
SAGEBRUSH STEPPE AND PINYON- JUNIPER ECOSYSTEMS - EFFECTS OF CHANGING FIRE REGIMES, INCREASED FUEL LOADS, AND INVASIVE SPECIES

Final Report to the Joint Fire Science Program

Project #00-1-1-03

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September 30, 2005

EXECUTIVE SUMMARY

Overview. Pinyon-juniper woodlands and Wyoming big sagebrush ecosystems have undergone major changes in vegetation structure and composition since settlement by European Americans. These changes are resulting in dramatic shifts in fire frequency, size and severity. Effective management of these systems has been hindered by lack of information on: (1) presettlement fire regimes and the spatial and temporal changes that have occurred in Intermountain Region woodlands and sagebrush ecosystems since settlement; (2) changes in fuel loads and the consequences for the ecosystem types and conditions that currently exist on the landscape; and (3) the environmental and ecological factors that influence community susceptibility to invasion by nonnative species. This project utilized an integrated, collaborative project of the Joint Fire Sciences Program to address each of these information needs. The project duration was from October 1, 2000 to September 30, 2005.

Approach. The first component of the project evaluated the spatial and temporal dynamics of the shrublands and woodlands prior to settlement by using a combination of plant community, stand structure, and dendrochronology data. It also examined the influence of varying environmental conditions on current shrubland and woodland dynamics. The second component was closely aligned with the first. It evaluated the changes that have occurred in fuel loads since settlement and provided a comparison of fuel loading characteristics for *Artemisia* ecosystems that are functional, at risk, or that have been converted to nonnative annual grasses and secondary weeds. It also examined how both vegetation type and successional stage affect fuel loads. The third component used mechanistic studies to examine key factors influencing community susceptibility to invasion by cheatgrass (*Bromus tectorum*) and the secondary weeds, squarrose knapweed (*Centaurea virgata* var. *squarrosa*) and Rush skeleton weed (*Chondrilla juncea*). The effects of both the presence and abundance of native species and fire on soil resource availability and, consequently, cheatgrass establishment and reproduction were examined for elevational gradients typical of *Artemisia* ecosystems in the Intermountain Region. The distribution of Rush skeleton weed within the Region was evaluated, and the effects of fire on its establishment and spread were examined. Similarly, the effects of fire and community composition on the establishment and spread of squarrose knapweed were evaluated.

Deliverables. The project has provided basic information on the rate and characteristics of woodland expansion into sagebrush dominated ecosystems across the Intermountain Region and on the implications of this expansion for both fuel loads and fire regimes. It also has provided information on the effects of fire and other disturbances such as overgrazing by livestock on the susceptibility of sagebrush ecosystems to cheatgrass and the secondary weeds, squarrose knapweed and Rush skeleton weed. This information has been used to develop guidelines for the management and restoration of these ecosystems. The deliverables are in the form of peer-reviewed publication, theses, and General Technical Reports. They also include numerous presentations at agency meetings, professional meetings, and symposia.

The project's initial findings were highlighted at a symposium on "Changing fire regimes, increased fuel loads, and invasive species: effects on sagebrush steppe and pinyon-juniper ecosystems" that was held at the Society of Range Management Meeting, Jan 25-29, 2004, in Salt Lake City, Utah. The project deliverables outlined in the initial proposal are provided below, and the titles and status of the specific products are given at the end of the report.

- **Woodland Expansion, Fuel Loads, and Fire Regimes -**
 - Information on the local and regional variation of the pre- and post settlement shrublands and woodlands, and descriptions of how vegetation structure and fuel loads have changed.*
 - Data on the characteristics of shrublands and woodlands at greatest risk of catastrophic fire, their associated communities, their environmental characteristics, and guidelines for their management.*
 - Guidelines on the fuel loads and other characteristics of communities most suitable for the use of prescribed fire or mechanical treatments.*
 - Comparative data on the fuel loads and fire susceptibility of intact communities, communities revegetated with introduced and native species, and communities invaded by cheatgrass or secondary weeds.*

- **Fire and Invasive Species -**
 - Data on the effects of both fire and disturbances that decrease native plant abundance on cheatgrass and secondary weed invasion.*
 - Identification of the types of management activities that are likely to increase community susceptibility to cheatgrass and secondary weed invasion.*

- **Management and Restoration -**
 - Identification of the types of fire and non-fire treatments (including revegetation) potentially useful for restoring more diverse communities in shrubland/woodland dominated areas.*

Organization of this Report. This report is organized into a series of extended abstracts that detail each of the integrated and collaborative studies that were conducted as a part of the project. The Table of Contents that follows gives the titles and authors for each abstract. The abstracts themselves provide descriptions of the studies, methods, results, and conclusions as well as a person who can be contacted for additional information. At the end of the document, a full list of both the initial and anticipated products has been compiled. These have been organized according to the deliverables that were included in the original proposal.

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WOODLAND EXPANSION, STAND DYNAMICS AND FUEL LOADS IN WESTERN JUNIPER (*JUNIPERUS OCCIDENTALIS*) WOODLANDS

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Study Description. It has been well documented that juniper and piñon woodlands have been rapidly expanding since the late 1800's in the Intermountain West. These semi-arid woodlands occupy around 19 million hectares, far less land than they are capable of under current climatic conditions. Woodlands have displaced shrub steppe communities, influenced wildlife habitat, and altered ecological processes including fire. However, little work has been conducted from a landscape perspective evaluating the process of expansion, its impacts on fuel loads, and the composition of relatively recent woodlands with presettlement stands. To describe the process of woodland expansion and its effects on fuel loads we evaluated four woodlands along an elevational gradient representing a wide array of environmental variables. Specific questions we addressed were: (1) do landscape variables such as topography and position of old-growth trees influence rate of tree establishment, (2) do landscape variables influence stand structure (i.e. cover and density), (3) how does woodland development influence fuel loads, and (4) how long does it take woodlands on different landscape positions to approach full development.

Methods. In 2001, 342 juniper stands were sampled in 0.5 ha plots along four transects ranging from 15 to 20 km in length in southeastern Oregon and southwestern Idaho. Phase of woodland development (i.e. early, intermediate, or late) was identified based on previously developed protocol. Tree density was measured and the three tallest trees were aged to estimate initiation of woodland age. Woodland development rate was determined as a function of woodland age and stand density. Environmental variables (e.g. slope, aspect, elevation, etc.) were recorded for each stand and regression models were fit to predict total juniper density, dominant juniper density, and woodland development rate on stand elevation and site exposure.

In 2002, thirty stands were revisited in southwestern Idaho to sample stand chronologies and fuels more intensively. Ten trees from four size classes were cored (n = 990). Ring widths on core samples were measured, relativized by the overall mean ring width, and resulting relative growth rates were plotted graphically to estimate years transpired since establishment until stand closure. All tree measurements were conducted in three 6 x 60 m belt transects. Basal circumference above the root crown was measured and used to estimate tree component biomass (Gholz 1980). The average depth and diameter of the litter mat under each aged tree was recorded and biomass estimates for duff were generated (Brown et al. 1982). The number of intersections by downed dead woody material were tallied by diameter class (one, ten, hundred and thousand hour fuels) along a 60 m transect and biomass was estimated (Brown 1974). Shrubs and grasses were measured within five regularly located belt transects (2 x 12 m) partitioned into 6, 1 m² microplots. Elliptical crown diameter and maximum height measurements were obtained on mountain big sagebrush to estimate crown area, percent cover, and shrub biomass (Rittenhouse and Sneva 1977). Perennial grasses and perennial forbs were measured and clipped in 20 microplots to determine basal area, percent cover, and biomass.

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Results and Discussion. The rate of woodland expansion and density of trees increased with elevation and northerly exposure (Fig. 1). The presence of at least one pre-settlement tree resulted in an earlier initiation of post settlement woodland expansion ($p < 0.0001$). We estimated that accelerated post settlement tree expansion began 24 years earlier in stands with at least one pre-settlement tree present relative to stands that lacked the presence of trees prior to settlement (95% C.I.: 16 to 32 years). The lightly shaded area in Figure 2 shows the estimated time period necessary for stands developing under varying environmental conditions to yield a minimum stocking of trees, that once mature, will lead to the dominance by juniper. Stands on mesic sites achieved a minimum stocking of over 250 trees/ha in less than 25 years, whereas 40 to 50+ years were required to achieve a minimum stocking of fewer than 100 mature trees/ha on drier sites. A significant reduction in relative growth rates in closed woodlands indicated intraspecific competition between juniper trees began in the late 1950s and early 1960s. We speculate that this was a period of canopy closure and rapid decline in understory vegetation, especially shrubs. The intermediately shaded area in Figure 2 represents the time period from stand establishment to stand closure given a site's environmental variables. It was found that drier sites support fewer total mature trees in a closed state than mesic sites, <340 trees/ha as opposed to >900 trees/ha, respectively. Stand closure occurred within 120 to 170 years on warm dry sites compared to 80 to 120 years on cool moist sites.

Only post-settlement trees were present in 63% of the stands sampled across the four transects. In 28% of the plots at least one pre-settlement tree was present and 4% of the plots were dominated by pre-settlement trees. Old-growth tree abundance and spatial distribution was greater in southwest Idaho than in southeast Oregon. Although post-settlement trees were present in 27% of the plots, these trees only accounted for 1 to 10% of the population. Trees established since the late 1800s accounted for 90% or more of the total population.

The pattern of woodland development can be described in three phases. Phase one is primarily a period of tree establishment (Figs. 2 & 3). During this first phase mountain big sagebrush and perennial grasses dominate the site with seedling and sapling trees scattered throughout. Phase two is a period of rapid tree encroachment and a period when trees are co-dominant with shrubs and herbs on the site. This phase is primarily an artifact of increased growth rates as established trees mature. Tree establishment continues through this period but generally slows as stands approach closure. Phase three is the final period corresponding to high tree dominance and low abundance of shrubs and herbs. An important threshold in this model occurs during the transition from phase two to three in which the fire regime changes and shrubs and grasses play a limited role in the tree-dominated site (Fig. 3).

There were significant differences across the three phases of woodland development in herbaceous, shrub, tree, duff, one hour, and thousand hour fuels biomass. Contrasts of early phase estimates with intermediate and late phase estimates are summarized in Table 1. Herbaceous and shrub biomass has declined markedly during the transition from open shrub steppe communities to closed woodlands. Shrub and herbaceous species averaged 96% of the total live fuels biomass in shrub steppe dominated communities, whereas this combination averaged less than 2% of the total live fuel biomass in late phase juniper stands, indicating a dramatic shift in the composition of live fuel loading.

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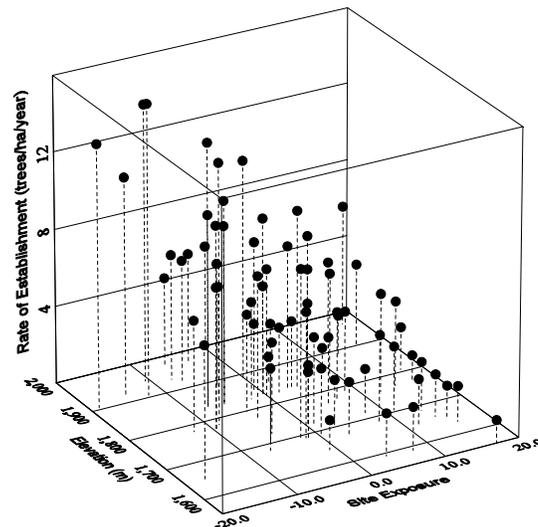
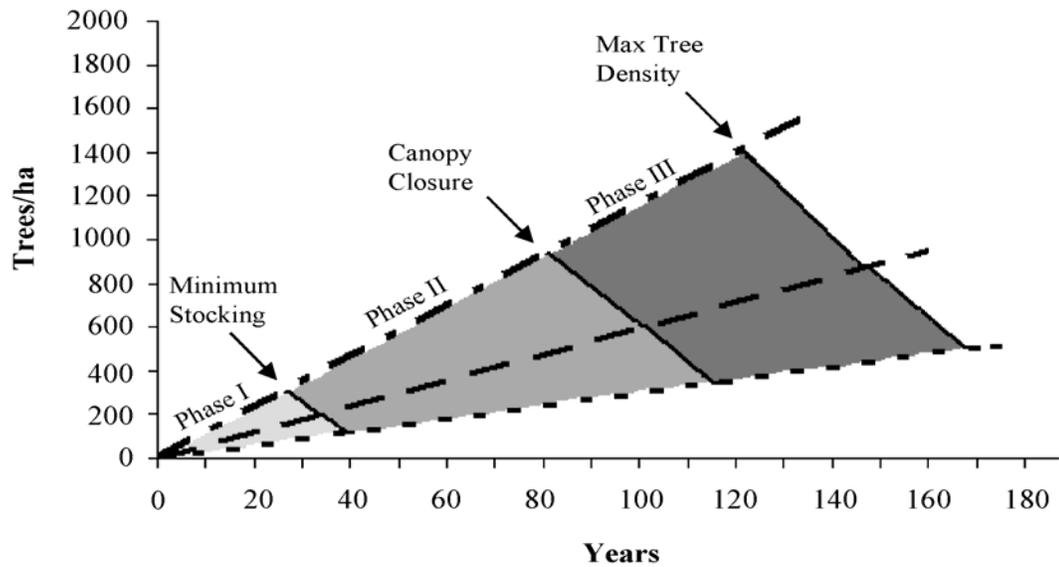


Figure 1. Relationship of rate of establishment with elevation (m) and site exposure in stands associated with mountain big sagebrush. Site exposure is an index based on aspect and slope calculated as follows: site exposure = slope * cosine(π * (aspect - 180)/180). Site exposure becomes increasingly warmer to the right.



- · - Estimated Rate of Tree Establishment for High Elevation/Low Exposure Sites
- - - Estimated rate of Tree Establishment for Intermediate Elevation and Exposure Sites
- Estimated Rate of Tree Establishment for Low Elevation/High Exposure Sites

Figure 2. Display showing the hypothesized time periods required from initial tree establishment to minimum stocking, stand closure, and estimated maximum tree density for stands developing under varying environmental conditions.

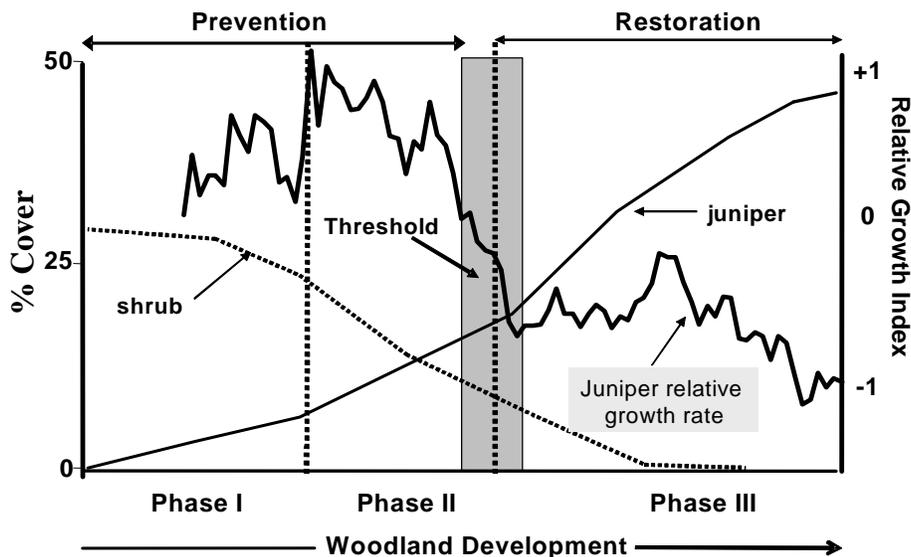


Figure 3. Display showing the relationship between shrub canopy cover, tree canopy cover and relative growth rates (i.e. ratio of annual ring width over mean ring width), and management intensity during different phases of woodland development. The threshold suggests the point where the fire regime significantly changes.

Table 1. Estimates of change in fuels biomass during transitions from early to intermediate and late phases of woodland development. 95 % confidence intervals for estimates are also presented.

Response	Early → Intermediate ^a		Early → Late ^b	
	Estimate (%)	95% C.I. (%)	Estimate (%)	95% C.I. (%)
Herbaceous	84 ^c	56 125	41	30 55*
Shrub	25	22 32*	5	3 8*
Tree	480	294 798*	5410	3700 7900*
Duff	405	329 490*	431	374 500*
One hour	115	64 204	166	108 257*
Ten hour	82	44 155	127	79 205
Hundred hour	50	9.7 255	39	12 134
Thousand hour	806	155 4200*	965	108 8626*

a

Estimated changes in biomass from early to intermediate phase

^b Estimated changes in biomass from early to late phase

^c Interpret as median herbaceous biomass in the intermediate phase is 84% of what it was in the early phase.

* 95% C.I.s that exclude 100% (i.e. no change in median biomass).

WOODLAND EXPANSION, STAND DYNAMICS AND FUEL LOADS IN SINGLE-NEEDED PINYON (*PINUS MONOPHYLLA*) AND UTAH JUNIPER (*JUNIPERUS OSTEOSPERMA*) WOODLANDS

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Study Description. Following European settlement, woodlands of the Great Basin have rapidly expanded into the sagebrush steppe. This expansion and eventual suppression of the invaded sagebrush community has resulted in considerable loss in area for these diverse and productive ecosystems. In many locations this has resulted in increased soil erosion and is increasingly resulting in the increase in the size and intensity of wildfire. This study is investigating the changes in plant community composition, structure and fuel loads that have resulted over time from this encroachment of sagebrush steppe by the woodland species. The specific objectives of this study include: 1) estimation of the spatial distribution of pre-settlement sagebrush steppe and pinyon-juniper woodland communities; 2) Determination of the spatial and temporal dynamics of woodland expansion during the past 130 years; 3) Determination of the patterns, the amounts, and the composition of the fuel load changes that have, and that are continuing to occur with woodland expansion; 4) Estimation of the possible differences between pre- and post-settlement fire regimes and how these are related to the ongoing woodland expansion and associated changes in fuel loads; and 5) In a central Nevada demonstration area the determination how variation in plant species composition and in fuel loads affects community response to prescribed fire. These studies have been conducted on the Shoshone Mountains in Nevada and the East Tintic Mountains in Utah.

Methods. In both Nevada and Utah, extensive sampling has occurred across the entire mountain range by using belt transects. These transects crossed the north-south trending mountain ranges in the east to west direction. They started 1 km below the present woodland on one side of the mountain, crossed over the mountain and ended 1 km below the present woodlands on the other side. Two belt transects were sampled in Nevada and one in Utah. One belt transect in Nevada was located on a wetter part, and one on a drier part of the range. Systematic samples occurred at regular intervals along three parallel transects located within each belt transect. Sampling occurred over two summers with a more rapid sampling procedure used the first summer and a more intensive sampling procedure the second summer on a subset of the first summer's plots.

At the additional site in Nevada sampling occurred on nine sites distributed the length of a west to east trending canyon. This location is being used as a demonstration area on the use of prescribed fire. The nine sites are located on north-facing slopes and alluvial fans at or near the bottom of the canyon and range in elevation from 2040 to 2400 m (6600 to 7800 feet) in elevation. Sampling was stratified on each site by the three tree dominance levels of low, medium and high. Following sampling four of the sites were burned using prescribed fire in May, 2002.

Percent cover was recorded by species for trees, shrubs, perennial grass, perennial forbs, and annuals. Measurements of plant size and height needed for determining fuel loads were also

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taken by species. On the cross-mountain transects in both Nevada and Utah, increment cores were taken from the three largest trees in each plot during the first summer's sampling to estimate when tree establishment on the site began. In the subset of plots sampled the second summer all the trees were sampled for age determination. For the more intensively sampled canyon in Nevada used as the demonstration area, increment cores or cross-sections were also taken from all the trees on one-third of the plots, stratified by tree dominance. This more intensive age sampling is being used to estimate how the patterns of establishment may have varied through time.

Results and Conclusions. A consistent pattern in the amounts and rates of the establishment of the trees into the sagebrush ecosystems has emerged from all study sites in both states. On all sites a major increase in tree dominance began at the end of the nineteenth century. In both Nevada and Utah tree establishment began about the same time (1880's) in both plots with pre-settlement age trees and plots with only post settlement age trees (Figs. 1 & 2). Establishment was more rapid in Utah than in Nevada but both sites generally reaching their peaks around or shortly after the middle of the twentieth century. Although differing in the rates of establishment, the patterns of establishment were essentially the same on both sites. The pattern and rates of establishment, and average density of trees in plots with pre-settlement age trees is essentially identical in both Nevada and Utah over about the 300 years previous to settlement (Fig. 1). At the time of sampling the average tree density was the same for pre- and post-settlement plots in Nevada (Fig. 3). In Utah the pre-settlement plots had an average density almost 50% higher than the post-settlement plots

On the Shoshone Mountains, Nevada about twice the current vegetation on the mountain is still sagebrush (32% of the plots) as on the East Tintic Mountains of Utah (13% of the plots) (Fig. 5). While woodlands in Nevada are not quite 70% of the vegetation on the mountain, in Utah they are nearly 80% of the vegetation (Fig. 5). In both Nevada and Utah the division between plots with pre-settlement age trees and those with only post-settlement age trees are essentially identical (Fig. 6) with over 75% of the plots post-settlement in age. The patterns of plot distribution between low, mid, and high tree dominance levels are also nearly identical (Fig. 7), with close to 50% of the plots in the mid-tree dominance category and another 30% in the early category. Overall, nearly 80 percent of these expansion woodlands established during the twentieth century and are now moving into the period of greatest biomass expansion.

Changes in fuel loads are directly related to the changes in tree dominance. As succession proceeds fuel loads contributed by the trees double between low and mid tree dominance levels. Simultaneously both the percent cover and the fuel loads contributed by the understory decrease by half. At this point the combined total fine fuels for the plot have nearly doubled. From mid to full tree dominance fuel loads contributed by the trees double again while the cover and fuel loads contributed by the understory decrease to one-fourth or less their level present at mid tree dominance. With total tree dominance fine fuels for the plot average double the level present at mid tree dominance, and have reached four to five times what was present without the trees.

In both Nevada and Utah less than one-fifth of the plots with trees are fully tree dominated (Fig. 7). More than half are in the mid-dominance category. These plots also have more than adequate tree density for the trees to dominate once they mature. Based on tree growth patterns present on the sites they will all be fully tree dominated in 40 to 50 years. This indicates that between 60 and 70 percent of the woodlands on the Shoshone Mountains in Nevada and the East Tintic Mountains in Utah will be tree dominated at that time. As a part of this the fuel loads on the mid-tree dominance sites, representing over half the current

woodlands, will double over the next 40 to 50 years. Because of the higher tree densities in Utah, this level of tree dominance across the landscape may be reached earlier than in Nevada.

Understory response following the prescribed fire in the demonstration canyon in Nevada indicates a possible threshold in the ability of the understory to rebound after the disturbance that occurs between 40 and 50 percent tree cover. Below 40 percent pre-fire tree cover, the understory response at the end of the first growing season after the fire appeared independent of the pre-fire tree cover. Beyond 60 percent pre-fire tree cover, the herbaceous understory cover present at the end of the first growing season after the fire is also independent of the pre-fire tree cover, but averages less than one-fourth the vegetation cover on sites that had less than 50 percent tree cover prior to the fire. Between 40 and 50 percent pre-fire tree cover there was a rapid decline in the ability of the understory to rebound after the disturbance.

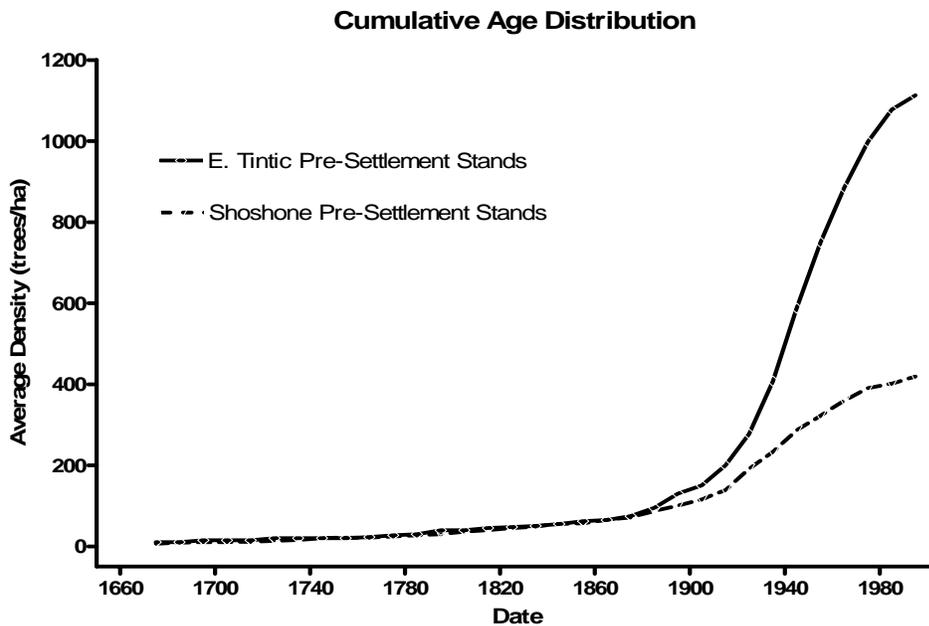


Figure 1. Cumulative age distributions on one site each in Nevada and Utah for the average density per plot recorded on pre-settlement aged plots (oldest tree greater than 140 years old) where all trees were aged.

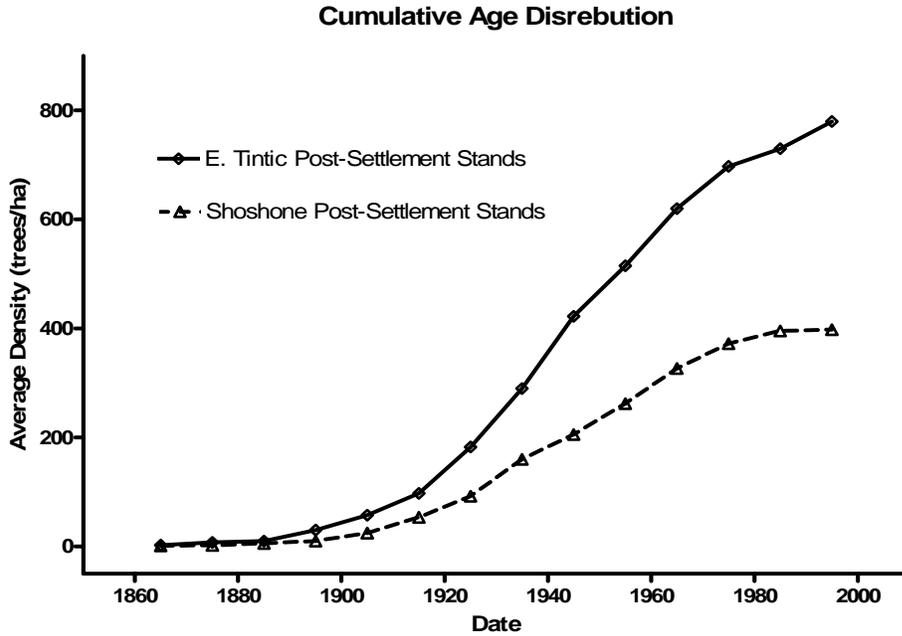


Figure 2. Cumulative age distributions on one site each in Nevada and Utah for the average density per plot recorded on post-settlement aged plots (all trees less than 140 years old) where all trees were aged.

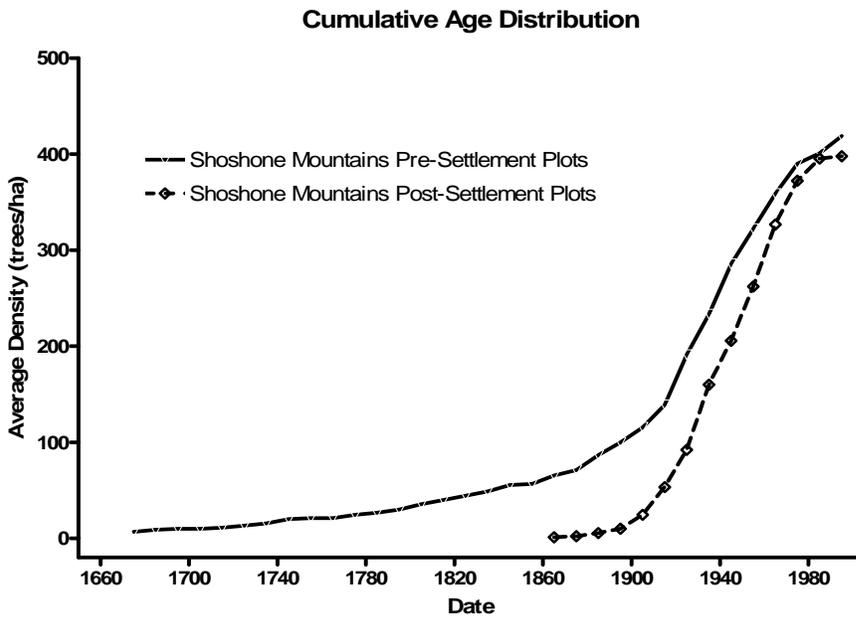


Figure 3. Comparison of cumulative age distributions on pre- (oldest tree greater than 140 years old) and post-settlement (all tree less than 140 years old) plots in Nevada for the average density per plot where all trees were aged.

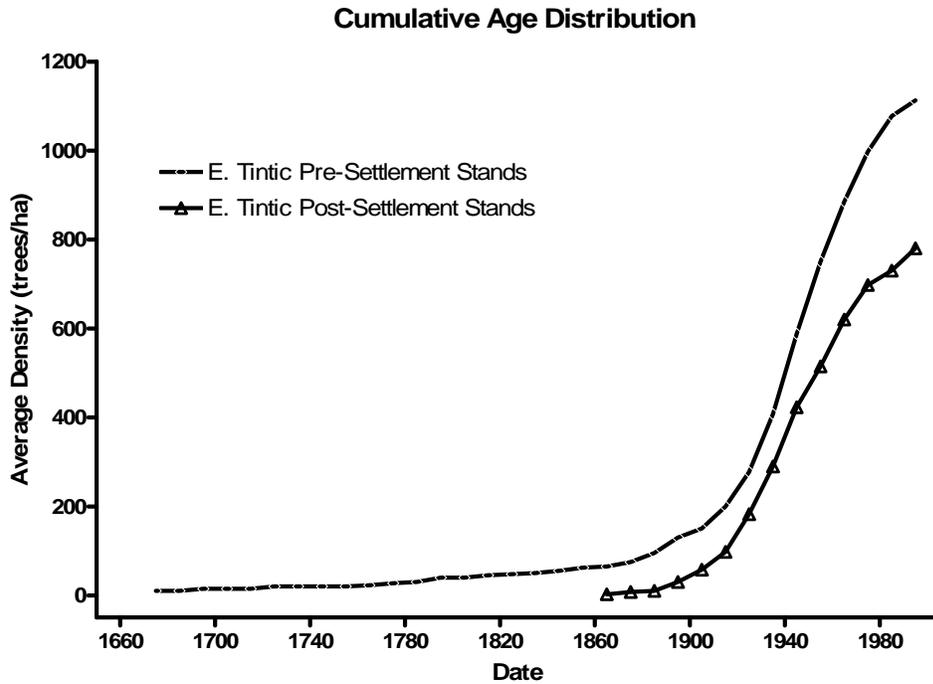


Figure 4. Comparison of cumulative age distributions on pre- (oldest tree greater than 140 years old) and post-settlement (all tree less than 140 years old) plots in Utah for the average density per plot where all trees were aged.

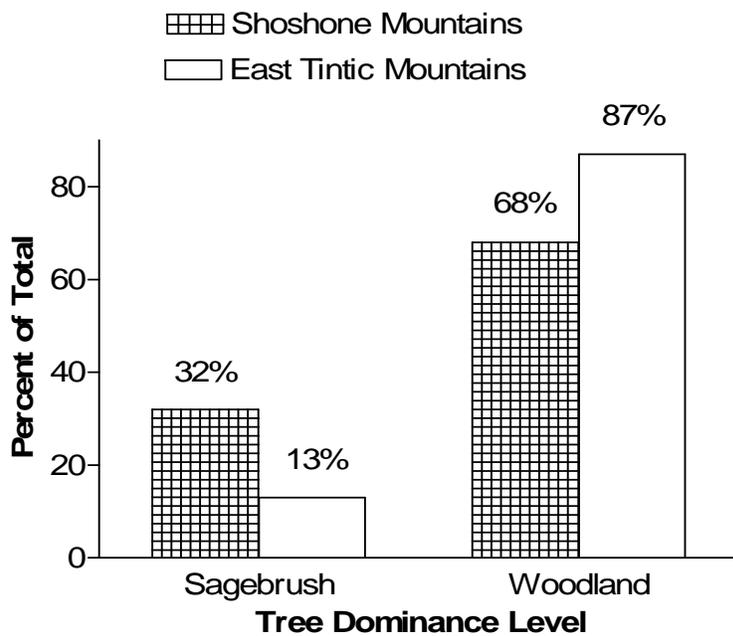


Figure 5. Comparison between the sampled plots in the Shoshone Mountains, Nevada, and the East Tintic Mountains, Utah for the presence or absence of woodland trees.

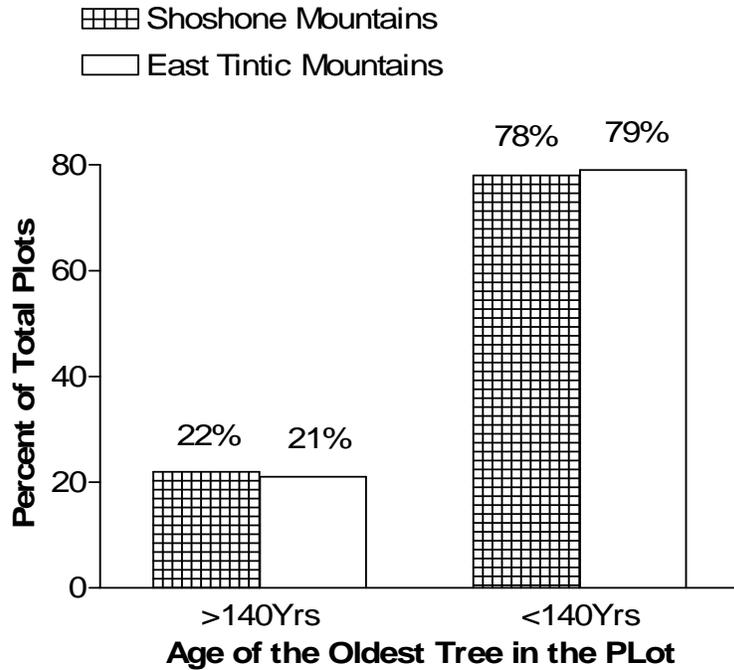


Figure 6. Comparison between the sampled plots in the Shoshone Mountains, Nevada, and the East Tintic Mountains, Utah for the presence or absence of trees greater than or less than 140 years old.

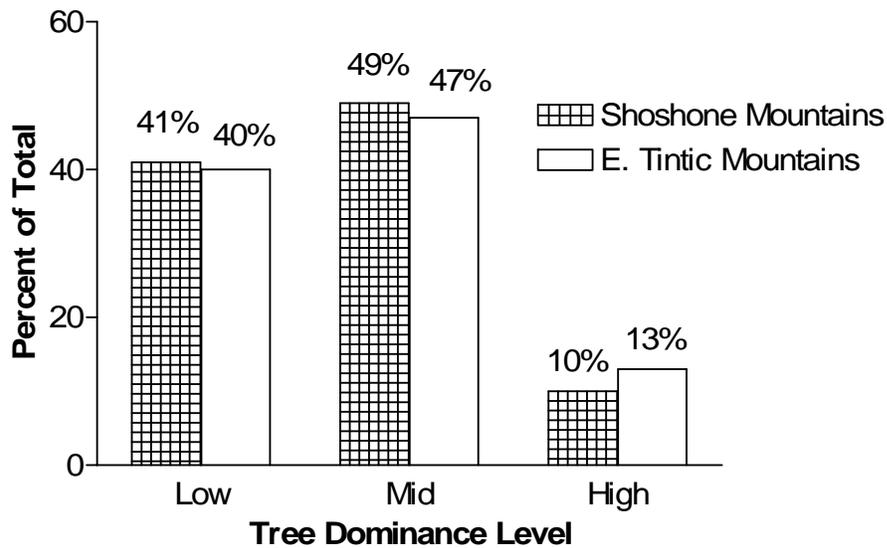


Figure 7. Comparison between the sampled plots in the Shoshone Mountains, Nevada, and the East Tintic Mountains, Utah for the percentage of the total plots that have low, mid or high tree dominance.

WESTERN JUNIPER (*JUNIPERUS OCCIDENTALIS*) SUCCESSION: CHANGING FUELS AND FIRE BEHAVIOR

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Study Description. Since European settlement, western juniper (*Juniperus occidentalis* spp. *occidentalis*) woodlands have had one of the most dramatic expansions of any plant community in the Intermountain West (Burkhardt and Tisdale 1969, Burkhardt and Tisdale 1976, Miller and Rose. 1994, Miller et al. 2000, 2005, Young and Evans 1981). The greatest expansion of western juniper in the Intermountain West occurred between 1870 and 1920 (Miller et al. 2005). Burkhardt and Tisdale (1976) indicated that increased rates of western juniper encroachment on the Owyhee Plateau of southwestern Idaho started around 1880.

The consequences of western juniper encroachment include changes in species composition and vigor (Burkhardt and Tisdale 1976, Miller et al. 2000, Young and Evans 1981). The changes in species composition and vigor could lead to changes in available fuel for fire, limiting the effectiveness of fire as a control for western juniper (Young and Evans 1981). To predict the effectiveness of fire, it is necessary to identify and understand the changes in fuel loading due to western juniper encroachment and succession.

The objectives of this study are to 1) measure changes in fuel loading due to encroachment and succession of western juniper in sagebrush communities; and 2) predict the resulting fire behavior using models to evaluate changes in fire spread and resulting fire size.

Methods and Study Area. The Owyhee Plateau is located in Owyhee County, Idaho between the towns of Grand View, Idaho and Jordan Valley, Oregon. The Owyhee Plateau is characterized by small mountains separated by deep canyons, rocky tablelands, and rolling plains ranging in elevation between 1200 and 2100 m. Average annual precipitation ranges between 30 and 56 cm, and is primarily received in fall, winter and early spring. Average temperatures range from -6.6 °C in January to 34.5 °C in July. Geologically, the area is mainly made up of a rhyolitic plateau. Soils vary from shallow rock outcrops to moderately deep gravelly, sandy, or silt loams.

Cover types were differentiated by encroachment and resulting successional gradients of western juniper in mountain big sagebrush (*Artemisia tridentata* subsp. *vaseyana*) and low sagebrush (*Artemisia arbuscula*) steppe. Five stages of encroachment and succession were used to classify the change from sagebrush steppe to mature western juniper for each sagebrush type. These successional stages included: 1) steppe (R3), 2) woodland initiation (W1), 3) open young woodland (W2), 4) young multi-story woodland (W4), and 5) old multi-story woodland (W5). Stages 2, 3 and 4 correspond to the Phase I, Phase II and Phase III stages described by Miller et al. (2005). Cover types were further separated by the associated sagebrush species.

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Fuels data were collected using methods described by Brown et al. (1982), with some modifications to the sample plot design. Herbaceous, shrub and downed dead woody biomass was estimated. BEHAVE was used to predict flame length and rate of spread for the new models created from the collected data. Fire behaviors for each fuel model were compared under a variety of environmental and topographic conditions. Predictions with BEHAVE were made at three fuel moisture levels: low (3%, 4%, 5% and 70%), medium (6, 7, 8 and 120%), and high (12, 13, 14 and 170%) for 1-, 10-, 100-hour and live fuel moisture conditions, respectively. Slope was set at 30%. Wind speeds were entered in as a range from 0 to 40 at 8 kph intervals.

Results and Discussion. Among the mountain big sagebrush cover types, changes in fuels due to western juniper encroachment and resulting succession are identified in Figure 1. The trend shows initial increases followed by slow declines, with minor exceptions in mountain big sagebrush W4 1-hour and W5 100-hour fuels. There is an initial increase in fuel bed depth followed by decline. As the cover type shifts from R3 to W1 there is a 62% increase in total fuels and a 12% increase in fuel bed depth, giving W1 the greatest total fuel load and fuel bed depth. The largest proportion of this increase is made up of 100-hour fuels (154%), corresponding with a 35% increase in shrub cover and in shrubs having basal diameters greater than 2 cm. An increase in 1-hour (41%) and 10-hour (55%) fuels can also be observed. The increase in 1-hour fuels is made up of litter and shrub material, which offset a decline in herbaceous biomass from its high in the R3 cover type.

The downward trend of total fuels and fuel bed depth initiate as succession continues from mountain big sagebrush W1 to W2 with declines of 31% and 29%, respectively. A reduction in total shrub biomass by 32% accounts for nearly all of the reduction in fuels and fuel bed depth. Shrub biomass continues to decline as the cover types shift from W2 to W4. However, the reduction in total shrub biomass (45%) is mitigated by a sharp increase in litter (440%). Litter in the W4 cover type is largely made up of western juniper needle cast found under the canopy. The overall result is a decline in total fuels (10%) and fuel bed depth (57%). At the final cover type, W5, a total fuels increase of 20% was primarily composed of a large increase in downed dead woody material mainly from dead mountain-mahogany. Fuel bed depth continued to decrease by 69% to a low at 0.4 cm.

The low sagebrush cover type also exhibits changes in fuels that follow encroachment and successional change (Figure 2). There is an increase in total fuels throughout the majority of the cover types except for the shift to W5 when there is a decline. A 90% increase in total fuel weight from R3 to W1 is followed by another 3% increase for W2. The only decline is observed between W2 and W5, with a 47% reduction in total fuels. The majority of the increase in total weight and fuel bed depth can be attributed to the increase in shrub biomass. There is a simultaneous increase in herbaceous biomass as well. The fuel bed depth follows total fuel weight closely; the only decline is in the shift from W2 to W5.

When total fuel weight and fuel bed depth for the mountain big sagebrush and low sagebrush encroachment and successional cover types were compared, it is apparent that they did not follow the same trends. Dissimilarities are related to the differing reactions of the sagebrush component to western juniper encroachment and succession. In the low sagebrush cover types the sagebrush component persisted in the mid- cover types whereas it declines in the mountain big sagebrush cover types with advancing succession.

The changes identified in fuel composition, total fuel weight, and fuel bed depth resulted in changes in modeled fire behavior. For mountain big sagebrush, flame length (FL) and rate of spread (ROS) increased between the R3 and W1 cover types and then declined throughout the remaining cover types, similar to changes observed in the 10-hour and live fuel loading (Figure 3, 4, 5 and 6). Flame lengths for R3 ranged from 0.5 to 2 m with winds ranging from 0 to 40 kph at high fuel moisture

conditions, which occur during prescribed burns. Low fuel moisture conditions occur during wildland fire conditions, with flame lengths reached 3.8 m.

Along the successional gradient from mountain big sagebrush R3 to W1, flame lengths increased as total fuel weight and fuel bed depth increased. Flame length averaged a 44% increase over R3, with values ranging from 0.7 m at 0 kph to 3.7 m at 40 kph and a maximum flame length of 6.8 m at 40 kph and low fuel moisture conditions. Rate of spread averaged an 82% increase from R3 to W1. Most of the increase was attributed to R3 reaching its modeled wind limit (the point when the model predicts there will no longer be an increase in FL or ROS with an increase in wind speed) at a lower wind speed than that of W1. At the point where R3 reaches the modeled wind limit (11.0 kph) there is only a 6% difference in ROS between the two cover types. Due to the modeled wind limit R3 had a maximum ROS 61% lower than that of W1 at high fuel moisture. The differences between the two modeled ROS is even greater at low fuel moisture conditions with W1 having a 137% increase over R3. The effects of the modeled wind limit are more pronounced when comparing ROS than FL.

The modeled fire behavior for low sagebrush cover types reflected the changes in fuel composition, total fuel weight, and fuel bed depth similar to those observed in the mountain big sagebrush cover types. Flame length increased between R3 and W2, first by an average of 130% from R3 to W1 and another 9% to W2, at high fuel moisture conditions (Figure 7). A 73% average reduction was observed from the high at W2 to the low at W5. Slightly lower rates of change are observed at low fuel moisture conditions due to the fuel model having higher wind limits (Figure 8).

Rate of spread had a pattern similar to that observed in the FL. ROS increased from low sagebrush R3 to W2 by an average of 270% from R3 to W1 and another 12% to W2 (Figure 9). The reduction in ROS from W2 to W5 was 80%. The percent change for ROS was similar at the low fuel moisture conditions even with the changing wind limits (Figure 10).

Mountain big sagebrush had higher FL and ROS than those observed in low sagebrush for the early-cover types (R3 and W1), attributed to the greater biomass and height associated with mountain big sagebrush. In the mid- and late-successional cover types (W2, W4, and W5), low sagebrush exhibits higher FL and ROS. This can be explained by the retention of the sagebrush component in the later cover types for low sagebrush cover types.

Of the fuel model parameters, changes in fuel bed depth appeared to have the largest influence on FL and ROS in mountain big sagebrush. For example, the difference between W1 and W5 was a 25% reduction in total fuel and a 91% reduction in fuel bed depth leading to the 91% decrease in FL and 99% decrease in ROS.

Summary and Conclusions: Changes in fuels and fire behavior are effected by western juniper encroachment and resulting succession. The mountain big sagebrush cover types were characterized by increasing fuel loads in the early- and early mid-successional stage cover types followed by declining fuel loads in the remaining mid- and late-successional stage cover types. Fuel loads were also dominated by differing components across the successional gradient, early-successional stage cover types by herbaceous and shrub components, mid-successional stage cover types by shrub and litter, and late-successional stage cover types by downed dead woody components.

The low sagebrush cover types were also affected by western juniper encroachment and resulting succession. Low sagebrush cover types had increasing fuel loads and fuel bed depth in the early- and mid-successional stage cover types and declining fuel loads in the late-successional stage cover type. Again, fire behavior was strongly related to the changes in fuel loads, composition, and fuel bed depth.

Changes in fuel loads, composition, and fuel bed depth changed fire behavior, which in turn affected fire size. The percent cover that consisted of mid- or late-successional stage cover types had the strongest influence on the resulting fire size.

Prescribed fires and fire use strategies will be more effective in controlling western juniper encroachment if they occur in the earlier stages of succession before the shift to declining fuel loads takes place. Higher fuel loads give the manager a wider range of environmental conditions under which the prescribed fire can achieve its objectives in a safe and contained manner. The combination of young western juniper being more susceptible to fire damage and fuel loads that allow the manager more opportunity to perform a prescribed burn increase the chances of minimizing the encroachment of western juniper into sagebrush grasslands.

Conclusions and management suggestions mentioned here should only be applied to western juniper woodlands without the presence of pinyon pine (*Pinus* spp.). Results may vary because juniper does not burn as readily as pinyon pine. This may be due to the higher flammability of pinyon pine foliage. Pinyon pine also grows in more mesic sites with more understory fuels and higher tree densities (Bradley et al. 1991).

Acknowledgements. The authors thank the USDI Bureau of Land Management, Lower Snake River District, Idaho for assistance in completing this study. Additional thanks to Renan Bagley, Jeremy Kleinsmith for field support and Eva Strand for GIS assistance. This research was supported by the USDA/USDI Joint Fire Sciences Program and the USDA Forest Service 01-JV-11222073-108.

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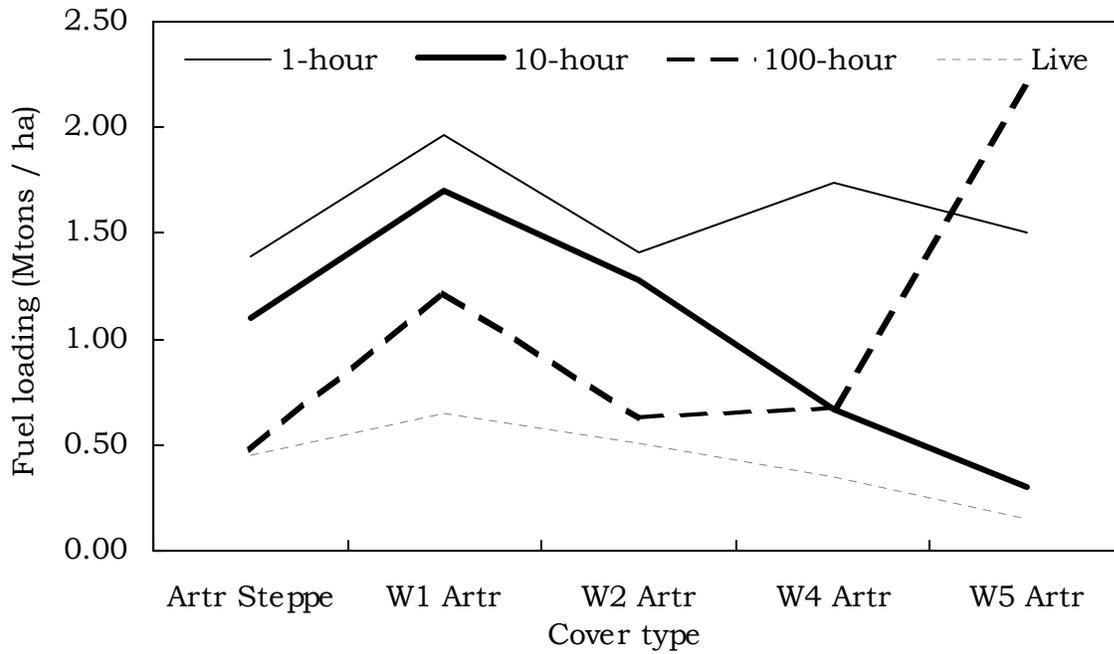


Figure 1. Changes in fuel loading associated with western juniper encroachment and advancing succession in mountain big sagebrush cover types (Artr).

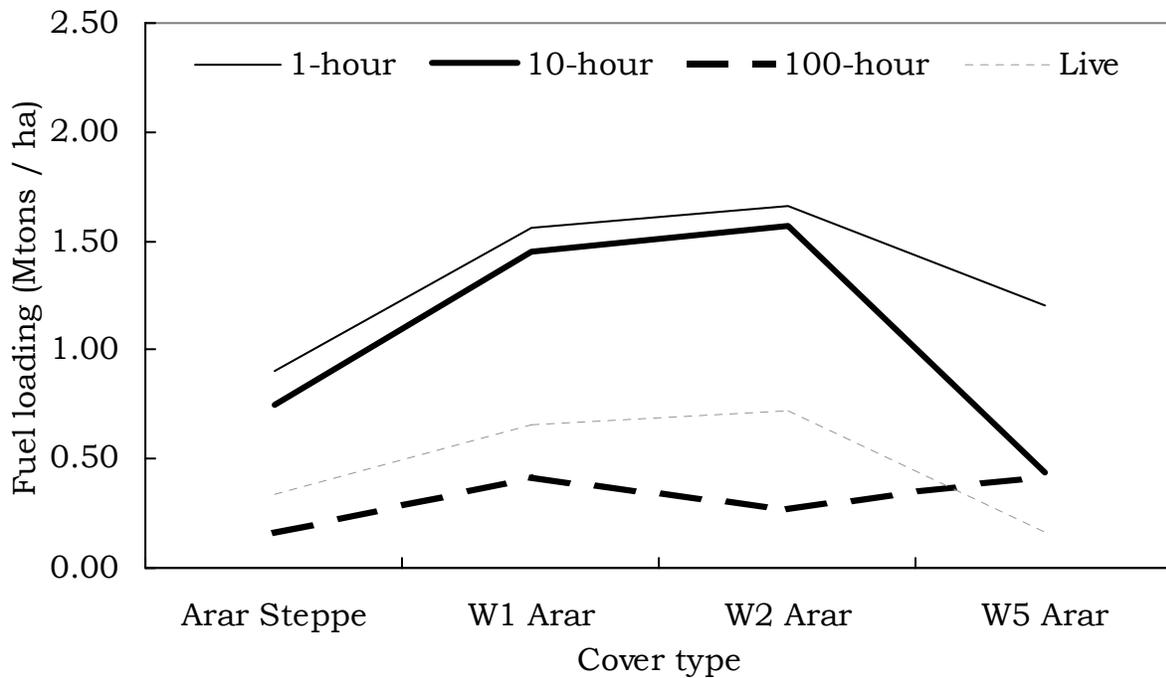


Figure 2. Changes in fuel loading associated with western juniper encroachment and advancing succession in low sagebrush cover types (Arar).

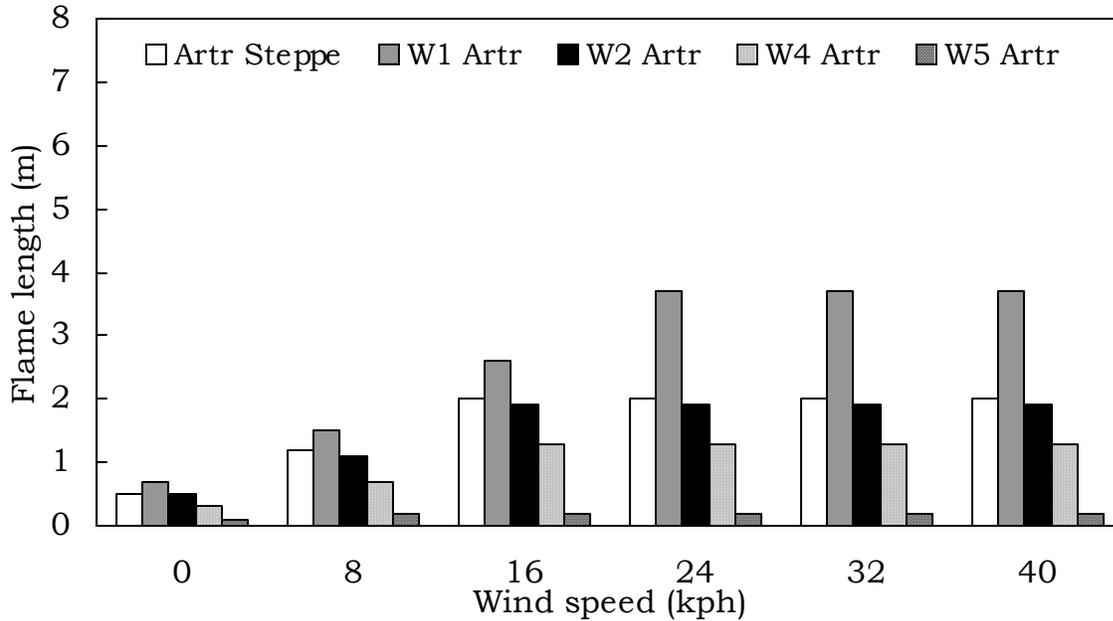


Figure 3. Changes in flame length compared among mountain big sagebrush (Artr) juniper encroachment and successional cover types at high fuel moisture conditions. Fuel moistures equal 12% for 1-hour, 13% for 10-hour, 14% for 100-hour, and 170% for live fuel.

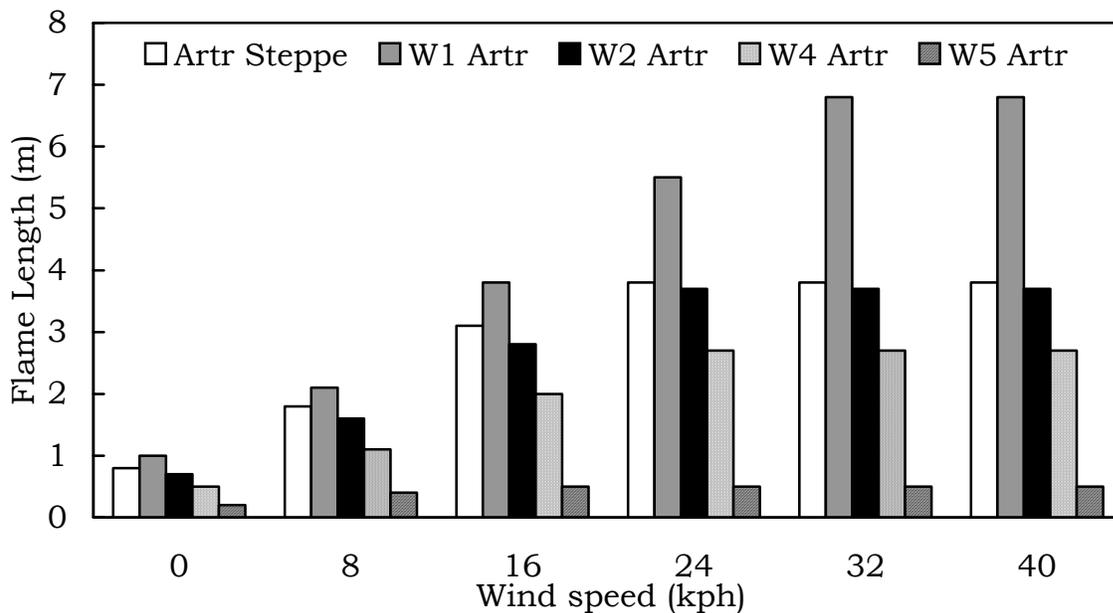


Figure 4. Changes in flame length compared among mountain big sagebrush (Artr) juniper encroachment and successional cover types at low fuel moisture conditions. Fuel moistures equal 3% for 1-hour, 4% for 10-hour, 5% for 100-hour, and 70% for live fuel.

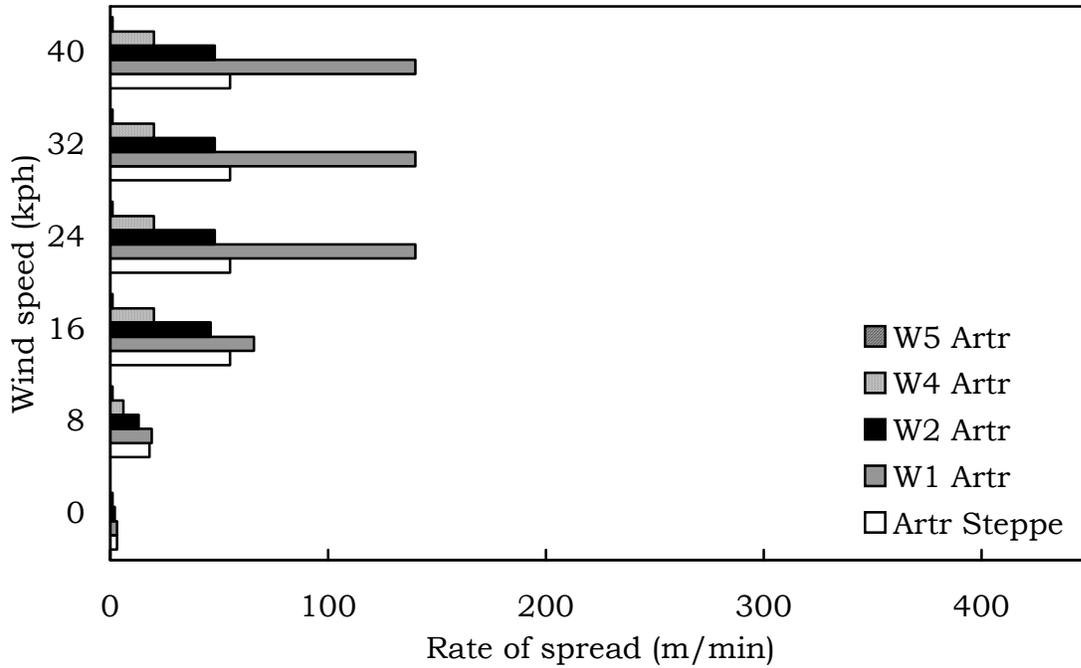


Figure 5. Changes in rate of spread compared among mountain big sagebrush (Artr) juniper encroachment and successional cover types at high fuel moisture conditions.

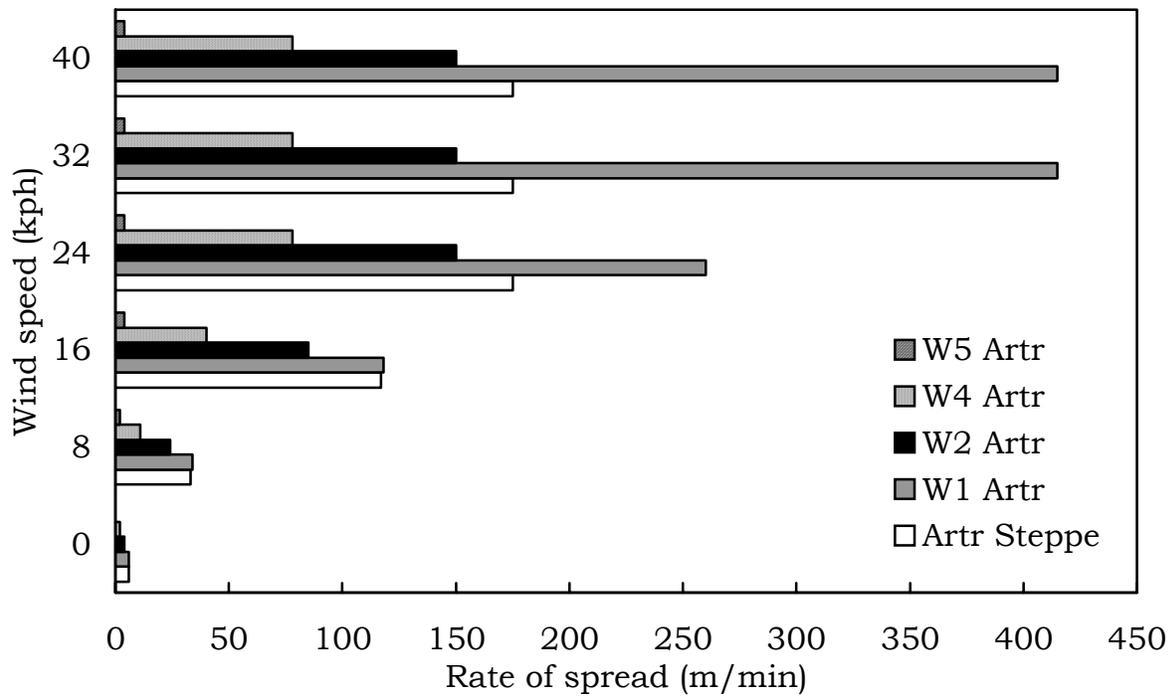


Figure 6. Changes in rate of spread compared among mountain big sagebrush (Artr) juniper encroachment and successional cover types at low fuel moisture conditions.

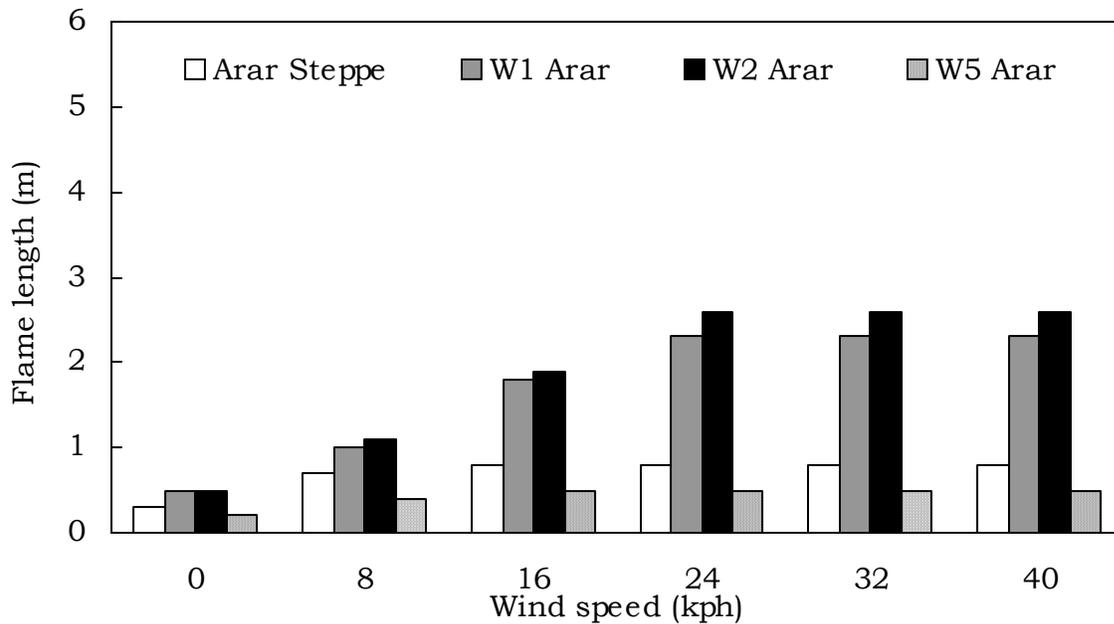


Figure 7. Changes in flame length compared among low sagebrush (Arar) juniper encroachment and successional cover types at high fuel moisture conditions. Fuel moistures equal 12% for 1-hour, 13% for 10-hour, 14% for 100-hour, and 170% for live fuel.

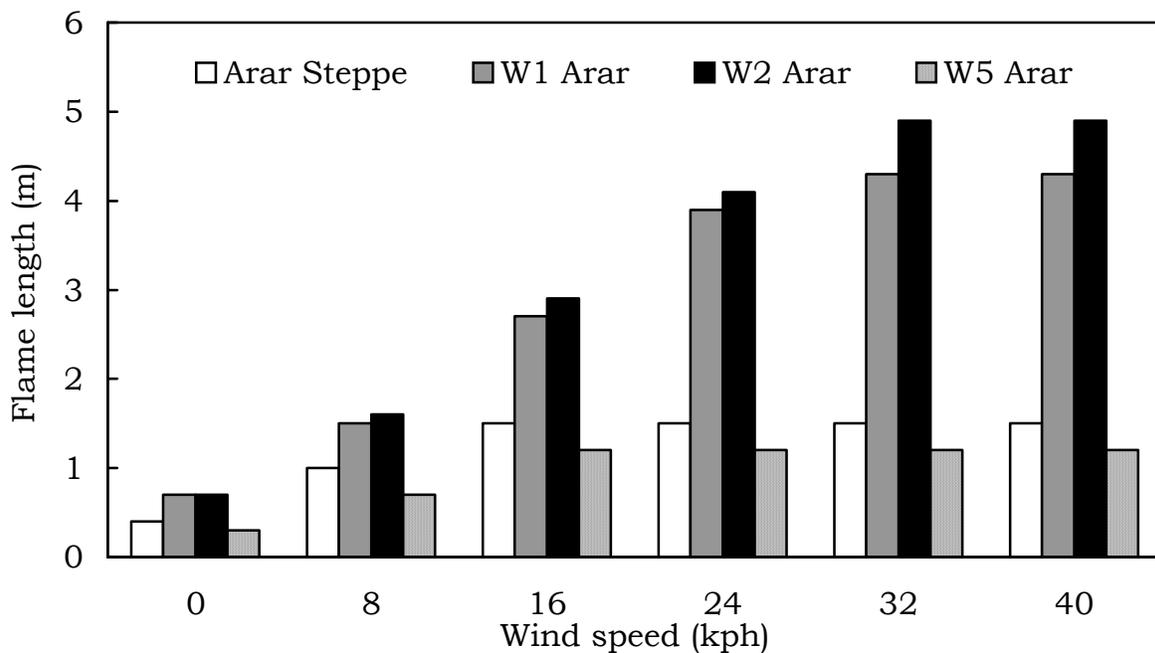


Figure 8. Changes in flame length compared among low sagebrush (Arar) juniper encroachment and successional cover types at low fuel moisture conditions. Fuel moistures equal 3% for 1-hour, 4% for 10-hour, 5% for 100-hour, and 70% for live fuel.

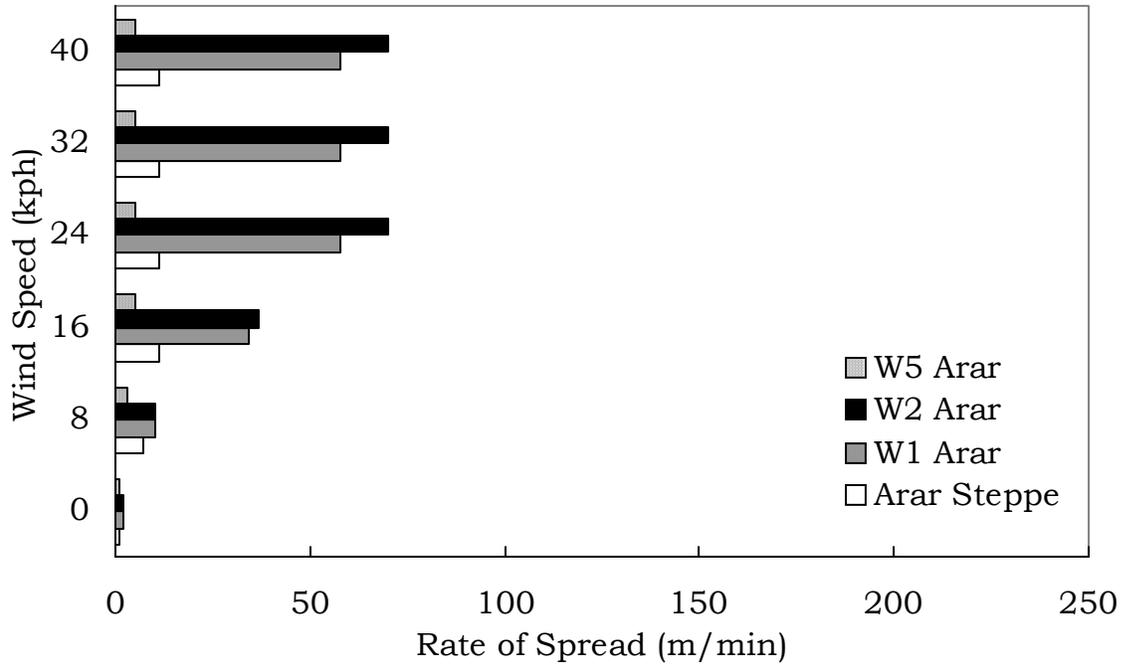


Figure 9. Changes in rate of spread compared among low sagebrush (Aara) juniper encroachment and successional cover types at high fuel moisture conditions.

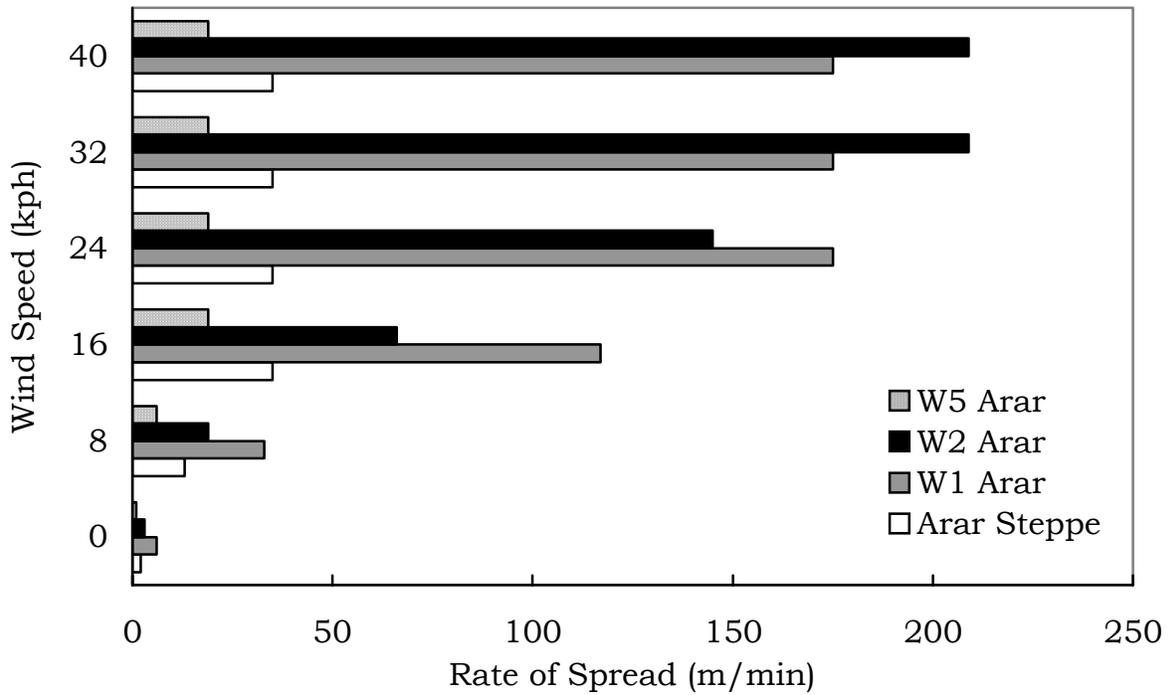


Figure 10. Changes in rate of spread compared among low sagebrush (Arar) juniper encroachment and successional cover types at low fuel moisture conditions.

EFFECTS OF HERBACEOUS SPECIES REMOVAL, FIRE AND CHEATGRASS (*BROMUS TECTORUM*) ON NUTRIENT AVAILABILITY IN SAGEBRUSH STEPPE

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Study Description. Soil nutrient availability (bioavailability) is a principle factor constraining vegetative growth and organizing competitive interactions among species (Goldberg, 1990; Grover, 1997). Availability of nutrients such as nitrogen and phosphorus determine if ecosystems can resist or are prone to invasion by exotic plant species (Herron, et al., 2001). Pulses of nutrient availability are robust predictors of ecosystem invasibility (Davis et al., 2000). If pulses coincide with optimal conditions for plant growth, a fast growing exotic species such as *Bromus tectorum* may co-opt nutrients and water faster than competing species. Soil nutrient levels in sagebrush-grass ecosystems of the western United States are highly spatially and temporally variable (Doescher et al., 1984; Burke, 1989). Does this fact imply that these ecosystems are inherently at risk of invasion? It is impossible to draw any conclusions, because we are unaware of any literature that defines a threshold level of nutrient availability at which an ecosystem becomes invasible. The purpose of this study was to investigate the role of soil nutrient availability in facilitating site invasion by *Bromus tectorum*

There are no universally accepted protocols or ideal methodologies to unambiguously gage soil nutrient availability. Phosphorus and nitrogen have proved especially difficult to correlate an extractable pool with true bioavailability (Saggar et al., 1992). Anion and cation resin exchangers integrate soil nutrient availability over time and as such have the potential to gage true bioavailability (Binkley et al., 1986). Such devices are not perfect in their own right (Sherrod et al., 2003; Johnson et al., 2005), but have in some cases been shown to relate to nutrient release kinetics and bioavailability of particular nutrients (van Raij et al. 1986; Searle, 1988; Yang and Skogley, 1992; Abrams and Jarrell, 1992).

Methods. We used resin capsules to integrate soil nutrient availability of N, P, Ca, Mg, Na, K, Fe, and Mn at two sagebrush-grassland elevational transects in the east Tintic range of Utah and the Shoshone range of Nevada. At each state, sites were selected to encompass a precipitation-vegetation gradient; 5 in Utah and 4 in Nevada. In the fall of 2001, treatments applied to replicated plots at each site included prescribed burning, herbaceous vegetation removal (2 levels Utah, 3 levels Nevada), and controls. *Bromus tectorum* was over-seeded in small subplots within each treatment. Following treatment application, resin capsules were installed in each plot at approximately 15 cm in shrub interspaces; one in an unvegetated area and one in a cheatgrass over-seeded area. Nutrient availability was integrated during 4 separate periods (over-winter and growing season) for 2 years.

Results. *Influence of herbaceous vegetation removal* - Relative to controls, complete herbaceous vegetation removal increased availability of NO_3^- (both states) and Ca^{+2} and Mg^{+2} (Nevada only), but only during the second sampling period (growing season) (Figure 1).

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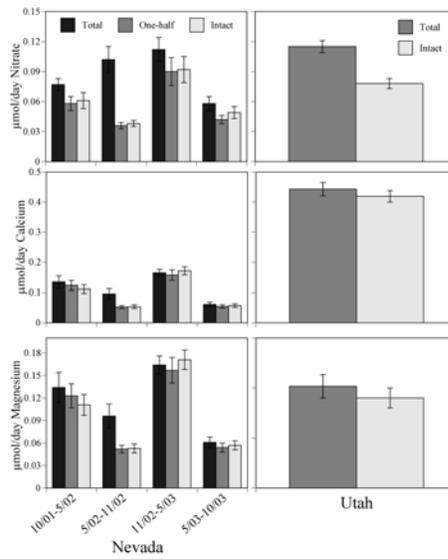


Figure 1. Influence of herbaceous vegetation removal on soil nutrient availability.

Influence of prescribed burning - Availability of K and ortho-P (both states) and NO_3^- (Nevada only) were greater on prescribed burned plots than the control plots (Figure 2).

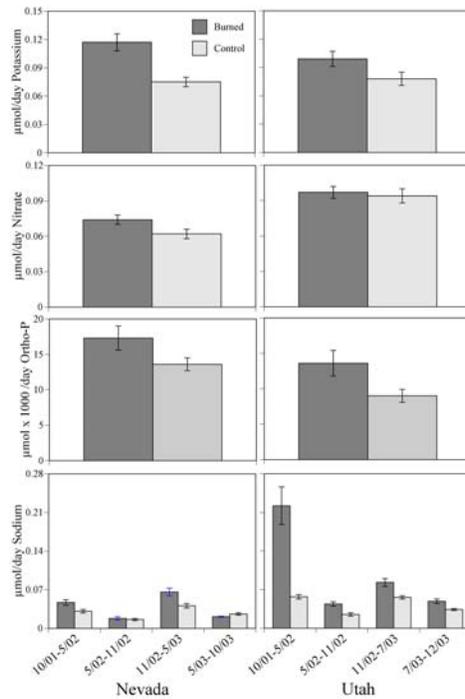


Figure 2. Influence of prescribed burning on soil nutrient availability.

Influence of elevation - For Utah, availability of ortho-P, K, Ca, Mg and Fe generally increased with increasing elevation. Nutrient availability did not display strong trends with elevation for Nevada (Figure 3).

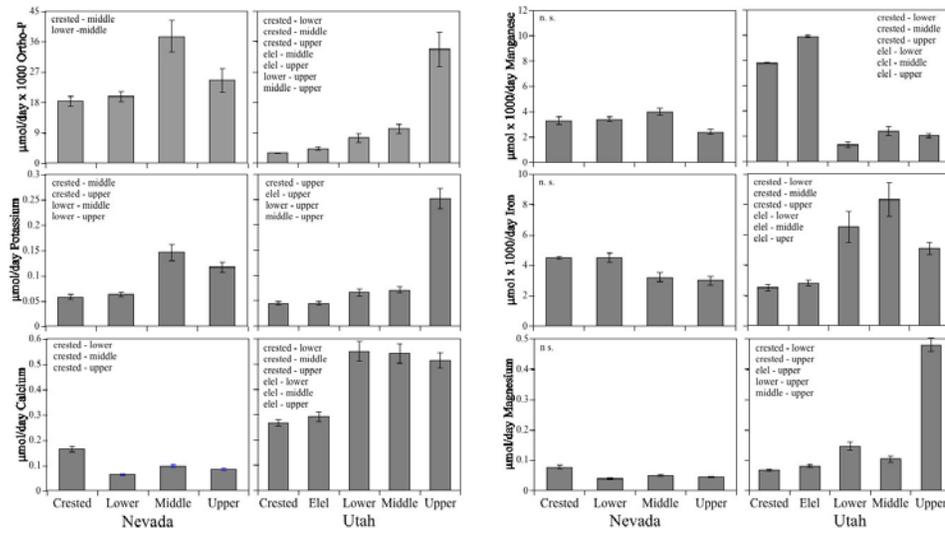


Figure 3. Influence of elevation on soil nutrient availability. Elevation pairs in panels denote significance.

Influence of season - For Nevada, availability of Ca, Mg, K, and Fe were greatest during over-winter integration periods; whereas for Utah, nutrient availability was more erratic among sampling periods (Figure 4).

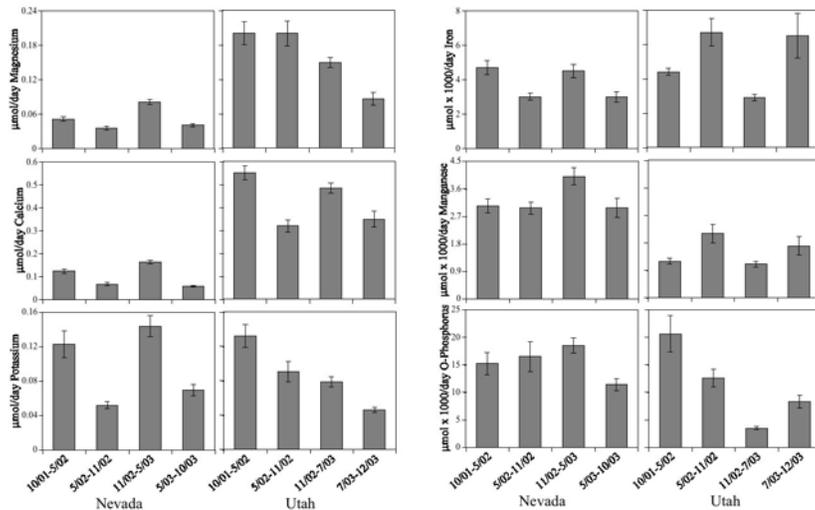


Figure 4. Soil nutrient availability during the four resin sampling periods.

Influence of over-seeding - Over-seeding with *B. tectorum* interacted with the burn treatment to influence availability of Ca, Mg, and Fe (Nevada sites only) (Figure 5).

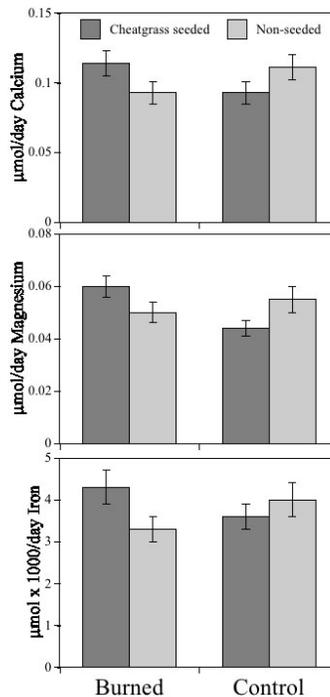


Figure 5. Influence of over-seeding of *B. tectorum* on soil nutrient availability (Nevada only).

Implications. Patterns of nutrient availability can be explained by a combination of decreased root uptake in relation to mineralization, differences in soil water content with season and elevation, and nutrient release from vegetation and soil as a consequence of prescribed burning. Our data suggest that removal of herbaceous vegetation via overgrazing, herbicide application and prescribed burning can raise NO_3^- availability and risk invasion by nitrophilic species such as *B. tectorum*. Moreover, for some soil nutrients, availability is out of phase with most plant growth, which suggests that plants capable of exploiting these nutrients during cold periods may have a competitive advantage. Additional needs are to determine threshold levels of soil nutrient availability which facilitate plant invasions.

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EFFECTS OF HERBACEOUS SPECIES REMOVAL, FIRE, AND CHEATGRASS (*BROMUS TECTORUM*) ON SOIL WATER AVAILABILITY IN SAGEBRUSH STEPPE

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Study Description. Over the past several decades, cheatgrass (*Bromus tectorum*) has been continually expanding in the sagebrush steppe ecosystem. There has not been a lot of research that examines why cheatgrass is able to invade these communities. This study looks at how changes in vegetation, and in consequence, changes in available soil water may play a part in this invasion. Available soil moisture is one of the main limiting factors in sagebrush steppe communities (Briones, Montana, and Ezeurra 1998). Many factors, such as vegetation, slope, aspect, and soil physical properties, can affect the total water availability (>-1.5MPa) in the soil. The objectives of this study were to determine the effects of elevation, differing amounts of vegetation, fire, and the presence of cheatgrass on total water availability in the soil.

Methods. In the spring of 2001 plots were setup on nine sites in Utah and Nevada. Three of the plots in both Utah and Nevada were located in sagebrush stands along an elevational gradient. In Utah the elevations were 5600', 6840', and 7460' and in Nevada they were 6400', 7180', and 7800'. The fourth sites in Utah and Nevada were located in a stand of crested wheatgrass (*Agropyron cristatum*) having elevations of 5340' and 6760', respectively. The final site was located in Utah in a stand of squirreltail (*Elymus elymoides*) at an elevation of 5240'. The treatments included a burn treatment (burned or not burned) and a vegetation removal treatment (intact or 100% removal). Within each combination of these two treatments, subplots were seeded with cheatgrass. Each treatment combination was replicated three times per site in a random complete block design for a total of 12 plots per site. The plots were circular with a diameter of 3.4m. The vegetation removal was accomplished by spraying the understory vegetation with Roundup in the spring of 2001. In the following October, the BLM and Forest Service fire crews carried out the burn treatments. Following the burn treatment, subplots within each plot were seeded with cheatgrass.

To measure soil moisture, gypsum blocks were installed in each plot both in the seeded subplot and in the unseeded area of the plot at three depths, 1-3, 13-15, and 28-30 cm. Thermocouples were also buried with the gypsum blocks to measure soil temperature. The sensors were read every minute and hourly averages were recorded onto CR-10X Campbell Scientific microloggers. Rain gauges and air temperature gauges were also installed on each site to record environmental conditions. The total days of available water for each treatment was totaled from April 1 to June 30 in 2002 and 2003. This is the period when cheatgrass and the other species on the site are actively growing and utilizing the soil moisture.

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Results and Discussion.

Elevation

Elevation was a significant factor on total time of available water. When comparing the sagebrush sites in both Nevada and Utah, time of available water was significantly higher on the higher elevation sites than the lower elevation sites (Fig.1). There was an average of about 18 more days of available water on the upper sites than the lower sites. These results were expected because of the differences in precipitation on these sites (Table 1). The more resources you add to a system the more resources available.

Vegetation Removal

Vegetation removal did not significantly affect the time of available water at the 1-3cm depth over all sites. Time of available soil water, over all sites, at the 13-15cm and 28-30cm depths was significantly higher on the 100% removal plots than the intact plots (Fig. 2). These differences were only seen during the first growing season after the treatments (2002). At the 13-15cm depth on the sagebrush sites, the average total days of available water between April 1 and June 30 on the intact and 100% removal plots were 53.5 and 60.7 days respectively. At the 28-30cm depth averages on the sagebrush sites for the intact and 100% removal plots were 60.3 and 68.4 days respectively. When looking at each individual site, the middle Utah site was the only site that vegetation removal resulted in a significant difference in available water with a difference of about 20 day of available soil water.

Water loss from the soil can occur in two ways, evaporation and transpiration. When vegetation is lost less transpiration and more evaporation occurs (Coronato and Bertiller 1996). The change in transpiration mainly affects the lower portion of the soil profile, whereas the upper portion of the soil profile is mainly affected by evaporation. The upper portion of the soil profile still experiences evaporation, which can be higher due to less surface cover from vegetation. This may explain why the deeper portions of the soil profile on the 100% removal plots had more days of available water than the intact plots.

Burn Treatment

Results for the burn treatment were very similar to the removal treatment. The burn treatment only showed a significant effect on the sites during the first year and only at the 13-15cm and 28-30 depths (Fig. 3). Burning resulted in significantly more days of available water compared to the unburned plots. At the 13-15cm depth, the average total days of available water between April 1 and June 30 on the no burn and burn plots were 53.8 and 60.3 days respectively. At the 28-30cm depth averages for the no burn and burn plots were 60.5 and 68.2 days respectively. This is also probably due to the loss of vegetation and a decrease in transpiration. By the second growing season (2003) the burned vegetation had a chance to start growing back, which may explain why there was not a significant difference during the second season.

Seeding Treatment

The seeding treatment did not produce any significant differences. This was slightly unexpected since cheatgrass appears to out-compete other species by using up available resources (Aguirre and Johnson 1991). Cheatgrass establishment in Utah during the first year was very low which may help explain these results.

Removal/Burn

The interaction of both the removal and the burn treatments had similar results as each treatment alone in that significant differences were only seen in the first growing season (2002) and at the 13-15cm and 28-30cm depths. The intact/no burn plot had the fewest days of available water followed by intact/burn, 100% removal/no burn, and then 100% removal/burn (Fig. 4).

Conclusion. The loss of vegetation through fire, drought, or grazing leaves extra resources available. This increase in available soil water may have an affect on the susceptibility of the community to invasion by weedy species. However the timing and the environmental conditions at the time of available water also play a part in the susceptibility of a community to weed invasion. In summary, water availability may increase, as vegetation is lost. Water availability also depends on the elevation and how much precipitation it receives. These changes in water availability can have an effect on the community and its susceptibility to weed invasion.

Acknowledgements. Funding was provided by Joint Fire Sciences Program and Pitman-Robbinson Federal Aide Project W-82-R. Assistance was provided by biology technicians at the USDA-Forest Service RMRS Shrub Sciences Laboratory – Stephanie Carlson, Suzette Clements, Matthew Pyne, and Jeff Taylor, and by the UDWR Great Basin Research Center.

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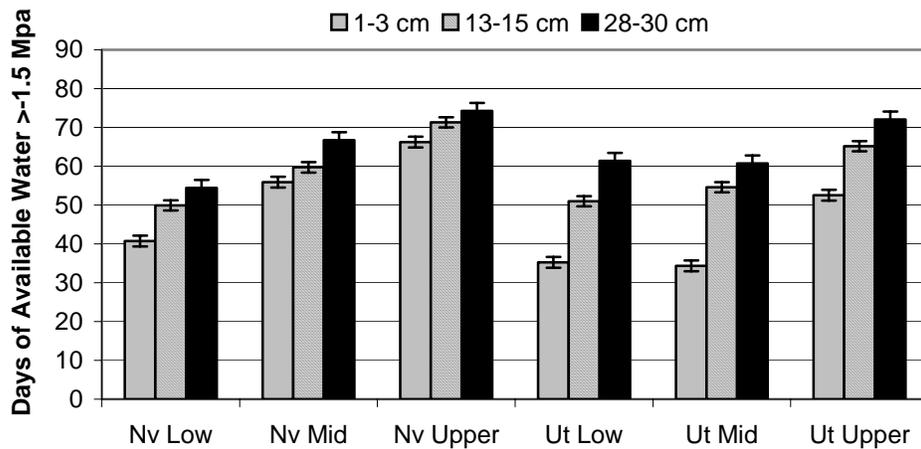


Figure 1. Average days of water availability at 1-3, 13-15, and 28-30 cm on all plots from April to June 2002 and 2003. Values are mean \pm 1 SE. Days of available soil water were measured when soil matric potential was $>$ -1.5 Mpa.

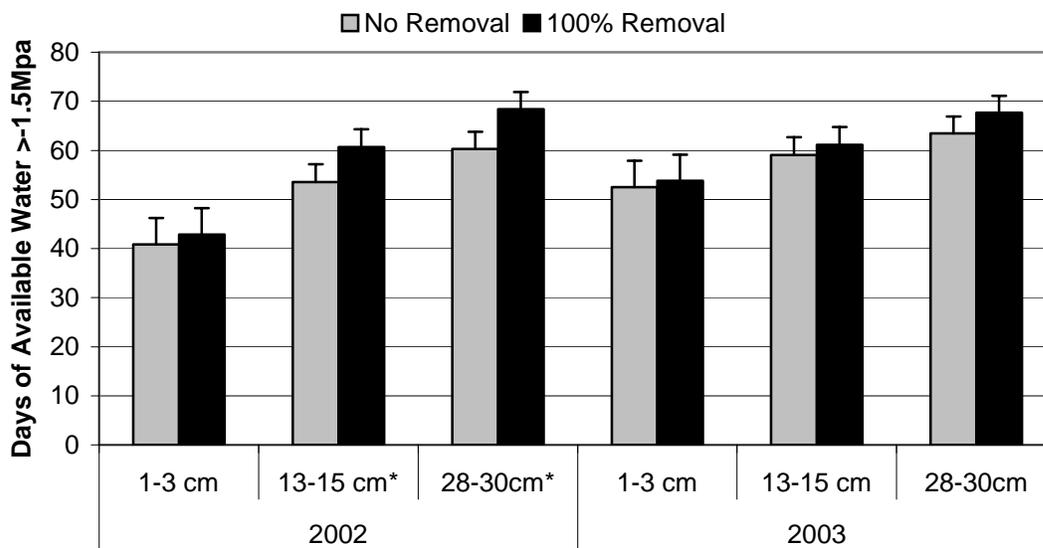


Figure 2. Average days of available water at 1-3, 13-15, and 28-30 cm on removal plots for the low, middle, and upper sites in Nevada and Utah. Values are mean \pm 1 SE. Days of available soil water were measured when soil matric potential was $>$ -1.5 Mpa. *Significant differences between removal treatments were only significant during 2002 at 13-15 and 28-30 cm.

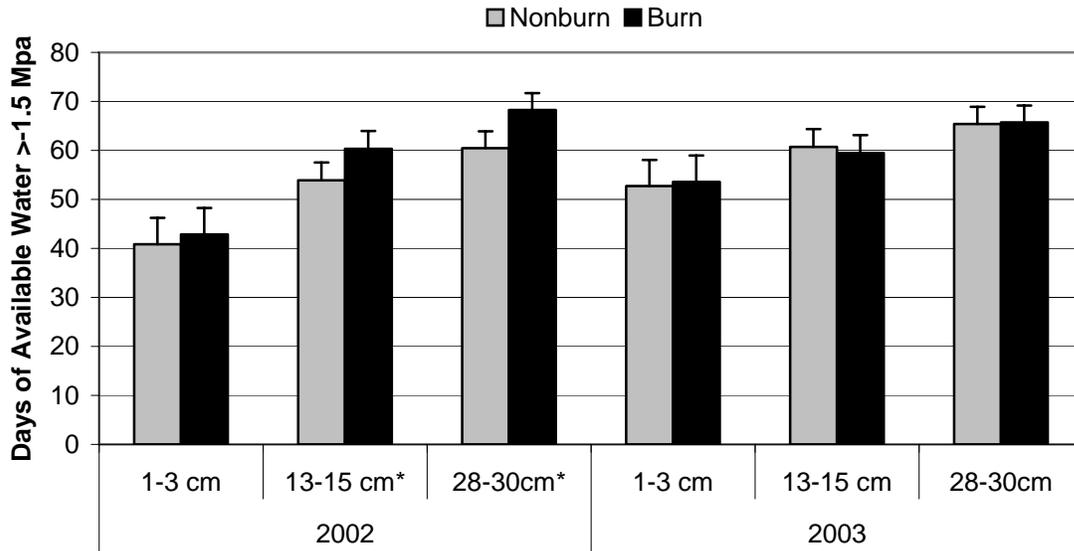


Figure 3. Average days of available water at 1-3, 13-15, and 28-30 cm on burn plots for the low, middle, and upper sites in Nevada and Utah. Values are mean \pm 1 SE. Days of available soil water were measured when soil matric potential was > -1.5 Mpa. *Significant differences between nonburn and burn were only seen during 2002 at 13-15 and 28-30 cm.

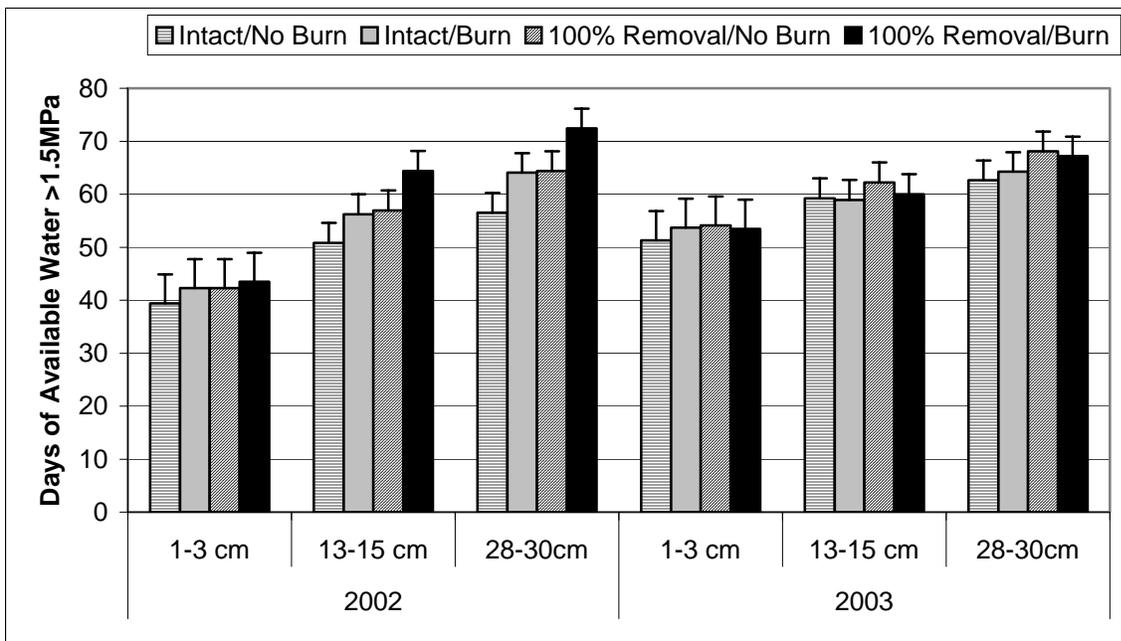


Figure 4. Average days of available water at 1-3, 13-15, and 28-30 cm on removal and burn plots for the low, middle, and upper sites in Nevada and Utah. Values are mean \pm 1 SE. Days of available soil water were measured when soil matric potential was > -1.5 Mpa.

Table 1. Precipitation for 2002 and 2003. *Closest long-term weather stations

		Precipitation (mm)					
Site		1 Oct-30 June			Annual		
		2002	2003	30 Yr Avg	2002	2003	30 Yr Avg
Nevada	<i>A. cristatum</i>	111.9054	139.908		144.436	194.775	
	Low	162.964	203.974		157.583	272.403	
	Middle	209.0773	252.123		187.4963	301.871	
	Upper	241.748	257.961		251.243	350.311	
	Reese River*	129.54	168.148	156.1801	149.352	209.804	203.1624
	Smokey Valley*	72.898	84.582	133.5019	88.646	116.078	183.1157
Utah	<i>A. cristatum</i>	91.904	158.596		124.013	232.7681	
	Low	83.63	147.354		193.216	244.674	
	Middle	154.518	226.874		201.347	322.73	
	Upper	198.7606	321.416		267.9143	505.551	
	Fairfield*	206.248	179.07	250.4402	332.6515	172.212	332.6515
	Vernon*	212.852	129.54	206.0442	272.9393	154.686	272.9393

FIELD GERMINATION POTENTIAL OF CHEATGRASS (*BROMUS TECTORUM*) IN RELATION TO DISTURBANCE AND ELEVATION

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Study Description. Over the past several decades, cheatgrass (*Bromus tectorum*) has been continually expanding in the sagebrush steppe ecosystem. Research on why cheatgrass is able to invade these communities has been limited. This study looks at how season, site, elevation, and disturbance affect germination potential of cheatgrass, as a function of seedbed soil water availability and temperature.

Methods. Plots were established in the spring of 2001 on 9 sites in Nevada and Utah. Three of the plots in both Nevada and Utah were located in sagebrush stands along an elevational gradient. In Nevada, the elevations were 1960, 2190, and 2380 m and in Utah were 1710, 2085, and 2274 m. The fourth sites were located in stands of crested wheatgrass (*Agropyron cristatum*) having elevations of 2065 m in Nevada and 1628 m in Utah. The final site was located in Utah in a stand of squirreltail (*Elymus elymoides*) at an elevation of 1597 m. Vegetation removal and burn treatments were applied in a factorial design to examine the separate and combined effects of perennial herbaceous vegetation removal and fire on seedbed environmental conditions. Individual study plots (3 m diameter) were located within each of the shrub-dominated study sites around a focal sagebrush and were usually 2 or more meters apart. For the crested wheatgrass and squirreltail study sites, plots were located within areas of relatively uniform grass cover. The removal treatments included leaving the herbaceous understory intact or spraying with glyphosphate (Roundup®) in spring of 2001 for complete kill. The burn treatments applied in fall 2001 included leaving the plot unburned or burning within 3.4 m diameter burn barrels around each plot, adding 4.5 kg of straw for consistent fuel loading, and lighting the plots with drip torches (Korfmacher et al. 2003). Three replicates of each treatment combination were applied per site (9 sites x 2 removal treatments x 2 burn treatments x 3 replicates = 108 plots). Also in fall 2001, subplots 1.5 m x 0.75 m within each of the 4 removal-burn treatment plots (live-unburned, live-burned, dead-unburned, dead-burned) in each replication were broadcast seeded with filled seeds (600 seeds/m²) of cheatgrass.

In summer 2001, thermocouples and gypsum blocks (Delmhorst, Inc.) were buried in each of the 4 removal-burn treatment plots and within subplots left unseeded or seeded in fall to cheatgrass. Both sensor types were buried at depths of 1-3, 13-15, and 28-30 cm. Thermocouples to sense soil temperatures were installed in 2 replicates and gypsum blocks to sense water potential were installed in all 3 replicates of the treatments at each study site. Soil temperature and water potential sensors were read every minute and hourly averages recorded using Campbell Scientific, Inc. CR-10X microloggers. Hourly air temperatures from a thermister in a gill shield and total hourly precipitation from an electronic tipping bucket rain gage were also recorded at each study site. Germination potential variables were calculated for 4 fall seasons (September through November for 2001 through 2004) and for 4 spring seasons (March through June for 2002 through 2005). Gypsum blocks sense soil water potential to about -1.5 MPa (CSI

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1983). Soil temperatures and water potentials from the 1-3 cm depth were used to calculate a number of environmental variables considered relevant to cheatgrass germination. Given a base temperature for germination of 0 ° C, the thermal time required for 50% germination of a population (D50) of imbibed seeds of cheatgrass ranges from 12 to over 100 degree days, depending on the incubation temperature and seed population (Hardegree et al. 2003, Taylor et al. 2004, Bair et. al 2005). A reasonable, average value for D50 for non-dormant cheatgrass is around 1000 degree hours or about 41.7 degree days. Wet degree days (WDD) were calculated by adding all hourly temperatures above 0 ° C for the entire time that the soil was wet (< -1.5 MPa) for a season and dividing by 24 hours/day. The number of potential germination wet periods (PGW) was calculated for each season by counting the number of times that the soil was wet and accumulated at least 41.7 degree days (1000 degree hours) typically required to germinate 50% of cheatgrass seeds (ref). The degree days for the maximum time (MDD) the soil was wet for a season was also calculated. To determine potential germination in time, wet degree days per day were calculated for fall and spring periods by adding up hourly temperatures over 0 ° C for each day that surface soil water potential was greater than -1.5 MPa at 1800 hours and dividing by 24 hours per day.

Average hourly thermocouple temperatures from the 2 replicates were used to calculate germination potential variables for each of the 3 soil water potential plot replicates for each site. Averages of all replicates for each treatment combination for a site were then analyzed in a mixed model (SAS Proc Mixed, Littell et al. 1996), with site considered random and year, removal, burn, and seed treatments considered fixed factors. Best unbiased linear prediction equations (BLUPS) were used to estimate germination potential values for sites for each year (Littell et al. 1996).

Results and Conclusions. Water-year precipitation increased with elevation and was least in Nevada in 2003-2004, least in Utah in 2001-2002, and highest for all sites in 2004-2005 (Fig. 1). An exception to the pattern of increasing precipitation with elevation was some fall or spring storms that produced similar precipitation at all sites.

The number of potential germination wet periods (PGW) and wet degree days (WDD) varied significantly ($P < 0.0001$) by year for both fall and spring (Fig. 2). Potential for cheatgrass germination was greater in spring than in fall, but was sufficient for germination for all seasons during all years except for the fall of 2001 ($PGW < 1$). In general, lower elevation sites had greater cheatgrass germination potential than middle and upper sites, but all sites had sufficient PGW and WDD for cheatgrass germination for most seasons and years. Burning slightly increased the number of WDD, but unburned plots had sufficient PGW and WDD for cheatgrass germination (Fig. 3). The earliest date of potential cheatgrass germination varied greatly among years, and was later in the fall and spring at upper than lower elevation sites in Nevada (Fig. 4). Even though there was generally sufficient potential for cheatgrass to germinate on all sites, the upper sites had little actual cheatgrass establishment, as did the lower site in Utah. Seedling death associated with cold temperatures or drought may have limited establishment on these sites.

Management Implications. Soil temperatures and surface soil water availability were sufficient for potential cheatgrass germination in fall or spring on 9 sagebrush zone sites during 4 years of study. Because cheatgrass is well-adapted to germinate throughout the sagebrush zone, this zone must be managed to maintain competitive residual perennial herbaceous vegetation to reduce invasion. Although residual vegetation may not reduce potential germination, it decreases subsurface soil water availability (Whittaker et al. 2006) and nitrate (Blank et al.

2006), thereby limiting cheatgrass growth and seed production (Chambers et al. 2006). The end result is a reduction in cheatgrass invasion potential.

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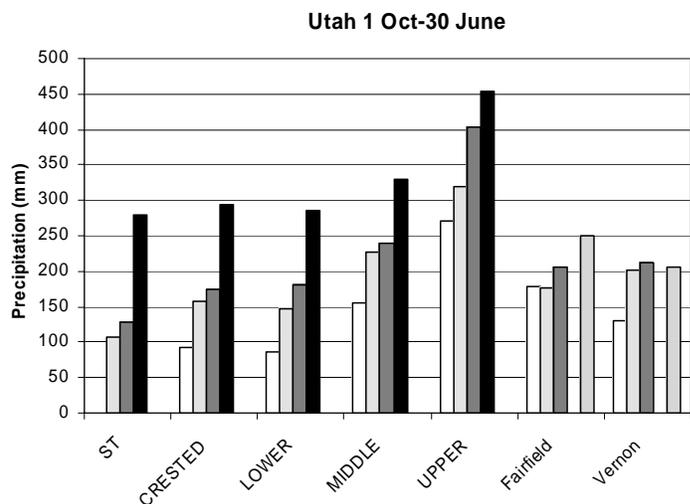
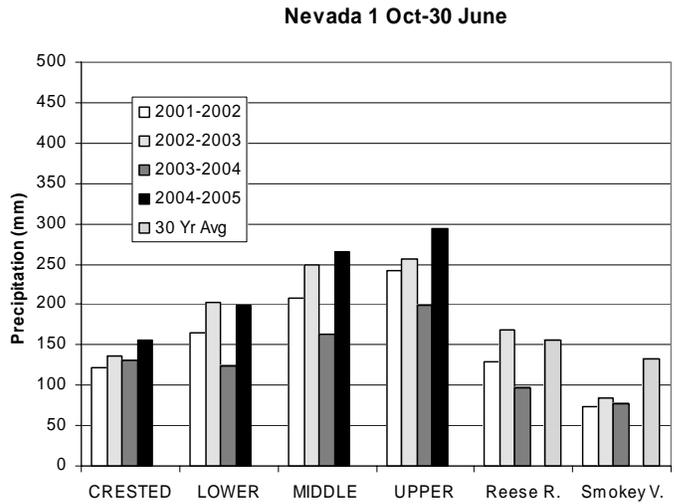


Figure 1. Water-year precipitation for squirreltail (ST), crested wheatgrass, and lower, middle, and upper elevation big sagebrush sites in Nevada and Utah.

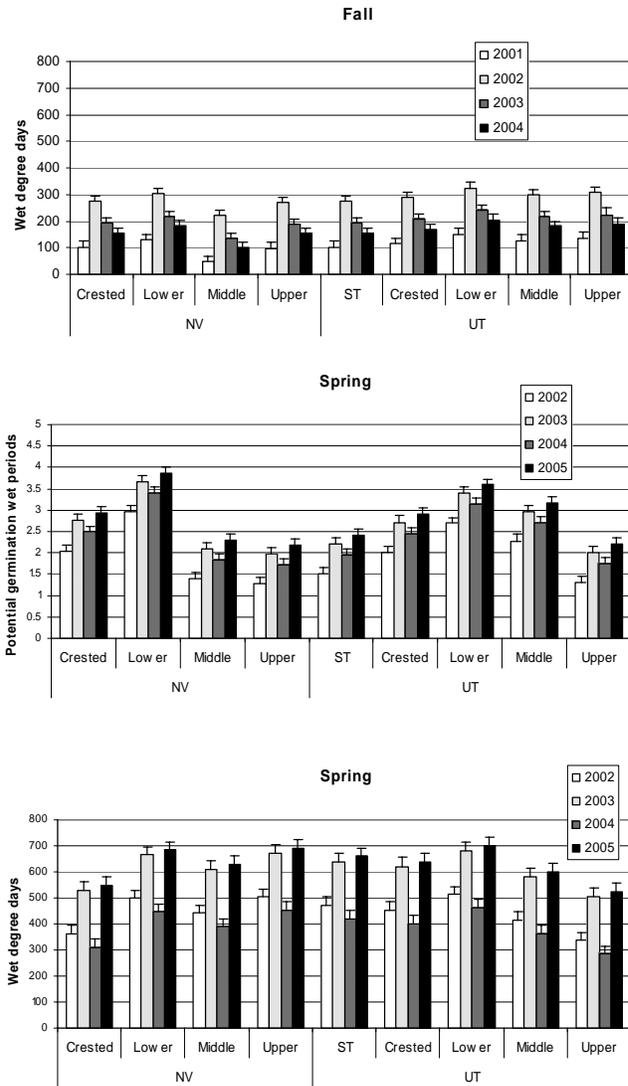


Figure 2. Potential germination wet periods and wet degree days for crested wheatgrass, squirreltail (ST), and lower, middle and upper elevation big sagebrush sites in Nevada and Utah.

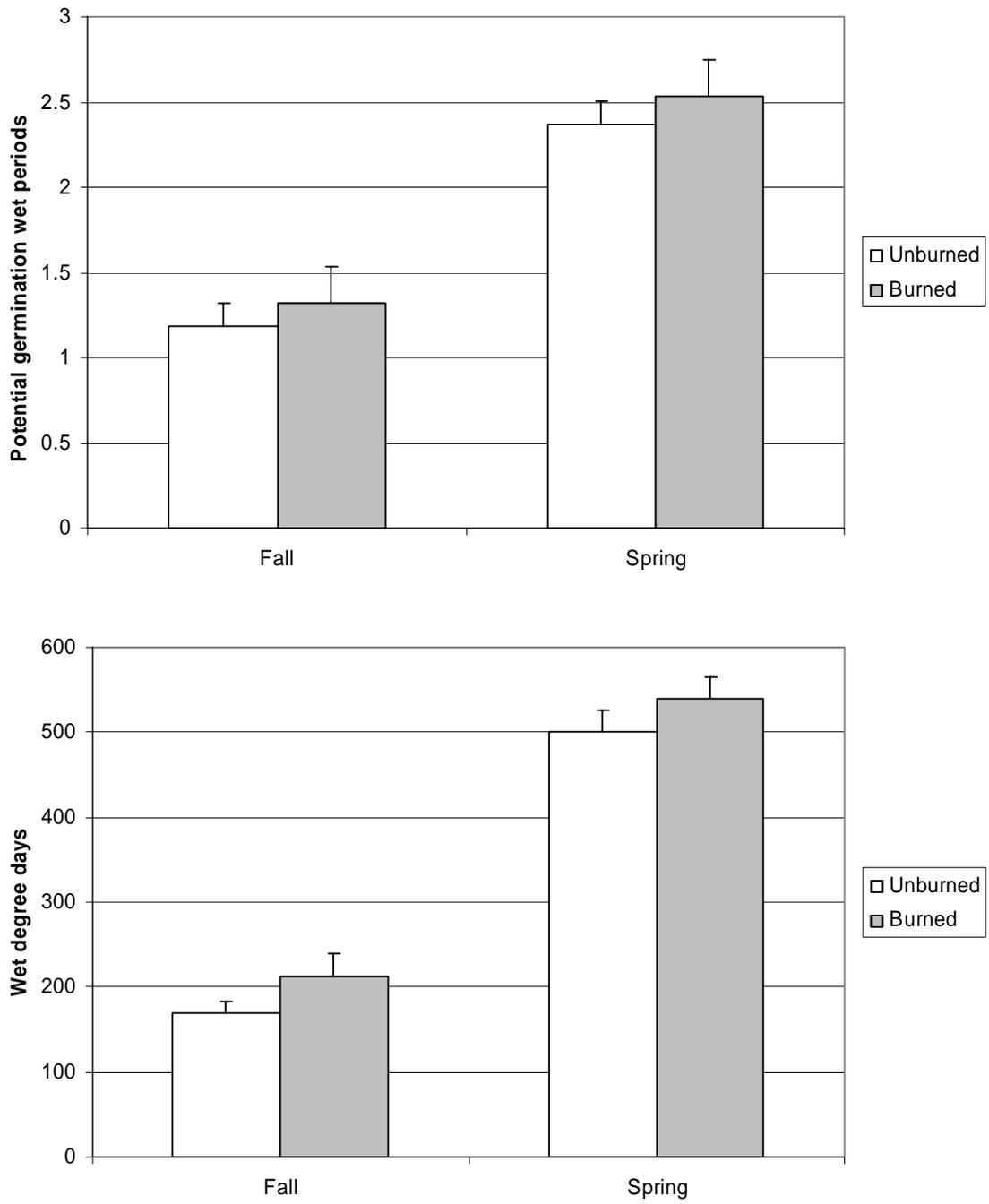


Figure 3. Potential germination wet periods and wet degree days for germination of cheatgrass in relation to fall burning on big sagebrush zone sites.

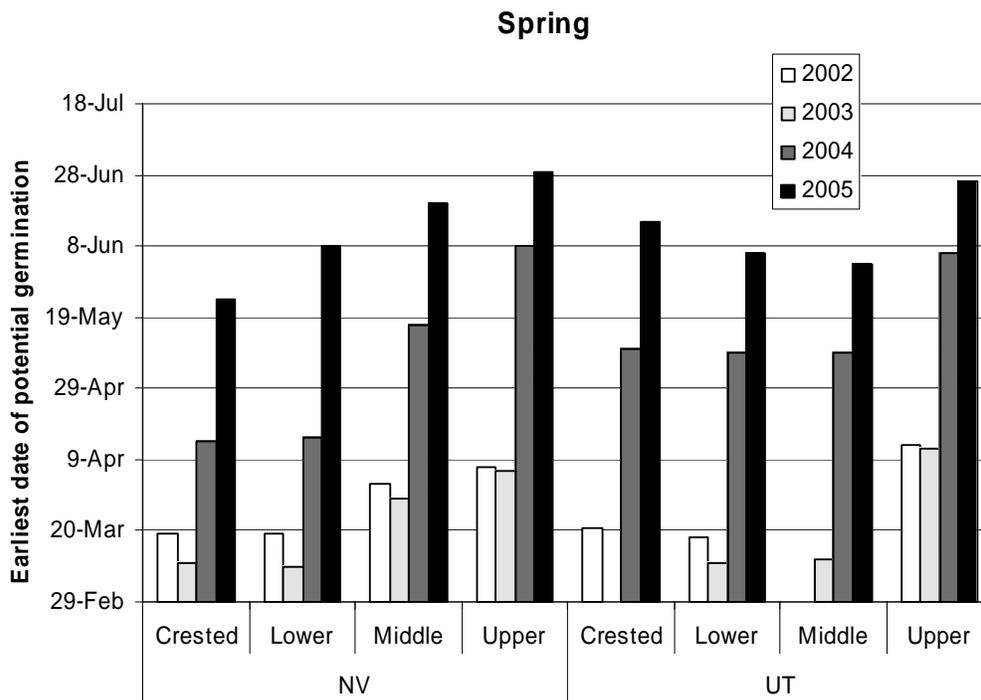
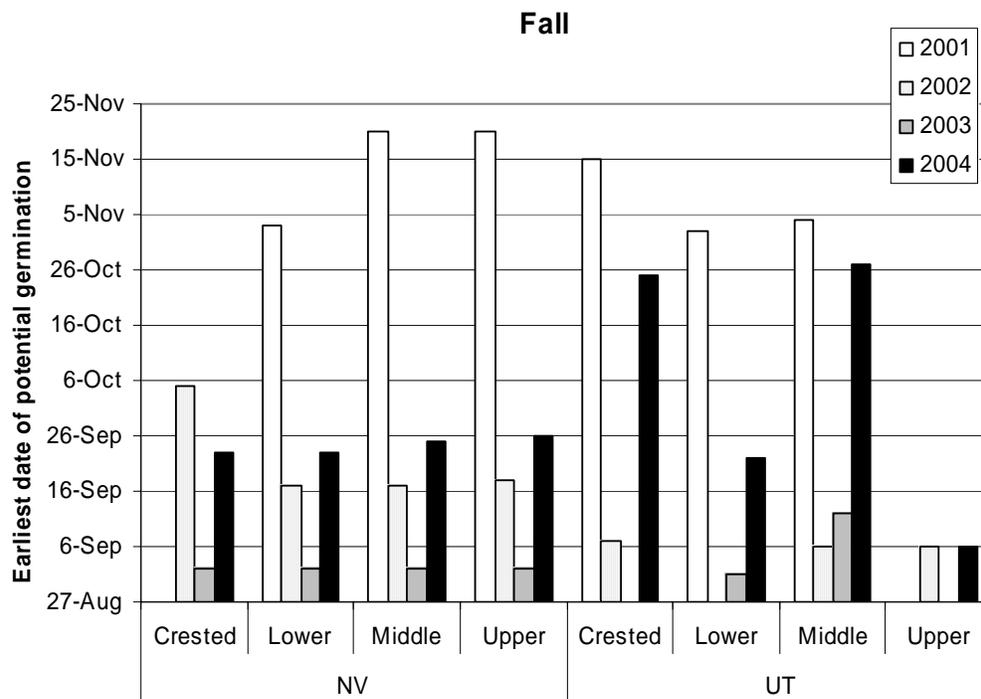


Figure 4. Earliest date of occurrence of sufficient wet degree days to germinate cheatgrass on crested wheatgrass and lower, middle, and upper elevation big sagebrush sites in Nevada and Utah.

SUSCEPTIBILITY OF SAGEBRUSH COMMUNITIES TO CHEATGRASS (*BROMUS TECTORUM*): EFFECTS OF NATIVE HERBACEOUS SPECIES REMOVAL AND FIRE

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Study Description. The exotic annual grass, cheatgrass (*Bromus tectorum*) is increasing in abundance throughout the sagebrush-dominated ecosystems of the United States, and is frequently assuming dominance following wildfire (D'Antonio and Vitousek 1992). Research and management have focused primarily on methods of controlling cheatgrass and restoring sustainable ecosystems following invasion. Consequently, little information exists on what makes sagebrush communities susceptible to invasion by cheatgrass. Increasing our understanding of the environmental and ecological factors that influence cheatgrass invasion and on invasion processes will allow us to develop management techniques aimed at preventing initial invasion or expansion. From a regional perspective, this approach may be as or more effective than management schemes designed to reduce or eliminate established populations of cheatgrass.

The invasion/expansion of cheatgrass is usually facilitated by an increase in resource availability. In general, cheatgrass invasion/expansion is occurring most rapidly at lower elevations within the Wyoming big sagebrush ecological type, in degraded rangelands with depleted understories, and in response to fire (Wisdom et al. 2003). This study examined the relative effects of elevation, understory species loss, and fire on cheatgrass establishment and reproduction. It is addressing the following questions: (1) What is the influence of elevation on resource availability and cheatgrass establishment and seed production within the sagebrush biome? (2) How does the relative abundance of native herbaceous species influence resource availability and cheatgrass establishment and reproduction? (3) How does fire influence resource availability and cheatgrass establishment and reproduction? (4) How do resource availability and cheatgrass invasibility differ between functional sagebrush communities and areas revegetated with crested wheatgrass? This aspect of the research focuses on cheatgrass establishment and reproduction. The effects of elevation, understory removal, and fire on soil water and nutrients are addressed in complimentary efforts (see syntheses by Blank et al. and Whittaker et al.).

Methods. To examine the effects of elevation on cheatgrass invasibility within the Region, study sites were located along an elevational gradient in the Shoshone Mountains of central Nevada and the Tintic Range of west-central Utah. The elevations of the study sites were 5600', 6840', 7460' in Utah and 6400', 7180', and 7800' in Nevada. In addition, crested wheatgrass study sites were located at elevations of 5340' in Utah and 6760' in Nevada. The lowest elevation sagebrush sites were characterized by *Artemisia tridentata wyomingensis*; the intermediate and upper elevation sites were dominated by *Artemisia tridentata vaseyana*. The understory vegetation was characterized by the perennial grasses *Pseudoroegneria spicata*, *Elymus elymoides*, and *Poa secunda* at low to intermediate elevations, and by *Festuca idahoensis*, and *Hesperochloa kingii* at upper elevations.

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Treatments included herbaceous species removal (0, 50, and 100% removed) and burning (burned and not burned). There were three replicates of each treatment combination. Circular treatment plots were 3.4 m in diameter and contained one sagebrush that was 1.0 to 1.5 m in diameter. The understory removal treatment consisted of spraying the vegetation with a mixture of Roundup and 2,4-D when the plants were actively growing at all sites (late May). The 50% removal treatment was obtained by spraying every other plant – one plant was protected with a small pot while the adjacent plant was sprayed. The 100% removal treatment was obtained by spraying all understory vegetation in the plot with a 2-gal. garden sprayer. The burning treatment was conducted in late October and consisted of placing a metal burn barrel around the plots, adding 4.5 kg of straw for a uniform burn, and igniting the contents of the barrel with a drip-torch (see Korfmacher et al. 2003). Treatments were applied in 2001 and 2002.

To evaluate cheatgrass establishment and reproduction within each treatment plot, 100 filled cheatgrass seeds were sown in 40 x 40 cm grids with 4 cm spacings to a depth of 0.5 to 1.0 cm in both interspace and undershrub microsites. Two grids (subsamples) were placed within each microsite and permanently marked. Cheatgrass seeds were from local collections, and had 98% or higher viability from standard tetrazolium tests. Seedling emergence and survival were monitored monthly from March through June. At seed ripe in late-June or early-July of 2002 and 2003, all plants within a plot were counted and harvested. The seeds were separated from the vegetative biomass, the number of filled seeds was counted, and the mass of both the seeds and vegetative portions of the plants were obtained.

Data were analyzed using mixed effects ANOVA models in the Mixed procedure of the SAS System for Windows.

Results and Discussion. Cheatgrass establishment (emergence and survival) varied among locations, elevations, and years (Fig. 1). The high elevation *A. tridentata vaseyana* sites, 7460' in Utah and 7800' in Nevada, generally had lower plant numbers than the low or intermediate elevation sites in both 2002 and 2003. Also, while most plants had emerged by the March census at low elevations, most emergence occurred between March and April at higher elevations (Fig. 2). Overall biomass and seed production of *Bromus tectorum* also was lowest on the high elevation sites in 2002 and 2003 (Figs. 3 and 4). The high elevation sites had the highest number of days of available soil water and among the highest levels of nitrate, but soil temperatures and number of degree days were considerably lower indicating colder and shorter growing seasons (Blank et al. *in process*, Chambers et al. *in process*, Whittaker et al. *in process*). Although *B. tectorum* exhibits relatively high germination at cold temperatures and has considerable ecotypic variation in optimal night/day germination temperatures (Meyer et al. 1997), ecophysiological limitations due to cold temperatures can restrict its growth and, consequently, reproduction at higher elevations and on cooler aspects. Site factors or precipitation and its effects on available soil water were the primary factors limiting cheatgrass success with decreases in elevation. In Nevada, the low elevation *A. tridentata wyomingensis* site exhibited relatively high establishment and similar biomass and seed production in 2002 and 2003 (Figs. 1 and 3). In Utah low establishment, biomass and seed production occurred on the low and mid elevation site in 2002 and the low elevation site exhibited almost no emergence in 2003 (Figs. 1 and 4). Lower establishment, growth and reproduction in 2002 can be explained by low precipitation and fewer days of soil water availability for the Utah than Nevada sites. However, in 2003 wet growing degree days, total days of available soil water, and soil water availability patterns in time were similar for the low and mid elevation sites in Nevada and Utah. The low elevation site in Utah had a limited amount of *B. tectorum* at the time of study initiation, and coarse sandy loam soils simply may not have been conducive to cheatgrass establishment.

The removal and fire treatments had relatively minor effects on emergence and survival and, consequently, almost no effects on number of plants at harvest (Fig. 1). The removal treatment resulted in plant mortality but because the dead plants were left in place, it had minimal effects on soil surface properties and, thus, the microenvironmental characteristics of the seed bed. In contrast to removal, burning resulted in higher emergence but lower survival of *B. tectorum* for certain sites and years. The largest effects on emergence and survival were due to microsite differences with interspaces having higher emergence and survival than undershrub locations (Fig. 2). Different mechanisms probably caused these microsite differences on burned vs. not burn plots. Burned undershrub microsites can provide a harsh environment for emergence and survival due to temperature extremes (Chambers and Linnerooth 2001). Undershrub microsites that are not burned are relatively cold, and can have litter layers with little available moisture for seed germination and seedling survival (Facelli and Pickett 1990). These results indicate that regardless of plant removal or fire, emergence is higher in interspaces than under shrubs and, as shown elsewhere, a high proportion of the seedlings that emerge survive to reproductive maturity (Pyke 1987).

In contrast to the results for seedling emergence and survival, both plant removal and fire had highly significant effects on biomass and seed production of *B. tectorum*. Biomass per plant and the number of seeds per plant exhibited responses similar to those for biomass and numbers of seeds per quadrat. Both biomass and seed numbers per plant were higher on burned than not burned plots, and on plots with 100% removal than 0% removal (Figs. 3 and 4). The effects of removal and burning on plant biomass and seed production were additive both for sites along the elevation gradient and for the *A. cristatum* sites. For sites with significant seedling emergence, biomass and seed numbers of *B. tectorum* typically increased 2 to 3 times after removal of all perennial herbaceous species, and 2 to 6 times after burning. Following both herbaceous species removal and burning, biomass and seed number increased from 10 to 30 times depending on site and year. These effects on biomass and seed production can be linked largely to increases in nitrate and especially soil water following plant removal and burning (Blank et al. *in process*; Whittaker et al. *in process*). *Bromus tectorum* has the capacity for high growth rates (Arredondo et al. 1998) and the ability to respond to increased nitrogen availability (Lowe et al. 2002) especially nitrate (Monaco et al. 2003) and soil water (Link et al. 1990, 1995).

Conclusions. The invasibility of Great Basin *Artemisia* ecosystems is dependent on environmental characteristics, and is the result of several interacting factors, including precipitation and temperature regimes, site conditions, past and present disturbance, and the competitive abilities of resident species. Underlying ecosystem properties, precipitation and temperature, had the greatest overall effects on the invasibility of *B. tectorum* during our two year study. *Bromus tectorum* was clearly limited by temperature at upper elevations. Precipitation, and its effects on available soil water, appear to be the primary controls on *B. tectorum* invasibility when temperature is not a factor. The effects of fire and species removal were consistent across the elevation gradient typical of *Artemisia* ecosystems in the Great Basin for *B. tectorum* establishment, growth and reproduction. Further, the effects of fire and species removal were additive for the growth and reproduction of *B. tectorum*. Our results clearly show that the effects of fire on *B. tectorum* invasion are much greater following the removal of the perennial herbaceous vegetation, and that the sustainability of these systems will depend on maintaining or restoring the perennial herbaceous species. Our species removals may not mimic the effects of overgrazing, especially in terms of soil nutrient dynamics, and additional research is needed to examine the effects of a one-time vs. chronic disturbance.

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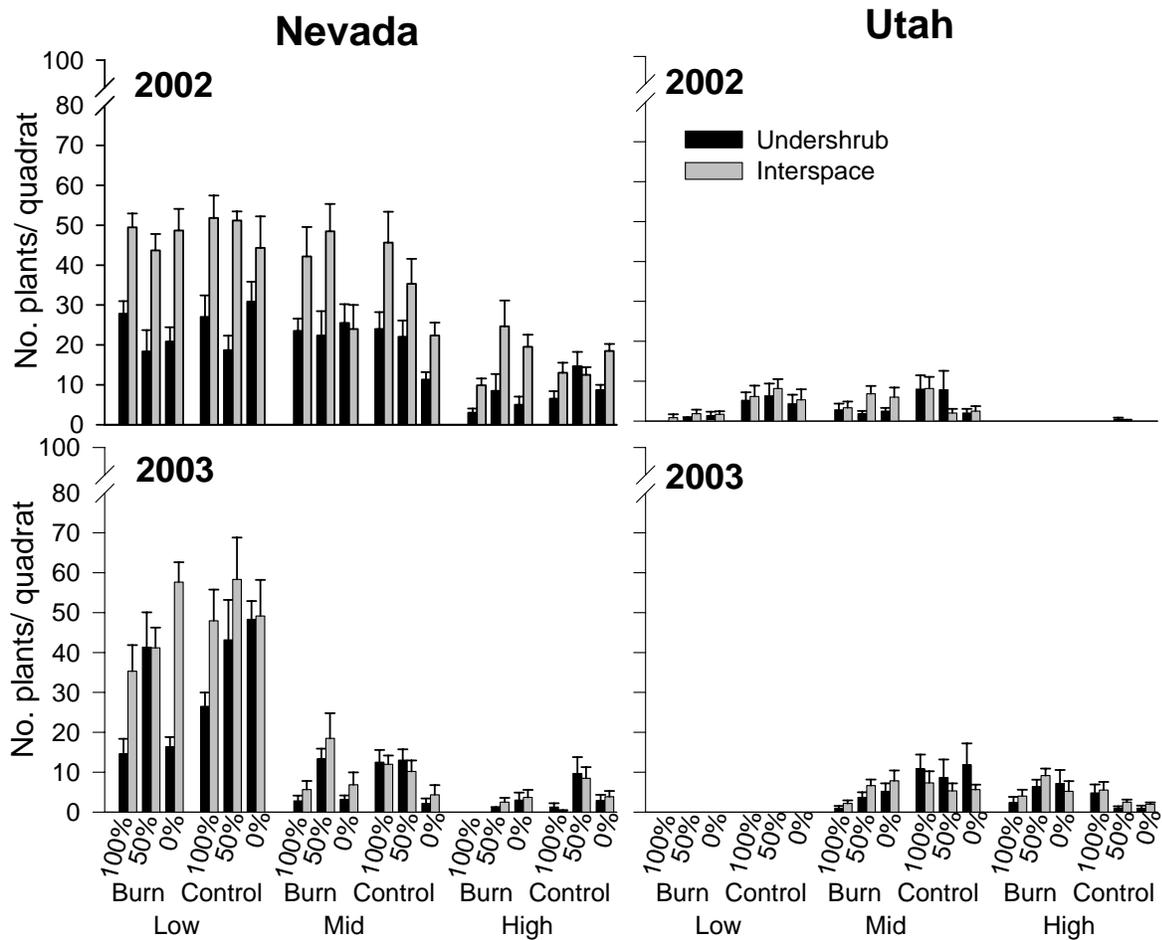


Figure 1. The number of plants at harvest (mean \pm SE) on the low, mid and high elevation sites in Nevada and Utah for 2002 and 2003.

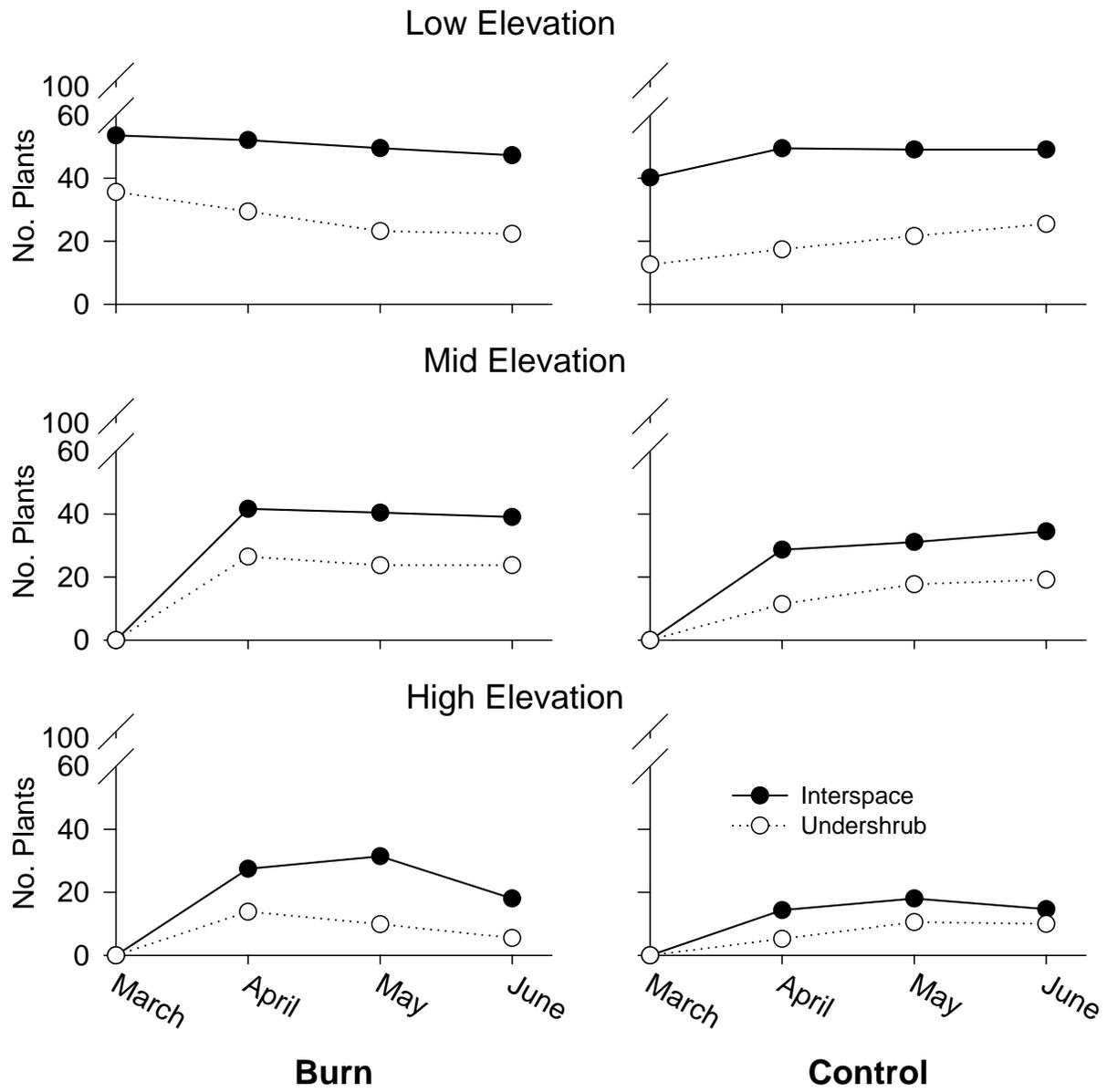


Figure 2. The number of plants alive for the March, April, May and June census dates on the low, mid and high elevation sites in 2002 in Nevada.

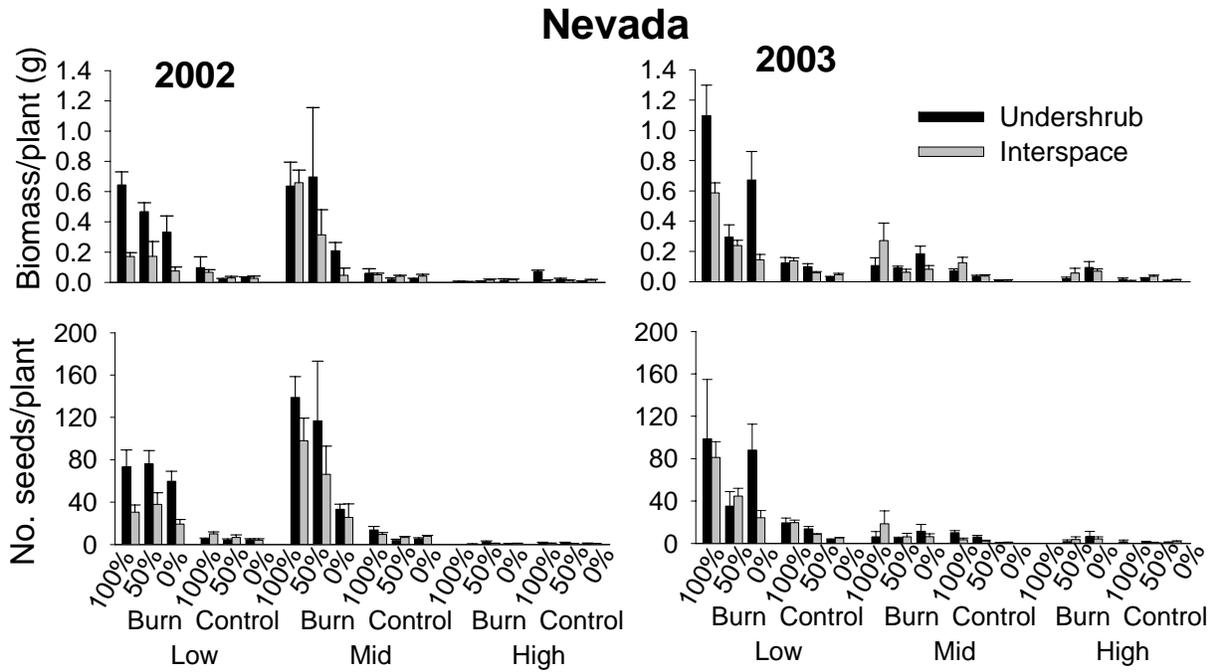


Figure 3. Biomass/plant and seeds/pant (mean \pm SE) on the low, mid and high elevation sites in Nevada in 2002 and 2003.

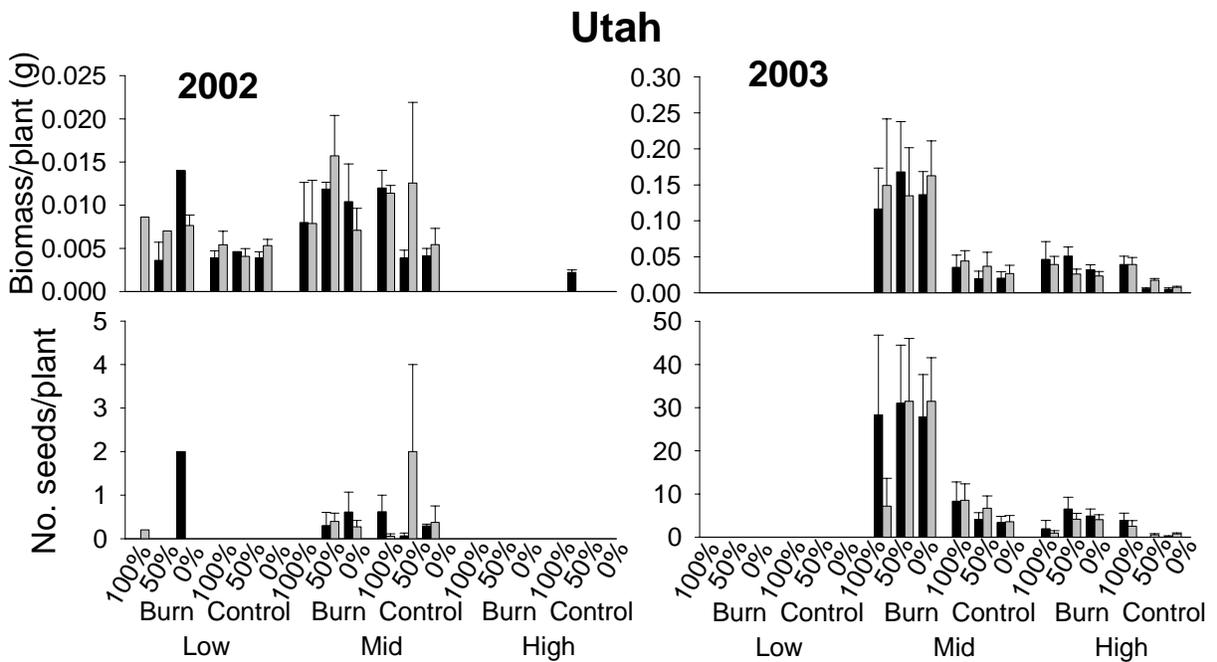


Figure 4. Biomass/plant and seeds/pant (mean \pm SE) on the low, mid and high elevation sites in Nevada in 2002 and 2003.

INVASION OF RUSH SKELETONWEED (*CHONDRILLA JUNCEA* L.) INTO SAGEBRUSH COMMUNITIES

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Characteristics of communities susceptible to secondary weed invasion. Rush skeletonweed (*Chondrilla juncea* L.), an invasive Eurasian apomictic perennial weed, has spread southward through Idaho and recently invaded sagebrush steppe communities of the Snake River Plain. Little is known of its fire ecology or its ability to invade sagebrush-steppe communities. A survey of the Snake River Plain conducted during the summers of 2004 and 2005 indicated that infestations occur in abandoned dryland cropping areas, road right-of-ways, and degraded crested wheatgrass seedings. Other occurrences noted in our field survey suggest that although rush skeletonweed may have gained an initial foothold on the Snake River Plain in agronomic settings, native rangeland is also invaded today. Rush skeletonweed is present in heavily grazed basin and Wyoming big sagebrush (*Artemisia tridentata* ssp. *tridentata* and *A. t.* ssp. *wyomingensis*) communities in a variety of vegetation states and along seasonally dry drainage ways extending out from the foothills to the north. Small, isolated populations have been found at considerable distances south and east of the expansion front, possibly established from seed carried by wind, vehicles, or animals. Although rush skeletonweed is generally considered most adapted to sandy or coarse soils, infestations are not uncommon on finer soils. Populations were found on slopes ranging from 0 to 70 percent, and although most common on upland sites and on west, south, or east facing slopes, populations were also found on disturbed north-facing slopes and on soil disturbances beneath cottonwoods (primarily *Populus trichocarpa*) within the Boise River floodplain. The densest and most extensive infestations occur near Emmett and Boise, Idaho.

Invasions are most common in areas where the soil has been previously disturbed and where vegetation is in early successional stages or replaced by cheatgrass (*Bromus tectorum*) or other invasive annuals. Populations are often prominent on recently burned annual weed sites. Invasion, however, is not limited to particular vegetative assemblages or disturbance histories. Rush skeletonweed has successfully entered many relatively undisturbed native shrubland communities, including big sagebrush (*Artemisia tridentata*) and antelope bitterbrush (*Purshia tridentata*) communities on the Snake River Plain. It was found in sagebrush communities with understories of perennial grasses including Sandberg bluegrass (*Poa secunda*), needle-and-thread (*Hesperostipa comata*), and Thurber needlegrass (*Achnatherum thurberianum*). In these communities, plants often establish on micro-disturbances such as gopher mounds or around badger holes. Spread in on such sites appears to occur more slowly than in surrounding weedy areas.

Rush skeletonweed seed ecology, population dynamics and response to fire. Rush skeletonweed flowering is indeterminate with flowers and seeds produced from early summer until first frosts. In a single season, mature plants produce up to 15,000 tiny seeds that are readily and widely dispersed by wind and other agents (Parsons 1973). Laboratory and field

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studies indicated that seeds lacked primary dormancy and did not acquire secondary dormancy. Seed germinated readily with total germination similar over a wide temperature range of 6 to 30 °C and the germination rate greatest at about 21 °C. Viability of seeds produced early in the season (mid-July) was low and increased through mid-September 2003 before declining until seed production ceased in October 2003. In Australia, McVean (1966) suggested that consistently hot summer weather (daily maxima exceeding 33 °C) with low relative humidity and low soil moisture reduces rush skeletonweed seed production and viability, and so is unfavorable for reproduction.

The seeds of rush skeletonweed are often targeted for control, but the actual recruitment role of seeds compared to root sprouts has not been quantified in the region, particularly following wildfire. We investigated recruitment processes in rush skeletonweed on 11 burned and unburned plot pairs located in rush skeletonweed stands straddling the boundaries of four 2003 wildfires on the Snake River Plain. Fall 2003 and 2004 soil seed banks averaged 93 to 164 emergents/m², considerably lower than reported for other invasive exotic Asteraceae in our area (McEvoy et al. 1991, Sheley and Larson 1994, Stringer 2003). Seeds sown in native soils and incubated in a growth chamber had markedly higher emergence when soils were sterilized (25 %) compared to unsterilized soils (9 %) indicating soil pathogens may have been responsible for some seed mortality. Seeds buried in bags in the field in late October 2003 displayed no secondary dormancy, reached peak germination of about 31 percent by mid-January 2004, and remained only 1 percent germinable by August 2004 (Figure 1). Germination was about 2 weeks earlier on unburned than on burned sites, but total germination did not differ between the two. Of 2,800 seeds sown in late October 2003, only four seedlings emerged by August 2004. Of 1,627 rosettes excavated in the field from fall 2003-spring 2005, only 112 (7 %) originated from seeds, and 108 of the 112 were found in spring 2005. This may have resulted from an unusually dry winter in which a large portion of the seeds did not imbibe, and were then able to germinate and emerge with spring rains. A higher proportion of rosettes from seeds were found on burned plots (32 %) than unburned plots (20 %) (Table 1). Other burn versus unburn comparisons did not differ.

Root sprouts, rather than seeds, were the main source of new plants from October 2003 to May 2005. Although initial density counts of adult rush skeletonweed plants showed no difference between burned and adjacent unburned stands, burned stands showed greater numbers of new rosettes and resprouted rosettes (1.9/m² vs. 0.2/m²) in fall 2003. Over 35 percent of tagged new rosettes died by mid-December 2003. Rosette emergence resumed by late March, peaking in late April before the annual die-off, again with greater densities in burned than in unburned sites (Figure 2). The increased density of plants following wildfire appears less pronounced in the second year following the fire (data yet to be analyzed). Thus it appears that once present on a site, density increases primarily through vegetative spread, a process favored by wildfires.

Management activities and secondary weed invasions. On the Snake River Plain, spread of rush skeletonweed is opportunistic, possibly limited by the availability of moisture and microsites suitable for seedling establishment. We have observed higher rates of invasion on formerly plowed sites than on adjacent unplowed sites, even though plowing ceased before rush skeletonweed arrived and so could not be implicated in increasing plant densities through root fragmentation. On study sites, we found many seedlings in mounds of soil that had been churned up by badgers, ground squirrels, or other fossorial mammals. It is plausible that these species are aiding seedling establishment of rush skeletonweed on wildland sites of the Snake River Plain. In Australia where rush skeletonweed forms dense stands in agricultural fields, but rarely spreads into uncultivated grasslands and shrublands, cultivation appears to provide a

more favorable seedbed for germination and seedling establishment (Cuthbertson 1967, 1970; Wells 1971). It is plausible that fossorial mammals are aiding seedling establishment in native communities on the Snake River Plain, while in Australia, absence of fossorial mammals may account, at least in part, for the lack of spread of rush skeletonweed on uncultivated lands.

In Australia, surface sown seeds were shown to have reduced germination and establishment in comparison to those buried at depths of 1 cm or less, and losses were attributed to desiccation or sealing of the soil surface as may occur following fires. Seeds in our region that lie on the soil surface likely suffer high mortality. Desiccation after germination but before emergence likely causes a substantial amount of mortality on some sites and in some years. In Australia, 3 to 6 weeks of continuous moisture availability was required to avoid seedling death by desiccation (Cullen and Groves 1977). In our region low seasonal precipitation may preclude germination in some years, while sporadic early fall precipitation may lead to precocious germination and loss of seedlings as a result of subsequent desiccation.

Control efforts aimed at reducing seed production of adult plants appear to be redundant with seed destruction currently occurring from biotic and biotic factors on the Snake River Plain. Furthermore, efforts to remove flowering plants can cause rush skeletonweed to increase. Mowing, burning, and other disturbance of aboveground portions of adult plants remove seeds and photosynthetic material, but have been shown to increase production of root buds and lead to particularly dense groups of plants in later years (McVean 1966; Cuthbertson 1967, 1972). Unless carried out at least several times annually for several years, cultivation also increases rush skeletonweed density by fragmenting the root system and causing new plants to develop from the fragments (Wapshere 1971). Herbicides such as picloram, clopyralid, and clopyralid/dicamba combinations are effective only when applied while rosettes are present in spring as the coarse stems take up little herbicide. Repeat treatments are often required in subsequent years in order to provide adequate control.

Fire may open sites to invasion by any exotic but does not directly favor the encroachment of rush skeletonweed by enhancing seedling recruitment unless favorable microsites and moisture conditions are available. Once on a site, populations of rush skeletonweed respond to wildfires by resprouting from root buds with the result that rosette and stem density are increased compared to nonburned areas. In general, fire history, (burns 1-2 years old) did not influence the germinable seed found in the seedbank on burned sites relative to unburned soils. However, in years when seedlings are recruited, spring 2005 in our study, burned sites did contain more recruitment from seed than unburned sites.

Occasional *C. juncea* recruitment from seed on the Snake River Plain is most likely the source of new infestations and appears to be possible in many vegetative communities, with and without major disturbance. Control efforts are best focused on mature plants where the species has already invaded. Additionally, efforts to protect intact sagebrush communities from the transport of seed to new locales should probably focus on biocontrol efforts. Burning and mowing are not suggested as management tools once rush skeletonweed has entered a site.

In the search for effective biocontrol agents, three agents released in Idaho in the 1970s are now widespread in our region: *Puccinia chondrillina*—a rust fungus of rosette leaves and seedlings, *Cystiphora schmidti*—a gall midge of leaves and stems, and *Eriophyes chondrillae*—a gall mite of terminal and axillary buds (Piper 1990 and refs therein). The rust fungus is virulent against the “Banks” genotype in our region, reduces seed viability and seedling survival (Cheney et al. 1981), and may have impacted some seed produced in populations included in our studies. The other two organisms reduce adult plant biomass and flower production

(Mendes 1982; Prather 1993), yet *C. juncea* continues to spread on the Snake River Plain. Root-boring larvae of the moth *Bradyrrhoa gilveolella* were introduced recently near Banks ID (Markin et al. 2003), and may hold promise if they can establish on the Snake River Plain. At any rate, there will likely always be some seed present where the exotic occurs and consequently a combination of biocontrol to target seed, coupled with root boring larvae to target vegetative spread may prove optimal. Biocontrol combined with herbicide treatments may be most effective where intensive treatment is feasible.

Limiting seed transport vectors would be an important proactive step to consider in order to protect native sagebrush stands on the Snake River Plain. Our study documents the great amount of variability in seedling recruitment from year to year. Reducing seed dispersal via mechanical and anthropogenic disturbances (quarantining livestock prior to movements to new areas, cleaning vehicle undercarriages, and similar precautions) could help to reduce the rate of spread in some areas. However, new infestations can be expected due to the high level of human activity near cities, the extent of livestock grazing on the Plain, and wind dispersal of the seed. Destroying small satellite infestations at and beyond the expansion front is essential and a feasible approach for reducing the rate of spread.

Our findings indicate that on the Snake River Plain, recruitment from seed is probably a rare event occurring in periods of opportune moisture and temperature. Spread may therefore be slower than in more mesic areas to the north or in Australia where both seedlings and root propagules are reportedly common means of establishment (Cuthbertson 1972, Cullen and Groves 1977), but the species has continued to expand its range into drier sagebrush communities typical of the northern Great Basin. Once plants are established, dense stands can develop over time through vegetative spread. Our findings are of interest because prolific seed production and establishment from seed has long been viewed as a character central to the ability of an introduced species to become invasive (Baker 1965, 1974). Seeds have undoubtedly contributed to the long-distance dispersal of rush skeletonweed in our region as shown by its rapid spread from the first known introduction site at Banks, Idaho, in the 1960s to sites that lie 50-80 km away, and the unlikely possibility that root fragments were the initial propagules on most of these sites. Clearly, Baker's (1965, 1974) additional trait of vegetation regeneration from root fragments is essential for dominance by rush skeletonweed in this area. This pattern has been noted, though perhaps in less obvious terms, for other invasive perennial composites of our region (Watson 1980, Moore 1975, Thomas and Dale 1974). Thus the potential for further expansion on the Snake River Plain exists so long as environmental variables do not limit both seed and vegetative spread to the extent that new populations cannot establish or maintain themselves.

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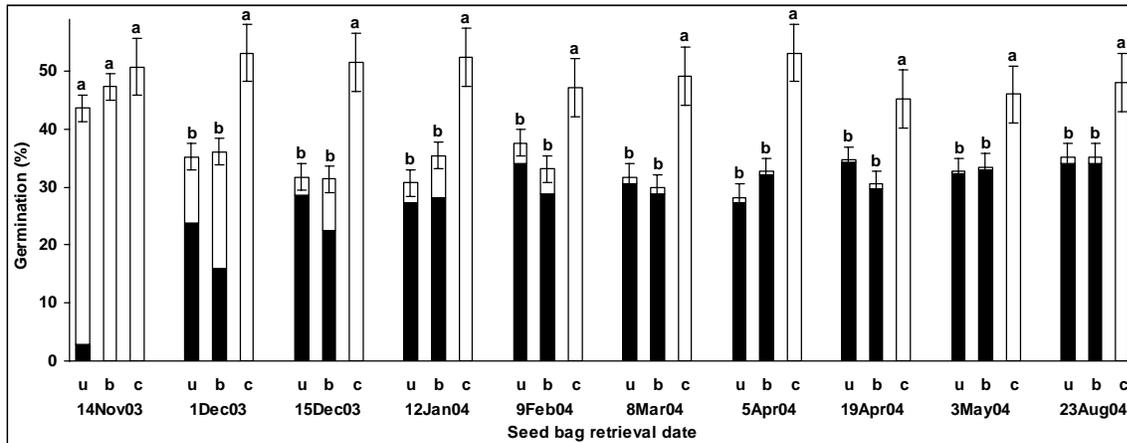


Figure 1. Germination of *C. juncea* seeds in bags buried 30-31 October 2004 on unburned (u) and burned (b) plots, and lab-stored controls (c). On nine dates, bags were retrieved and analyzed for field germination (filled bars), and additional germination in laboratory following retrieval (open bars). Error bars indicates standard error for combined field and laboratory germination.

Table 1. Source of 1,509 excavated *C. juncea* rosettes from root bud or seed, on 11 unburned and burned plot pairs on the Snake River Plain, ID.

Season	Unburn		Burn		Total (Unburn + Burn)	
	Root	Seed	Root	Seed	Root	Seed
Fall 03	78 (100.0 %)	0 (0.0 %)	463 (99.6 %)	2 (0.4 %)	541 (99.6 %)	2 (0.4 %)
Spring 04	209 (100.0 %)	0 (0.0 %)	364 (99.5 %)	2 (0.5 %)	573 (99.7 %)	2 (0.3 %)
Fall 04	45 (100.0 %)	0 (0.0 %)	64 (100.0 %)	0 (0.0 %)	109 (100.0 %)	0 (0.0 %)
Spring 05	137 (79.7 %)	35 (20.3 %)	155 (68.0 %)	73 (32.0 %)	292 (73.0 %)	108 (27.0 %)
Total (03-05)	469 (93.1 %)	35 (6.9 %)	1046 (93.1 %)	77 (6.9 %)	1515 (93.1 %)	112 (6.9 %)

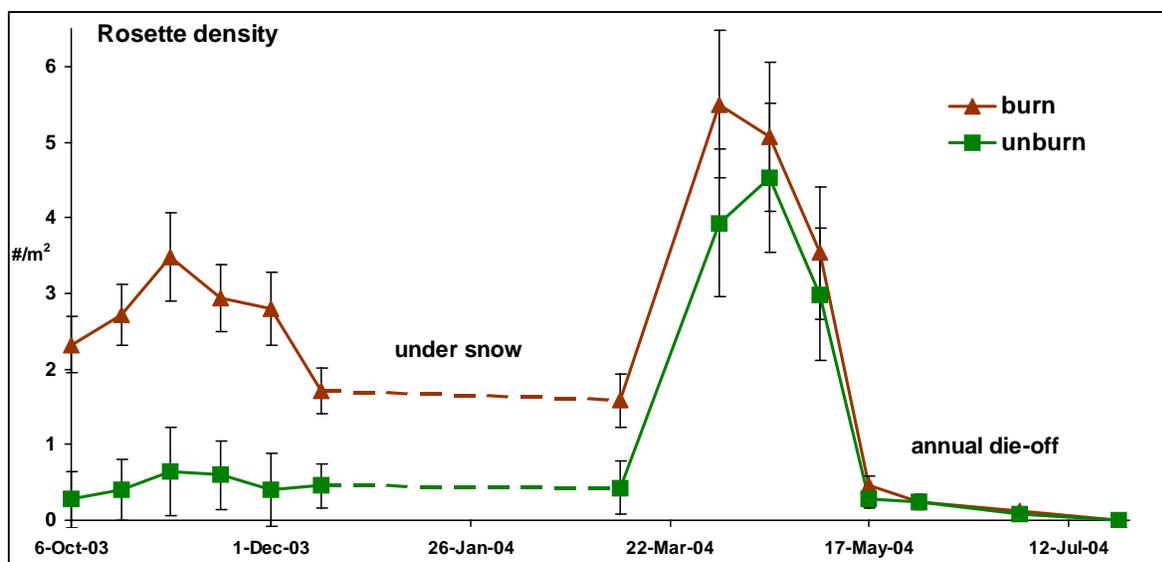


Figure 2. Density of rosettes (no./m²) from October 2003 to July 2004 on 11 burned and unburned plot pairs on the Snake River Plain following 2003 wildfires.

EFFECTS OF FIRE AND RESTORATION SEEDING ON ESTABLISHMENT OF SQUARROSE KNAWWEED (*CENTAUREA VIRGATA* VAR. *SQUARROSA*)

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Study Description. In the late 1920's squarrose knapweed (*Centaurea virgata* var. *squarrosa*) was introduced into Utah. It was first seen in Tintic Valley, Utah near a grain silo where it was just a single plant. It now covers thousands of acres in at least six counties and is listed on the Utah noxious weed list. It has invaded abandoned farmland, crested wheatgrass seedings, and degraded sagebrush communities. It is also invading intact sagebrush communities and communities that have been converted to cheatgrass. The purpose of this study was to examine the effect that fire has on squarrose knapweed and its invasion into different communities and also the effect that squarrose knapweed has on fuel loading. The three components of this study are: 1) Determine the effect of squarrose knapweed on fuel loading in cheatgrass communities, crested wheatgrass communities, degraded sagebrush stands and intact sagebrush stands; 2) Determine the effect of fire on the susceptibility to knapweed invasion in these same communities; and 3) Determine the effect that the season of burning has on cheatgrass-knapweed dominated communities.

Methods. The study sites are within several different community types located in Tintic Valley, Utah. The first portion of the study looks at the changes in fuel loading caused by squarrose knapweed in four different communities. These communities are cheatgrass, crested wheatgrass, degraded sagebrush, and intact sagebrush. Four sites of each community type were selected along with four sites dominated by knapweed, and combinations of cheatgrass and knapweed, crested wheatgrass and knapweed, and sagebrush and knapweed, for a total of 32 sites. During the summers of 2001 and 2002, a 100m transect was placed on each site and vegetation cover was estimated using a 1m² 20-point quadrat that was placed along the transect every 5 meters. Biomass was also collected from these quadrats. Fuel moisture information was also collected from these sites by collecting vegetation samples every two weeks from the first of June to the end of August and then recording the green and dry weights.

The second portion of the study looks at these same four community types to determine their susceptibility to invasion by squarrose knapweed and how fire plays a role in that. This part of the study had two treatments, a burn treatment (burned and not burned) and then within the burn treatment a seeding treatment (seeded with knapweed and not seeded). Three sites for each community type were located in the fall of 2000. Each site has four plots that were 400 m². Two of these plots, either burned or not burned, were treated in 2002 and the other two plots were treated in 2003. Forty randomly selected 1m² subplots were placed within the plots. Twenty of these subplots were designated to be seeded with squarrose knapweed and the other twenty were designated as unseeded. Vegetation cover on the subplots was estimated before treatment. The plots were then burned in June and then the subplots that were designated as seeded subplots were seeded the following fall.

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The objective of the third portion of the study was to determine the effect that the season of burning had on areas dominated by cheatgrass and squarrose knapweed. The treatment consisted of three different times of burning and a control. The first burn time was at the first of June just before cheatgrass seed heads shattered. The second burn time was in the middle of July after the cheatgrass heads had shattered but before the knapweed seed heads shattered. The final burn time was in October after the knapweed seed heads shattered. Four sites that were dominated by cheatgrass and knapweed were located in fall of 2000. On each site eight 900m² plots were randomly located. Four plots on each site were pre-inventoried in 2001 by estimating cover from twenty randomly located permanent quadrats. Biomass was also measured by clipping a ¼ m² quadrat adjacent to the quadrats used to measure cover. Due to fire restrictions in 2001 the plots were not treated. In 2002, these four plots were pre-inventoried again and then treated. However, due to fire restrictions during the summer, the mid-season burns were not completed. In 2003 the other four plots were pre-inventoried and treated.

Results and Conclusions. There were a few significant differences in fuel moisture between cheatgrass sites and cheatgrass/knapweed sites and between crested wheatgrass sites and crested wheatgrass/knapweed sites. Fuel moisture content during the first two weeks of the summer was significantly lower in cheatgrass and knapweed when they were growing together than in a monoculture (Fig. 1). Fuel moisture content in knapweed was also lower during the first two weeks when it was growing with crested wheatgrass than when it was growing in a monoculture (Fig. 2). There were no significant differences in fuel moisture between the degraded sagebrush sites, the intact sagebrush sites, and the sagebrush/knapweed sites.

Results from the second portion of the study were not significant. 2002 and 2003 were very dry and so seed germination and establishment was greatly reduced. In 2003, squarrose knapweed establishment in other knapweed research in Tintic valley was 1/3 of the amount that established in 2000 and 2001 (unpublished data). This low germination really affected the results. When a community is disturbed and loses vegetation the resources that the vegetation was using should become available. Many weedy species will take advantage of the open resources and establish (Burke and Grime 1996; Foster and Dickson 2004). In 2002 and 2003 there was very little resources available and so the effect of the fire was not as significant as we expected.

Data from the third portion of the study also did not show any significant differences. Again this can be mainly explained by the drought conditions that occurred in 2002 and 2003. In previous years these sites had high densities of cheatgrass and knapweed but due to the drought cheatgrass and knapweed establishment was low on these sites.

Acknowledgements. Funding was provided by Joint Fire Sciences Program and Pitman-Robbinson Federal Aide Project W-82-R. Assistance was provided by the UDWR Great Basin Research Center.

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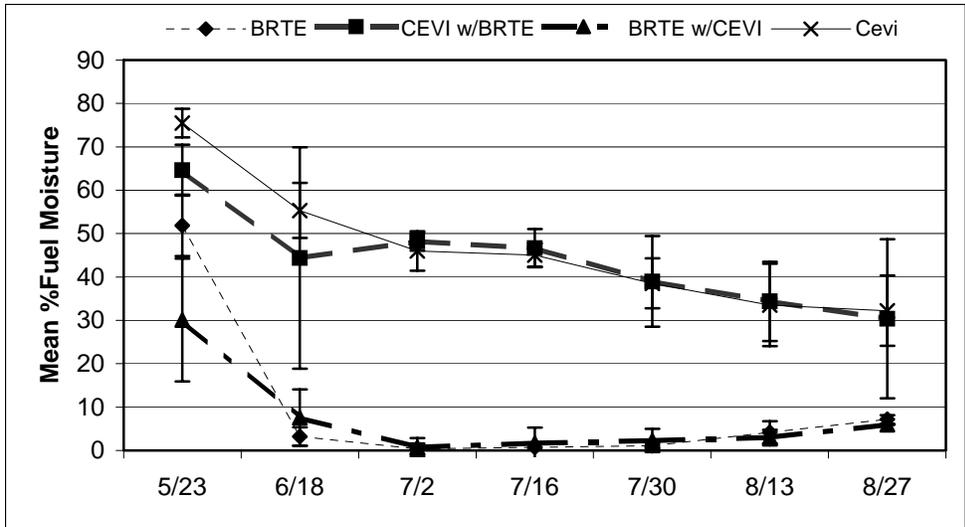


Figure 1. Mean %fuel moisture of cheatgrass and squarrose knapweed in monocultures and mixed stands. Samples were taken every two weeks from May 23 to August 27 2001.

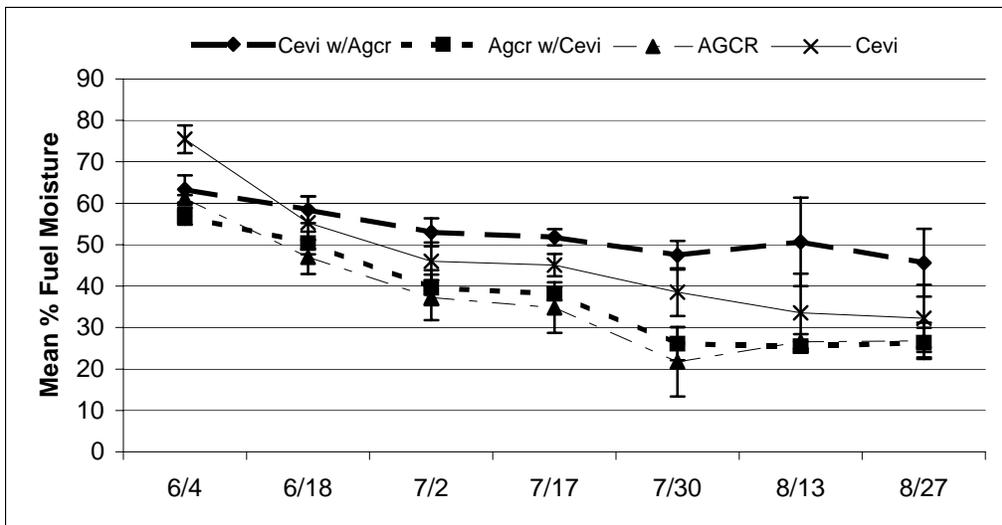


Figure 2. Mean %fuel moisture of crested wheatgrass and squarrose knapweed in monocultures and mixed stands. Samples were taken every two weeks from May 23 to August 27 2001.

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Publications:

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Rush Skeletonweed (*Chondrilla juncea* L.)

Publications:

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