.81$ ;  $P=0.691$ ) similar to those for the GB treatment in 2005.

*Fire behavior modeling*

Simulations using the BehavePlus 3.0 low and high load dry climate grass models provided estimations of fire behavior under differing environmental conditions. The actual fuel moisture (6%) during the prescribed burns in October 2005 was well above peak fire condition (July-August) fuel moisture (2%), and the actual wind speed (3 km h<sup>-1</sup>) during the prescribed burns was well below peak fire condition wind speed (20 km h<sup>-1</sup>). Under prescribed burn conditions, the low load dry climate grass model accurately estimated flame lengths for the GB treatment (actual = 0.2 m, modeled = 0.2 m), and the high load dry climate grass model also accurately estimated flame lengths for the NGB treatment (actual = 2.0 m, modeled = 2.2 m). Under peak fire conditions, predicted mean flame lengths (0.6 m) in the GB treatment were one-eighth as long as those predicted in the NGB treatment (4.8 m) (Fig. 2.4). The mean actual and modeled rates of spread were similar for the GB treatment (actual = 7.3 m min<sup>-1</sup>, modeled = 5.8 m min<sup>-1</sup>) and the NGB treatment (actual = 12.4 m min<sup>-1</sup>, modeled = 13 m min<sup>-1</sup>) under prescribed burn conditions (Fig. 2.5). Modeling at peak fire conditions generated rates of spread 16 times faster in the NGB treatment (231 m min<sup>-1</sup>) as compared to the GB treatment (13 m min<sup>-1</sup>).

Simulations using the conditions during the October 2006 prescribed burn resulted in flame lengths and rates of spread in the NGB treatment similar to those in the 2005 GB treatment. The GB treatment in 2006 was incapable of supporting a fire under actual conditions and peak fire conditions due to low fuel loads.

**Discussion**

Poorly managed cattle grazing on arid and semiarid rangelands in the western

USA can lead to a simplification of plant communities (Fuls 1992), and in conjunction with fire, can result in sites dominated by species such as *B. tectorum* (D'Antonio and Vitousek 1992). *B. tectorum*-dominated sites are characterized by high levels of fine fuel deposition, creating a contiguous and volatile fuel bed (McAdoo *et al.* 2007a). When sites such as these are burned, *B. tectorum* communities will continue to dominate (Young *et al.* 1987). However, targeted grazing of annual grasses at the boot stage can suppress seed production and plant yield (Young *et al.* 1987; Mosley and Roselle 2006; McAdoo *et al.* 2007b), and lead to a decrease in fuel biomass and connectivity, thus moderating fire behavior (Taylor 1994).

Targeted grazing reduced the total fuel load and percent cover of *B. tectorum* in the GB treatment. These reductions in the fuel load, *B. tectorum* percent cover, and fuel bed depth, and the increase in bare soil, resulted in a reduction in fuel connectivity. While the existing litter cover did not significantly change with grazing, the removal of *B. tectorum* plants and cattle hoof action led to an increase in bare soil, creating a patchy litter bed that still allowed fire to spread through the GB treatment but resulted in islands of unburned vegetation.

Targeted sheep grazing and herbicide application are also capable of altering the fuel characteristics of *B.tectorum*-dominated sites in the northern Great Basin (Smith *et al.* 2000; Kury *et al.* 2002). Smith *et al.* (2000) used 90 sheep ha<sup>-1</sup> to reduce herbaceous fuel loads and create a 60 m wide fuel break along a portion of the urban/wildland interface surrounding Carson City, Nevada. The plant community was dominated by *B. tectorum*, with interspersed crested wheatgrass (*Agropyron desertorum*) and *P. secunda*. Intensive sheep grazing reduced biomass by 73%, litter by 60% and fuel bed depth by

75%. These findings are similar to ours, in terms of biomass reduction and fuel bed depth; however, litter reduction appears to be higher with the use of sheep.

Similarly, Kury *et al.* (2000) used varying levels of Plateau® (imazapic ammonium salt) to suppress *B. tectorum* south of Boise, Idaho. Their site was similar to ours in terms of climatic conditions and species composition, being dominated by *B. tectorum* with low levels of *P. secunda*. The use of 437 ml ha<sup>-1</sup> of herbicide resulted in maximum *B. tectorum* suppression, reducing *B. tectorum* biomass by 80-90%, as did our GB treatment.

Unlike these previous studies (Smith *et al.* 2000; Kury *et al.* 2002), our study evaluated the fire behavior of fuel reduction treatments with prescribed burning. As the fuel loads and fuel connectivity were reduced in the GB treatment, flame lengths were reduced to one-eighth of those in the NGB treatment. The decrease in flame length resulted in a reduction of fuel preheating adjacent to the flame front (Pyne *et al.* 1996). While a 2-m flame length, as in the NGB treatment, is capable of preheating fuels up to 2-m ahead of the flame front, a 0.25-m flame length, as in the GB treatment, is only capable of preheating directly adjacent fuels. The low and erratic wind speeds during the prescribed burn resulted in vertical flame lengths; thus, only directly adjacent fuels were preheated, and rates of spread between the GB and NGB treatments were not significantly different (Pyne *et al.* 1996).

Even though we were able to evaluate fire behavior of grazed and non-grazed treatments with prescribed burning, the lack of fire suppression personnel during the peak fire season (July-August) did not allow us to burn until October in 2005 and 2006. However, the October burns did provide baseline data for fire behavior modeling under

different fuel and climatic conditions. The simulations based on October 2005 fuel and climatic conditions mirrored the actual fire behavior during the prescribed burn. Given the high degree of similarity between actual and modeled fire behavior under October conditions, the models have high inferential value for peak fire conditions in July and August. Modeling of fire behavior based on our simulations indicates that at lower fuel moisture (2%) and higher wind speed (20 km h<sup>-1</sup>), flame lengths up to 4.5 m play a significant role in preheating fuels. At high wind speeds in grasslands, flame lengths can become near horizontal (Pyne *et al.* 1996). Under peak fire conditions, a fire in the NGB treatment could spread at up to 231 m min<sup>-1</sup>, while a fire in the GB treatment is only capable of spreading at up to 12.4 m min<sup>-1</sup>. The combination of a high fuel load, high *B. tectorum* cover and a contiguous litter bed, as in the NGB treatment, results in a high rate of spread. While litter cover is similar between the two treatments, litter depth is much lower and bare soil is higher in the GB treatment, due to the reduced cover of *B. tectorum*. This shallow and less contiguous litter cover, along with a lower fuel bed depth, reduces the potential for fast moving fires in the GB treatment.

Simulations based on the 2006 GB fuel conditions (resulting from targeted grazing in May 2005 and 2006, and prescribed burning in 2005) indicate that even under peak fire conditions, a flame front would not carry in this treatment. The simulation models indicate that under peak fire conditions, the 2006 NGB treatment will support a burn with low flame lengths and slowed rate of spread, as observed in the 2005 GB treatment.

While this is the first study to evaluate *B. tectorum* fuel reduction with grazing and the consequences for fire behavior, others have used modeling and observed



estimates to evaluate fire behavior for other fuel reduction treatments (Smith *et al.* 2000; Kury *et al.* 2002). While the Plateau<sup>®</sup> treated site in Idaho was not burned, fire behavior was modeled using a short grass prairie model in BehavePlus (Kury *et al.* 2002). The resulting simulations of fire behavior indicated that flame lengths in the herbicide treatment would not exceed 0.5 m, while the untreated control would exceed 4 m. The model also estimated rates of spread not exceeding 12 m min<sup>-1</sup> in the herbicide treatment and up to 231 m min<sup>-1</sup> in the control. Running their fuel parameters in our models indicated that these are slight overestimations of fire behavior. These overestimations are due to the use of an earlier fuel model and the lack of an actual burn parameterized model. Thus, the treatment effects on modeled fire behavior in the herbicide study were very similar to those in our study.

The fuel break created with targeted sheep grazing along the urban/wildland interface surrounding Carson City, Nevada did burn in a wildfire event (Chapman pers. comm. 2007). Observed estimates during the burn indicate that the grazing treatment suppressed flame lengths from above 2 m to below 1 m and slowed the rate of spread by three-quarters. While the model used by Kury *et al.* (2002) may have been inappropriate and the estimates of fire behavior by Chapman were not modeled, they demonstrate a similar change in fire behavior (flame length and rate of spread) associated with similar types of fuel reduction treatments (Personal communication 2007).

In summary, targeted cattle grazing of *B. tectorum* at the boot stage has the potential to moderate the flame length and rate of spread of wildfires. This grazing treatment reduced percent cover of *B. tectorum*, fuel bed depth and fuel loading, and thus the flame length and rate of spread. By basing our simulations on the actual burning of

the treatments, we provide a high degree of inference to peak fire season estimates. These simulations of fire behavior indicate a decrease in predicted wildland fire behavior (flame length and rate of spread) under peak fire conditions. These findings constitute an initial step in reducing the threat of catastrophic wildfires on *B. tectorum*-dominated rangelands in the northern Great Basin.

### Management Implications

The reduction of wildfire flame lengths and rates of spread through targeted grazing leads to a need for fewer fire suppression resources. Flame lengths above 1 m, as observed in the NGB treatment, require the use of indirect attack methods. Flame lengths near 0.25 m, as observed in the GB treatment, can be managed via direct attack (Pyne *et al.* 1996). Fewer fire resources reduce the cost of suppression, and release resources for higher severity wildfires.

While a single targeted grazing event as in May 2005 has the potential to slow a wildfire, the combination of grazing and burning, followed by a second targeted grazing as in May 2006 has the potential to stop a wildfire. The reduction in fuel load, fuel bed depth and *B. tectorum* cover along with an increase in bare soil accomplished with a graze-burn-graze treatment appears to virtually eliminate the spread of a wildfire. The reduction in fuel loading is likely only viable for 1 to 2 years.

Targeted grazing of *B. tectorum* at the boot stage is capable of reducing flame length and rates of spread, but how should grazing be implemented? Grazing management limits the application of this method in space and time. The temporal window for *B. tectorum* treatment is narrow, between the short boot stage and soft dough

stage (2-3 weeks in spring), and the treatment must be repeated within that window (Mosley and Roselle 2006). The spatial extent of this grazing treatment is limited both by the time of grazing and the management goals for the grazers (McAdoo *et al.* 2007b). Given these temporal and spatial constraints, we recommend that the two-thirds ha GB treatment plot size used in this study can be scaled up in two ways: as a strip or as a large block. A strip 100m wide could be used as a brown strip fire break around desirable plant communities or simply to supplement current fire breaks such as roads (McAdoo *et al.* 2007b). The maintenance of Bureau of Land Management green strips (primarily *Agropyron desertorum*) in Idaho is reliant on livestock to reduce fuel loads (Davison 1996). Targeted grazing of large blocks and strips could be used as the initial step in the revegetation of *B. tectorum*-dominated sites (Miller 2006). Grazing *B. tectorum* may allow for native or non-native desirable plants to be seeded into the site and establish with reduced competitive pressure from *B. tectorum* (Svejcar 1990). Targeted grazing of *B. tectorum*-dominated sites can be applied as a first step in breaking the cheatgrass/fire cycle via removal of fire disturbance. However, caution must be used in applying this level of biomass removal in a community that retains some native or desirable plant species.

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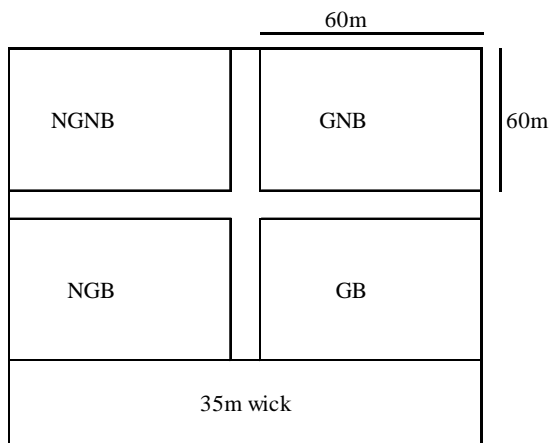
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**Table 2.1.** Mean values ( $\pm$  SE) for plants, litter and soil cover, *B. tectorum* and total plant biomass and fuel characteristics, including fuel load, litter depth, and fuel bed depth for GB (graze and burn) and NGB (no-graze and burn) treatments in 2005 and 2006.

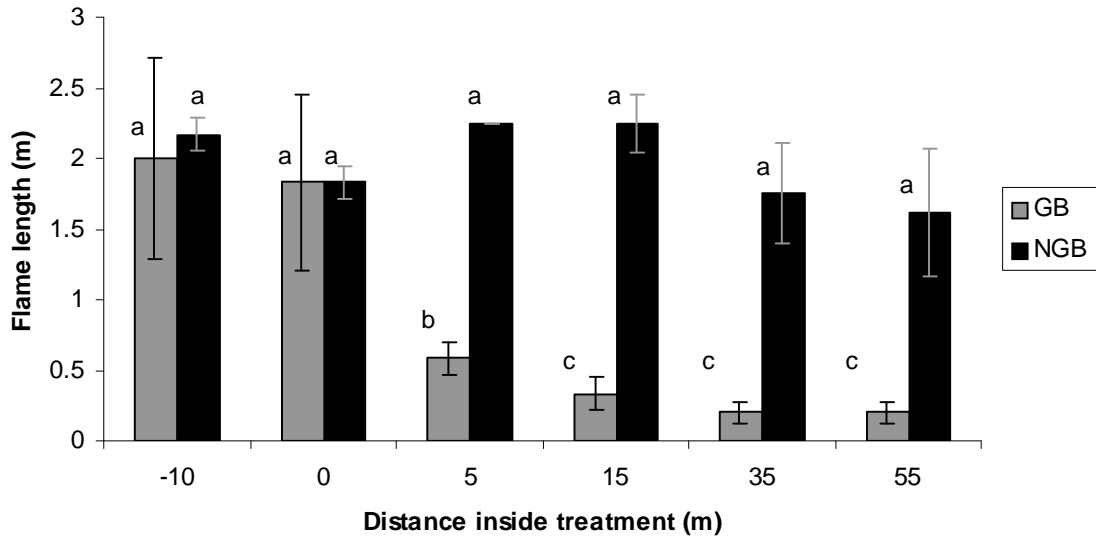
Attributes *	2005		2006	
	GB	NGB	GB	NGB
<i>B. tectorum</i> cover (%)	33 $\pm$ 4.3	52 $\pm$ 4.3	16 $\pm$ 4	51 $\pm$ 4.3
<i>P. Secunda</i> cover (%)	1 $\pm$ 1.3	7 $\pm$ 1.3	10 $\pm$ 1.1	11 $\pm$ 1.1
Forb cover (%)	25 $\pm$ 1.8	16 $\pm$ 1.8	6 $\pm$ 1.5	3 $\pm$ 1.5
Litter cover (%)	10 $\pm$ 3.7	10 $\pm$ 3.7	18 $\pm$ 3.7	9 $\pm$ 3.5
Soil cover (%)	31 $\pm$ 4.4	15 $\pm$ 4.4	45 $\pm$ 4.2	17 $\pm$ 4.1
<i>B. tectorum</i> Biomass (kg h <sup>-1</sup> )	340 $\pm$ 47	896 $\pm$ 47	34 $\pm$ 43	101 $\pm$ 42
Total Biomass (kg h <sup>-1</sup> )	550 $\pm$ 36	1225 $\pm$ 36	63 $\pm$ 36	122 $\pm$ 33
Fuel load (kg h <sup>-1</sup> )	445 $\pm$ 84	1503 $\pm$ 227	150 $\pm$ 53	260 $\pm$ 72
Litter depth (cm)	1 $\pm$ 0.1	2 $\pm$ 3	0.8 $\pm$ 0.1	0.6 $\pm$ 0.2
Fuel bed depth (cm)	20 $\pm$ 1.1	60 $\pm$ 21	10 $\pm$ 0.7	40 $\pm$ 13

\* Cover and biomass measurements taken in July; fuel load litter depth and fuel bed depth taken just prior to burning in October.

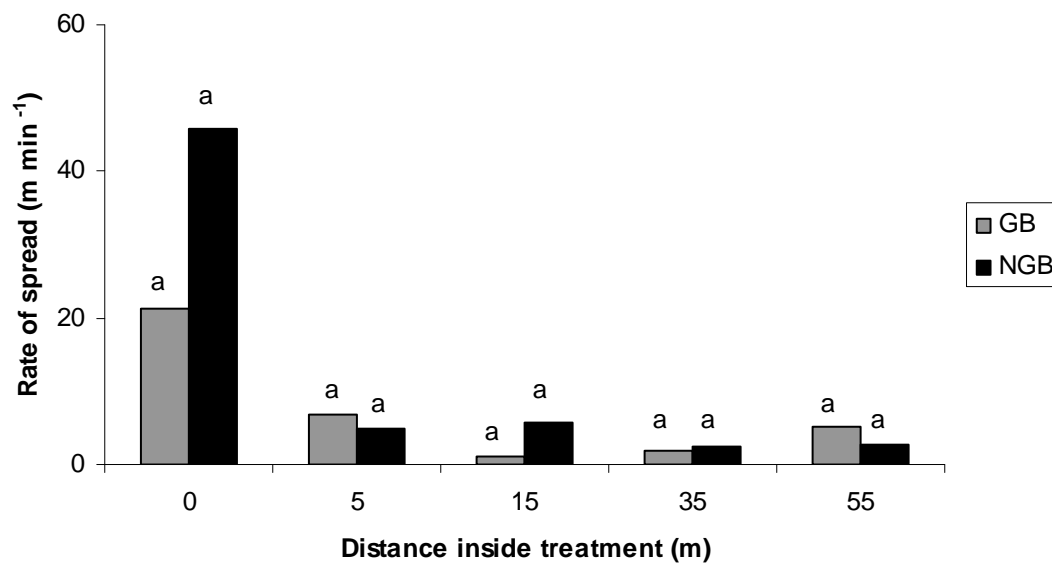


**Figure 2.1.** Treatment plot layout within block; southern end of each block has a 35m *B.tectorum* wick to carry the fire into burning treatments. GB = graze and burn, GNB = graze and no-burn, NGB = no-graze and burn, and NGNB = no-graze and no-burn.

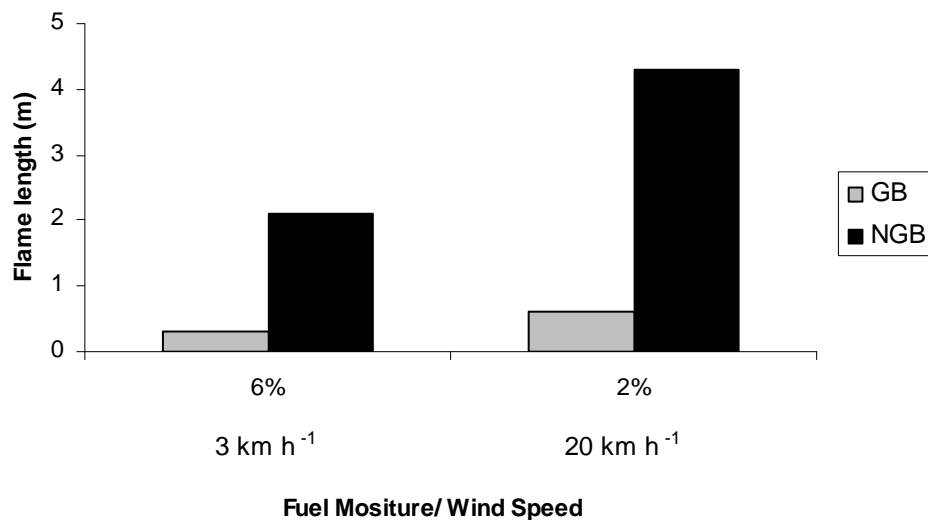




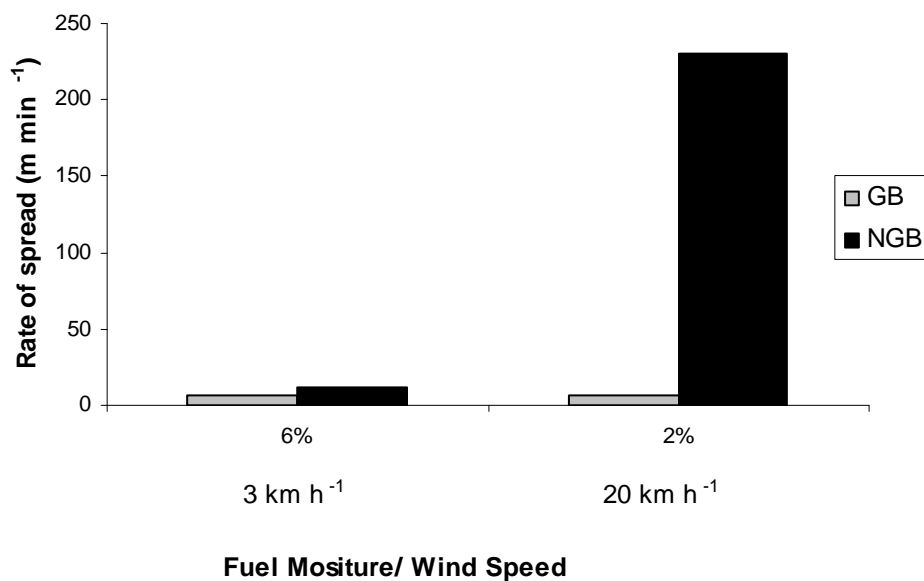
**Figure 2.2.** Mean flame length for treatments GB = graze and burn, NGB = no-graze and burn at Robel pole distances inside treatment plots during prescribed burn in October 2005. Distance begins at -10m, in the wick; at 0m, fire is at the wick-treatment interface. Error bars represent the standard error and different letters indicate significant differences at  $P < 0.05$ .



**Figure. 2.3.** Mean rate of spread for treatments GB = graze and burn, NGB = no-graze and burn at Robel pole distances inside treatment plots during prescribed burn in October 2005. Distance begins at 0m at the wick-treatment interface. There were no significant differences at  $P < 0.05$ .



**Figure 2.4.** Simulated flame length for treatments GB = graze and burn, NGB = no-graze and burn based on actual prescribed burn conditions (6% fuel moisture and 3 km h<sup>-1</sup> wind) and peak fire season conditions (2% fuel moisture and 20 km h<sup>-1</sup> wind) during July-August 2005.



**Figure 2.5.** Simulated rate of spread for treatments GB = graze and burn, NGB = no-graze and burn based on actual prescribed burn conditions (6% fuel moisture and 3 km h<sup>-1</sup> wind) and peak fire season conditions (2% fuel moisture and 20 km h<sup>-1</sup> wind) during July-August 2005.

CHAPTER 3  
EFFECTS OF TARGETED GRAZING AND PRESCRIBED BURNING  
ON COMMUNITY AND SEED DYNAMICS OF A  
CHEATGRASS-DOMINATED LANDSCAPE<sup>1</sup>

*Abstract.* Cheatgrass (*Bromus tectorum*)-dominated communities can remain as stable states for long periods, even with frequent disturbance by grazing and fire. However, properly timed (targeted) grazing and prescribed burning can modify seed dispersal and storage, site availability for establishment, and species performance, possibly directing the trajectory of succession to a more desirable community. The objective of this study was to determine the effectiveness of using targeted cattle grazing and late-season prescribed burning, alone and in combination, to reduce *B. tectorum* seed input and seed bank density, and alter aboveground community dynamics (species composition, cover biomass and density) on a *B. tectorum*-dominated landscape in northern Nevada. Cattle removed 80-90% of standing biomass in grazed plots in May of 2005 and 2006 when *B. tectorum* was in the boot (phenological) stage. Grazed and ungrazed plots were burned in October 2005 and 2006. The combined grazing-burning treatment was more effective than either treatment alone in reducing *B. tectorum* cover, biomass, plant density, and seed density, and in shifting species composition from a community dominated by *B. tectorum* to one composed of a suite of species, with *B. tectorum* as a component rather than a dominant.

<sup>1</sup> Coauthored by J.M. Diamond, C.A. Call, and N. Devoe.

## INTRODUCTION

The Great Basin now has more than 6.8 million ha dominated by *Bromus tectorum* (cheatgrass) and an additional 25 million ha with *B. tectorum* as a component species (Morrow and Stahlman 1984; Pellant and Hall 1994). Over 20% of former Great Basin sagebrush-grassland communities are infested with *B. tectorum* to levels that preclude reestablishment of native perennial species (Knapp 1996). This community degradation is due primarily to overgrazing by domestic livestock, the associated invasion of *B. tectorum*, and the resulting grass/fire cycle (Young et al. 1987; Whisenant 1990; Miller et al. 1994). *B. tectorum* provides the fine fuel that facilitates frequent, large wildfires (Upadhyaya et al. 1986; D'Antonio and Vitousek 1992). While historical fire frequencies in Great Basin sagebrush-grassland communities prior to *B. tectorum* invasion were 50-100 years (Harniss and Murray 1973; Wright and Bailey 1982), they are now 2-15 years (Yensen 1981; Whisenant 1990). This grass/fire cycle, in conjunction with plant traits such as early, prolific growth and high seed production, promote *B. tectorum* dominance (Uresk 1979 et al.; Melgoza and Nowak 1991; Nasri and Doescher 1995; Rice 2005).

Invasive species such as *B. tectorum* can have large propagule pools. Depending on plant densities and environmental conditions, individual *B. tectorum* plants can produce 10-6,000 seeds (Hulbert 1955; Young and Evans 1978). Seed bank densities of 2,400-8,300 seeds m<sup>-2</sup> have been reported for over-grazed sagebrush communities (Young and Evans 1975), and 4,800-19,000 seeds m<sup>-2</sup> for unburned, *B. tectorum*-dominated communities in the Great Basin (Humphrey and Schupp 2001; Hempy-Mayer

and Pyke 2008). After summer dispersal (late-May to late June, depending on moisture conditions), most seeds germinate the following fall through spring; however, seeds can remain viable in the soil for up to 2-5 years (Burgert et al. 1971; Wicks et al. 1971; Mack and Pyke 1983; Smith et al. 2008). The majority (90%) of seeds disperse a short distance from the parent plant by dropping to the soil surface and being moved along the soil surface via wind (Mack and Pyke 1983). The short rigid hairs on the awns and lemmas facilitate long distance dispersal by attaching to animal fur and human clothing (Pyke and Novak 1994). These dispersal methods allow *B. tectorum* to readily colonize disturbed sites and adjacent communities.

*B. tectorum*-dominated communities in the Great Basin are characterized by near monotypic stands of *B. tectorum* with interspersed annual forbs such as *Sisymbrium altissimum* (tumble mustard) and *Lepidium perfoliatum* (clasping pepperweed), and remnant bunchgrasses such as *Poa secunda* (Sandberg bluegrass). *S. altissimum* and *L. perfoliatum* colonize sites after disturbance and modify the seed bed to allow for *B. tectorum* germination, while *P. secunda* tolerates disturbance. Litter deposited by *S. altissimum* and *L. perfoliatum* moderates temperature and moisture conditions on the soil surface, providing safe sites for *B. tectorum* germination (Piemeisel 1951; Young et al. 1969; Facelli and Pickett 1991). The two forb species can comprise 10% (wet year) to more than 40% (dry year) of a *B. tectorum*-dominated community, while *P. secunda* remains a small component even under heavy grazing and frequent burning (Hironaka and Tisdale 1963; Young et al. 1970).

Vegetation manipulation methods such as targeted grazing and prescribed burning have the potential to reduce *B. tectorum* dominance by altering seed and aboveground

community dynamics. Intensive sheep grazing at the boot stage reduces the seed production, biomass, density, and cover of annual grasses (Mosley and Roselle 2006; McAdoo et al. 2007). Grazing, however, does not significantly alter site characteristics (litter cover and depth) that facilitate *B. tectorum* establishment, because it only removes standing biomass. Prescribed burning has the potential to alter *B. tectorum* site characteristics by removing or reducing the litter bed (Humphrey and Schupp 2001). Fire also kills many of the seeds in the litter and on the soil surface; however, this reduction in seed bank density may last for only 1 year (Evans and Young 1984).

Targeted grazing and prescribed burning can be viewed as designed disturbances used to modify the causes of succession, i.e. site availability (safe sites for establishment), species availability (propagule dispersal and storage), and species performance (growth and reproduction), in the successional weed management framework developed by Sheley et al. (1996, 2006). This framework allows for a more process-based approach to address *B. tectorum*-dominated rangeland management.

The objective of this study was to determine the effectiveness of using targeted cattle grazing and late-season prescribed burning alone, and in combination, to reduce *B. tectorum* dispersal (seed input) and seed bank density, and alter aboveground community dynamics (species composition, cover, biomass and density) on a *B. tectorum*-dominated landscape in northern Nevada.



## METHODS

*Site Description*

The study site is located in northwestern Nevada, 20 km southeast of McDermitt (E. 455618, N. 4641643), within the Quinn River Management Area of the Bureau of Land Management Winnemucca Field Office. It is on a 5% slope with a western aspect at 1400 m elevation. Average annual precipitation is 228 mm, most of which falls as snow from November through March. Mean maximum and minimum temperatures are 17 and  $-1^{\circ}\text{C}$ , respectively. The site has 50-60% *B. tectorum* cover. Other primary species include *L. perfoliatum*, *S. altissimum* and *P. secunda*. Secondary species include annual and biennial forbs [*Onopordum acanthium* (Scotch thistle), *Ceratocephala testiculata* (bur buttercup), *Chorispora tenella* (blue mustard), *Alyssum desertorum* (desert alyssum), *Erodium cicutarium* (filaree) and *Lactuca serriola* (prickly lettuce)], the perennial grass *Poa bulbosa* (bulbous bluegrass) and the annual grass *Vulpia octoflora* (sixweek fescue). The *B. tectorum*-dominated site is part of a 19,830 ha grazing allotment that is divided into 15 pastures characterized by *Agropyron desertorum* (crested wheatgrass) and grazed in a rest-rotation/deferment system, where pastures are used early (March 1 to May 15), late (May 15 to August 31), deferred (July 1 to August 31), or fall/winter (October 1 to February 28), or receive complete rest in alternating years (USDI-BLM 1998). About 1,500 cow/calf pairs are divided into four distinct herds, each of which is generally grazed in separate pastures throughout the grazing season. Historically, herbaceous forage utilization estimates have ranged between 20-40% for the pastures. The site has burned in 1972, 1985, 1994 and 1996 as the result of wildfires.

Soils are characteristic of the McConnel series (sandy-skeletal, mixed, mesic Xeric Haplocambids). These are deep soils formed with mixed rock particles and components of loess and volcanic ash over lacustrine deposits or gravelly alluvium fans extending into the Quinn River Valley (USDA, NRCS 1997). These soils correspond to Loamy, Claypan, and Droughty Loam ecological sites in the 200-350 mm precipitation zone (USDA, NRCS 1997).

### *Treatments and Experimental Design*

Four grazing-burning treatments were arranged in a 2 X 2 factorial design in a block, and replicated 3 times (Fig. 3.1). Treatment plots were 60 X 60 m. Shred lines, mowed to 4-8 cm high and 10 m wide, were placed between treatments to reduce the potential of fire spread between treatments. The southern edge of each block had a 35m-wide *B. tectorum* “wick” to carry fires into the treatment plots (Fig. 3.1).

The four treatments included: graze and no-burn (GNB), graze and burn (GB), no-graze and burn (NGB), and no-graze and no-burn (NGNB). These treatments were designed not only to assess seed and aboveground community dynamics but also the resulting fire behavior (see Ch. 2). The GB and GNB treatments were intensively grazed (equivalent of 83 cow-calf pairs ha<sup>-1</sup>) during the boot stage of *B. tectorum* in early May 2005 and 2006. The plots were grazed to 80-90% removal of aboveground biomass over a 32-40 h period. Cool temperatures and frequent precipitation promoted regrowth and additional germination of *B. tectorum*, so intensive grazing (same duration and stocking density) was repeated on GB and GNB treatments in late-May to maintain 80-90% removal of aboveground biomass. While the initial study design called for peak fire

season (July-August) prescribed burns, Hurricane Katrina in 2005 and an active fire season in 2006 diverted fire management resources. Thus, the October burns, after peak fire season in 2005 and 2006, were implemented during the optimum time period for prescribed burning of sagebrush-grassland communities in the Great Basin (Bunting et al. 1987). The NGB treatment was burned in mid-October 2005 and 2006. The GB treatment was also burned in mid-October 2005, but did not have enough fuel (biomass and continuity) to carry a fire in mid-October 2006. The GB and NGB treatments were located at the southern end of all three blocks (Fig. 3.1). The ignition point for all prescribed burns was a 35-m wide *B. tectorum* wick (Fig. 3.1), which allowed fires to reach peak behavior (flame length and rate of spread) before contacting the interface of the two burn treatments. Rate of spread was consistent across treatments, while flame length was greater in the NGB treatment than in the GB treatment (Table 3.1).

#### *Aboveground Community Dynamics*

The effects of grazing and fire on the aboveground plant community were evaluated in three ways: biomass, percent cover, and density. To evaluate the biomass, we clipped the vegetation in 10, 0.5 m x 0.5 m quadrats alternating along a 30 m transect. Vegetation was clipped at the soil surface, separated by species, and dried at 60°C for 48 hours. Two transects were randomly placed within each of the treatments in all three blocks. Biomass samples were collected three times per year: pre-graze (late-April), post-graze (late-May) and at peak biomass (late-June). Species composition for the four grazing-burning treatments was determined each year (at peak biomass) by dividing the biomass for each species by the total biomass. To evaluate species density and percent

cover, we used a nested plot technique. Measurements were taken at the same temporal and spatial scales described above. We measured density and cover using 10 sample points alternating along three permanent 30 m transects in each treatment plot. Cover (live plant by species, litter, rock and soil) was measured with a 10-pin frame on a 0.5 X 0.5 m quadrat. *P. secunda* primarily reproduces vegetatively (tillering) and individual plants are commonly fragmented, thus density was not measured due to the difficulty of identifying individual plants post-fire (Young and Evans 1978). *B. tectorum* density was counted in a 10 X 10 cm quadrat nested in a corner of the 0.5 X 0.5 m quadrat. *S. altissimum* and *L. perfoliatum* individuals were enumerated in a nested 25 X 25 cm quadrat, and all other species were counted in a 0.5 X 0.5 m quadrat. All density data were scaled to plants per m<sup>2</sup>. Aboveground community dynamics data were analyzed in a 3-way factorial (fire X grazing X year + period). This is a split-split-plot design with whole plots in blocks. We used mixed-model ANOVA's to examine the effects of grazing, burning, and the combined effect of year and the period of collection on community dynamics. We used Fisher's Protected LSD and t-tests ( $P \leq 0.05$ ) to evaluate differences among treatment means (SAS Institute 2005).

### *Seed Dynamics*

Effects of grazing and fire on the seed dynamics of the four primary species, *B. tectorum*, *P. secunda*, *L. perfoliatum* and *S. altissimum*, were evaluated in two ways: seed bank density and seed input. We estimated seed bank density and composition for each of the treatments by collecting soil cores (5 cm in depth, 2.5 cm in diameter) and associated surface litter at 40 points in a 5 x 8 grid of sampling points 2.5 m apart within each

treatment plot. Samples were taken at 2.5-m intervals within the matrix. In 2005, a single soil core and associated litter was collected at each of the 40 points; however, in 2006, five soil cores and associated litter (sub-samples) were collected at each of the 40 points to reduce variance. The sub-samples were then mixed, and a composite soil/litter sample (1/5 of the volume) was used to evaluate the seed bank density. We repeated this sampling in each of the treatments three times per year: post-graze (late-May), peak biomass (late-June) and post-burn (mid-October). After soil/litter samples were stored in a cold room (3-4°C) for three months to meet dormancy requirements, each soil/litter sample was placed on top of 4.5 cm of sand in a 10 cm x 10 cm x 10 cm (w x l x h) container. Soil/litter samples were spread evenly across the sand surface. Containers were placed in a greenhouse with a 12 h light period and 12 h dark period, and a mean high temperature of 27°C and a mean low temperature of 15°C. After a 2 month growing period, the individuals of each species were identified, counted and removed. The soil/litter samples were then dried in the greenhouse for 1 month, and each soil/litter sample was mixed and placed back on the sand surface as before. The greenhouse study was then repeated. Some seeds that did not germinate in the first growing period germinated in the second growing period, allowing for a more complete estimation of the number of viable seeds per unit area. Seed bank data were analyzed in a 3-way factorial (fire X grazing X year + period). This is a split-split-plot design with whole plots in blocks. We used mixed-model ANOVA's to examine the effects of grazing, burning, and the combined effect of year and the period of collection on the seed bank. We used Fisher's Protected LSD and t-tests ( $P \leq 0.05$ ) to evaluate differences among treatment means (SAS Institute 2005).

Seed input was measured by collecting seed rain beginning in late-May (post-graze) and ending mid-July (just prior to the originally scheduled burn during peak fire season) in 2006. Seeds were collected using 10 sample points alternating along each of three randomly placed 30 m transects in each treatment plot. Each sample point consisted of a 10 cm diameter funnel buried to 1 cm above the soil surface to only allow seed entrance from above (methods modified from Chabrerie and Alard 2005). By only allowing seed entrance vertically, we are only measuring Phase I dispersal and avoiding the difficulty of measuring phase II dispersal (Chambers and MacMahon 1994). The funnel exit was sealed with cotton to allow water to pass through but no solids. The funnel was filled with fine gravel, which entrapped and incorporated the seeds over time. At the termination of the collection period, samples were stored in a cold room (3-4°C) for 3 months to meet dormancy requirements. Seeds from each sample were then sorted by species and placed on saturated filter paper in a 5 cm x 10 cm x 2.5 cm (w x l x h) germination container. A 12 h light period and a 12 h dark period, at mean high and low temperatures of 27°C and 15°C, respectively, were used in a germination chamber. Germination was evaluated for grasses with emergence of the radicle and coleoptile, and for forbs with the emergence of the radicle and the cotyledon. Seed input was not measured for *S. altissimum* because dispersal for this species does not occur until fall, after sampling was discontinued (Young et al. 1970). The effects of intensive grazing and prescribed burning on seed input were evaluated in a two-way factorial (grazing x burning). This is a split-split-plot design with whole plots in blocks. We used mixed-model ANOVA's and Fisher's Protected LSD test ( $p \leq 0.05$ ) to examine the effects of grazing and burning on seed input (SAS Institute 2005).

## RESULTS

### *Aboveground Community Dynamics*

*B. tectorum* cover ( $F = 29.73$ ;  $P < 0.001$ ), biomass ( $F = 29.51$ ;  $P < 0.001$ ), and density ( $F = 87.03$ ;  $P < 0.001$ ) differed significantly among the four grazing-burning treatments at peak biomass in 2005, 2006 and 2007 (Tables 3.2, 3.3 and 3.4; Table A.3). In 2005, the graze and no-burn (GNB) and graze and burn (GB) treatments had lower cover, biomass and density than the no-graze burn (NGB) and no-graze and no-burn (NGNB) treatments (Tables 3.2, 3.3 and 3.4). In 2006 and 2007, the GB treatment had lower cover and density than all other treatments, and lower biomass than the NGNB treatment. *B. tectorum* cover ( $F = 10.72$ ;  $P < 0.001$ ), biomass ( $F = 30.89$ ;  $P < 0.001$ ) and density ( $F = 8.65$ ;  $P < 0.001$ ) decreased within each treatment from 2005 to 2007 (Tables 3.2, 3.3 and 3.4; Table A.3). The magnitude of these decreases was greatest for the GB treatment, followed by the NGB, GNB and NGNB treatments, respectively.

There were also significant differences in *S. altissimum* cover ( $F = 14.99$ ;  $P < 0.001$ ), biomass ( $F = 11.41$ ;  $P < 0.001$ ), and plant density ( $F = 20.68$ ;  $P < 0.001$ ) among the four grazing-burning treatments in 2006 and 2007 (Tables 3.2, 3.3 and 3.4; Table A.4). In 2006, the GNB treatment had higher cover and density than all other treatments. In 2007, the GNB treatment had significantly higher biomass and density than all other treatments. There were also significant differences in *S. altissimum* cover ( $F = 4.14$ ;  $P < 0.001$ ), biomass ( $F = 5.17$ ;  $P < 0.001$ ), and plant density ( $F = 11.91$ ;  $P < 0.001$ ) within treatments across years (Tables 3.2, 3.3 and 3.4; Table A.4). The GB treatment decreased

in cover and density from 2005 to 2007, while biomass decreased from 2005 to 2006 but recovered by 2007. Cover and density peaked in 2006, while biomass was at its lowest value for the GNB treatment. The NGB treatment decreased in cover and density from 2005 to 2007, while biomass decreased from 2005 to 2006 and recovered by 2007. All three vegetation attributes decreased from 2005 to 2007 in the NGNB treatment.

*L. perfoliatum* cover ( $F = 16.53$ ;  $P < 0.001$ ), biomass ( $F = 7.40$ ;  $P < 0.001$ ), and density ( $F = 5.23$ ;  $P < 0.001$ ) differed significantly among the four grazing-burning treatments in 2005 and 2006 (Tables 3.2, 3.3 and 3.4; Table A.5). In 2005, the GB treatment had at least twice the cover of the other treatments (Table 3.2). By 2006, the GNB treatment had at least three times the cover of the other treatments. Trends were not as evident among the treatments for biomass and density. In 2005 and 2006, the GB treatment had greater biomass than the NGB treatment and greater density than the GNB treatment. *L. perfoliatum* cover ( $F = 8.36$ ;  $P < 0.001$ ), biomass ( $F = 7.40$ ;  $P < 0.001$ ) and plant density ( $F = 7.21$ ;  $P < 0.001$ ) varied significantly within treatments across years (Table A.5). In the GB treatment, cover and biomass decreased from 2005 to 2006, and by 2007 cover increased slightly and biomass decreased by half. Plant density did not differ from 2005 to 2006 but decreased by a factor of 10 by 2007 in the GB treatment. Cover in the GNB treatment increased 3-fold from 2005 to 2006 and decreased similarly from 2006 to 2007. Biomass in the GNB treatment decreased by 3-fold from 2005 to 2006 and remained at that level in 2007. Plant density in the GNB treatment did not differ from 2005 to 2006; however, by 2007, density had decreased 3-fold. In general all three vegetation attributes decreased from 2005 to 2007 in the NGB and NGNB treatments.

*P. secunda* cover ( $F = 63.78$ ;  $P < 0.001$ ) and biomass ( $F = 7.29$ ;  $P = 0.02$ ) varied



significantly across treatments within year (Tables 3.2 and 3.3; Table A.6). The NGB and NGNB treatments had 7 times the cover and almost 10 times the biomass of the GB and GNB treatments in 2005 (Tables 3.2 and 3.3). By 2006, the GB and NGB treatments had 5 times the cover of the GNB and NGNB treatments; however, there were no significant differences in biomass among the grazing-burning treatments. The GB and NGB treatments had higher cover and biomass than the GNB and NGNB treatments in 2007. Cover ( $F = 11.88$ ;  $P < 0.001$ ) and biomass ( $F = 41.82$ ;  $P < 0.001$ ) of *P. secunda* also varied significantly across years within treatments (Tables 3.2 and 3.3; Table A.6). Cover increased in all grazing-burning treatments from 2005-2007, except for the NGNB treatment, where cover was highest in 2005. Biomass response was highly variable, with the GB treatment having at least 10 times the biomass in 2007 than in the previous 2 years, and the NGB and NGNB treatments having almost 25 times the biomass in 2005 than in 2006. There were no significant differences in biomass for the GNB treatment from 2005 to 2007.

The other forb species (*Onopordum acanthium*, *Ceratocaphala testiculatus*, *Chorispora tenella*, *Alyssum desertorum*, *Erodium cicutarium* and *Lactuca serriola*) were minor components of the *B. tectorum*-dominated community at the study site, and thus were not well distributed throughout the treatment plots. When analyzed as a group, there were no significant differences in cover and density across treatments within year, or across years within treatment (Tables 3.2 and 3.4). Biomass varied ( $F = 1.64$ ;  $P = 0.133$ ) across treatments within year, with the NGB treatment having higher biomass than the other three treatments all three years (Table 3.3; Table A.7). Biomass also varied ( $F = 3.97$ ;  $P < 0.001$ ) across years within treatment, with the NGB and NGNB treatments

having higher biomass in 2005 than in 2006 and 2007 (Table 3.3; Table A.7).

As with the other forb species, the other grass species (*Poa bulbosa* and *Vulpia octoflora*) were also minor components in the treatment plots. There were few significant differences in cover ( $F = 13.40$ ;  $P = 0.005$ ) and no significant differences in biomass ( $F = 1.13$ ;  $P = 0.409$ ) and density ( $F = 0.38$ ;  $P = 0.769$ ) across treatments within year (Tables 3.2, 3.3 and 3.4; Table A.8). In 2005, the NGB treatment had at least 3 times the biomass of the other treatments. In 2006, only the GB and NGB treatments had measurable cover. Cover ( $F = 22.00$ ;  $P < 0.001$ ) and biomass ( $F = 6.69$ ;  $P < 0.001$ ) varied within treatment across years (Table A.8). Both the GB and NGB treatments had significant increases in cover from 2005 to 2006, but cover decreased to initial levels for both treatments by 2007. In the NGB and NGNB treatments, there was a significant reduction in biomass from 2005 to 2006, and biomass remained low in 2007.

Grazing and burning, in combination, had the greatest impact on species composition from 2005 to 2007 (Fig. 3.4). In the GB treatment, *B. tectorum* composition decreased from  $> 50\%$  to  $< 10\%$ , *P. secunda* increased from  $< 5\%$  to  $> 50\%$ , *S. altissimum* increased from  $5\%$  to  $> 20\%$ , *L. perfoliatum* decreased from  $20\%$  to  $5\%$ , and other forbs and grasses increased from  $< 2\%$  to  $> 15\%$ . In the GNB treatment, *B. tectorum* decreased from  $> 60\%$  to  $< 50\%$ , *P. secunda* and *L. perfoliatum* remained at  $10\%$ . *S. altissimum* increased from  $10\%$  to  $> 20\%$  and other forbs and grasses increased from  $< 2\%$  to  $> 10\%$ . In the NGB treatment, *B. tectorum* composition decreased from  $60\%$  to  $< 50\%$ , *P. secunda* increased from  $2\%$  to  $10\%$ , *S. altissimum* increased from  $5\%$  to  $> 15\%$ , *L. perfoliatum* decreased from  $5\%$  to  $2\%$  and other forbs and grasses increased from  $< 2\%$  to  $> 10\%$ . Species composition in the NGNB treatment did not vary markedly

throughout the study. *B. tectorum* accounted for 60-90% of the composition, *P. secunda* from 1-2%, *S. altissimum* from 2-10%, *L. perfoliatum* from 1-10%, and other forbs and grasses accounted for 1-5%.

Changes in plant species cover were associated with changes in soil cover (bare soil) and litter cover over the 3-year study period. Soil cover ( $F = 24.42$ ;  $P < 0.001$ ) and litter cover ( $F = 6.09$ ;  $P = 0.029$ ) varied significantly across treatments within year (Table 3.2; Table A.9). In 2005, the GB and GNB treatments had twice the amount of bare soil as did the NGB and NGNB treatments. By 2006, the amount of bare soil in the GB treatment was 3 times that of the GNB and NGB treatments, and 15 times that of the NGNB treatment. In 2007, the GB treatment had the highest amount of bare soil, followed in decreasing order by the NGB, GNB and NGNB treatments. This trend for bare soil aligns well with the opposite trend for litter cover in 2007, where the NGNB treatment had the highest litter cover, followed in decreasing order by the GNB, NGB and GB treatments. Soil cover ( $F = 16.52$ ;  $P < 0.001$ ) and litter cover ( $F = 18.58$ ;  $P < 0.001$ ) also varied significantly within treatment across years (Table 3.2; Table A.9). Bare soil was higher in 2007 than in 2005 in the GB and NGB treatments and the opposite trend occurred in the GNB and NGNB treatments. Interestingly, the highest litter cover in all four grazing-burning treatments occurred in 2007, and the lowest litter cover occurred in the GB, GNB and NGNB treatments in 2005.

### *Seed Dynamics*

*B. tectorum* seed bank density varied significantly ( $F = 8.68$ ;  $P < 0.001$ ) across treatments within sampling period, i.e. post-graze (late-May), peak biomass (late-June),

and post burn (mid-October), ranging from 66 to 4278 seeds m<sup>-2</sup> (Fig. 3.2a; Table A.19). Post-graze 2005, the GB and GNB treatments had less than half the seed density of the NGB and NGNB treatments. At peak-biomass 2005, after most of the seeds had dispersed, seed densities increased in all four grazing-burning treatments, but densities in the GB and GNB treatments remained significantly lower than in the NGB and NGNB treatments. Following burning in 2005, seed densities in the GB and NGB treatments were at least 100 times lower than those in the GNB and NGNB treatments. By post-graze 2006, after most of the seeds in the seed bank had germinated, seed densities were low in all four grazing-burning treatments. By peak biomass 2006, following seed dispersal, seed densities increased in all four treatments, with densities significantly lower in the GB and GNB treatments than in the NGB and NGNB treatments. This trend held through the last sampling period after burning in 2006. *B. tectorum* seed bank density also varied significantly ( $F = 13.57$ ;  $P < 0.001$ ) across sampling time periods within treatment (Table A.10). Seed densities were lowest in the GB and NGB treatments after burning in 2005; and within 1 year, they increased by 4-fold and 20-fold, respectively, after burning in 2006. Seed densities were lowest in the GNB treatment after grazing in 2005 and 2006. Since the NGNB treatment was not disturbed by grazing or burning, seed densities remained high during most sampling periods, except for post-graze 2005 and 2006, where densities were lowest after germination of seeds in the seed bank.

*P. secunda* seed densities were typically an order of magnitude lower than *B. tectorum* densities, ranging from 6 to 276 seeds m<sup>-2</sup>, and they differed across treatments ( $F=9.39$ ;  $P<0.001$ ) within sampling period and across time periods within

treatment ( $F=30.61$ ;  $P<0.001$ ) (Fig. 3.2b; Table A.13). The effects of grazing and burning on *P. secunda* seed densities were similar to those for *B. tectorum*, with the GB and GNB treatments generally having significantly lower seed densities than the NGB and NGNB treatments at the post-graze 2005 and peak biomass 2005 and 2006 sampling periods, and the GB and NGB treatments having lower seed densities than the GNB and NGNB treatments at the post-burn 2005 sampling period. And, *P. secunda* seed densities were low in all four grazing-burning treatments by the post-graze 2006 sampling period. There were no significant differences in seed densities among the four treatments after burning in 2006.

*L. perfoliatum* seed densities ranged from 2 to 289 seeds  $m^{-2}$ , and varied significantly across treatments within time period ( $F=8.68$ ;  $P<0.001$ ) and across time periods within treatment ( $F=60.1$ ;  $P<0.001$ ) (Fig. 3.2c; Table A.12). Densities were lowest (and similar) for all four grazing-burning treatments at the post-graze 2005 and 2006 sampling periods after seeds had germinated in the seed bank. By peak biomass 2005, following the dispersal of most *L. perfoliatum* seeds, densities increased in all treatments and were highest in the NGB treatment. Following burning in 2005, seed densities were lowest in the GB treatment. It appears that seed dispersal extended over a longer period in 2006, with densities increasing in the four grazing-burning treatments at peak biomass 2006, and further increasing after burning in 2006.

*S. altissimum* seed densities ranged from 19 to 261 seeds  $m^{-2}$ , and varied significantly ( $F = 7.41$ ;  $P < 0.001$ ) across treatments within time periods and across time periods within treatment ( $F=44.69$ ;  $P<0.001$ ) (Fig. 3.2c; Table A.11). Seed densities were low in all four grazing-burning treatments at the post-graze 2005 sampling period after

seeds had germinated in the seed bank. Seed densities increased in the GB and GNB treatments and remained unchanged in the NGB and NGNB treatments at the peak biomass period 2005 and post-burn 2005 sampling periods. Following seed dispersal from tumbling *S. altissimum* plants in the fall and winter, seed densities increased in all four treatments (with significant increases in the GNB, NGB and NGNB treatments) by post-graze 2006. Seed density remained highest in the GNB treatment at all sampling periods in 2006.

Seed input for *B. tectorum*, *P. secunda* and *L. perfoliatum*, extrapolated from seeds collected in funnel traps from late-May to mid-July 2006, varied across the four grazing-burning treatments (Fig. 3.3). *B. tectorum* seed input ranged from 250 to 2870 seeds per m<sup>-2</sup>, and was significantly reduced ( $F = 63.95$ ;  $P < 0.001$ ) by recent grazing in May 2006, with GB and GNB treatments having one-tenth the input of NGB and NGNB treatments (Table A.14). *P. secunda* seed input was much lower than for *B. tectorum*, ranging from 2 to 123 seeds m<sup>-2</sup>. The NGB treatment had at least 4 times greater seed input ( $F = 0.45$ ;  $P = 0.76$ ) than the GB and GNB treatments but also the NGNB treatment, which was never grazed or burned (Table A.16). *L. perfoliatum* seed input was even lower than that for *P. secunda*, ranging from 7 to 15 seeds m<sup>-2</sup>. Recent grazing in May 2006 did not reduce ( $F = 0.54$ ;  $P = 0.67$ ) the seed input of this forb as it did for *B. tectorum* and *P. secunda*. In fact, the NGB treatment had the lowest seed input (Table A.15).

## DISCUSSION

Anomalies in vegetation attributes over the 3-year study period are due, in part, to

data collection methods and treatment effects. Plant, litter and soil cover, and plant density measurements were recorded in 0.5 X 0.5 m quadrats along three, randomly-selected, permanent transects in each treatment. Using a 10-pin frame for cover measurements in quadrats is more objective than an ocular estimate; however, point estimates do not adequately account for the cover of minor species without using a very large number of points (Bonham 1989). Aboveground plant biomass was collected from 0.5 X 0.5 m quadrats along two, randomly selected, temporary transects (different locations for each sampling period) in each treatment; thus, biomass values often do not align with density and cover values. The graze and no-burn (GNB), no-graze and burn (NGB), and no-graze and no-burn (NGNB) treatments were implemented according to the study design in 2005 and 2006; however, the graze and burn (GB) did not have the fuel load or continuity to carry fires further than 5 m into treatment plots during the second prescribed burn in October 2006. The GB treatment therefore, was a graze (May 2005) / burn (October 2005) / graze (May 2006) treatment rather than the intended graze (May 2005) / burn (October 2005) / graze (May 2006)/ burn (October 2006) treatment. And, grazing and burning treatments created a more patchy distribution of vegetation, resulting in sample points on the same transects differing by an order of magnitude in density or biomass values. Thus, trends in cover, density and biomass were often inconsistent for treatments and sampling periods from 2005 to 2007.

Grazing and fire, and their interaction, have the potential to drastically alter plant communities, as observed in the decline of sagebrush-grasslands (Young et al. 1987; Whisenant 1990; Miller et al. 1994; Peters and Bunting 1994; Knick and Rotenberry 1997). Any recovery of plant communities in the northern Great Basin requires

management of these disturbances (Evans and Young 1984). Grazing has the potential to reduce *B. tectorum* dominance via the removal of biomass (fuel for frequent fires) and the suppression of reproductive potential (Daubenmire 1940; Mack and Pyke 1984; Pyke 1986). Prescribed burning also has the potential to suppress *B. tectorum* by killing/damaging seeds in the litter and modifying microsites for germination and establishment via litter removal (Evans and Young 1984; Humphrey and Schupp 2001). In our study, the combination of intensive grazing at the boot stage and a late-season prescribed burn reduced *B. tectorum* cover, biomass, plant density, seed input and seed bank density. Grazing and burning shifted species composition from a community dominated by *B. tectorum* to a community dominated by *P. secunda* and *S. altissimum*. Our results are supported by findings from other studies, where: 1) targeted sheep grazing was used to suppress *B. tectorum* and other annual grasses (Mosley and Roselle 2006); 2) *P. secunda* tolerated grazing and increased in cover following fire (Hironaka and Tisdale 1963); 3) *S. altissimum* increased in cover after burning (Young et al. 1972); and 4) *B. tectorum* seed bank density was reduced following fire (Evans and Young 1984; Humphrey and Schupp 2001).

The suppression of *B. tectorum* cover, biomass and plant density by two consecutive years of grazing (GNB treatment) led to increases in the biomass, density and cover of other species. The resultant community was composed of approximately 50% *B. tectorum*, 30-40% *S. altissimum*, 8-10% *L. perfoliatum*, 5% *P. secunda* and a small percentage of other grasses and forbs (*Ceratocephala testiculata*, *Chorispora tenella*, *Alyssum desertorum*, *Erodium cicutarium* and *Lactua serriola*, *Poa bulbosa* and *Vulpia octoflora*). Daubenmire (1940) observed an increase in *S. altissimum* on intensively



grazed sites in a bunchgrass prairie in southeastern Washington. These increases in *S. altissimum* are likely due to reduced competition from palatable species such as *B. tectorum*. Grazing not only suppressed *B. tectorum*, it also fragmented the litter bed via hoof action. The reduction in competition and increase in bare soil facilitated an increase in annual forbs. The increase in *S. altissimum* following grazing is short-lived because even though fragmented, remaining litter will provide safe sites for subsequent *B. tectorum* establishment. However, the persistent seed bank of *S. altissimum* allows for reestablishment following the next disturbance (Rickard 1985).

The gaze no-burn treatment (GNB) reduced *B. tectorum* seed bank density during most sampling periods in 2005 and 2006. The reduction of seed density following intensive grazing in May 2005 was likely due to cattle hoof action and environmental conditions. Trampling compressed and redistributed litter, increasing seed-soil contact for *B. tectorum* seeds suspended in the litter (Heady and Child 1994; Allen et al. 1995). Cool temperatures and frequent precipitation between the first (early-May) and second (late-May) grazing events promoted germination. Although not directly observed, trampling during both events may have also injured a portion of the seeds in the litter and on the soil surface. *B. tectorum* seeds are actually fairly large, elongated florets (lemma body 9-13 mm long) that could be susceptible to breakage under intensive hoof action. Seed bank density was low in all four grazing-burning treatments at the post-graze sampling period in May 2006. Germination of most of the seed in the seed banks in all treatments was promoted by favorable moisture conditions in fall 2005 and spring 2006 prior to grazing in May 2006. In a seed bank carryover study in western Utah, Smith et al. (2008) observed that most of the *B. tectorum* seed bank (96%) can germinate during the first fall

and spring after seed dispersal. Grazing at the boot stage reduced seed input into the seed bank, as reflected in lower seed bank densities at peak biomass sampling periods in late-June 2005 and 2006.

Two years of consecutive grazing also altered the seed dynamics of *P. secunda*, *L. perfoliatum*, and *S. altissimum*. Although an order of magnitude lower, *P. secunda* seed input and seed bank density for the GNB treatment essentially followed the same pattern as for *B. tectorum*. Both species flower and set seed at approximately the same time (Blaisdell 1958; Pyke and Novak 1994); therefore, *P. secunda* is also susceptible to having flower heads removed or trampled during intensive cattle grazing in early and late May, reducing potential seed input into the seed bank. Hoof action also increased seed-soil contact for *P. secunda* seeds or florets (lemma body 4-5 mm long) suspended in litter, enhancing germination. The palatability of *L. perfoliatum* and *S. altissimum* is low for cattle, with some limited use when plants are young (Dittberner and Olson 1983); thus, these species were avoided when they were in the flowering stage during grazing events in May 2005 and 2006. Although some of these annual forbs were trampled during the grazing events, most plants flowered and set seeds in the GNB treatment. The small, smooth, ovate seeds of *L. perfoliatum* (2 mm long X 2 mm diameter) and *S. altissimum* (1mm long X 0.8 mm diameter) allow for good seed-soil contact without trampling (Evans and Young 1970; Young et al. 1970). However, redistribution of litter by hoof action reduced litter depth and exposed small patches of bare soil, creating sites more favorable to both species (Young et al. 1970). In its native range in Russia, *L. perfoliatum* germination is much higher on bare soil than under litter (Volodina 1992). Similarly, *S. altissimum* germination is much higher on bare soil in shrub interspaces than

in litter near shrubs in the Great Basin (Young and Evans 1975). Seed bank densities of *S. altissimum* were not only influenced by seed input from plants that remained on the treatment plots (did not break at the base and tumble), but also from plants that tumbled onto the treatment plots from the GB and NGB treatments and adjacent untreated areas. Dried plants break off at the stem base and tumble great distances, dispersing seeds throughout the fall and winter (Kostivkovsky and Young 2000).

Two consecutive years of late-season prescribed burns (NGB treatment) resulted in initial decreases in *B. tectorum* cover, biomass and density from 2005 to 2006, and further decreases from 2006 to 2007. The first year decrease was likely due to removal of much of the litter bed and seed bank via fire, thus reducing the number of potential safe sites for germination (Evans and Young 1970). The continued decrease in cover, biomass and plant density for *B. tectorum* and the annual forbs (*L. perfoliatum* and *S. altissimum*) from 2006 to 2007, can be attributed to 2007 spring drought conditions. Only 75 mm of rain fell from February through May in 2007, compared to 126 mm in 2006, and 121 mm in 2005.

Prescribed burning altered soil surface characteristics (litter cover and depth, and amount of bare soil), and resulted in an increase in *S. altissimum*. *S. altissimum* has long been recognized as an early colonizer of disturbed areas due to its previously mentioned ability to disperse seeds via tumbling. In addition, seeds germinating on bare soils form a mucilage around the seed when moistened, preventing moisture loss (Evans and Young 1970). *S. altissimum* has been associated with facilitating *B. tectorum* invasion and dominance (Hironaka and Tisdale 1963). Annual forbs add to the litter layer, which facilitates *B. tectorum* establishment and recovery following a fire (Evans and Young

1984). Once *B. tectorum* becomes established under the conditions provided by *S. altissimum*, the latter is readily out-competed for nutrients and water (Daubenmire 1940; Young and Evans 1978).

*P. secunda* responded to the prescribed burns by increasing in cover and biomass from 2005 to 2007. It is generally unharmed by fire, because its small bunch size and sparse litter reduce the amount of heat transferred to perennating buds at or below the soil surface (Kellogg 1985). Wright and Klemmedson (1965) noted that *P. secunda* basal cover increased 2-fold the first year after a fire in a sagebrush-grassland community in southern Idaho. Daubenmire (1975) observed that *P. secunda* competed more successfully than other native perennials with *B. tectorum* as a result of increased tillering following the reduction of litter and improved insolation caused by fire, but he noted that post-fire gains lasted only a few years, after which *B. tectorum* resumed its pre-fire dominance. Young and Evans (1978) also described a similar response after a fire in northern Nevada, where *P. secunda* cover initially increased after a fire, but quickly decreased by 2 years post-fire, indicating that this method is a first step in integrated management.

Prescribed burning in 2005 significantly reduced *B. tectorum* seed density. By burning in October (after seed shatter), fire has the potential to kill/damage much of the seed suspended in the litter bed (Young and Evans 1975; Thill et al. 1984). However, following a late-season burn, *B. tectorum* can rapidly reestablish dominance on a site via increased seed production from plants that develop from seeds protected in the soil (Daubenmire 1975). Seed production can increase from 10-250 seeds per plant prior to a burn, to 960-6000 seeds per plant after a burn, due to lower plant densities and greater

resource availability (Young and Evans 1978). Thus, within 1 year, *B. tectorum* seed density in the burn only treatment (NGB) had recovered to pre-burn conditions. A similar trend was reported by Humphrey and Schupp (2001) in the West Desert of Utah, where an initial drop in seed density the first year after burning was followed by a quick recovery the following year. In a review article, Rasmussen (1994) stated that initial decreases in *B. tectorum* seed densities after burning events are typically followed by a return to pre-burn seed densities conditions within 2 years. Hassan and West (1986) documented a doubling of the *B. tectorum* seed bank 1 year after a fire in a sagebrush community in Central Utah.

As with grazing, *P. secunda* and the annual forbs, *L. perfoliatum* and *S. altissimum*, responded differently to the burn only (NGB) treatment, in terms of seed dynamics. Burning in October reduced the seed bank density of *P. secunda* in 2005 but not in 2006. Fire effects on the seed bank of this grass species are not well documented, but fire may kill some seeds in the upper soil layers. Champlin (1982), using a burning chamber in a mountain big sagebrush community in eastern Oregon, found that *P. secunda* seedling emergence (from seeds in the top 1-2 cm of the soil) was significantly reduced by cool (104° C) and hot (416° C) prescribed burns. The first prescribed burn in October 2005 in our study had a greater fuel load and continuity, and thus was hotter than the second prescribed burn in October 2006. After a fire, both *L. perfoliatum* and *S. altissimum* establish from soil-stored seeds, and *S. altissimum* can also establish from seed dispersed by tumbling plants (Young and Evans 1981; Everett and Ward 1984). Research on fire's impact to the seed banks of these two species is lacking, but the tiny seeds, particularly of *S. altissimum*, can easily fall into fire-safe microsites such as soil

crevices (Young and Evans 1981). Fire is likely to kill some seed, but the overall effect to the seed banks of these two species is probably negligible compared to the effect on the larger, more elongate seeds (florets) of *B. tectorum*, and to some degree *P. secunda*, which have a greater tendency to remain in the litter or on the soil surface.

The combination of grazing and burning (GB) resulted in reduced *B. tectorum* cover, biomass, plant density, and seed input, as with the graze only (GNB) treatment, and reduced seed density, as with the burn only (NGB) treatment. The magnitude of these changes, however, was greater for the GB treatment. The decrease in *B. tectorum* cover, biomass and density was due to a reduction in the seed input via grazing, and the removal of the litter bed and associated seeds via burning. The GB treatment shifted a *B. tectorum*-dominated community with high seed inputs, large seed banks and a deep and contiguous litter bed to a *P. secunda* and *S. altissimum*-dominated community with low *B. tectorum* seed input, almost no *B. tectorum* seed bank, and islands of litter in a matrix of bare soil.

Successional weed management was used as a framework to understand the plant strategies and ecological processes influencing community dynamics on *B. tectorum*-dominated landscapes. The successional management model, proposed by Pickett et al. (1987) and applied to rangeland weed management by Sheley et al. (1996, 2006), identifies three causes of succession (site availability, species availability and species performance), the ecological processes and components primarily responsible for controlling the causes of succession, and the factors that modify those processes and components. For *B. tectorum* invasion and expansion to occur, safe sites (litter, soil cracks and depressions) must be present, propagules (seeds) need to occupy those sites,

and plants must perform successfully in the available sites (Krueger-Mangold et al. 2006). Knowledge about the causes of succession and the associated ecological processes, components and modifying factors can help in identifying methods for reducing *B. tectorum* dominance and creating more desirable plant communities. The graze only (GNB) treatment altered species availability by significantly limiting seed production, and reduced species performance by removing photosynthetic leaf area and restricting regrowth. Targeted grazing alone, however, does not significantly alter litter depth and continuity, and thus the site availability that facilitates *B. tectorum* recovery and dominance. The burn only (NGB) treatment altered site availability by reducing the litter bed and increasing the amount of bare soil, but its impact on species availability and performance was short-lived. By integrating targeted grazing and burning (GB treatment), all three causes of succession were addressed, and the trajectory of a *B. tectorum*-dominated community was changed to a community dominated by less flammable species. This change opens a management window for *B. tectorum*-dominated communities.

## IMPLICATIONS

Grazing and burning alone, and in combination, have the potential to alter *B. tectorum* dominated landscapes but only on limited temporal and spatial scales. The required stocking density for the grazing treatment and the short temporal window of the boot stage limit the use of this technique. The reduction in available forage also limits the use of targeted grazing on a sustained basis. Given the spatial and temporal limits of this treatment, any use must be at a small scale. Targeted grazing could be used to create fuel

breaks to protect remnant native, desirable vegetation, to tie into existing fuel breaks such as roads and streams, or to reduce biomass in existing fuel breaks. The short-lived (1-2 years) effect of targeted grazing indicates that it should be used as a component of an integrated management program rather than a stand-alone method. Targeted grazing could also be used to prepare a site for reseeded, if we assume a threshold seed bank of 330 *B. tectorum* seeds m<sup>-2</sup>, below which reseeded success increases due to decreased competition (Hempy-Mayer and Pyke 2008). However, seed densities in the GNB (1032 seeds m<sup>-2</sup>) and GB (616 seeds m<sup>-2</sup>) treatments are above that threshold after 2 years of targeted grazing.

After using intensive and repeated clipping treatments during the boot stage to reduce *B. tectorum* seed input and seed bank densities in sagebrush-steppe communities in eastern Oregon, Hempy-Mayer and Pyke (2008) questioned the potential of livestock to remove *B. tectorum* competition in preparation for the successful reestablishment of native plants through artificial seeding. They indicated that four questions should be investigated: 1) whether livestock are able to achieve equal or greater seed reductions using the defoliation parameters in their study (clipping plants short at the boot stage, and again 2 weeks later); 2) what the range of environmental conditions would be for this treatment to be effective; 3) whether livestock would be practical for larger-sized projects; and 4) whether defoliation effectiveness could be improved by increasing the defoliation intensity, repeating the defoliation for at least 2 years, and/or using an integrated weed management approach with other control methods such as herbicide application or prescribed burning. Although our study did not specifically address the potential of using cattle to prepare a *B. tectorum*-dominated community for native plant



reseeding, it did address many aspects of the questions above. Intensive grazing was implemented at the boot stage in early-May, and 3 weeks later in late-May, but cattle utilizing 80-90% of *B. tectorum* biomass do not remove every reproductive culm, as can be done with clipping in a small plot study. In terms of environmental conditions, cattle are most effective at removing reproductive tissues when culms are well elongated, as they were in May 2005 and 2006, in response to favorable spring moisture. Defoliation by cattle could be less effective under drought conditions, when little forage is available, and reproductive culms are very reduced in stature. Our demonstration plots were 2/3 ha in size and had a stocking density of 83 cow-calf pairs ha; thus, they provide an indication of the effectiveness of cattle grazing at a larger scale. In order to create fuel breaks or prepare seedbeds on larger areas, an operator would have to consider how to manipulate a large herd or several small herds to defoliate plants twice during the boot stage. This would include consideration of fencing and water requirements, animal movement patterns, and animal behavior at high stocking density for several weeks. By grazing for two consecutive years, we were able to reduce *B. tectorum* seed bank densities further than by grazing for just one year. And, we demonstrate that *B. tectorum* suppression is more effective if integrated with prescribed burning. Thus, our study addressed some of the questions posed by Hempy-Mayer and Pyke (2008); however, further work is required before targeted cattle grazing (and other integrated work) can be effectively used to shift the community and seed dynamics of *B. tectorum*-dominated landscapes.

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**Table 3.1.** Mean values ( $\pm$  SE) for litter depth, and flame length and rate of spread for treatments (GB = graze and burn, GNB = graze and no-burn, NGB = no-graze and burn, and NGNB = no-graze and no-burn) in 2005 and 2006.

Attributes	2005				2006			
	GB	GNB	NGB	NGNB	GB	GNB	NGB	NGNB
Litter depth (cm)	1 $\pm$ 0.1	1 $\pm$ 0.3	2 $\pm$ 0.3	2 $\pm$ 0.5	0.8 $\pm$ 0.1	1.5 $\pm$ 0.4	0.6 $\pm$ 0.2	2.5 $\pm$ 4
Flame Length (m)	0.25 $\pm$ 0.1	___*	2.3 $\pm$ 0.2	___*	0	___*	0.5 $\pm$ 0.2	___*
Rate of Spread (m min <sup>-1</sup> )	7 $\pm$ 6	___*	7 $\pm$ 4	___*	0	___*	7 $\pm$ 4	___*

\* No burn treatment; no data collected

**Table 3.2.** Mean percent cover of *Bromus tectorum* (BRTE), *Sisymbrium altissimum* (SIAL), *Lepidium perfoliatum* (LEPE), *Poa secunda* (POSE), all other forbs combined (other forbs), all other grasses combined (other grasses), soil and litter. These data were collected for three consecutive years at peak biomass in four treatments: graze and burn (GB), graze and no-burn (GNB), no-graze burn (NGB), and a no-graze and no-burn control (NGNB). Upper case letters indicate statistical significance within treatments across years. Lower case letters indicate statistical significance within year across treatments. Fishers Protected LSD ( $p < 0.05$ ).

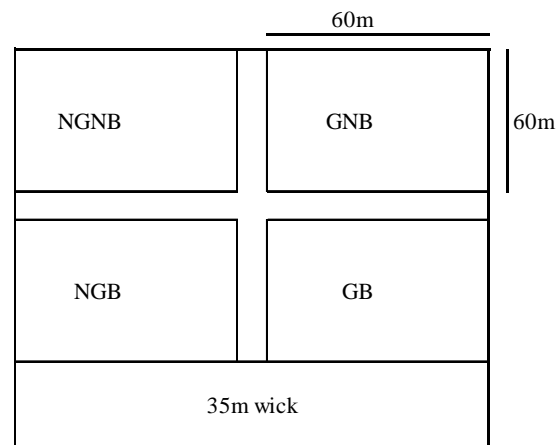
Treatment	Year	BRTE	SIAL	LEPE	POSE	Other forbs	Other grasses	Soil	Litter
GB	2005	33 Ab	6 Aa	15 Aa	1 Cb	4 Aa	0 Ba	31 Ba	10 Ca
	2006	16 Bc	1 Bb	0 Bb	10 Ba	0 Aa	10 Aa	45 Aa	18 Bb
	2007	5 Cc	1 Ba	2 Ba	19 Aa	1 Aa	0 Ba	45 Aa	27 Ab
GNB	2005	34 Ab	3 Ba	3 Bb	0 Bb	1 Aa	0 Aa	42 Aa	17 Ca
	2006	27 Ab	15 Aa	11 Aa	2 Bb	1 Aa	2 Ab	12 Bbc	30 Ba
	2007	25 Bb	2 Ca	2 Ba	7 Ac	0 Aa	0 Aa	19 Bb	46 Aa
NGB	2005	52 Aa	5 Aa	8 Ab	7 Ba	2 Aa	1 Ba	15 Bb	10 Ba
	2006	51Aa	0 Bb	1 Bb	11 Aa	1 Aa	10 Aa	17 Bb	9 Bb
	2007	15 Bb	2 Ba	3 Ba	14 Ab	0 Aa	0 Ba	37 Aa	29 Ab
NGNB	2005	55 Aa	6 Aa	7 Ab	7 Aa	1 Aa	0 Aa	14 Ab	10 Ba
	2006	73 Aa	4 Bb	4 Bb	0 Bb	4 Aa	0 Ab	3 Bc	12 Bb
	2007	40 Ba	1 Ba	1 Ba	1 Bd	1 Aa	1 Aa	5 Ac	50 Aa

**Table 3.3.** Mean biomass (kg ha<sup>-1</sup>) for *Bromus tectorum* (BRTE), *Sisymbrium altissimum* (SIAL), *Lepidium perfoliatum* (LEPE), *Poa secunda* (POSE), all other forbs combined (other forbs), and all other grasses combined (other grasses). These data were collected for three consecutive years at peak biomass in four treatments: graze and burn (GB), graze and no-burn (GNB), no-graze and burn (NGB), and a no-graze and no-burn control (NGNB). Upper case letters indicate statistical significance within treatments across years. Lower case letters indicate statistical significance within year across treatments. Fishers Protected LSD ( $p < 0.05$ ).

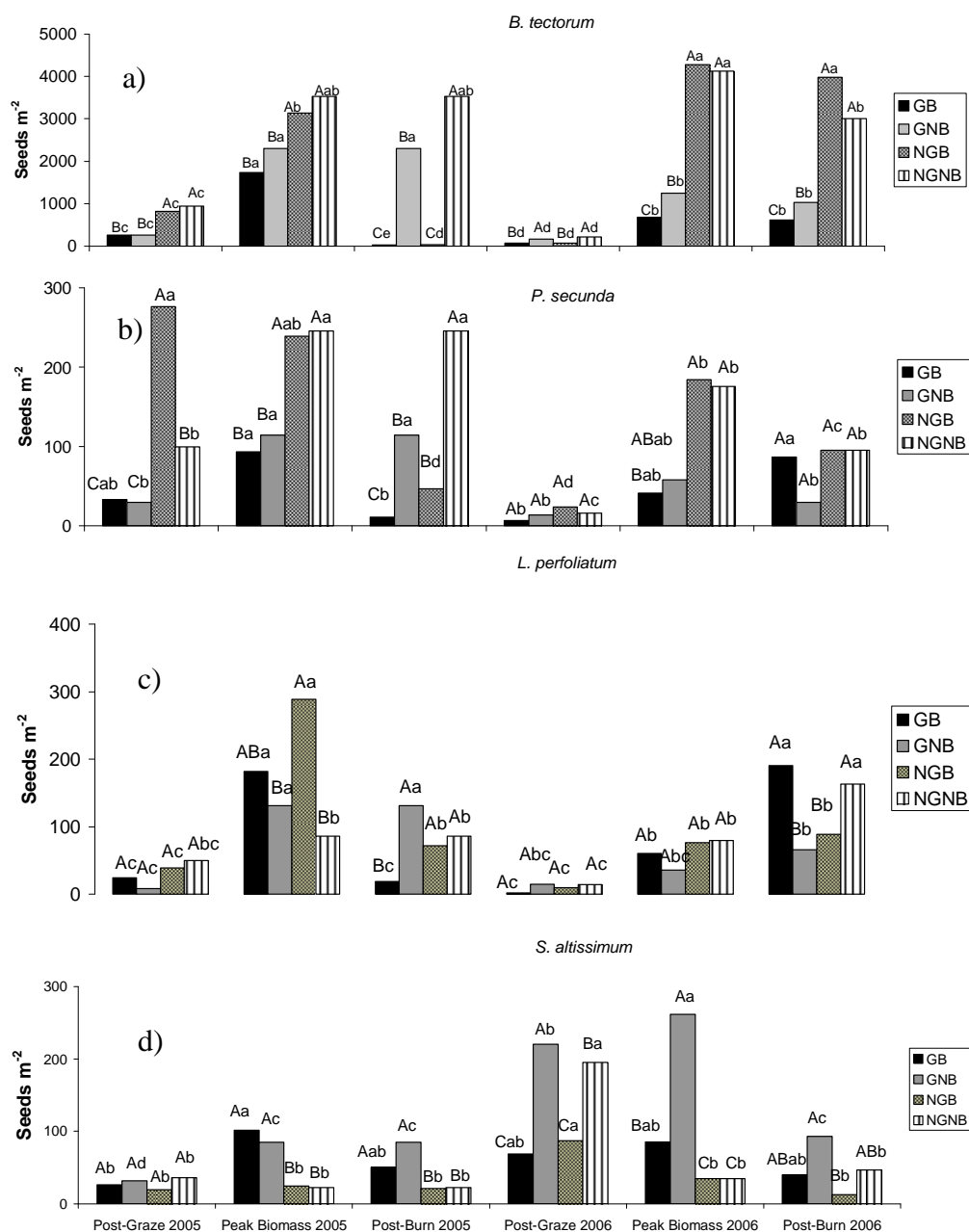
Treatment	Year	BRTE	SIAL	LEPE	POSE	Other forbs	Other grasses	Total
GB	2005	340 Ab	32 Ab	160 Aa	5 Bb	11 Ab	2 Ab	550 Aa
	2006	34 Bc	5 Bb	15 Ba	1 Ba	8 Ab	0 Aa	63 Ba
	2007	15 Bb	33 Ab	6 Ca	48 Aa	15 Aab	7 Aa	125 Ca
GNB	2005	378 Ab	49 Aab	64 Ab	5 Ab	13 Ab	3 Ab	511 Aa
	2006	34 Bc	5 Bb	15 Ba	2 Aa	8 Ab	0 Aa	63 Ba
	2007	80 Bab	43 Aa	10 Ba	2 Ac	6 Ab	11 Aa	154 Cab
NGB	2005	896 Aa	65 Aa	108 Ab	49 Aa	57 Aa	46 Aa	1221 Ab
	2006	101 Bb	3 Cb	6 Ba	2 Ca	17 Ba	2 Ba	131 Bc
	2007	85 Bab	36 Bb	7 Ba	16 Bb	28 Ba	4 Ba	177 Cbc
NGNB	2005	952 Aa	54 Aab	177 Aa	46 Aa	21 Aab	11 Aa	1261 Ab
	2006	578 Ba	13 Ca	9 Ba	2 Ba	8 Bb	5 Ba	616 Bd
	2007	146 Ca	23 Bb	19 Ba	0 Bc	15 Aab	0 Ba	202 Cc

**Table 3.4.** Mean density (plants m<sup>-2</sup>) for *Bromus tectorum* (BRTE), *Sisymbrium altissimum* (SIAL), *Lepidium perfoliatum* (LEPE), all other forbs combined (other forbs), and all other grasses combined (other grasses). These data were collected for three consecutive years at peak biomass in four treatments: graze and burn (GB), graze and no-burn (GNB), no-graze and burn (NGB), and a no-graze and no-burn control (NGNB). Upper case letters indicate statistical significance within treatments across years. Lower case letters indicate statistical significance within year across treatments. Fishers Protected LSD (p< 0.05).

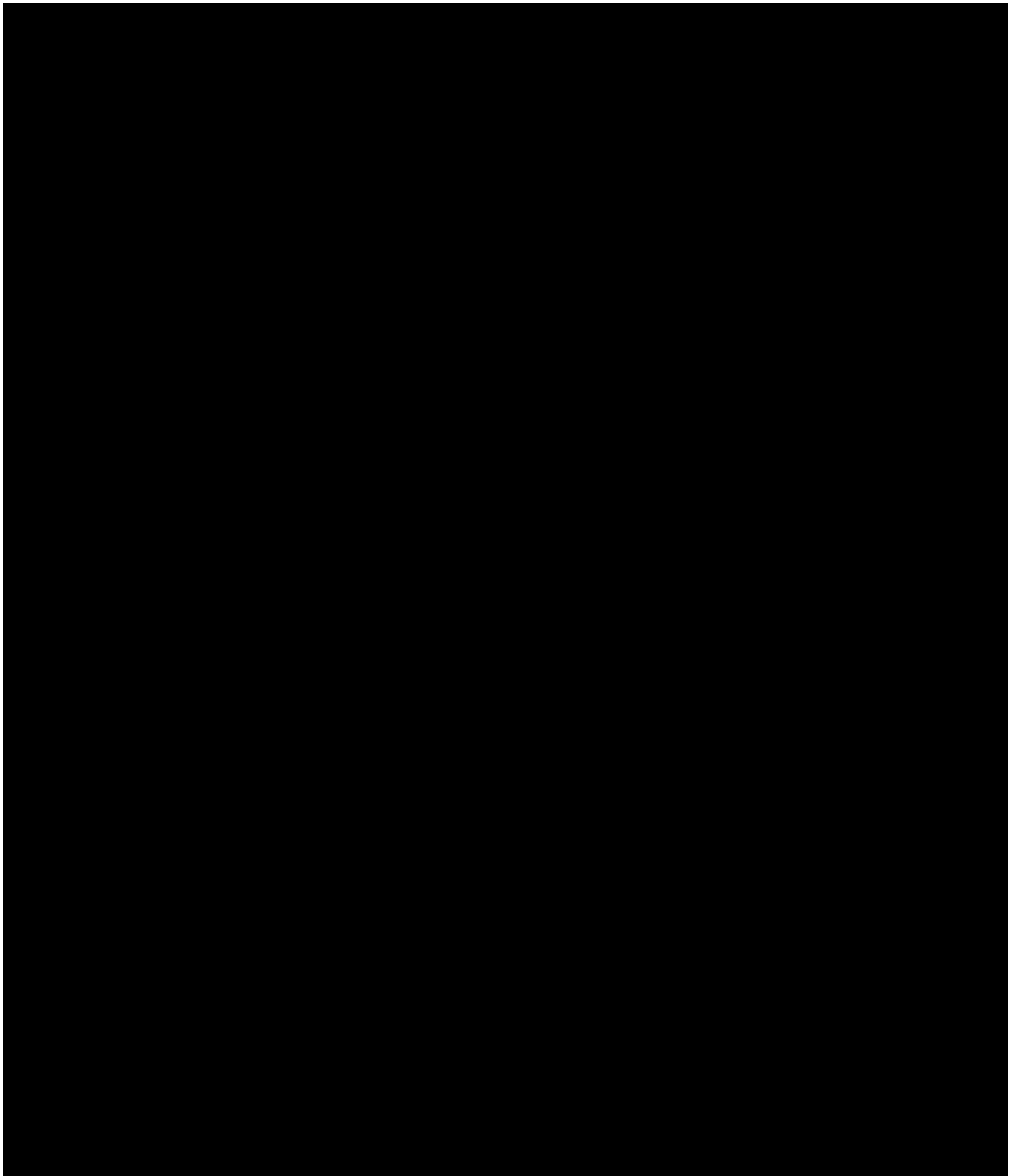
Treatment	Year	BRTE	SIAL	LEPE	Other forbs	Other grasses
GB	2005	138 Ab	6 Aa	22 Aa	2 Aa	0 Aa
	2006	67 Bd	3 Ab	30 Aa	6 Aa	0 Aa
	2007	34 Cc	2 Ab	3 Ba	3 Aa	0 Aa
GNB	2005	131 Ab	8 Ba	13 Aa	1 Aa	0 Aa
	2006	118 Ac	27 Aa	14 Ab	5 Aa	0 Aa
	2007	89 Bb	8 Ba	4 Ba	2 Aa	0 Aa
NGB	2005	349 Aa	10 Aa	19 Aa	10 Aa	1 Aa
	2006	213 Bb	2 Bb	11 Ab	7 Aa	0 Aa
	2007	101 Cb	3 Bb	2 Ba	5 Aa	0 ABa
NGNB	2005	475 Ba	8 Aa	11 Aa	4 Aa	1 Aa
	2006	672 Aa	6 Ab	16 Aab	8 Aa	0 Aa
	2007	216 Ca	2 Bb	5 Ba	6 Aa	0 Aa



**Figure 3.1.** Treatment plot layout within block; southern end of each block has a 35 m *B. tectorum* wick to carry the fire into burning treatments. GB = graze and burn, GNB = graze and no-burn, NGB = no-graze and burn, and NGNB = no-graze and no-burn.

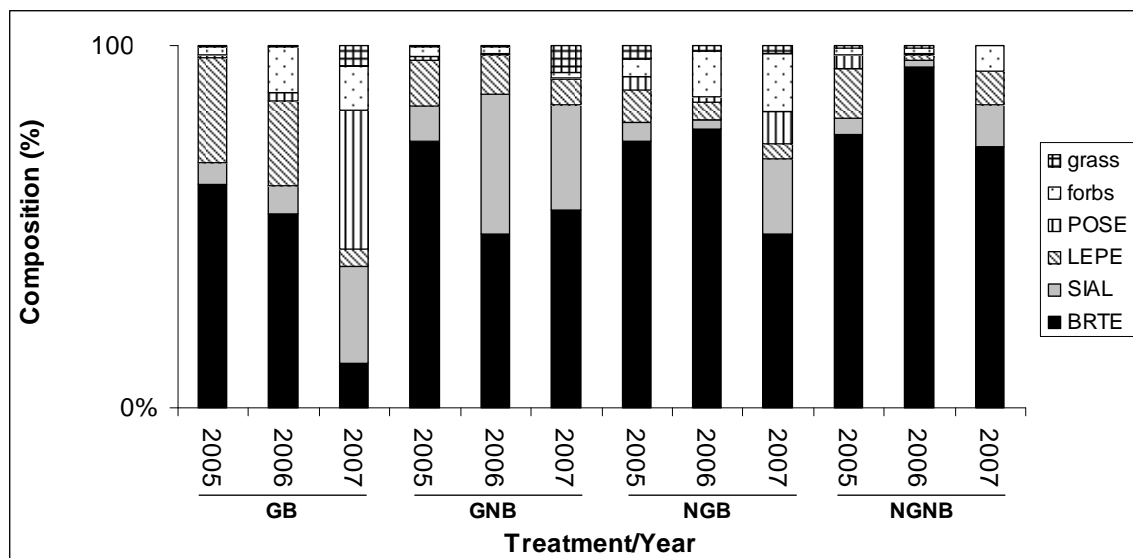


**Figure 3.2.** Seed density for a) *B. tectorum*, b) *P. secunda*, c) *L. perfoliatum* and d) *S. altissimum* for four treatments: graze and burn (GB), graze and no-burn (GNB), no-graze and burn (NGB) and a no-graze and no-burn treatment (NGNB), and six periods: post-graze (May 2005 and 2006), peak-biomass (July 2005 and 2006), and post-burn (October 2005 and 2006). Capital letters indicate mean comparisons across treatments within time period, and lower case letters indicate mean comparisons across time periods within treatment. Means with different letters are significantly different ( $P < 0.05$ ).



**Figure 3.3.** *B. tectorum*, *P. secunda* and *L. perfoliatum* seed input in 2006 for four treatments: graze and burn in 2005 and graze 2006 (GB), graze in 2005 and 2006 and no-burn (GNB), no-graze and burn in 2005 (NGB) and a no-graze and no-burn control (NGNB). Means with different letters are significantly different ( $p < 0.05$ ).





**Figure 3.4.** Community composition based on proportion of total biomass for *Bromus tectorum* (BRTE), *Sisymbrium altissimum* (SIAL), *Lepidium perfoliatum* (LEPE), *Poa secunda* (POSE), all other forbs combined (forbs) and all other grasses combined (grass). These data were collected for three consecutive years at peak biomass in four treatments: graze and burn (GB), graze and no-burn (GNB), no-graze and burn (NGB) and a no-graze and no-burn control (NGNB).

## CHAPTER 4

COST EFFECTIVENESS OF FUEL BREAK TREATMENTS IN CHEATGRASS-  
DOMINATED LANDSCAPES<sup>4</sup>

*Abstract.* The increase in size and frequency of wildfires in the northern Great Basin is related to the increase in cheatgrass (*Bromus tectorum*)-dominated plant communities. To address these changes in wildfire characteristics, federal wildland fire policy recommends research on, and economic analysis of, fuel reduction treatments. The objective of this research was to compare the cost effectiveness of using Plateau<sup>®</sup> (imazapic) herbicide and targeted cattle grazing to create a fuel break on a *B. tectorum*-dominated landscape. We used the fuel characteristics measured for a targeted cattle grazing study south of McDermitt, Nevada and a Plateau<sup>®</sup> study south of Kuna, Idaho to parameterize fire behavior models. Wildfire rate of spread and flame length were simulated for peak fire conditions using the BEHAVE Plus fire modeling system. Cost-effectiveness analysis was used to compare the relative costs and outcomes of a 3-year application of the two fuel reduction treatments under five fuel loading scenarios. Targeted cattle grazing and Plateau<sup>®</sup> treatments had similar reductions in flame length and rate of spread. Cattle grazing had high fixed costs (primarily fencing), and was more cost-effective than application of Plateau<sup>®</sup> under all scenarios except for three consecutive years of low fuel loads.

<sup>4</sup> Coauthored by J.M. Diamond, C.A. Call, and N. McCoy.

## INTRODUCTION

The 1995 Federal Wildland Fire Management Policy and Program Review report (updated, USDI and others 2001) encourages a more proactive approach to reduce the threat of catastrophic wildfires on rangelands in the western USA. The report states that strategic landscape-scale fuel management will require the integration of a variety of treatment methods (chemical and biological), and recommends research on fuel reduction alternatives and their economic viability.

Cheatgrass (*Bromus tectorum*) is an invasive annual grass that originated in Eurasia and is now dominant on many sagebrush-grassland communities in the western USA (Mack 1981). Invasion has set in motion a grass/fire cycle where *B. tectorum* provides the fine fuel necessary for the initiation and propagation of fire (D'Antonio and Vitousek 1992). *B. tectorum* recovers more rapidly than native species and facilitates more frequent and larger fires (D'Antonio and Vitousek 1992). Often, fires that ignite in *B. tectorum*-dominated communities spread readily into desirable communities such as remnant sagebrush-grasslands (Whisenant 1990; Davison 1996). To maintain these remnant patches of native vegetation, the connectivity with *B. tectorum* dominated sites must be reduced (Bunting et al. 1987; Davison 1996). The creation of fuel breaks with livestock grazing or herbicides may be an effective way to not only protect remnant native vegetation but also prepare seed beds for revegetation (Davison 1996). The economic impacts of these changes, in terms of fire suppression and rehabilitation are great, and thus, management techniques must be evaluated for their cost effectiveness (Pellant and Hall 1994; Knapp 1996).

Recent research in the northern Great Basin has shown that targeted cattle grazing in spring can reduce *B. tectorum* cover, biomass, and density (see Ch. 3). Targeted grazing at the most susceptible phenological stage (boot stage) can remove biomass and reduce subsequent regrowth of *B. tectorum* during the remainder of the growing season (Olson and Richards 1989; Vallentine and Stevens 1994). Cattle readily consume *B. tectorum* at the boot stage because it is at peak nutritional and caloric value (Murray et al. 1978; Thill et al. 1984). Calves 1-5 months old gain an average of 0.91 kg daily when nursing from cows on a spring diet of *B. tectorum* (Murray et al. 1978; National Research Council 1984; Mayland et al. 1992). Thus, cattle readily use *B. tectorum* in the spring when it meets their nutritional requirements and intensive grazing has the highest potential to suppress *B. tectorum* (Olson and Richards 1989; Mayland et al. 1992; Vallentine and Stevens 1994).

The herbicide Plateau<sup>®</sup> (Imazapic) can also suppress *B. tectorum* cover, biomass, and thus fuel loads (Kury et al. 2002). Plateau<sup>®</sup> is an amino acid inhibitor that leads to cessation of meristem growth and decreased root growth, leading to plant death. Plateau<sup>®</sup> application prior to seedling emergence is the most effective method of application (Kury 2002). Plateau<sup>®</sup> can have a residual effect on *B. tectorum* for up to 2 years following treatment, depending on the rate of application (Davison and Smith 2007). Herbicide treatments can reduce fuel build-up in *B. tectorum*-dominated landscapes, but they can be costly and have real or perceived effects on environmental quality, such as residue build-up in soil or leaching into adjacent water sources (Vallentine 1989; Trainor and Bussan 2001).

While both grazing and herbicides have the potential to create and maintain fuel

breaks, the economic viability of the treatments has yet to be compared. An emerging method for analyzing the effect of alternative fuel treatments is cost-effectiveness analysis. It was developed in the military and first applied to health care in the mid-1960s. More recently, this analysis method has been used to assess the economic viability of alternative practices in several natural resource management programs, including stream restoration (Frimpong et al. 2006), threatened animal conservation (Cullen 2005), forest plantation management (Dampier et al. 2006; van Landingham et al. 2008), and forest fuel reduction (Rideout et al. 1999). Cost-effectiveness analysis allows scientists and managers to select treatments which provide the maximum effectiveness per level of cost or which require the least cost per level of effectiveness (Levin and McEwan 1992). Effectiveness can be measured as the percent change of a common variable (Kline 2004). Rideout et al. (1999) suggested that alternative fuel reduction treatments could be evaluated for effectiveness with percent reduction of flame length, and that treatments could be compared with a high degree of inference (Rideout et al. 1999). Cost-effectiveness analysis has yet to be utilized to evaluate fuel reduction treatments on rangelands.

The objective of this research was to compare the cost effectiveness of using Plateau<sup>®</sup> herbicide and targeted cattle grazing to create fuel breaks on *B. tectorum*-dominated landscapes. Applications of the two fuel reduction treatments were simulated along a potential fuel break at the interface of a *B. tectorum* community and a native sagebrush-grassland community.

## METHODS

*Fuel Break Treatments*

The grazing and herbicide treatments compared in this study are based on findings from targeted cattle grazing (see Ch. 2) and Plateau<sup>®</sup> (Kury et al. 2002) studies conducted in the northern Great Basin. The grazing study site is located within the Bureau of Land Management (BLM) Quinn River Management Area, 20 km southeast of McDermitt, Nevada (E. 455618, N. 4641643). It is on a 5% slope at 1400 m elevation. Average annual precipitation is 228 mm, much of which falls as snow from November through March. The predominant soil is a McConnel fine sandy loam (sandy-skeletal, mixed mesic Xeric Haplocambid). The soil corresponds to the Loamy ecological site in the 200-350 mm precipitation zone. The site has 50-60% *B. tectorum* cover. Other species include tumble mustard (*Sisymbrium altissimum*) clasping pepperweed (*Lepidium perfoliatum*) and Sandberg bluegrass (*Poa secunda*). Four grazing-burning treatments (graze and no-burn, graze and burn, no-graze and burn, and no-graze and no-burn) were arranged in a split-plot design in a block, and replicated three times. Treatment plots were 60 X 60 m. Fire behavior from the graze and burn and the no-graze and burn treatments were used for cost-effectiveness analysis. The graze and burn treatment plots were intensively grazed (equivalent of 83 cow-calf pairs ha<sup>-1</sup>) during the boot stage (inflorescence emergence from the leaf sheath) of *B. tectorum* in early May 2005. The plots were grazed to 80-90% removal of aboveground biomass over a 32-40 h period. Cool temperatures and frequent precipitation promoted regrowth and additional germination of *B. tectorum*, so intensive grazing (same duration and stocking density) was repeated in late May to

maintain 80-90% removal of aboveground biomass. The graze and burn and no-graze and burn treatments were burned in mid-October 2005 to assess the effects of fuel reduction on flame length and rate of spread. On the day of the burn, climatic and fuel variables were recorded to enable an accurate prediction of fire behavior for the fuel type, fuel loading, fuel moisture, fuel bed depth, and weather conditions.

The herbicide study site is located within the BLM-administered portion of the Snake River Birds of Prey National Conservation Area, 13 km south of Kuna, Idaho (E. 546985, N. 4815734). It is on a 0-4% slope at 900 m elevation. Annual average precipitation is 246 mm, much of which falls as snow from November-February. The predominant soil is a Power silt-loam (fine-silty, mixed superactive, mesic Xeric Calciargid). Plant species composition was dominated by *B. tectorum* (up to 90% on some areas of the study site), and secondary species included *S. altissimum*, *L. perfoliatum* and *P. secunda*. In November 1999, Plateau<sup>®</sup> herbicide was applied at seven rates (0, 146, 292, 437, 585, 731 and 877 mL ha<sup>-1</sup>) in a randomized block design, replicated three times. Treatment plots were 3 X 15 m. The 437 mL ha<sup>-1</sup> rate resulted in maximum *B. tectorum* reduction (80-90%) with minimum product use (Kury et al. 2002).

### *Fire Behavior Modeling*

Since the plots in the targeted grazing study were burned in October after peak fire season (July-August), and the plots in the Plateau<sup>®</sup> study were not burned, fire behavior in both studies was simulated for peak fire conditions using the BehavePlus 3.0 fire modeling system (Andrews et al. 2003). Fuel models were created by substituting fuel parameters (fuel load, surface area:volume ratio, fuel bed depth, heat content, and

extinction moisture) into the low load, dry climate (GR2 dynamic) fuel model for the graze and burn treatment and the 437 mL ha<sup>-1</sup> Plateau<sup>®</sup> treatment (Scott and Burgan 2005). The high load, dry climate grass (GR7 dynamic) fuel model (Scott and Burgan 2005) was used for the no-graze and burn treatment and the 0 mL ha<sup>-1</sup> Plateau<sup>®</sup> treatment, both of which served as controls. These models were selected due to their similarities to actual fuel conditions within each treatment, and they represent a reduction in fuel from a high to a low biomass level. An additional model, the moderate load, dry climate grass (GR 4) fuel model (Scott and Burgan 2005) was used to represent a reduction in fuel from a moderate to a low (GR2) biomass level. And the low load, dry climate grass (GR2) model was used to represent a reduction in fuel from a low to a very low biomass level. Flame lengths and rates of spread were then simulated for peak fire conditions (2% fuel moisture and a 20 km h<sup>-1</sup> wind speeds) in the northern Great Basin.

#### *Cost-effectiveness Analysis*

An area adjacent to the grazing study plots southeast of McDermitt, Nevada was used as a model site to simulate a fuel break created by targeted cattle grazing and Plateau<sup>®</sup> herbicide. The fuel break runs along the interface of a *B. tectorum*-dominated community and a remnant sagebrush-grassland community, and is 100 m wide by 4.2 km long, with a perennial stream at each end and an access road along its entire length (Fig. 4.1).

The costs for each treatment were estimated based on 2008 market prices (Table 4.1). The herbicide treatment cost estimates were for a fall (September) application of Plateau<sup>®</sup> at 437 mL ha<sup>-1</sup> with a helicopter, resulting in a total volume of 18.35 L of



Plateau<sup>®</sup> applied on the entire fuel break (Table 4.2). Unlike the Plateau<sup>®</sup> treatment, the grazing treatment requires fencing to contain the cattle at the proper stocking density. For this simulation, a 4-strand barbed wire fence already existed at the interface of the *B. tectorum*-dominated community and the remnant sagebrush-grassland community. Grazing paddocks were created along the fuel break by constructing electric fence (two-strand, Flash cable perimeter and single-strand interior) on three sides and using the existing barbwire fence as the fourth side. Material costs were financed on a 5-year amortization schedule at a 6% interest rate (Table 4.3). To facilitate intensive grazing for a relatively short period of time during the *B. tectorum* boot stage, the grazing treatment was arranged in six replicate blocks, with each block consisting of two, 3.5 ha paddocks (Fig. 4.1). All six blocks were grazed simultaneously with separate groups of cattle. Cattle graze in each paddock for 1 week to achieve 80-90% biomass removal, resulting in approximately 2 weeks of grazing per block. Cattle are then moved back to the first paddock in each block to repeat another 2-week grazing cycle to utilize regrowth and maintain 80-90% biomass removal. Thus, targeted grazing occurs over a period of 4 weeks (during May). Drop gates (part of the interior single-strand electric fence) facilitate cattle movements between paddocks. A portable water tank (1200 L) is moved with the cattle to each paddock. The water tank can be filled as frequently as every 2.5 hours for a total of ~5,300 L day<sup>-1</sup> to meet the water requirements for up to 83 cow-calf pairs ha<sup>-1</sup> (64 L lactating cow<sup>-1</sup> day<sup>-1</sup>) (Lardy and Stoltenow 1999). A 4000-L water truck is used to deliver water from the perennial streams at each end of the fuel break to paddocks in all six blocks from 0800 h to 1900 h each day.

Since *B. tectorum* aboveground biomass production can vary from year to year

with fluctuations in climatic conditions, the grazing treatment and Plateau<sup>®</sup> treatment costs were estimated for high (950 kg ha<sup>-1</sup>), moderate (580 kg ha<sup>-1</sup>), and low (150 kg ha<sup>-1</sup>) biomass levels, which were documented at the grazing study site in 2005, 2006, and 2007, respectively. We regressed cold season (Sept-May) precipitation with the biomass recorded in 2005, 2006 and 2007. We then used this regression to produce an index of biomass production based on 30 years of precipitation data from the McDermitt, Nevada weather station. Precipitation > 25 cm was categorized as high biomass, between 25 and 18 cm as moderate and < 18 cm as low. We used these categories to produce a probability analysis of biomass in year 1 to year 2 and year 3. Five high probability biomass scenarios were created from this analysis: scenario one high biomass in year 1, moderate in year 2, and low in year 3; scenario two was moderate, low, and moderate biomass; scenario three was moderate, moderate, and low biomass; scenario four was low, moderate, and low biomass; and the final scenario was low biomass in years 1, 2, and 3. In order to remove 80-90% of *B. tectorum* biomass in 1 week in a paddock, stocking densities for the grazing treatments would be 83, 51 and 13 cow-calf pairs per paddock, respectively, for the high, moderate, and low biomass levels.

The grazing treatment and the Plateau<sup>®</sup> treatment, for the five biomass scenarios, were used to create and maintain fuel breaks over a 3-year period, where targeted cattle grazing occurs in May of year 1 with follow-up grazing in years 2 and 3, and Plateau<sup>®</sup> is applied in September of year 1 with a follow-up application in September of year 3. Both the grazing and Plateau<sup>®</sup> treatments evaluated fire behavior only with the high biomass level, thus, the remaining two levels (moderate and low) were modeled based on an 85% biomass reduction at each retreatment.

We used percent reduction in flame length and rate of spread to evaluate the cost effectiveness of the Plateau<sup>®</sup> application and targeted cattle grazing. The United States Office of Management and Budget (OMB) recommends a discount rate of 4%, and other natural resource management studies have utilized a rate as low as 3% and as high as 7% (Aldrich et al. 2005; Frimpong et al. 2006). These rates were considered to be inappropriate for this simulation because it involves both federal and non-federal resources (e.g. private livestock). Thus, a 6% discount rate is a more appropriate representation of the opportunity costs associated with using privately owned resources. We used discounted costs (C) to calculate the present value (PV) of each treatment scenario; retreatment was simulated for every year for 3 years. PV is described by the formula:

$$PV = \sum_{t=1}^3 \frac{C_t}{(1 + 6\%)^t}$$

The grazing treatment creates an output (cattle gain); thus, we evaluated this gain along with the costs. Cattle gains were based on a mean daily calf weight gain of 0.91 kg (Murray et al. 1978; National Research Council 1984; Mayland et al. 1992). The cattle herd used in this analysis is managed as a cow-calf operation that sends all calves to market at 8 months of age to be sold as feeders. We assumed the forage typically used by these cattle was utilized by other cattle outside of the treatment, and breeding stock and the support facilities and associated equipment already exist. Mean market value per hundred weight (CWT) (45 kg) over the last 19 years was \$97.94 (AMS-USDI 2008). A total of 498, 304 and 79 cow-calf pairs were used to treat our three biomass levels (high,

moderate, and low), respectively. Each of those calves was assumed to gain 0.91-kg day<sup>-1</sup> for the 30 days of the treatment, for a total gain of 27.3 kg. Mean calf gains were 61% of CWT, the relationship is described by the formula:

$$CV = (\# \text{ cattle } ) \times (\text{Cost per CWT}) \times (\text{Percentage cattle gain})$$

Income from cattle gain (CV) was then subtracted from the cost (C) of the grazing treatment prior to the discount rate calculation. Cost-effectiveness analysis was performed in Microsoft<sup>®</sup> Excel.

## RESULTS

The grazing treatment and Plateau<sup>®</sup> treatment were similar in their suppression of flame length and rate of spread under peak fire conditions (Figs. 4.2 and 4.3). Grazing and Plateau<sup>®</sup> reduced flame length by 94% at the high biomass level, 72% at the moderate level, and 33% at the low level, and rate of spread was reduced by 94, 70 and 21%, respectively, at the high moderate and low biomass levels (Figs. 4.2 and 4.3).

Animal weight gains led to profits in four of the five biomass scenarios for the grazing treatment: high-moderate-low (\$24,153.91), moderate-low-moderate (\$13,303.96), moderate-moderate-low (\$13,303.96), and low-moderate-low (\$2,086.92), while the low-low-low biomass scenario resulted in net costs (\$9,092.15) (Table 4.4). The high-moderate-low scenario had nearly twice the profit of the moderate-low-moderate and moderate-moderate-low scenarios, and 12 times the profit of the low-moderate-low scenario. Profits from the high-moderate-low scenario were 2 times the total cost of the grazing treatment. The moderate-low-moderate and moderate-moderate-

low scenarios resulted in cattle weight gain profits 1.5 times greater than treatment costs. Cattle weight gains for the low-moderate-low biomass scenario offset 1.2 times the treatment costs. The low-low-low grazing scenario offset only 60% of the project costs (Table 4.4). The Plateau<sup>®</sup> treatment was more cost-effective than the targeted grazing treatment for the low-low-low scenario (Table 4.4).

## DISCUSSION

Both Plateau<sup>®</sup> and targeted cattle grazing are effective treatments for reducing wildfire flame length and rate of spread. The reduction in flame length from above 4 m to below 1 m reduces the fire suppression resources required to manage wildfires. A fire with flame lengths < 1.2 m can typically be fought with fire crews using hand tools; one with flame lengths of 1.2-2.4 m requires equipment such as plows, dozers and pumps; and a fire with flame lengths > 2.4 m typically requires fire retardant drops from aircraft (Pyne et al. 1996). Simulations show that targeted cattle grazing and Plateau<sup>®</sup> reduced flame lengths well below 1.2 m at high, moderate and low biomass levels, and offered some protection to the adjacent sagebrush-grassland community.

Since targeted cattle grazing and Plateau<sup>®</sup> treatments essentially provide the same level of fuel reduction and change in fire behavior, cost-effectiveness analysis can be used to select the treatment which requires the least cost per level of effectiveness (Kline 2004). Targeted cattle grazing was more cost-effective than Plateau<sup>®</sup> except when fuel loads were low in all three years. This reflects how the market value of calf weight gains over the 30-day grazing period offsets the costs associated with fencing and animal maintenance. Calf weight gains offset all costs associated with the grazing treatment in

all scenarios except the low-low-low biomass scenario. Thus, weight gains from only 13 cow/calf pairs year<sup>-1</sup> under the low-low-low biomass scenario do not adequately cover the costs associated with implementing the grazing treatment. The Plateau<sup>®</sup> treatment is only more cost-effective than the grazing treatment for the low-low-low biomass scenario.

The primary costs of grazing are in the initial fencing while the secondary costs of managing the cattle during the grazing period (watering and maintaining fencing) were only 1/10 of the overall spending. The costs of the Plateau<sup>®</sup> treatment do not change with future reapplication. If fuel break re-treatment is necessary, the cost of the treatment will be comparable to 100% of initial Plateau<sup>®</sup> costs. In contrast, the grazing re-treatment requires only the secondary costs.

Our cost-effectiveness analysis indicates that Plateau<sup>®</sup> is more cost-effective than targeted cattle grazing when fuel loads are low for three consecutive years. Plateau<sup>®</sup> can persist in the soil for 120 days or more, impacting germinating seeds of annual grasses (Kyser et al. 2007), and it can reduce the biomass of *B. tectorum* for at least 2 years after application (Davison and Smith 2007). In order to reduce the existing *B. tectorum* seed bank and future seed input (see Chapter 3), Plateau<sup>®</sup> would probably have to be applied every other fall for at least 3 years. Based on findings in Chapter 3, targeted cattle grazing would also have to be implemented every May for several years. Cost-effectiveness comparisons for the two fuel reduction treatments across multiple years indicate that grazing is more cost-effective than Plateau<sup>®</sup> when fuel loads are not low. Cattle have been grazing *B. tectorum* as a primary spring forage in the Great Basin for many decades as part of traditional grazing management programs. With much of the grazing

infrastructure in place, cattle operators could intensively graze *B. tectorum*-dominated areas when needed, as part of a more flexible grazing management plan.

Cost-effectiveness analysis allowed us to evaluate *B. tectorum* fuel reduction treatments that would provide some measure of protection for an adjacent sagebrush-grassland community, which is difficult to monetize. It is impractical to try to estimate a dollar value for protecting the biodiversity or wildlife habitat attributes of a native plant community. In management situations where the benefits of fuel reduction treatments are difficult to measure, effectiveness proxies, such as fuel loading changes, flame length and rate of spread provide a measure of effectiveness that can be used in a cost-effectiveness analysis (Rideout et al. 1999).

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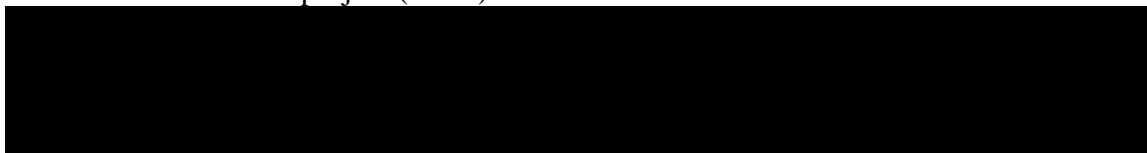
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**Table 4.1.** Itemized costs of targeted cattle grazing treatment for 3.5-ha paddocks, 7-ha blocks and the total project (42 ha). Costs for materials are based on 2008 MSRP. Labor costs are derived from personnel communications with federal contractors.

<b>Item</b>	<b>Unit cost</b>	<b>Paddock</b>	<b>Block</b>	<b>Total project</b>
Two employees, 4 h paddock <sup>-1</sup> (fence installion)	\$13.60	\$108.80	\$217.60	\$1,305.60
1000 m of Flash cable	\$0.23	\$230.00	\$1,610.00	\$9,660.00
63 "t" posts paddock <sup>-1</sup> @ 20' m intervals	\$8.00	\$504.00	\$1,008.00	\$6,048.00
99 line posts - 3/8" @ 5 m intervals	\$1.29	\$127.71	\$255.42	\$1,532.52
3 Grounding rods	\$9.00	\$13.50	\$27.00	\$162.00
Wire Splicer tool and 100 sleeves	\$18.22			\$18.22
2 insulators clips post <sup>-1</sup>	\$0.15	\$18.90	\$37.80	\$226.80
1200 L water tank	\$500.00	\$250.00	\$500.00	\$3,000.00
Electric fence charger	\$60.00	\$30.00	\$60.00	\$360.00
12v battery	\$100.00	\$50.00	\$100.00	\$600.00
<b>Sub-total for project set up</b>	<b>N/A</b>	<b>\$1,332.91</b>	<b>\$3,815.82</b>	<b>\$22,913.14</b>
One employee 40 week <sup>-1</sup> for 1 month (grazing and watering)	\$13.60	\$51.81	\$362.67	\$2,176.00
75 L fuel week <sup>-1</sup> @ \$1.05 L <sup>-1</sup> (water truck)	\$320.00	\$7.62	\$53.33	\$320.00
<b>Sub-total for project during grazing</b>	<b>N/A</b>	<b>\$59.43</b>	<b>\$416.00</b>	<b>\$2,496.00</b>
<b>Total</b>	<b>N/A</b>	<b>\$1,392.34</b>	<b>\$4,231.82</b>	<b>\$25,409.14</b>

**Table 4.2.** Itemized costs (BASF 2008) of Plateau® treatment for 1-ha paddocks, 7-ha blocks and the total project (42 ha).

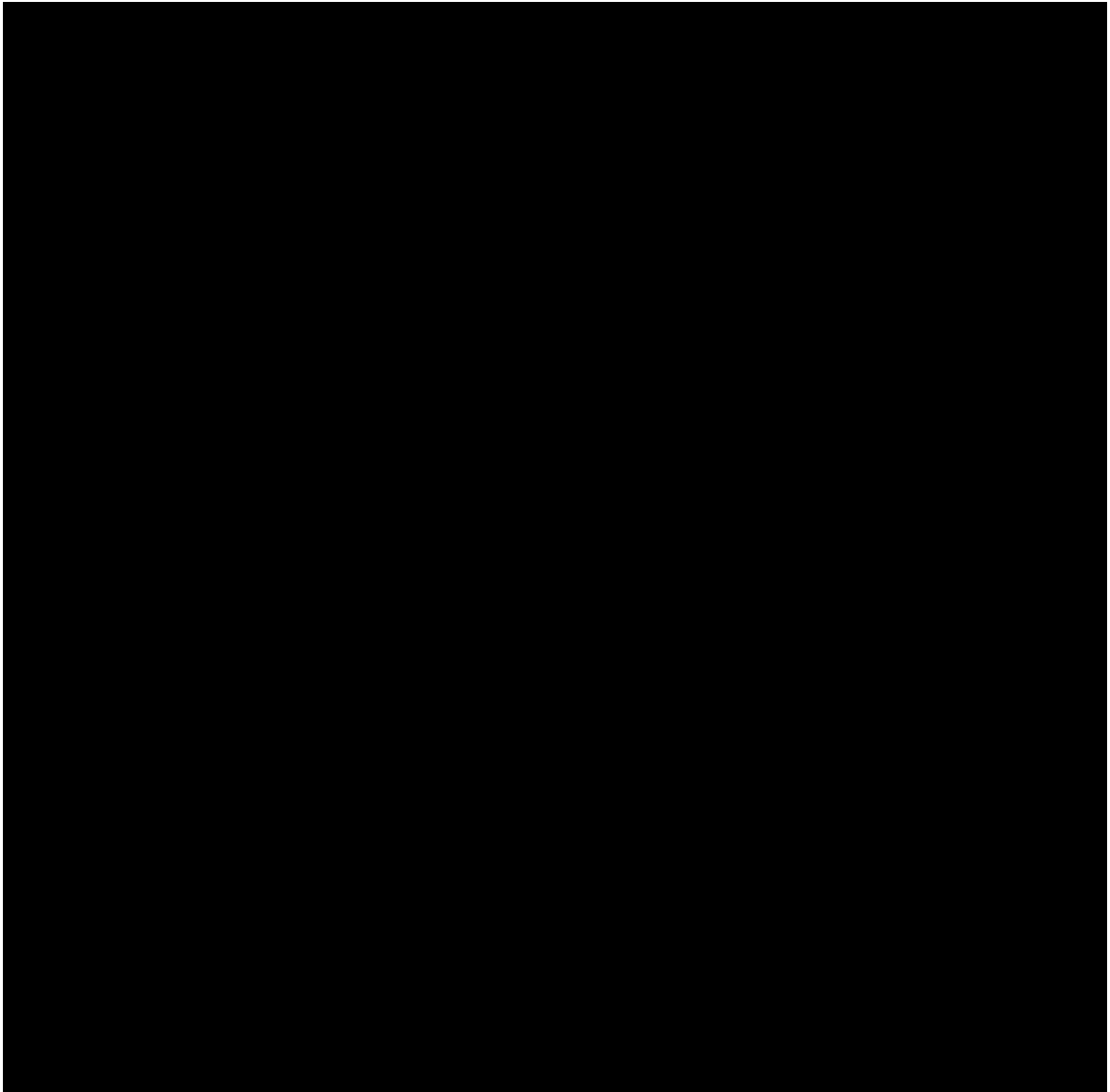


**Table 4.3.** Amortization schedule for grazing costs financed across five years.

Loan Amortization Schedule	
Loan amount	\$22,913.14
Interest rate	6.00%
Loan period in years	5
Monthly cost	\$442.98
Annual cost	\$5,315.70
Total interest	\$3,665.37
Total cost	\$26,578.51

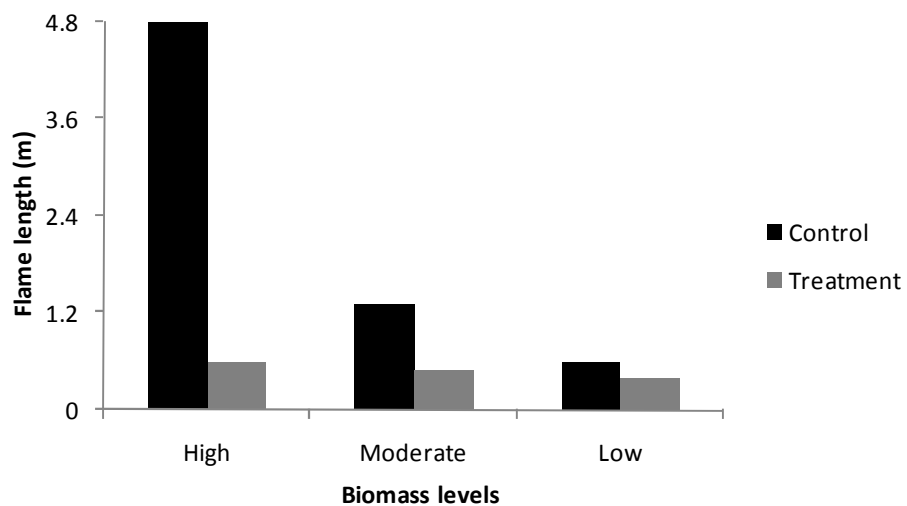
**Table 4.4.** Cost-effectiveness analysis for multiple treatments of targeted cattle grazing or Plateau<sup>®</sup> under five fuel loading scenarios across three years; first scenario is high year 1 moderate year 2 and low year 3. Second scenario is moderate, low moderate. Third scenario is moderate, moderate low. Fourth scenario is low moderate low. And the final scenario is low, low, and low. Negative values indicate cost-effectiveness that exceeds input costs and produces profit.

Fuel reduction	Grazing	Plateau <sup>®</sup>
Discounted cost, high-Moderate-Low biomass levels	-\$24,153.91	\$3,769.56
Discounted cost, Moderate-Low-Moderate biomass levels	-\$13,303.96	"
Discounted cost, Moderate-Moderate-Low biomass levels	-\$13,303.96	"
Discounted cost, Low-Moderate-Low biomass levels	-\$2,086.92	"
Discounted cost, Low-Low-Low biomass levels	\$9,092.15	"

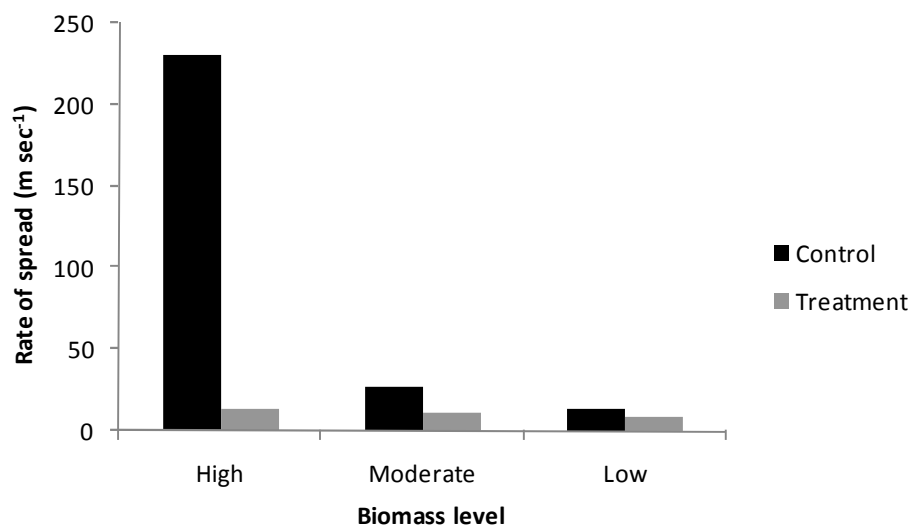


**Figure 4.1.** Grazing treatment layout consists of two 3.5-ha paddocks bordered by an existing barbed wire fence on the east side and electric fence on the other three sides. The electric fence separating the paddocks has a 5 m drop gate adjacent to the barbed wire fence to allow cattle to be moved between paddocks. Six replicate blocks are laid out end to end along the existing barbed wire fence connecting to roads and streams at each end for a total length of 4.2 km and a total area of 42 ha.





**Figure 4.2.** Modeled flame length (m) prior to grazing or Plateau<sup>®</sup> application (control), compared to the flame length following a fuel load reduction (treatment) of 85%. The comparison of the control to the treatment was used to calculate the percent reduction for three scenarios; [high moderate and low biomass levels (fuel load)].



**Figure 4.3.** Modeled rate of spread (m sec<sup>-1</sup>) prior to grazing or Plateau<sup>®</sup> application (control), compared to the rate of spread following a fuel load reduction (treatment) of 85%. The comparison of the control to the treatment was used to calculate the percent reduction for three scenarios; [high moderate and low biomass levels (fuel load)].

## CHAPTER 5

## SYNTHESIS

The degradation of native sagebrush grasslands in the northern Great Basin is due primarily to overgrazing by domestic livestock, the associated invasion of *B. tectorum*, and the resulting grass/fire cycle (Young et al. 1987; Whisenant 1990; Miller et al. 1994). *Bromus tectorum*-dominated communities burn more often and at a higher intensity than native plant communities (Platt and Jackman 1946; Pyne et al. 1996). These short return intervals and high intensity burns reduce the potential for shifts in trajectories away from *B. tectorum*-dominated rangelands. While grazing and burning are strongly associated with the decline of sagebrush grasslands (Costello 1944; Piemeisel 1951; Miller et al. 1994), they can be used as tools for community recovery. Burning and grazing management strategies, to be effective, require an understanding of fire behavior, community dynamics and treatment feasibility.

The overall goal of this research project was to determine the effectiveness of using cattle and prescribed burning as tools to reduce fire hazards and *B. tectorum* dominance on rangelands in the northern Great Basin. Complementary field studies were conducted in northern Nevada during 2005, 2006 and 2007 to determine: 1) the effects of targeted cattle grazing and prescribed burning on fire behavior, 2) the impact of targeted cattle grazing and prescribed fire on the seed dynamics of *B. tectorum* and associated species, 3) the effects of targeted cattle grazing and prescribed fire on aboveground community dynamics, and 4) the economic effectiveness of using targeted cattle grazing and herbicide to create fuel breaks.

Results from the fire behavior study (Chapter 2) indicate that targeted cattle grazing of *B. tectorum* at the boot stage is capable of moderating the flame length and rate of spread of wildfires. This grazing treatment reduced percent cover of *B. tectorum*, fuel bed depth and fuel loading, and thus the flame length and rate of spread. While our burns were not carried out under peak wildfire conditions, simulation modeling provided a high degree of inference to fire behavior under these conditions. Simulations of fire behavior indicate a significant decrease in flame length and rate of spread during peak fire season. These findings constitute an initial step in reducing the threat of catastrophic wildfires on *B. tectorum*-dominated rangelands in the northern Great Basin.

Results from the aboveground community and seed dynamics study (Chapter 3) indicate that grazing, burning, and the combination of the two, have the potential to alter the trajectory of *B. tectorum*-dominated communities. Knowledge of the causes of succession (site availability, species availability and species performance) and the associated ecological processes, components and modifying factors helped to identify methods and the timing of their application, for reducing *B. tectorum* dominance and creating more desirable (less flammable) plant communities (Pickett et al. 1987; Sheley et al. 1996 and 2006). Grazing alone altered species availability by significantly limiting seed production, and it reduced species performance by removing photosynthetic leaf area and restricting plant regrowth. Grazing alone, however, did not significantly alter litter depth and continuity, and thus the available sites that facilitate *B. tectorum* recovery and dominance. Burning alone altered site availability by reducing the litter bed and increasing the amount of bare soil, but its impact on species availability and performance was short-lived. By combining grazing and burning, we addressed all three causes of

succession, which resulted in a change in community trajectory from a *B. tectorum*-dominated community to one dominated by less flammable species.

In the final study (Chapter 4), cost-effectiveness analysis allowed us to evaluate *B. tectorum* fuel reduction treatments that would provide some measure of protection for an adjacent sagebrush-grassland community. It is impractical to try to estimate a dollar value for protecting the biodiversity or wildlife habitat attributes of a native plant community. In management situations where the benefits of fuel reduction treatments are difficult to measure, effectiveness proxies, such as fuel loading changes, flame length and rate of spread provide a measure of cost effectiveness (Rideout et al. 1999). Simulation modeling indicated that targeted grazing is an economically viable method for creating fuel breaks between a *B. tectorum*-dominated community and a remnant sagebrush-grassland community when *B. tectorum* biomass levels (fuel loads) are high or moderate. However, when biomass levels are consistently low, it is more cost-effective to use Plateau<sup>®</sup> herbicide to reduce flame lengths and rate of spread in fuel breaks.

Collectively, findings from these studies show that targeted cattle grazing and prescribed burning can be used as management tools to reduce *B. tectorum* dominance and associated fire hazards on rangelands in the northern Great Basin. To be most effective, these vegetation manipulation methods should be combined at appropriate times (stages) in the life cycle of *B. tectorum*, so that the strengths of one method can compensate for the weaknesses of the other, and vice-versa. Intensive cattle grazing in May, when inflorescences are emerging from leaf sheaths, suppresses vegetative growth and seed input into the seed bank; however, hoof action during grazing does not effectively disrupt the existing litter bed that provides safe sites for establishment from

carryover seeds in the seed bank. Prescribed burning in October, after seed shatter and plant senescence, does not suppress vegetative growth or seed input, but it can remove much of the litter bed and seeds suspended in the litter. Thus, when integrated accordingly, grazing and burning can place *B. tectorum* at a disadvantage with associated species, such as perennial grasses (*Poa secunda*) and annual forbs (*Sisymbrium altissimum* and *Lepidium perfoliatum*), which respond differently to these disturbances. Decreasing *B. tectorum* cover, biomass, and density leads to lower fuel loads and continuity, resulting in fires that have lower flame lengths and spread more slowly. Lower severity fires can be fought with fewer resources, reducing suppression costs. However, these changes in community dynamics and fire behavior are only temporary, as *B. tectorum* can rapidly regain dominance from new and carryover propagules, and from litter buildup which facilitates their establishment. It is appealing to use a readily available resource such as cattle to alter community dynamics and fire behavior on private and public lands; however, the cost effectiveness of using cattle as fuel reduction agents depends on the amount of forage that is available to generate weight gains that offset the costs associated with intensive grazing management.

Although these studies indicate that targeted cattle grazing and prescribed burning can alter the fire behavior and community dynamics of a *B. tectorum*-dominated landscape, in a cost effective manner under certain conditions, there are several aspects of this type of research that require further investigation:

1. The effects of grazing treatments on fire behavior were supposed to be characterized by conducting burns during the peak fire season (July-August) in the northern Great Basin. Due to the unavailability of fire suppression personnel

in July and August 2005 and 2006, burns were implemented in October 2005 and 2006, and we were forced to rely on simulation modeling to estimate flame lengths and rates of spread under peak fire conditions. While our simulation estimates were based on actual fuel conditions and fire behavior during the October burns, they cannot duplicate fire behavior under peak fire conditions. Therefore, we recommend that similar grazing treatments be implemented and burned in July or August, and monitored for differences in flame length and rate of spread.

2. Changes in seed bank dynamics were monitored for 2 years, and changes in aboveground community dynamics were monitored for 3 years, primarily during the period when grazing and burning treatments were implemented. Monitoring should be extended over a longer period to document the effects of these treatments on plant density, cover, biomass, species composition, and seed bank dynamics. Targeted grazing should be extended beyond 2 consecutive years to further examine its impact on *B. tectorum* seed bank dynamics and forage availability.
3. Even though grazing and burning treatments were implemented in fairly large plots (2/3 ha), there is a need to increase the scale of this type of demonstration research to a more realistic landscape level. A multiple year study at a larger scale, similar to the simulated 4.2 km fuel break described in Chapter IV, would provide an opportunity to account for fencing and animal management costs, and determine how targeted grazing fits into a public land grazing management program.

4. Revegetation needs to be incorporated with fuel reduction treatments. Further research should investigate how desirable native and introduced species establish in grazed and burned sites where species composition has temporarily shifted from *B. tectorum* as the dominant species to *Poa secunda* and annual forbs as the major species.

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APPENDIX

**Table A.1.** Analysis of flame length in 2 treatments, 6 distances and 2 years.

Type 3 Tests of fixed effects

Source of Variation	Num. DF	Den. DF	F	P
Treatment	1	46	140.39	0.001
Distance	5	46	18.25	0.001
Distance*Year	5	46	0.46	0.801
Distance*Treatment	5	46	0.46	0.801
Distance*Treatment*Year	5	46	0.54	0.751

**Table A.2.** Analysis of rate of spread in 2 treatments, 6 distances and 2 years

Type 3 Tests of fixed effects

Source of Variation	Num. DF	Den. DF	F	P
Distance*Treatment	5	46	3.46	0.069
Distance*Year	5	46	0.81	0.691

**Table A.3.** Analysis of *Bromus tectorum* cover, biomass and plant density in 4 treatments, 6 periods and 3 years.

Type 3 Tests of fixed effects					
Data Type	Source of Variation	Num. DF	Den. DF	F	P
Cover	Treatment	3	6	29.71	0.001
Cover	Treatment*Period	3	15	10.72	0.001
Biomass	Treatment	3	6	29.51	0.001
Biomass	Treatment*Period	3	15	30.89	0.001
Density	Treatment	3	6	87.03	0.001
Density	Treatment*Period	3	15	8.65	0.001

**Table A.4.** Analysis of *Sisymbrium altissimum* cover, biomass and plant density in 4 treatments, 6 periods and 3 years

Type 3 Tests of fixed effects					
Data Type	Source of Variation	Num. DF	Den. DF	F	P
Cover	Treatment*Period*year	2	1797	14.99	0.001
Cover	Treatment*Period	15	1797	4.14	0.001
Biomass	Treatment*Period*year	6	1359	11.41	0.001
Biomass	Treatment*Period	15	1359	5.17	0.001
Density	Treatment*Period*year	3	1826	20.68	0.001
Density	Treatment*Period	15	1826	11.91	0.001

**Table A.5.** Analysis of *Lepidium perfoliatum* cover, biomass and plant density in 4 treatments, 6 periods and 3 years.

Type 3 Tests of fixed effects					
Data Type	Source of Variation	Num. DF	Den. DF	F	P
Cover	Treatment*Period	6	1797	16.53	0.001
Cover	Treatment*Period*year	15	1797	8.36	0.001
Biomass	Treatment*Period	6	1359	13.4	0.001
Biomass	Treatment*Period*year	15	1359	7.4	0.001
Density	Treatment*Period	6	1826	5.24	0.001
Density	Treatment*Period*year	15	1826	7.21	0.001

**Table A.6.** Analysis of *Poa secunda* cover, biomass and plant density in 4 treatments, 6 periods and 3 years.

Type 3 Tests of fixed effects					
Data Type	Source of Variation	Num. DF	Den. DF	F	P
Cover	Treatment	3	6	63.78	0.001
Cover	Treatment*Period	15	1797	11.88	0.001
Biomass	Treatment	3	6	7.29	0.02
Biomass	Treatment*Period	15	1359	41.82	0.001



**Table A.7.** Analysis of all other forbs combined cover, biomass and plant density in 4 treatments, 6 periods and 3 years.

Type 3 Tests of fixed effects					
Data Type	Source of Variation	Num. DF	Den. DF	F	P
Biomass	Treatment*Year	6	1368	1.64	0.133
Biomass	Treatment*Period	15	1368	3.97	0.001

**Table A.8.** Analysis of all other grasses combined cover, biomass and plant density in 4 treatments, 6 periods and 3 years.

Type 3 Tests of fixed effects					
Data Type	Source of Variation	Num. DF	Den. DF	F	P
Cover	Treatment	3	6	13.4	0.005
Biomass	Treatment	3	6	1.13	0.409
Density	Treatment	3	6	0.38	0.769
Cover	Treatment*Period	15	1797	22	0.001
Biomass	Treatment*Period	15	1368	6.69	0.001

**Table A.9.** Analysis of soil and litter cover in 4 treatments, 6 periods and 3 years.

Type 3 Tests of fixed effects					
Data Type	Source of Variation	Num. DF	Den. DF	F	P
Soil cover	Treatment	3	6	24.42	0.001
Soil Cover	Treatment*Period	15	1797	16.52	0.001
Litter Cover	Treatment	3	6	6.09	0.029
Litter Cover	Treatment*Period	15	1797	18.58	0.001

**Table A.10.** Analysis of *Bromus tectorum* seed density 4 treatments, 6 periods and 2 years.

Type 3 Tests of fixed effects					
Data Type	Source of Variation	Num. DF	Den. DF	F	P
Seed Density	Treatment	3	6	13.57	0.001
Seed Density	Treatment*Period*Year	6	2528	8.68	0.001

**Table A.11.** Analysis of *Sisymbrium altissimum* seed density in 4 treatments, 6 periods and 2 years.

Type 3 Tests of fixed effects					
Data Type	Source of Variation	Num. DF	Den. DF	F	P
Seed Density	Period	5	2528	44.69	0.001
Seed Density	Treatment*Period	15	2528	7.41	0.001

**Table A.12.** Analysis of *Lepidium perfoliatum* seed density in 4 treatments, 6 periods and 2 years.

Type 3 Tests of fixed effects					
Data Type	Source of Variation	Num. DF	Den. DF	F	P
Seed Density	Period	5	2528	60.1	0.001
Seed Density	Treatment*Period	15	2528	8.68	0.001

**Table A.13.** Analysis of *Poa secunda* seed density in 4 treatments, 6 periods and 2 years.

Type 3 Tests of fixed effects					
Data Type	Source of Variation	Num. DF	Den. DF	F	P
Seed Density	Period	5	2528	30.61	0.001
Seed Density	Treatment*Period	5	2528	9.39	0.001

**Table A.14.** Analysis of *Bromus tectorum* seed input 4 treatments.

Type 3 Tests of fixed effects					
Data Type	Source of Variation	Num. DF	Den. DF	F	P
Seed Input	Treatment	3	6	63.95	0.001



**Table A.15.** Analysis of *Lepidium perfoliatum* seed input in 4 treatments.

Type 3 Tests of fixed effects					
Data Type	Source of Variation	Num. DF	Den. DF	F	P
Seed Input	Treatment	3	6	0.54	0.675

**Table A.16.** Analysis of *Poa secunda* seed input in 4 treatments.

Type 3 Tests of fixed effects					
Data Type	Source of Variation	Num. DF	Den. DF	F	P
Seed Input	Treatment	3	6	0.45	0.79

## CURRICULUM VITAE

Joel M. Diamond

EDUCATION

- Ph.D., Ecology and Range Science Pending 2009  
 Utah State University, Logan, Utah  
 Effects of targeted grazing and prescribed burning on fire behavior, seed & community dynamics and economic allocation of cheatgrass (*Bromus tectorum*) dominated landscapes.
- M.S., Wildlife Biology 2007  
 Utah State University, Logan, Utah  
 Monitoring bat activity: Evaluating the effectiveness of motion detectors.
- B.Sc., Zoology 2001  
 Southern Utah State University, Cedar City, Utah

TEACHING & ADVISORY EXPERIENCE

- |  |                                     |
|--|-------------------------------------|
| Instructor, General Ecology                      | Utah State University, Spring 2008  |
| Co-Instructor Wildland Fauna                     | Utah State University, 2004 – 2006  |
| Instructor, Vertebrate Biodiversity              | Utah State University, Spring 2005  |
| Advisor, Student Chapter of The Wildlife Society | Utah State University, 2005         |
| Advisor, Undergraduate Bat Research              | Utah State University, 2006 -- 2007 |

PUBLICATIONS & PROFESSIONAL PARTICIPATION

- Effects of targeted cattle grazing on fire behaviour of cheatgrass-dominated rangeland in the northern Great Basin, USA.* In review, The International Journal of Wildland Fire, July 2008.
- Effects of intensive cattle grazing on fire behavior, community assembly and seed dynamics of cheatgrass dominated landscapes, 2005 – 2008.* Utah State University. Identifying the impacts of physiologically targeted grazing on cheatgrass and the associated plant community through seed bank, community assembly and prescribed burning analysis.
- Evaluating the impacts of survey personnel on obligate cave roosting bat species in northern Utah, 2006 – present.* Utah State University. Directing undergraduate students in data collection and analysis.

- Monitoring bat activity: Evaluating the effectiveness of motion detectors*, 2002 – 2007  
Utah State University. Behavioral observation, data analysis, and handling of obligate mine roosting bat species.
- DNA fingerprinting of Burrowing Owl (Athene cunicularia)*, 2001- 2002. Southern Utah University. Extraction, digestion, and fingerprinting of Burrowing Owl DNA from blood samples. Analysis of genetic relatedness between individuals and groups of animals.
- Silver Reef and Tushar Mountain abandoned mine and bat behavior study*, 1999- 2003. Southern Utah University. Behavioral observation, data analysis, and handling of obligate mine roosting bat species in Silver Reef and Tushar mining districts of southern Utah.
- Poster presentation, “*Effects of intensive grazing on fire behavior, seed banks and plant communities: managing cheatgrass (Bromus tectorum) dominated landscapes*,” Society of International Wildland Fire Ecology, San Diego CA (2006).
- Guest lecturer, “*Bats of Northern Utah*,” Stokes Nature Center, general bat identification, natural history and field observations to the public, Logan Utah, (2003, 2004, 2005, 2006 and 2007).
- Panel speaker, “*Monitoring bat activity: evaluating the effectiveness of motion detectors*,” Utah Chapter of The Wildlife Society, Provo Utah (2002).
- Poster presentation, “*Monitoring bat activity: evaluating the effectiveness of motion detectors*,” North American Symposium on Bat Research, Salt Lake City Utah, (2001, 2004).
- Guest field lecturer, “*Batting Around: western bat identification and general ecology*,” Desert Wildlife Festival, St. George Utah, (2001).
- Poster presentation, “*Evaluating the effectiveness of bat compatible gates*,” Utah Chapter of The Wildlife Society, St. George Utah, (2001).
- Poster presentation, “*Evaluating the effectiveness of bat compatible gates*” and “*Evaluating Utah Prairie Dog behavior in translocation sites*,” Undergraduate Research Symposium, Southern Utah University, Cedar City Utah, (2001).
- Guest field lecture, “*Western bat ecology and identification*,” Cedar Mountain Science Center, Cedar Mountain Utah, (2000).

#### PROFESSIONAL EXPERIENCE

Bat Biologist, UDOGM

August 2000--present

Wildlife Technician, USFS

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Biological Technician, UDWR 2001	November 2000--March 2001
Utah Prairie Dog Translocation Project, UDWR	April 1999--September 2000

#### AWARDS & GRANTS

Utah Division of Oil, Gas and Mining	\$20,000 – 127,000
Utah Chapter of The Wildlife Society Research Grant, 2002	\$1,000
NASA Space Grant Consortium Fellowship, 2001	\$1,000
Biology Departmental Scholarship, 2000-2001	\$5,000

#### MANUSCRIPT REVIEW

Western North American Naturalist

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International Wildland Fire  
Utah Bat Conservation Cooperative  
Wildlife Society, Utah Chapter  
Western Bat Working Group  
Legacy Bat Cooperative