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Soil Physical, Chemical and Understory Plant Nutritional Response to Pinyon-juniper Encroachment and Prescribed Fire in a Central Nevada Woodland.

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Abstract

Pinyon-juniper encroachment into sagebrush-grassland communities has produced a need for viable restoration treatments of sagebrush-grasslands. In the following manuscripts we document the effects woodland expansion and spring prescribed fire used as a restoration treatment on soil physical properties particularly infiltration and water repellency, soil chemical properties, and understory aboveground biomass and tissue nutrient concentrations. We observed that water infiltration and the development of water repellency was effected by cover type, soil coarse mineral particles, and burning. Soil chemistry was affected by cover type, time, and burning. It was also determined that plant aboveground biomass and nutrient concentrations are species dependent, and that these factors are effected by time and burning. Each species had a unique response to burning, although they may follow similar trends in temporal variation. There were direct links between burning, soil chemistry, and plant aboveground biomass and tissue nutrient concentrations.

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TABLE OF CONTENTS

Page

Abstract…….....i

Acknowledgements……..ii

Introduction…….....1

Hydrologic response of a central Nevada pinyon-juniper……...6
(*Pinus monophylla-Juniperus osteosperma*) woodland to prescribed fire

Pinyon-juniper expansion and prescribed fire effects on……...38
soil in Great Basin sagebrush ecosystems

Effects of pinyon (*Pinus monophylla*) and juniper (*Juniperus osteosperma*)……...84
expansion and prescribed fire on understory biomass and nutrient concentration in
Great Basin sagebrush ecosystems

Conclusions……..…135

Introduction

Sagebrush-grasslands are home to abundant and diverse native plant and animal species, some of which are completely or partially dependent on sagebrush for their existence (Connelly et al. 1988, Fischer et al. 1996, Brussard and Funari 2001).

Pinyon-juniper woodlands occupy greater than 30 million ha in the American west with about 3.6 million ha in the state of Nevada (Miller and Tausch 2001). Woodlands in Nevada and the Great Basin are expanding their range into sagebrush-bunchgrass communities due to post European settlement over grazing, fire suppression, and climatic warming at the end of the little ice age (Miller and Tausch 2001). Woodland expansion can lead to habitat elimination for native animal species, reduction of understory biomass, increased risk of catastrophic wildfire, and conversion to annual grass dominated communities (Young and Evans 1973). Pinyon expansion could also bring about changes in biogeochemical cycles and increase the risk of soil erosion (Norris et al 2001; Davenport et al. 1998).

Re-instating pre-European settlement fire regimes has been proposed to slow the rate of woodland expansion in the Great Basin. Historic fire regimes in these woodlands are incredibly variable and range from 30 years in Wyoming big sagebrush dominated systems to 400 years in old-growth woodlands found on steep rocky slopes (Baker and Douglas 2004; Miller and Tausch 2001).

We have chosen an area typical of expanding central Great Basin woodlands where the mean fire return interval is thought to be between 15 and 90 years to determine if spring prescribed burning is a viable option for reducing woodland cover. The woodland is located in central Nevada on the Humboldt-Toiyabe National Forest and contains a mix

of tree dominated areas ranging from low (12% cover, 2152 kg/ ha) to high tree dominance (74% cover, 14213 kg/ ha) (Reiner 2004). This study examines the impacts of pinyon expansion and spring prescribed burning on soil physical properties, soil chemical properties, understory biomass, and understory tissue nutrient concentrations.

In the first experiment, we conducted single ring infiltrometer experiments on three cover types typical of transitional woodlands (under tree canopy, under shrub canopy, and interspace) to determine how cover type effected infiltration of water into soil. Experiments were also conducted under tree canopies of different tree dominance to determine if percent tree cover influenced infiltration. The experiment was conducted over an elevation gradient typical of woodlands in this area to determine elevation and soil characteristic impacts on infiltration. Experiments were performed one year before the spring prescribed burn and then two months afterward to determine how cover type, cover percent, and elevation influenced burn induced changes in surface soil water repellency. Supplemental water drop penetration time experiments were also performed in order to determine how soil heating during spring prescribed fire at the three cover types affected the development of water repellency.

The second experiment was designed to determine spatial, temporal, and burn induced changes in soil nutrient availability in a woodland with intermediate tree dominated woodland. A control and burn plot were sampled in the fall before the spring prescribed burn and two subsequent fall seasons afterward at the three cover types described in the previous experiment. Soil was sampled at four depth increments (0-8, 8-23, 23-38, and 38-52cm) to determine the extent of fertile island effects and track the fate of vertically mobile nutrients. A second set of soil samples was collected from all cover types at two

smaller depth increments (0-3 and 3-8cm) two days prior to the spring burn and then two days afterward to determine the more immediate effects of burning on soil fertility.

The third experiment was designed to compliment the second experiment and to determine the temporal and spatial influence of a spring burn on understory plant aboveground biomass and tissue nutrient concentrations, and to explore the link between burning, soil fertility, and plant nutrition. Six understory plant species (*Lupinus argenteus*, *Crepis acuminata*, *Eriogonum umbellatum*, *Eriogonum elatum*, *Poa secunda secunda*, and *Festuca idahoensis*) were collected from locations corresponding to those described in the second experiment. Plants were collected in the summer prior to the spring burn and two summer seasons following.

The experimental designs, methods, and results are discussed in detail in the following three chapters.

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Hydrologic response of a central Nevada pinyon-juniper (*Pinus monophylla-Juniperus osteosperma*) woodland to prescribed fire

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Keywords: Surface hydrology, pinyon-juniper woodlands, prescribed fire

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Abstract

This study was conducted to further determine the effect of prescribed fire on surface hydrology in Great Basin pinyon-juniper woodlands. The study was conducted 35 miles south of Austin, Nevada in the Shoshone mountain range. Infiltration rates were measured using a single ring infiltrometer over an elevation gradient (2103, 2225, and 2347m), at three microsites (under tree canopy, under shrub canopy, and interspace) in Aug. 2001 before a spring prescribed burn conducted in May 2002 and then following the prescribed burn in Aug. 2002. Data collected from infiltration experiments were used to calculate saturated hydraulic conductivity ($K(\theta_s)$) rates. Soil was collected and water drop penetration times were determined to evaluate the development of water repellent soils. Final infiltration rates before the burn were higher at the low elevation than at the mid and high elevation study sites, although the data could not be numerically analyzed due to the inability to pond water on the soil surface at the low elevation. Before burning, the final infiltration and saturated hydraulic conductivity ($K(\theta_s)$) rates measured on the interspace and under shrub canopy microsites were lower than on those microsites located beneath tree canopies at the mid-elevation study site (2225m). Following burning, only the intermediate tree cover under tree canopy microsites had higher final infiltration rates than interspace microsites, all other microsites were similar. No significant differences in $K(\theta_s)$ rates were found among the three microsites after burning. However, on the higher elevation study site before the burn the interspace microsites had lower final infiltration rates than those located under the tree canopy, and burning caused no significant deviation from this trend. Saturated hydraulic conductivity rates at the high elevation did not differ by microsite before the burn, but after burning interspace

microsites had lower $K(\theta_s)$ rates than under tree microsites. Burning increased water repellency of surface soils (0-3 cm) for all cover types.

Introduction

Pinyon-juniper woodlands in the Great Basin currently occupy about 18 million ha (Miller and Tausch 2001), an increase of roughly 60 to 94 percent from the area occupied prior to European settlement approximately 140 years ago (Gruell 1999). Although pinyon-juniper woodlands have expanded and receded several times over the last 5,000 years, the rate of expansion over the last 140 years is unprecedented, and less than 10 percent of current woodlands are older than 140 years (Miller and Tausch 2001). Prior to European settlement, wildfire return intervals were 25 to 130 years restricting woodlands to steep rocky terrain with limited understory vegetation. These fire resistant sites currently retain woodlands of age classes greater than 140 years (Miller and Tausch 2001). Domestic livestock grazing, warming at the end of the little ice age, and fire suppression explain current woodland expansion (Miller and Wigand 1994; Miller and Rose 1995, 1999).

Increasing tree dominance and crown cover eliminates understory vegetation, and increases crown fuel continuity across the landscape (Tausch 1999a, b). Crown cover exceeding 50 percent is sufficient to carry high intensity fire during dry or windy periods. Woodlands with this coverage now occupy 25 percent of the current area, and are expected to double over the next 50 years (Miller and Tausch 2001).

High severity wildfires have been reported to produce detrimental hydrologic responses in arid and semi-arid woodland ecosystems (Covington and DeBano 1990). Illg and Illg (1997) and Savage et al. (1998) have reported increased water repellency, flooding, and debris flows following wildfire in ponderosa pine (*Pinus ponderosa*) forest. Poff (1989) has found severe water repellency in white fir (*Abies concolor*) forest after wildfire, and

Pierson et al. (2001, 2002) described decreased infiltration and increased sediment yield in sagebrush-steppe following wildfire. Little information regarding post wildfire surface hydrology has been reported in pinyon-juniper woodlands where landscape response to fire is highly dependent on the timing of precipitation. In the Southwest, for example, a late spring fire season is followed by the summer monsoon characterized by short duration high intensity rainfall (Mohrle 2003). By contrast, in the Great Basin most fires occur in late summer, and precipitation comes mostly as winter snow and spring rains.

Infiltration rates are often high in mature pinyon-juniper woodlands, but prescribed burning can reduce infiltration and increase sediment yield (Buckhouse and Gifford 1976). The effects of prescribed fire on infiltration and runoff may be less pronounced in pinyon-juniper than in other cover types such as oak (*Quercus virginiana*), bunchgrass, and shortgrass (Hester et al. 1997). Burning can have variable effects on soil hydrologic response based on fire severity and antecedent soil and fuel moisture content (DeBano 2000). Pinyon-juniper woodlands are spatially heterogeneous, and under shrub and under tree canopy microsites tend to have more litter and soil organic carbon than interspace microsites (Davenport et al. 1998, Chambers 2001). Burning volatilizes organic hydrocarbon compounds, which are often drawn into the soil profile due to steep temperature gradients, and condense onto soil particles creating a water repellent layer (DeBano et al. 1970). The depth and extent to which water repellent conditions develop is determined by soil moisture and soil specific surface area (DeBano et al. 1970, 1976). Pinyon-juniper woodlands occur in areas of variable particle size dependent on landscape position and relief (Davenport et al. 1998) (D. Zamudio, pers. com. 2001; USDA Soil Scientist Lakeview, OR).

Burning may also alter soil hydrologic properties by the addition of ash to the soil. Ash may fill small soil pores, or cause dispersion of expandable clays by increasing potassium carbonate (Durgin 1985). Intense burning may reduce soil elasticity and plasticity by consumption of organic matter and formation of coarse-grained silica aggregates (Ulery and Graham 1993; Ulery et al. 1996).

Prescribed fire has been suggested as a management tool to decrease pinyon-juniper expansion and reduce the risk of high severity wildfire. In this study we investigate the soil hydrologic response (water repellency, final infiltration rate, and saturated hydraulic conductivity) of pinyon-juniper woodland to a prescribed burn. Specific objectives were to: 1) Determine the hydrologic response of soil over the typical elevation gradient found in pinyon-juniper woodlands; and 2) Determine how spatial variability and cover type within woodlands affect the soils hydrologic response to burning.

Materials and Methods

Study Area

The study area is located in the Shoshone Mountain Range on the Humboldt-Toiyabe National Forest (Austin Ranger District) in Nye and Lander Counties, Nevada. Underdown Canyon (39°15’11” N, 117°35’83” W) is oriented east to west and contains infrequent springs and an ephemeral stream near the top of the drainage. Average annual precipitation ranges from 23cm at the bottom to 50cm at the top of the drainage and arrives mostly in winter as snow and spring rains (USDA Forest Service unpublished data). Average annual temperature recorded in Austin, NV ranges from –7.2°C in January to 29.4°C in July. Lithology of the Shoshone range consists of welded and non-welded

silica ash flow tuff. Soils developed on alluvial fans in this study are classified as Coarse loamy mixed frigid Typic Haploxerolls. Coarse mineral particles decease and silt and sand particles generally increase with increasing elevation (Table 1). The slope of these side-valley alluvial fans ranges from 5 to 20 percent.

Vegetation is dominated by single leaf pinyon (*Pinus monophyllla*), and associated sagebrush (*Artemisia tridentata*) communities. Utah juniper (*Juniperus osteosperma*) occurs in minor amounts throughout the study area. Wyoming sagebrush (*Artemisia tridentata wyomingensis*), Sandberg bluegrass (*Poa secunda secunda*), squirrel tail (*Eeleymus elymoides*), and needle and thread grass (*Stipa comata*) dominate lower elevations. Mountain big sagebrush (*Artemisia tridentata vaseyana*) occurs at mid to high elevations, as does Idaho fescue (*Festuca idahoensis*) and bluebunch wheatgrass (*Pseudoroegneria spicata*). Forbs in the study area include *Eriogonum* species, western hawksbeard (*Crepis acuminata*), longleaf phlox (*Phlox longifolia*), pale agoseris (*Agoseris glauca*), tailcup lupine (*Lupinus argenteus*), *Penstemon* species, and others in lower abundance. Cheatgrass (*Bromus tectorum*), an invasive annual grass is not a large component of the study area. The vegetation occurs in patches of variable tree dominance and is classified as low (12% cover, 2152 kg ha^{-1}), intermediate (38% cover, 6722 kg ha^{-1}), and high tree dominance (74% cover, 14213 kg ha^{-1}) (Reiner 2004).

Study Design

The study was a split plot design with sub-sampling. The study sites were located on north-east facing alluvial fans at elevations of 2103, 2225, and 2347m. Three plots were sampled at each elevation. Plots were characterized by intermediate tree dominance at all elevations (n=9) with three additional high tree dominance plots occurring at the 2225m

site. All plots measured approximately 1000m^2 and contained a mixture of trees, shrubs, and interspaces. Two sub-samples were located in each of three microsites (under shrub canopy, under tree canopy, and interspace) on all plots except for the high tree dominance plots where only the under tree canopy microsite was sampled (n=60). USDA Forest Service fire personnel burned the study plots on 11-14, May 2002 under favorable weather conditions. Maximum soil temperatures were recorded during the fire using heat sensitive paints on metal strips. Strips were placed parallel to the soil surface at 0, 2, and 5cm soil depths at all microsites (Korfmacher et al. 2002).

Field Data Collection

To characterize the soil, pits were dug at all three elevations. Soil horizons were identified and samples were collected for particle size analyses (Table 1). Samples were sieved to 2mm; organic matter was removed with H_2O_2, and particle size distribution determined by the pipette method (Black 1965).

Infiltration was measured using a 345mm diameter Single Ring Infiltrometer (SRI) before the burn in Aug. 2001 and again in Aug. 2002 following the prescribed burn (n=120). The ring was placed at each microsite (under tree canopy, under shrub canopy, and interspace) so that the soil surface was not disturbed. The ring was inserted into the mineral soil surface to a depth of approximately 50mm on localized slopes from 0 to 10 percent. A Scotch™ pad was placed in the center of the ring and 2L of water were poured onto the pad in order to pre-wet the soil and prevent dispersion at the soil surface. Water was then ponded to a depth of 20mm in the center of the ring and maintained for the duration of the test. Volume measurements of water added were taken every two minutes for thirty minutes (Bouwer 1986). Cumulative infiltration (I = mm^3 mm^{-2}) was plotted

against time (t). After the first year, it was determined that the slope of I vs. t rapidly reached a constant value such that it could be obtained by shortening the total test time from 30 to 16 min. By plotting the slope ($\Delta I/\Delta t$) between each measurement interval over time (mm min^{-1} vs. time) a standard infiltration curve (i vs. t) was obtained. Because i was not constant, final infiltration rates (i_f) for each experiment were obtained by taking the mean value of the 10-16 minute intervals (Pierson et al. 2002).

A sub-set of soil samples was collected from the intermediate elevation (2225m) study site in order to assess the development of water repellency. Soils were collected two days before the burn from all plots and microsites at depths of 0-3 and 3-8 cm (n=24). Collection sites were marked with a metal stake so that they were easily located after the prescribed burn. Samples were again collected two days after the burn from the same locations (n=48). Soils were brought back to the lab, air dried, and sieved to 2mm. Water Drop Penetration Times (WDPT) were performed by pipetting one drop of water onto each soil sample's surface (Letey 1969). The time for each drop to be absorbed onto the soil was recorded with a minimum and maximum value of 1 and 180s, respectively. Three replicates were performed on each sample (n=144).

Theory

Saturated hydraulic conductivity $K(\theta_s)$ rate (mm min^{-1}) was estimated when there was no change in the volume infiltration over time. The following relationship for 1-dimensional saturated infiltration in the vertical direction was applied:

$$Q/At = -K(\theta_s)[(\Delta z+\Delta p)/L] \qquad [1]$$

where Q/At (mm^3 mm^{-2}) is the measured parameter, Q is the volume of water flowing per unit of time (mm^3 min^{-1}), A is the cross sectional area (mm^2) of the SRI, t is time (min),

$K(\theta_s)$ is the saturated hydraulic conductivity (mm min^{-1}), z is the gravitational potential (mm), p is the pressure potential (mm), L is the column length (mm), and the change of z and p over depth $[(\Delta z+\Delta p)/L]$ is the hydraulic gradient. For vertical infiltration the gravitational gradient $\Delta z/L = 1$ and as L approaches infinity the pressure gradient $(\Delta p/L)$ approaches zero. For long-term saturated vertical infiltration Eq. [1] may then be reduced to

$$Q/At = -K(\theta_s)\ (1) \qquad [2]$$

Statistical Analyses

Differences in infiltration rate and estimated saturated hydraulic conductivity among elevation, microsites, and dates were evaluated by SAS™ using mixed effects models (SAS Institute 2004). Elevation was treated as a main effect, microsite was a split-plot within elevation, and treatment was a split-split-plot within elevations and microsite. Differences in water drop penetration times among microsites, depth, and dates were also evaluated using the same procedure. Microsite was treated as a main effect, depth was a split-plot within microsite, and treatment was a split-split-plot within depths and microsites (SAS Institute 2004).

Results

Soil heating and Water Repellency

Soil heating was greatest in under shrub canopy microsites at the soil surface and at the 2cm depth. Under tree canopy microsites had slightly lower temperatures, and interspace microsite soils were only heated on the surface (Table 2).

Water drop penetration times were affected by depth, treatment, treatment-microsite interactions, treatment-depth interactions, and treatment-microsite-depth interactions (Table 3). The water repellency of the soils was not statistically different for any microsite or depth prior to the prescribed burn. After the prescribed burn the surface soil (0-3cm) at all microsites exhibited increases in water drop penetration time but microsite differences did not occur (Figure 1). Sub-surface soil (3-8cm) remained unchanged by fire in all but the under shrub canopy microsite. Soil at this depth and microsite was found to be statistically similar to surface soils after the burn (Figure 1).

Final Infiltration and Saturated Hydraulic Conductivity Rates

Pre-burn soil final infiltration rates were highest on the lower elevation (2103m) study site (Table 4). Extremely coarse soil prevented ponding of water on the soil surface in most cases and for this reason the study site at 2103m was not included in the statistical analyses. After the burn, the infiltration of water into soils was reduced on all the different vegetative cover types at 2103m (Table 4). However, because there was little pre-burn numeric data for comparison the results are strictly observational.

Results of the ANOVA show that final infiltration rates were affected by microsite, treatment, treatment-microsite interactions, and treatment-elevation-microsite interactions (Table 5). Final infiltration rates on the intermediate elevation (2225m) study site were affected by microsite and treatment (Table 4, Figure 2). Before the burn, interspace and under shrub microsites had statistically lower final infiltration rates than under tree microsites. After the burn, under shrub and interspace microsites retained comparably low infiltration rates. High tree dominance under tree microsites, however were no longer

statistically different than any other microsite, and intermediate tree dominance under tree microsites were similar to under shrub microsites (Table 4, Figure 2).

At the highest elevation (2347m) study site, the final infiltration rates were affected by microsite (i.e. cover) but not by burning treatment. (Table 4, Figure 1). Interspace microsites had the lowest final infiltration rate before and after the burn, but were only statistically different than under tree canopy microsites.

Estimated $K(\theta_s)$ rates were similar to the mean final infiltration rates in most cases. However, a separate comparative analyses was warranted because the calculated $K(\theta_s)$ rate values are not a direct transformation of the final infiltration rates (i_f,) but are instead based on cumulative final infiltration rate (I_f).

The low number of data points on the lower elevation (2103m) study site again rendered statistical comparisons impossible. However, the observations suggest that large decreases in $K(\theta_s)$ rates occurred after the fire on all of the microsites (Table 4).

Results of the ANOVA show that $K(\theta_s)$ rate is affected by microsite, treatment, and treatment-microsite interactions (Table 5). At the intermediate elevation (2225m) study site $K(\theta_s)$ rates were affected by microsite and treatment. The soils on microsites beneath tree canopies had statistically higher $K(\theta_s)$ rates than soils under interspace and shrub vegetation before the burn, but following the burn no differences in $K(\theta_s)$ rates were present among the three vegetative cover types (Table 4).

For the high elevation (2347m) study site, all microsites were statistically similar before the burn, but after the burn interspace microsites differed from under tree canopy microsites (Table 4).

Discussion

Soil Heating and Water Repellency

Water repellency following prescribed fire was affected by soil texture and soil heating. Fire behavior resulting from the different cover conditions (i.e. under tree and shrub canopies and interspaces) affected soil heating and post-fire water repellency patterns.

The differences in particle size distribution among the three study sites over the elevation gradient most likely resulted from the localized variation in climate. Although only 244m in elevation separated the lower and upper elevation study sites, the temperature and precipitation regimes were dramatically different. The average annual precipitation ranges from 23cm at the bottom of the watershed to 50cm at the top of the drainage and snow frequently accumulates at the higher elevations because of cooler temperatures (USDA Forest Service unpublished data). Over time these differences in climate have resulted in differential soil development and the differences in soil texture measured at the different study sites. In general, the percentage of greater than 2mm and silt sized mineral soil particles decreased with elevation compared to percent clay and sand sized mineral particles which increased (Table 1). For example, about 76 percent of the surface soil at the lower elevation (2103m) study sight was made up of particles greater than 2mm, and 53 percent of the less than 2mm mineral soil particles were sand-sized particles. These coarse textured soils not only affected water repellency, but also interfered with the infiltration trials. In contrast, the surface soil at the higher elevation (2347m) study site contained only about 42 percent of particles greater than 2mm, and almost 8 percent of the less than 2mm mineral soil particles was made up of clay (Table

1). Variation in course fragment and soil specific surface area affects the development of water repellency (DeBano et al. 1970).

During the spring prescribed burn maximum soil temperatures measured with temperature-sensitive paints indicated that soil heating was generally limited to less than 5cm soil depth, and that soil heating was greatest under the shrub canopies both at the soil surface and at the 2cm soil depth (Table 2). Soil heating occurred under the tree canopies at the soil surface and 2cm depth, but maximum observed temperatures were less than under shrub canopies. In the interspace microsite soil temperatures were raised only at the soil surface and that increase was less than occurred under either the shrub or tree canopies. Although sagebrush canopies burn at higher temperatures than understory vegetation (Neary et al. 1999), they do not support a substantial litter layer (< 5cm) as compared to that found under pinyon tree canopies (6-10cm) (Reiner 2004; USDA Forest Service unpublished data 2004). At the time of the prescribed fire, the fuel moisture content of the thicker surface litter layers under the pinyon trees was high (≈ 45%) and insulated the soil at the 2cm depths or greater from the intense soil heating that occurred under the flaming sagebrush canopies (Table 2) (USDA Forest Service unpublished data 2004). Insulation of the soil surface from intense heating may not occur if prescribed burning occurs in the fall. Fuel moisture is typically lower during fall burns and litter mats may be completely consumed, resulting in more intense soil heating under tree canopies.

Water repellency is a common phenomenon found in soils of arid regions (both burned and unburned conditions) because the hydrophobic substances that produce water repellency can be stored in the litter on the soil surface until they are either leached

downward by water on unburned sites or are volatilized and moved downward in the soil during a fire (Parks and Cundy 1989; Brock and DeBano 1990; Covington and DeBano 1990). In this study, the water repellency produced after the fire differed among the three study sites. At the highest elevation (2347m), there was only minimal water repellency produced as a result of the fire. This may have occurred because of the low content (42%) of coarse mineral particles greater than 2mm in the soils found on this site. Soils having smaller particle sizes and higher specific surface areas are generally less susceptible to becoming water repellent than coarser textured soils because they do not allow the organic distillate to penetrate as deeply into the soil. Those substances that do penetrate do not effectively coat the larger surface area of the mineral particles (DeBano et al. 1970; Brock and DeBano 1990). The soils collected from at the 0-3cm depth on the mid-elevation (2225m) study site immediately before the burn, were found to be slightly water repellent or not water repellent (Figure 1) (Robichaud and Hungerford 2000). However, following the burn the surface soils on all of the microsites on the mid-elevation (2225m) study site were found to be more water repellent (Figure 1). The only sub-surface soil layer (3-8cm) affected by fire was under shrub canopies (Figure 1). At this microsite, the maximum soil temperature measurements at the 2cm depth were the highest that were measured during the study (Table 2). This intense soil heating under the shrub canopy most likely produced large temperature gradients which moved volatilized hydrophobic substances more deeply in the soil profile (DeBano et al. 1976; DeBano 2000).

Final Infiltration and Saturated Hydraulic Conductivity Rates

Before the prescribed burn (in 2001), the soil differences among the study sites at the three different elevations made it difficult to successfully measure infiltration and

hydraulic conductivity rates. For example, the lower elevation study site (2103m) contained a high proportion of greater than 2mm particles (76%), making it nearly impossible to pond water on the surface during the infiltration trials (Table 1). Therefore, the few infiltration tests that were completed were made only in the interspaces (2 tests) and under shrub canopies (1 test), but not under tree canopies (Table 4). These tests did not provide enough numerical data to include into the statistical analyses. Although, the surface soils at the intermediate elevation (2225m) study site were coarse-textured and were made up of about 60 percent of particles greater than 2mm (Table 1), it was possible to determine both final infiltration rates and hydraulic conductivity rates (Table 4, Figure 2). On these intermediate elevation study sites, both the final infiltration and hydraulic conductivity rates were greater under the tree canopies than under shrub canopies and in interspaces (Table 4, Figure 2). Tree dominance did not affect final infiltration or $K(\theta_s)$ rates under tree canopies (Table 5, Figure 2). On the higher elevation study site (2347m) the final infiltration rates under the tree canopies were greater than in the interspaces but those under shrub canopies were no different from under tree canopy or interspace microsites (Table 4, Figure 2). High spatial variability of infiltration and hydraulic conductivity rates (such as occurred at the different elevation and cover conditions in this study) has also been reported (Parks and Cundy 1989; Hester et al. 1997; Pierson et al. 2002). Some investigators suggest that this high heterogeneity could be a result of spatial variability in soil organic matter and root density between microsites (Hester et al. 1997; Davenport et al. 1998).

The effect of spring burning on the different study sites and microsites was highly variable. However, the infiltration and $K(\theta_s)$ rates were reduced by burning at all

microsites on the lower elevation (2103m) study site and by burning under the tree canopies on the intermediate-elevation (2225m) study site (Table 4, Figure 2). Tree dominance did not affect the response under tree canopy microsites to burning. After the fire final infiltration and $K(\theta_s)$ rates were less variable than before burning (Figure 2), indicating that fire generally reduces the heterogeneity of landscapes in natural systems (Parks and Cundy 1989). The combined effect of soil surface pore space reduction by ash and the condensation of hydrophobic substances onto mineral surfaces appear to be the major mechanisms responsible for the reduced infiltration and $K(\theta_s)$ rates. Because infiltration and $K(\theta_s)$ rates are reduced but cannot become negative the landscape appears more heterogeneous.

Conclusions

The results of this study and others (Hester et al. 1997) indicate that prescribed burning in pinyon-juniper woodlands can affect soil hydrologic characteristics. We have shown that fire induced water repellency can be spatially variable after prescribed burning, and it is therefore important that managers consider the soil characteristics, type and percentage cover of vegetation, and climate of the woodland with which they are dealing. Development of water repellent soils is affected by elevation gradients due to differences in surface soil particle size distribution, especially coarse mineral particles (Table 1). Water repellency is also affected by spatial variability in surface and soil organic matter associated with the type of vegetation and its percentage canopy cover. Spring burning in sagebrush canopy microsites may have little or no effect on surface soil hydrology if the surface soil coarse mineral particles are less than 70 percent. However, burning in pinyon canopy microsites may cause decreases in infiltration and $K(\theta_s)$ rates if the surface soil

coarse mineral particles are greater 40 percent. The degree of tree dominance does not affect under tree canopy response to burning, but it will influence the percentage of the landscape affected by fire induced water repellency. Fall burning may result in more widespread and intense soil heating because fuel and soil moisture are typically lower resulting in greater development of water repellency (Robichaud and Hungerford 2000).

The effects of reduced infiltration, $K(\theta_s)$, and the development of water repellency must be considered in the context within which they occur. Soil with coarse mineral particles greater than 40 percent will likely have high infiltration and $K(\theta_s)$ rates. The probability of a precipitation event exceeding $K(\theta_s)$ should decrease as $K(\theta_s)$ increases within a climatic region. It must also be noted that heat induced water repellency can be short lived, and broken down by light intensity precipitation and spring wetting (Morris and Moses 1987; McNabb et al. 1989).

Alluvial fans of Underdown canyon in central Nevada are high in coarse mineral particles. Also precipitation occurs dominantly as winter snow and spring rain, and obvious evidence of overland flow is lacking. High intensity summer monsoon events are infrequent in contrast to areas in Colorado, Arizona, and New Mexico (Mohrle 2003). Saturated hydraulic conductivity data collected from this site and data from the NOAA indicates that a five-minute storm event intense enough to exceed the lowest levels of conductivity occurs on an interval greater than one thousand years (Table 6). It is therefore unlikely that prescribed fire in an area similar to Underdown Canyon will cause detrimental hydrologic response unless unusually high intensity precipitation occurs.

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Table 1. Soil particle sizes of soils at elevations of 2103 m, 2225 m, and 2347 m.

depth	>2mm%	sand%	silt%	clay%
---------- 2103 m ----------				
0-15 A	76.2	53.2	44.1	2.7
15-50 AB	72.0	52.3	43.3	4.4
50-70 B_w	81.7	63.6	31.0	5.4
70-100 BC	75.0	64.4	33.5	2.1
---------- 2225 m ----------				
0-16 A_1	60.5	67.6	23.9	8.5
16-28 A_2	55.3	59.3	33.5	7.2
28-40 BA	47.8	65.9	27.1	7.1
40-70 B_w	47.0	53.9	21.4	24.6
70-100 BC	31.1	64.5	24.8	10.7
---------- 2347 m ----------				
0-15 A_1	41.6	67.6	24.9	7.5
15-38 A_2	35.3	62.9	30.9	6.2
38-65 B_w	53.7	63.2	29.0	7.8
65-100 BC	55.9	65.1	29.8	5.1

The >2mm% is of the total soil. The sand, silt, and clay% make up 100% of the <2mm soil fraction Letters following soil depth refer to the soil horizon nomenclature.

Table 2. Mean (n≈31) maximum soil temperatures for surface, 2, and 5cm soil depths at all microsites.

Microsite	Avg. temperature (°C)
----------surface----------	
Interspace	206
Under shrub	369
Under tree	304
----------2 cm----------	
Interspace	40
Under shrub	86
Under tree	77
----------5 cm----------	
Interspace	40
Under Shrub	40
Under Tree	44

Table 3. ANOVA results for water drop penetration times in pinyon-juniper soils subjected to prescribed burning under different vegetation cover (microsites).

Factor	**DF**	**F**	***P***
Microsite	2	1.48	0.2785
replicate*microsite	9		
Depth	1	18.79	0.0019
microsite*depth	2	1.01	0.4007
replicate*microsite*depth	9		
Treatment	1	96.28	<.0001
treatment*microsite	2	4.68	0.0112
treatment*depth	1	37.06	<.0001
treatment*microsite*depth	2	7.14	0.0012
treatment*replicate*microsite*depth	111		

Table 4. The average final infiltration and saturated hydraulic conductivity [K(θ_s)] rates (mm minute^{-1}) of pinyon-juniper soils located at different elevations and under different vegetation cover (microsite) prior to (2001) and following (2002) a prescribed burn. Means followed by the same letter do not differ. (LSM; $P \leq 0.05$).

Elevation	**2103m**		**2225m**		**2347m**	
Year	**2001**	**2002**	**2001**	**2002**	**2001**	**2002**
----------Mean Final Infiltration Rate (mm minute^{-1})----------						
Intermediate						
Interspace	35	16	4 CD	5 CD	4 CD	3 D
Under shrub	65	18	8 BC	9 BC	7 BCD	6 BCD
Under tree	∞	21	18 A	10 B	9 B	10 B
High						
Under tree	x	X	20 A	6 BCD	x	x
----------Estimated Saturated Hydraulic Conductivity (mm minute^{-1})----------						
Intermediate						
Interspace	75	17	5 BC	5 BC	5 BC	4 C
Under shrub	75	26	8 BC	9 BC	7 BC	6 BC
Under tree	∞	25	18 A	11 B	8 BC	11 B
High						
Under tree	x	x	20 A	7 BC	x	x

The “Intermediate” and “High” headings refer to the stage of tree dominance where the under tree canopy tests were performed. The symbols “x” and “∞” refer to microsites that did not occur on the particular study site and inability to pond water on the soil surface respectively.

Table 5. ANOVA results for final infiltration and saturated hydraulic conductivity [K(θ_s)] rates (mm minute^{-1}) of pinyon-juniper soils subjected to prescribed burning at different elevations and under different vegetation cover (microsites).

	final infiltration rate			saturated hydraulic conductivity		
Factor	**DF**	**F**	***P***	**DF**	**F**	***P***
Elevation	1	3.22	0.1471	1	5.02	0.0887
replicate(elevation)	4					
Microsite	3	10.10	0.0023	3	8.9	0.0036
elevation*microsite	2	0.52	0.6104	2	0.78	0.4857
microsite*replicate(elevation)	10					
Treatment	1	38.45	<0.0001	1	8.19	0.0061
treatment*microsite	3	14.96	<0.0001	3	3.31	0.0272
treatment*elevation	1	1.81	0.1797	1	0.38	0.5383
treatment*elevation*microsite	2	7.20	0.0009	2	1.15	0.3243
treatment*microsite*replicate(elevation)	51					

Table 6. Estimated frequencies (years) and rates (mm minute^{-1}) of different precipitation events at two areas near the study site in Nevada. Modified from:(http://dipper.nws.noaa.gov/hdsc/pfds/)

Duration	5 min	10 min	15 min	30 min	1 hr
Freq (yrs)					
----------Nevada 39.022ºN 117.488ºW 2055 m--------					
10	1.0	0.8	0.6	0.4	0.3
100	2.0	1.5	1.2	0.8	0.5
1000	3.6	2.7	2.2	1.5	0.9
----------Nevada 39.426ºN 117.085ºW 2413 m--------					
10	1.1	0.8	0.7	0.5	0.3
100	2.1	1.6	1.4	0.9	0.6
1000	4.0	3.0	2.5	1.7	1.0

Figure 1. Mean infiltration rates (mm minute-1) over time for interspace (IS), under shrub (US), and under tree (UT) microsites on intermediate tree dominance plots at 2225m and 2347m, and for under tree microsites on high tree dominance plots at 2225m.

Figure 2. Mean drop penetration times and standard errors before and after prescribed fire for interspace, under shrub, and under tree microsites at 0-3 cm and 3-8 cm. Different letters indicate differences among treatment, microsites, and depths (LSM; $P < 0.05$).

Figure 1.

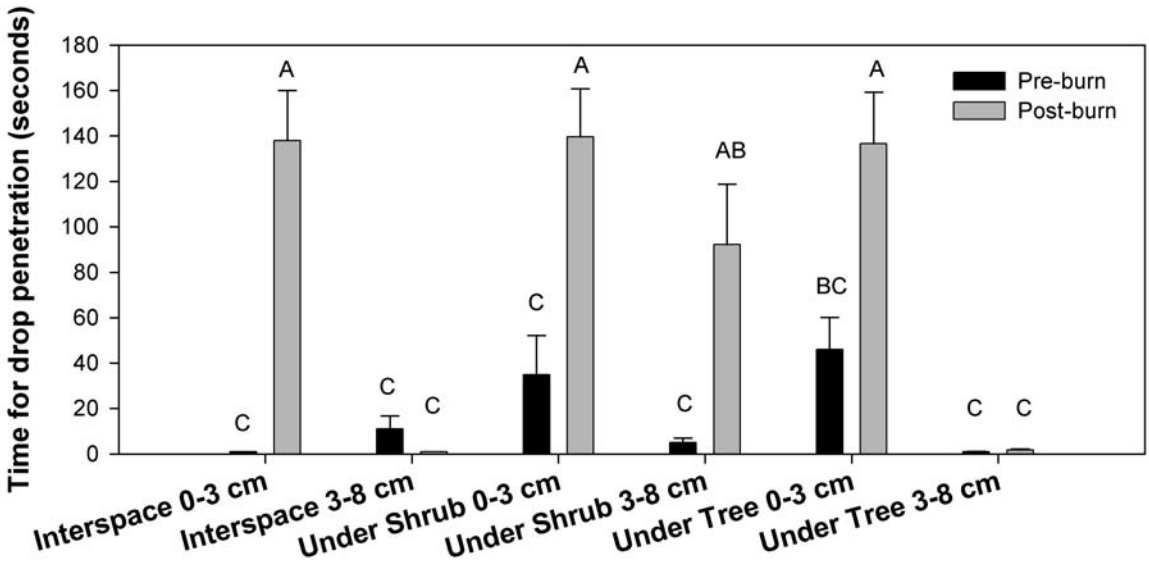
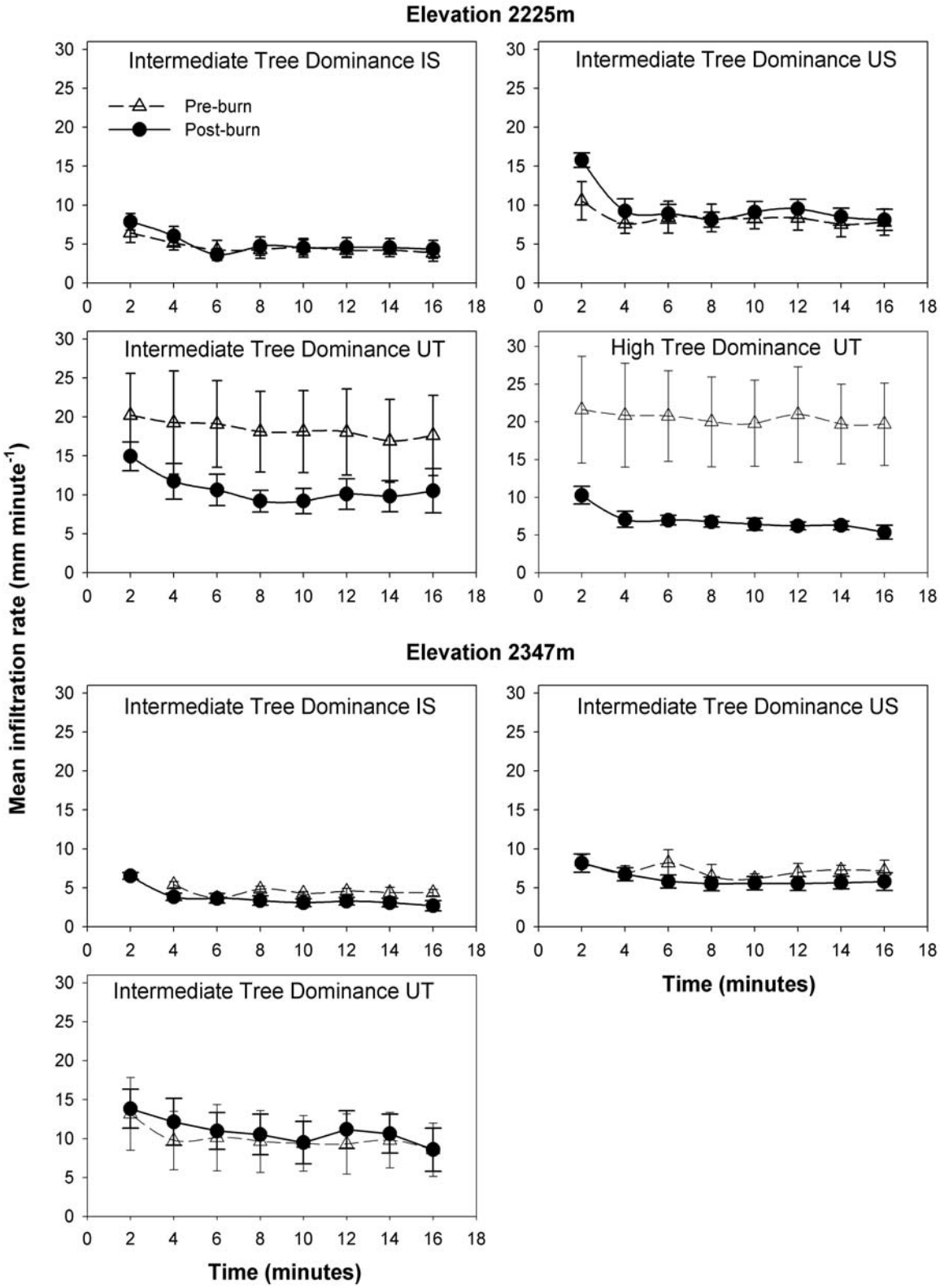

Time for drop penetration (seconds)
180
160
140
120
100
80
60
40
20
0
Pre-burn
Post-burn
A
A
A
AB
BC
C
C
C
C
C
C
C
C
Interspace 0-3 cm
Interspace 3-8 cm
Under Shrub 0-3 cm
Under Shrub 3-8 cm
Under Tree 0-3 cm
Under Tree 3-8 cm

Figure 2.


Elevation 2225m
Intermediate Tree Dominance IS
Pre-burn
Post-burn
Intermediate Tree Dominance US
Intermediate Tree Dominance UT
High Tree Dominance UT
Elevation 2347m
Intermediate Tree Dominance IS
Intermediate Tree Dominance US
Intermediate Tree Dominance UT
Mean infiltration rate (mm minute-1)
Time (minutes)
Time (minutes)

Pinyon-juniper expansion and prescribed fire effects on soil in Great Basin sagebrush ecosystems

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ABSTRACT

In the central Great Basin, loss of herbaceous vegetation through livestock grazing combined with wildfire suppression and aspects of global change have facilitated expansion of Pinyon-juniper (*Pinus monophylla-Juniperus osteosperma*) woodlands. We attempt to quantify the effects of pinyon-juniper woodland expansion and prescribed fire, on soil at three microsites (under tree, under shrub, and shrub interspace). Soils were sampled in November of 2001 to determine spatial pre-burn soil nutrient characteristics. Additional soil samples were taken two days before and two days after the burn in May 2002 to determine immediate fire effects. Soils also were sampled in November of 2002 and 2003 to determine temporal and extended burn effects on soils. Fire resulted in immediate increases in soil pH, ortho-P, NH_4^+, Ca^{2+}, Mg^{2+}, Mn^{2+}, and Zn^{2+}. Immediate losses of NO_3^- also occurred. Chemical changes persisted into November 2003 with the exception of Ca^{2+}, and large increases in available NO_3^- occurred. Prescribed burning in woodlands can cause immediate and persistent effects on soil available nutrients and pH. Burning effects can be limited to surface soils and a particular microsite, or they can occur over the entire burn area and be observed at soil depths of 52 cm. Spring burning in Pinyon-juniper woodlands similar in topography, soils, and climate to Underdown Canyon should be a viable method for reducing woodland expansion and preventing catastrophic wildfire which may reduce erosion in these woodlands.

Introduction

Vegetation changes associated with climate shifts and anthropogenic disturbance are thought to have major impacts on biogeochemical cycling and soils (Schimel et al.1991, 1994). Much of the Great Basin is currently dominated by sagebrush semi-desert ecosystems. These communities are heterogeneous in regards to vegetation and soil chemistry. Shrubs are associated with fertile islands of soil found to be lower in bulk density and higher in organic matter, available nutrients, and total N than the surrounding interspaces (Bolton et al. 1990; Doescher et al. 1987; Jackson and Caldwell 1993; Chambers 2001). At higher elevations, sagebrush semi-desert is increasingly influenced by pinyon and juniper expansion. Pinyon (*Pinus monophylla*) and juniper (*Juniperus osteosperma*) woodlands have expanded their pre-European settlement range in the Great Basin by more than 60% over the last 140 years due to a combination of climate change, fire suppression, and overgrazing by livestock (Gruell 1999; Miller and Wigand 1994: Miller and Rose 1999). Although pinyon-juniper woodlands have expanded and receded several times over the last 5,000 years, the current rate of expansion is unprecedented. Less than 10% of current woodlands are of age classes exceeding 140 years (Miller and Tausch 2001). As pinyon-juniper woodlands increasingly dominate sagebrush semi-desert communities, they compete for available resources and often eliminate most understory vegetation (Reiner 2004). The expansion of pinyon-juniper woodlands and reduction of sagebrush semi-desert may alter biogeochemical cycles and the distribution of soil nutrients across the Great Basin by reducing landscape heterogeneity and increasing aboveground biomass.

Pinyon and Juniper expansion has resulted in increased crown fuel continuity across the landscape (Tausch 1999a, b). Crown cover exceeding 50% is sufficient to carry high intensity fire during dry or windy periods. Woodlands with this coverage now occupy 25% of the current range, and the area is expected to double over the next 50 years (Miller and Tausch 2001). High intensity wildfires combined with reduced understory vegetation may leave a burned area susceptible to exotic invasive species such as cheatgrass (*Bromus tectorum*). Invasion by cheatgrass has been documented to increase fire frequency on the landscape, and may shift community composition almost to monocultures (Young and Evans 1973). Increased fire severity, fire frequency, and conversion to invasive monocultures can further alter biogeochemical cycling and nutrient distribution in the Great Basin.

Fire characteristics and fuel properties determine the effect of fire on soils. Fire intensity is a measure of the amount of energy released by combustion over a given time interval (Neary et al. 1999). Fire severity is a measure of the impact fire has on soils and vegetation, and severity is a function of burn duration, fuel loading, degree of oxidation, vegetation type, weather, topography, soil texture and moisture, soil organic matter content, time since last burn, the area burned, and burn intensity (Neary et al. 1999). Tree dominance, burn intensity, and burn severity may be positively correlated during wildfire in pinyon-juniper woodlands because most wildfires occur in late summer when relative humidity (RH) and fuel and soil moisture are low. Above ground biomass and surface fuel loads increase with increasing pinyon and juniper dominance of sagebrush semi-desert (Reiner 2004; R. Tausch unpublished data 2004 USDA RMRS Range Scientist Reno, NV). Thus, conversion of sagebrush semi-desert to pinyon-juniper woodlands

increases landscape fuel loads (Reiner 2004). During dry conditions increased fuel loads may prolong soil heating and increase fire severity (Neary et al. 1999). Because of the high specific heat of water (40.656 kJ mol^{-1}), it is more difficult to raise soil temperature during times when moisture content is high (DeBano 1976).

High severity fire has variable effects on soil in pinyon-juniper woodlands. Nitrogen, P, C, and S in above ground biomass and litter can be volatilized during combustion. The magnitude of nutrient loss is dependent on fuel loads and efficiency of combustion (Covington and DeBano 1988). Potassium, Ca, and Mg are often conserved on the site, and are deposited as oxides in ash (Blank et al.1996). During extremely high severity events ash can is removed via convection (Neary et al. 1999). Similar trends have been observed for soil nutrients following fire, but losses are more dependent on soil heating than combustion. Nutrients become volatile at different temperatures and start with N at 200° , K > 760°, P > 774°, S > 800°, Na > 880°, Mg > 1107°, and Ca > 1240°C (Weast 1988). The high temperatures necessary to volatilize soil nutrients other than N are rare and found only for short durations during wildfire or during slash burning (Neary et al. 1999; Gifford 1981).

Prescribed fire has been suggested as a management tool to decrease the rate of pinyon-juniper expansion and reduce the risk of high severity wildfire. Effective use of prescribed fire requires increasing our understanding of the effects of tree expansion and prescribed fire on nutrient cycling of pinyon-juniper woodlands and their associated sagebrush semi-desert ecosystems in the Great Basin. Because most prescribed burning in the Great Basin is done during spring, RH, fuel, and soil moisture contents are relatively high, and fire intensity and severity are low to moderate. Lower severity fire and soil

temperatures associated with prescribed burning less than 400°C consumes less above ground biomass, and often produce substantial increases in available nutrients (Blank et al. 1994, 1996; Covington and DeBano 1988; Klopatek et al. 1991; DeBano and Klopatek 1988). Simulated burning studies show that maximum solute concentrations occur after soil is heated to 350°C at 5 to 15 minutes of heating (Blank et al. 1996). Decreases from the maximum solute concentration occur after soil is heated to greater than 450°C. Increases in available nutrients occur due to deposition of ash onto the soil surface, release of ortho-P and NH_4^+ from organic matter, decomposition of below-ground biomass, and further oxidation of NH_4^+ to NO_2^- and then NO_3^- by bacteria (Blank and Zamudio 1998; DeBano and Klopatek 1988; Covington et al. 1991; Hobbs and Schimel 1984).

Pinyon-juniper woodlands and their associated sagebrush semi-desert ecosystems are spatially and temporally heterogeneous in regards to soil nutrients. Concentrations of N, P and K are highest under shrubs and trees where they accumulate due to litter fall (Chambers 2001; Covington and DeBano 1988). As tree cover changes it is likely that the distribution of soil nutrients in these systems will also change as will the response to burning. In this study we document the soil nutrient response to prescribed burning, and the spatial and temporal variability of soil in a central Nevada pinyon-juniper woodland. Our research questions are: 1) Do soil available nutrients in pinyon-juniper woodlands vary spatially in regard to location, microsite (under tree, under shrub, interspace), or depth (0-8, 8-23, 23-38, 38-52 cm)? 2) Do soil available nutrients in pinyon-juniper woodlands vary temporally? 3) Are soil available nutrients in pinyon-juniper woodlands affected by prescribed burning?

Materials and Methods

Study area

The study area is in the Shoshone Mountain Range on the Humboldt-Toiyabe National Forest (Austin Ranger District) in Nye and Lander Counties, Nevada. Underdown Canyon (39°15'11" N 117°35'83" W) is oriented east to west and contains infrequent springs and an ephemeral stream near the top of the drainage. Average annual precipitation ranges from 23 cm at the bottom to 50 cm at the top of the drainage and arrives mostly as winter snow and spring rains. Average annual temperature recorded in Austin, NV ranges from –7.2°C in January to 29.4°C in July. Lithology of the Shoshone range consists of welded and non-welded silica ash flow tuff. Soils developed on alluvial fans in this study are classified as Coarse loamy mixed frigid Typic Haploxerolls.

The vegetation is characterized by sagebrush (*Artemisia tridentata vasayana*) and single leaf pinyon (*Pinus monophylla*) with lesser cover of Utah juniper (*Juniperus osteosperma*). Herbaceous species include the grasses, *Poa secunda secunda*, *Elymus elymoides*, *Stipa comata*, *Festuca idahoensis*, and *Pseudoroegneria spicata,* and the forbs, *Eriogonum* species, *Crepis acuminata*, *Phlox longifolia*, *Agoseris glauca*, *Lupinus argenteus*, and *Penstemon* species. *Bromus tectorum*, an invasive annual grass, is not a large component of the study area. The vegetation occurs in patches of variable tree dominance and is classified based on low (12% cover, 2152 kg/ ha), intermediate (38% cover, 6722 kg/ ha), and high tree dominance (74% cover, 14213 kg/ ha) (Reiner 2004).

Study design and data collection

The study was a split plot design with repeated measures. The study sites were located on northeast facing alluvial fans at elevations of 2195 m and 2225 m. The site at elevation 2195 m was a control, and the site at 2225 m received a spring burn treatment. Four replicate plots were sampled on both the control and treatment sites. Plots were characterized by intermediate tree cover at both elevations (n = 2 x 4 = 8) and contained a mix of trees, shrubs, and interspaces. To characterize the 2195 m control and 2225 m burn treatment sites, soil pits were dug to a depth of 100 cm, and the soil horizons were identified. Depth increments for sampling were assigned to the approximate center of the soil A_1 horizon and subsequent 15 cm increments (0-8, 8-23, 23-38, and 38-52 cm). Soil samples were taken from each of three microsites (under tree, under shrub, interspace) for each depth of using a 10 cm diameter bucket auger (n = 2 x 4 x 3 x 4 = 96). Sampling was conducted in November 2001 through 2003 to determine temporal, spatial, and treatment differences in soil available nutrients (n = 2 x 4 x 3 x 4 x 3 = 288). Soil samples also were collected at soil depths 0-3 and 3-8 cm using a hand trowel to determine the immediate effects of burning and the spatial variability of soil available nutrients. Soil was collected on the burn treatment site from each microsite on May 11, 2002 immediately before the burn (n = 4 x 3 x 2 x 2= 48). Collection sites were marked with a metal stake so that they could be located and sampled after the prescribed burn. Samples were again collected on May 15, 2002 after the prescribed fire. USDA Forest Service fire personnel burned the study plots on May 11-14, 2002 under favorable weather conditions. Soil temperatures were recorded during the fire using heat sensitive paints on metal strips (Korfmacher 2002). Strips were placed at 0, 2, and 5 cm soil depths at all microsites.

All soil was brought back to the lab, air-dried, and sieved to less than 2 mm. Sub-samples were analyzed for DTPA-extractable micronutrients, KCl-extractable NH_4^+ and NO_3^-, $NaHCO_3$-extractable P, and NH_4OAc-extractable metals (Lindsay and Norwell 1978; Keeney and Nelson 1982; Olsen and Sommers 1982; Thomas 1982). Soil pH was measured in $CaCl_2$ (0.1 M) using a gas electrode (Schofield and Taylor 1955). Extractable metals were determined using atomic absorption and atomic emission spectrophotometry. Extractable NH_4^+ and NO_3^- were determined using flow injection, and extractable P was determined using molybedenate-blue chemistry.

To evaluate year by treatment differences on a landscape level bulk density samples were collected from each microsite and depth using a 93 cm^3 soil core, and percent cover by microsite was measured using three line transects on each replicate plot (Elzinga et al. 1998). Data was then transformed into Kg ha^{-1} by using the formula

$$\text{Kg ha}^{-1} = (d)(Db)[1 - (>2\text{ mm}\%)](Conc)(F)$$

Where d = depth (cm) of the soil horizon, Db = bulk density (g cm^{-3}) of that horizon, >2 mm% is the volume percentage coarse fragment of that horizon, Conc = nutrient concentration (ug g^{-1}), and F = conversion factor (0.1cm^2 ug^{-1}). Kg ha^{-1} was summed by depth and microsite. The mass for each microsite was then weighted by the microsites' cover percentage on intermediate tree dominance plots, and the three microsites were then summed to give the total kg ha^{-1} on each site.

Statistical analyses

The Kolmogorov-Smirnov test was used to test for data normality. Although some of the data was normally distributed, the results were highly variable and dependent on time and treatment. Therefore, all data was natural log transformed to meet the assumption

that the data was normally distributed. All comparisons were evaluated using SAS™ mixed effects models. Initial differences in available soil nutrients between control and treatment sites, microsites, and depths were evaluated by treating location as a main effect, microsite as a split-plot within location, and depth was a split-split-plot within location and microsite. Temporal differences on the control and treatment sites were individually evaluated. Year was treated as a main effect, microsite was a split-plot within year, and depth as a split-split-plot within microsite. Immediate prescribed burn effects on the treatment site were evaluated with treatment as a main effect, microsite was a split-plot within treatment, and depth was a split-split-plot within microsite. Year by site interactions were measured by treating year as a main effect location was a split plot within year and microsite as a split-split-plot within location. All results were Tukey-Kramer adjusted and considered significant at the 95% confidence level.

Results and Discussion

Results of the ANOVA comparing initial site, microsite, and depth differences showed that all soil nutrients and pH were affected by spatial variation as shown by significant site, microsite, and depth interactions (Table 1). Temporal comparisons on the control site (Table 2) indicated that spatial variation remains similar over time, but that nutrients were subject to temporal changes. Immediate burn effects on the treatment site (Table 3) indicated that nutrients and pH are affected by burning but at different spatial scales. Temporal comparisons on the treatment site (Table 2) indicated that nutrients and pH were affected by interactions among the treatment and spatial and temporal components of the system (Table 4).

Spatial variation

Our experimental sites were highly variable even though the two sites were only separated by 500 m laterally and 30 m vertically. Soil extractable K^+, Ca^{2+}, and Na^+ were higher on the treatment site while soil extractable ortho-P, Mg^{2+}, and Zn^{2+} were higher on the control site (Table 1, Figure 1-6). Soil extractable NH_4^+, NO_3^-, Mn^{2+}, $Fe^{2+/3+}$, and pH were similar on both the control and treatment sites.

The percent cover by microsite was similar on both the control (45% interspace, 33% under shrub, 22% under tree) and treatment (46% interspace, 34% under shrub, 20% under tree) sites, making landscape scale comparisons possible. Spatial variability among microsites also was apparent at the initial comparison. NO_3^- was highest in interspace microsites, lower in under shrub microsites. The lowest levels were found under trees (Table 1; Figure 7). NH_4^+ displayed the same results on the treatment site, but values for interspace and under shrub microsites did not differ on the control site (Figure 8). Under shrub and under tree microsites typically contain more total and inorganic N than interspace sites (Chambers 2001), however these levels change seasonally in response to plant uptake and the decreasing NH_4^+ and NO_3^- availability with shrub and tree cover seen in November most likely reflects a full growing season of plant uptake, and microbial immobilization of these nutrients during organic matter decomposition (Booth et al. 2003; Compton and Boone 2002). Seasonal variability in available nutrients can be examined by comparing the May and November sampling dates. Ortho-P was lowest at interspace microsites on both control and treatment sites, and under shrub and under tree microsites had higher levels of ortho-P on the control site. Only under tree microsites displayed elevated ortho-P on the treatment site (Table 1, Figure 4). The

higher levels of ortho-P found under shrubs and trees are most likely due to release of P from litter during decomposition (Taylor et al. 1993). Mg^{2+} was significantly higher under shrubs and under trees than in interspaces, indicating that both species are capable of changing surface soils through litter fall (Table 1, Figure 5). Under tree microsites contained higher levels of Na^{+} than both under shrub and interspace microsites. Also it was observed that under shrub microsites contained higher levels of Zn^{2+} than either of the other two microsites (Table 1, Figure 6). No microsite differences were observed for K^{+}, Mn^{2+}, or $Fe^{2+/3+}$ indicating these nutrients may not be cycled preferentially by shrubs or trees.

All soil extractable nutrients were affected by soil depth with the exception of Mn^{2+} however, not all nutrients vary in the same way. Mg^{2+}, NO_3^{-}, and NH_4^{+} were highest in the surface 0-8 cm of soil (Table 1; Figure 5,7,8). No significant differences were noted below that depth, although the concentrations do generally appear to decrease with depth. These data suggest that N and Mg^{2+} are being deposited on the soil surface as litter, and that movement into the soil profile is inhibited by microbial activity, decomposition, plant uptake, or exchange mechanisms (Compton and Boone 2002; Gaston and Selim 1991). It also appears that the vegetation is not preferentially removing N, Ca^{2+}, and Mg^{2+} from a particular soil horizon. Soil ortho-P was higher at the surface 0-8 cm than at the 38-52 cm depth (Table 1; Figure 4). Concentrations of ortho-P were not different for surface soil and the 8-23 cm or 23-38 cm depths. Ortho-P declines steadily with depth so that the transition is subtle and not apparent except for the two horizons separated by the greatest distance. The nutrients Zn^{2+} and $Fe^{2+/3+}$ are highest at the surface, significantly lower at 8-23 cm, and then significantly lower on the interval from 23-52 cm (Table 1;

Figure 6,9). This indicates significant deposition and retention at the surface, as well as steady decrease with depth. Na^+ displays the opposite trend as $Fe^{2+/3+}$ and Zn^{2+} as well as most of the other nutrients. Na^+ concentrations are lowest in the surface 0-23 cm, significantly higher in the 23-38 cm horizon, and highest at the 38-52 cm depth (Table 1; Figure 3). Because Na^+ is so easily leached through soils the higher concentration at depth is probably a result of the limited precipitation in the region. The 23-52 cm depth may represent the maximum depth water penetrates before being utilized by vegetation during evapotranspiration (Salama et al. 1993). Additionally there is a B_w horizon beginning at 40cm, which may limit Na^+ migration. Soil pH was lowest at the surface but did not differ below 8 cm depth (Table 1; Figure 10). Plants releasing H^+, organic anions, CO_2 respired from roots, and higher soil organic matter most likely explains pH distribution (Hinsinger et al. 2003).

Temporal variation

Temporal variation on the control site was nutrient specific, and although there was some variation in the spatial distribution of nutrients from one year to the next, the general trends discussed above remain applicable over time (Table 2; Figure 1-10). Extractable Ca^{2+} was higher in 2003 than in 2001. Soil extractable ortho-P and Mn^{2+} was higher in 2001 and 2002 than in 2003. (Table 2; Figure 2,4,11). Soil extractable K^+ was higher in 2003 than in 2001 or 2002 and soil extractable Mg^{2+} was higher in 2003 than in 2002, and (Table 2; Figure 1,5). Extractable $Fe^{2+/3+}$ was higher in 2001 than in 2002 (Table 2; Figure 9). Natural yearly variation had no effect on the nutrients NH_4^+, NO_3^-, Na^+, Zn^{2+} and soil pH. The variation in extractable nutrients could be related to climatic variables such as precipitation, which affects soil moisture (Singh 2003). Yearly rainfall and its

distribution over plant uptake, which may affect soil extractable nutrients. Total precipitation in 2001 (27.25 cm) and 2002 (26.5 cm) was similar, but lower than precipitation in 2003 (34.47 cm). The distribution of rainfall during these years also differed. In 2001 and 2002 most precipitation occurred during winter and spring, but in 2003 there was significant additional precipitation in late summer.

Immediate effects of burning

Comparisons of immediate fire effects on soil extractable nutrients concentrated on the surface 0-3 and 3-8 cm of soil (Table 3; Figure 12). The initial differences described for depth changed little or were eliminated, because the 0-8 cm sample zone was subdivided. The nutrients NH_4^+, NO_3^-, Na^+, and Zn^{2+} no longer differed among microsites. However, pH was now highest in the under shrub microsite (Table 3; Figure 12). Soil extractable ortho-P remained higher under shrubs than in interspaces and extractable Mg^{2+} was higher under trees than interspaces (Table 3; Figure 12).

Burning resulted in significant increase in extractable NH_4^+, ortho-P, Ca^{2+}, Mg^{2+}, Mn^{2+}, Zn^{2+}, and pH (Table 3; Figure 12). The immediate increases in NH_4^+ and ortho-P were probably due to heat induced release from organic matter, and the increases in Mg^{2+}, Mn^{2+}, Zn^{2+}, and pH were most likely due to deposition of oxides and ash onto the soil surface (Covington and DeBano 1991; DeBano and Klopatek 1988; Blank and Zamudio 1998). Burning also resulted in significant decreases of extractable NO_3^- and Na^+(Table 3; Figure 12). The decrease in NO_3^- was probably due to its low volatilization temperature (Neary et al. 1999), however the decrease in extractable Na^+ is not typical of most burns. Differences were limited to the surface 0-3 cm of soil, and burning affected each microsite similarly. This most likely reflects the observed soil heating trends. Burn

temperature at the surface and 2 cm depth was highest in under shrub and under tree microsites. Interspace soils were heated only on the surface (Table 5). Soil was not heated below 2 cm. Because the burn was conducted when fuel moisture in the litter was high, litter mats under trees were not completely consumed and insulated soil from intense heating.

Extended effects of burning

Burning had extended effects on soil extractable nutrients and pH on temporal and spatial scales (Table 2; Figure 1-11), with the exceptions of K^+, Ca^{2+}, Na^+, and $Fe^{2+/3+}$. Following the burn in 2002, extractable NH_4^+ increased in the surface 0-8 cm of soil, but only on under shrub and under tree microsites. By 2003, extractable NH_4^+ had returned to pre-burn levels on under tree microsites, but remained elevated on under shrub microsites (Table 2; Figure 8). After a small initial loss of extractable NO_3^- immediately following the burn, NO_3^- rebounded and significantly increased over pre-burn levels on all microsites and soil depths by November 2002 (Table 2, 3; Figure 12, 7). Conversion of NH_4^+ to NO_3^- by bacteria is the most likely mechanism explaining the trends in NH_4^+ and NO_3^- (Covington and DeBano 1991). The observed increases at depth may be a result of root decomposition and chemical transport through the soil profile (Neary 1999). Elevated extractable NO_3^- persisted in 2003 at all microsites and soil depths, and may be facilitated from increased N-fixation by legumes (Hendricks and Boring 1999). Soil extractable ortho-P was elevated on under shrub and under tree microsites in November of 2002 after the burn, but only at the soil surface. In 2003 extractable ortho-P in the surface 0-8 cm of soil had returned to levels similar to pre-burn in 2001, however, significant increases were observed for all microsites and soil depths below 8 cm (Table

2; Figure 4). Concentrations of ortho-P increased with increasing depth, representing a reverse of pre-burn trends. The large increases in sub-surface ortho-P concentrations and accumulation at depth may be due to root decomposition, or it is possible that P migrated through the soil profile in the colloidal fraction (Hens and Merckx 2001). The soils on alluvial fans in Underdown Canyon are coarse grained and may allow for this type of transport mechanism. Soil pH was also found to remain elevated in 2002 and 2003 over pre-burn levels (Table 2; Figure 10). Extractable Mg^{2+} did not significantly increase until 2003, although there is an increasing trend in 2002 (Table 2; Figure 5). Increases in extractable Mg^{2+} over time were limited to the surface 0-8 cm of soil and to the under shrub and under tree microsites (Table 2; Figure 5). Burning resulted in increased extractable Zn^{2+} in 2002 and 2003. The increase was limited to the surface 0-8 cm of soil, but was found at all microsites As on the control extractable Mn^{2+} appeared to decrease through time when averaged over the entire profile. However, it was apparent that Mn^{2+} significantly increased in the surface 0-8 cm of soil over the same time frame after burning. (Table 2; Figure 5,11).

The total mass of available nutrients to 52 cm soil depth across the study area was affected for NO_3^- and ortho-P only (Table 4; Figure 13). This indicates that burning may only be capable of causing small changes in nutrient availability on a landscape scale with the exception of N and P, which are typically the limiting nutrients in most systems.

Conclusions

From the results comparing initial site, microsite, and depth differences we can draw a few general conclusions about the spatial heterogeneity of pinyon-juniper woodlands and their associated sagebrush semi-desert communities. Nutrient availability can change

over short distances due to inherent site characteristics that affect soil development such as parent material, shading, minor variation in precipitation or snow cover, and vegetation cover. It also is apparent that vegetation type influences nutrient distribution on the landscape, and although shrubs and trees display similar characteristics for nutrient cycling, changes in the percent cover of either of these components may affect landscape scale soil nutrient cycling and shift the distribution of nutrients contained in above ground biomass. Soil nutrients also vary with depth. Nutrient concentrations generally decrease with depth with the exception of Na^+, which increases with depth. The distribution of these nutrients within the soil profile is controlled by litter fall, decomposition, microbial immobilization, exchange mechanisms, and climate.

The temporal comparison of the control site indicates that soil nutrients change annually, but that the variation is nutrient specific and may be affected by climatic variables. It is possible that climatic variables influence nutrient availability through mineral weathering and decomposition, or that climate influences vegetation which alters uptake and influences nutrient availability. Trends observed on the treatment site suggest that nutrient availability changes seasonally as well. Our data may also suggest that nutrients are correlated. An increase in available Ca^{2+} was coincident with decreasing ortho-P. Increasing Ca^{2+} may provide sites for P precipitation (Stout et al. 2003). Although there are minor variations in the spatial distribution of nutrients over time the trends do not differ greatly from those described in our discussion of spatial variability.

Burning had immediate positive effects on most soil extractable nutrients and pH with the exception of decreased NO_3^- and Na^+. The initial effects of burning were limited to the surface 3 cm of soil due to high fuel and soil moisture at the time of the burn, and the

effects of burning were observed at all microsites. It is noteworthy that the micronutrients Mn^{2+} and Zn^{2+} respond to burning. This has rarely been observed or reported in previous literature, and may prove to be important for enzymatic processes during photosynthesis and DNA replication (Marschner 1999; Samarah et al. 2004).

Longer-term temporal and spatial changes associated with burning are evident in Underdown Canyon. Burning increased extractable NH_4^+, NO_3^-, ortho-P, Mg^{2+}, Zn^{2+}, and pH into the fall of 2003. In the case of NH_4^+, Mg^{2+}, Mn^{2+}, and Zn^{2+} the effects of burning were only observed near the soil surface, and the effects on Mg^{2+} were limited to under shrub and under tree microsites. NH_4^+ is easily converted to NO_3^- and Mg^{2+}, Mn^{2+}, and Zn^{2+} are probably inhibited from migration due to exchange mechanisms. Significant spatial and temporal changes occurred for NO_3^- and ortho-P, in all microsites displaying large increases throughout the soil profile after burning. The large spatial changes in NO_3^- distribution were expected due to decomposition of roots, mineralization of NH_4^+, and the high mobility of the NO_3^- anion. The persistence of elevated ortho-P and the large changes in spatial distribution have been less documented. We believe the dominant mechanism for P accumulation in the sub-surface is burning-induced release from organic matter, root decomposition, and colloid movement through coarse soil. Persistence of elevated Mn^{2+} and Zn^{2+} following the burn is also noteworthy due to its lack of documentation and possible implications for enzyme activity and drought tolerance (Marschner 1995; Samarah et al. 2004).

Pinyon-juniper woodlands and their associated sagebrush semi-dessert communities are highly variable in regards to the spatial and temporal distribution of mineral nutrients. As woodlands encroach into semi-arid shrub communities we can expect to see a

homogenization of soil nutrients across the landscape as tree crown cover becomes more continuous, and a shift in distribution of nutrients to the soil surface associated with litter fall. Increasing tree cover will also increase the proportion of nutrients stored in above ground biomass. These effects could prove to be deleterious if associated with high intensity wildfire, because losses from volatilization, convection, and erosion could be increased. The concentration of soil available nutrients in woodlands changes from one year to the next, due to climatic variability this complicates the analysis of fire effects. Prescribed burning in woodlands can cause immediate and persistent effects on soil available nutrients and pH. Because fuel and soil moisture is high during spring prescribed burning, soil heating only occurs to shallow depths. The results are positive changes in soil available nutrients. Burning effects can be limited to surface soils and particular microsites, or they can occur over the entire burn area and be observed at soil depths of 52 cm. From a soil nutrient standpoint prescribed burning in pinyon-juniper woodlands can be a successful treatment for restoring sagebrush semi-desert ecosystems. It is important however to consider the state of existing understory vegetation before burning, due to its influence on soil stability and site recovery post-burn (Reiner 2004).

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Figure 1. Mean extractable soil K^+ concentrations (mmol kg^{-1}) and standard errors in November 2001, 2002, and 2003 for interspace (IS), under shrub (US), and under tree (UT) microsites and 0-8, 8-23, 23-38, and 38-52 cm soil depths at the control (2195 m) and treatment (2225 m) sites. The dotted line indicates the burn treatment.

Figure 2. Mean extractable soil Ca^{2+} concentrations (mmol kg^{-1}) and standard errors in November 2001, 2002, and 2003 for interspace (IS), under shrub (US), and under tree (UT) microsites and 0-8, 8-23, 23-38, and 38-52 cm soil depths at the control (2195 m) and treatment (2225 m) sites. The dotted line indicates the burn treatment.

Figure 3. Mean extractable soil Na^+ concentrations (mmol kg^{-1}) and standard errors in November 2001, 2002, and 2003 for interspace (IS), under shrub (US), and under tree (UT) microsites and 0-8, 8-23, 23-38, and 38-52 cm soil depths at the control (2195 m) and treatment (2225 m) sites. The dotted line indicates the burn treatment.

Figure 4. Mean extractable soil ortho-P concentrations (mmol kg^{-1}) and standard errors in November 2001, 2002, and 2003 for interspace (IS), under shrub (US), and under tree (UT) microsites and 0-8, 8-23, 23-38, and 38-52 cm soil depths at the control (2195 m) and treatment (2225 m) sites. The dotted line indicates the burn treatment.

Figure 5. Mean extractable soil Mg^{2+} concentrations (mmol kg^{-1}) and standard errors in November 2001, 2002, and 2003 for interspace (IS), under shrub (US), and under tree

(UT) microsites and 0-8, 8-23, 23-38, and 38-52 cm soil depths at the control (2195 m) and treatment (2225 m) sites. The dotted line indicates the burn treatment.

Figure 6. Mean extractable soil Zn^{2+} concentrations (mmol kg^{-1}) and standard errors in November 2001, 2002, and 2003 for interspace (IS), under shrub (US), and under tree (UT) microsites and 0-8, 8-23, 23-38, and 38-52 cm soil depths at the control (2195 m) and treatment (2225 m) sites. The dotted line indicates the burn treatment.

Figure 7. Mean extractable soil NO_3^- concentrations (mmol kg^{-1}) and standard errors in November 2001, 2002, and 2003 for interspace (IS), under shrub (US), and under tree (UT) microsites and 0-8, 8-23, 23-38, and 38-52 cm soil depths at the control (2195 m) and treatment (2225 m) sites. The dotted line indicates the burn treatment.

Figure 8. Mean extractable soil NH_4^+ concentrations (mmol kg^{-1}) and standard errors in November 2001, 2002, and 2003 for interspace (IS), under shrub (US), and under tree (UT) microsites and 0-8, 8-23, 23-38, and 38-52 cm soil depths at the control (2195 m) and treatment (2225 m) sites. The dotted line indicates the burn treatment.

Figure 9. Mean extractable soil $Fe^{2+/3+}$ concentrations (mmol kg^{-1}) and standard errors in November 2001, 2002, and 2003 for interspace (IS), under shrub (US), and under tree (UT) microsites and 0-8, 8-23, 23-38, and 38-52 cm soil depths at the control (2195 m) and treatment (2225 m) sites. The dotted line indicates the burn treatment.

Figure 10. Mean extractable soil pH and standard errors in November 2001, 2002, and 2003 for interspace (IS), under shrub (US), and under tree (UT) microsites and 0-8, 8-23, 23-38, and 38-52 cm soil depths at the control (2195 m) and treatment (2225 m) sites. The dotted line indicates the burn treatment.

Figure 11. Mean extractable soil Mn^{2+} concentrations (mmol kg^{-1}) and standard errors in November 2001, 2002, and 2003 for interspace (IS), under shrub (US), and under tree (UT) microsites and 0-8, 8-23, 23-38, and 38-52 cm soil depths at the control (2195 m) and treatment (2225 m) sites. The dotted line indicates the burn treatment.

Figure 13. Mean extractable soil nutrient concentrations (mmol kg^{-1}) and standard errors pre and post-burn in May 2002 for interspace (IS), under shrub (US), and under tree (UT) microsites and 0-3 and 3-8 cm soil depths at the treatment (2225 m) site. The dotted line indicates the burn treatment.

Figure 14. Mean extractable soil nutrient content (Kg ha^{-1}) in November 2001, 2002, and 2003 summed by microsite and depth for the control and treatment sites.

Table 1. Results of the ANOVA for site, microsite, and depth differences.

		NH_4^+		NO_3^-		ortho-P		K^+		Ca^{2+}		Mg^{2+}	
Effect	**DF**	**F**	**P**	**F**	**P**	**F**	**P**	**F**	**P**	**F**	**P**	**F**	**P**
Site	1	2.1	0.1977	3.62	0.1057	10.27	0.0185	33.49	0.0012	8.46	0.027	40.29	0.0007
replicate(site)	6												
Microsite	2	3.92	0.049	41.2	<.0001	4.05	0.0453	1	0.3966	2.42	0.1306	22.25	<.0001
site*microsite	2	0.18	0.8353	0.16	0.8538	3.01	0.0873	0.01	0.9928	1.06	0.3763	2.99	0.0881
microsite*replicate(site)	12												
Depth	3	97.6	<.0001	129.4	<.0001	4.87	0.0045	23.43	<.0001	7.15	0.0004	48.56	<.0001
site*depth	3	0.77	0.5178	2.66	0.0575	0.78	0.5096	6.05	0.0013	9.01	<.0001	12.63	<.0001
microsite*depth	6	1.73	0.1316	33.68	<.0001	1.14	0.3534	2.21	0.056	2.99	0.0139	11.09	<.0001
site*microsite*depth	6	0.93	0.4799	0.54	0.7737	0.74	0.6195	1.2	0.3185	0.75	0.6094	2.75	0.021
depth*microsite*replicate(site)	54												

		Na^+		Mn^{2+}		$Fe^{2+/3+}$		Zn^{2+}		pH	
Effect	**DF**	**F**	**P**	**F**	**P**	**F**	**P**	**F**	**P**	**F**	**P**
Site	1	11.4	0.0149	2.28	0.1818	4.94	0.0679	19.21	0.0047	0.73	0.425
replicate(site)	6										
Microsite	2	7.84	0.0066	1.01	0.3938	2.41	0.1316	9.07	0.004	0.6	0.5624
site*microsite	2	3.03	0.0859	0.9	0.432	0.46	0.6418	4.45	0.0358	0.06	0.9442
microsite*replicate(site)	12										
Depth	3	49.7	<.0001	2.07	0.1144	83.51	<.0001	100.2	<.0001	14.2	<.0001
site*depth	3	26.6	<.0001	0.88	0.4551	4.65	0.0058	13.19	<.0001	1.38	0.259
microsite*depth	6	1.23	0.308	2.3	0.0474	3.54	0.005	5.69	0.0001	1.32	0.2641
site*microsite*depth	6	0.98	0.4464	0.27	0.9466	1.83	0.1098	5.25	0.0003	1.79	0.1186
depth*microsite*replicate(site)	54										

Table 2. Results of the ANOVA for microsite, depth, and year differences at sites 2195 m and 2225 m.

		NH_4^+				NO_3^-				ortho-P				K^+			
		2195 m		2225 m		2195 m		2225 m		2195 m		2225 m		2195 m		2225 m	
Effect	**DF**	**F**	**P**	**F**	**P**	**F**	**P**	**F**	**P**	**F**	**P**	**F**	**P**	**F**	**P**	**F**	**P**
Microsite	3	4.26	0.05	9.32	0.004	2.75	0.1172	2.68	0.1104	8.1	0.0096	4.02	0.0454	0.7	0.5287	1.3	0.3314
replicate*microsite	9																
Depth	3	8.6	0.0004	79.7	<.0001	3.29	0.0359	22.2	<.0001	9.3	0.0002	9.28	0.0002	15	<.0001	29	<.0001
microsite*depth	6	1.51	0.2114	6.69	0.0002	1.06	0.4093	0.6	0.7283	1.1	0.3825	12.9	<.0001	2.7	0.0344	4.6	0.0025
replicate*microsite*depth	27																
Year	2	1	0.3733	7.94	0.0008	0.59	0.5576	48.5	<.0001	9.6	0.0002	102	<.0001	9.8	0.0002	0.4	0.6976
microsite*year	4	2.35	0.0655	1.49	0.2166	1.18	0.3276	1.63	0.1789	2.5	0.0539	5.4	0.0008	1.7	0.1607	0.2	0.912
depth*year	6	3.8	0.003	3.21	0.0082	0.71	0.6442	3.99	0.002	7	<.0001	39.8	<.0001	0.7	0.6519	0.1	0.9991
microsite*depth*year	12	1.7	0.0905	1.42	0.1819	0.92	0.5339	1.48	0.1565	0.8	0.6701	3.2	0.0011	0.7	0.7034	0.2	0.9986
year*replicate*microsite*depth	68																

		Ca^{2+}				Mg^{2+}				Na^+				Mn^{2+}			
		2195 m		2225 m		2195 m		2225 m		2195 m		2225 m		2195 m		2225 m	
Effect	**DF**	**F**	**P**	**F**	**P**	**F**	**P**	**F**	**P**	**F**	**P**	**F**	**P**	**F**	**P**	**F**	**P**
Microsite	2	2.12	0.1755	0.43	0.6637	15.9	0.0011	12.7	0.0014	3.1	0.0928	3.32	0.0706	5.7	0.0248	1.9	0.1985
replicate*microsite	9																
Depth	3	30.302	<.0001	1.37	0.2726	49.3	<.0001	108	<.0001	36	<.0001	65	<.0001	1.2	0.3354	15	<.0001
microsite*depth	6	5.43	0.0009	0.82	0.5647	7.4	<.0001	27.9	<.0001	1.6	0.1754	3.11	0.019	0.3	0.924	5.6	0.0007
replicate*microsite*depth	27																
Year	2	6.09	0.0036	0.52	0.5965	4.13	0.0202	25	<.0001	4.3	0.017	3.97	0.0233	22	<.0001	13	<.0001
microsite*year	4	2.27	0.0703	0.95	0.4422	1.99	0.105	1.76	0.1469	1.2	0.321	0.52	0.7243	0.1	0.982	0.8	0.5262
depth*year	6	0.68	0.6688	0.24	0.9611	0.56	0.7605	5.11	0.0002	1.2	0.305	0.21	0.9714	0.7	0.6532	2	0.0733
microsite*depth*year	12	1.6	0.1126	0.50	0.9069	0.19	0.9987	0.92	0.5359	0.4	0.9741	0.59	0.8404	1	0.4416	1.7	0.0985
year*replicate*microsite*depth	63																

		$Fe^{2+/3+}$				Zn^{2+}				pH			
		2195 m		2225 m		2195 m		2225 m		2195 m		2225 m	
Effect	**DF**	**F**	**P**	**F**	**P**	**F**	**P**	**F**	**P**	**F**	**P**	**F**	**P**
Microsite	3	0.69	0.5277	0.78	0.5362	5.19	0.0317	2.6	0.1167	0.8	0.4769	6.02	0.0156
replicate*microsite	9												
Depth	3	40.9	<.0001	94.7	<.0001	103	<.0001	83.7	<.0001	4	0.0179	2.59	0.0737
microsite*depth	6	5.08	0.0013	0.97	0.4622	10.2	<.0001	3.98	0.0056	4.1	0.005	4.64	0.0023
replicate*microsite*depth	27												
Year	2	4.32	0.017	0.73	0.4864	0.57	0.568	6.02	0.0039	23	<.0001	5.55	0.0058
microsite*year	4	1	0.4142	0.82	0.5172	2.99	0.0244	0.81	0.5245	1.8	0.144	0.93	0.4546
depth*year	6	1.09	0.3784	1.22	0.3052	1.35	0.248	3.6	0.0038	0.4	0.8822	0.36	0.901
microsite*depth*year	12	1.42	0.1762	0.45	0.9357	2.76	0.004	1.2	0.2995	0.6	0.8219	0.56	0.8666
year*replicate*microsite*depth	68												

Table 3. Result of the ANOVA for depth, microsite, and immediate fire effects on soil nutrients

		NH_4^+		NO_3^-		ortho-P		K^+		Ca^{2+}		Mg^{2+}	
Effect	**DF**	**F**	**P**	**F**	**P**	**F**	**P**	**F**	**P**	**F**	**P**	**F**	**P**
Microsite	2	2.93	0.1047	0.1	0.9082	15.39	0.0012	0.33	0.7291	7.38	0.0127	8.34	0.0089
replicate*microsite	9												
Depth	1	44.4	<.0001	4.85	0.0551	18.81	0.0019	1.62	0.2347	7.57	0.0225	13.48	0.0051
microsite*depth	2	3.02	0.0994	0.3	0.7497	0.94	0.4267	0.22	0.8029	22.82	0.0003	9.05	0.007
replicate*microsite*depth	9												
Treatment	1	62	<.0001	4.87	0.0424	27.75	<.0001	1.99	0.1772	5.12	0.0413	14.39	0.0015
microsite*treatment	2	9.99	0.0015	2.53	0.1107	1.18	0.3325	0.95	0.4093	0.09	0.9164	0.42	0.6622
depth*treatment	1	12.6	0.0027	1.15	0.2988	4.53	0.0492	1.26	0.2779	7.18	0.0189	3.31	0.0863
microsite*depth*treatment	2	1.89	0.1834	0.54	0.5925	1.24	0.3156	0.09	0.9171	0.81	0.4678	0.37	0.6958
replicate*microsite*depth*treatment	16												

		Na^+		Mn^{2+}		$Fe^{2+/3+}$		Zn^{2+}		pH	
Effect	**DF**	**F**	**P**	**F**	**P**	**F**	**P**	**F**	**P**	**F**	**P**
Microsite	2	0.31	0.7416	0.8	0.4789	0.12	0.8899	0.81	0.4746	8.24	0.0092
replicate*microsite	9										
Depth	1	26.8	0.0006	97.2	<.0001	12.97	0.0057	305	<.0001	1.96	0.1949
microsite*depth	2	0.03	0.9733	8.73	0.0078	0.89	0.4444	40.8	<.0001	0.12	0.8886
replicate*microsite*depth	9										
Treatment	1	7.51	0.0145	14.3	0.0016	0.68	0.4203	11	0.0047	14.9	0.0014
microsite*treatment	2	2.74	0.0946	4.76	0.0239	2.14	0.1503	0.29	0.7551	2.09	0.1564
depth*treatment	1	0.98	0.3366	7.41	0.0151	0.37	0.5532	5.73	0.0302	3.84	0.0678
microsite*depth*treatment	2	0.55	0.5851	0.94	0.4118	0.6	0.5632	0.57	0.5748	0.65	0.5328
replicate*microsite*depth*treatment	16										

Table 4. Results of the ANOVA for year and site differences in units of kg ha^{-1}.

		NH_4^+		NO_3^-		ortho-P		Ca^{2+}	
Effect	**DF**	**F**	**P**	**F**	**P**	**F**	**P**	**F**	**P**
year	2	0.58	0.5734	12.72	0.0011	7.12	0.0091	0.39	0.6877
replicate(site)	12								
site	1	0.01	0.9221	6.29	0.0461	3.54	0.1088	2.6	0.1579
year*replicate(site)	6								
year*site	2	2.90	0.0937	5.99	0.0157	15.80	0.0004	0.18	0.8345
year*replicate*microsite(site)	12								

		Mg^{2+}		K^+		Na^{2+}	
Effect	**DF**	**F**	**P**	**F**	**P**	**F**	**P**
year	2	2.21	0.1524	0.05	0.9552	0.46	0.6407
replicate(site)	12						
site	1	1.94	0.2133	11.24	0.0154	15.32	0.0079
year*replicate(site)	6						
year*site	2	0.40	0.6787	0.22	0.8073	0.34	0.7208
year*replicate*microsite(site)	12						

		Mn^{2+}		$Fe^{2+/3+}$		Zn^{2+}	
Effect	**DF**	**F**	**P**	**F**	**P**	**F**	**P**
year	2	14.97	0.0005	0.55	0.5926	1.16	0.3455
replicate(site)	12						
site	1	1.33	0.2928	2.02	0.2053	3.21	0.1234
year*replicate(site)	6						
year*site	2	0.11	0.8936	0.44	0.6547	1.63	0.2357
year*replicate*microsite(site)	12						

Table 5. Mean soil temperatures for surface, 2, and 5 cm soil depths at all microsites.

Microsite	Avg. temperature (°C)
----------surface----------	
Interspace	206.1
Under shrub	369.4
Under tree	303.9
----------2 cm----------	
Interspace	39.5
Under shrub	86.0
Under tree	76.8
----------5 cm----------	
Interspace	39.5
Under Shrub	39.5
Under Tree	43.9

Figure 1.

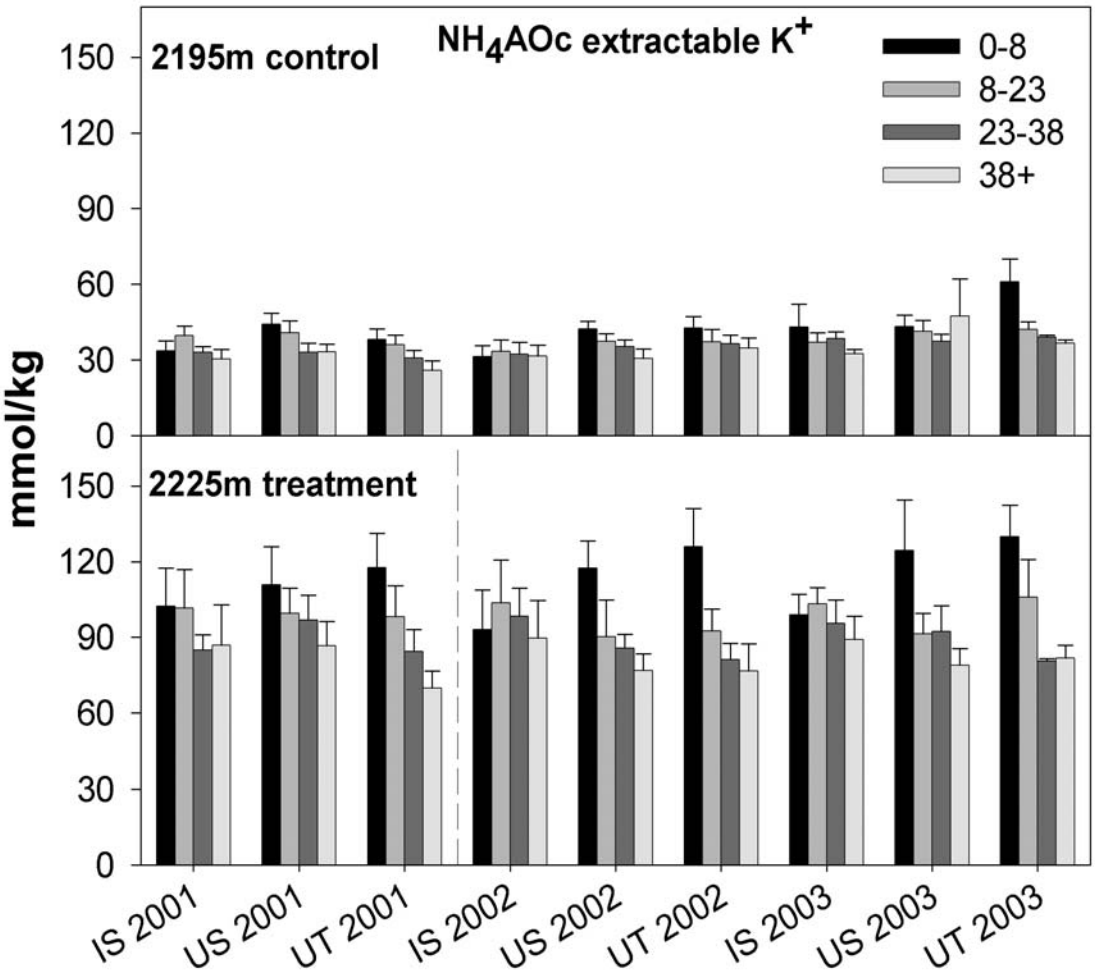

2195m control
NH4AOc extractable K+
0-8
8-23
23-38
38+
2225m treatment
mmol/kg
150
120
90
60
30
0
IS 2001
US 2001
UT 2001
IS 2002
US 2002
UT 2002
IS 2003
US 2003
UT 2003

Figure 2.

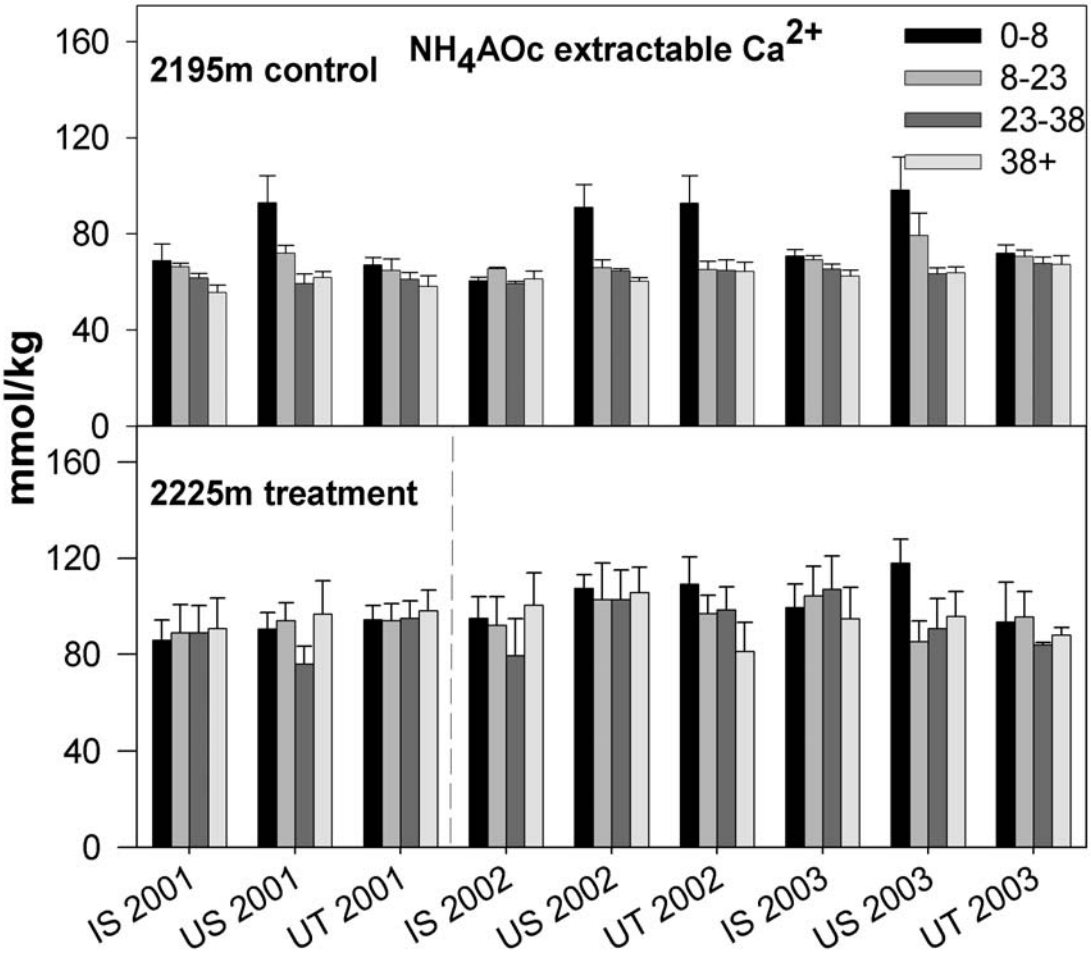

2195m control
NH4AOc extractable Ca2+
0-8
8-23
23-38
38+
mmol/kg
2225m treatment
0
40
80
120
160
IS 2001
US 2001
UT 2001
IS 2002
US 2002
UT 2002
IS 2003
US 2003
UT 2003

Figure 3.

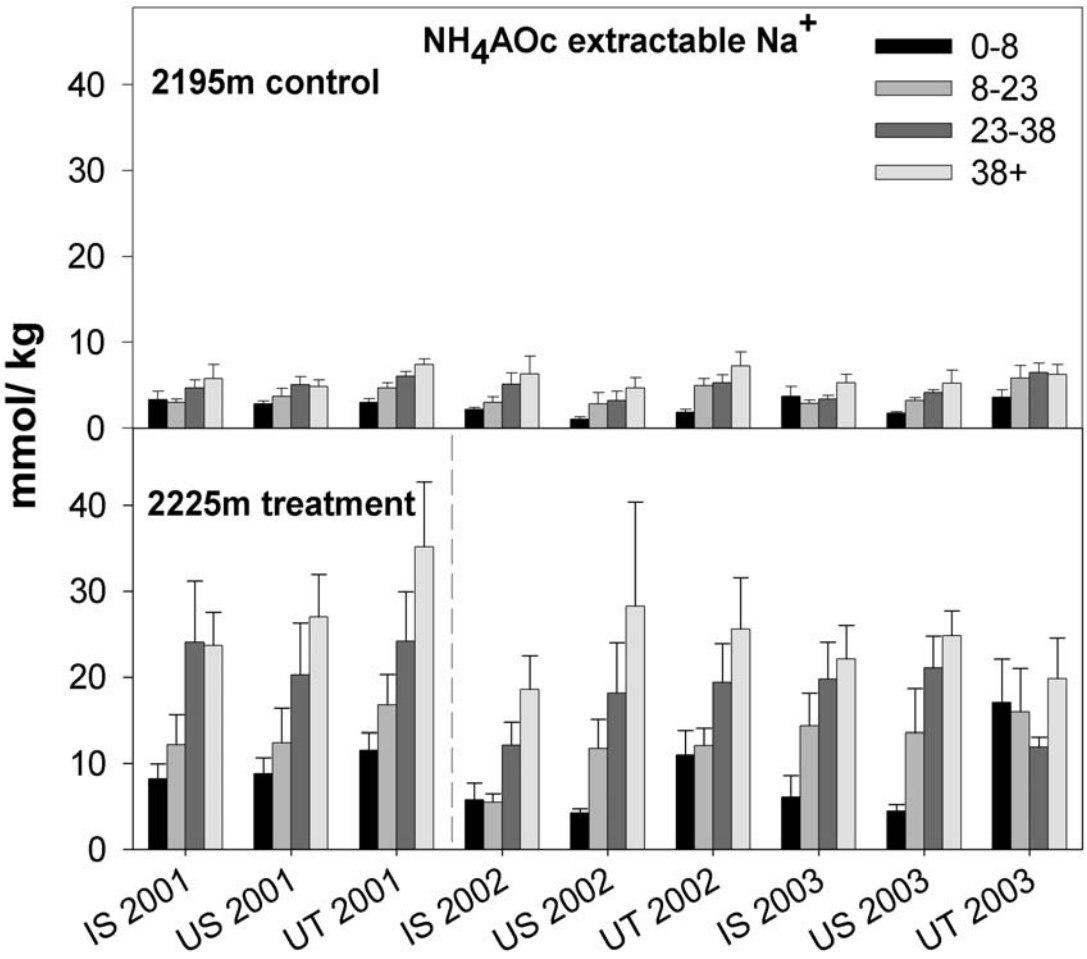

NH4AOc extractable Na+
0-8
8-23
23-38
38+
2195m control
2225m treatment
mmol/ kg
0
10
20
30
40
IS 2001
US 2001
UT 2001
IS 2002
US 2002
UT 2002
IS 2003
US 2003
UT 2003

Figure 4.

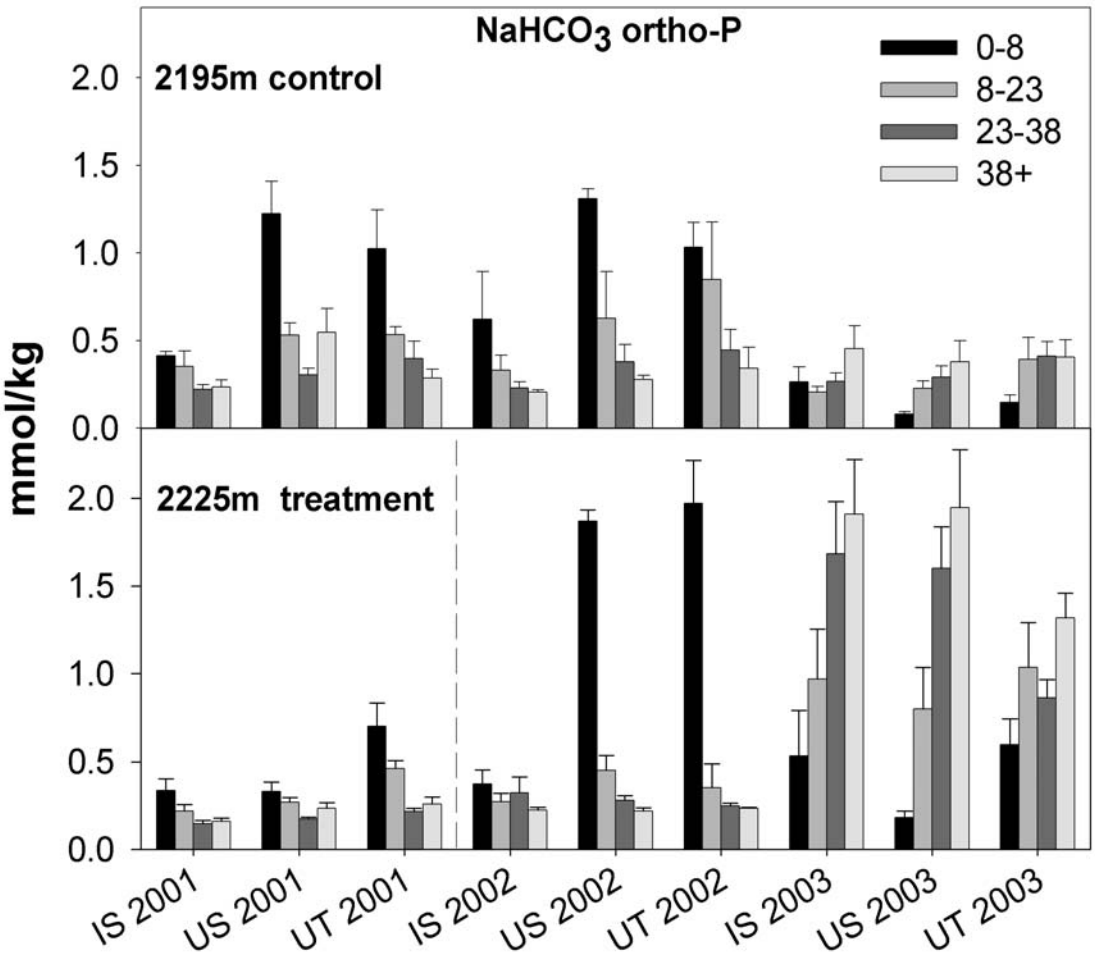

NaHCO3 ortho-P
0-8
8-23
23-38
38+
2195m control
2225m treatment
mmol/kg
2.0
1.5
1.0
0.5
0.0
IS 2001
US 2001
UT 2001
IS 2002
US 2002
UT 2002
IS 2003
US 2003
UT 2003

Figure 5.

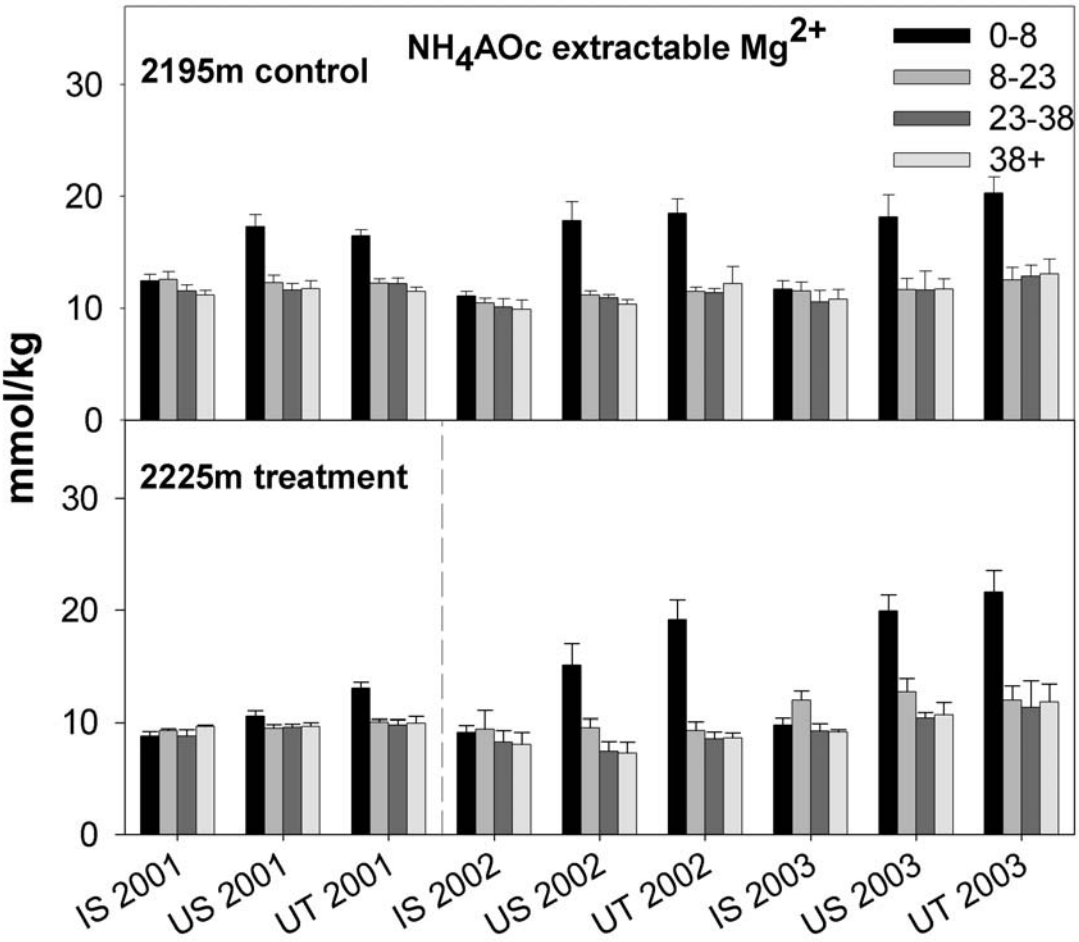

2195m control
NH4AOc extractable Mg2+
0-8
8-23
23-38
38+
2225m treatment
mmol/kg
0
10
20
30
IS 2001
US 2001
UT 2001
IS 2002
US 2002
UT 2002
IS 2003
US 2003
UT 2003

Figure 6.

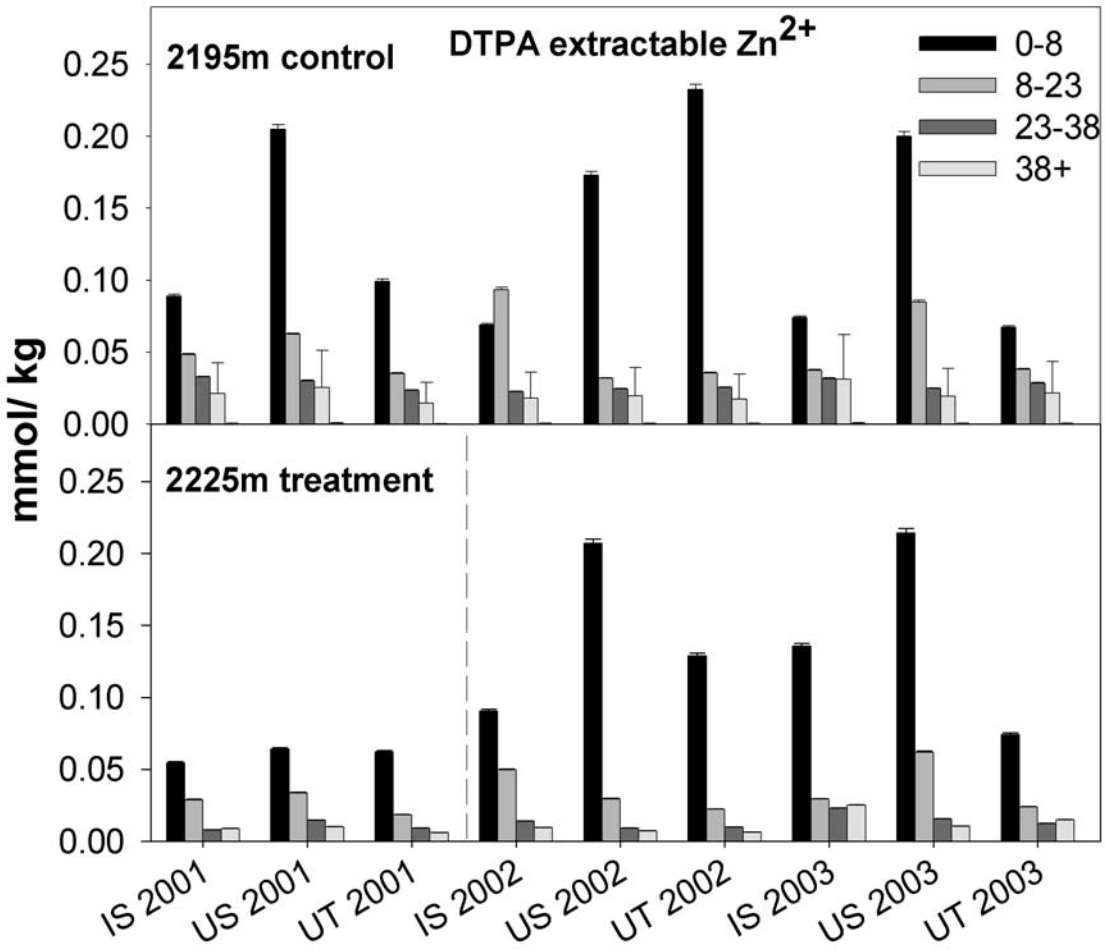

2195m control
DTPA extractable Zn2+
0-8
8-23
23-38
38+
mmol/ kg
0.25
0.20
0.15
0.10
0.05
0.00
2225m treatment
IS 2001
US 2001
UT 2001
IS 2002
US 2002
UT 2002
IS 2003
US 2003
UT 2003

Figure 7.

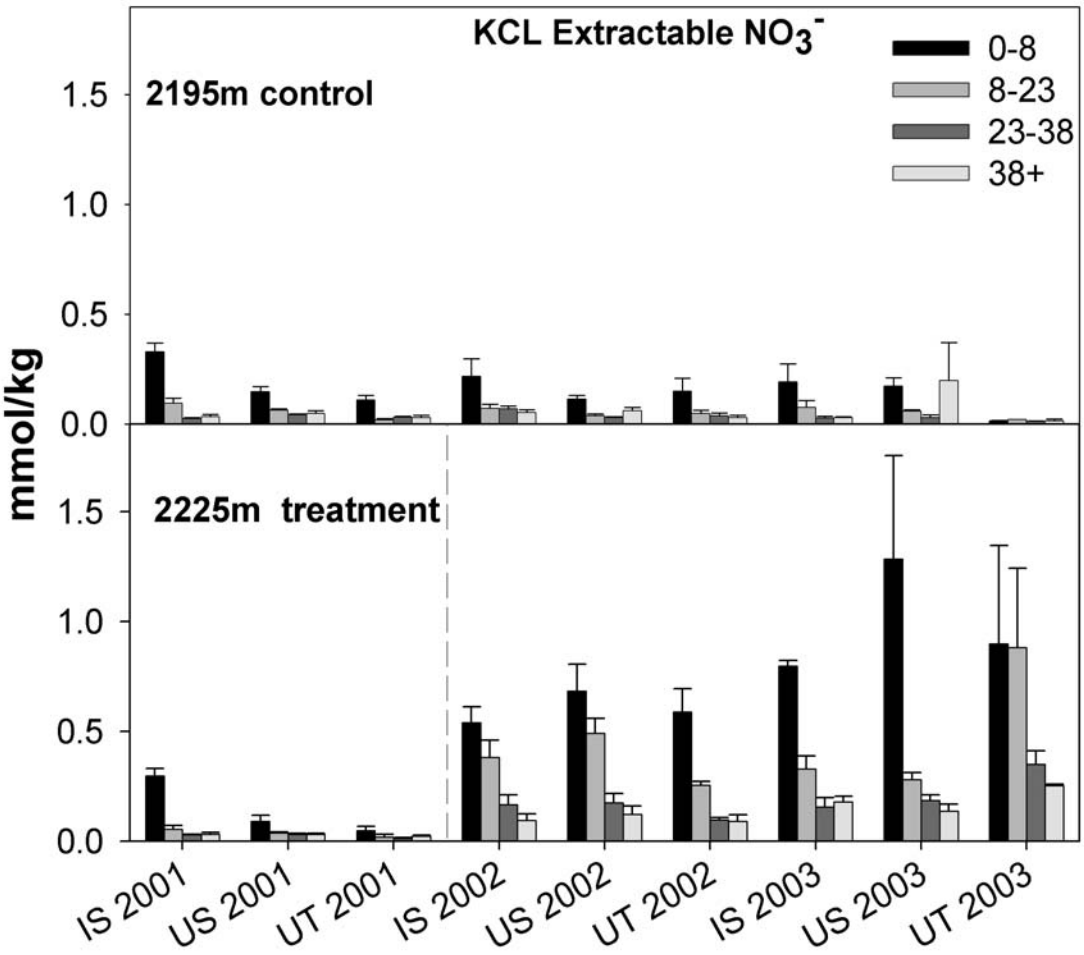

KCL Extractable NO_3^-
0-8
8-23
23-38
38+
2195m control
2225m treatment
mmol/kg
1.5
1.0
0.5
0.0
IS 2001
US 2001
UT 2001
IS 2002
US 2002
UT 2002
IS 2003
US 2003
UT 2003

Figure 8.

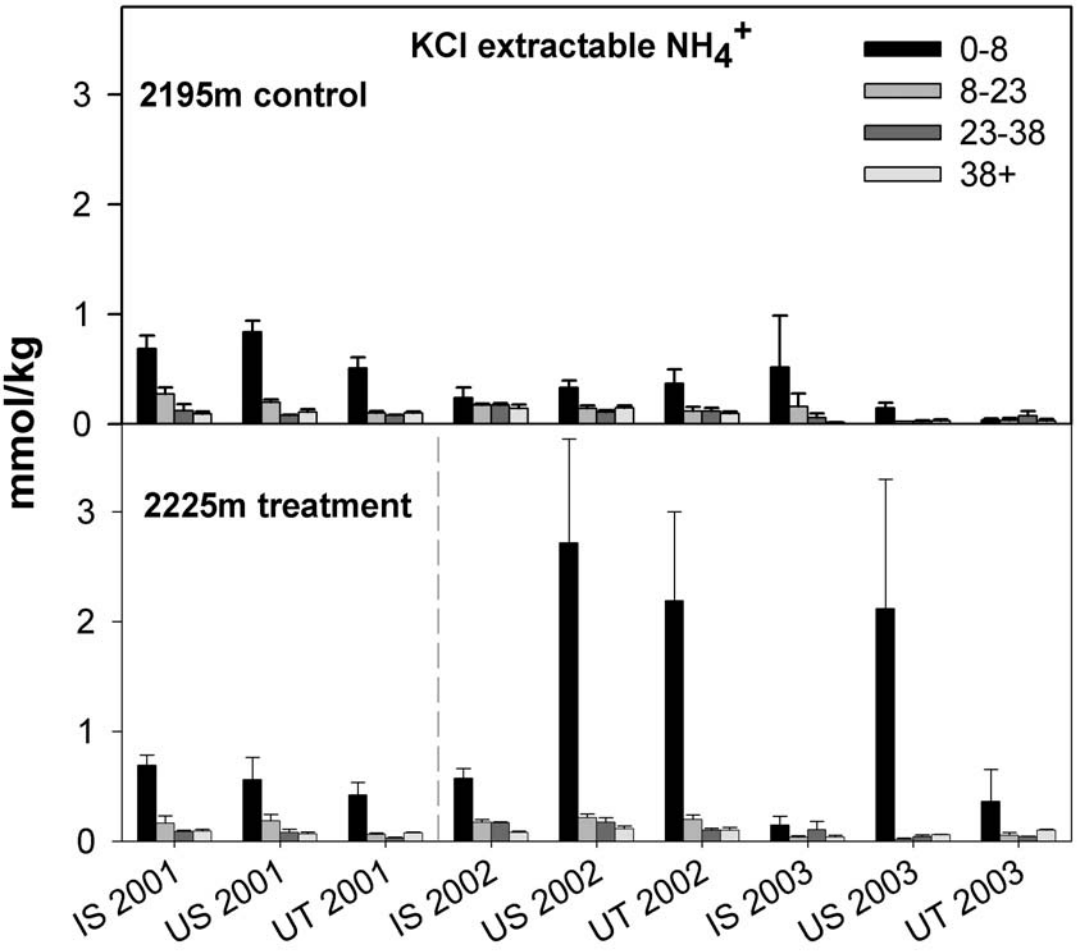

KCl extractable NH_4^+
0-8
8-23
23-38
38+
2195m control
2225m treatment
mmol/kg
0
1
2
3
IS 2001
US 2001
UT 2001
IS 2002
US 2002
UT 2002
IS 2003
US 2003
UT 2003

Figure 9.

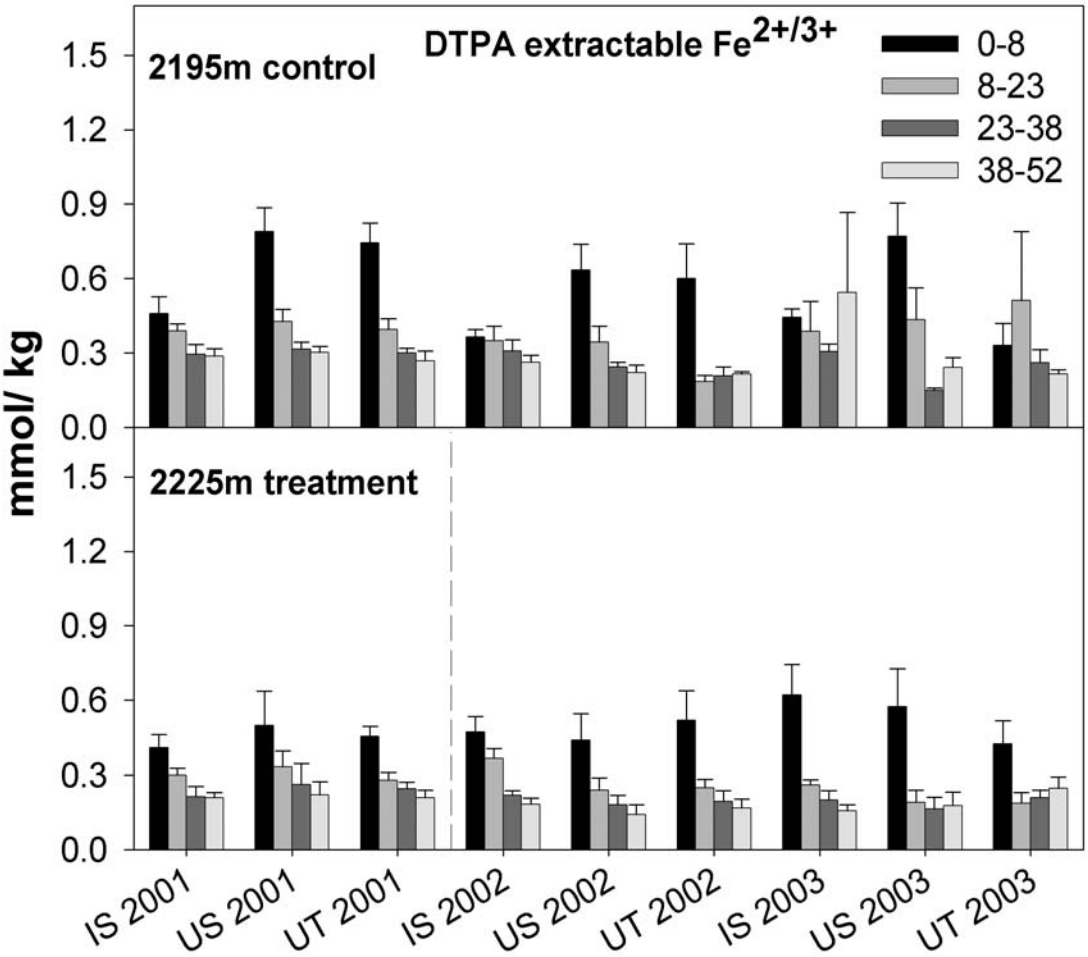

DTPA extractable Fe2+/3+
0-8
8-23
23-38
38-52
2195m control
2225m treatment
mmol/ kg
1.5
1.2
0.9
0.6
0.3
0.0
IS 2001
US 2001
UT 2001
IS 2002
US 2002
UT 2002
IS 2003
US 2003
UT 2003

Figure 10.

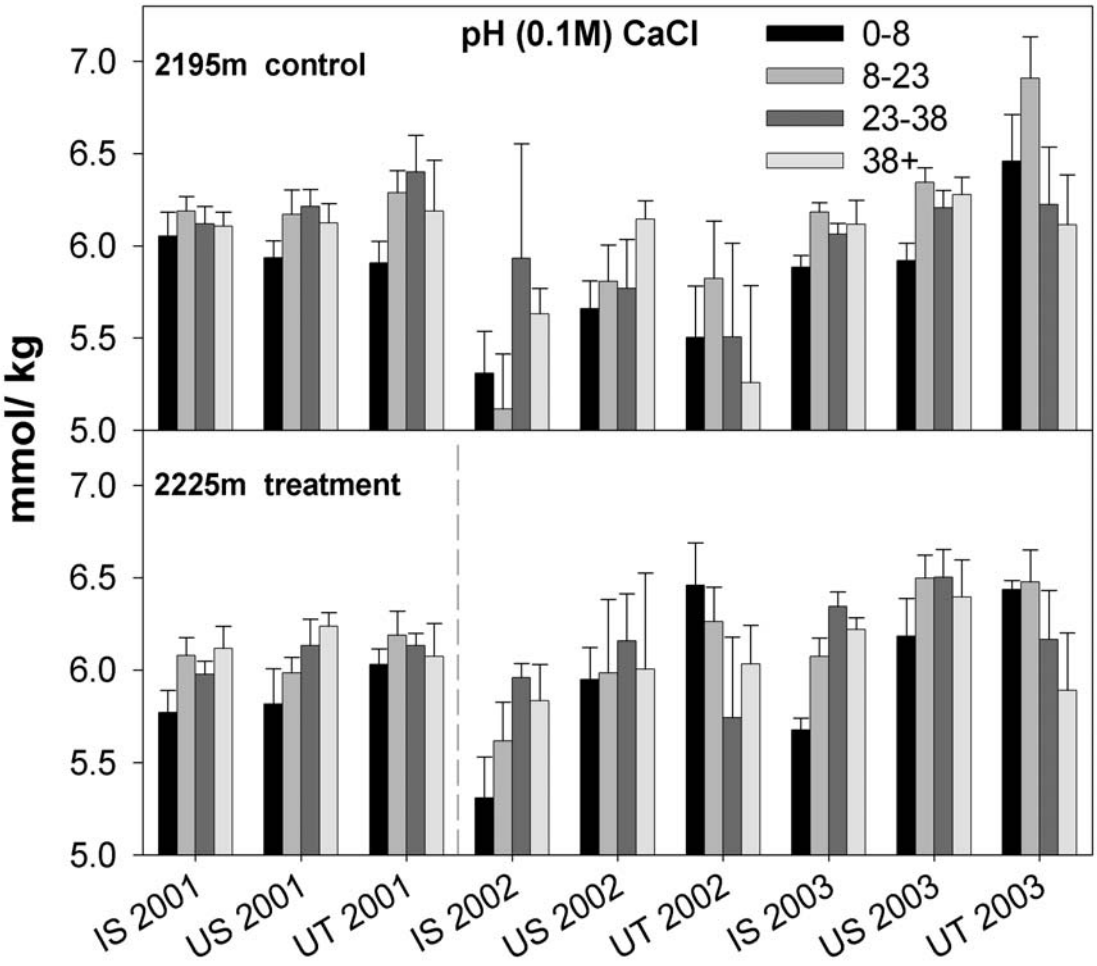

pH (0.1M) CaCl
0-8
8-23
23-38
38+
2195m control
2225m treatment
mmol/ kg
7.0
6.5
6.0
5.5
5.0
IS 2001
US 2001
UT 2001
IS 2002
US 2002
UT 2002
IS 2003
US 2003
UT 2003

Figure 11.

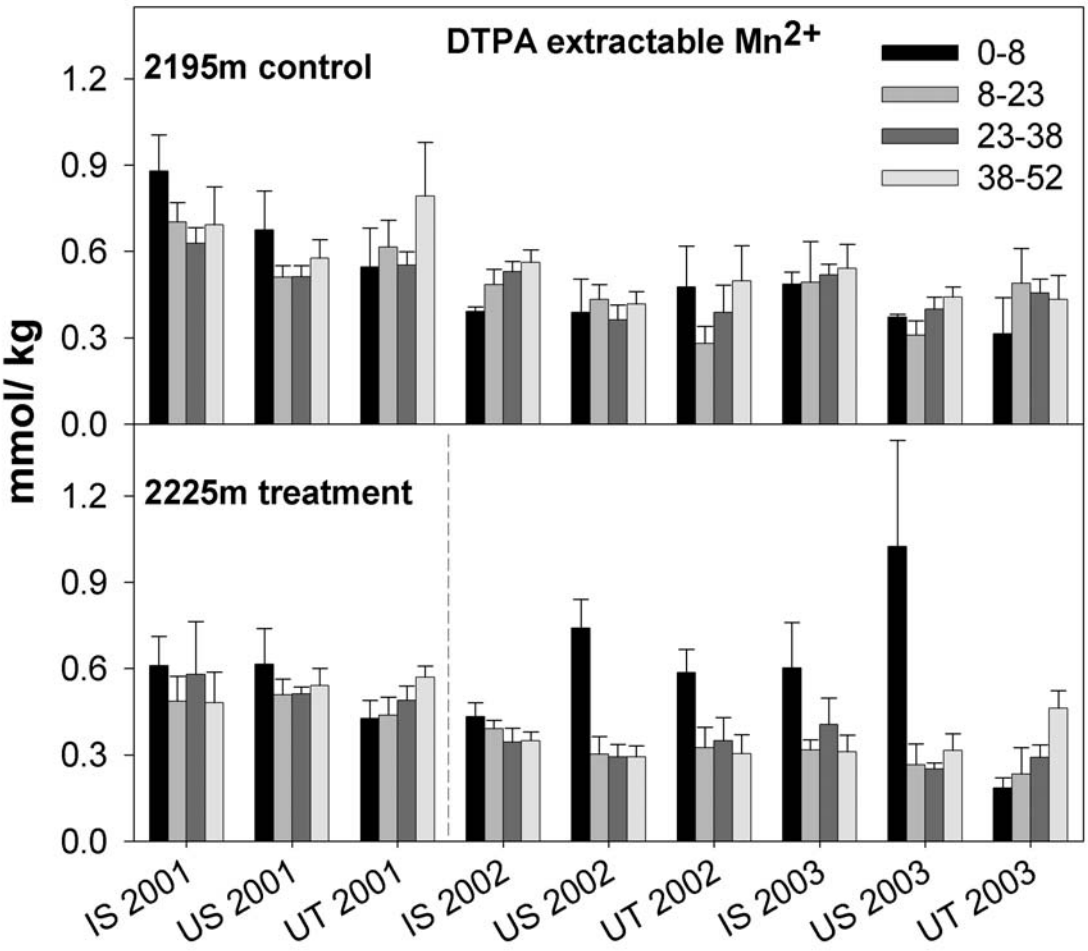

DTPA extractable Mn2+
2195m control
2225m treatment
0-8
8-23
23-38
38-52
mmol/ kg
1.2
0.9
0.6
0.3
0.0
IS 2001
US 2001
UT 2001
IS 2002
US 2002
UT 2002
IS 2003
US 2003
UT 2003

Figure 12.

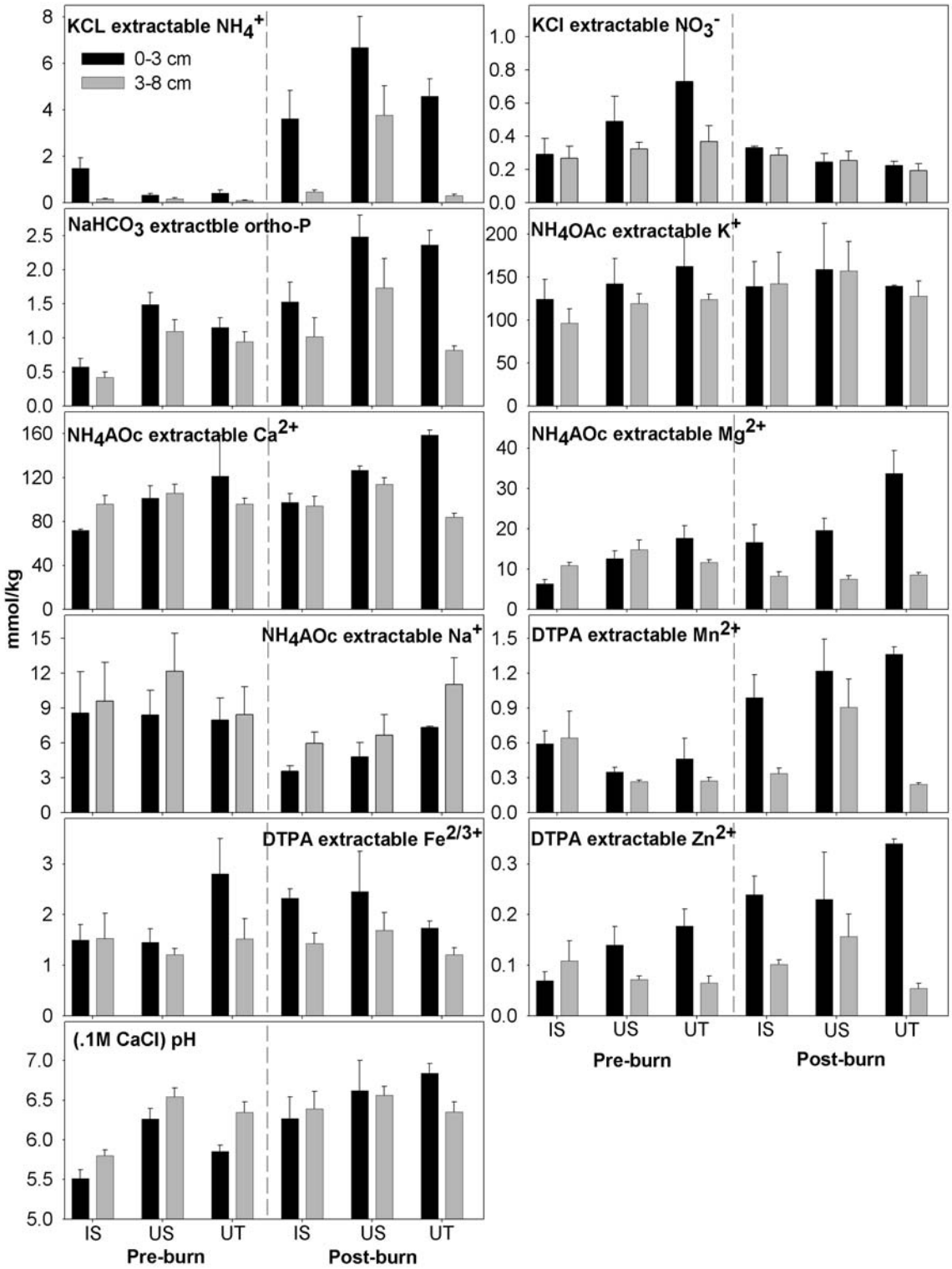
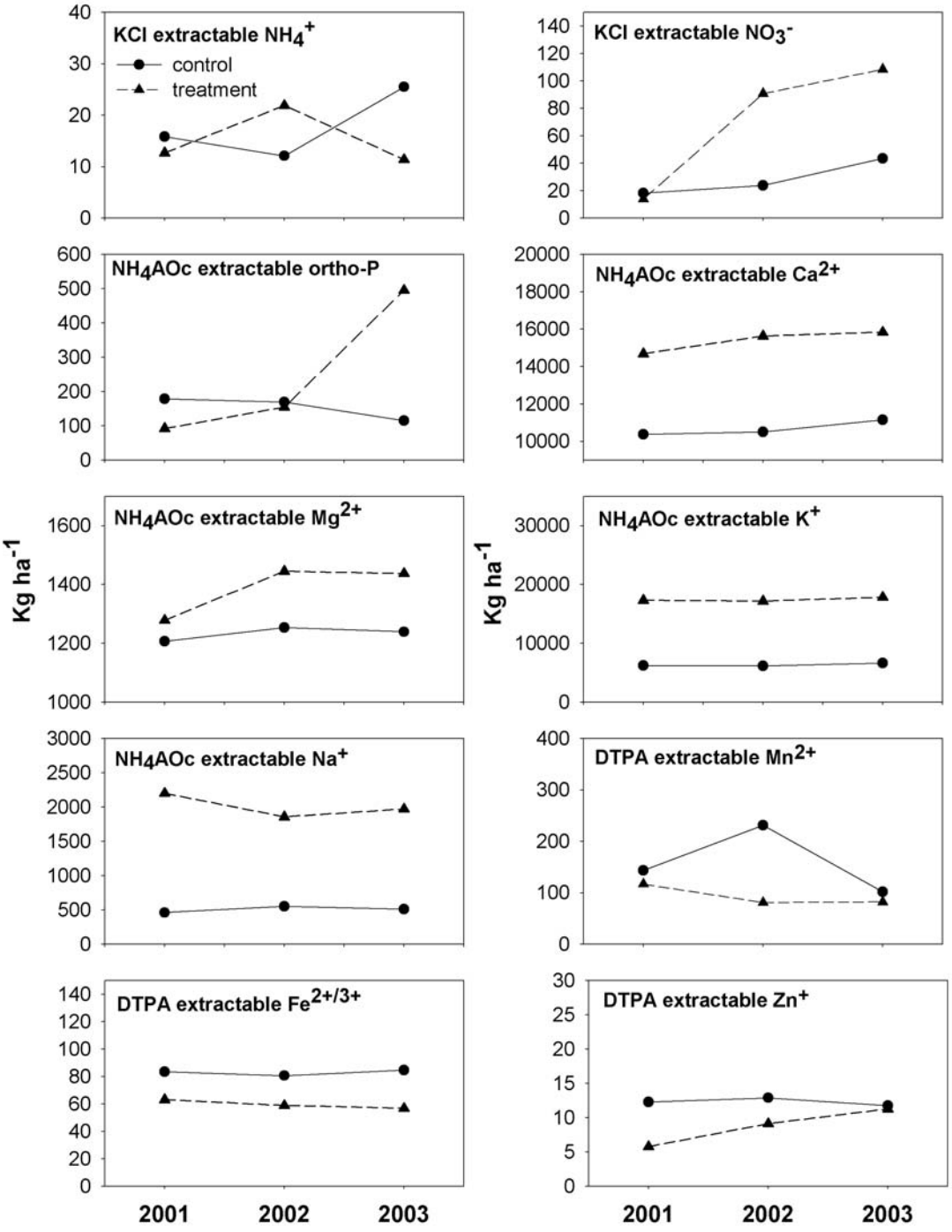

KCL extractable NH4+
0-3 cm
3-8 cm
KCl extractable NO3-
NaHCO3 extractble ortho-P
NH4OAc extractable K+
NH4AOc extractable Ca2+
NH4AOc extractable Mg2+
NH4AOc extractable Na+
DTPA extractable Mn2+
DTPA extractable Fe2/3+
DTPA extractable Zn2+
(.1M CaCl) pH
mmol/kg
IS
US
UT
Pre-burn
Post-burn

Figure 13.


KCl extractable NH4+
control
treatment
KCl extractable NO3-
NH4AOc extractable ortho-P
NH4AOc extractable Ca2+
NH4AOc extractable Mg2+
NH4AOc extractable K+
NH4AOc extractable Na+
DTPA extractable Mn2+
DTPA extractable Fe2+/3+
DTPA extractable Zn+
Kg ha-1
Kg ha-1
2001
2002
2003
2001
2002
2003

Effects of pinyon (*Pinus monophylla*) and juniper (*Juniperus osteosperma*) expansion and prescribed fire on understory biomass and nutrient concentration in Great Basin sagebrush ecosystems

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Keywords: Pinyon-juniper woodlands, prescribed fire, soil nutrients, plant nutrition

Abstract

Piyon-juniper encroachment into sagebrush-grasslands has been shown as detrimental to understory vegetation and therefore to native foragers. In this experiment we explore the effectiveness of spring prescribed fire on restoration of sagebrush-grasslands by documenting burn effects on understory aboveground biomass and tissue nutrient concentrations. Six native understory plant species important for native sagebrush obligate foragers (*Crepis acuminata*, *Eriogonum umbellatum*, *Eriogonum elatum*, *Poa secunda secunda*, *Festuca idahoensis*, and *Lupinus argenteus*) were chosen to represent the understory plant community. *L. argenteus* is also important for system nutrient cycling and nitrogen fixation. Plants were collected from three microsites common in transitional woodlands (under tree canopy, under shrub canopy, and interspace) the summer before a spring prescribed burn and each of two summers following the burn. Soils were also collected from corresponding locations at two depths (0-8 and 0-52cm) in order to determine burning-soil-plant interactions. All species studied had unique nutritional characteristics and independent responses to burning. Burning did not affect aboveground plant biomass or nutrient concentrations in the first year with the exception of *F. idahoensis* which had increased tissue P. By the second year all species except *C. acuminata* had statistically significant responses to burning. The most common response was for increased aboveground plant weight and tissue N concentrations. Microsite had an impact on soil nutrients but was typically not reflected by plant tissue concentrations with the exception of *F. idahoensis.* Burning caused increases in soil surface and profile ortho-P and inorganic-N in both years following fire. Plants response to burning appeared

to be best linked to the burn treatment itself, and the soil variable profile ortho-P which was influenced by burning.

Introduction

Reduction of understory herbaceous vegetation associated with the expansion of Pinyon-juniper woodlands may reduce available forage and habitat for sagebrush obligate wild ungulates, smaller mammals, and rare bird species such as sage grouse (*Centrocercus urophasianus*) (Brussard and Funari 2001). Much of the Great Basin is currently dominated by sagebrush semi-desert ecosystems. These communities are heterogeneous in regards to vegetation and forage type. At higher elevations, sagebrush semi-desert is increasingly influenced by pinyon and juniper expansion (Miller and Tausch 2001). Many native Great Basin animal species such as the pygmy rabbit (*Brachylagus idahoensis*), pronghorn (*Antilocarpa americana*), sage thrasher (*Oreoscoptes montanus*), sage sparrow (*Amphispiza belli*), and sage grouse are dependent on sagebrush and the associated communities for cover and food (Connelly et al. 1988, Fischer et al. 1996, Brussard and Funari 2001). During spring and summer these species rely on understory vegetation as a food source (Connelly et al. 1988, Fischer et al. 1996, Brussard and Funari 2001). The expansion of pinyon-juniper woodlands and reduction of sagebrush semi-desert is resulting in the loss of habitat for sagebrush obligate species (Connelly et al. 2004). Although pinyon-juniper woodlands have expanded and receded several times over the last 5,000 years, the current rate of expansion is unprecedented. Less than 10 percent of current woodlands are of age classes exceeding 140 years (Miller and Tausch 2001). The recent rapid expansion of Pinyon pine (*Pinus monophylla*) and juniper (*Juniperus osteosperma*) woodlands is due to a combination of warming after the little ice age, fire suppression, and overgrazing by livestock (Gruell 1999; Miller and Wigand 1994: Miller and Rose 1999). As pinyon-juniper woodlands increasingly

dominate sagebrush semi-desert communities, they compete for available resources and often eliminate most understory vegetation (Reiner 2004).

Pinyon and juniper expansion has resulted in increased crown fuel continuity across the landscape (Tausch 1999a, b). Crown cover exceeding 50 percent is sufficient to carry high intensity fire during dry or windy periods. Twenty five percent of all pinyon-juniper woodlands have this crown cover, and the area is expected to double over the next 50 years (Miller and Tausch 2001). High intensity wildfires combined with reduced understory vegetation may leave burned areas susceptible to exotic invasive species such as cheatgrass (*Bromus tectorum*). Invasion by cheatgrass has been documented to increase fire frequency on the landscape, and may shift community composition almost to monocultures (Young and Evans 1973). Cheatgrass monocultures are little value as forage to migrating animals and may further fragment migration corridors (Connelly et al. 2004).

Prescribed fire has been suggested as a mechanism for slowing pinyon-juniper expansion, preventing catastrophic wildfire, and restoring understory vegetation quantity and quality. It is thought that burning in semi-arid environments produces increased available nutrients and reduces competition for resources from woody species (Sturgis 1993). Specifically burning has been shown to increase available soil N, P, Ca, Mg, Mn, and Zn in semi-arid systems (Blank et al. 1994a, 1996; Covington and DeBano 1988; DeBano and Klopatek 1988; Rau 2005). These factors contribute to increased quantity and nutrient content of understory vegetation (Sturgis 1993; Cook et al. 1994; Blank et al. 1994b). Previous studies of understory vegetation response to prescribe fire in arid regions show variable results dependent on burn severity and climatic conditions

following fire. Cook et al. (1994) found that production of perennial herbs was two times greater on burned than control sites following both prescribed and wildfire in sagebrush communities. They also concluded that perennial herb crude protein levels were higher through late summer for two years following fire (Cook et al. 1994). Blank et al. (1994b) found that native and exotic shrubs and grasses had more above ground biomass and higher concentrations of N, P, and SiO_2 when grown on previously burned soils. All species sampled in Spanish Mediterranean shrubland were found to have higher concentrations of mineral nutrients immediately after fire, and then experienced steady declines with time following burn (Carreira and Niell 1992). On a central Florida sand dune, three of four herb species responded positively to fire having increased vegetative growth and flowering, and displaying increased tissue concentrations of N and P (Anderson and Menges 1997). Following burning in Australian sub-tropical semi-arid grassland concentrations of N and P were higher in aboveground plant material four months after burning, but total aboveground biomass did not improve due to dryer than normal conditions (Bennett et al 2002).

In this study we document understory vegetation above ground biomass and nutrient concentration response to prescribed burning on spatial (cover type) and temporal (before and each of two years following burning) scales in a central Nevada pinyon-juniper woodland. We chose a control and treatment location containing a mixture of tree, shrub, and interspace cover types typical of expanding woodlands, and sampled understory vegetation on both sites and all cover types before and after a prescribed burn. Our research questions are: 1) Do understory plant biomass and nutrient concentration in pinyon-juniper woodlands vary spatially in regard to microsite (under tree canopy, under

shrub canopy, interspace)? 2) Are understory plant biomass and nutrient concentration in pinyon-juniper woodlands affected by prescribed burning? 3) Do understory plant biomass and nutrient concentration in pinyon-juniper woodlands vary temporally (time before and after burning)? 4) Does soil nutrient status affect understory plant biomass and nutrient concentration?

Materials and Methods

Study Area

The study is a demonstration area in the Shoshone Mountain Range on the Humboldt-Toiyabe National Forest (Austin Ranger District) in Nye and Lander Counties, Nevada. Underdown Canyon (39°15’11” N 117°35’83” W) is oriented east to west and contains infrequent springs and an ephemeral stream near the top of the drainage. Average annual precipitation ranges from 23cm at the bottom to 50cm at the top of the drainage and arrives mostly as winter snow and spring rains. Average annual temperature recorded in Austin, NV ranges from –7.2°C in January to 29.4°C in July. Lithology of the Shoshone range consists of welded and non-welded silica ash flow tuff. Soils developed on alluvial fans in this study are classified as Coarse loamy mixed frigid Typic Haploxerolls. The soils are extremely coarse grained and have weak to moderate structure. The high clay content of the B_w horizon appears to be from vertical transport rather than formation on ped faces (Table 1).

The vegetation is characterized by sagebrush (*Artemisia tridentata vasayana*) and single leaf pinyon (*Pinus monophylla*) with lesser cover of Utah juniper (*Juniperus osteosperma*). Herbaceous species include the grasses, *Poa secunda secunda*, *Elymus*

elymoides, *Stipa comata*, *Festuca idahoensis*, and *Pseudoroegneria spicata,* and the forbs, *Eriogonum umbellatum*, *Eriogonum ovalafolium*, *Eriogonum elatum*, *Eriogonum heracleoides*, *Crepis acuminata*, *Phlox longifolia*, *Agoseris glauca*, *Lupinus argenteus*, and *Penstemon* species (http://plants.usda.gov/index.html). *Bromus tectorum*, an invasive annual grass, is not a large component of the study area. The vegetation occurs in patches of variable tree dominance typical of intermediate age class woodlands in the central Great Basin and ranges from low (12% cover, 2152 kg/ ha) to high tree dominance (74% cover, 14213 kg/ ha) (Reiner 2004).

Yearly average precipitation measured over the three year study period in a standing rain gauge ranged from 27 to 34cm during the water year (Table 2). Precipitation was similar among years with most precipitation arriving between October 15th April 15th. Little rain (0 to 4cm) fell during the active growing season (April 15th to July 16th).

Study design and data collection

The study was a split-plot design with sub-sampling. The study sites were located on northeast facing alluvial fans at elevations of 2195 and 2225m. The location at elevation 2195m was a control, and the treatment location at 2225m was burned by USDA Forest Service fire personnel on May 11-14, 2002 under favorable weather conditions. Because soil and fuel moisture were relatively high during the time of burning the vegetation and duff were consumed in patches creating a landscape of burned and unburned islands. Four replicate plots were sampled on both the control and treatment locations. Plots were characterized by intermediate tree cover (38% cover, 6722 kg/ ha), and contained a mix of trees, shrubs, and interspaces. This cover type represents the transition period from shrub dominated systems to tree dominated systems, and may be a threshold for

understory vegetation recovery following fire (Tausch and West 1995). To characterize soils on the 2195m control and 2225m burn treatment locations, soil pits were dug to a depth of 100cm and the soil horizons were identified (Table 1).

Five understory plant species were chosen for sampling because of their value as forage to native animal species: *E. umbellatum, E. elatum, C. acuminata, F. idahoensis,* and *P. secunda secunda* (Barnett and Crawford 1994; Fischer et al. 1996; and Drut et al. 1994). A sixth species was chosen for its importance to ecosystem nutrient cycling and N-fixation, *L. argenteus* (Hainds et al. 1999; Hendricks and Boring 1999). Two adult plants of the six species were collected at similar phenology (peak biomass with flowers or seed heads) from each of three microsites (under tree canopy, under shrub canopy, and interspace) using a shears to remove all aboveground biomass. All clipped vegetation was marked with a nail and washer to avoid future sampling of the same plant. *E. umbellatum* and *elatum* were sampled only in interspace and under shrub microsites. Sampling was conducted in June 2001, 2002, and 2003 to determine temporal, spatial, and treatment differences in understory plant biomass and nutrient concentration. Plants were collected from all locations, replicate plots, microsites, and in all years. The overall study included 2 locations (burned and control), 3 years, 3 microsites, 4 replicates, 6 species, and 2 sub-samples ($n \approx 864$).

Plants were collected at similar phenology (i.e. peak biomass and had flowers or seed heads) returned to the lab, and dried at 60°C for 48 hours then weighed.

Dried plant samples were ground in an Udi™ mill and placed into labeled vials. All samples were placed in a dessiccator for 72 hours prior to analyses. Approximately 0.5 grams of dry plant material is placed into a crucible ashed in a muffle furnace for four

hours at 500°C, solublized with 20ml of 1.0 N HCl solution, and diluted with de-ionized water in a 100ml volumetric flask (Miller 1998). Extractable metals were determined using atomic absorption and atomic emission spectrophotometry (Miller 1998). Solution P was determined using vanomolybedenate-blue chemistry (Miller 1998). Plant total N was determined using a standard Kjeldahl digest and flow injection chemistry (Horneck and Miller 1998).

Soil samples were collected from the same locations, microsites, and replicates as plants during early November over the same three year period at four depths: 0-8, 8-23, 23-38, and 38-52cm (Rau 2005). All soil was brought back to the lab, air-dried, and sieved to 2 mm. Sub-samples were analyzed for DTPA extractable micronutrients, KCl extractable NH_4^+ and NO_3^-, $NaHCO_3$ extractable P, and NH_4OAc extractable metals (Lindsay and Norwell 1978; Keeney and Nelson 1982; Olsen and Sommers 1982; Thomas 1982). Soil pH was measured in $CaCl_2$ (0.1 M) using a gas electrode (Schofield and Taylor 1955). Extractable metals were determined using atomic absorption and atomic emission spectrophotometry. Extractable NH_4^+ and NO_3^- were determined using flow injection, and extractable P was determined using molybedenate-blue chemistry.

Statistical Analyses

The study was analyzed as a split-split plot nested within a paired before after control impact (BACI) design (Stuart-Oaten et al. 1986). Location and year were considered main effects in the BACI design. Differences between the change from pre-burn (2001) to post-burn (2002 and/or 2003) of the burn and control locations were considered to imply a burn treatment effect (Smith 2002; Underwood 1994; but see Hurlbert 1984). Microsite was treated as a split-plot within location and year, and depth as a split-split

plot within location, year and microsite. The data were analyzed using a mixed effects model in SAS™ (SAS™ 2004). Differences in surface soils and the soil profile were examined by creating two depth increments from the original data. These included the 0-8 cm surface depth and a 0-52 cm soil profile depth. Significant differences existed among surface soils and the soil profile for most of the response variables, and the differences among location, year and microsite were evaluated using a reduced model for each depth.

Differences in plant biomass and nutrient concentrations among locations, dates, microsites, and species were evaluated using the same design that was used for the soils data. Location and year were treated as main effects. Microsite was treated as a split-plot within location and year, and species as a split-split plot within location, year and microsite. The data were analyzed using a mixed effects model in SAS™ (SAS™ 2004. Significant differences existed among species for most of the response variables, and the differences among location, year and microsite were evaluated using a reduced model for each species.

The Kolmogorov-Smirnov test was used to test for data normality. All data were natural log transformed to meet the assumption that the data were normally distributed. Mean comparisons for both the complete and reduced models were Tukey-Kramer adjusted and considered significant at the 95% confidence level.

Relationships among soil and plant nutrients for individual species were explored using SAS™ Canonical correspondence procedures. Soil surface and soil profile variables (Mn^{2+}, Zn^{2+}, $Fe^{2/3+}$, Ca^{2+}, Mg^{2+}, K^{+}, Na^{+}, pH, ortho-P, and inorganic-N concentrations) were used as predictors for the plant variables (above ground weight, tissue concentration

Mn, Zn, Fe, Ca, Mg, K, Na, P, and N). Because it is difficult to separate the effect of woody vegetation removal and fire induced soil changes on understory plant characteristics, we created a binary variable named “treatment.” This binary variable was used to take into account all attributes related to burning not measured in this study. All vegetation collected from the control location as well as vegetation collected from the burn treatment location before the burn occurred was designated “unburned”. Vegetation collected from the burn treatment location after the prescribed burn was designated “burned”. Unmeasured variables that may fall into this category include soil water content, competition for mineral nutrients, shading from overstory vegetation, soil temperature, and removal or production of organic compounds which inhibit or promote vegetation or microbial communities (Neary et al. 1999; Sturges 1993; Certini 2005). In our canonical correspondence models the binary variable “treatment” and the soil chemical parameters measured were used as predictor variables for understory plant biomass and nutrient characteristics. Therefore, if the variable “treatment” had a higher partial correlation with the canonical predictor variable than measured soil variables, it is possible that variables we did not measure are having a significant influence over understory plant characteristics.

Pearson Product-Moment Correlations were used to identify relationships among plant nutrients for individual species. This allowed us to determine whether plant tissue nutrient concentrations were related solely to soil nutrients or if physiological constraints regarding nutrient ratios were also driving changes in plant tissue nutrients.

Results

Plants

The combined model revealed differences in plant weight, tissue P, K, Ca, Fe among years for the burn and control location, and for individual species as indicated by year-location-species interactions (Table 3). Strong differences among species for all measured plant characteristics confirmed that each species should be considered independently (Table 4). The analyses of individual species indicated that microsite and microsite-burning interactions were only a factor for *F. idahoensis*.

The individual species analyses showed the forb, *L. argenteus*, had increased plant weight, tissue N, and P by the second year after burning on the treated location. Plant concentrations of Fe decreased in the first and second year after burning on the treated location, and the overall effect of burning was significant for weight, N, P, and Fe (Figure 1).

C. acuminata, a perennial forb, had no statistically significant response to burning, although there were trends towards increased plant weight and tissue N (Figure 2).

The woody forb, *E. umbellatum*, did not re-sprout in the first season following the spring prescribed burn on the burned location. By the next spring the forb had re-sprouted and had higher tissue N and K but lower Fe on the burned location (Figure 3). A trend toward higher P and Mg was also apparent.

E. elatum, a low growing fleshy forb, re-sprouted shortly after the prescribed fire on the treated location but had no significant nutritional response. By the second year after burning plant weight and tissue Na had increased as a result of the fire (Figure 4). Overall burning increased plant weight and plant Na (Figure 4).

The perennial grass, *P. secunda secunda*, also did not re-sprout in the first season following the spring prescribed fire on the burned location. By the second year following the prescribed fire, *P. secunda secunda*, had re-sprouted and had increased plant weight, N, and K as a result of burning (Figure 5). There were also trends of increased tissue Mg and Zn, and decreased Fe.

F. idahoensis, another perennial grass, did re-sprout the first season after the fire on the burned location and had immediate increases in tissue N and P (Figure 6). In the second year after burning *F. idahoensis*, had increased tissue N and decreased Mg, but P was not dissimilar than control location plants (Figure 6). Overall burning increased plant tissue N, and P with trends toward increased K, and a trend toward decreasing Fe (Figure 6). It appears that burning eliminated inherent microsite differences that affected plant weight (plants were typically larger under tree and shrub canopies before burning) and disproportionately increased the concentrations of plant Na in under shrub canopy microsites by the second season after burning as indicated by the year-location-microsite interaction ($P < 0.05$).

Soils

The combined model revealed differences in inorganic-N, ortho-P, Mn^{2+}, Zn^{2+}, and $Fe^{2/3+}$ among years for the burn and control location, and for the soil surface and soil profile as indicated by year-location-depth interactions (Table 5). Near surface soils typically had higher concentrations of mineral nutrients than the entire profile with the exception of Na^{+} which increased with depth (Table 5; Figure 7). The model split by depth indicated that burning increased near surface soil (0-8cm) inorganic-N and ortho-P in both 2002 and 2003 as indicated by the year-location interaction ($P < 0.05$, DF = 2 and

36). It also appears there is a trend toward increased pH, Mg^{2+}, Ca^{2+}, Mn^{2+}, Zn^{2+}, and $Fe^{2+/3+}$ after burning (Figure 7). Microsites vary spatially in nutrient availability depending on cover type, but relationships among nutrient availability remain similar over time on the unburned control location. Burning significantly increased near surface inorganic-N only at under shrub canopy microsites and only in 2003 as indicated by the year-site-microsite interaction ($P < 0.05$, DF = 4 and 31). Burning only increased near surface ortho-P at under shrub and under tree canopy microsites in the first year following the prescribed burn, but near surface ortho-P was not different than pre-burn levels in the second year ($P > 0.05$, DF = 4 and 34).

The analyses for soil profile nutrients (0-52cm) revealed that burning only increased inorganic-N and ortho-P and that these effects were apparent in both years post-burn and for the overall burn effect as indicated by year-site interactions ($P < 0.05$, DF = 2 and 36; Figure 7).

Plant and Soil Interactions

Results from canonical correspondence analyses performed for individual plant species indicate that burning, soil nutrients and plant nutrients are related.

Three canonical soil and plant axis had significant correlation coefficients for the forb *L. argenteus* (Table 6a). The first set of variables relate burn treatment, soil surface K^+, profile K^+, and soil profile ortho-P with plant weight, tissue P, and N ($R^2 = 0.90$, proportion of variance = 0.47, $P < 0.001$). The second canonical axis relates soil surface ortho-P with plant P and K ($R^2 = 0.74$, proportion of variance = 0.15, $P = 0.0012$). The third canonical axis relates plant K to soil surface Ca^{2+}, and ortho-P, soil profile ortho-P, and profile pH ($R^2 = 0.69$, proportion of variance = 0.11, $P = 0.0196$).

The perennial forb *C. acuminata* had two canonical soil and plant axis with significant correlation coefficients (Table 6a). The first axis best relates burn treatment and soil profile Mn^{2+} to plant Ca, Mg, Na, and P ($R^2 = 0.89$, proportion of variance = 0.41, $P < 0.0001$). The second canonical variable relates surface soil ortho-P and soil profile pH with plant P ($R^2 = 0.83$, proportion of variance = 0.25, P = 0.0083).

One canonical axis had a significant correlation coefficient for the woody forb *E. umbellatum* (Table 6a) ($R^2 = 0.99$, proportion of variance = 0.76, P = 0.0098). The axis does not have strong partial correlations with any particular predictor or response variables.

E. elatum also had only one set of significant canonical variables relating treatment, surface soil K^+, soil profile K^+, Na^+, ortho-P, pH, plant K, and plant N (Table 6b) ($R^2 = 0.97$, proportion of variance = 0.63, P = 0.0220).

The cool season perennial grass, *P. secunda secunda*, had two significant canonical axes (Table 6b). The first axis relates treatment, soil profile ortho-P, Ca^{2+}, Mg^{2+}, and K^+ with plant Mg and plant N ($R^2 = 0.93$, proportion of variance = 0.53, $P < 0.0001$). The second canonical axis best relates soil Zn^{2+} and plant tissue Mn concentrations ($R^2 = 0.82$, proportion of variance = 0.15, P = 0.0073).

The perennial grass *F. idahoensis* had only one significant canonical axis primarily relating treatment, surface soil K^+, soil profile K^+, Ca^{2+}, and Na^+ with plant N, K, and P (Table 6b) ($R^2 = 0.89$, proportion of variance = 0.52, P = 0.0214).

Species also had partial correlations for groups of nutrients and nutrient ratios for N, P, K, and Mg as indicated by Pearson Product-Moment Correlations ($P < 0.05$). *P. secunda secunda* had significant partial correlations for plant Mg and K, plant Mg and P, plant Mg

and N, plant K and P, and plant K and N (Table 7). *E. umbellatum* had partial correlations for plant Mg and K, plant Mg and P, plant K and P, plant K and N, and plant N and P (Table 7). *F. idahoensis* had partial correlations for plant Mg and K, plant Mg and N, plant K and P, plant K and N, and plant N and P (Table 7). The other species examined have simpler relationships to nutrient ratios. *Lupines* and *C. acuminata* have significant plant Mg and P ratios and *L. argenteus* had a significant N and P ratio (Table 7).

Discussion

Plants

Plant species differed in their nutritional requirements and aboveground tissue concentrations (Table 3). These differences may be exploited by native foragers. The effect of microsite on aboveground plant characteristics was minimal in this study. Burning had a statistically significant effect on all of the plant species in the study with the exception of *C. acuminata*. Individual species differed in response to burning most likely due to burn tolerance, root characteristics, mycorrhizal associations, and ecophysiological characteristics (Steuter and McPherson 1995; Anderson and Menges 1997). Species dependent responses to burning has been documented by Cook et al. (1994) and Anderson and Menges (1997). All of the perennial vegetation in our study is capable of re-sprouting after fire. Because the fire occurred in mid May of 2002 and plants were collected in late June 2002 two of the plant species, *E. umbellatum* and *P. secunda secunda*, were not present in large numbers at the time of the 2002 sample date. *E. umbellatum* is a woody forb with low shrub characteristics, and although this species has extensive below ground biomass allocation, it did not re-sprout in the first months

following the burn. A larger fuel component of this shrub-like forb at the time of burning could have increased temperatures above *E. umbellatum*'s root system and delayed vegetative recovery (Steuter and McPherson 1997). *P. secunda secunda* is a cool season grass and likely did not re-sprout in the first year because conditions were not favorable in the weeks following the burn. Of the plants that did re-sprout in the first year, there were no statistically significant differences in above ground plant biomass, N, K, Mg, Ca, Na, Mn, or Zn at control or burned locations (Figure 1-6). However, *F. idahoensis*, did have higher plant P concentrations in the first season after the burn, and *L. argenteus* had decreased Fe on the burn location (Figure 1, 6). Increases in plant nutrients have been reported in the first growing season following fire by Bennett et al. (2002), Anderson and Menges (1997), and Cook et al. (1994) for spring burns. Fescue's increased nutrient response to burning in the first season may be because it generally had lower above ground biomass on the burned location. Smaller and younger plants have been documented to have higher above ground concentrations of mineral nutrients because of the low percentage of structural tissues in smaller younger plants (Schaffers 2002).

By the second year after burning, large scale increases in plant biomass and nutrient concentrations were evident (Figure 1-6). Although most plant species re-sprouted immediately following fire they were delayed in phenology and growth period compared to plants on the unburned location. In the second year following the prescribed fire, plants on both burned and control locations were sampled at identical phenology after similar periods of growth. Preliminary results indicate that the percent cover of the six species studied remains unchanged or increased after burning (Reiner 2004; Dhaemers 2005). If cover remains constant or increases after burning, and plant weight and nutrient

concentrations increase, then the above ground biomass and nutrient pools should increase as a result of burning. This would be favorable to native animal species utilizing these sites (Connelly et al. 2004; Cook et al. 1994). Increased aboveground biomass and nutrient concentrations are most likely a result of complex interactions between removal of woody vegetation and changes in nutrient and water availability after burning. Without the advantage of soil moisture data and vegetation removal experiments, it is difficult to separate the effects of increased soil nutrients, increased soil moisture, and plant competition on plant biomass and nutrition. Sturges (1993) documented that removal of woody vegetation in arid sagebrush-steppe communities resulted in increased soil moisture and doubled herbaceous vegetation in the first three years following brush removal. The effects of brush removal were evident twenty years following treatment.

Soils

Burning induced increases in soil inorganic-N and ortho-P are due to release of N and P from organic matter during heating or oxidation, and decomposition of incompletely consumed above and below ground biomass (Blank and Zamudio 1998; DeBano and Klopatek 1988; Covington et al. 1991; Hobbs and Schimel 1984). Microsites respond differently to burning because of variations in plant uptake, litterfall, and microbial communities (Rau 2005; Chambers 2001; Booth et al. 2003; Compton and Boone 2002).

The increase in inorganic-N though the soil profile is due to release of NH_4^+ from organic matter during burning and decomposition, and subsequent microbial conversion of NH_4^+ to NO_3^- (Covington and DeBano 1991). The NO_3^- ion is highly mobile and easily moves through the soil profile (Chorover et al. 1994). The large increase in soil profile ortho-P may be a result of a release of phosphorus during combustion and decomposition

of below ground biomass after burning (Neary et al 1999). Also, although phosphorus is not normally thought to be mobile, the alluvial fans in Underdown canyon are extremely coarse grained (2mm mineral particles > 60%) and vertical transport of colloidal P may explain the large increase at lower depth (Hens and Merckx 2001; Rau 2005).

Plant and Soil interactions

The predictor variable "treatment" explains most of the variance related to *L. argenteus* variables for the first canonical axis. Plant P appears to be following a temporal pattern, but was also influenced by the prescribed burn (Figure 1). However, plant weight and plant N almost directly track changes in soil profile ortho-P indicating a direct positive relationship (Figure 1, 7). The relationship between soil surface and profile K^+ and plant characteristics does not appear direct (Figure 1, 7). In the second canonical axis a distinct positive relationship between temporal variation in soil surface ortho-P and plant P is evident, although a burn effect is also apparent (Figure 1, 7). The variable plant K does not display a clear relationship with soil surface ortho-P, but may be best explained by a combination of all soil variables that are less dynamic over time and treatment. The third canonical axis supports the idea that plant K is controlled by a combination of several variables. *L. argenteus* is a deep rooted N-fixing legume and appears to be directly influenced by fire induced changes in soil surface and profile ortho-P as well as soil variables not measured in this study. In addition to physiological changes, prescribed burning increased *L. argenteus* cover on the treated location (Dhaemers 2005). The increased cover and improved physiological performance of *L. argenteus* on burned sites may be important for carbon and nitrogen cycling in arid systems (Hendricks and Boring 1999; Hainds et al. 1999).

The two sets of significant canonical variables for *C. acuminata* are poorly correlated with any one particular variable, and relationships between soil and plant variables are not immediately apparent. However the second canonical axis does support the idea that a temporal factor is operating on soil pH which influences ortho-P availability, and in turn is reflected by plant uptake of P.

The two sets of statistically significant canonical variables for the woody forb *E. umbellatum* were also poorly correlated with any one independent variable. Burning did have direct effects on plant N and K, but no relationship to these variables is suggested (Figure 3). The weak partial correlation of independent variables is not surprising given the complexity of plant mineral nutrition, and the direct relationship of soil chemistry to plant characteristics is not clear for *E. umbellatum*. Perhaps interactions between soil variables drive plant characteristics for this species.

E. elatum displayed a clear positive relationship between treatment, soil profile ortho-P, and plant K and N (Figure 4, 7). It appears as though *E. elatum*'s uptake of mineral nutrients, especially N, has been influenced by burn induced changes in soil properties particularly profile ortho-P.

For the cool season perennial grass *P. secunda secunda* there is a strong positive relationship between the variables treatment, soil profile ortho-P, and plant N (Figure 5, 7). Although not statistically significant there are temporal and possible burn effects of increased soil profile Mg^{2+} on plant Mg (Figure 5, 7).

The perennial grass *F. idahoensis* has a clear positive relationship between treatment and the plant variables N and K (Figure 6, 7). There may also be an effect of the

unmeasured variables in “treatment” on plant P in the first year after burning. Other soil relationships appear more influential but they are not statistically supported.

Plant nutrition and plant interactions with soil are complex and not well understood in non-agricultural systems (Schaffers 2002). Unlike agricultural systems wild or native plant species have developed adaptations to nutrient poor environments, and changes in soil fertility may not directly influence the corresponding plant nutrient (Hayati and Procter 1990, 1991; Schaffers 2002). Also wild systems, especially those in arid regions, are subject to intense plant competition for soil resources further complicating plant soil interactions (Sturgis 1993).

Although each plant species in our study responded differently to fire the most common response of understory vegetation to burning was for increased plant weight (*L. argenteus, E. elatum,* and *P. secunda secunda)* and increased tissue concentrations of N (*L. argenteus, E. umbellatum,* and *P. secunda secunda*). Interestingly the large increases in soil inorganic-N, at the surface or through the profile, do not statistically correspond with increases in plant weight or plant tissue N concentrations. Schaffers (2002) suggests that plant tissue N concentrations are not driven by soil inorganic-N pools, but rather by soil N-mineralization potential. In our study plant weight and tissue N were more closely related to the blanket variable “treatment” and soil surface or profile ortho-P concentrations. The variable “treatment” comprises a complex series of variables not measured in this study, although it could be argued that fire reduced competition and increased soil moisture by reduction of woody biomass. Removal of woody biomass has been shown to increase soil moisture and herbaceous biomass in arid rangelands (Sturges 1993; Chambers and Linerooth 2001; Wright and Chambers 2002). The concurrent

increase in aboveground herbaceous biomass and tissue N concentrations has been reported previously following fire, but is typically thought of as an inverse relationship due to “dilution effects” (Cook et all 1994; Carreira and Niell 1992; Schaffers 2002). The increased tissue N concentrations could also be related to an increase in photosynthesis often associated with re-sprouting after burning. Re-sprouts following fire are less susceptible to photo oxidation during times of drought stress because they can increase their stomatal conductance in lieu of higher soil moisture, and it has been demonstrated that photosynthetic activity is positively correlated with leaf tissue N concentrations (Fleck et al. 1998; Ripullone et al. 2003). Soil ortho-P especially in the profile may have positive influences on plant weight and leaf tissue N due to its effect on root density and morphology (Marschner 1995). Localized increases in soil P have been shown to increase root density and the number of root hairs per root in areas of elevated P (Marschner 1995). Large increases in soil profile ortho-P following fire may result in the reallocation of biomass to root structures deeper in the soil profile. This would allow plants to avoid drought stress as surface soil dries out, and make them more efficient at nutrient uptake by reducing the transport distance to individual root hairs (Marschner 1995). Despite the lack of statistical evidence relating soil inorganic-N, aboveground plant weight, and plant tissue N we believe that the two are explicitly linked. The immediate increase in soil inorganic-N was not reflected in plant tissue N until the second year after the fire. This makes a linear statistical comparison difficult and may explain the low correlation between soil and plant N.

The partial correlations of soil K^+, Na^+, and possibly Ca^{2+} to aboveground plant weight and tissue N are most also likely statistical artifacts mainly because K^+, Na^+, and Ca^{2+}

were higher on the burn location even before treatment and therefore are somewhat co-linear with burning.

The temporal variation in plant tissue P represented in all of the plant species studied seems more closely related to surface soil ortho-P concentrations and temporal or fire mediated variation in soil pH especially for the forb *E. umbellatum* which appears to be more influenced by surface soil characteristics than the other species studied. Shaffers (2002) found that P uptake was directly related to soil pH and was highest between pH-$CaCl_2$ 5.5 and 6.0. It is not clear at this time what is influencing the temporal pattern of soil pH and plant P uptake.

The relationships with fire induced increases in plant tissue Mg, K, and soil properties is not well defined however it is possible that increases and plant K and Mg are directly and indirectly related to a small but insignificant fire induced increase in soil profile Mg^{2+}. Cations in soil have been positively and directly correlated with plant uptake of the corresponding cation (Schaffers 2002). Additionally it has been observed that cation concentrations in soil do not necessarily affect plant uptake of the corresponding cation, but rather directly influences the uptake of a different cation (Hayati and Proctor 1990). These relationships are not well understood, but may be a result of simultaneous and complex nutrient limitation in native soils (Hayeti and Proctor 1990). It is also possible that plants adjust the uptake of K^+ or Mg^{2+} in response to changes in plant tissue N in order to maintain N:P:K:Mg ratios (Marschner 1995). This appears to be the case for *P. secunda secunda*, *E. umbellatum*, and *F. idahoensis*. These were the three species with Mg or K tissue response. These ratios may be important for maintaining proper function of photosynthetic mechanisms and increased stomatal conductance (Marschner 1995).

Conclusions

Plant nutritional response to fire has been documented by several authors yet few attempts have been made to determine soil chemical changes direct influence on plant nutrition (Carreira and Niell 1992; Blank et al. 1994; Cook et al. 1994; Anderson and Menges 1997; and Bennett et al. 2002). Results from our study indicate that soils may change temporally in response to yearly climatic fluctuations, and plants growing on these soils respond to those changes. It has also been determined that fire may have significant impacts on soil nutrients immediately following fire and for at least one year post burn (Rau 2005). Subsequently the vegetation that re-sprouts or germinates on burned locations responds to dramatic changes in plant competition, soil water content, organic compounds, microbial communities, and soil nutrients (Carreira and Niell 1992; Blank et al. 1994; Cook et al. 1994; Anderson and Menges 1997; and Bennett et al. 2002).

The direct influence of soil nutrients on plant nutrition in wild systems is not well understood but appears to be species dependent. Previous studies have found that soil nutrient supply or content has direct positive or negative relationships with plant tissue concentrations of that nutrient, or other related nutrients (Schaffers 2002; Hayati and Proctor 1990). In our system the effect of burning and changes in soil surface and profile ortho-P concentrations had dramatic influences on plant weight and tissue N concentrations. We also believe that fire induced increases in soil inorganic-N play a large role in elevated plant weight and tissue N concentrations after fire. Temporal and fire mediated variations in soil surface and profile pH and soil surface ortho-P influenced plant tissue P concentrations. Increased levels of Mg^{2+} in soil after fire may be related to

plant tissue Mg and K concentrations. Plants in our study, with the exception of *E. umbellatum,* may have responded better to changes that affected the entire soil profile because of their rooting structure. Most perennial species in arid environments have deep root systems which allow them to access soil moisture longer into the growing season (Schwinning and Ehleringer 2001). *E. umbellatum* may have root morphology more similar to shrubs such as sagebrush which have fine roots near the surface, and are able to take advantage of small rain events that wet only surface soil in arid environments (Leffler et al. 2004).

In addition to plant physiological changes the percent cover of species studied was not affected or increased following fire (Dhaemers 2005). Unchanged or increased cover coupled with increased understory plant biomass and tissue nutrient concentrations should increase above ground pools of mineral nutrients, and produce sites more favorable to native foragers including sage grouse (Connelly et al. 2004; Cook et al. 1994).

It appears that spring prescribed burning is a plausible option for slowing pinyon-juniper expansion as long as sufficient understory vegetation is present on locations to be treated (Reiner 2004). Spring burning increased understory plant biomass and nutrient concentration for most species in our study. However, burning should be done in a patch like fashion so that protective cover is not eliminated for native species such as sage grouse (Connelly et al. 1998). Fall burning in these woodlands may not have the same beneficial effects due to decreased soil and fuel moisture and increased fire severity (Cook et al. 1994).

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Figure 1. Mean above-ground plant weight (g), tissue nutrient concentrations (ug g^{-1}), and standard errors for *Lupinus argenteus* in 2001, 2002, and 2003 on the control (2195m) and burned (2225m) locations.

Figure 2. Mean above-ground plant weight (g), tissue nutrient concentrations (ug g^{-1}), and standard errors for *Crepis acuminata* in 2001, 2002, and 2003 on the control (2195m) and burned (2225m) locations.

Figure 3. Mean above-ground plant weight (g), tissue nutrient concentrations (ug g^{-1}), and standard errors for *Eriogonum umbellatum* in 2001, 2002, and 2003 on the control (2195m) and burned (2225m) locations.

Figure 4. Mean above-ground plant weight (g), tissue nutrient concentrations (ug g^{-1}), and standard errors for *Eriogonum elatum* in 2001, 2002, and 2003 on the control (2195m) and burned (2225m) locations.

Figure 5. Mean above-ground plant weight (g), tissue nutrient concentrations (ug g^{-1}), and standard errors for *Poa secunda* in 2001, 2002, and 2003 on the control (2195m) and burned (2225m) locations.

Figure 6. Mean above-ground plant weight (g), tissue nutrient concentrations (ug g^{-1}), and standard errors for *Festuca idahoensis* in 2001, 2002, and 2003 on the control (2195m) and burned (2225m) locations.

Figure 7. Mean extractable soil nutrient concentrations (ug g^{-1}) and standard errors in 2001, 2002, and 2003 for the soil surface (0-8cm) and soil profile (0-52cm) at the control (2195m) and burned (2225m) locations.

Table 1. Soil profile depths and particle sizes of soils at the control (2195m) and burn (2225m) locations.

Depth (cm)	>2mm%	sand%	silt%	clay%
0-16 A_1	60.5	67.6	23.9	8.5
16-28 A_2	55.3	59.3	33.5	7.2
28-40 BA	47.8	65.9	27.1	7.1
40-70 B_w	47.0	53.9	21.4	24.6
70-100 BC	31.1	64.5	24.8	10.7

The >2mm% is of the total soil. The sand, silt, and clay% make up 100% of the <2mm soil fraction Letters following soil depth refer to the soil horizon nomenclature.

Table 2. Mean precipitation (cm) collected on the study sites over the three year study period.

Dates	Mean precipitation (cm)		
	2001	2002	2003
Oct. 15th to Apr. 15th	23	24	26
Apr. 15th to July 15th	4	0.5	3
Total water year	27	27	34

Table 3. Results for the ANOVA comparing plant weight and nutrient concentrations for all locations, microsites, years, and species.

		Weight		N		P		K		Ca	
Effect	**DF**	**F**	**P**	**F**	**P**	**F**	**P**	**F**	**P**	**F**	**P**
Location	1	18.8	0.0049	22.2	0.0033	4.4	0.0812	4.6	0.0767	0.8	0.4054
replicate(location)	6										
Year	2	5.9	0.0162	19.4	0.0002	55.2	<.0001	71.6	<.0001	9.3	0.0036
year*location	2	28.2	<.0001	5.5	0.0206	1.0	0.4057	6.5	0.0122	1.6	0.2502
year*replicate(location)	12										
Microsite	2	8.0	0.0014	0.4	0.7078	3.9	0.0297	10.2	0.0004	1.7	0.1946
year*microsite	4	1.4	0.2601	1.7	0.1699	2.2	0.09	0.8	0.5195	0.1	0.9694
location*microsite	2	0.6	0.5338	1.4	0.2521	3.7	0.0358	0.5	0.6391	0.9	0.4253
year*location*microsite	4	1.5	0.2258	2.0	0.1115	0.9	0.4953	0.7	0.5747	1.3	0.2896
Microsite*year*replicate(location)	33										
Species	5	57.3	<.0001	73.8	<.0001	142.5	<.0001	226.7	<.0001	193.4	<.0001
year*species	10	15.3	<.0001	3.4	0.0002	10.9	<.0001	8.1	<.0001	15.1	<.0001
location*species	5	15.0	<.0001	2.4	0.0349	3.2	0.0072	7.6	<.0001	0.9	0.4706
year*location*species	8	10.4	<.0001	0.9	0.4815	2.5	0.0115	2.2	0.0244	2.3	0.0206
Microsite*species	8	6.1	<.0001	2.1	0.0365	1.1	0.338	0.9	0.4828	0.8	0.5652
year*microsite*species	16	1.8	0.0353	1.0	0.4628	0.6	0.91	1.5	0.1058	1.4	0.1369
location*microsite*species	8	0.8	0.6021	0.8	0.633	1.6	0.1302	0.9	0.5212	1.6	0.1175
year*location*microsite*species	11	2.0	0.0266	0.7	0.7714	0.5	0.9082	0.9	0.5595	0.8	0.6672
species*microsite*year*replicate(location)	532										

		Mg		Na		Mn		Fe		Zn	
Effect	**DF**	**F**	**P**	**F**	**P**	**F**	**P**	**F**	**P**	**F**	**P**
Location	1	1.3	0.2968	0.1	0.7828	1.0	0.3623	0.0	0.908	0.2	0.7059
replicate(location)	6										
Year	2	29.8	<.0001	5.8	0.0177	22.0	<.0001	17.2	0.0003	16.9	0.0003
year*location	2	3.0	0.0887	1.0	0.388	1.3	0.3	6.1	0.0151	0.8	0.4884
year*replicate(location)	12										
Microsite	2	2.6	0.0877	2.4	0.1102	0.5	0.6024	0.5	0.6042	2.5	0.0954
year*microsite	4	2.0	0.1199	2.1	0.1056	0.6	0.702	0.7	0.5838	1.0	0.4166
location*microsite	2	0.5	0.5989	1.9	0.1739	2.2	0.1244	0.2	0.8636	0.4	0.699
year*location*microsite	4	0.5	0.7302	0.4	0.8205	3.1	0.0284	1.5	0.2212	0.5	0.7703
Microsite*year*replicate(location)	33										
Species	5	358.9	<.0001	16.8	<.0001	16.2	<.0001	17.9	<.0001	9.4	<.0001
year*species	10	7.3	<.0001	1.3	0.2389	4.6	<.0001	5.9	<.0001	3.9	<.0001
location*species	5	4.6	0.0004	0.8	0.5386	2.5	0.031	2.1	0.0603	1.7	0.1309
year*location*species	8	0.3	0.9526	1.6	0.128	1.5	0.1698	4.7	<.0001	1.6	0.1215
Microsite*species	8	1.3	0.2517	1.1	0.3856	1.9	0.0645	1.5	0.1546	1.6	0.1152
year*microsite*species	16	3.3	<.0001	1.6	0.0544	1.9	0.0167	0.9	0.5575	1.7	0.049
location*microsite*species	8	0.8	0.5997	0.9	0.5361	0.2	0.9923	1.3	0.2367	0.7	0.7374
year*location*microsite*species	11	0.7	0.7146	1.8	0.0479	0.5	0.9045	1.6	0.0951	1.3	0.2396
species*microsite*year*replicate(location)	532										

Table 4. Means for aboveground plant weight (g) and tissue nutrient concentrations (ug g^{-1}) for vegetation on control and burned locations in 2001 before the prescribed fire. Means followed by different letters within columns differ statistically (Tukey-Cramer, $P < 0.05$).

Species	Weight	Ca	Mg	K	Na	Fe	Mn	Zn	P	N
L. argenteus	10 b	6522 a	2939 a	15154 b	129 b	186 ab	65 ab	59 b	5042 c	30837 a
C. acuminata	4 d	6562 a	1870 b	23909 a	123 b	106 cd	73 a	76 a	10080 a	25361 b
E. umbellatum	13 a	7114 a	1814 b	10533 cd	162 a	200 a	72 a	28 d	4310 d	18098 cd
E. elatum	7 c	5349 b	3004 a	25765 a	135 b	68 d	58 b	40 cd	6849 b	24171 b
P. secunda secunda	6 c	2311 c	927 c	11538 c	97 c	138 bc	71 a	53 bc	4657 cd	19048 c
F. idahoensis	10 b	2061 c	1004 c	9224 d	93 c	225 a	68 ab	48 bc	4498 d	15908 d

Table 5. Results for the ANOVA comparing measured soil variables at all locations, microsites, years, and soil depths.

		pH		Inorganic-N		Ortho-P		K^+		Mg^{2+}	
Effect	**DF**	**F**	**P**	**F**	**P**	**F**	**P**	**F**	**P**	**F**	**P**
Location	1	1.30	0.2976	18.40	0.0052	4.24	0.0853	89.9	<.0001	1.87	0.2202
replicate(location)	6										
Year	2	5.65	0.0187	2.90	0.0940	12.81	0.0011	0.18	0.8372	3.95	0.048
year*location	2	2.36	0.1366	6.92	0.0100	16.52	0.0004	0.01	0.9934	0.84	0.4566
year*replicate(location)	12										
Microsite	2	13.45	<.0001	2.79	0.0746	18.52	<.0001	10.1	0.0003	32.65	<.0001
year*microsite	4	3.26	0.0223	2.52	0.0583	10.71	<.0001	0.84	0.5101	2.16	0.0935
location*microsite	2	3.19	0.0533	2.78	0.0752	2.40	0.1048	0.47	0.6292	0.34	0.7155
year*location*microsite	4	1.49	0.2252	1.34	0.2732	4.20	0.0068	1.58	0.1994	0.12	0.9738
Microsite*year*replicate(location)	36										
Depth	1	14.78	0.0003	42.94	<.0001	24.97	<.0001	43.17	<.0001	111.6	<.0001
year*depth	2	1.04	0.3600	2.48	0.0944	91.55	<.0001	0.6	0.5503	6.04	0.0044
location*depth	1	0.27	0.6085	17.37	0.0001	2.22	0.1424	13.22	0.0006	0.36	0.5515
year*location*depth	2	0.72	0.4897	5.73	0.0058	18.72	<.0001	0.68	0.5131	1.08	0.3481
Microsite*depth	2	4.78	0.0125	0.97	0.3854	10.57	0.0001	5.96	0.0047	21.45	<.0001
year*microsite*depth	4	1.74	0.1544	1.24	0.3047	5.72	0.0007	1.36	0.2592	2.62	0.0455
location*microsite*depth	2	3.13	0.0520	5.21	0.0089	5.54	0.0066	0.68	0.5132	0.02	0.9759
year*location*microsite*depth	4	0.25	0.9091	3.84	0.0086	3.80	0.0088	2.59	0.0471	0.61	0.6566
depth*microsite*year*replicate(location)	52										

		Ca^{2+}		Na^+		Mn^{2+}		Zn^{2+}		$Fe^{2+/3+}$	
Effect	**DF**	**F**	**P**	**F**	**P**	**F**	**P**	**F**	**P**	**F**	**P**
Location	1	21.87	0.0034	38.67	0.0008	0.02	0.9065	3.07	0.1302	2.72	0.1501
replicate(location)	6										
Year	2	0.61	0.5584	0.11	0.8989	7.07	0.0094	3.95	0.0482	1.12	0.3584
year*location	2	0.05	0.9484	0.23	0.7951	2.76	0.1035	3.69	0.0565	2.19	0.1549
year*replicate(location)	12										
Microsite	2	8.49	0.001	15.5	<.0001	3.62	0.0371	11.88	0.0001	3.9	0.0293
year*microsite	4	1.38	0.2592	1.6	0.1962	1.95	0.1238	2.83	0.0386	1.93	0.127
location*microsite	2	0.73	0.4872	6.47	0.004	3.46	0.0424	1.98	0.1525	4.59	0.0168
year*location*microsite	4	0.54	0.7091	0.85	0.5059	0.99	0.423	1.66	0.1806	0.48	0.7473
Microsite*year*replicate(location)	36										
Depth	1	30.16	<.0001	81.1	<.0001	10.24	0.0023	166.3	<.0001	107.4	<.0001
year*depth	2	1.04	0.3597	0.48	0.6193	0.41	0.6674	3.81	0.0285	0.01	0.9897
location*depth	1	7.07	0.0104	26.81	<.0001	11.14	0.0016	0.11	0.7467	0.12	0.733
year*location*depth	2	0.62	0.5424	0.59	0.5602	4.58	0.0148	3.57	0.0353	2.79	0.0708
Microsite*depth	2	7.68	0.0012	2.05	0.1389	5.37	0.0076	12.66	<.0001	4.64	0.0139
year*microsite*depth	4	2.68	0.0419	2.99	0.0271	3.51	0.013	3.45	0.0143	2.18	0.084
location*microsite*depth	2	0.84	0.439	3.05	0.0557	2.44	0.097	1.93	0.1559	2.95	0.061
year*location*microsite*depth	4	1.01	0.4086	2.69	0.0413	1.44	0.2344	1.43	0.2383	0.15	0.9606
depth*microsite*year*replicate(location)	52										

Table 6a. Partial correlation coefficients for canonical variables vs. their partial component variables, and vs. their contrasting partial component variables.

Partial Variables	Canonical Variables					
Lupinus argenteus	**Soil 1**	**Plant 1**	**Soil 2**	**Plant 2**	**Soil 3**	**Plant 3**
Treatment	**0.8644**	0.8206	-0.0140	-0.0121	-0.0454	-0.0377
Surface Mn^{2+}	0.1138	0.1081	-0.4094	-0.3526	0.0631	0.0523
Surface Ca^{2+}	0.4923	0.4673	-0.0483	-0.0416	**-0.3384**	-0.2807
Surface K^{+}	**0.5493**	0.5214	-0.0375	-0.0323	-0.1815	-0.1506
Surface ortho-P	0.0758	0.0720	**0.5382**	0.4635	**-0.3576**	-0.2967
Surface inorganic-N	0.4946	0.4695	-0.0437	-0.0376	-0.0785	-0.0651
Profile Mn^{2+}	-0.4673	-0.4437	-0.3915	-0.3372	0.0856	0.0711
Profile $Fe^{2+/3+}$	-0.3873	-0.3677	-0.3944	-0.3396	-0.0747	-0.0620
Profile Ca^{2+}	0.5319	0.5049	-0.0897	-0.0773	-0.3099	-0.2572
Profile K^{+}	**0.5682**	0.5394	-0.1398	-0.1204	-0.2237	-0.1856
Profile ortho-P	**0.6331**	0.6010	0.2453	0.2113	**0.3847**	0.3192
Profile inorganic-N	0.4197	0.3984	-0.1764	-0.1517	-0.2499	-0.2073
Profile pH	-0.0159	-0.0151	-0.0924	-0.0795	0.3970	0.3294
Plant weight	0.6551	**0.6901**	-0.3707	-0.4304	0.3247	0.3913
Plant Ca	0.4447	0.4684	0.3950	0.4586	-0.1135	-0.1368
Plant K	-0.2862	-0.3014	0.5825	**0.6764**	0.5108	**0.6156**
Plant Na	0.2443	0.2573	0.4152	0.4821	0.0723	0.0872
Plant Fe	-0.3146	-0.3314	0.0127	0.0147	-0.1691	-0.2038
Plant Mn	0.1967	0.2072	0.0389	0.0452	0.3798	0.4578
Plant P	0.5038	**0.5307**	0.6333	**0.7353**	-0.2640	-0.3182
Plant N	0.6980	**0.7353**	0.0144	0.0167	0.1659	0.2000
Crepis acuminata						
Treatment	**0.5260**	0.4952	0.3630	0.3304		
Surface ortho-P	0.1789	0.1684	**-0.6015**	-0.5474		
Surface inorganic-N	0.4079	0.3840	0.1570	0.1429		
Profile Mn^{2+}	**-0.4931**	-0.4642	0.0210	0.0191		
Profile inorganic-N	0.2999	0.3101	0.2857	0.2067		
Profile pH	-0.2159	-0.2033	**0.4829**	0.4395		
Plant weight	0.2020	0.2145	0.4252	**0.4671**		
Plant Ca	0.5543	**0.5888**	-0.1869	-0.2054		
Plant Mg	0.5365	**0.5699**	-0.3041	-0.3342		
Plant K	0.3351	0.3559	0.4014	0.4411		
Plant Na	0.5415	**0.5752**	0.3264	0.3587		
Plant Mn	0.3572	0.3794	0.4354	**0.4784**		
Plant P	0.5548	**0.5893**	-0.6337	**-0.6963**		
Plant N	0.1656	0.1759	0.3504	0.3850		
Eriogonum umbellatum						
Surface Mn^{2+}	**-0.3515**	-0.3503				
Surface ortho-P	**0.3917**	0.3903				
Surface inorganic-N	**-0.3544**	-0.3532				
Surface pH	0.266	0.265				
Profile Mg^{2+}	**0.2977**	0.2967				
Profile ortho-P	0.2512	0.2503				
Profile pH	0.2611	0.2602				
Plant Mg	0.2918	**0.2928**				
Plant K	-0.1454	-0.1459				
Plant Fe	-0.1361	-0.1366				
Plant P	-0.3105	**-0.3116**				

Table 6b. Partial correlation coefficients for canonical variables vs. their partial component variables, and vs. their contrasting partial component variables.

Partial Variables	Canonical Variables					
Eriogonum elatum	**Soil 1**	**Plant 1**	**Soil 2**	**Plant 2**	**Soil 3**	**Plant 3**
Treatment	**0.6057**	0.5967				
Surface K^+	**0.6226**	0.6132				
Surface ortho-P	-0.4072	-0.4011				
Surface inorganic-N	0.3166	0.3118				
Profile Ca^{2+}	0.4141	0.4079				
Profile K^+	**0.5631**	0.5547				
Profile Na^+	**0.5038**	0.4962				
Profile ortho-P	**0.5781**	0.5695				
Profile inorganic-N	0.3192	0.3144				
Profile pH	**0.4612**	0.4543				
Plant weight	0.8392	**0.8266**				
Plant K	0.5747	**0.5661**				
Plant Mn	0.4111	0.4050				
Plant N	0.5512	**0.5430**				
Poa secunda						
Treatment	**0.8767**	0.8498	-0.1794	-0.1623		
Surface Zn^{2+}	0.1793	0.1738	**0.5211**	0.4713		
Surface Ca^{2+}	0.4281	0.4150	0.2884	0.2609		
Surface K^+	0.4459	0.4322	-0.0626	-0.0566		
Surface Na^+	0.4131	0.4005	-0.1730	-0.1565		
Profile Mn^{2+}	-0.4090	-0.3965	0.2663	0.2409		
Profile Zn^{2+}	0.2137	0.2072	0.4055	0.3668		
Profile Ca^{2+}	**0.5697**	0.5523	0.1606	0.1452		
Profile Mg^{2+}	**0.5219**	0.5059	0.0237	0.0214		
Profile K^+	**0.5161**	0.5003	-0.0789	-0.0714		
Profile ortho-P	**0.6653**	0.6449	0.1682	0.1521		
Profile inorganic-N	0.2160	0.2094	-0.1018	-0.0921		
Plant Ca	0.4736	0.4886	-0.0868	-0.0959		
Plant Mg	0.7180	**0.7406**	0.2301	0.2543		
Plant K	0.6978	**0.7199**	-0.1495	-0.1653		
Plant Fe	-0.4818	-0.4970	0.0908	0.1004		
Plant Mn	0.3269	0.3372	-0.4882	**-0.5398**		
Plant N	0.8122	**0.8379**	-0.3219	-0.3558		
Festuca idahoensis						
Treatment	**0.7793**	0.7360				
Surface K^+	**0.6202**	0.5857				
Profile Mn^{2+}	-0.4649	-0.4390				
Profile $Fe^{2+/3+}$	-0.4213	-0.3979				
Profile Ca^{2+}	**0.5779**	0.5458				
Profile K^+	**0.6192**	0.5848				
Profile inorganic-N	0.3503	0.3308				
Profile Na^+	**0.5337**	0.5040				
Plant K	0.4975	**0.5268**				
Plant Fe	-0.2585	-0.2737				
Plant Mn	-0.4446	-0.4708				
Plant Zn	-0.4534	-0.4801				
Plant P	0.4369	**0.4626**				
Plant N	0.7769	**0.8227**				

Table 7. Pearson correlations for the plant nutrients Mg, K, N, and P for each of the study species.

Correlation		*L. argenteus*	*C. acuminata*	*E. umbellatum*	*E. elatum*	*P. secunda*	*F. idahoensis*
Mg vs. K	r =	0.0526	0.1657	0.3853	0.3182	0.7176	0.6106
	P =	0.6776	0.177	0.0141	0.0293	<.0001	<.0001
	n =	65	68	40	47	60	60
Mg vs. P	r =	0.4761	0.4063	0.6798	0.5626	0.3276	0.1675
	P =	<.0001	0.0006	<.0001	<.0001	0.0106	0.2008
	n =	65	68	40	47	60	60
Mg vs. N	r =	0.0864	-0.0727	0.2823	0.2693	0.4328	0.5664
	P =	0.4937	0.5588	0.0776	0.0672	0.0007	<.0001
	n =	65	67	40	47	58	60
K vs. P	r =	0.1493	-0.1719	0.6675	0.0942	0.4413	0.4825
	P =	0.2353	0.161	<.0001	0.5288	0.0004	<.0001
	n =	65	68	40	47	60	60
K vs. N	r =	-0.1473	0.2237	0.7904	0.6531	0.554	0.6233
	P =	0.2416	0.0687	<.0001	<.0001	<.0001	<.0001
	n =	65	67	40	47	58	60
P vs. N	r =	0.2929	-0.1481	0.5042	0.0837	0.0707	0.3888
	P =	0.0179	0.2316	0.0009	0.5758	0.5978	0.0021
	n =	65	67	40	47	58	60

Figure 1.

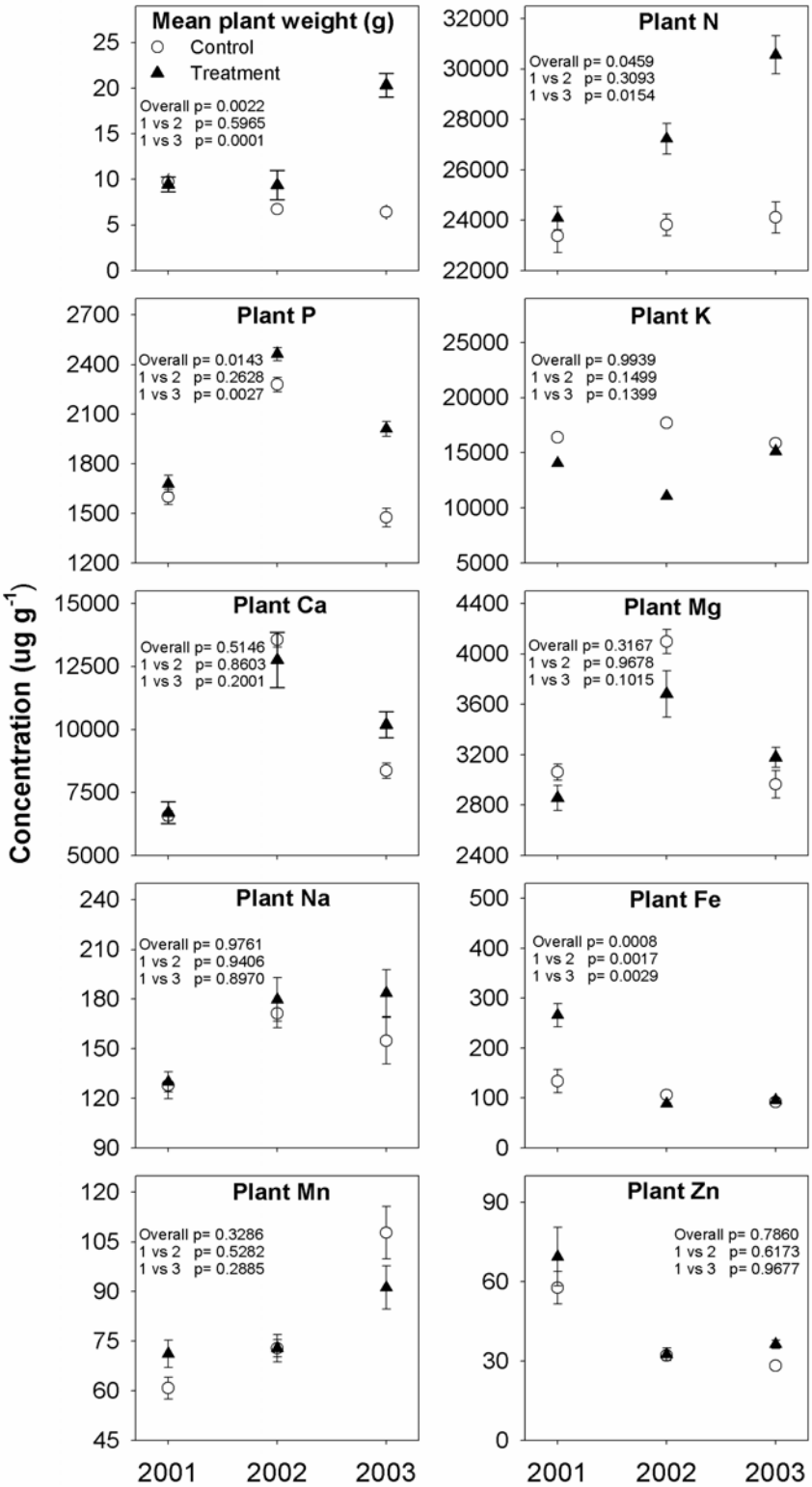

Mean plant weight (g)
Control
Treatment
Overall p= 0.0022
1 vs 2 p= 0.5965
1 vs 3 p= 0.0001
Plant N
Overall p= 0.0459
1 vs 2 p= 0.3093
1 vs 3 p= 0.0154
Plant P
Overall p= 0.0143
1 vs 2 p= 0.2628
1 vs 3 p= 0.0027
Plant K
Overall p= 0.9939
1 vs 2 p= 0.1499
1 vs 3 p= 0.1399
Plant Ca
Overall p= 0.5146
1 vs 2 p= 0.8603
1 vs 3 p= 0.2001
Plant Mg
Overall p= 0.3167
1 vs 2 p= 0.9678
1 vs 3 p= 0.1015
Plant Na
Overall p= 0.9761
1 vs 2 p= 0.9406
1 vs 3 p= 0.8970
Plant Fe
Overall p= 0.0008
1 vs 2 p= 0.0017
1 vs 3 p= 0.0029
Plant Mn
Overall p= 0.3286
1 vs 2 p= 0.5282
1 vs 3 p= 0.2885
Plant Zn
Overall p= 0.7860
1 vs 2 p= 0.6173
1 vs 3 p= 0.9677
Concentration (ug g-1)
2001
2002
2003

Figure 2.

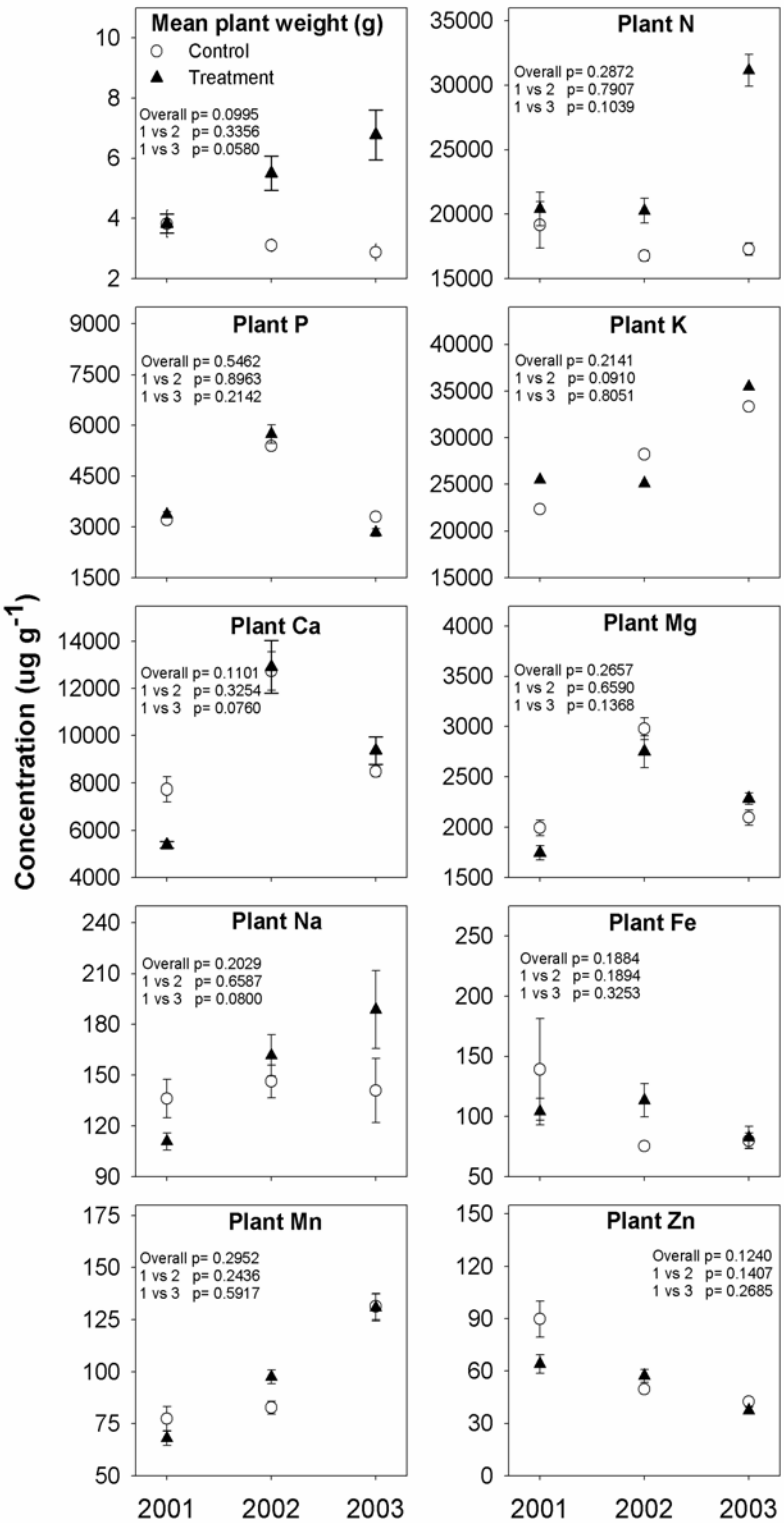

Mean plant weight (g)
Control
Treatment
Overall p= 0.0995
1 vs 2 p= 0.3356
1 vs 3 p= 0.0580
Plant N
Overall p= 0.2872
1 vs 2 p= 0.7907
1 vs 3 p= 0.1039
Plant P
Overall p= 0.5462
1 vs 2 p= 0.8963
1 vs 3 p= 0.2142
Plant K
Overall p= 0.2141
1 vs 2 p= 0.0910
1 vs 3 p= 0.8051
Plant Ca
Overall p= 0.1101
1 vs 2 p= 0.3254
1 vs 3 p= 0.0760
Plant Mg
Overall p= 0.2657
1 vs 2 p= 0.6590
1 vs 3 p= 0.1368
Plant Na
Overall p= 0.2029
1 vs 2 p= 0.6587
1 vs 3 p= 0.0800
Plant Fe
Overall p= 0.1884
1 vs 2 p= 0.1894
1 vs 3 p= 0.3253
Plant Mn
Overall p= 0.2952
1 vs 2 p= 0.2436
1 vs 3 p= 0.5917
Plant Zn
Overall p= 0.1240
1 vs 2 p= 0.1407
1 vs 3 p= 0.2685
Concentration (ug g-1)
2001
2002
2003

Figure 3.

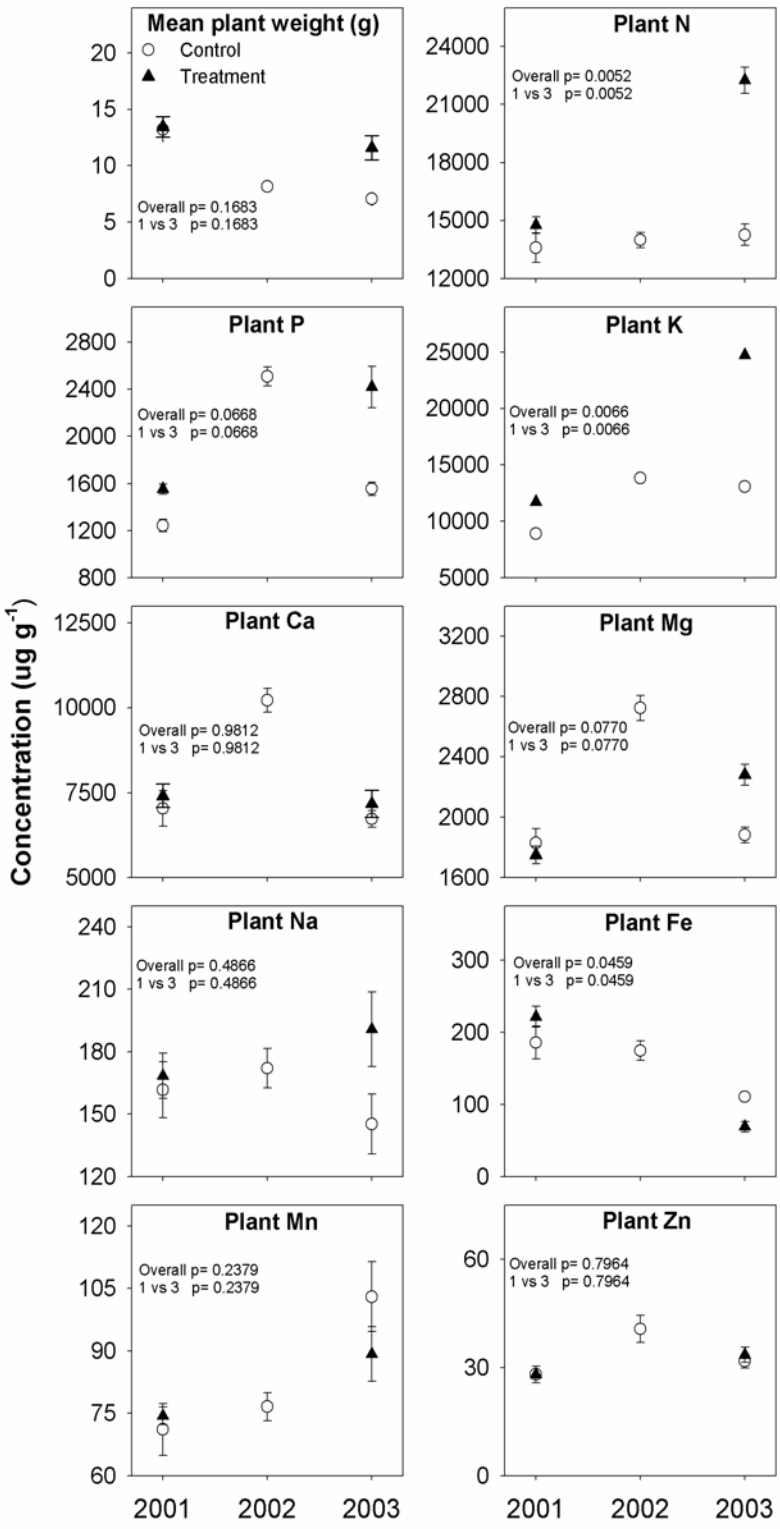

Mean plant weight (g)
Control
Treatment
Overall p= 0.1683
1 vs 3 p= 0.1683
Plant N
Overall p= 0.0052
1 vs 3 p= 0.0052
Plant P
Overall p= 0.0668
1 vs 3 p= 0.0668
Plant K
Overall p= 0.0066
1 vs 3 p= 0.0066
Plant Ca
Overall p= 0.9812
1 vs 3 p= 0.9812
Plant Mg
Overall p= 0.0770
1 vs 3 p= 0.0770
Plant Na
Overall p= 0.4866
1 vs 3 p= 0.4866
Plant Fe
Overall p= 0.0459
1 vs 3 p= 0.0459
Plant Mn
Overall p= 0.2379
1 vs 3 p= 0.2379
Plant Zn
Overall p= 0.7964
1 vs 3 p= 0.7964
Concentration (ug g-1)
2001
2002
2003

Figure 4.

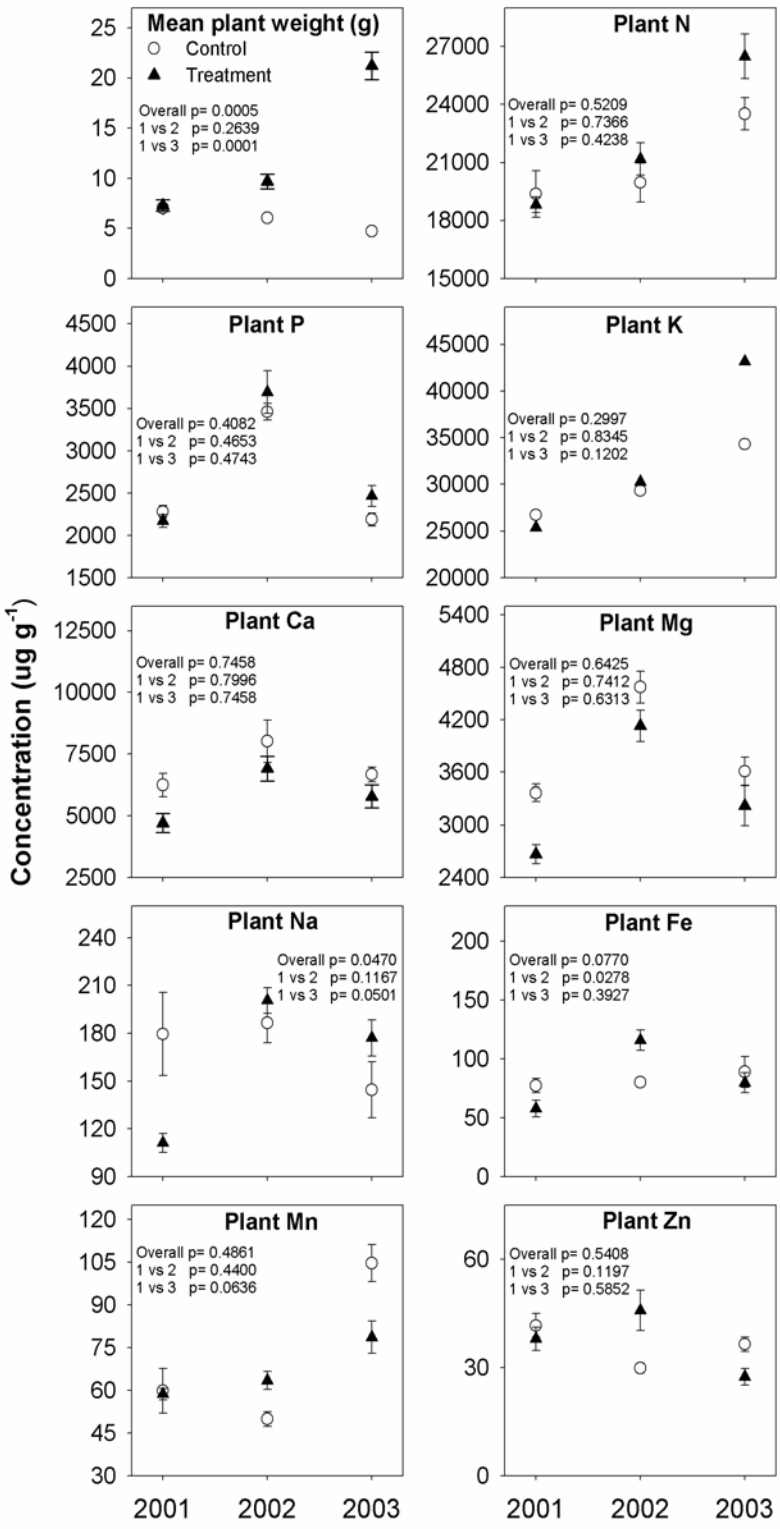

Mean plant weight (g)
Control
Treatment
Overall p= 0.0005
1 vs 2 p= 0.2639
1 vs 3 p= 0.0001
Plant N
Overall p= 0.5209
1 vs 2 p= 0.7366
1 vs 3 p= 0.4238
Plant P
Overall p= 0.4082
1 vs 2 p= 0.4653
1 vs 3 p= 0.4743
Plant K
Overall p= 0.2997
1 vs 2 p= 0.8345
1 vs 3 p= 0.1202
Plant Ca
Overall p= 0.7458
1 vs 2 p= 0.7996
1 vs 3 p= 0.7458
Plant Mg
Overall p= 0.6425
1 vs 2 p= 0.7412
1 vs 3 p= 0.6313
Plant Na
Overall p= 0.0470
1 vs 2 p= 0.1167
1 vs 3 p= 0.0501
Plant Fe
Overall p= 0.0770
1 vs 2 p= 0.0278
1 vs 3 p= 0.3927
Plant Mn
Overall p= 0.4861
1 vs 2 p= 0.4400
1 vs 3 p= 0.0636
Plant Zn
Overall p= 0.5408
1 vs 2 p= 0.1197
1 vs 3 p= 0.5852
Concentration (ug g-1)
2001
2002
2003

Figure 5.

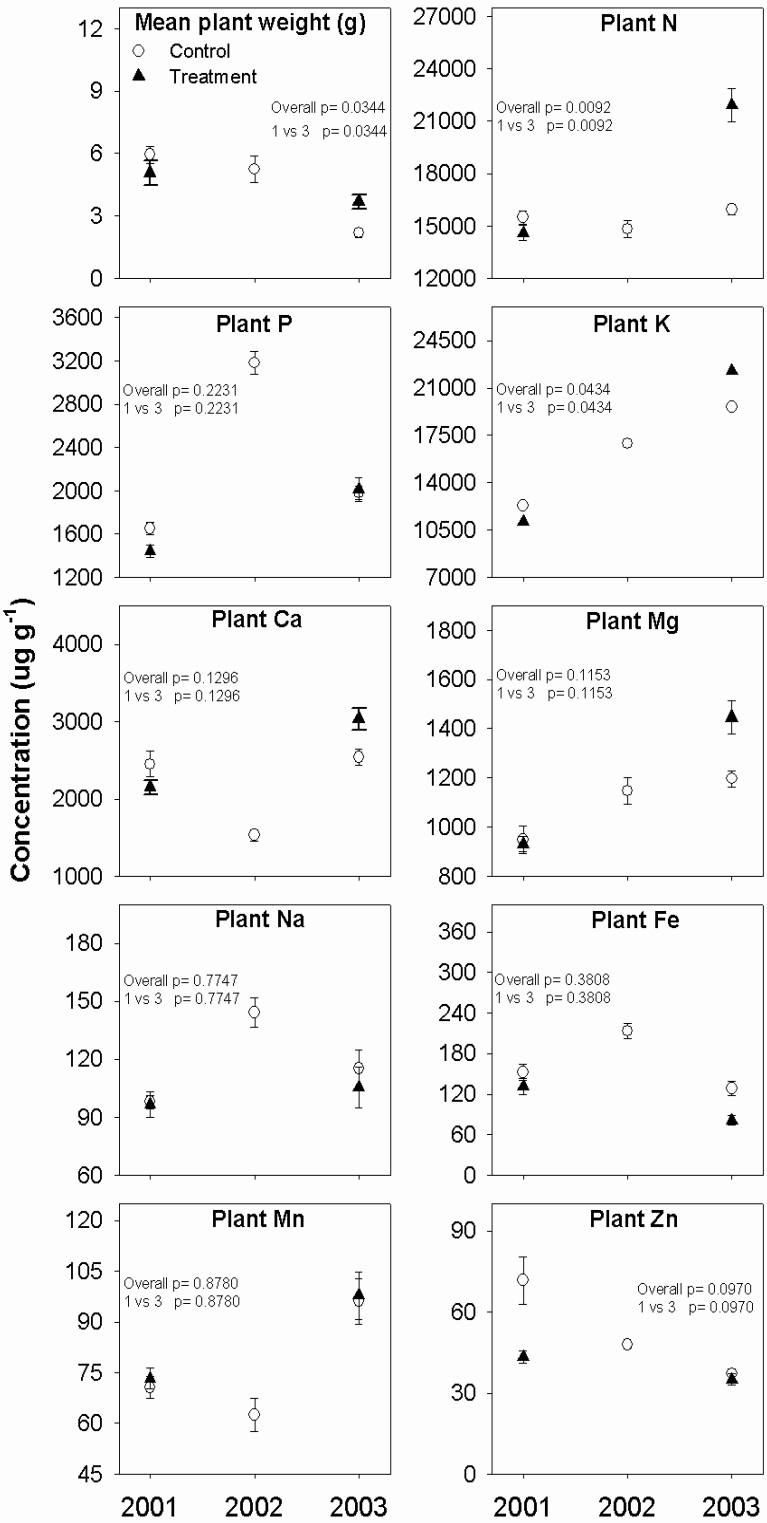

Mean plant weight (g)
Control
Treatment
Overall p= 0.0344
1 vs 3 p= 0.0344
Plant N
Overall p= 0.0092
1 vs 3 p= 0.0092
Plant P
Overall p= 0.2231
1 vs 3 p= 0.2231
Plant K
Overall p= 0.0434
1 vs 3 p= 0.0434
Plant Ca
Overall p= 0.1296
1 vs 3 p= 0.1296
Plant Mg
Overall p= 0.1153
1 vs 3 p= 0.1153
Plant Na
Overall p= 0.7747
1 vs 3 p= 0.7747
Plant Fe
Overall p= 0.3808
1 vs 3 p= 0.3808
Plant Mn
Overall p= 0.8780
1 vs 3 p= 0.8780
Plant Zn
Overall p= 0.0970
1 vs 3 p= 0.0970
Concentration (ug g-1)
2001
2002
2003

Figure 6.

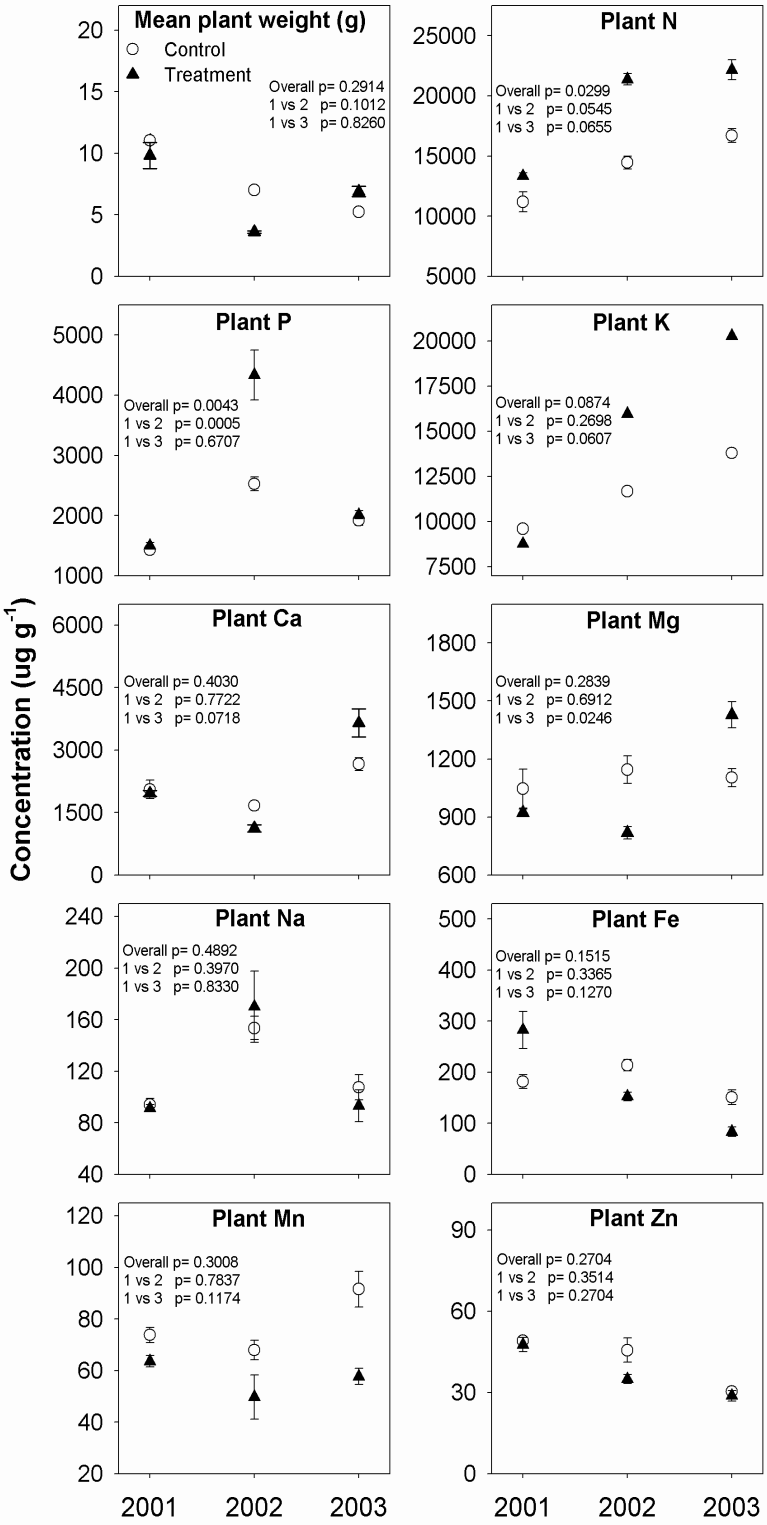
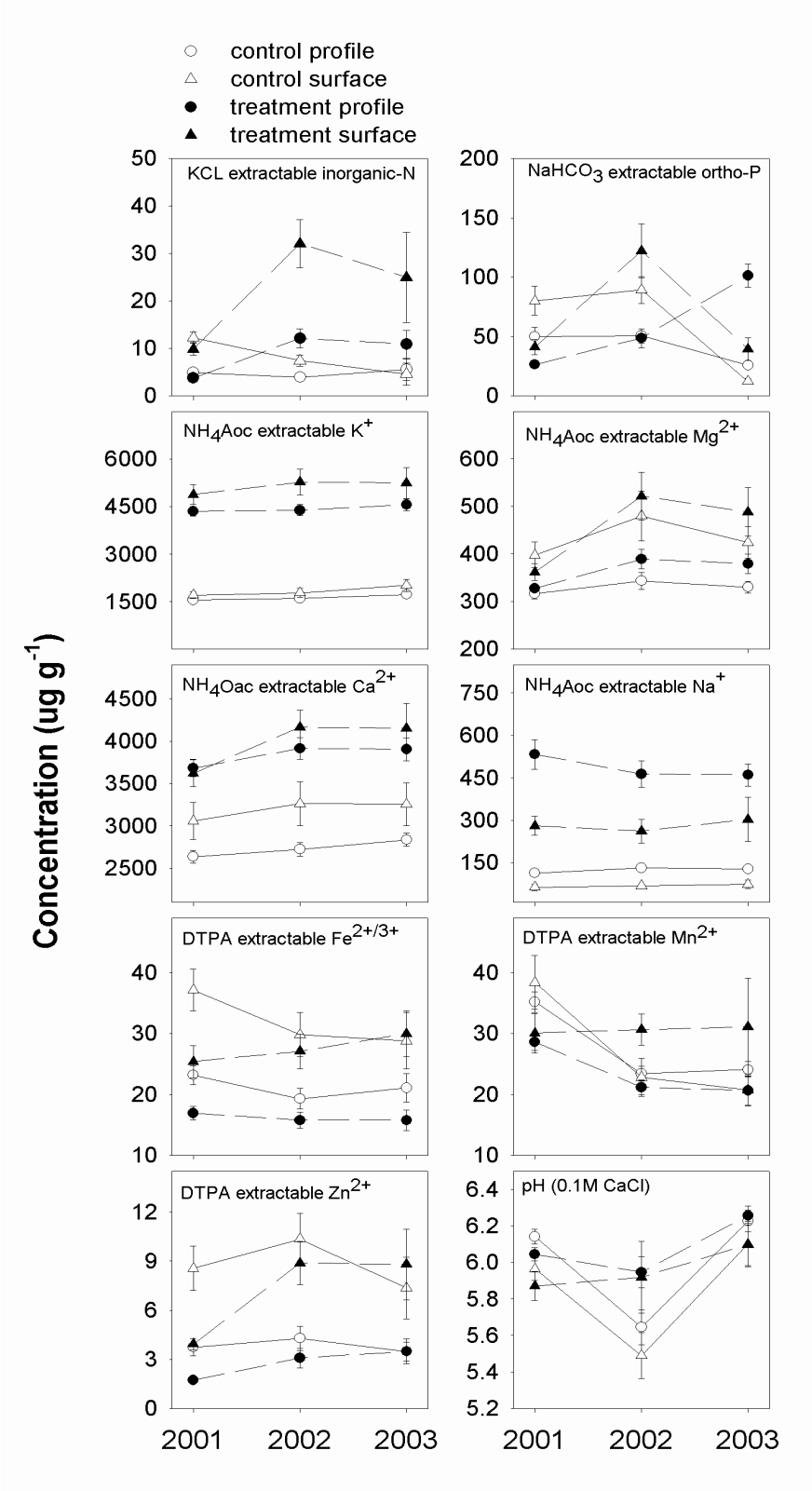

Mean plant weight (g)
Control
Treatment
Overall p= 0.2914
1 vs 2 p= 0.1012
1 vs 3 p= 0.8260
Plant N
Overall p= 0.0299
1 vs 2 p= 0.0545
1 vs 3 p= 0.0655
Plant P
Overall p= 0.0043
1 vs 2 p= 0.0005
1 vs 3 p= 0.6707
Plant K
Overall p= 0.0874
1 vs 2 p= 0.2698
1 vs 3 p= 0.0607
Plant Ca
Overall p= 0.4030
1 vs 2 p= 0.7722
1 vs 3 p= 0.0718
Plant Mg
Overall p= 0.2839
1 vs 2 p= 0.6912
1 vs 3 p= 0.0246
Plant Na
Overall p= 0.4892
1 vs 2 p= 0.3970
1 vs 3 p= 0.8330
Plant Fe
Overall p= 0.1515
1 vs 2 p= 0.3365
1 vs 3 p= 0.1270
Plant Mn
Overall p= 0.3008
1 vs 2 p= 0.7837
1 vs 3 p= 0.1174
Plant Zn
Overall p= 0.2704
1 vs 2 p= 0.3514
1 vs 3 p= 0.2704
Concentration (ug g-1)
2001
2002
2003

Figure 7.


control profile
control surface
treatment profile
treatment surface
KCL extractable inorganic-N
NaHCO3 extractable ortho-P
NH4Aoc extractable K+
NH4Aoc extractable Mg2+
NH4Oac extractable Ca2+
NH4Aoc extractable Na+
DTPA extractable Fe2+/3+
DTPA extractable Mn2+
DTPA extractable Zn2+
pH (0.1M CaCl)
Concentration (ug g-1)
2001
2002
2003

Conclusions

Spring prescribed burning in central Great Basin pinyon-juniper woodlands may be a viable restoration strategy for sagebrush-grasslands if sufficient understory is still in place at the time of burning. The results of these three experiments give us a partial understanding of the consequences of woodland expansion and prescribed burning on soil physical and chemical properties and plant nutritional characteristics.

Infiltration of water into surface soils and $K\theta_s$ in our experiments was influenced by vegetation cover type. Under tree canopy microsites had the highest final infiltration rates and $K\theta_s$ followed by under shrub canopy microsites and interspaces before burning. Final infiltration and $K\theta_s$ was also influenced by variation in soil coarse mineral particles over an elevation gradient typical of central Great Basin woodlands. Soil at lower elevations had higher percentage of coarse mineral particles and higher final infiltration and $K\theta_s$ rates. Coarse mineral particles decreased with increasing elevation as did final infiltration and $K\theta_s$ rates. Burning resulted in increased water repellency of soils at all microsites, and burning decreased final infiltration and $K\theta_s$ rates on under tree canopy microsites with coarse mineral particles greater than sixty percent. The percentage of tree cover did not affect the under tree canopy microsite response to burning, but did affect the percentage of the landscape effected by burn induced water repellency. Burning also decreased final infiltration and $K\theta_s$ rates on under shrub canopy and interspace microsites but only when coarse mineral particles were greater than seventy percent. However, soils with these textures usually have extremely high final infiltration and $K\theta_s$ rates before and after burning as was observed in this study. We believe that the climatic characteristics of the

central Great Basin are such that low intensity spring prescribed burning will not have severe detrimental effects on surface hydrology in expanding woodlands.

Soil chemistry was influenced by vegetation cover type, location, and depth. Soils typically had higher concentrations of mineral nutrients under tree and shrub canopies than in interspaces, but only near the soil surface. Increasing tree cover may change the distribution of nutrients near the soil surface and increase pools of aboveground nutrients. Concentrations of nutrients were typically higher at the soil surface and decreased with depth with the exception of Na^+ which increased with depth. Nutrients were somewhat variable over time, particularly those with strong biological cycles such as ortho-P and NH_4^+. Burning resulted in immediate increases in most soil base cations, pH, micronutrients, NH_4^+, and ortho-P at all microsites, but only near the soil surface. Burning resulted in an immediate small but significant decrease in surface soil NO_3^- and Na^+ near the soil surface. Over longer periods burning resulted in increased NH_4^+ Mg^{2+}, Mn^{2+}, and Zn^{2+} at the soil surface and in the case of Mg^{2+} and NH_4^+ only at under shrub and or tree canopy microsites. Burning also resulted in a longer term increase of NO_3^- and ortho-P through the soil profile. Spring burning appears to create a favorable site for re-sprouting or new vegetation due to increases in available soil nutrients at the soil surface and though the profile.

Plant aboveground biomass and tissue nutrient concentrations are species-dependent, as are their responses to prescribed burning. Plant characteristics varied temporally, but were not highly influenced by microsite. The most common response to burning was increased aboveground plant weight and tissue N concentrations, although increases in tissue P and cations, and a decrease in tissue Fe were also observed. Burning and climate variation

directly influenced soil nutrient availability, and plants appear to respond to these changes. The direct effect of burning, and the increase in soil ortho-P caused by the burn seamed to have the largest impact on increasing plant weight and tissue N. However, we believe that the large fire-induced increases in soil inorganic-N also had a large impact on plant weight and N concentrations.

This study gives a broad background for the possible effects of spring prescribed burning in central Great Basin woodlands. The results of this study may be applicable to areas of similar climatic, physical, chemical, and biological composition within the Great Basin. Pinyon-juniper woodlands in the Great Basin and the rest of western United States cover a broad range of geographic and climatic conditions. We believe that the impacts of spring prescribed burning are beneficial for soils and understory herbaceous vegetation, and that these impacts will occur over a broad range of geography within the Great Basin. The limited impact of spring burning on surface hydrology may not be as widely applicable. Areas where soils, climate, vegetation associations, and fire histories are not similar may not have a similar response to burning as the woodland in our experiment. A large scale series of inventories and experimental burns covering the variation found in western woodlands will help us to better understand the effects of burning under these variable conditions.