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Prescribed fire in a Great Basin sagebrush ecosystem: Dynamics of soil extractable nitrogen and phosphorus

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Abstract

Pinyon and juniper have been expanding into sagebrush (*Artemisia tridentata*) ecosystems since settlement of the Great Basin around 1860. Herbaceous understory vegetation is eliminated as stand densities increase and the potential for catastrophic fires increases. Prescribed fire is increasingly used to remove trees and promote recovery of sagebrush ecosystems. We quantified the effects of prescribed fire, vegetation type, and time following fire on soil KCl extractable nitrogen and $NaHCO_3$ extractable phosphorus in a pinyon–juniper woodland and its associated sagebrush ecosystem immediately before and for 4 years after a spring prescribed burn. Potassium chloride extractable NH_4^+ and total inorganic-N increased immediately following prescribed fire, and extractable NO_3^- decreased immediately after the burn. In the surface layer (top 8 cm), extractable NH_4^+ remained elevated compared to the control through year 2 after the burn. By the first fall post-burn extractable NO_3^- and total extractable inorganic-N increased and remained elevated over the control through year 3 after the burn in the surface layer. For the entire soil profile (52 cm), the burn had no effect on NH_4^+, and the effects on total extractable inorganic-N were no longer significant after year 1. However, NO_3^- remained elevated over the control through year 2 post-fire for the soil profile. Near surface $NaHCO_3$ extractable *ortho*-P increased immediately following fire, and remained elevated through year 2 post-fire. No fire effects were observed for extractable *ortho*-P in deeper horizons. Our data show that plant available nitrogen can remain elevated for extended

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periods following prescribed fire. This can influence regrowth and seedling establishment of native plant species, invasion of exotic plant species and, ultimately, site recovery potential.

Keywords: Burning; Ecosystem succession; Juniper; Pinyon; Soil nutrients

1. Introduction

Soil nutrients in pinyon–juniper woodlands and their associated sagebrush ecosystems are spatially and temporally heterogeneous. Concentrations of N and P are typically highest under shrubs (US) and trees where they accumulate due to litter fall (Chambers, 2001; Covington and DeBano, 1990, pp. 78–86; Jackson and Caldwell, 1993). 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 ecosystems (*Artemisia tridentata* ssp. (Rydb.) Boivin). At higher elevations, sagebrush ecosystems are increasingly influenced by pinyon (*Pinus monophylla* Torr. & Frém.) and juniper (*Juniperus osteosperma* Torr.) expansion. Pinyon and juniper woodlands have expanded their pre-European settlement range in the Great Basin by more than 60% since 1860 due to a combination of climate change, fire suppression, and overgrazing by livestock (Gruell, 1999; Miller and Rose, 1999; Miller and Wigand, 1994). Although pinyon–juniper woodlands have expanded and receded several times over the last 5000 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). 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 that area is expected to double over the next 50 years (Miller and Tausch, 2001). As pinyon–juniper woodlands increasingly dominate sagebrush ecosystems, they compete for available resources and often eliminate most understory vegetation (Reiner, 2004). High-severity wildfires combined with reduced understory vegetation may leave burned areas susceptible to exotic invasive species such as cheatgrass (*Bromus tectorum* L.). 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).

Fire characteristics, fuel properties, and fire return intervals 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 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). High-severity fire has variable effects on soil in sagebrush and pinyon–juniper ecosystems. 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, 1990, pp. 78–86). 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 (Gifford, 1981; Neary et al., 1999).

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 extended effects that prescribed fire has on nutrient levels in pinyon–juniper woodlands and their associated sagebrush ecosystems in the Great Basin. Plant community recovery following prescribed fire could be highly correlated with soil nutrients for several years following burning. Prescribed burning in the Great Basin can be done during spring when, relative humidity, 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 of less than 400 °C consumes less above ground biomass, but often produces substantial increases in available nutrients (Blank et al., 1994, 1996; Covington and DeBano, 1990, pp. 78–86; Covington et al., 1991; DeBano and Klopatek, 1988). 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; Covington et al., 1991; DeBano and Klopatek, 1988; Hobbs and Schimel, 1984). Most studies documenting the effects of prescribed fire on soil nutrients are limited to 1 or 2 years following the burn. We have collected data 1 year before, and four continuous years following a spring prescribed burn in a pinyon–juniper woodland. This type of intermediate to longer term data is needed to effectively link soil processes and plant community succession following fire. We asked three questions: (1) what were the immediate effects of burning on extractable soil N and P in under tree (UT), under shrub and interspace (IS) microsites? (2) What was the duration and distribution of the burn effects? (3) What was the range of temporal variability found in the woodland study area?

2. Materials and methods

2.1. Experimental area

The study is located within a Joint Fire Sciences Program 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 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 soils are extremely coarse grained and have weak to moderate structure.

The vegetation is characterized by sagebrush (*Artemisia tridentata vaseyana*) and single leaf pinyon (*P. monophylla*) with lesser cover of Utah juniper (*J. osteosperma*). Herbaceous species include the grasses, *Poa secunda secunda* J. Presl, *Elymus elymoides* Swezey, *Stipa comata* Trin. & Rupr., *Festuca idahoensis* Elmer, and *Pseudoroegneria spicata* (Pursh)

A. Löve, and the forbs, *Eriogonum umbellatum* Torr., *Eriogonum ovalifolium* Nutt., *Eriogonum elatum* Dougl. ex Benth., *Eriogonum heracleoides* Nutt., *Crepis acuminata* Nutt., *Phlox longifolia* Nutt., *Agoseris glauca* (Pursh) Raf., *Lupinus argenteus* Pursh, and *Penstemon* species. *B. 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^{-1}) to high tree dominance (74% cover, 14,213 kg ha^{-1}) (Reiner, 2004).

2.2. 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 and 2225 m. The site at elevation 2195 m was a control, and the site at 2225 m received a spring burn treatment. Each of the study sites were approximately 4.5 ha. Four sub-plots (30×30 m) were sampled on both the control and treatment sites. Plots were characterized by intermediate tree cover (38% cover, 6722 kg ha^{-1}) at both elevations and contained a mix of trees, shrubs, and interspaces with herbaceous species. 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, and interspace) for each depth using a 10 cm diameter bucket auger. Sampling was conducted the first week of November in 2001–2005 to determine temporal, spatial, and treatment differences in soil available nutrients. A second series of soil samples 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. These samples were collected on the burn treatment site from each microsite on May 11, 2002 immediately before the burn. 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 (air temp $<32\,^{\circ}C$, RH $>15\%$, wind speed $<9\,m\,s^{-1}$, and fuel moisture $\approx 40\%$). Maximum soil temperatures were recorded during the fire using heat sensitive paints on metal strips with a range of 39–788 °C in 28 °C increments. Strips were placed parallel to the soil surface at 0, 2, and 5 cm soil depths at all microsites (Korfmacher et al., 2002).

All soil was brought back to the lab, air dried, and sieved to 2 mm. Sub-samples were analyzed for KCl extractable NH_4^+ and NO_3^-, and $NaHCO_3$ extractable *ortho*-P (Keeney and Nelson, 1982; Olsen and Sommers, 1982; Thomas, 1982). Extractable NH_4^+ and NO_3^- were determined using flow injection, and extractable *ortho*-P was determined using molybedenate-blue chemistry. Data were then transformed into kg ha^{-1} by using the formula:

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

where d is the depth (cm) of the soil horizon, Db the bulk density ($g\,cm^{-3}$) of that horizon, >2 mm% is the volume percentage coarse fragment of that horizon, Conc the nutrient concentration ($\mu g\,g^{-1}$), and F the conversion factor ($0.1\,cm^2\,\mu g^{-1}$).

To evaluate year by treatment differences at the landscape scale, 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 30 m line-intercept transects on each replicate plot (Elzinga et al., 1998). The mass calculated for each microsite was then weighted by the microsites' cover percentage on intermediate tree dominance plots. For the surface 8 cm, $kg\,ha^{-1}$ was calculated as the weighted sum of the three microsites. For the soil profile $kg\,ha^{-1}$ was calculated as the weighted sum of the three microsites and four depths to 52 cm.

2.3. 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 were natural log transformed to meet the assumption that the data were normally distributed. All comparisons were evaluated using SAS™ mixed effects models with repeated measures. Immediate prescribed burn effects on the treatment site were evaluated with treatment as a main effect, microsite as a split-plot within treatment, and depth as a split-split-plot within microsite (Appendix A). Overall differences in available soil nutrients between control and treatment sites, microsites, depths, and years were evaluated by considering treatment as a main effect, microsite as a split-plot within treatment, depth was a split-split-plot within treatment and microsite, and year was a split-split-split-plot within treatment, microsite, and depth (Appendix A). This overall analysis was not ideal for measuring treatment and year effects across the entire study area because mean values for microsite and depth do not necessarily reflect the sum or distribution of these sample locations on the landscape. Therefore, year by site interactions were assessed at the two depth integrals described in the methods above (0–8 and 0–52 cm) by considering year and treatment as main effects (Appendix A). Means comparisons were made with Duncan's test ($p<0.05$) after confirming significant main effects and interactions with the mixed models ($p<0.05$).

3. Results

3.1. Immediate effects of burning by vegetation type

Soil heating was greatest under shrub canopy microsites at the soil surface and at the 2 cm depth. Tree canopy microsites had slightly lower temperatures, and interspace microsite soils were only heated on the surface (Fig. 1).

In the soil samples taken for immediate burning effects, burning produced increases in KCl extractable NH_4^+ at all microsites in our study with the largest increases occurring under shrubs ($p<0.05$) (Fig. 1). Ammonium was higher at the surface 0–3 cm after the fire than in the 3–8 cm depth except for under shrubs where large increases of NH_4^+ also occurred at depth 3–8 cm ($p<0.05$) (Fig. 1). Burning decreased KCl extractable NO_3^- across all microsites ($p<0.05$) (Table 1). The largest losses occurred in the surface 3 cm and under trees ($p<0.05$) (Fig. 1). Because of the large increases of NH_4^+, burning increased total extractable inorganic-N, especially in sagebrush microsites ($p<0.05$) (Fig. 1). Burning increased bicarbonate extractable *ortho*-phosphate across all microsites, and as deep as 8 cm ($p<0.05$) (Fig. 1). Sagebrush microsites had the greatest *ortho*-P before and after treatment ($p<0.05$) (Fig. 1).

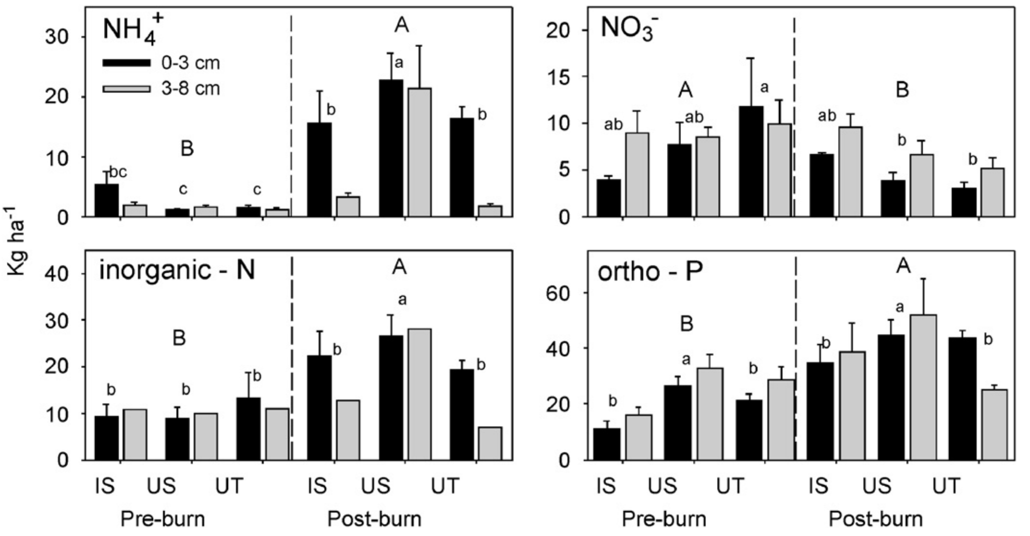

Fig. 1. Means and standard errors for pre- and post-burn soil nutrients on the treatment site at two depths (0–3 and 0–8 cm) and three microsites, interspace (IS), under shrub (US), and under tree (UT). Means not represented by a common letter are significantly different. Capital letters indicate treatment effects. Lower case letters indicate treatment–microsite interactions. Ammonium and inorganic-N both had a significant depth effect with 0–3 > 3–8 cm.

Table 1
Mean ($n = 31$) maximum soil temperatures and SE for surface, 2, and 5 cm soil depths for all three microsites

Microsite	Average temperature (°C)
Surface	
Interspace	206 ± 24
Under shrub	369 ± 33
Under tree	304 ± 26
2 cm	
Interspace	40 ± 0
Under shrub	86 ± 24
Under tree	77 ± 11
5 cm	
Interspace	40 ± 0
Under shrub	40 ± 0
Under tree	44 ± 3

3.2. Temporal variability and long-term burn effects

On unburned control plots near surface (top 8 cm) NH_4^+ trended downward after the first year and was lower in 2003 and 2005 than in the first year 2001. Interspace microsites tended to have more extractable NH_4^+ than tree canopy microsites ($p < 0.05$) (Fig. 2). In the surface soil, KCl extractable NH_4^+ on the burn site was significantly higher than the control in year 1 (2002) post-fire, and although declining over time, remained significantly

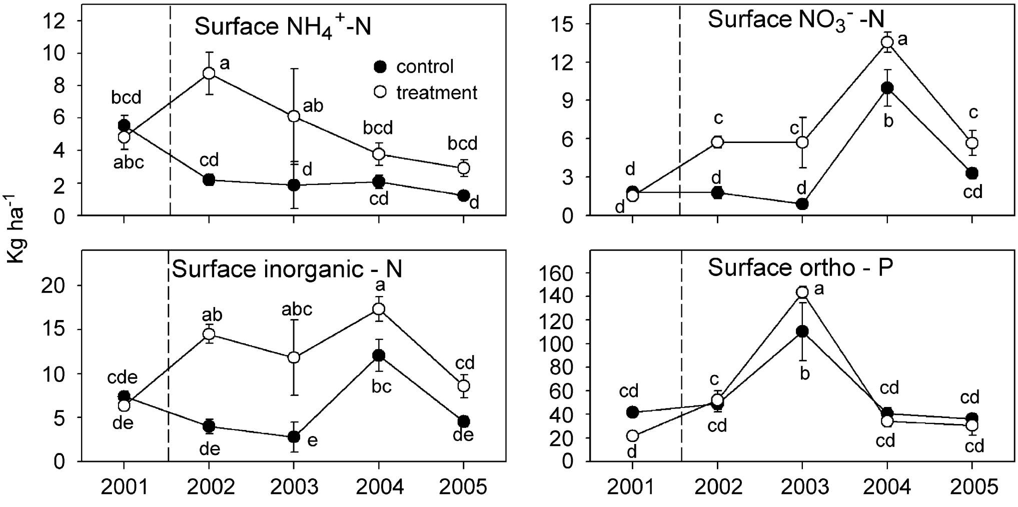

Fig. 2. Means and standard errors for 5 years of pre- and post-burn near surface (0–8 cm) soil nutrients on the control and treatment sites. Means not represented by a common letter are significantly different.

greater than the control through 2 years (2003) post-burn ($p<0.05$) (Fig. 2). Most increases in NH_4^+ occurred near the surface under the shrub and tree canopy microsites at least through year 1 (2002) post-burn ($p<0.05$).

KCl extractable NO_3^- in near surface soil varied considerably over time in both the control and burn treatments. Extractable NO_3^- remained similar on the control site from pre-fire through year 2 (2003) post-fire, then increased sharply in year 3 (2004) post-fire, and returned to previous levels in year 4 (2005) post-fire ($p<0.05$) (Fig. 2). Long-term trends in microsite differences were not significant. Burning caused an increase in extractable NO_3^- compared to the control by year 1 (2002) post-fire, and remained elevated over the control through year 3 (2004) post-fire ($p<0.05$) (Fig. 2). Increased extractable NO_3^- was observed at all microsites following fire ($p<0.05$). The temporal variation observed on the control in year 3 (2004) post-fire was also observed in the treatment site, but differences between burned and unburned were significantly different nevertheless ($p<0.05$) (Fig. 2). The results for total mineral N were similar to those for extractable NO_3^- alone, but burn effects were magnified in 2002 and 2003 by increased NH_4^+ ($p<0.05$) (Fig. 2).

Bicarbonate extractable *ortho*-P in near surface soil was similar in the control and burn sites before burning in 2001 and in year 1 post-fire (2002) ($p<0.05$) (Fig. 2). However, extractable *ortho*-P on the burn site was greater than on the control in year 2 post-fire (2003) ($p<0.05$) (Fig. 2). There was a dramatic increase on both sites in 2003, with levels on the burn site exceeding those measured on the control site ($p<0.05$) (Fig. 2). Long-term trends indicate tree and shrub canopy microsites contain more extractable *ortho*-P than interspace microsites ($p<0.05$).

When soil nutrient contents for the entire sampled profile (0–52 cm) were calculated, results were similar to those for the surface horizon, but burn effects were attenuated. The initial NH_4^+ spike following the fire is noticeable (Fig. 3), but no significant temporal or treatment effects were found for NH_4^+. Burning significantly increased soil extractable

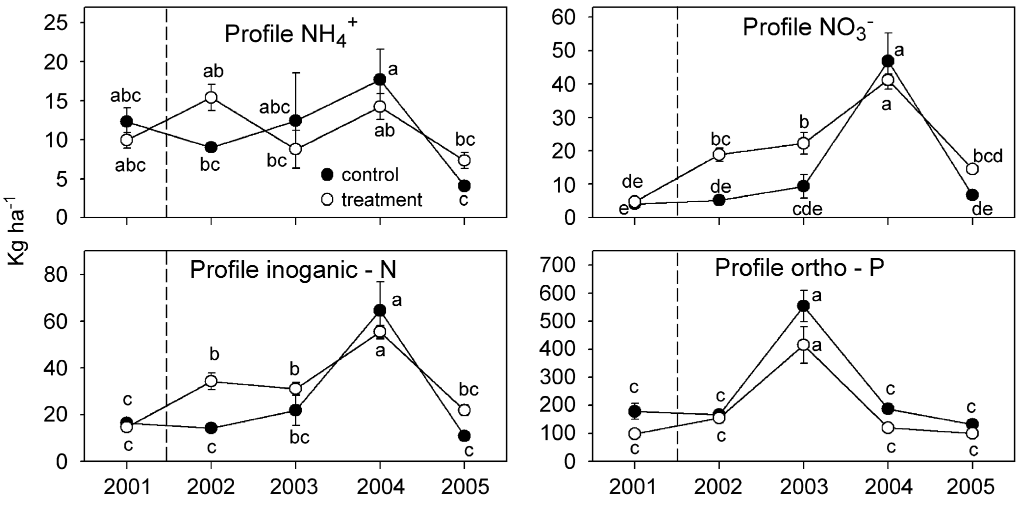

Fig. 3. Means and standard errors for 5 years of pre- and post-burn soil nutrients through the soil profile (0–52 cm) on the control and treatment sites. Means not represented by a common letter are significantly different.

NO_3^- during years 1 and 2 post-fire but by year 3, when the spike in contents occurred, the differences were no longer significant ($p<0.05$) (Fig. 3). Burning caused increased total extractable inorganic-N content in year 1 following the fire ($p<0.05$), but differences were not significant thereafter (Fig. 3). The same temporal spike in 2004 observed for extractable NO_3^- is also apparent in total extractable inorganic-N ($p<0.05$) (Fig. 3). Burn effects on soil extractable *ortho*-P were not significant, but a temporal increase on both sites was observed in year 2 post-fire (Fig. 3).

4. Discussion

4.1. Immediate effects of burning by vegetation type

Burning increased extractable NH_4^+ immediately following the spring prescribed burn on our demonstration site. Most of the increases appeared limited to the first 3 cm of soil except for under sagebrush where increases occurred though 8 cm of soil. Increased NH_4^+ is common following prescribed fire for several semi-arid systems (Neary et al., 1999). Ammonium is released from organic matter during incomplete combustion or from heating (Covington et al., 1991). The pattern observed in our study for increased NH_4^+ is most likely related to patterns of soil heating. We observed that soil temperatures during the fire were elevated highest under sagebrush. The heating patterns are also no doubt related to litter mass and moisture content. Although pinyon pines had litter mats up to 20 cm deep, the moisture content of the mats at the time of the fire was approximately 40%. Most mats were not completely consumed (Rau, 2005; Reiner, 2004), and the unburned litter insulated soil from extensive heating. Sagebrush have much smaller litter mats, (less than 5 cm deep), which were completely consumed, allowing heat to be transferred directly to the soil (Rau, 2005; Reiner, 2004). The interspace microsites had little if any litter and fuel to transfer large quantities of heat into the soil profile (Rau, 2005; Reiner, 2004).

The initial burn-induced losses in NO_3^- observed in our study are not unique in semi-arid systems (Neary et al., 1999). The NO_3^- ion is the most volatile form of nitrogen in soil systems becoming gaseous around 200 °C (Weast, 1988). The pattern of NO_3^- loss is best explained by the initial distribution of NO_3^- (which was greatest under trees) and soil heating which was limited largely to the first 2 cm of soil under trees (Fig. 1, Table 1). Immediate losses in NO_3^- were easily offset by the gains in extractable NH_4^+, resulting in a net gain in total extractable-N following the spring burn.

Burning immediately increased extractable *ortho*-P across all microsites in this study, but this finding is not universal in arid environments (Fig. 1) (Neary et al., 1999). Combustion of organic matter releases phosphorus as ash which can be redistributed by convection during fire (DeBano and Klopatek, 1988). Heating of organic matter also releases phosphorus (DeBano and Klopatek, 1988), and pH increases caused by ash cause the release of *ortho*-P bound with iron and aluminum (Blank and Zamudio, 1998; DeBano and Klopatek, 1988).

4.2. *Temporal variability and long-term burn effects*

The increase in extractable NH_4^+ following fire was largest immediately following the burn, and the subsequent decreases were likely due to uptake by recovering vegetation and nitrification (Covington et al., 1991). Nitrification was somewhat delayed on our sites, however, as evidenced by the fact that NH_4^+ remained significantly elevated on the burn site into year 2 post-fire, and tended to be elevated into year 4. This delay may be due to the arid conditions on the site and the timing of moisture. In the Great Basin, microbial activity is probably greatest during the winter and spring months when most precipitation occurs, but temperatures are low (Agehara and Warncke, 2005; Murphy et al., 1998).

Burn induced increases in extractable NO_3^- were obvious at all microsites by the fall of 2002 and into 2003 through the entire soil profile. In 2004, extractable NO_3^- near the surface remained elevated over controls but the whole profile returned to levels similar to the control. The wide spread and sustained increase in NO_3^- following fire is not surprising given the mobility of the NO_3^- ion and the continued conversion of NH_4^+ to NO_3^- by nitrification. However, the trend for increased NO_3^- 4 years following prescribed fire is rarely if ever documented. We believe these sites are capable of retaining NO_3^- for longer periods due to the arid climate (Salama et al., 1993). Even during spring melt wetting fronts in the soil only extend to 50–70 cm. Therefore, NO_3^- leaching from the system is probably minimal in ‘‘normal’’ years. It is notable on these sites that NO_3^- increased dramatically in year 3 post-fire on both sites and, although treatment differences remained significant in surface soils, these increases were larger in magnitude than the increases initially caused by the fire. Evaluating large temporal spikes in nutrient availability must be done with caution. Because these samples were taken at a single time each year, the antecedent conditions highly influenced the observed nutrient availability. The large increase in NO_3^- in 2004 may be the result of an unusually wet fall when soils were sampled (Table 2). Soil moisture dramatically affects N availability due to changes in nitrification, and nitrification rates are typically higher under moist conditions (Agehara and Warncke, 2005). This indicates that in this system temporal variability may play an equally if not more significant role in nutrient availability than burn treatments, but that fire effects are still discernable despite large temporal climate fluctuations.

Table 2
Total monthly precipitation in centimeters from a NOAA climate station (39°04′N 117°25′W 1996.4 m) located in the Reese River Valley, Nevada 11 km south of Underdown Canyon

Year	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP	OCT	NOV	DEC	Annual
2001	1.8	1.5	1.9	2.7	0.3	0.0	2.4	0.0	0.4	**1.2**	1.5	3.3	17.1
2002	0.4	1.3	1.0	2.3	0.9	0.9	1.4	0.3	1.0	**1.8**	2.2	1.3	14.9
2003	0.4	2.7	1.5	5.2	1.0	0.7	1.1	4.4	0.7	**0.1**	0.4	2.7	21.0
2004	0.6	1.1	0.1	2.7	0.8	1.1	1.5	1.3	1.4	**5.0**	2.1	0.6	18.4
2005	2.5	1.9	0.8	1.3	4.1	0.2	0.6	0.5	0.5	**1.3**	1.0	1.9	16.4

Values preceding sample dates are bolded.

Burning increased total extractable inorganic-N 3 years following fire at the surface and for a year following fire through the soil profile. The treatment trends obviously reflect both initial increases in NH_4^+ and NO_3^- followed by subsequent conversion of NH_4^+ to NO_3^-. The treatment and temporal similarities between total extractable inorganic-N and extractable NO_3^- indicate that over longer periods available NO_3^- dominates inorganic-N dynamics in this system. Total extractable inorganic-N tends to be elevated on the treatment relative to controls through year 4 post-fire. This indicates that burning produced a long-term increase in extractable-N which should be beneficial to vegetation recovering on the site.

There is some evidence that *ortho*-P increased near the surface as a result of burning. It appears as though near surface *ortho*-P was elevated through year 2 post-fire on the burn plots as a result of burning. Also notable for *ortho*-P was the large temporal increase in year 2 (2003) post-fire. This peak is evident on both sites through the profile and is greater in magnitude than any burn effects. This peak in 2003 does not correspond with the peak observed in NO_3^- during 2004. At this time it is not clear what causes this high degree of temporal variability in extractable *ortho*-P, although 2003 was the driest fall in our study. Phosphorus dynamics are not well understood and can be influenced by soil pH and base cation availability (DeBano and Klopatek, 1988). The high variance in phosphorus availability over time may be important for vegetation found on these sites as phosphorus can be limiting if adequate nitrogen is present.

The temporal patterns observed in this study indicate that soil nutrients change at least 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 microbial decomposition, or that climate influences vegetation which alters uptake and influences nutrient availability.

Soils in pinyon–juniper woodlands and their associated sagebrush ecosystems are highly variable in regards to the spatial and temporal distribution of mineral nutrients. As woodlands encroach into semi-arid shrub communities we may 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 increase. Soil available nutrients in woodlands change from one year to the next, due to climatic variability. This complicates the analysis of fire effects, and emphasizes the need for representative controls.

4.3. Conclusions

Burning had immediate, positive effects on extractable NH_4^+, inorganic-N, and *ortho*-P. The initial effects of burning were mostly limited to the surface 3 cm of soil due to high fuel and soil moisture at the time of the burn. Burning also caused increased extractable NH_4^+ and *ortho*-P into year 2 post-fire and increased NO_3^- and total extractable inorganic-N into year 3 post-fire. In the case of NH_4^+ and *ortho*-P, the effects of burning were only observed near the soil surface. Significant spatial and temporal changes occurred for NO_3^- and extractable inorganic-N which displayed large increases throughout the soil profile and at all microsites 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.

Prescribed burning on this site caused both immediate and persistent effects on soil mineral N and P. This study is unique in that we have documented persistent effects up to 4 years following a prescribed burn. This may prove beneficial to vegetation recovering from burns in arid locations where water and nutrients can be scarce. More intermediate and long-term studies are necessary to accurately document nutrient dynamics due to temporal climate changes and burn treatments. These studies will help us better understand the interaction between soil nutrients and vegetative recovery.

Acknowledgments

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Appendix A

See Table A1–A3.

Table A1
Results of the ANOVA comparing immediate treatment effects, microsite, and soil depth

Effect	d.f.	NH_4^+		NO_3^-		Inorganic-N		*ortho*-P	
		F	*P*	*F*	*P*	*F*	*P*	*F*	*P*
Microsite	2	4.16	0.0525	0.13	0.8826	2.55	0.1325	5.99	0.0221
Depth	1	12.49	0.0064	4.58	0.0611	5.59	0.0423	0.32	0.5878
Microsite × depth	2	2.65	0.1246	1.29	0.3227	3.33	0.0827	1.29	0.3207
Replicate (microsite)	9								
Treatment	1	40.83	<0.0001	4.7	0.0446	14.11	0.0016	20.29	0.0003
Treatment × microsite	2	6.83	0.0066	3.21	0.0655	4.57	0.0257	1.18	0.3323
Treatment × depth	1	5.44	0.0322	0.38	0.5436	2.19	0.1569	1.31	0.2688
Treatment × microsite × depth	2	1.07	0.3637	0.37	0.6944	0.61	0.5551	1.34	0.2883
Depth × replicate (microsite)	17								

Table A2
Results for the ANOVA comparing overall differences in available soil nutrients between control and treatment sites, microsites, depths, and years

Effect	d.f.	NH_4^+		NO_3^-		Inorganic-N		*ortho*-P	
		F	*P*	*F*	*P*	*F*	*P*	*F*	*P*
Site	1	0.26	0.6280	7.35	0.0350	3.13	0.1272	8.57	0.0270
Replicate (site)	6								
Microsite	2	7.31	0.0084	3.32	0.0713	5.62	0.0190	9.75	0.0031
Site × microsite	2	2.78	0.0102	8.16	0.0058	6.75	0.0108	0.08	0.9215
Microsite × replicate (site)	12								
Depth	3	4.66	0.0057	2.09	0.0713	3.52	0.0209	6.86	0.0031
Microsite × depth	6	1.50	0.1960	1.14	0.3506	1.45	0.2132	2.15	0.0625
Site × depth	3	7.21	0.0004	1.99	0.1261	3.86	0.0142	2.66	0.0576
Site × microsite × depth	6	3.33	0.0072	1.94	0.0909	2.88	0.0166	1.12	0.3611
Depth × microsite × replicate (site)	54								
Year	2	3.18	0.0144	3.12	0.0023	19.42	<0.0001	58.71	<0.0001
Year × depth	6	3.06	0.0005	1.24	0.2576	1.74	0.0597	1.25	0.2507
Year × microsite	4	3.39	0.0010	3.12	0.0023	2.00	0.0465	1.36	0.2141
Year × microsite × depth	12								
Year × site	2	3.03	0.0184	1.87	0.1166	1.46	0.0827	0.77	0.5461
Year × site × depth	6	1.63	0.0369	1.45	0.1426	1.29	0.2267	0.94	0.5081
Year × site × microsite	4	1.58	0.1306	7.53	<0.0001	4.93	<0.0001	1.14	0.3396
Year × site × microsite × depth	12	2.44	0.0052	1.50	0.0690	1.70	0.2530	0.42	0.9933
Year × depth × microsite × replicate (site)	288								

Table A3
Results of the ANOVA comparing year and treatment effects on two different soil depth integrals

Effect	d.f.	NH_4^+		NO_3^-		Inorganic-N		*ortho*-P	
		F	*P*	*F*	*P*	*F*	*P*	*F*	*P*
0–8 cm									
Treatment	1	13.56	0.0030	19.66	0.0040	22.74	0.0031	<0.0001	0.9738
Replicate (treatment)	6								
Year	4	3.14	0.0349	40.87	<0.0001	6.41	0.0014	51.81	<0.0001
Year × treatment	4	2.43	0.0783	2.69	0.0578	3.29	0.0296	3.04	0.0389
Year × replicate (treatment)	22								
0–52 cm									
Treatment	1	0.00	0.9847	8.08	0.0295	3.52	0.1099	15.06	0.0082
Replicate (treatment)	6								
Year	4	2.69	0.0579	47.68	<0.0001	29.53	<0.0001	47.66	<0.0001
Year × treatment	4	1.31	0.2954	3.14	0.0349	2.82	0.0500	1.04	0.4107
Year × replicate (treatment)	22								

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