

University of Nevada Reno

Assessing Carbon and Nitrogen in a Central Nevada Pinyon Woodland with Tree
Encroachment and Prescribed Fire

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by

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Abstract

Long term climate change, land use history, woodland encroachment, exotic grass invasions, and altered fire regimes is dramatically influencing vegetation and biogeochemical cycles in the Great Basin, U.S. Currently there is a lack of data on carbon pools and fluxes within this region and the closely linked nitrogen cycle. Increasing concern toward global climate change may drive land management policy over the coming decades. It is therefore increasingly important to understand the current state of carbon (C) and nitrogen (N) pools, their trends, and the impact of land management on those pools. This research focuses on those goals by refining methods for estimating belowground C and N, and estimating whole ecosystem C and N associated with woodland encroachment into sagebrush (*Artemisia tridentate spp.*) ecosystems, and the effects of land management, particularly prescribed fire, on ecosystem C and N. We found that our use of a soil coring device to estimate belowground biomass and C and N gives similar estimates to those obtained from quantitative soil pits. However, the core device is more efficient in regards to sampling time and data processing. It is noted that belowground biomass may exceed 1/3 of total biomass in transitional woodlands, and that soil contains more than 80 % ecosystem C and 90 % ecosystem N at low tree cover. As tree cover increases, biomass and C accumulates rapidly on the landscape, and closed canopy woodlands may exceed 60 % of ecosystem C in aboveground biomass, but less than 15 % N. Prescribed burning at low tree cover releases some C and N to the atmosphere, but a portion is incorporated into soil pools. Burning at increasing tree cover results in more C and N lost to the atmosphere and less C and N incorporated into soils.

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Introduction

The sagebrush (*Artemisia tridentata* spp.) steppe is the most expansive shrubland ecotype in the U.S., and is considered to be one of the most threatened ecosystems in North America (Noss et al. 1995). In the Great Basin of the western U.S. dramatic shifts in vegetation are occurring due long term climate change, livestock grazing, rapid population growth, fire suppression, and introduction of exotic annual grasses (Miller and Tausch 2001). Sagebrush steppe ecosystems face declining herbaceous perennial understory components, expansion of pinyon (*Pinus monophylla*, *Pinus edulis*) and juniper (*Juniperus osteosperma*, *Juniperus occidentalis*) woodlands on more mesic sites, and invasion of exotic annual grasses such as Cheatgrass (*Bromus tectorum*) on drier sites (Miller and Tausch 2001; Chambers et al. 2007). Vegetation shifts can have large impacts on biogeochemical cycles in established ecosystems including changes in carbon (C) and nitrogen (N) accumulation and retention (Turner and Lambert 2000). The sagebrush steppe is a fire adapted ecosystem and fire return intervals ranging from 20 – 90 years along with appropriate livestock grazing can maintain healthy perennial understory vegetation and may keep elemental pools stable on the landscape (Miller and Tausch 2001). Currently most C and N in these arid ecosystems is stored in soils (Hooker et al. 2008). However, woodland expansion may increase aboveground C and N pools that are susceptible to loss from higher severity wildfires. In contrast, annual grasses may reduce aboveground C and N pools, but decrease fire return interval again resulting in C and N losses (D'Antonio and Vitousek 1992). Further, conversion to woodland or annual grass dominance could alter below ground interactions related to litter decomposition and alter root exudates and soil biota including micro-organisms (Schlesinger 1977). Changes in

these processes alter the input vs. respiration balance which drives belowground C and N retention (Turner and Lambert 2000). Ultimately, much is still unknown about these semi arid systems and their ability to retain C and N on the landscape. I conducted a series of studies in a pinyon woodland in central Nevada which is representative of the transition from sagebrush to tree dominance. The goals were to identify the mass of C and N stored in several ecosystem components, and to understand how woodland expansion affects C and N pools. Additionally I attempt to better understand how land management activities like prescribed fire influences C and N pools.

The following four chapters outline my efforts to quantify C and N changes in a central Nevada woodland with tree and expansion and following prescribed fire. Chapter 1 focuses on quantifying soil C and N to a depth of 52 cm. It examines the effects of prescribed fire over a six year period and includes pre- and post-fire measures. Chapter 2 evaluates root biomass, C, and N in these systems. Very little is currently known about belowground biomass in sagebrush or pinyon- juniper ecosystems, and current methods are destructive and time consuming. I compare traditional quantitative pit methods with a newer method developed by Don Todd of the Oak Ridge National Laboratory and described by USDA researchers Ponder and Alley (1997). Chapter 3 uses all of the data collected from belowground and aboveground pools on intermediate tree density plots to estimate whole ecosystem C and N pools and how they are affected by prescribed fire. Finally in chapter 4 I use available data from multiple stages of tree encroachment to develop a simple empirically derived spreadsheet calculator which estimates C and N pools with tree cover ranging from 0 to 100 %. The model incorporates prescribed fire as a management option, and quantifies C and N changes at each tree cover after a burn.

This model may eventually prove useful to land managers who are increasingly asked to consider C retention when planning projects.

Ultimately, these data set and series of manuscripts lay a foundation for future work, and will lead to a better understanding of C and N in these semi-arid ecosystems.

Soil Carbon and Nitrogen in a Great Basin Pinyon-juniper Woodland; Influence of Vegetation, Burning, and Time

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Abstract

Much of the Great Basin, U.S. is currently dominated by sagebrush (*Artemisia tridentata* ssp. (Rydb.) Boivin) ecosystems. At intermediate elevations, sagebrush ecosystems are increasingly influenced by pinyon (*Pinus monophylla* Torr. & Frém.) and juniper (*Juniperus osteosperma* Torr.) expansion. Some scientists and policy makers believe that increasing woodland cover in the intermountain western US will create new carbon storage on the landscape; however, little is currently known about the distribution of carbon on these landscapes. This is especially true of below ground pools. Our objectives were to quantify the spatial distribution of soil carbon in expansion woodlands, and to determine prescribed fire's effect on soil C and N. We looked at two treatments (control and burn), three microsites (undertree, undershrub, and interspace), and four soil depths (0-8, 8-23, 23-38, and 38-52 cm). The study was conducted over a six year period with one year pre-fire and five years post-fire data. Results for both carbon and nitrogen were similar, indicating the close relationship between the two elements in this ecosystem. Undershrub microsites had higher soil C and N concentrations than interspace and undertree microsites; however, under tree microsites had higher C:N ratio than interspace and undershrub microsites. Carbon and nitrogen concentration tended to decrease with increasing depth at both control and burn sites. Prescribed burning caused immediate increases in surface soil C and N concentration, but over intermediate to longer periods of time no statistically detectable change in soil C or N content occurred from burning.

Key words: prescribed fire, woodland expansion, carbon storage, spatial heterogeneity

1. 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 (*Artemisia tridentate ssp.* (Rydb.) Boivin) ecosystems. At intermediate 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 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). 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).

A growing concern in forest and rangeland ecosystems is the effects of altered vegetation composition and fire regimes on both carbon and nitrogen balances. Recent data from semi-arid forests and woodlands indicate that changes in stand densities and especially fire regimes have significant and often immediate effects on carbon and nitrogen balances (Norris et al. 2001, Johnson et al. 1998). Catastrophic wildfire can

cause changes in ecosystem C budgets in a single day that overwhelm and supersede many decades of photosynthesis, respiration, and decomposition, setting the ecosystem on a new vector that is very different from the one it was on before the fire (Johnson et al.1998).

The role of fire in ecosystem C changes is complex. Nitrogen is the growth-limiting nutrient in nearly all western ecosystems, and thus has a major effect on the long-term C budgets of these systems. The effects of fire and post-fire vegetation especially N-fixers can have profound, long-term effects on ecosystem C sequestration (Johnson and Curtis 2001).

As pinyon-juniper woodlands increasingly dominate sagebrush ecosystems, they compete for available resources and often eliminate most understory vegetation (Reiner 2004). High intensity wildfires combined with reduced understory vegetation may leave a burned area 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). The new annual grasslands store considerably less carbon than sagebrush steppe or woodland ecosystems (Bradley 2006)

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 cycling in pinyon-juniper woodlands and their associated sagebrush ecosystems in the Great Basin.

We have collected data one year before and several years following a spring prescribed burn in a pinyon woodland. This type of data will give insight to management effects on soil C and N over short to intermediate time periods. Our aims were to determine: 1) how soil C and N varies spatially between microsites and with soil depth a pinyon woodland; 2) the immediate effects of burning on soil C and N; and 3) changes in soil C and N pools over time since burning.

2. Materials and Methods

2.1 Experimental area

The study is 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 intermittent 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 loamy skeletal mixed frigid Typic Haploxerolls.

The vegetation is characterized by sagebrush (*Artemisia tridentata vaseyana*) and single leaf pinyon (*Pinus monophylla*) with lesser cover of Utah juniper (*Juniperus 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. *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 (5% cover, 5,630 kg/ ha) to high tree dominance (86% cover, 115,000 kg/ ha) (Reiner 2004).

2.2 Study design and data collection

The study was a split plot design with repeated measures. The study plots were located on northeast facing alluvial fans at elevations of 2195 m and 2225 m. Each alluvial fan in the study was approximately 2 ha. The plots at elevation 2195 m were a control, and the plots at 2225 m received a spring burn treatment. Four sub-plots were sampled on both the control and burn treatment. Plots were characterized by intermediate tree cover (38% cover, 6722 kg/ ha) at both elevations and contained a mix of trees, shrubs, and interspaces. To characterize the 2195 m control and 2225 m burn treatment plots, 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 A₁ 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 using a 10 cm diameter bucket auger. Sampling was conducted in November 2001 through 2004 and again in 2006 to determine temporal, spatial, and treatment differences in soil carbon and nitrogen. A second set of 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 carbon and nitrogen. These samples were collected on the burn treatment plots from each microsite on May 11, 2002 immediately before the burn. Collection locations 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°C, RH > 15%, wind speed < 9 m•s⁻¹, and gravimetric fuel moisture ≈ 40%). 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. Fire behavior during the prescribed burn was characterized by creeping ground fire with individual and group tree torching. Some short crown runs were also observed. Sustained crown runs were not frequent due to low wind speeds and discontinuous fuels. Soil temperatures were recorded during the fire using heat sensitive paints on metal strips (Korfmacher et al. 2002). Strips were placed at 0, 2, and 5 cm soil depths at all microsites.

All soil was brought back to the lab, dried, and sieved to 2 mm. Soils were then ground in a Wiley[®] mill and analyzed for total carbon and nitrogen concentration using a LECO Truspec[®] CN determinator. In order to look at landscape scale changes in C and N content, data was transformed into kg ha⁻¹ by using the formula

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

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 ($\mu\text{g g}^{-1}$), and F = conversion factor ($1 \text{ g } 1,000,000 \mu\text{g}^{-1} * 1 \text{ kg } 1,000 \text{ g}^{-1} * 10,000 \text{ cm}^2 1 \text{ m}^{-2} * 10,000 \text{ m}^2 \text{ ha}^{-1}$).

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

2.3 Statistical analyses

The Kolmogorov-Smirnov test was used to test for data normality. All data was natural log transformed to meet the assumption that the data was normally distributed. All comparisons were evaluated using SASTM mixed effects models. Overall differences in C and N concentration between control and burn treatment plots, microsites, depths, and years were determined by evaluating 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.1). Immediate prescribed burn effects on soil C and N concentration within the treatment plots 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.2). The overall analysis was not ideal for measuring burn effects across the landscape because means 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 for soil C and N content were assessed

at the two depth integrals described in the methods above (0-8 and 0-52 cm) by treating year and treatment as main effects (Appendix A.3). 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 Distribution of Carbon and Nitrogen

Over the 6 year study period, almost all of the terms in the overall mixed model for carbon were significant at the $P < 0.05$ level (Appendix A.1). Means comparisons revealed that mean soil carbon concentration to depth of 52 cm was higher on the control plots than in burn plots (Figure 1). Carbon concentrations under shrubs were typically higher than at undertree and interspace microsites on both control and burn plots (Figure 1). Carbon was highest near the soil surface and decreased with depth across the all measurements (Figure 1). Along temporal scales carbon was higher in 2002 and 2004 than in 2001 and 2006 (Figure 1). The higher level interaction terms indicate that most of the spatial and temporal variation within the system occurred near the soil surface (Figure 1).

Most terms in the mixed model for nitrogen concentrations were also significant at the $P < 0.05$ level (Appendix A.1). Mean nitrogen concentrations over the six year study period were slightly higher on the control plots than on burn plots (Figure 2). Nitrogen concentrations were higher under shrubs than at interspace and undertree microsites, and N concentrations decreased with increasing depth (Figure 2). Along temporal scales N was slightly higher in 2004 than in 2001 and 2006 (Figure 2). As with carbon the higher

interaction terms with nitrogen indicate that most temporal and spatial variation occurs near the soil surface (Figure 2).

The mean ratio of C and N concentration across all samples was higher on the control than burn plots ($P < 0.05$) (Figure 3). Carbon nitrogen ratios were highest under tree microsites, lower under shrub microsites and lowest at interspace microsites. The C:N ratio of soil tended to decrease with depth to 38 cm (Figure 3). Carbon to nitrogen ratio was highest in 2002 and 2004, lower in 2003 and 2006, and lowest in 2001 (Figure 3). Higher level interactions showed that changes in C:N ratio were evident through the soil profile (Figure 3).

3.2 Effect of burning on carbon and nitrogen

Burning had identical immediate effects on both carbon and nitrogen. Burning resulted in an immediate increase of both carbon and nitrogen concentration at the soil surface (0-3 cm) (Figure 4). However, no change in soil just below the surface (3-8 cm) was observed (Figure 4). Burning did not have a significant interaction with microsite for carbon or nitrogen.

Burning had no statistically significant longer term effect on total soil carbon or nitrogen content at the soil surface (0-8 cm) or to a depth of 52 cm as indicated by the year by site interaction term in the mixed model (Appendix A.3) (Figure 5 & 6). There were no significant temporal influences on surface (0-8 cm) or soil profile (0-52 cm) total C or N content during the study period.

4 Discussion

4.1 Distribution of Carbon and Nitrogen

Over the entire six year study period the control plots had higher carbon and nitrogen concentrations than burn plots. It is unclear exactly why C and N concentrations are different between the two sites, and it is noteworthy that the absolute value of the difference is small. Similarly when C and N contents are compared no site differences are significant (Appendix A.3). The difference in C and N concentration between the two sites appears directly related to differences observed in surface soils below shrub canopies. Most other measurements of C and N concentration are similar (Figure 1, 2). The observed difference could be related to past fire history, vegetation history, nutrient availability, or microclimatic differences which affect shrub productivity, and microbial respiration and biomass (Klopatek et al. 1991, Norris et al. 2001, Hibbard et al. 2003, Johnson et al 2003). We compared nutrient availability on both sites and found that the control site had lower available Ca^{2+} , K^+ , and Na^+ compared to the burn site, but we are unsure how this would affect total soil C or N (Rau et al. 2008). Because we have no data related to fire history or microclimate for each individual site further explanation of site differences would be speculative.

Undershrub microsites had higher C and N concentrations than undertree and interspace microsites, but only near the soil surface (Figure 1, 2). The tendency for highest carbon and nitrogen at undershrub microsites contrasts slightly with observations from data obtained for root biomass in this system (Rau et al. 2009). It was determined that both undershrub and undertree microsites contained 25% more root biomass or approximately 960 kg ha^{-1} more C and 40 kg ha^{-1} more N than interspace microsites (Rau et al. 2009). Although the root biomass accounts for only a small fraction of total C and N at each microsite ($< 2\%$ C and $< 0.5\%$ N) root turnover, incorporation of root exudates, and

incorporation of surface litter into soil C and N are major long term factors affecting soil C and N, these results suggest an interesting dichotomy (Sturges 1977, Jackson et al. 1996). Because this site is considered expansion woodland, and has been most recently dominated by sagebrush grassland, it is possible that the increased root mass at undertree microsites resulting from tree establishment is a recent phenomenon and the processes that result in increased carbon concentration have not had adequate time to produce measurable change in C and N. This may be supported by the C:N ratio of soil observed at the undertree microsite which was greater than undershrub and interspace. If the roots of trees have a higher C:N ratio than shrub or herbaceous roots then decomposition and incorporation of tree root carbon into soils will be delayed.

Nearly identical patterns exist for both carbon and nitrogen distribution through the soil profile to 52 cm (Figure 1, 2). Carbon and nitrogen concentration and C:N ratio generally decrease with depth. However, the C:N ratio below tree canopies decreases less with increasing soil depth compared to undershrub and interspace (Figure 3). The decrease in C, N, and C:N with increasing depth in this system is probably related to litterfall, and possibly past and current rooting density (Sturges 1977, Jackson et al. 1996). The distribution of total C and N corresponds relatively well with the distribution of root biomass below shrubs and interspaces (Rau et al. 2009). However, there is a discrepancy again at the undertree microsite. Root biomass under trees was typically concentrated at lower depth (23-52 cm) near the lithic contact (Rau et al. 2009). More data from additional sites will be needed to further understand how tree encroachment influences soil C and N pools in arid woodlands.

4.2 Effect of burning on carbon and nitrogen

Burning resulted in an immediate increase in both carbon and nitrogen concentration at the soil surface (0-3 cm) (Figure 4). This contrasts with some observations of both prescribed and wildland fire, but is consistent with others (Johnson and Curtis 2001). Fire generally oxidizes organic matter on and near the soil surface driving off C and N as CO, CO₂, NO₂, N₂O, etc. (Johnson et al. 2004). Temperature data from the fire indicates that surface temperatures reached 200-300 °C; hot enough to oxidize carbon and nitrogen. However, temperatures may not been hot enough to oxidize carbon just below the soil surface (Rau et al. 2005). Temperature data and results from chemical analysis confirm that the fire was not hot enough (< 80 °C) to produce significant changes in total soil carbon and nitrogen just below the soil surface (2 cm) (Rau et al. 2005). The result could have been that ash and partially combusted material from above ground biomass was deposited on the soil surface and incorporated into the soil profile. These materials could have contributed the additional C and N observed after burning.

Through the six year study period burning had no statistically detectable influence on total soil carbon or nitrogen content near the soil surface (0-8 cm) or to a depth of 52 cm as indicated by the mixed model. However, data from immediate measurements and close inspection of Figure 4 and 5 suggest burning increased C and N contents to levels similar to the control plots. Because burning only increased C and N within the first 0-3 cm of soil it is likely that this change was not detectable when integrated into the 0-8 cm or 0-52 cm increments. Similarly, the processes that effect larger changes in soil profile total C and N following fire, such as N-fixation, microbial respiration, and incorporation of litter or root materials into soil, may not have had a long enough period to detectably influence landscape scale C and N pools on this site.

A related study from this site observed a large increase in legume cover following the burn, and an increase in extractable nitrogen adjacent to the legume, *Lupinus argenteus* (Goergen and Chambers, submitted). We hypothesized that the large increase in legume cover following the prescribed burn would eventually lead to recovery of N lost from fire or increase N following fire (Johnson et al. 2004). At this time neither of those scenarios has proved true. However, it is possible that over longer periods we may see significant changes in surface soil 0-8 cm N (Figure 5).

4.3 Conclusions

Although this data is only applicable at this location it suggests interesting implications for carbon storage. Current paradigm suggests that as woodlands encroach into arid landscapes, carbon storage on the landscape should increase (Norris et al. 2001, Hibbard et al. 2003). This may be true for aboveground biomass, but may not be applicable for soil carbon. Data from our study on root distribution in this woodland indicate that tree roots and vertical carbon distribution are not synchronous (Rau et al. 2009). It is feasible that tree encroachment in this woodland was relatively recent and uncommon for long periods prior to our study. For these reasons the soil profile C and N does not reflect current vegetation distribution.

Fire is an integral part of semi-arid sagebrush and woodland systems. Years of fire suppression in these landscapes has increased fuel loads and left ecosystems open to exotic invasions. Prescribed fire in these transition woodlands has been shown as an effective way to re-establish native herbaceous understory biomass (Dhaemers 2006). Although prescribed fire releases carbon and nitrogen from litter and aboveground

biomass, this study suggests it may not have a large impact on soil pools, as has been found in other studies (Klopatek et al. 1991, Caldwell et al. 2003, Johnson et al. 2004).

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Appendix A.1 Results of the mixed model comparing soil C and N concentrations and C:N over the 2 treatments, 4 depths, 3 microsities and 5 years.

	<i>df</i>	Carbon		Nitrogen		CN	
		<i>F</i>	<i>P</i>	<i>F</i>	<i>P</i>	<i>F</i>	<i>P</i>
Treatment	1	23.54	0.0028	9.31	0.0225	47.99	0.0004
Replicate (Treatment)	6						
Microsite	2	30.34	<.0001	38.05	<.0001	17.06	0.0003
Microsite* Treatment	2	7.56	0.0075	10.92	0.0020	1.40	0.2846
Microsite*Replicate (Treatment) Error A	12						
Depth	3	248.41	<.0001	252.67	<.0001	70.54	<.0001
Depth* Treatment	3	2.66	0.0574	3.17	0.0316	1.13	0.3450
Depth*Microsite	6	23.74	<.0001	20.23	<.0001	3.57	0.0047
Depth* Treatment *Microsite	6	3.30	0.0077	2.54	0.0309	1.05	0.4053
Depth*Microsite*Replicate (Treatment) Error B	54						
Year	4	9.78	<.0001	5.01	0.0007	14.67	<.0001
Year* Treatment	4	3.99	0.0037	5.34	0.0004	1.80	0.1288
Year*Microsite	8	1.77	0.0838	2.08	0.0382	2.18	0.0296
Year*Depth	12	4.02	0.0002	3.24	0.0015	2.40	0.0162
Year* Treatment *Microsite	8	7.32	<.0001	4.83	<.0001	4.89	<.0001
Year*Microsite*Depth	24	1.29	0.2216	1.16	0.3161	1.52	0.1177
Year* Treatment *Depth	12	2.72	<.0001	2.09	0.0026	1.25	0.2011
Year* Treatment *Microsite*Depth	24	1.63	0.0354	1.77	0.0164	0.80	0.7356
Year*Depth*Microsite*Replicate (Treatment) Error C	262						

Appendix A.2 Results of the mixed model comparing immediate pre and post burn C and N concentrations at 3 microsites and 2 depths.

	<i>df</i>	Carbon		Nitrogen	
		F	P	F	P
Concentration					
Microsite	2	1.10	0.3734	0.62	0.5571
Replicate (Microsite) Error A	9				
Depth	1	42.77	0.0001	32.97	0.0003
Depth*Microsite	2	3.27	0.0855	1.37	0.3022
Depth*Replicate (Microsite) Error B	9				
Treatment	1	10.09	0.0055	11.37	0.0037
Treatment*Depth	1	7.69	0.0130	7.27	0.0153
Treatment*Microsite	2	0.09	0.9117	0.41	0.6674
Treatment*Depth*Microsite	2	0.07	0.9371	0.22	0.8080
Treatment*Depth*Replicate (Microsite) Error C	17				
Content					
Microsite	2	0.05	0.9479	0.23	0.7956
Replicate (Microsite) Error A	9				
Treatment	1	7.99	0.0198	8.2	0.0187
Treatment*Microsite	2	0.29	0.7562	0.74	0.505
Treatment*Replicate (Microsite) Error B	9				

Appendix A.3 Results from the mixed model comparing C and N content in surface soil and through the soil profile on control and burn plots over 5 years.

	<i>df</i>	Carbon		Nitrogen	
		F	P	F	P
Soil Surface					
Treatment	1	2.99	0.1344	0.58	0.4754
Replicate (Treatment) Error A	6				
Year	4	1.52	0.2268	1.74	0.1736
Year* Treatment	4	0.50	0.7393	0.42	0.7936
Year*Replicate (Treatment) Error B	24				
Soil Profile					
Site	1	2.99	0.1344	0.58	0.4754
Replicate (Treatment) Error A	6				
Year	4	1.52	0.2268	1.74	0.1736
Year*Site	4	0.50	0.7393	0.42	0.7936
Year*Replicate (Treatment) Error B	24				

Figure 1 Means and standard errors for the concentration of soil carbon at three microsites (interspace, undeshrub, and undertree), four depths (0-8, 8-23, 23-38, 38-52), and five years.

Figure 2 Means and standard errors for the concentration of soil nitrogen at three microsites (interspace, undeshrub, and undertree), four depths (0-8, 8-23, 23-38, 38-52), and five years.

Figure 3 Means and standard errors for the soil C:N ratio at three microsites (interspace, undeshrub, and undertree), four depths (0-8, 8-23, 23-38, 38-52), and five years.

Figure 4 Means and standard errors for pre- and post-burn soil carbon and nitrogen concentration on the treatment plots at two depths (0-3 and 0-8 cm). Means not represented by a common letter are significantly different. Capital letters indicate treatment effects. Lower case letters indicate treatment-depth interactions.

Figure 5 Means and standard errors for five years of pre- and post-burn near surface (0-8 cm) and soil profile to 52 cm total soil carbon. Means not represented by a common letter are significantly different.

Figure 6 Means and standard errors for five years of pre- and post-burn near surface (0-8 cm) and soil profile to 52 cm total soil nitrogen. Means not represented by a common letter are significantly different.

Figure 1.0

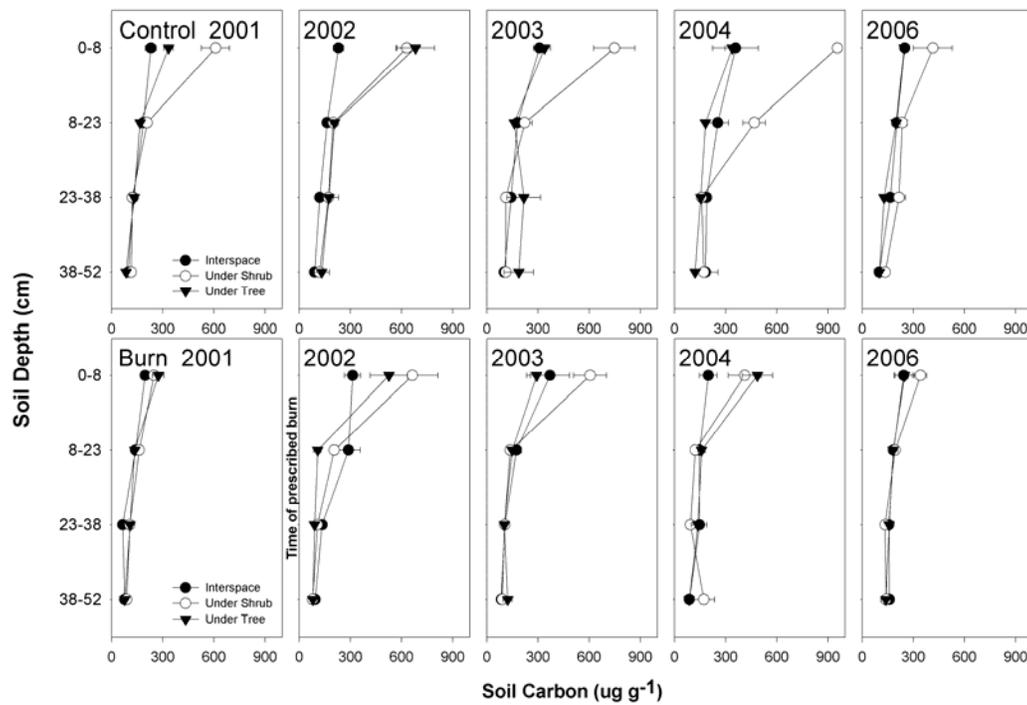


Figure 2.0

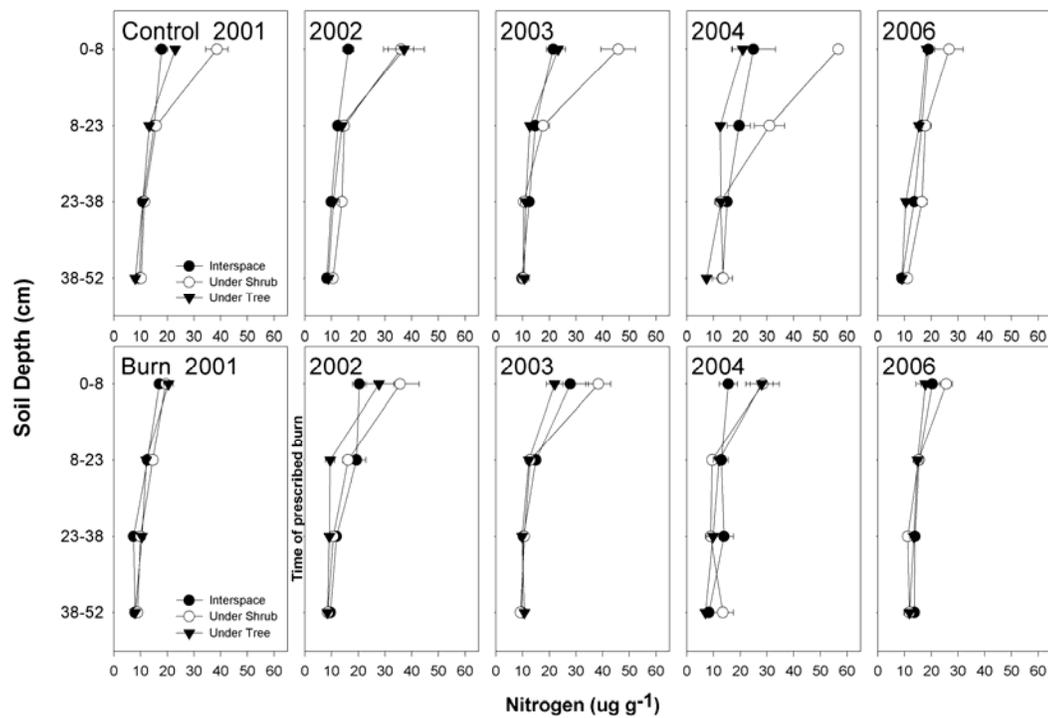


Figure 3.0

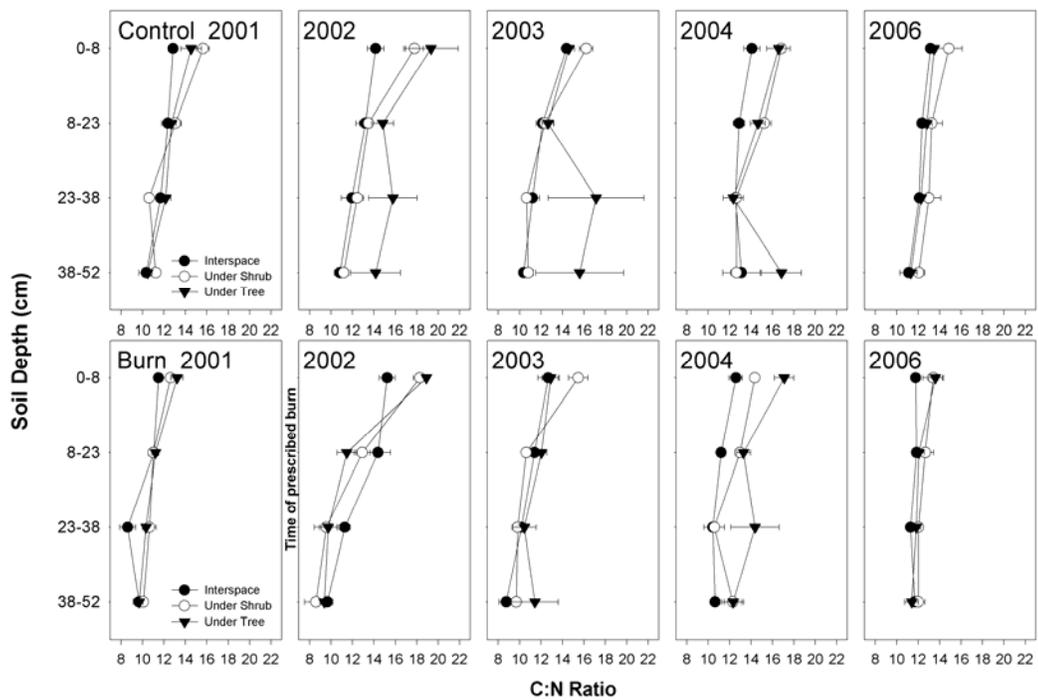


Figure 4.0

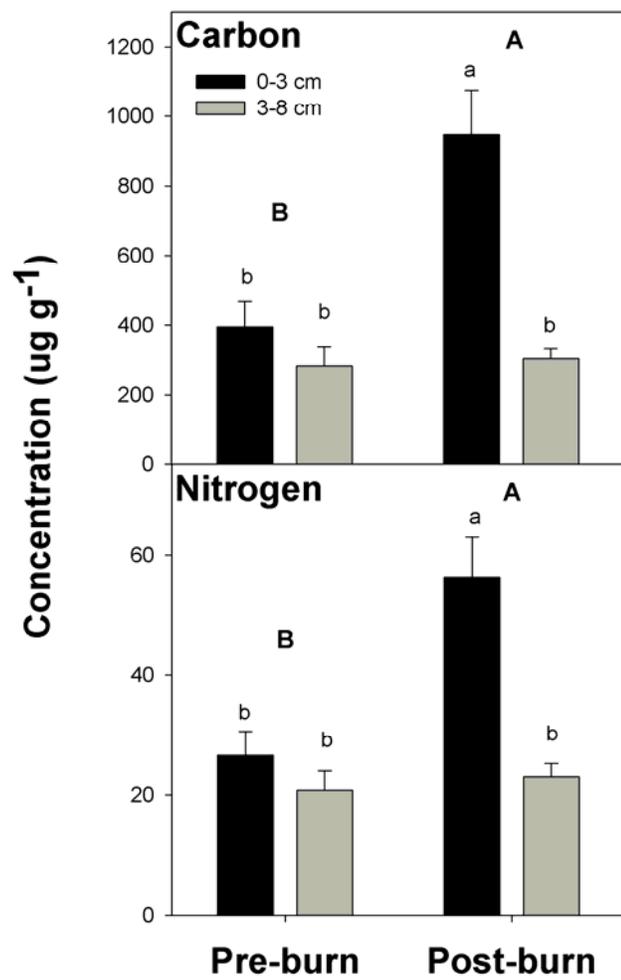


Figure 5.0

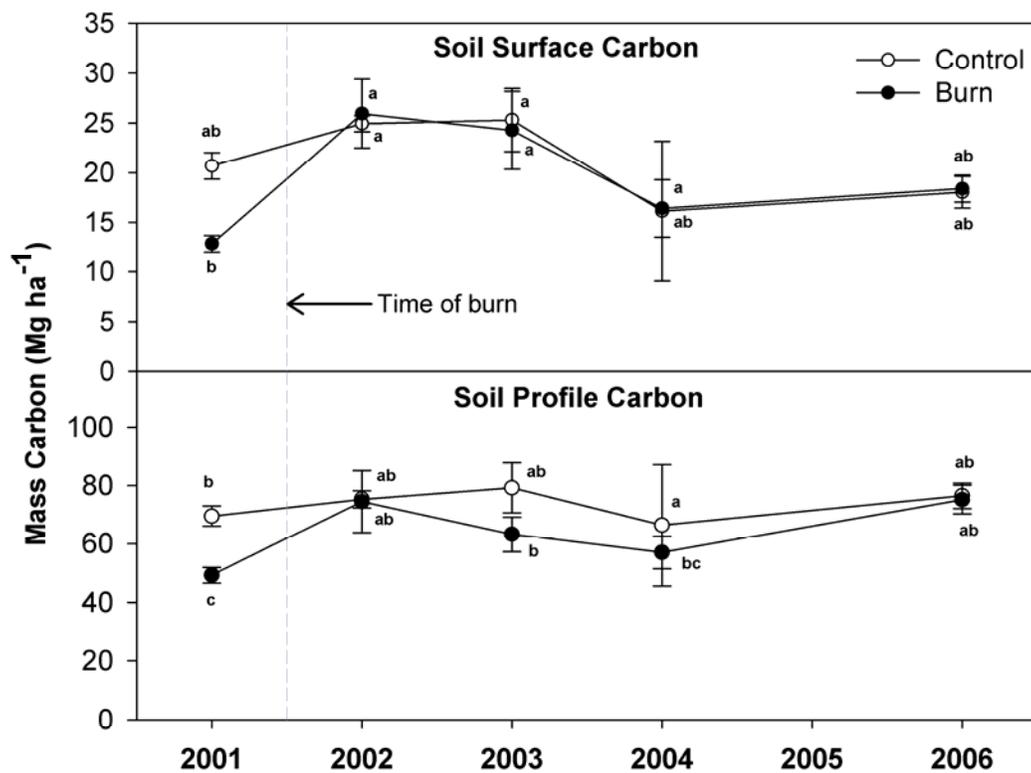
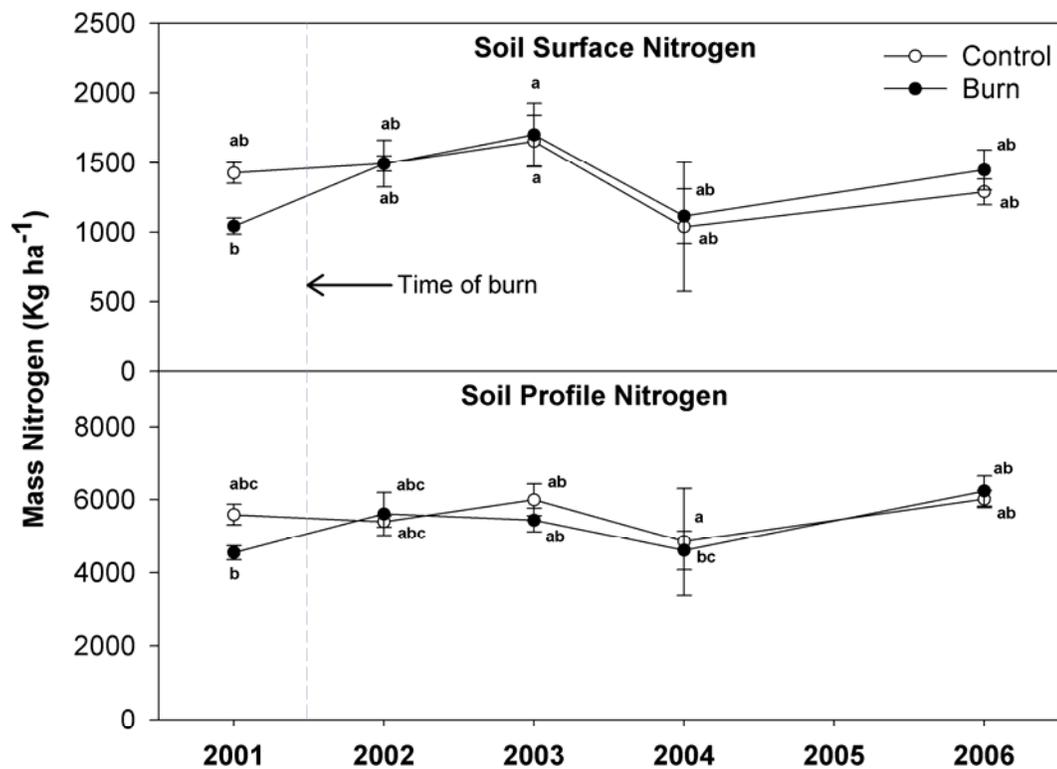


Figure 6.0



Estimating Root Biomass and Distribution after Fire in a Great Basin Woodland Using Cores and Pits

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Abstract

Quantifying root biomass is a critical component of estimating and understanding ecosystem net primary production, biomass partitioning, and carbon storage. We compared a new soil coring technique with traditional quantitative pits for determining root biomass. We conducted the study in an existing Joint Fire Sciences Demonstration Area in the central Great Basin that is representative of a shrub (sagebrush) ecosystem exhibiting tree (pinyon and juniper) expansion. The Demonstration Area had a prescribed burn implemented four years prior to our study, and we sampled both control and burned plots. The samples were stratified across three microsites (interspace, shrub, and tree) and four soil depths (0-8, 8-23, 23-38, and 38-52 cm) to determine the effects of plant life form and burning on root biomass. We found that total root biomass estimates were similar for quantitative pits and our new soil core. However, cores tended to show a more even distribution of root biomass across all microsites and depths than did pits. Overall results indicated that root biomass differs significantly among microsites and soil depths, and that the amount of root biomass at a given depth differs among microsites. Burning reduced root biomass in our study by 23 percent and altered the spatial distribution of root mass.

Introduction

Quantifying root biomass is a critical component of estimating and understanding ecosystem net primary production, biomass partitioning, and carbon storage. In heterogeneous landscapes such as arid and semi-arid shrublands exhibiting tree expansion quantifying root biomass is necessary for understanding the changes that occur in belowground biomass with increasing tree dominance. It also is necessary for understanding effects of both natural and managed fire on ecosystem C storage. Little data exist on root biomass mostly because of the difficulty in making accurate measurements. The two most common methods are excavation of large soil pits and extraction of soil cores.

Excavating soil pits is a labor intensive process but has been used to quantify soil nutrient and biomass pools. However, excavating soil pits can be challenging when sampling rocky soils (Hamburg 1984, Johnson et al. 1991). In soils that contain large rock fragments, vertical wall pits are difficult to excavate. The pit method suffers from the assumption that estimates of soil bulk density taken from small samples are applicable to the entire soil regolith which may contain large rock fragments. Estimating the pit's total volume is difficult because of large rocks intruding into the side of the pit. Alternative methods for measuring pit volume have been suggested, such as filling pits with ping pong balls, sand, or water, but each has limitations. Supplying a large volume of water is extremely difficult in remote or difficult to access areas. The same is true for sand. Ping pong balls are lighter and easier to transport, but may be difficult to level at the soil surface or on slopes, and may not pack uniformly. With quantitative pits, frequently the entire monolith is not sampled

due to the extensive effort and cost. Consequently a large amount of error is introduced into estimates by the necessity to process sub-samples, make moisture corrections to field measurements, and make extensive back calculations to obtain pit volume and root biomass. These measurements are not particularly complex, but assume that sub-samples are uniform, and can be extrapolated to the entire pit.

Soil cores have received limited use in measuring root biomass because of the devices currently available for coring. Traditionally, cores were large truck mounted impact or rotary devices or small hand driven hollow rods which could not be used to collect large diameter roots, and could not penetrate past large coarse fragments (Vogt and Persson 1991). For this study we chose a newer soil core device (Ponder and Alley 1997; Don Todd. USDOE Oak Ridge National Laboratory, personal communication). The device uses a rotary core drill that is diamond tipped and designed for reinforced concrete applications. It is powered by a 6.5 hp two person rotary power head which delivers 120 ft lbs of torque at the output (Figure 1). The core allows for relatively easy sampling of rocky soils and removes most large roots and rocks in its path (Figure 1). This study was part of a Joint Fire Sciences Demonstration Project designed to examine the use of prescribed fire as a tool to slow pinyon and juniper tree expansion and maintain sustainable sagebrush ecosystems. Our objectives were to: 1) test the new soil coring device, and compare our results to traditional soil pit methods; and 2) determine if prescribed burning has an effect on the distribution of root biomass.

Methods

Experimental Area

The study is 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 varies in elevation from 2,072 m to 2,346 m. 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 (35 miles from the site) 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 are classified as coarse loamy mixed frigid Typic Haploxerolls (Rau et al. 2005). The soils are extremely coarse grained and have weak to moderate structure.

Vegetation is characterized by sagebrush (*Artemisia tridentata vaseyana*) and single leaf pinyon (*Pinus monophylla*) with lesser cover of Utah juniper (*Juniperus osteosperma*). Herbaceous species include grasses, *Poa secunda secunda* J. Presl, *Elymus elymoides* [Swezey](#), *Stipa comata* Trin. & Rupr., *Festuca idahoensis* [Elmer](#), and *Pseudoroegneria spicata* ([Pursh](#)) [A. Löve](#), and 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. *Bromus tectorum*, a common invasive annual grass in the region, 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).

Study Design and Data Collection

Two \approx 4 ha treatment plots were established in summer 2001 on northeast facing alluvial fans at elevations of 2195 m and 2225 m. The plot at elevation 2195 m was an unburned control, and the plot at 2225 m received a spring burn treatment. Four 0.1 ha sub-plots were evenly distributed in both control and burn plots and contained a mix of trees (20 % cover), shrubs (34 % cover), and interspaces (46 % cover) (Figure 2). Percent surface cover by microsite was measured using three 30 m line-intercept transects on each replicate plot (Elzinga et al. 1998).

Soil pits were dug at each microsite to a depth of 100 cm, and the soil horizons were identified. Depth increments for sampling included the approximate depth of the soil A₁ horizon and subsequent 15 cm increments (0-8, 8-23, 23-38, and 38-52 cm). The final increment ending at 52 cm was chosen because it corresponded with the transition to the C horizon which is dominated by large alluvial material.

USDA Forest Service fire personnel burned the treatment plots on May 11-14, 2002 (Air temp $< 32^{\circ}\text{C}$, RH $> 15\%$, wind speed $< 9 \text{ m}\cdot\text{s}^{-1}$, and gravimetric fuel moisture $\approx 40\%$). 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. Fire behavior during the prescribed burn was characterized by creeping ground fire with individual and group tree torching. Some short crown runs also were observed. Sustained crown runs were not frequent due to low wind speeds and discontinuous fuels.

Soil Pits

For this study, eight total sub-plots were sampled in fall of 2005 to collect root biomass; four on both the control and treatment sites (Figure 2). Twenty four soil pits were excavated, one at each of three microsites (interspace, under shrub, under tree) within each sub-plot on both the control and treatment plots (Figure 2). Pits were located 1 m from tree boles (under tree), centered on shrub boles (under shrub), and evenly spaced between shrubs and trees (interspace). Individual pits measured 50 x 50 x 52 cm and were excavated in four consecutive depth increments (0-8, 8-23, 23-38, and 38-52 cm) for a total of 96 samples. A 100 cm³ bulk density sample was collected from each of the four depth increments using a impact sampler, and all material from each depth increment was removed from pits and sieved to 12 mm. Roots were manually separated from rocks > 12 mm. The soil and rock fractions were weighed in the field using a Pesola[®] spring scale.

Sub-samples of less than 12 mm soil from each depth increment weighing approximately 2 kg were returned to the lab weighed and sieved to 2 mm. Roots were separated from rocks by floatation. Roots were then dried at 60° C for 24 hours or until they no longer lost mass and a final weight was recorded. The volume of the soil sub-sample was estimated using bulk density measurements, sample mass, and a moisture correction derived from a separate soil sub-sample dried at 100° C for 24 hours or until the sample no longer lost mass. Sub-sample volume was used to calculate root density less than 12 mm but greater than 2 mm. This rooting density was multiplied by the total volume of soil from each pit depth increment to estimate total root mass less than 12 mm, but greater than 2 mm. This mass was added to the

root mass greater than 12 mm to obtain total root mass greater than 2 mm for each depth increment (Figure 3). Root weights were corrected for embedded mineral fraction by ashing samples at 500° C for four hours.

Total pit volume was calculated for each depth increment by adding the estimated > 12 mm rock volume ($> 12 \text{ mm rock mass} / 2.60 \text{ g cm}^{-3}$), the < 12 mm soil volume ($< 12 \text{ mm soil mass moisture corrected} / \text{Db}_{\text{soil}}$), and > 12 mm root volume ($> 12 \text{ mm dry root mass} / \text{Db}_{\text{root}}$).

Soil Cores

Twenty four soil cores were extracted one at each microsite (interspace, under shrub, under tree) and sub-plot on control and treatment plots using a method similar to the one described by Ponder and Alley (1997). The method uses a 7.62 cm diameter diamond tipped core device manufactured by Diteq™. The device is driven by a two person, rotary, Briggs and Stratton™ power head allowing it to core through rocks and soil with minimal compaction (Figure 1). This auger differs very little from the device described by Ponder and Alley (1997). However, we were able to successfully use the diamond tipped core drills which they reported as unreliable. Four soil samples corresponding to the depth increments excavated in pits were removed from each bore hole for a total of 96 samples. Each sample increment was extracted before the core was augered to the next depth increment. This methodology should help to further minimize compaction of each depth increment, but could result in some soil from upper layers being incorporated into lower cores. Cores were bagged individually, brought back to the lab, dried at 100° C for 48 hours, and weighed. Cores were then sieved to 2 mm. Roots were separated from rocks by

flotation, dried again at 60° C, and weighed to obtain total root mass > 2 mm for each depth increment (Figure 4). Root weights were corrected for embedded mineral fraction by ashing samples at 500° C for four hours.

Statistical Analyses

All data were natural log transformed to meet the assumption that the data were normally distributed. All comparisons were evaluated using SASTM mixed effects models. Treatment, microsite, depth, and sample type differences were evaluated using treatment as a main effect, microsite as a split-plot within treatment, depth as a split-plot within treatment and microsite, and sample type as a split-plot within treatment, microsite, and depth (Table 1). This overall analysis was not ideal for measuring treatment and microsite effects across the entire study area because mean values for microsite and depth do not necessarily reflect the coverage of these sample locations on the landscape. Therefore, a second set of comparisons was made using the sum of root biomass through the soil profile. To evaluate overall treatment differences on root mass, the mass calculated for each microsite was weighted by the microsites' cover percentage on the study plots. Treatment was evaluated as a main effect, microsite was a split-plot within treatment, and sample type was a split plot within treatment and microsite (Table 2). Mean comparisons were made with Tukey's test ($p < 0.05$) after confirming significant main effects and interactions with the mixed models ($p < 0.05$).

Results

Pits vs. Cores

There was no difference between the mean value of root biomass per sample increment for soil pits ($2,206 \text{ kg ha}^{-1} \pm 205$) vs. soil cores ($2,324 \text{ kg ha}^{-1} \pm 171$), and no difference between the sum of root biomass through the soil profile for soil pits ($8,828 \text{ kg ha}^{-1} \pm 499$) vs. soil cores ($9,297 \text{ kg ha}^{-1} \pm 806$). However, cores tended to show a more uniform vertical distribution of roots than pits at each microsite measured as indicated by the sample type-depth and sample type-microsite-depth interactions (Table 1, Figure 5).

Spatial Distribution of Roots

The sum of root biomass differed across microsites with tree ($9,878 \text{ kg ha}^{-1} \pm 803$) and shrub ($9,875 \text{ kg ha}^{-1} \pm 888$) microsites having more total root biomass than interspace microsites ($7,514 \text{ kg ha}^{-1} \pm 826$) ($P < 0.05$) (Table 2). Root biomass across all microsites typically decreased with depth ($P < 0.05$), but depth patterns varied by microsite as indicated by the microsite-depth interaction (Table 1). Interspace microsites had most roots concentrated in the top 8 cm of soil; shrubs had most roots within the first 23 cm, and trees concentrated root biomass in the 23-38 cm soil layer (Figure 6).

Effect of Burning

Burning reduced the mass of individual samples twenty-three percent (Table 1) from $2,566 \text{ kg ha}^{-1} \pm 183$ to $1,981 \text{ kg ha}^{-1} \pm 183$. Burning may have also reduced root biomass through the profile by twenty-three (Table 2) from $10,266 \text{ kg ha}^{-1} \pm 695$ to $7,925 \text{ kg ha}^{-1} \pm 649$. There was a significant treatment-depth interaction (Table 1) indicating that burning. Root mass decreased in all depths below 8 cm (Figure 7).

Discussion

Pits vs. Cores

We expected that pits and cores would provide relative indices of root biomass which were inherently different due to sampling specific bias. The lack of difference between the two sampling methods at the individual sample and soil profile levels contrast with other studies which have documented differences of as much as twenty seven percent (Park et al. 2007). Our results also consistent with results reported for large intact monoliths from a shortgrass steppe in Northeastern CO USA (LeCain et al. 2006). We believe that the more consistent results between methods are due to the type of coring device used. The new device was designed to core through concrete, and is well suited to soil sampling. This device will core through rock fragments which eliminates problems described by other researchers who were unable to push a simple punch core through coarse fragments and large diameter roots (Vogt and Persson 1991). The device is also small enough that it can be transported into rugged terrain unlike truck mounted units. This device also should yield a more accurate estimate of sample volume and whole soil bulk density. Some researchers have reported increases in bulk density using traditional cores and related it to compaction of the sample (Vogt and Persson, 1991). However, Ponder and Alley (1996) reported no increase in soil bulk density when using a device similar to the one we employed. Some compaction of the sample may have occurred with the new device, increasing whole soil bulk density, but it is likely that a more proportionate (larger) volume of large coarse fragment was sampled because the diamond tipped bit cuts cleanly

through large coarse fragments. Because large coarse fragment has a mean bulk density of 2.6 g cm^{-3} , the bulk density of the samples was probably increased.

Our higher level interactions for sample type-depth, and sample type-depth-microsite show a significant influence of the sample method on root mass. Soil pits typically show much larger variability across depths and microsites, while soil cores indicate a more even distribution across depths and microsites. We attribute this discrepancy to sample areas which are unique to each method. A soil core is only 7.62 cm diameter giving it an area of 45.6 cm^2 . A soil pit measured 50 cm on a side giving it an area of 2500 cm^2 . Thus, for a given depth a pit samples an area 55 times larger than a core. Further investigation is necessary to determine how the area sampled affects estimates of root distribution on the landscape. A smaller sample area will have a larger edge effect where roots may be included or excluded from the sample if they are not cut cleanly at the sample margins. This could increase the error associated with root biomass estimates.

The coring method modified from Ponder and Alley (1996) shows significant promise for streamlining belowground biomass sampling and sample processing. Quantitative soil pits can take as long as three hours to excavate depending on depth obtained and the number of large coarse fragments encountered. Pits also require transportation of large sieves, tarps, buckets, scales, and excavation tools. In the case of large monoliths heavy equipment may be necessary. In addition to field time, processing of moisture samples, soil sub-samples, and computation time increase the effort required by quantitative pits (Figure 3). By contrast a soil core can typically be extracted in less than one half hour. The core device does require similar amounts of

equipment, but it can be easily transported by three people into remote areas. Sample processing and data calculations associated with core samples are considerably simpler than with quantitative pits reducing the possibility for error (Figure 4). The core device setup can be assembled for approximately \$1,200 U.S. at this time utilizing currently available products, and materials common at most metal fabricators. We believe the core gives more accurate estimates of sample volume, rock fragment, and bulk density than previously utilized methods. This method should allow researchers to utilize larger sample numbers over a broader range of soil conditions, and improve our estimates of belowground biomass. Additional consideration should be given to the effect of core diameter on estimated root distribution, and it would be worthwhile to make further comparison of pits and cores across more locations and ecosystems.

Spatial Distribution of Roots

Cold deserts have been shown to have some of the most extensive belowground biomass of any ecosystem (Jackson et al. 1996). However, the spatial distribution of roots in arid sagebrush ecosystems has been documented infrequently over the last century, and studies typically have focused on a single plant species, particularly sagebrush (Robertson 1943; Frischknecht 1963; Sturges 1977; Richards and Caldwell 1987). Our data show that both the distribution and amount of belowground root biomass may change with conversion from shrubland to woodland or from woodland to grassland with fire.

Tree and shrub microsites typically had higher root biomass in the top 50 cm of the soil profile than interspace microsites. We suspect that root mass is concentrated

beneath trees and shrubs, but it is likely these roots are spreading laterally into areas without tree or shrub cover. Sturges (1977) observed that sagebrush produced lateral roots extending over 1 m from the shrub bole. It is likely that pinyon pine and juniper have roots which extend even farther laterally (Kramer et al. 1996).

Our data show that herbaceous vegetation found in interspace has most of its root mass in surface horizons (0-8 cm). Sagebrush in this study also tend to maximize root density near the surface, but they also partition large amounts of root mass just below the surface (8-23 cm). We observed that shrubs have a long taproot extending well into the soil profile, at least to the bottom of our pits. Finally, trees trended toward maximum root density near the sub-surface (23-52 cm) or lithic contact. This pattern has been noted by other researchers in arid environments and has been linked to high soil carbon content at depth in mature woodlands (McDaniel and Graham 1992). Trees tended to have the lowest root density near the surface (0-8 cm) which is typical of trees growing in semi-arid regions with low summer precipitation (Williams and Ehleringer 2000).

Effect of Burning

Burning resulted in a statistically significant reduction in root biomass measured to 52 cm four years afterward. However, the absolute magnitude of the reduction was relatively small (13-23 %). What is perhaps more interesting is how burning influenced the distribution of roots. The control plots averaged across all microsites displayed a root distribution more representative of the tree and shrub microsites. Conversely, the burned site had a distribution representative of interspace microsites or herbaceous vegetation. The rapid change in distribution may be related to partial or

complete decomposition of roots after the death of trees and shrubs, and the re-growth and establishment of herbaceous vegetation on the site. Following the burn herbaceous vegetation on the site increased in cover, biomass, and nutrient content (Rau et al. 2008, Goergen and Chambers 2009).

It is of interest to determine how burning will influence turnover of belowground carbon in these arid heterogeneous ecosystems. Prescribed burning volatilizes considerable aboveground biomass and more carbon will eventually be lost through microbial respiration. Additionally, studies have documented the effects of fire on near surface soil carbon (Murphy et al 2006, Caldwell et al. 2002, and Klopatek et al. 1991). However, very little consideration has been given to turnover and storage of belowground carbon especially in root biomass.

Conclusions

Estimates of root biomass using quantitative pits were similar to estimates of root biomass using soil cores. However, pits yielded higher variability in spatial distributions and burn effects. We believe that the area of each sample type influences these results possibly due to edge effects. More work is needed to better understand how sample area influences root biomass estimates.

Root biomass was found to be higher under tree and shrub canopies than in interspaces. Root biomass was highest near the soil surface at interspace and undershrub microsites and decreased with depth. However, root biomass was concentrated near the lithic contact under tree canopies.

Burning appears to have reduced root biomass on our plots, and altered the distribution of roots so that it is more representative of the profile observed at interspace microsites.

Table 1.0 Results from the mixed effects model comparing treatment, microsite, depth, and sample type (*Pits vs. Cores*) means.

Comparison	DF	F	P
Treatment	1	10.50	0.0177
Sub-plot (Treatment) error A	6		
Microsite	2	4.23	0.0407
Treatment x Microsite	2	0.39	0.6848
Microsite x Sub-plot (Treatment) error B	12		
Depth	3	7.10	0.0004
Treatment x Depth	3	3.56	0.0200
Microsite x Depth	6	4.36	0.0012
Treatment x Microsite x Depth	6	0.33	0.9200
Depth x Microsite x Sub-plot (Treatment) error C	54		
Sample Type	1	2.11	0.1532
Sample Type x Treatment	1	2.05	0.1586
Sample Type x Microsite	2	0.00	0.9972
Sample Type x Treatment x Microsite	2	0.13	0.8752
Sample Type x Depth	3	3.01	0.0392
Sample Type x Treatment x Depth	3	1.14	0.3416
Sample Type x Microsite x Depth	6	3.28	0.0087
Sample Type x Treatment x Microsite x Depth	6	1.21	0.3195
Sample Type x Depth x Microsite x Sub-plot (Treatment) error D	48		

Table 2.0 Results from the mixed effects model comparing treatment, microsite, and sample type (*Pits vs. Cores*) summed over all depth increments.

Comparison	DF	F	P
Treatment	1	6.61	0.0546
Sub-plot (Treatment) error A	6		
Microsite	2	3.68	0.0029
Treatment x Microsite	2	0.31	0.1052
Microsite x Sub-plot (Treatment) error B	12		
Sample Type	1	2.45	0.2810
Sample Type x Treatment	1	2.26	0.2243
Sample Type x Microsite	2	0.03	0.9701
Sample Type x Treatment x Microsite	2	0.13	0.8757
Sample Type x Microsite x Sub-plot (Treatment) error C	12		

Figure 1. Photos of the rotary core bit, the adapter shaft used to connect it to the power head, and the power head. Note how cleanly the large coarse fragment has been cut by the core device. Top scale is in inches, bottom scale is in cm. Models are J.J. Klima and the author at USFWS, Hart Mountain Wildlife Refuge, OR.

Figure 2. Diagram depicting the two treatment plots, eight sub-plots, and three microsites within the study area.

Figure 3. Flow diagram of the Quantitative Soil Pit method and the process used to calculate root biomass estimates.

Figure 4. Flow diagram of the Soil Core method and the process used to calculate root biomass estimates.

Figure 5. Tukey tested means comparison for microsite-depth-sample type interactions. Means not represented by the same letter are significantly different. Bars represent estimated > 2 mm root biomass for three microsites (under tree, under shrub, and interspace), four soil depths (0-8, 8-23, 23-38, and 38-52 cm), and two sample types (pits and cores). Means and standard errors are calculated from core and pit samples separately.

Figure 6. Tukey tested means comparison for microsite-depth-interactions. Means not represented by the same letter are significantly different. Bars represent estimated > 2 mm root biomass for three microsites (under tree, under shrub, and interspace) and four soil depths (0-8, 8-23, 23-38, and 38-52 cm). Means and standard errors are calculated from core and pit samples combined.

Figure 7. Tukey tested means comparison for treatment-depth interactions. Means not represented by the same letter are significantly different. Bars represent estimated > 2

mm root biomass for three microsites (under tree, under shrub, and interspace), four soil depths (0-8, 8-23, 23-38, and 38-52 cm), and two treatments (control and burned). Means and standard errors are calculated from core and pit samples combined.

Figure 1.0



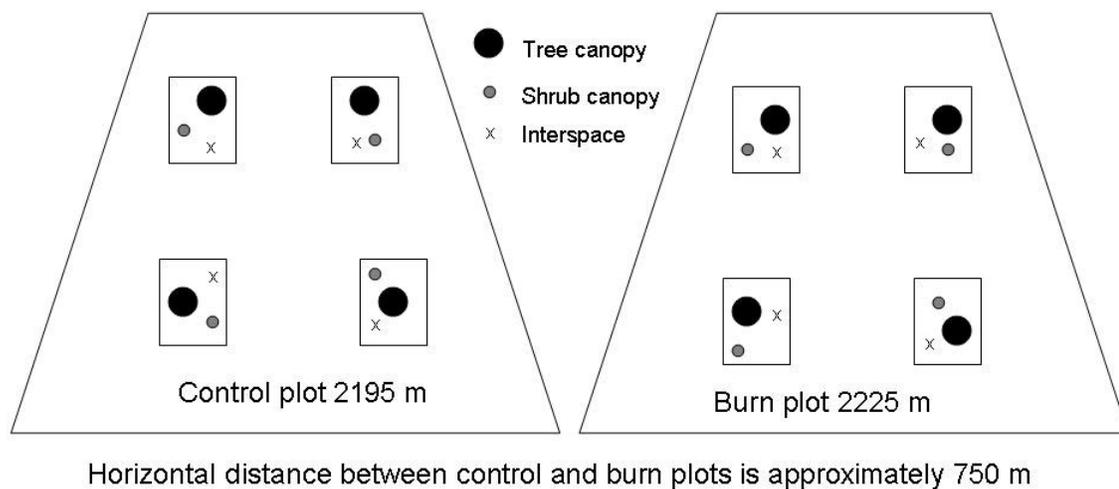
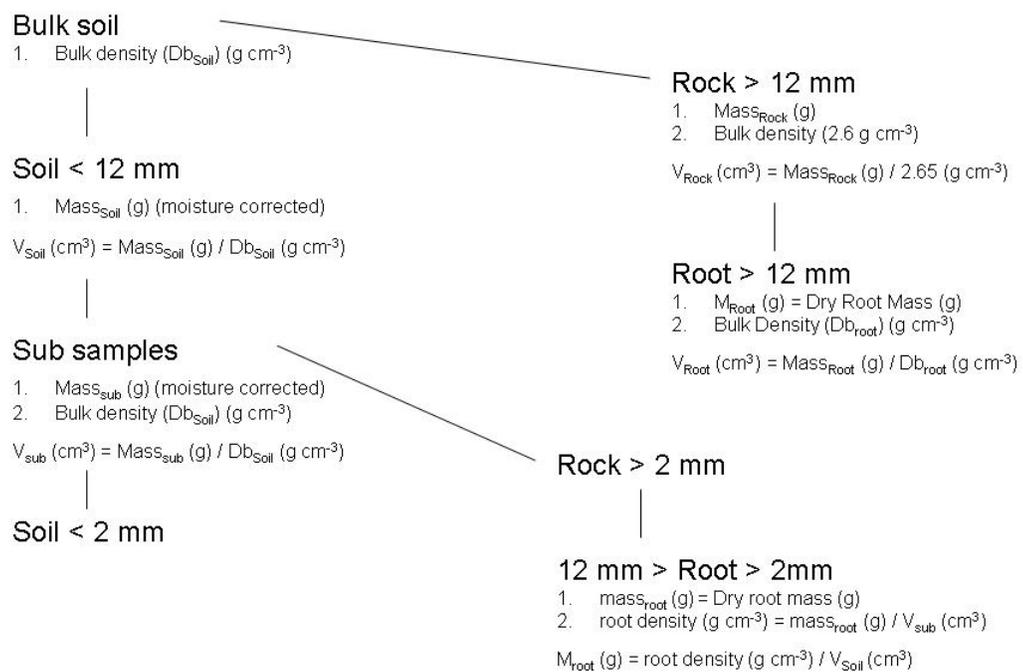
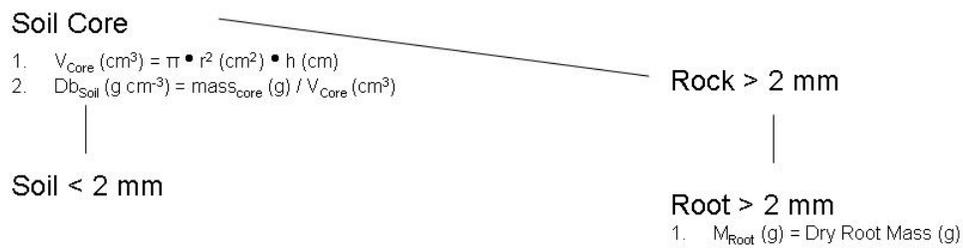
Figure 2.0

Figure 3.0



$$V_{pit} (cm^3) = V_{Rock} (cm^3) + V_{Soil} (cm^3) + V_{Root} (cm^3)$$

$$Roots > 2\ mm\ (Kg\ ha^{-1}) = \{M_{Root} (g) + M_{root} (g) / V_{pit} (cm^3)\} \cdot d (cm) \cdot 100,000 (cm^2\ g^{-1})$$

Figure 4.0

$$\text{Roots} > 2 \text{ mm (Kg ha}^{-1}\text{)} = \{M_{\text{Root}} (\text{cm}) / V_{\text{core}} (\text{cm}^3)\} \cdot d (\text{cm}) \cdot 100,000 (\text{cm}^2 \text{g}^{-1})$$

Figure 5.0

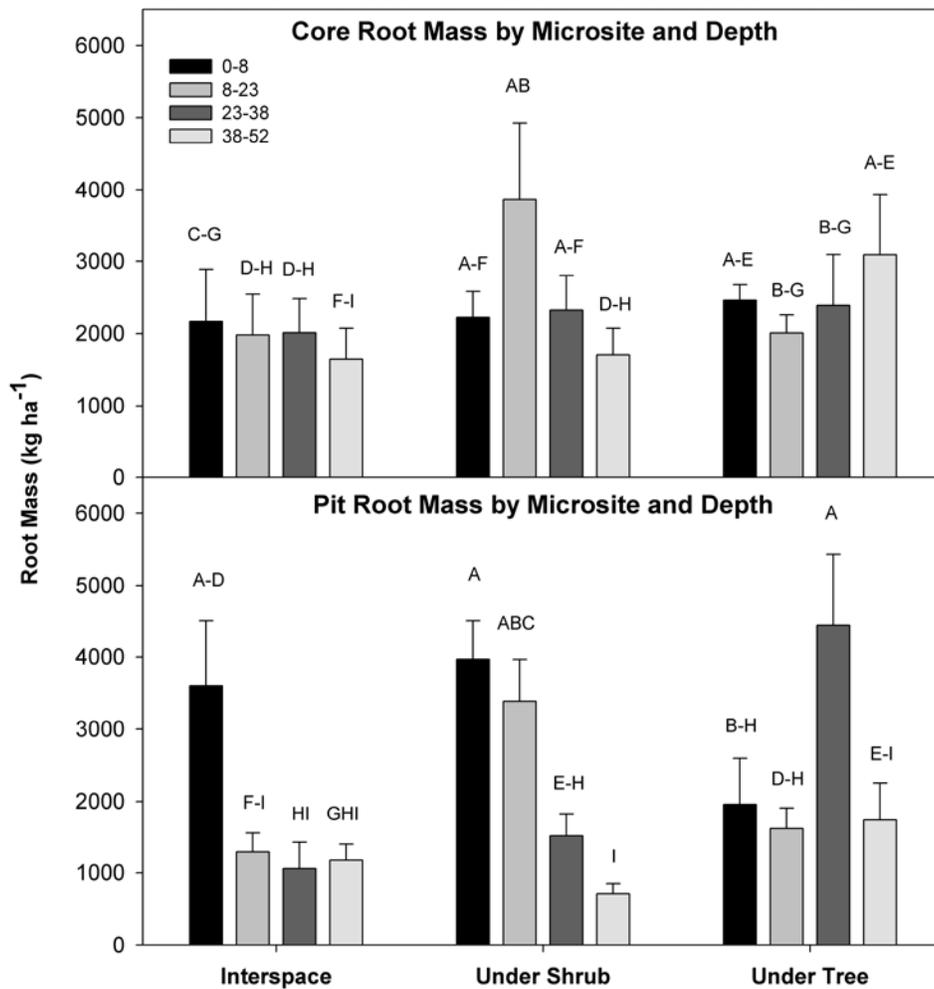


Figure 6.0

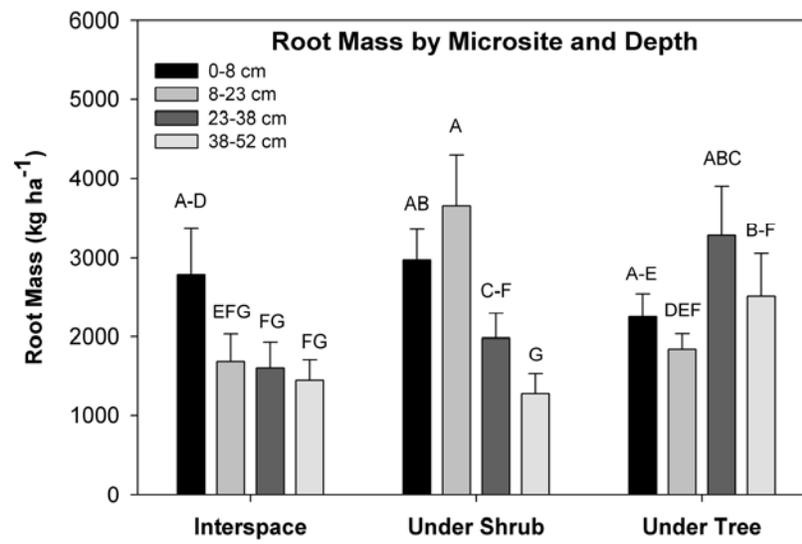
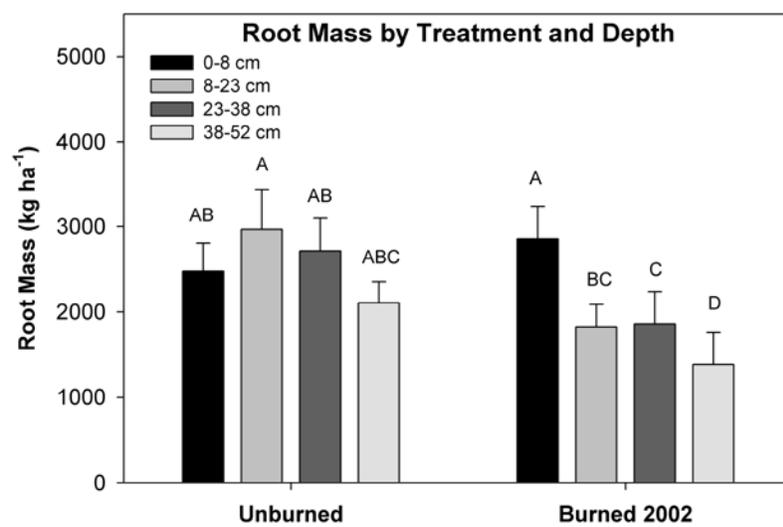


Figure 7.0



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Influence of Prescribed Fire on Ecosystem Biomass, Carbon, and Nitrogen in a Pinyon Juniper Woodland

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Abstract

Increases in pinyon and juniper woodlands associated with land use history are suggested to provide offsets for carbon emissions in arid regions. However, the largest pools of carbon in arid landscapes are typically found in soils, and aboveground biomass cannot be considered long term storage in fire prone ecosystems. Also, the objectives of carbon storage may conflict with landscape management for ecosystem services and fuels reduction. We quantified effects of prescribed fire as a fuels reduction and ecosystem maintenance treatment on fuel loads, ecosystem carbon, and nitrogen in a pinyon-juniper woodland in the central Great Basin. We found that plots containing 30 % tree cover averaged nearly 40,000 kg ha⁻¹ in total aboveground biomass, 80,000 kg ha⁻¹ in ecosystem C, and 5,000 kg ha⁻¹ in ecosystem N. Only 25 % of ecosystem C and 5 % of ecosystem N resided in aboveground biomass pools. Prescribed burning resulted in a 65 % reduction in aboveground biomass, a 68 % reduction in aboveground C, and a 78 % reduction in aboveground N. However, due to temporal variation in soil C and N pools, their size relative to aboveground pools, and the difficulty in obtaining consistent measurements no statistically significant change in ecosystem C occurred, and, although unrealistic, a statistical increase in ecosystem N occurred after the burn. We believe this is in part due to our stratified sampling design which may over estimate islands of fertility including high N concentrations under shrubs and trees following fire.

Key words: prescribed fire, carbon storage, nitrogen, ecosystem maintenance, fuels management,

1. Introduction

Vegetation changes associated with longer term climate change and anthropogenic disturbance are thought to have major effects on soils, vegetation, and biogeochemical cycling (Schimel et al. 1991, 1994). Much of the Great Basin is currently dominated by sagebrush (*Artemisia tridentata ssp.*) ecosystems, but at intermediate elevations with more mesic climatic regimes, sagebrush ecosystems are increasingly influenced by pinyon (*Pinus monophylla*, *Pinus edulis*) and juniper (*Juniperus osteosperma*, *Juniperus occidentalis*) 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 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).

Some scientists hypothesize that woodland expansion could result in large increases in carbon (C) storage within the interior west (Canadell and Raupach 2008; Hibbard et al. 2003; Norris et al. 2001). It is possible that increasing tree cover could temporarily increase biomass and C storage; however, most C in semi-arid systems is contained belowground in soils (Birdsey 1992). In addition, aboveground biomass in expansion woodlands should not be considered long-term C storage due to the frequency of wildfires in semi-arid systems that may cause C stored in biomass to return to the atmosphere (Hurteau and North 2009; Canadell and Raupach 2008). Earlier snowmelt and warmer temperatures due to climate change may extend the wildfire season and

increase the frequency and duration of large wildland fires (Westerling et al. 2006). Wildfires are estimated to produce 40 % of CO₂ emissions globally, and longer fire seasons and larger fires could further increase CO₂ emissions (Running 2006).

Increasing tree cover in sagebrush ecosystems can lead to a detrimental decrease in herbaceous understory biomass which facilitates ecosystem recovery following fire (Suring et al. 2005, Chambers et al. 2007). Landscapes with high tree density and compromised understory vegetation are susceptible to catastrophic wildfire and exotic grass invasions (Young and Evans 1973, Miller and Tausch 2001, Chambers 2005). Once exotic grasses such as cheatgrass (*Bromus tectorum*) establish on the landscape, a shift to an annual grass dominance may result in considerably lower potential to store C, and may create a C source by decreasing the fire return interval (Young and Evans 1973, Bradley 2006).

Re-establishing fire as an ecosystem process through planned or unplanned fires could reduce woodland cover on the landscape, maintain herbaceous species, and discourage exotic grass invasion. Carbon storage decisions must consider not only the current vegetation state, but also future states and the potential effects of climate change and exotic invasion on the fire regime associated with those states (Hurteau and North 2009). Because nitrogen (N) is often the limiting factor in semi-arid systems after water, and because C and N cycling are so closely linked, it is important to understand potential changes in N cycling as well (Johnson and Curtis 2001).

In this study we addressed three questions. 1) How does prescribed fire affect biomass and fuel loads in expansion woodlands? 2) How does prescribed fire influence the release of C and N from the system? 3) Which pools of C and N are most responsive to

prescribed fire? To address these questions we measured vegetation and soil C and N pools in a sagebrush ecosystem prior to and following prescribed fire.

2. 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. Elevation ranges from 2,072 m to 2,346 m. 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 located 35 miles from the site 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 and are classified as coarse loamy skeletal mixed frigid Typic Haploxerolls (Rau et al. 2005).

The vegetation is characterized by sagebrush (*Artemisia tridentata vaseyana*) and single leaf pinyon (*Pinus monophylla*) with lesser cover of Utah juniper (*Juniperus 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., [E. ovalifolium](#) Nutt., *E. elatum* [Dougl. ex Benth.](#), *E. heracleoides* Nutt., *Crepis acuminata* Nutt., *Phlox longifolia* Nutt., *Agoseris glauca* ([Pursh](#)) [Raf.](#), *Lupinus argenteus* Pursh, and *Penstemon* species. *Bromus tectorum*, an invasive annual grass, is not a large component

of the study area. 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, 21,000 kg/ ha) to high tree dominance (74% cover, 98,000 kg/ ha; Reiner 2004).

2.2 Study design and data collection

The study plots were located on northeast facing alluvial fans at elevations of 2195 m and 2225 m. The plots at elevation 2195 m were a control, and the plots at 2225 m received a spring prescribed burn. Four 0.1 ha sub-plots were located on both the control and burn treatment. Plots were characterized by intermediate tree cover ($\approx 30\%$ cover, 40,000 kg/ ha) at both elevations and contained a mix of trees, shrubs, and interspaces (Figure 1). USDA Forest Service fire personnel burned the study plots on May 11-14, 2002. (Air temp $< 32^{\circ}\text{C}$, RH $> 15\%$, wind speed $< 9 \text{ m}\cdot\text{s}^{-1}$, and gravimetric fuel moisture $\approx 40\%$). Fire behavior was characterized by creeping ground fire with some single and group tree torching (Figure 2). Soil surface temperatures during the fire reached 370°C under shrub canopies, 300°C under tree canopies, and 200°C at interspaces. Soil temperatures were elevated slightly at 2 cm depth (86°C under shrubs, and 77°C under tree canopies), but measurable quantities of heat were not transferred below 2 cm (Rau et al. 2005).

2.3 Soil and roots

Soil pits were used to characterize the study plots. Pits were dug to a depth of 52 cm and the soil horizons were identified. Depth increments for sampling were assigned to the approximate center of the soil A₁ horizon and subsequent 15 cm increments (0-8, 8-23, 23-38, and 38-52 cm). Bulk density samples were collected from each depth using a 93 cm³ soil core. Soil samples were taken from three microsites (under tree, under shrub, interspace) for each depth using a 10 cm diameter bucket auger (Figure 1). Sampling was

conducted in November 2001 and again in 2002 to determine fire effects on soil C and N. All soil was brought back to the lab, dried at 60°, and sieved to 2 mm. The fine earth fraction was then ground in a Wiley® mill and analyzed for total C and N concentration using a LECO Truspec® CN analyzer. To examine site level changes in C and N content, data was transformed into kg ha⁻¹ by using the formula

$$\text{kg ha}^{-1} = (t)(\text{Db})[1 - (>2 \text{ mm}\%)](\text{conc})(F)$$

where t = thickness (cm) of the soil horizon, Db = bulk density (g cm^{-3}) of that horizon, $>2 \text{ mm}\%$ is the volume percentage coarse fragment of that horizon, conc = nutrient concentration ($\mu\text{g g}^{-1}$), and F = conversion factor ($1 \text{ g } 1,000,000 \mu\text{g}^{-1} * 1 \text{ kg } 1,000 \text{ g}^{-1} * 10,000 \text{ cm}^2 1 \text{ m}^{-2} * 10,000 \text{ m}^2 \text{ ha}^{-1}$).

Root biomass was estimated for each sub-plot, microsite, and depth post-hoc in 2005 using a core device (Rau et al. 2009). Roots were separated from soil and rock by sieve and flotation, dried, weighed, then ground in a Wiley® mill and analyzed for total C and N concentration using a LECO Truspec® CN analyzer.

For the soil profile, soil and root C and N (kg ha^{-1}) was summed by the four soil depths to 52 cm. Then the total mass of soil and root C and N at each microsite was weighted by that microsites cover percentage on the sub-plot. The sum of all three weighted microsites was the sum of soil and root C and N on each sub-plot.

2.4 Understory biomass and litter

Regression analyses were used to model biomass by species based on field measurements of plant size and cover, and estimates of plant weight. Understory vegetation in each sub-plot was sampled in fifty 1 x 2 m micro-plots. Ten plots were located contiguously along five belt transects perpendicular to the long axis of the sub-plot (Figure 3). Sampling

occurred in summer 2001 and again in summer 2002 to evaluate biomass lost to prescribed burning. All shrubs rooted in the micro-plots were measured for the longest crown diameter, the crown diameter perpendicular to the longest, total plant height, the crown height of live foliage and the basal diameter (stem diameter just above ground level). Perennial forbs were measured for two crown diameters and the total height. Perennial grasses were measured for two basal diameters and the total height. When herbaceous plants were small and abundant, grasses and forbs were sampled by species in each micro-plot by estimating their percent cover plus a measurement of average height.

For each shrub, grass, or forb measured on a transect, an individual of that species was randomly located outside the sub-plot and clipped to ground level. Also, one micro-plot on each transect was randomly selected and clipped to obtain the biomass by species for which the percent cover was determined. Clipped shrub and herbaceous biomass was returned to the lab, dried, separated into live and dead categories of 1 hour (< 6.35 mm), 10 hour ($6.35 - 25.4$ mm), and 100 hour ($25.4 - 76.2$ mm) timelag fuels and weighed.

Individual regression models were created for each grass and forb species when sample size was sufficient (Reiner 2004). Volume vs. mass regression models were created using non-linear regressions (Tausch and Tueller 1988). The ellipsoid volume was calculated from the two diameters and total plant height and used in regression analysis (Reiner 2004). Crown dimension measurements were used to calculate the ellipsoid crown volume used in regression equations for the forbs and large grass species with measured crowns. When the percent cover measurement method was used, it was multiplied by micro-plot area to compute an area in cm^2 . Biomass for each species was then estimated

by micro-plot, and the sum of biomass for all micro-plots was reported for each sub-plot as grams per 100 m².

Shrub litter mats were sampled under 18 rabbitbrush *Chrysothamnus viscidiflorus* Hook. (Nutt.) and 36 sagebrush (a combination of *A. tridentata vaseyana* and *A. tridentata wyomingensis*) plants. A 100 cm² frame was placed approximately halfway between the stem and the outer edge of the litter mat of each shrub to collect samples representative of the entire litter mat (Brown 1982). The litter in each frame was collected from the Oi and Oe horizons. Full crown and litter mat dimensions were taken for each shrub. Each litter sample was floated to remove rocks, then dried and weighed. Because shrub litter samples were taken to be representative of the entire litter mat, the weight per cm² computed for each 100 cm² litter sample was multiplied by the entire shrub litter mat area to derive an estimated total litter mat weight. Shrub litter weight was estimated for sagebrush and rabbitbrush species using regression equations based on relationships developed between shrub crown area and litter mat weight (Reiner 2004).

2.5 Tree biomass and litter

Tree biomass was estimated in each sub-plot by measuring total tree height, crown height, longest crown diameter, crown diameter perpendicular to the longest diameter, and trunk diameter just above the root crown for each individual tree rooted within the sub-plot. Crown measurements were the portion of the crown containing foliage. Tree measurements were collected in summer 2000. Volume vs. mass regression equations were used to estimate biomass from tree volume (Tausch 2009). Volume-mass regression equations were developed for pinyon and juniper trees by measuring and harvesting 18 trees from just outside of sub-plots prior to and after burning. Trees ranged in size from

0.5 m to 6.3 m in height. Trees up to about 2.5 m tall were collected intact. Tarps were spread under the trees prior to sampling. All parts of the above ground portion of the trees were returned to the lab for processing and weighing. Larger trees were cut into as few sub-samples as possible. All collected parts were wrapped in plastic tarps to prevent branch or needle loss. The trunk was cut off at ground level, and divided into only as many segments as necessary for transport. Individual trees were processed by separating live and dead foliage from branches, and then all parts of each tree were separated by fuel timelag size class (Tausch 2009). The biomass of each live and dead fuel size class was then calculated pre and post-burn based on regression equations derived from these measurements (Tausch 2009).

Tree litter mats were sampled under 17 pinyon pines with crown diameters ranging from 1.8 to 7.0 m. Complete crown and litter mat dimensions were collected in order to derive relationships between litter mat area and litter mat mass. One to three 33.5 cm diameter rings were evenly spaced on either side of the tree bole depending on its size. Litter was removed by horizon (O_i , O_e , and O_a), bagged, and brought back to the lab. Litter samples were floated to remove mineral fragments, separated by fuel time lag size, dried and weighed. Regressions were then developed between litter biomass and tree crown area to estimate total sub-plot tree litter mass (Table 1).

2.6 Biomass carbon and nitrogen

Six sub-samples were randomly selected from each biomass component (grass, forb, shrub litter, live and dead shrub 1-100 hr fuels, tree litter, live and dead tree 1-1000 hr fuels) pre and post-burn for chemical analyses. Samples were ground in a Wiley[®] mill and analyzed for total C and N concentration using a LECO Truspec[®] CN analyzer. The

percent C and N for each biomass component was multiplied by the total mass of that component in each sub-plot. Biomass C and N were scaled to kg ha^{-1} .

2.7 Statistical analyses

The Kolmogorov-Smirnov test was used to test for data normality. All data were natural log transformed to meet the assumption that the data was normally distributed.

Comparisons were evaluated using SASTM mixed effects models with year and treatment as fixed effects and sub-plot as a random effect. The year by treatment interaction term was used to identify changes caused by the prescribed fire ($P < 0.05$).

3. Results

3.1 Biomass and fuels

Prior to the prescribed fire in summer 2001, cover and biomass were similar on control and treatment plots averaging approximately 30 % tree cover, 30 % shrub cover, and 40 % bare ground or herbaceous cover (Table 2; Figure 2). Total aboveground biomass across all plots averaged over $38,000 \text{ kg ha}^{-1}$ (Table 2). Herbaceous biomass accounted for approximately 1 % of total biomass on plots, shrub biomass was less than 10 %, shrub and tree litter accounted for another 10 %, and trees contained the remaining 80 % of aboveground biomass (Table 2). Roots within the control and treatment plots to a depth of 52 cm averaged approximately $13,000 \text{ kg ha}^{-1}$ or about 1/3 of aboveground biomass (Table 2).

Mixed model results showed significant year by treatment interactions for herbaceous understory, tree foliage, tree 1 hr fuels and aboveground total ($P < 0.05$). Herbaceous understory fuels were reduced 91 % post-burn (Table 2; Figure 2). Aerial fuels including tree foliage and 1 h fuels were reduced by 75 and 93 %, respectively (Table 2; Figure 2).

Reductions in shrub fuels, and tree 10, 100, and 100 h fuels and shrub and tree litter were not statistically significant (Table 2). Overall aboveground biomass on the site was significantly reduced by 63 % (Table 2; Figure 2). Additionally, measurements from root biomass taken in 2005 indicates that fire did not significantly reduce total root mass ($P > 0.05$) (Table 2).

3.2 Ecosystem Carbon

Pre-burn C concentrations varied by biomass component, ranging from 32 % for roots to 55% for tree 1 h fuels (Table 3). Soil C was most concentrated near the surface (2.4 %) and decreased with depth (Table 3). There was no consistent pattern in how prescribed fire affected C concentrations. Soil C concentration tended to increase near the surface, herbaceous and standing dead shrub fuels tended to have increased C concentration, and standing dead tree fuels tended to have lower C concentration following the burn (Table 3).

Soil pools accounted for over 65 % of ecosystem C with the next largest pool contributed by aboveground mass of trees (20 %), then roots (6 %), litter, shrubs, and herbaceous biomass (Table 4). Burning did not have a significant overall effect on soil C as described by the year by treatment interaction term ($P > 0.05$).

Prescribed burning significantly reduced aboveground C including herbaceous C and shrub C as indicated by the year by treatment interaction term ($P < 0.05$). Burning resulted in a net loss of approximately 13,000 kg C ha⁻¹ from aboveground biomass (Table 4). Herbaceous and shrub C decreased by 91 % and 83 %, respectively (Table 4). The year by treatment interaction term for roots was marginally significant ($P = 0.0812$).

Total ecosystem C was not significantly changed by the prescribed fire ($P > 0.05$) (Table 5).

3.3 Ecosystem Nitrogen

Pre-burn N concentration varied by biomass component ranging from 0.17 % for tree 1000 h fuels to 1.55% for shrub litter (Table 3). Soil N was most concentrated near the soil surface (0.25 %) and decreased with depth (Table 3). There was no consistent pattern in how prescribed fire affected N concentrations. Soil N concentration increased slightly after burning. Roots, herbaceous understory, litter, and foliage tended to have increased N concentration, and woody fuels tended to have lower N concentration following the burn (Table 3).

Prior to burning, mean ecosystem N contents in control and burn plots averaged $> 5,000 \text{ kg ha}^{-1}$ (Table 4). Soil pools accounted for over 90 % of ecosystem N with trees (2 %) and roots (2 %) being the next largest pool followed by litter, shrubs, and herbaceous biomass (Table 4). The year by treatment interactions were significant for all but roots, indicating significant effects of fire ($P < 0.05$).

Fire caused the expected decreases in biomass and litter N contents, but seemed to also cause an increase in soil N content (Table 4). Estimated N losses from the total aboveground N pool was 227 kg N ha^{-1} from aboveground biomass with at least 121 N kg ha^{-1} lost from trees alone (Table 4). Litter, herbaceous, and shrub N were all significantly reduced by 92, 87, and 86 % respectively (Table 4). Because of the increase in soil N of $> 1,500 \text{ kg ha}^{-1}$ (Table 4), total ecosystem N also increased significantly by nearly $1,300 \text{ kg ha}^{-1}$ following the fire in the burned plots (Table 4).

4. Discussion

4.1 Biomass and fuels

Tree cover values on our plots average 30 % and represent total aboveground biomass approaching 40,000 kg ha⁻¹. Trees represent over 80 % of total biomass in these expansion woodlands. Total production varies with climatic and other site conditions for both trees and understory species, and tree abundance relative to that of the understory can be expected to increase as the stand matures (Chambers et al. 2007; Miller and Tausch 2001; Reiner 2004).

Prescribed burning removed nearly 65 % of total aboveground biomass, including 90 % of herbaceous fuels. Observations from the site show that herbaceous fuels recovered relatively quickly following the burn (Dhaemers 2006). Burning also removed 56 % of tree biomass with over 90 % of foliage and 1 h fuels being removed leaving predominantly 10, 100, and 1000 h fuels. Post burn aboveground biomass on our plots was estimated at less than 15,000 kg ha⁻¹. These plots should remain in a state of perennial grass and forb dominance for several years following the fire.

Root biomass in our plots was measured to be 1/3 of aboveground biomass. This is less than some estimates of root shoot ratios in cold semi-arid systems (Jackson et al. 1996). Differences in our estimates compared to other reported values could be influenced by the depth of roots sampled, we sampled to 52 cm, or the type of system sampled, most other reports for cold deserts come from sagebrush dominated stands. Prescribed burning did not have a statistically significant effect on root biomass on our plots even though measurements were taken four years following the burn. In a companion study focusing on root biomass dynamics we determined that there were statistically detectable changes

in root biomass caused by fire, but those results were obtained from quantitative soil pits, and the same results could not be statistically verified using soil cores (Rau et al. 2009). Root biomass is not easily measured, and not well documented in sagebrush or pinyon woodland ecosystems. Better and more frequent measurements of root biomass will allow us more completely understand competition for resources, and ecosystem C dynamics.

4.2 Ecosystem Carbon

As is usually the case, soil pools dominated total ecosystem C. The magnitude of soil C contained in our plots (50,000 - 75,000 kg ha⁻¹) is similar to values obtained from sagebrush plots (62,000 kg ha⁻¹) (Hooker et al. 2008), for semi-arid Jeffrey pine (*Pinus jeffreyii*) plots (38,000 - 65,000 kg ha⁻¹) (Johnson et al. 2007, 2008), and more mesic forest types (Baird et al. 1999, Neill et al. 2007). This indicates that some arid and semi-arid systems have similar potential to more mesic systems for soil C retention. The degree of variation between plots in all studies is also similar to our observed variation. High inherent variability in soil C for geographically similar and co-located plots is common and increases the difficulty of detecting soil C change and the effects of treatments like prescribed fire on soil C.

Aboveground biomass on our plots accounted for just over 20 % of ecosystem C. Trees contained the most C by far. Our estimates for C stored in aboveground biomass in transition woodlands are 30 – 400 % of values reported by similar studies. Additional measurements from this study show that as stands mature and crown cover increases, aboveground biomass increases to nearly 140,000 kg ha⁻¹ in closed canopy pinyon stands (Rau unpublished). These values are comparable to more mesic forests despite lower precipitation

Root C accounts for about 5 % of total ecosystem C, but may play a very important role in long term C storage (Schimel 1995; Strand et al. 2008). Roots exude carbohydrates and fine roots can turnover very rapidly in soils providing a substrate for microorganisms to convert these exudates to less labile forms of soil C (Schlesinger 1977).

Burning released 13,000 kg ha⁻¹ C from aboveground biomass on our plots. This is consistent with the only other study we could find measuring C loss from fire in pinyon and juniper woodlands, but amounts to only half of the C released from wildfire in a Sierra Nevada Jeffrey pine stand (Klopatek et al. 1991; Johnson et al. 2007). However, as tree cover increases in expansion woodlands we can expect wildland fire to release as much if not more C than in some mesic forests. Burning consumed 90 % of herbaceous and shrub C and 90 % of fine aerial fuel C (foliage & 1 hr), leaving predominantly 10, 100, and 1000 hr woody C. Over time these residues likely will fall to the ground and decompose. A large portion of this remaining pool will be lost as microbial respiration, but some will be incorporated into soils (Johnson and Curtis 2001).

Prescribed burning removed C from aboveground pools, but no significant reduction in root C was observed 3 years following the burn. We have evidence that the C:N ratio of root material is greatly decreased following fire, indicating that microbial respiration of root material is occurring (Table 4). However, it is also possible that fine roots from perennial herbaceous vegetation re-establishing on the site are offsetting decomposition of dead tree and shrub roots. Observations from our companion study that focused on root biomass show that rooting depth profiles across all burned sub-plots more closely resemble herbaceous species rooting profiles following fire than the rooting profiles of trees or shrubs (Rau et al. 2009).

Managing semi-arid woodlands or fire prone forests for C retention may not be as simple as growing more trees on the landscape (Hurteau and North 2009). Most C in semi-arid ecosystems is stored in soils, and C found in aboveground biomass varies depending on disturbance regime and land use history (Johnson and Curtis 2001). Ironically, the best way to manage ecosystems with historically frequent fire regimes for C retention may be to re-introduce fire into the system (Hurteau et al. 2008). Prescribed fire or woodland thinning can maintain fewer large trees on the landscape and reduce the risk of intense fires which release large amounts of C and leave ecosystems susceptible to exotic invasive species (Hurteau and North 2009; Miller and Tausch 2001). Conversely, other studies suggest that frequent prescribed burning releases more C than an infrequent large wildfire (Johnson et al. 1998). In the Great Basin site conversion to exotic annual grass at low to mid-elevations within the woodland zone could have a similar result. Annual grass monocultures eliminate ecosystem services such as wildlife forage, habitat, and ecosystem stability, and could cause further C losses by increasing fire frequency (Bradley et al. 2006).

4.3 Ecosystem Nitrogen

Prior to burning total ecosystem N was not statistically different between control and burn plots and averaged greater than $5,000 \text{ kg ha}^{-1}$, with over 90 % of ecosystem N found as total N in the soil. Trees and roots contributed approximately equal amounts of N, but accounted for less than 6 % of total N. The large discrepancy in above vs. belowground N emphasizes the importance of soil assessments. Fire and harvest often considered to remove large amounts of ecosystem nitrogen and leave a system in N deficit. This could be true for some systems; however, researchers have shown that ecosystem N lost to fire

or harvest can be replaced and exceeded very rapidly by N-fixing vegetation (Johnson et al. 2005). On our plots, the nitrogen fixing leguminous forb, *Lupinus argenteus* rapidly increased in abundance following the fire (Dhaemers 2006) and is associated with higher soil N contents (Goergen and Chambers 2009).

Soil N on our plots is six times higher than values reported by Klopatek et al. (1991) for pinyon and juniper woodlands and three times higher than pitch pine and oak forests reported by Neill et al. (2007), although these studies only sampled soil to 20 cm. Soil N on our plots is similar to sagebrush plots sampled to 100 cm by Hooker et al. (2008), and some Jeffrey pine plots sampled to 60 cm on andesite derived soils in the Sierra Nevada (Johnson et al. 2008), but 50 % higher than drier Jeffrey pine plots sampled to 40 and 100 cm on granitic derived soils in the Sierra Nevada (Johnson et al. 2005, 2007). These observations emphasize that the current standard of soil nutrient inventories (20 cm) may be drastically insufficient to characterize soil N or C (Johnson and Rau 2009). Further, the variation in sampling depths makes comparisons between studies difficult at best.

Prescribed burning removed 227 kg ha^{-1} or roughly 80 % of aboveground N. Nearly 90 % of herbaceous, litter, and shrub N was removed. This value is similar to values reported for other pinyon and juniper woodlands (Klopatek et al. 1991), and for Jeffrey pine forest (Johnsen et al. 2008). Our estimates for aboveground N lost by prescribed fire are roughly half of the values reported for a wildfire in Jeffrey pine forest (Johnson et al. 2007). We can assume that as woodland biomass increases additional N will be lost during fire. Although a large proportion of aboveground N was lost during the fire, the amount of N removed from aboveground biomass represents less than 5 % of total ecosystem N. It is of interest to determine how quickly N can be replaced in Great Basin

woodland ecosystems, the role of N-fixers or N-fixation rates. As mentioned above, a large increase in the legume *Lupinus argenteus* occurred following prescribed fire on our site (Georgen and Chambers 2009). Perhaps over time this legume, other N-fixers, and atmospheric deposition will replace what was lost due to burning, and may even increase ecosystem N over longer periods of time as observed in more mesic systems (Johnson et al. 2005). Thus, fire can drive both N loss and accumulation with the balance depending on biomass accumulation at the time of the fire, fire frequency, and the presence or absence of N-fixers on the site.

Although prescribed burning removed N from aboveground pools no significant reduction in root N was observed 3 years following the burn. We have evidence that the C:N ratio of root material is greatly decreased following fire (Table 4). The respiration of carbon and immobilization of nitrogen by microorganisms during root decomposition could account for these observations.

Despite the loss of aboveground N, burning resulted in a significant 30 % increase in soil N. As a result of the large increase in soil N total ecosystem N also increased. Soil N in the burned plots increased by 1,533 kg ha⁻¹. This gain represents more than the amount lost from aboveground biomass (227 kg ha⁻¹). These anomalous results may be partly due to sampling protocols. Peterson and Calvin (1986) note that there are three sources of error in soil sampling: 1) sampling error, or that error associated with the fact that only a selected sub-sample of the entire population of samples is taken; 2) selection error, where some sample types are not adequately represented (i.e., rocky areas or deeper horizons), and 3) measurement error, where the value measured is not the true value for the unit. It is impossible to sample the exact location twice, and due to the extreme heterogeneity of

soils in semi-arid woodlands sample variation is inevitable. We believe that the main source of error resulting in the over-estimate of increased soil N results from the stratified sampling design. Because soils were sampled by microsite, and the concentration of C and N is not constant across the microsite, estimation errors are possible. We typically sampled 1 m from tree boles and directly over shrub boles. The concentration of C and N obtained from these locations may be higher than C and N concentrations near the microsite margins. Hence applying the estimated C and N from cores to the entire microsite may overestimate changes in C and N. Johnson and Curtis (2001) have summarized several studies with increased N following fire. However, these increase typically take years to identify, and are the result of N-fixing vegetation and incorporation of partially burned residues into soil. Temporal variation in soil N estimates, whether real or related to sampling error, is not uncommon but presents a serious challenge when analyzing these types of data sets. Rau et al. (2009) measured temporal variation in estimated total soil N ranged from 3 – 24 % inter-annually over six years without any influence from prescribed fire. Unfortunately these results make total ecosystem N accounting difficult. We believe this further stresses the need for longer-term data sets, and improved sampling protocols.

5. Conclusions

Prescribed burning was effective at reducing fuel loads by 63 %, and maintaining ecosystem diversity within our central Nevada study plots (Dhaemers 2006). Burning released 13,000 kg ha⁻¹ C and 227 kg ha⁻¹ N from aboveground biomass accounting for 16 and 4.5 % of total ecosystem pools, respectively. Fire effects on soil C pools were not

statistically significant, but some undetected changes (either increases or decreases) could have occurred. Burning appeared to have significantly increased soil pools of N. However, the magnitude of this apparent increase is far greater than could be expected from any known inputs from either aboveground char or atmospheric deposition leading us to suspect sampling bias. Inter-annual variability and measurement uncertainty in soil C and N pools is a challenge when assessing ecosystem changes associated with land management because of the large proportion of these elements in soil. This appears especially true for N. While prescribed fire caused immediate decreases in aboveground C content, this short-term loss must be placed into perspective with regard to the risks of wildfire when assessing effects of burning on the long-term ecosystem C balance and its contribution to the global C cycle.

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Table 1.0 Regression models, r^2 coefficients, and P values for individual and total litter mass based on crown diameter (x).

Litter component	Regression	r^2	P
Needles	$0.6139 + 0.03125x$	0.5	0.01
1 hr stick	$-0.214 + 0.01905x$	0.67	0.001
10 hr stick	$-5.5264 - 0.4209x$	0.24	0.05
Oi	$-4.8412 + 0.46583x$	0.27	0.05
Oe	$-106.994 + 5.36184x$	0.55	0.001
Oa	$-111.835 + 5.8277x$	0.57	0.001
Total biomass	$-137.17 + 6.6740x$	0.62	0.001

Table 2.0 Means, standard errors (S.E.), mass change, and % change for individual biomass components before and after the prescribed burn on control and burn plots.

	Pre-burn Mass (kg / ha)				Post-burn Mass (kg / ha)				Mass Change (kg / ha)		Mass Change %	
	<u>Control</u>		<u>Burn</u>		<u>Control</u>		<u>Burn</u>		<u>Control</u>	<u>Burn</u>	<u>Control</u>	<u>Burn</u>
	<u>Mean</u>	<u>S.E.</u>	<u>Mean</u>	<u>S.E.</u>	<u>Mean</u>	<u>S.E.</u>	<u>Mean</u>	<u>S.E.</u>				
Roots 0-52 cm	13,242	1,606	13,242	1,606	13,242	1,606	8,808	559	0	-4,433	0%	-33%
Herbaceous understory	388	37	297	24	309	44	26	8	-78	-271	-20%	-91%
Shrub litter	1,052	136	1,304	241	804	271	130	24	-248	-1,173	-24%	-90%
Shrub foliage	435	64	575	104	283	90	58	10	-152	-517	-35%	-90%
Shrub 1h	1,020	163	1,322	237	549	181	132	24	-471	-1,190	-46%	-90%
Shrub 10h	693	82	835	145	460	150	83	14	-234	-751	-34%	-90%
Shrub 100h	1,031	126	1,151	212	651	217	288	53	-380	-863	-37%	-75%
Shrub 1000h	10	3	0	0	1	1	0	0	-8	0	-85%	0%
Shrub total	2,779	591	3,490	910	1,555	648	538	100	-1,224	-2,951	-44%	-85%
Tree litter	3,184	1,124	3,830	1,178	2,819	1,018	307	61	-365	-3,523	-11%	-92%
Tree foliage	4,684	1,004	5,659	914	4,809	858	410	111	125	-5,248	3%	-93%
Tree 1h	2,725	718	3,807	758	2,975	656	937	219	250	-2,870	9%	-75%
Tree 10h	4,566	956	4,531	875	4,659	813	2,734	413	92	-1,797	2%	-40%
Tree 100h	5,486	1,380	5,013	1,490	5,694	1,185	3,266	812	208	-1,748	4%	-35%
Tree 1000h	9,496	2,769	10,985	2,038	10,116	2,421	4,139	1,101	620	-6,845	7%	-62%
Tree total	30,981	6,359	30,849	5,731	31,701	5,422	13,542	2,402	720	-17,308	2%	-56%
Aboveground total	38,029	6,643	39,356	5,907	36,879	5,938	14,485	2,350	-1,151	-24,871	-3%	-63%

Table 3.0 Means and standard errors for soil and biomass component carbon and nitrogen before and after burning.

	Pre-burn %C		Post-burn %C		Pre-burn %N		Post-burn %N	
	<u>Mean</u>	<u>S.E.</u>	<u>Mean</u>	<u>S.E.</u>	<u>Mean</u>	<u>S.E.</u>	<u>Mean</u>	<u>S.E.</u>
Soil 0-8 cm	2.39	0.19	5.00	0.72	0.19	0.01	0.28	0.03
Soil 8-23 cm	1.46	0.08	2.00	0.35	0.13	0.01	0.15	0.02
Soil 23-38 cm	0.93	0.08	1.10	0.12	0.09	0.01	0.10	0.01
Soil 38-52 cm	0.81	0.04	0.83	0.08	0.08	0.00	0.09	0.01
Roots 0-52 cm	32.18	0.08	32.04	1.29	0.83	0.03	1.13	0.05
Herbaceous understory	43.59	0.17	45.08	0.20	1.48	0.06	2.11	0.05
Shrub litter	33.35	1.79	32.64	1.75	1.40	0.11	1.92	0.15
Shrub foliage	50.35	0.16	45.70	0.13	1.55	0.04	1.86	0.05
Shrub 1h	48.07	0.13	50.10	0.10	0.61	0.03	0.55	0.02
Shrub 10h	49.96	0.16	53.38	0.55	0.32	0.02	0.50	0.02
Shrub 100h	49.87	0.09	52.27	0.33	0.32	0.01	0.19	0.01
Shrub 1000h	49.87	0.09	52.27	0.33	0.32	0.01	0.19	0.01
Tree litter	47.20	0.49	43.63	2.63	1.14	0.05	1.56	0.09
Tree foliage	53.78	0.05	48.82	0.19	0.89	0.02	1.06	0.01
Tree 1h	55.57	0.02	51.58	0.26	0.37	0.00	0.45	0.02
Tree 10h	54.78	0.17	50.98	0.13	0.35	0.02	0.23	0.01
Tree 100h	51.55	0.16	50.92	0.12	0.21	0.00	0.19	0.01
Tree 1000h	51.67	0.28	51.90	0.43	0.17	0.01	0.20	0.02

Table 4.0 Means, standard errors (S.E.), mass change, and % change for ecosystem component carbon and nitrogen mass before and after the prescribed fire on control and burn plots.

	Pre-burn Carbon Mass (kg / ha)				Post-burn Carbon Mass (kg / ha)				Mass Change (kg /ha)		Mass Change %	
	Control		Burn		Control		Burn		Control	Burn	Control	Burn
	Mean	S.E.	Mean	S.E.	Mean	S.E.	Mean	S.E.				
Soil	66,789	3,238	47,053	2,678	75,337	3,953	75,764	9,777	8,548	28,711	13%	61%
Roots	4,242	509	4,288	717	4,242	509	2,956	147	0	-1,332	0%	-31%
Litter	1,475	472	2,204	318	1,863	730	166	20	388	-2,038	26%	-92%
Herbaceous	118	28	130	8	90	31	12	4	-27	-118	-23%	-91%
Shrub	1,412	146	1,616	154	943	85	275	7	-469	-1,341	-33%	-83%
Tree	16,744	2,563	15,901	1,117	13,723	3,763	5,889	728	-3,021	-10,012	-18%	-63%
Aboveground Biomass	19,748	2,051	19,851	1,489	16,620	4,565	6,342	717	-3,129	-13,509	-16%	-68%
Total Ecosystem	90,779	4,467	71,192	2,008	96,198	6,566	85,062	8,934	5,420	13,870	6%	19%
	Pre-burn Nitrogen Mass (kg / ha)				Post-burn Nitrogen Mass (kg / ha)				Mass Change (kg /ha)		Mass Change %	
	Control		Burn		Control		Burn		Control	Burn	Control	Burn
	Mean	S.E.	Mean	S.E.	Mean	S.E.	Mean	S.E.				
Soil	5,342.4	246.8	4,341.7	196.9	5,113.7	226.4	5,874.9	268.5	-229	1,533	-4%	35%
Roots	110.2	13.8	112.2	19.3	110.2	13.8	101.9	3.4	0	-10	0%	-9%
Litter	65.8	18.3	99.9	7.7	60.5	18.7	8.3	0.7	-5	-92	-8%	-92%
Herbaceous	3.5	0.8	3.3	0.2	2.8	0.7	0.4	0.1	-1	-3	-18%	-87%
Shrub	11.6	1.3	12.9	1.8	10.0	1.3	1.8	0.0	-2	-11	-13%	-86%
Tree	160.8	11.2	174.6	8.8	166.8	21.5	52.7	1.1	6	-122	4%	-70%
Aboveground Biomass	241.5	13.6	290.7	16.0	240.2	18.6	63.3	1.3	-1	-227	-1%	-78%
Total Ecosystem	5,694.1	244.4	4,744.6	218.3	5,464.0	230.7	6,040.1	267.6	-230	1,295	-4%	27%

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Figure 1.0 Diagram which represents the two treatment plots, eight sub-plots, and three microsites within the study area.

Figure 2.0 Photo series of the burn plots before the prescribed fire, during the burn implementation, and post fire.

Figure 3.0 Diagram of the sub-plot layout including the five belt transects.

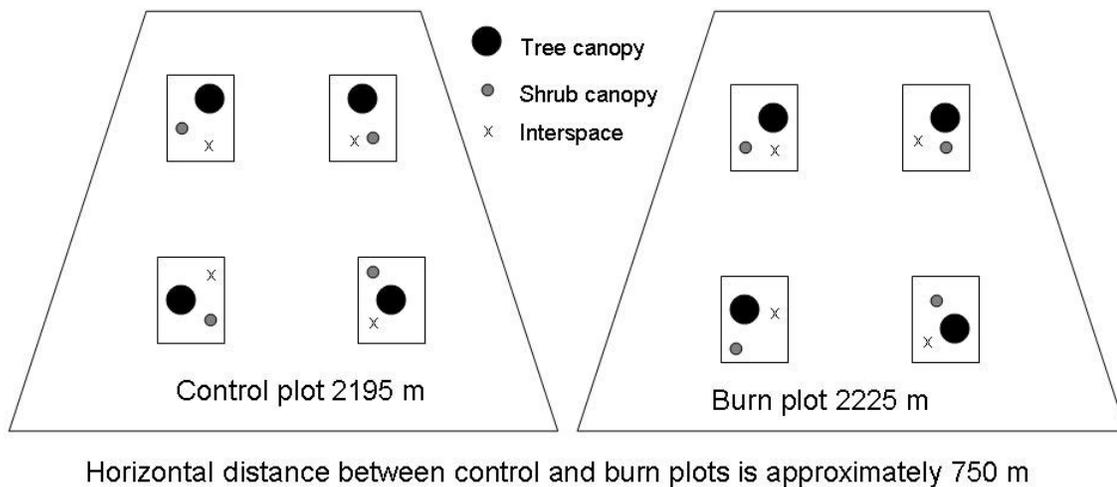
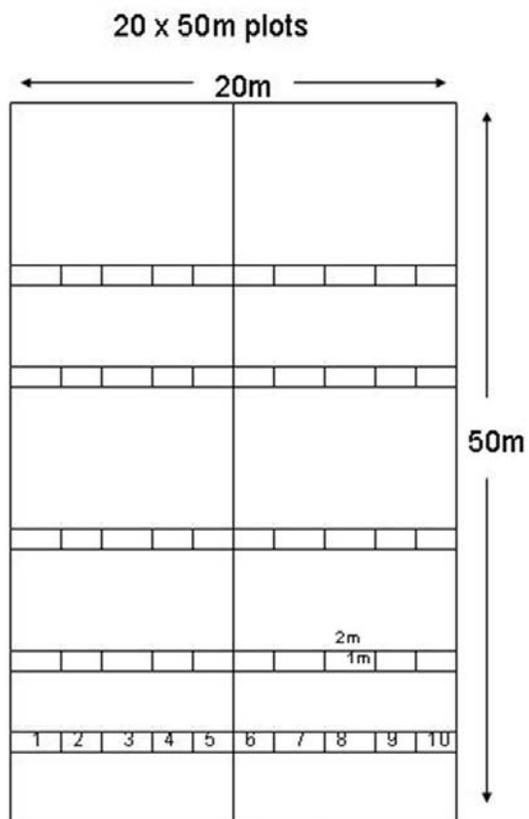
Figure 1.0

Figure 2.0



Figure 3.0



Modeling Effects of Pinyon-juniper Expansion and Prescribed Fire on Ecosystem Biomass, Carbon, and Nitrogen Using an Empirically Derived Spreadsheet Calculator

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Abstract

Sagebrush ecosystems are one of the most threatened ecosystems in North America due to woodland expansion and exotic annual grass invasion. Some policy makers and interest groups have suggested that woodland expansion will lead to increased carbon storage on the landscape. This might be true temporarily, but woodland encroachment could ultimately turn these sensitive systems into carbon sources as a result of increased susceptibility to wildfire. To assess this potential we used empirically derived data from a Joint Fire Sciences Program Demonstration Area to develop a Microsoft Excel™ based ecosystem biomass, carbon, and nitrogen calculator. The model uses input for tree cover, soil chemistry, soil physical properties, and vegetation chemistry to estimate biomass, carbon, and nitrogen accumulation on the landscape with woodland expansion. The model also estimates C and N losses associated with prescribed burning. We estimate that aboveground biomass accounts for < 10 % of ecosystem C and N at low tree cover, but can be nearly 60 % ecosystem C and 14 % ecosystem N in closed canopy woodlands. Prescribed burning removes aboveground biomass at all tree cover estimates, but may increase soil C and N at low tree covers, and decrease soil C and N at high tree cover. The model serves as a tool by which we are able to assess our understanding of the system and identify knowledge gaps which exist for these ecosystems. We believe that further work is necessary to quantify herbaceous biomass, root biomass, and soil C and N with woodland encroachment and prescribed fire.

Keywords: carbon sequestration, nitrogen balance, prescribed fire, woodland encroachment, fuels management

1. Introduction

Carbon storage and management of ecosystems for carbon retention has become an important issue for private and public land managers (Canadell and Raupach 2008). The issue is not limited to tropical or mesic environments, but also is a consideration in management of semi-arid and arid ecosystems (Anser et al. 2003). Sagebrush-steppe ecosystems in the intermountain western U.S. are currently threatened by expansion of pinyon pine (*Pinus monophylla* and *Pinus edulis*) and juniper (*Juniperus osteosperma* and *Juniperus occidentalis*) at intermediate elevations (Miller and Tausch 2001). Some have argued that tree expansion will provide an offset for greenhouse gas emissions (Norris et al. 2001, Canadell and Raupach 2008). However, tree expansion eliminates sagebrush vegetation, which can provide forage for native animal species and reduce surface erosion. Tree encroachment also facilitates high intensity wildfire and exotic annual grass invasion (Miller and Tausch 2001, Chambers et al. 2007). Ultimately little is currently known about ecosystem C and N within expansion woodlands, nor about the processes which drive accumulation and retention.

Fire has an integral role in maintaining sagebrush steppe ecosystems, and land managers would like to use prescribed fire as a tool for ecosystem maintenance. Because these systems are prone to fire, dense woodlands cannot be maintained on the landscape without severely increasing the risk of high intensity wildfire (Hurteau et al. 2008). When wildfire burns though dense woodlands large amounts of aboveground carbon and some soil carbon are released back to the atmosphere. Prescribed fire also releases carbon to the atmosphere, but may do so at a more moderate rate (Hurteau et al. 2008).

Ecological models can be an important tool for scientists seeking to understand ecosystem processes and land managers planning and implementing projects (Verburg and Johnson 2001). However, some models are complex to use and difficult to calibrate (Tiktak and Van Grinsven 1995; Van Tongeren 1995). Scientists and managers need a tool that will allow them to account for carbon in expansion pinyon and juniper woodlands and the effects of land management on carbon loss and retention. In this manuscript we develop such a tool based on our current knowledge of expansion woodlands, and we utilize empirical data collected from a Joint Fire Sciences Program Demonstration area. The goals of this tool are to provide scientists with feedback on the current state of knowledge in expansion woodlands, and to identify where knowledge gaps exist, so that we can modify our data collection efforts.

2. Methods

2.1 Experimental area

The data for the calculator is derived from 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 intermittent 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 skeletal mixed frigid Typic Haploxerolls (Rau et al. 2005).

The vegetation is characterized by sagebrush (*Artemisia tridentata vaseyana*) and single leaf pinyon (*Pinus monophylla*) with lesser cover of Utah juniper (*Juniperus 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. *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 (1% crown cover, 10,000 kg/ ha) to high tree dominance (87% crown cover, 115,000 kg/ ha) (Reiner 2004).

2.2 Study design and data collection

Study plots were located on northeast facing alluvial fans at elevations ranging from 2,072 m and 2,346 m. Forty-seven 0.1 ha sub-plots were sampled before the prescribed burn treatment. USDA Forest Service fire personnel burned half of the study plots on May 11-14, 2002 under favorable weather conditions (Air temp < 32°C, RH > 15%, wind speed < 9 m•s⁻¹, and gravimetric fuel moisture ≈ 40%). Fire behavior was characterized by creeping ground fire with single and group tree torching.

2.2.1 Soil and roots

Eight sub-plots at elevation 2195 m and 2225 m were sampled to determine soil characteristics. The four plots at 2195 m were controls and the four plots at 2225 m received the prescribed burn treatment. Soil pits were dug to a depth of 52 cm, and the

soil horizons were identified. Depth increments for sampling were assigned to the approximate center of the soil A₁ horizon and subsequent 15 cm increments (0-8, 8-23, 23-38, and 38-52 cm). Bulk density samples were collected from each depth using a 93 cm³ soil core. Soil samples were taken from each of three microsites (under tree, under shrub, interspace) for each depth using a 10 cm diameter bucket auger. Sampling was conducted in fall 2001 and again in 2002 to determine fire effects on soil C and N (Rau et al. 2009a). Mass of soil was determined using the formula

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

where t = thickness (cm) of the soil horizon, Db = bulk density (g cm^{-3}) of that horizon, $> 2 \text{ mm } \%$ is the volume percentage coarse fragment of that horizon, and F = conversion factor ($1 \text{ g } 1,000,000 \text{ ug}^{-1} * 1 \text{ kg } 1,000 \text{ g}^{-1} * 10,000 \text{ cm}^2 1 \text{ m}^{-2} * 10,000 \text{ m}^2 \text{ ha}^{-1}$). The sum of all horizons constitutes the entire mass of soil sampled.

Root biomass was estimated on each sub-plot, microsite, and soil depth post-hoc in 2005 using a core device (Rau et al. 2009b). Roots were separated from soil and rock by sieve and flotation, dried, and then weighed. The mass of roots was determined using the formula

$$\text{Kg ha}^{-1} = (d)(Dr)(G)$$

where d = depth increment (cm) of the core, Dr = rooting density (g cm^{-3}), and G = conversion factor ($100,000 \text{ cm}^2 \text{ g}^{-1}$). The sum of all depth increments sampled constitutes the entire mass of roots sampled.

2.2.2 Understory biomass

Understory vegetation in each sub-plot was sampled in fifty 1 x 2 m micro-plots located contiguously along belt transects perpendicular to the long axis of the sub-plot (Reiner

2004). Sampling occurred in summer 2001 and again in summer 2002 to determine how much biomass was lost due to prescribed burning. Understory cover and volume were estimated by species. Volumes vs. mass regression models were created using non-linear regressions (Tausch and Tueller 1988). Biomass was reported as live and dead categories of 1 hour (< 6.35 mm), 10 hour (6.35 – 25.4 mm), and 100 hour (25.4 – 76.2 mm) timelag fuels (Reiner 2004). Biomass regression r^2 coefficients ranged from 0.31 for some small grass species to 0.82 for common shrubs such as *Artemisia* and *Chrysothamnus spp.*

Shrub litter mats were sampled under 18 rabbitbrush (*Chrysothamnus viscidiflorus*) and 36 sagebrush (*A. tridentata vaseyana* and *A. tridentata wyomingensis*) shrubs using a 100 cm² plot frame (Brown 1982). Shrub litter weight was estimated for rabbitbrush and sagebrush species using regression equations based on relationships developed between shrub crown area and litter mat weight (Reiner 2004). Regressions produced r^2 values of 0.74 and 0.75 respectively.

2.2.3 Tree biomass

Tree biomass was estimated in each sub-plot by measuring total tree height, crown height, longest crown diameter, crown diameter perpendicular to the longest diameter, and trunk diameter just above the root crown for each individual tree rooted within the sub-plot. Crown measurements were of the portion of the crown containing foliage. Tree measurements were collected in summer 2000. Volume vs. mass regression equations were used to estimate biomass from tree volume (Tausch in press). Volume-mass regression equations were developed for pinyon and juniper trees by measuring and

harvesting 18 trees from just outside of sub-plots prior to and after burning. Trees ranged in size from 0.5 m to 6.3 m in height (Tausch in press).

Tree litter mats were sampled under 17 pinyon pines with crown diameters ranging from 1.8 to 7.0 m. Complete crown and litter mat dimensions were collected in order to derive relationships between litter mat area and litter mat mass. One to three 33.5 cm diameter rings were evenly spaced on either side of the tree bole depending on its size. Litter was removed by horizon (O_i , O_e , and O_a), bagged, and brought back to the lab. Litter samples were floated to remove mineral fragments, dried, separated by fuel time lag size and weighed. Equations were then developed between litter biomass and tree crown area to estimate total sub-plot tree litter mass.

2.3 Carbon and nitrogen

All soil and root samples and six vegetation sub-samples were randomly selected from each biomass component (grass, forb, shrub litter, live and dead shrub 1-100 hr fuels, tree litter, live and dead tree 1-1000 hr fuels) pre and post-burn for chemical analyses. Samples were ground in a Wiley[®] mill and analyzed for total carbon and nitrogen concentration using a LECO Truspec[®]. The percent C and N for soil and each biomass component was multiplied by the total mass of that component in each sub-plot.

2.4 Model Description

The carbon and nitrogen calculator is a simple spreadsheet model based on empirical soil and biomass information. The model requires inputs for tree cover, soil physical characteristics, soil chemical characteristics, and vegetation chemical characteristics. Default data is provided if users are unable to generate all of the needed parameters (Figure 1). The model was created in Microsoft Excel[™] and uses 28 regression equations

embedded in 10 worksheets for calculating different ecosystem component biomass, C and N (Table 1). The model uses individual regression equations to calculate soil mass, root biomass, herbaceous biomass, shrub litter, shrub, foliage, shrub woody fuels, tree litter, tree foliage, and tree woody fuels before and after prescribed fire. The model also provides data on soil, root, litter, herbaceous, shrub, and tree biomass, carbon and nitrogen. Biomass and nutrient pools are calculated from regression models based on tree cover. The model will create 6 chart graphics based on the input, and return data on ecosystem biomass, carbon, and nitrogen, before and after prescribed fire based on tree cover (Figures 1-4).

It was determined that crown area or tree cover had the highest correlation with all ecosystem components and was the easiest parameter to measure within the sub-plot area. Therefore, the model was designed so that all ecosystem components could be estimated using tree cover. A series of regression equations were developed using observed tree cover from the 41 sub-plots as the independent variable or x – axis, and treating all other ecosystem components as dependant variables or y - axis. Individual regressions were developed for pre-burn shrub cover, pre-burn herbaceous cover, and pre and post-burn herbaceous biomass, shrub litter mass, shrub foliage mass, shrub 1 - 1000 hr fuel mass, tree litter mass, tree foliage mass, and tree 1 – 1000 hr fuel mass (Table 1). In most cases the best fitting regression was a second order polynomial (Table 1). Regressions were then used to predict the mass of individual ecosystem components based on tree cover ranging from 0 – 100 %. Because a second order polynomial has minimum value represented by the apex of a parabola, regression predictions were truncated once the equation crossed the x – axis or the minimum value was obtained. The regressions for

shrub cover vs. tree cover were used to determine the weighting for soil and root mass at each microsite. Where interspace cover was determined to be $[100 - (\text{tree cover} + \text{shrub cover})]$.

2.5 Verification

Six sub-plots were withheld from the development of regressions to verify the predicted values for tree biomass based on tree cover. Observed vs. predicted values were plotted against each other for tree biomass components to determine how well the regression equations predicted field observed data. The plots used to verify the predictions of tree biomass were fit relatively well by the model. The slope of the regression between observed and predicted biomass ranged from 0.84 to 0.99 for pre and post burn fuels (Table 2). The model is very good at predicting fuels at intermediate tree cover, but additional sample points will be needed to verify high and low tree cover values (Figure 5). The regression equations developed for tree cover vs. herbaceous biomass and shrub fuels had relatively low r^2 values compared to equations developed for tree fuels (Table 1). We believe this is due to the variability in abiotic conditions across our plots. For example plots at low elevation have more coarse grained soil and lower annual precipitation than plots near the top of the canyon. Further work is need to determine how these abiotic factors influence herbaceous and shrub fuels. It may be necessary to develop a more complex series of regressions using abiotic factors to more accurately estimate herbaceous biomass. The regression equations developed for tree fuels fit surprisingly well across the environmental gradient of our plots (Table 1, 2; Figure 5). This may be less influenced by variation in site characteristics and grow somewhat uniformly over a range of abiotic conditions. We intend to test our regression equations and overall model

on plots currently being measured from a broad geographical area with a range of climatic conditions. Like our Underdown Canyon plots, treatments include controls and burns. The new plots also should allow us to better estimate the influence of woodland expansion and burning on soil and roots.

3. Results and discussion

3.1 Soil and roots

3.1.1 Influence of woodland encroachment on soil, roots, carbon, and nitrogen

Because soils were sampled by microsite, and then weighted by the microsites cover on the landscape, the model produces only small variation in soil C and N due to the small absolute differences in C and N content found between microsites. Although not shown by the calculator: modeled soil mass decreases per unit depth with increasing tree cover. This is because soil bulk density is highest at vegetation interspaces, and lower under tree and shrub canopies. To correct for this phenomenon under shrub and under tree canopy, microsites could be sampled to a deeper depth to account for the increase in volume (Federer et al. 1992). As soil bulk density decreases on our plots, concentrations of soil C and N increase. Because content is a function of mass and concentration the soil pools of carbon (48,000 – 50,039 kg ha⁻¹) and nitrogen (3, 866 – 3,662 kg ha⁻¹) remain relatively constant as tree cover increases from 0 – 100 % (Figure 2a, 3a). To better account for changes in soil C and N with tree encroachment, a more rigorous sampling regime would need to be implemented which samples soils over a range of tree cover. We are currently undertaking these efforts to improve the model.

Similarly, root biomass remains relatively constant with increasing tree cover in our model (13,500 – 14,500 kg ha⁻¹) (Figure 4a). By collecting root samples from individual microsites within a sub-sample of plots with consistent tree cover we have most likely missed true increases or reductions in root biomass associated with increasing tree dominance. Roots can extend several meters from their point of origin and cannot be isolated by species using the methods we used for sample collection (Sturges 1977). Therefore, our measurements of root biomass could be more a function of the species composition on our plots and do not accurately represent shifts in tree dominance. We are working with a more extensive data set from across a range of tree covers which indicates that root biomass may actually increase substantially with tree expansion (Rau unpublished data). Because modeled root mass remains relatively constant with increasing tree cover, the pools of C and N in roots remains relatively constant as well (Figure 2a – 4a).

3.1.2 Effects of prescribed fire on soil, roots, carbon, and nitrogen

Prescribed burning resulted in an increase in soil C and N concentrations, but only near the soil surface (Rau et al. 2009a). Carbon and nitrogen concentrations may have slightly decreased below the soil surface under trees (Rau et al. 2009a). The observed increase in near surface soils is likely due to the incorporation of unburned residues and deposition of organic distillates (Johnson and Curtis 2001). At low tree cover our model predicts that prescribed burning increases soil C and N due to the increased C and N concentration of near surface soils at interspace and undershrub microsites. The mass of C and N incorporated into soils is maximized at very low tree cover (9,250 kg ha⁻¹ C and 412 kg ha⁻¹ N) and decreases with increasing tree cover to values near 70%. Once this 70% tree

cover threshold has been crossed, prescribed fire results in net loss of C and N due to the slight decrease in subsurface C and N concentrations under trees. The greatest C and N losses occur at 100 % cover ($-5,888 \text{ kg ha}^{-1} \text{ C}$ and $-342 \text{ kg ha}^{-1} \text{ N}$) (Figure 2, 3). These relationships indicate that as tree cover increases across the landscape the likelihood of fire removing soil C and N may increase.

Root biomass, C, and N were all reduced by fire in our model. Biomass and carbon losses increased with tree cover, but nitrogen losses tended to decrease with increasing tree cover and fire (Figure 2 - 4). The results for root nitrogen pools reflect the decomposition process which over time reduces the C:N ratio of root material remaining after fire (Entry and Backman 1995). Herbaceous biomass and shrub roots started with lower C:N values than tree roots. Therefore, these materials will lose more nitrogen over a shorter period of time. Tree roots with a relative high C:N ratio will take longer before they begin to lose N from decomposition processes (Entry and Backman 1995).

3.2 Aboveground biomass, carbon, and nitrogen

3.2.1 Influence of woodland encroachment on aboveground biomass, carbon, and nitrogen

Understory herbaceous and shrub biomass are maximized at the lowest values of tree cover with our site supporting 400 and $6,394 \text{ kg ha}^{-1}$ of herbaceous and shrub biomass respectively when modeled without trees (Figure 4a). Understory biomass accounts for $4,537 \text{ kg ha}^{-1} \text{ C}$ and $263 \text{ kg ha}^{-1} \text{ N}$ at zero tree cover, with shrubs and shrub litter accounting for nearly all C and N. Aboveground C and N constitute 8 and 6 % of total ecosystem pools, respectively. We recognize that these values are probably site specific, and are dependent on abiotic factors including soil texture, soil nutrient availability,

precipitation, and temperature. This was evident in our data set which spans 300 m of elevation and 20 cm of annual precipitation. We will require more data from multiple sites across the range of abiotic conditions to incorporate these kinds of variables into the model.

Adding trees to the landscape rapidly increases total biomass and shifts the balance of mass distribution to aerial mass. The model estimates that at only 10 % tree cover, tree biomass exceeds total understory biomass, and total aboveground biomass nearly triples to 19,531 kg ha⁻¹ (Figure 4a). At 15 % tree cover, litter becomes the largest portion of understory biomass compared to shrubs and herbaceous vegetation.

Understory vegetation biomass continues to decline toward zero as tree cover increases with shrub biomass decreasing more rapidly than herbaceous biomass (Figure 4a). By the time tree cover reaches 70 %, aboveground biomass is estimated at 91,181 kg ha⁻¹ (48,529 kg ha⁻¹ C). At this cover, aboveground pools of carbon equal or exceed soil carbon pools (Figure 2a). This level of tree dominance represents a stage where herbaceous understory has been eliminated through competition, and the site is at risk for catastrophic wildfire and conversion to exotic annual grasses (Miller and Tausch 2001).

At 100 % tree cover, the model predicts our site would exceed 136,000 kg ha⁻¹ in aboveground biomass and 73,000 kg ha⁻¹ in aboveground biomass and aboveground C. Aboveground C pools would then represent 57 % of total ecosystem carbon (Figure 2a). Aboveground nitrogen pools at 100 % tree cover would near 750 kg ha⁻¹ which is still only 14 % of total ecosystem N (Figure 3a).

3.2.2 Effects of prescribed fire on aboveground biomass, carbon, and nitrogen

Prescribed burning at tree cover values near zero removes 83 % of aboveground biomass and carbon, or 7,943 and 3,777 kg ha⁻¹ respectively (Figure 2, 4). Burning releases slightly higher percentage (87 %) aboveground nitrogen (230 kg ha⁻¹) due to the high N concentration in herbaceous biomass at zero tree cover (Figure 3). The values for N lost due to burning are lower than the estimated N increase for soil pools after burning. We feel that additional data will need to be collected to solve this discontinuity in the model at low tree cover values.

As tree cover increases to 50 %, prescribed burning results in a 61 % reduction in aboveground biomass including 90 % of herbaceous vegetation, litter, and shrubs (Figure 4). Roughly 55 % of tree biomass is consumed (> 90 % foliage and 1 h stick); however, the majority of 10, 100, and 1000 hr tree fuels remain following the fire. Biomass consumed by burning accounts for 21,477 kg ha⁻¹ of C and 246 kg ha⁻¹ N released by combustion. These estimates represent 33 and 6 % of total ecosystem C and N. Not all C and N lost from biomass escapes in gaseous form, but a portion is incorporated into soils as ash, charcoal, and organic distillates (Johnson and Curtis 2001). The result is smaller losses of ecosystem C and N. Our model predicts that actual losses may be closer to 23 % C and 2 % nitrogen. Tree cover near 50 % is often considered a threshold for recovery of healthy sagebrush steppe ecosystems following fire (Miller and Tausch 2001).

Although prescribed fire is not likely to be used at tree cover approaching 100 %, it is worth contemplating this scenario as it may apply to certain wildfire situations. Burning at 100 % tree cover is modeled to remove 56, 59, and 46 % of aboveground biomass, C, and N respectively (Figure 2 - 4). The loss of C from aboveground biomass by this type of burn (43,259 kg ha⁻¹) is more than twice the amount lost at 50 % tree cover and more

than 10 times the amount lost from burning at tree cover near zero. From a whole ecosystem perspective this represents a 40 % decrease in ecosystem carbon from the existing pool. However, this may not be a net loss when viewed from the perspective of a woodland developing from a treeless sagebrush stand. The ultimate trend of the ecosystem as a source or sink may depend on the vegetation which develops after the fire. If a healthy sagebrush system returns then perhaps carbon is gained. If an annual grass monoculture returns and the fire return interval decreases, then the ecosystem may become a C source. More data on standing dead and root decomposition rates are needed before more solid conclusions can be made.

Because of the low concentration of nitrogen in large woody materials the amount of nitrogen lost at high tree cover is not much greater than at intermediate tree cover. Our model estimates that 289 kg ha^{-1} nitrogen is lost from aboveground biomass by burning at high tree cover this is less than 8 % of total ecosystem N.

3.3 Ecosystem level budgets

At low tree cover, aboveground biomass represents a small portion (< 10 %) of total ecosystem C and N. As tree cover increases biomass rapidly increases on the landscape. By the time an ecosystem recovery threshold is crossed near 50 % tree cover, aboveground biomass increases seven fold and accounts for 40% of ecosystem C and 10 % ecosystem N. In a closed canopy woodland biomass is 14 times higher than on treeless plots, and accounts for 57 and 14 % of total ecosystem C and N.

Burning of low density woodlands effectively removes aboveground biomass, but has positive effects on soil C and N pools. Our model currently over estimates the amount of C and N which can be moved from aboveground compartments into soil pools at low tree

cover, but we believe the process is correct. Burning woodlands near the threshold value of 50 % tree cover is also effective at reducing aboveground biomass, and still has positive effects on soil C and N pools. We believe that further losses of remaining roots and large woody fuels will occur via decomposition, but that this is a relatively slow process due to the arid nature of the system (Contant et al. 1998). In closed canopy woodlands, large amounts of biomass, C, and N are removed by fire. Our model estimates that burning also will reduce soil C and N in closed canopy woodlands resulting in total net losses from all ecosystem pools of carbon and nitrogen.

3.4 Assessing variable burn frequency

A single prescribed fire does not maintain an ecosystem in a permanent state, and so it is necessary to determine how burn frequency will influence ecosystem C and N budgets. We present here three different fire return intervals in to assess repeated management treatments. The model estimates that if our woodland plots were burned every time they reached 25 % tree cover they would lose 12,124 kg ha⁻¹ C and 236 kg ha⁻¹ N from aboveground biomass (Figure 6a, 7a). If the plots were burned every time they reached 50 % tree cover they would lose 21,477 kg ha⁻¹ C and 246 kg ha⁻¹ N from aboveground biomass (Figure 6a, 7a). Finally assuming that the plots burned under closed canopy conditions the model predicts that 43,259 kg ha⁻¹ C and 289 kg ha⁻¹ N would be lost from aboveground biomass. Assuming that managers would need to burn the 25 % cover plots four times and the 50 % cover plots twice before the closed canopy plots burned once, we estimate that burning at 25 % cover removes 48,496 kg ha⁻¹ C and 944 kg ha⁻¹ N from aboveground biomass, and burning at 50 % cover removes 42,954 kg ha⁻¹ C and 492 kg ha⁻¹ N. This suggests that multiple burns release roughly the same amount of C as one

high intensity burn, but that multiple burns could release 2 - 3 times more nitrogen than one high intensity burn. These scenarios may not truly reflect C and N losses from burning because they assume that no ash, organic distillates, or charcoal are incorporated into soils. These scenarios also assume that standing dead biomass does not decompose after the burn.

To better understand how burn return intervals influence ecosystem C and N we re-ran these scenarios incorporating our estimates for soil and root changes due to fire. The new runs present a much different outcome. Burning at 25 % cover resulted in ecosystem losses of 6,526 kg ha⁻¹ C and a gain of 76 kg ha⁻¹ N (Figure 6b, 7b). We accept that a gain of ecosystem N is not possible immediately following the burn, but assume that some N is incorporated into soil. Over four burn cycles total ecosystem C loss is 26,104 kg ha⁻¹ and N gain totals 304 kg ha⁻¹. The model estimates burning at 50 % tree cover results in loss of 19,191 kg ha⁻¹ ecosystem C and 84 kg ha⁻¹ ecosystem N (Figure 6b, 7b). Over two burn cycles this totals 38,380 kg ha⁻¹ C and 168 kg ha⁻¹ N. Burning in closed canopy stands may result in total ecosystem losses of 50,908 kg ha⁻¹ C and 630 kg ha⁻¹ N. By considering soil and root processes with prescribed fire we may hypothesize that more frequent burning may actually result in lower losses of ecosystem C and N because some aboveground biomass is incorporated into soil as ash, organic distillates, and charcoal. However, the results presented here still do not incorporate decomposition of standing dead biomass after fire. Additionally the discontinuities in our model which incorporate more N into soil than could be gained from aboveground biomass at low tree cover require reconsideration of sampling protocols and model assumptions. For this reason we must assume that either the current model or our sample regime is inadequate to properly

estimate changes in soil C and N, and we must make an effort to collect better data on soils, roots, and decomposition. Ultimately a model which is more process based rather than empirically derived may provide better estimates of biogeochemical transitions associated with prescribed fire in pinyon-juniper woodlands. We continue to collect a wider range of data and encourage other researchers to pursue these measurements as well.

4. Conclusions

Sagebrush ecosystems are one of the most threatened ecotypes in North America (Noss et al. 1995). Woodland encroachment and the threat of exotic invasions presents land managers with difficult decisions on how best to maintain these ecosystems on the landscape and meet the goals of carbon retention.

The empirically derived C and N calculator we have developed is an initial effort to estimate carbon and nitrogen pools in expansion pinyon and juniper woodlands. We believe the calculator provides insight into our lack of knowledge of belowground processes and decomposition, and provides us with additional hypotheses that need to be addressed. The model shows that most C and N on our plots is stored in soils at low tree cover, but that as tree cover increases aboveground biomass can account for nearly 60 % of ecosystem C and 14 % ecosystem N. Tree cover approaching 50 % represents a possible threshold which once passed leaves expansion woodlands prone to catastrophic wildfire and exotic invasion (Miller and Tausch 2001). Both intense wildfire and changing fire regimes associated with exotic annual grass invasion could lead these systems to become carbon sources rather than sinks (Bradley et al. 2006). We will continue to collect data over a large range of abiotic conditions in order to test and refine

this model. Ultimately a process based model could replace this empirically derived version.

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Table 1.0 Best fit regression equations and r^2 values for tree cover vs. shrub and herbaceous cover and biomass components from unburned and burned plots

Cover	Pre-Burn		Post-Burn	
	Regression Equation	r^2	Regression Equation	r^2
Herbaceous	$-0.0011x^2 - 0.0039x + 12.096$	0.38		
Shrub	$0.0059x^2 - 1.1437x + 55.37$	0.75		
Biomass				
Herbaceous	$-0.0133x^2 - 2.9919x + 402.67$	0.54	$-0.2796x + 36.245$	0.04
Shrub litter	$0.6108x^2 - 92.772x + 3559.7$	0.71	$0.1106x^2 - 14.88x + 496.9$	0.85
Shrub foliage	$0.0538x^2 - 13.881x + 780.89$	0.71	$0.0121x^2 - 2.2412x + 103.68$	0.78
Shrub 1h	$0.1801x^2 - 39.392x + 2055.7$	0.80	$0.0343x^2 - 5.9694x + 262.07$	0.87
Shrub 10h	$0.1593x^2 - 30.55x + 1475.7$	0.76	$0.027x^2 - 4.3368x + 179.23$	0.90
Shrub 100h	$0.1927x^2 - 40.755x + 2072.5$	0.66	$0.0927x^2 - 14.864x + 606.27$	0.89
Shrub 1000h	$0.0176x^2 - 1.665x + 42.02$	0.61	$0.0099x^2 - 0.8313x + 17.518$	0.99
Tree litter	$0.0037x^2 + 176.19x$	0.79	$-0.0493x^2 + 13.485x$	0.69
Tree foliage	$0.4714x^2 + 159.14x$	0.99	$0.0718x^2 + 13.51x$	0.99
Tree 1h	$0.2626x^2 + 125.2x$	0.98	$0.0986x^2 + 30.908x$	0.99
Tree 10h	$0.4499x^2 + 141.16x$	0.99	$0.2576x^2 + 82.369x$	0.99
Tree 100h	$0.7038x^2 + 174.27x$	0.97	$0.3588x^2 + 112.95x$	0.99
Tree 1000h	$0.8396x^2 + 351.88x$	0.96	$1.2953x^2 + 146.75x$	0.97

Table 2.0 Slope of the correlation and r^2 for observed vs. predicted values on the model verification plots

	Slope	r^2
Pre-burn litter	0.84	0.42
Post-burn litter	0.96	0.76
Pre-burn foliage	0.97	0.92
Post-burn foliage	0.95	0.98
Pre-burn 1 hr	0.93	0.84
Post-burn 1 hr	0.95	0.99
Pre-burn 10 hr	0.96	0.86
Post-burn 10 hr	0.96	0.99
Pre-burn 100 hr	0.94	0.82
Post-burn 100 hr	0.95	0.99
Pre-burn 1000 hr	0.99	0.82
Post-burn 1000 hr	0.95	0.92

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Figure 1.0 A screen shot of the calculator input and output window. Once the input parameters for soil and vegetation are set tree cover can be manipulated. Changes in biomass, C and N after burning can be viewed in the lower pane.

Figure 2.0 The output graphics produced for ecosystem C pre burn (panel a) and post burn (panel b). Post burn graphics represent the residual carbon found on the site immediately after burning at any given cover percentage. Mass C kg ha^{-1} is stacked, adding values for each ecosystem component starting at zero, and ending at cumulative ecosystem C mass.

Figure 3.0 The output graphics produced for ecosystem N pre burn (panel a) and post burn (panel b). Post burn graphics represent the residual nitrogen found on the site immediately after burning at any given cover percentage. Mass N kg ha^{-1} is stacked, adding values for each ecosystem component starting at zero, and ending at cumulative ecosystem N mass.

Figure 4.0 The output graphics produced for ecosystem biomass pre burn (panel a) and post burn (panel b). Post burn graphics represent the residual biomass found on the site immediately after burning at any given cover percentage. Biomass kg ha^{-1} is stacked, adding values for each ecosystem component starting at zero, and ending at cumulative ecosystem biomass.

Figure 5.0 Graphics showing the predicted biomass based on regression equations, and the observed values obtained from individual verification plots.

Figure 6.0 Estimated Carbon accumulation in aboveground biomass (panel a.) and the entire ecosystem (panel b.) under three different fire return scenarios.

Figure 7.0 Estimated Nitrogen accumulation in aboveground biomass (panel a.) and the entire ecosystem (panel b.) under three different fire return scenarios.

Figure 1.0

Soil Physical Properties:

Enter values in box

Microsite	Horizon	Thickness (cm)	Db (g/cm3)	% >2mm
Under tree	A1	8	0.85	38.41
	A2	15	0.80	36.04
	B1	15	0.94	42.44
	B2	15	0.92	41.41
Under shrub	A1	8	0.80	35.98
	A2	15	1.00	45.16
	B1	15	0.95	42.72
	B2	15	1.03	46.12
Interspace	A1	8	1.08	48.66
	A2	15	1.17	52.80
	B1	15	1.29	57.97
	B2	15	1.25	56.20

Pre-Burn Vegetation Chemical Properties

Enter values in box

Vegetation Type	C%	N%
Herbaceous Roots	30.84	0.86
Herbaceous	44.01	1.58
Shrub Roots	32.61	0.89
Shrub litter	33.35	1.40
Shrub foliage	50.35	1.55
Shrub 1h	48.07	0.61
Shrub 10h	49.96	0.32
Shrub 100h	49.87	0.32
Shrub 1000h	49.87	0.32
Tree Roots	33.10	0.74
Tree litter	47.20	1.14
Tree foliage	53.78	0.89
Tree 1h	55.57	0.37
Tree 10h	54.78	0.35
Tree 100h	51.55	0.21
Tree 1000h	51.67	0.17

Pre-Burn Soil Chemical Properties (Total Concs):

Enter values in box

Microsite	Horizon	C%	N%
Under tree	A1	4.30	0.27
	A2	1.70	0.13
	B1	1.38	0.11
	B2	0.95	0.08
Under shrub	A1	4.96	0.31
	A2	1.90	0.15
	B1	1.33	0.12
	B2	1.07	0.10
Interspace	A1	2.19	0.17
	A2	1.63	0.13
	B1	1.04	0.09
	B2	0.89	0.09

Post-Burn Vegetation Chemical Properties

Enter values in box

Vegetation Type	C%	N%
Herbaceous Roots	31.00	1.11
Herbaceous	45.08	2.11
Shrub Roots	33.74	1.17
Shrub litter	32.64	1.92
Shrub foliage	45.70	1.86
Shrub 1h	50.10	0.55
Shrub 10h	53.38	0.50
Shrub 100h	52.27	0.19
Shrub 1000h	52.27	0.19
Tree Roots	31.39	1.13
Tree litter	43.63	1.56
Tree foliage	48.82	1.06
Tree 1h	51.58	0.45
Tree 10h	50.98	0.23
Tree 100h	50.92	0.19
Tree 1000h	51.90	0.20

Post-Burn Soil Chemical Properties (Total Concs):

Enter values in box

Microsite	Horizon	C%	N%
Under tree	A1	5.25	0.28
	A2	1.08	0.09
	B1	0.90	0.09
	B2	0.80	0.08
Under shrub	A1	6.63	0.36
	A2	2.05	0.16
	B1	0.98	0.10
	B2	0.78	0.09
Interspace	A1	3.13	0.20
	A2	2.88	0.19
	B1	1.34	0.12
	B2	0.93	0.10

Enter tree cover % = 50.0

Estimated Biomass, C, and N

Class	Pre-Burn			Post-Burn			Change		
	Biomass kg/ha	C kg/ha	N kg/ha	Biomass kg/ha	C kg/ha	N kg/ha	Biomass kg/ha	C kg/ha	N kg/ha
Soil	N/A	46,695	3,621	N/A	50,522	3,791	N/A	3,828	170
Roots	13,482	4,357	108	8,912	2,816	100	-4,570	-1,541	-7
Litter	9,464	4,312	107	590	298	1	-8,874	-4,014	-106
Herbaceous	220	97	97	22	10	10	-198	-87	-87
Shrub	1,623	801	9	196	95	2	-1,427	-706	-7
Tree	54,044	28,813	189	24,530	12,143	142	-29,514	-16,670	-47
Aboveground Biomass	65,351	34,023	402	25,338	12,546	156	-40,013	-21,477	-246
Total Biomass	78,834	38,380	510	34,250	15,362	256	-44,583	-23,018	-254
Total		85,074	4,131		65,884	4,047		-19,190	-84

Figure 2.0

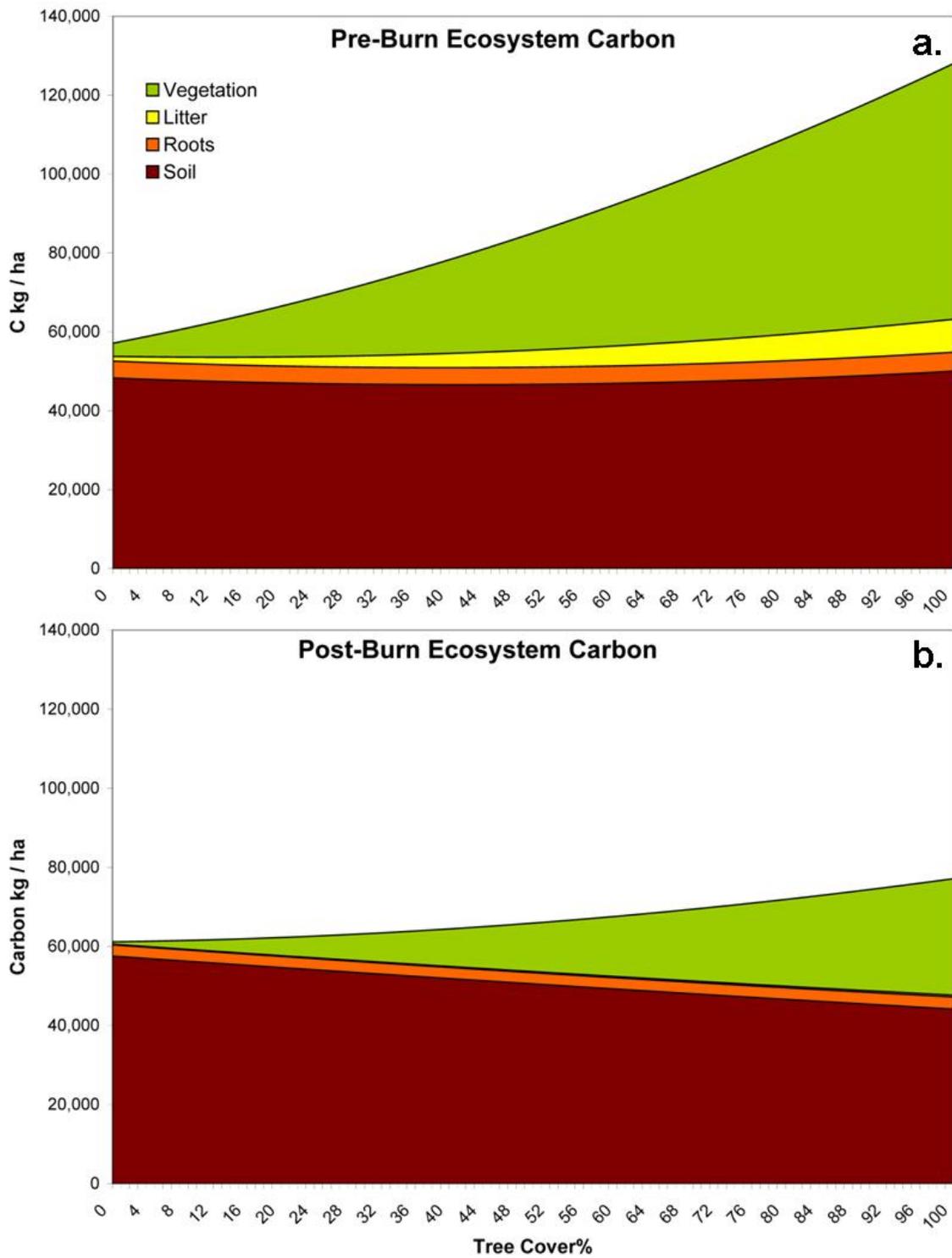


Figure 3.0

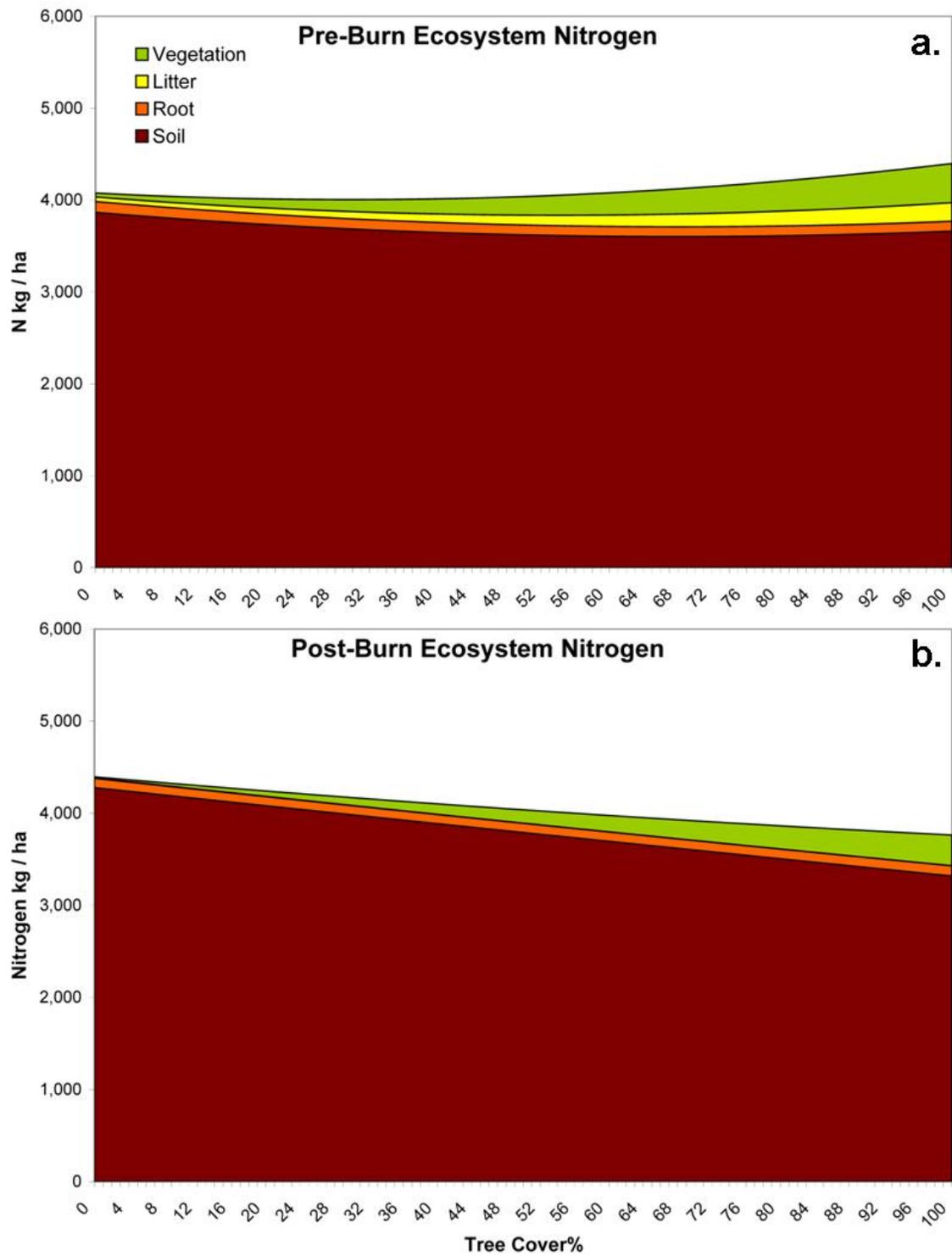


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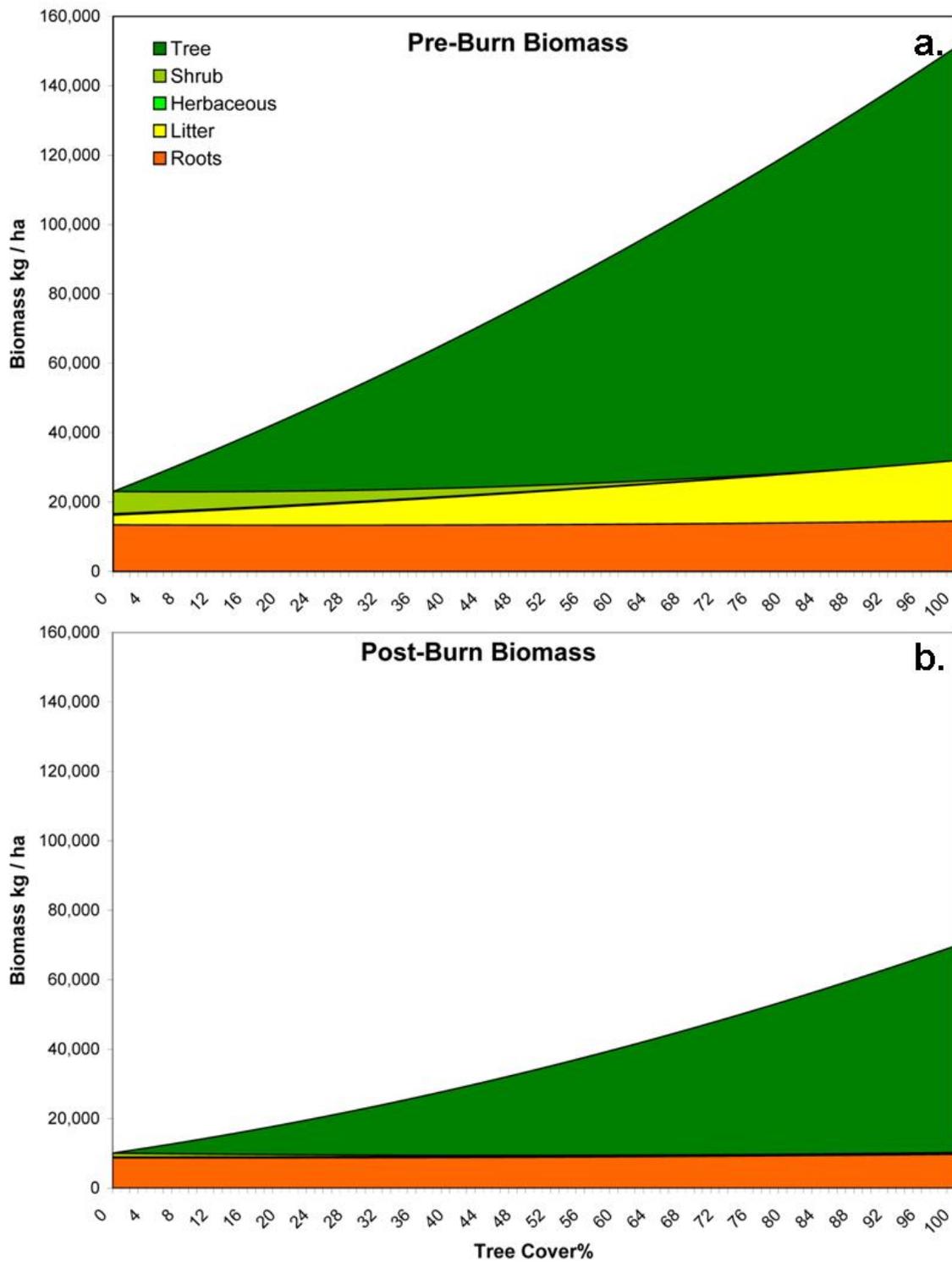


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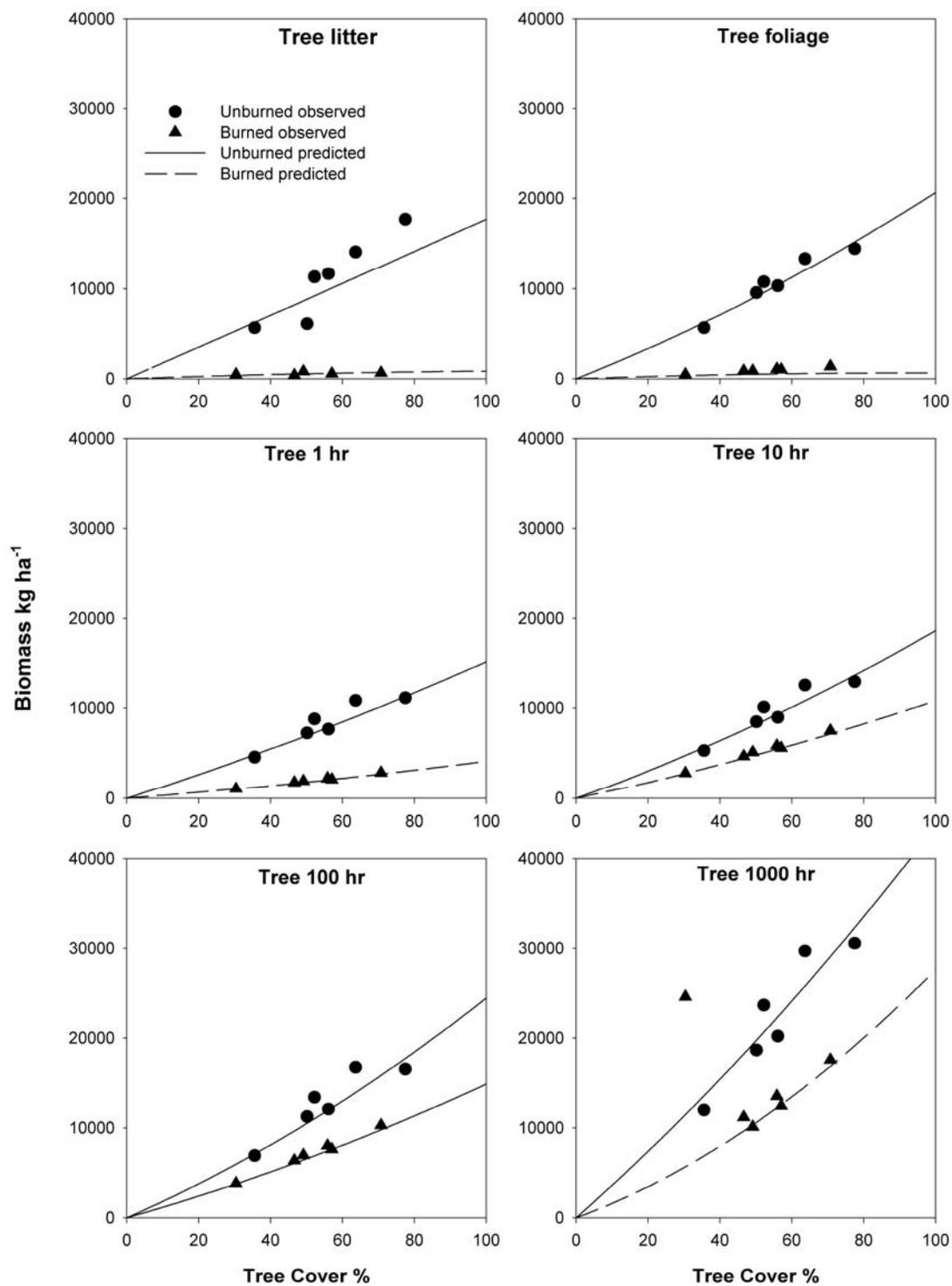


Figure 6.0

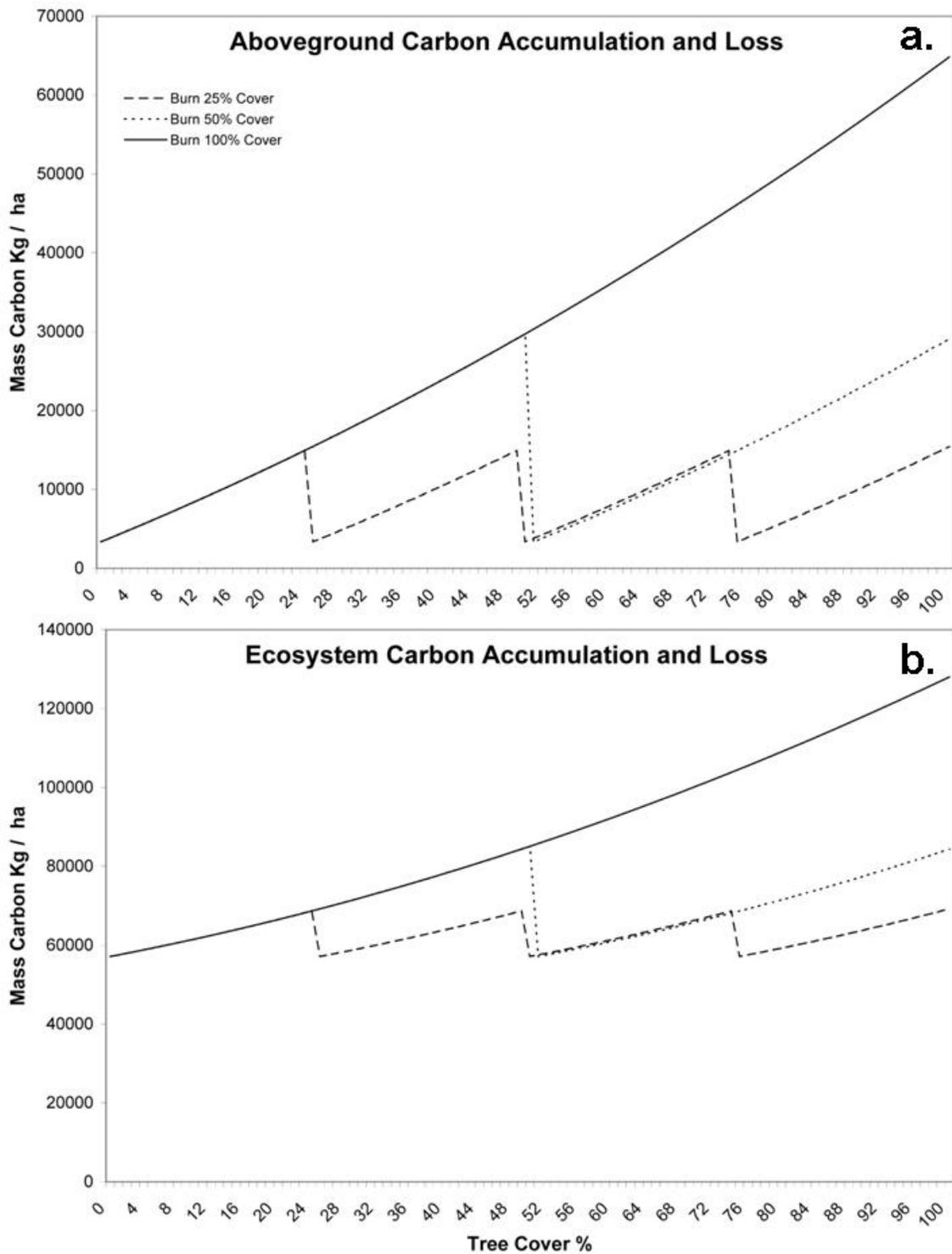
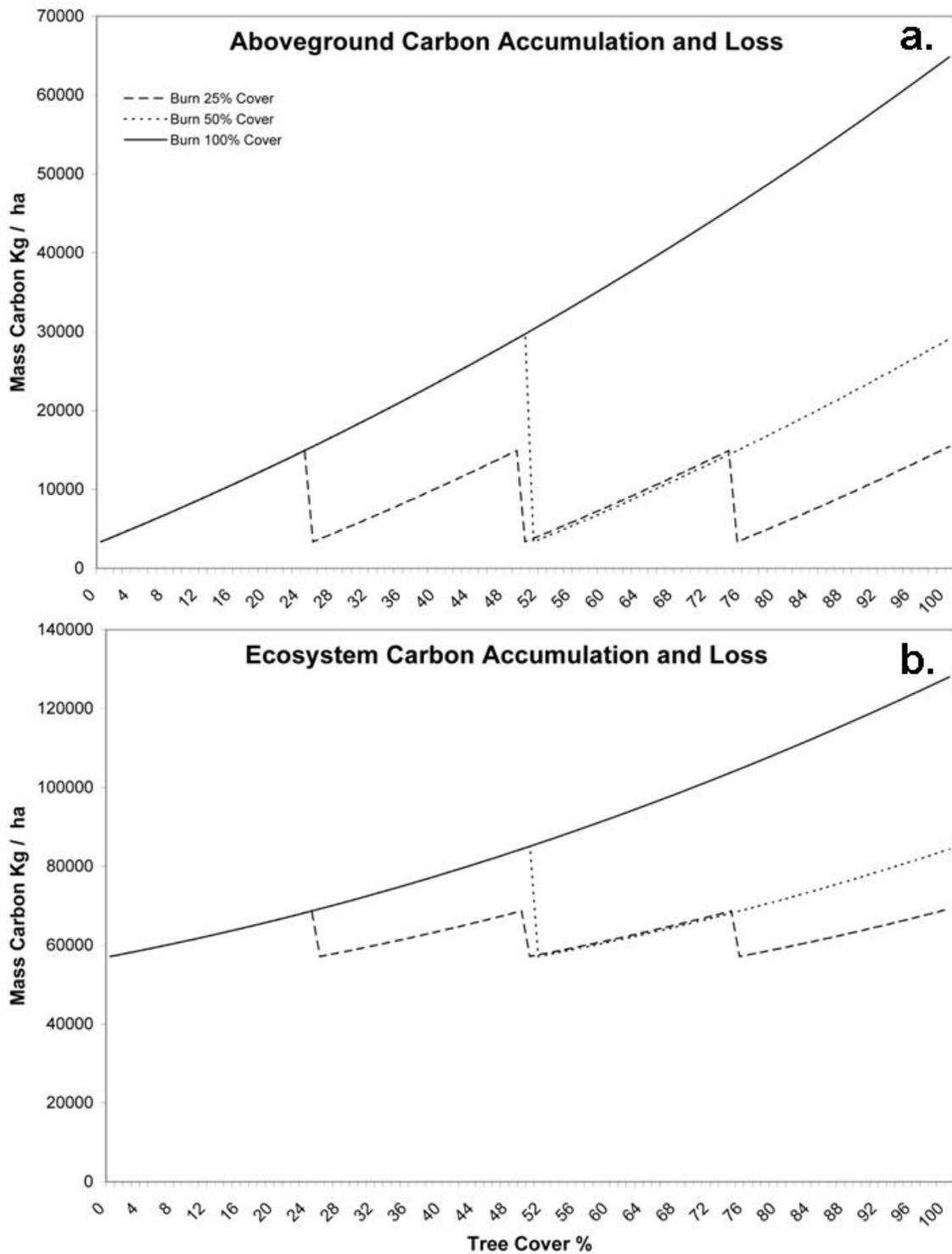


Figure 7.0



Summary

In the preceding chapters I have taken steps towards quantifying total ecosystem C and N in pinyon-juniper woodlands. We have outlined methods for estimating root biomass and quantified the influence of woodland expansion and prescribed fire on ecosystem C and N. We have also initiated an effort to develop a simple spreadsheet calculator which will assist land managers in their effort to assess project impacts on carbon retention.

The soil core we have proposed for measuring root biomass as an alternative to quantitative pits was originally described by Ponder and Alley (1997), and suggested to us by Donn Todd of the Oakridge National Labs. We found the core device gives estimates of root biomass consistent to estimates made from quantitative pits, but is much less labor and calculation intensive.

Soil pools typically dominate ecosystem C and N budgets in sagebrush ecosystems, and root biomass may be as much as 50 % of total ecosystem biomass (Hooker et al. 2008, Jackson et al. 1996). However, as trees encroach onto the landscape biomass C and N rapidly accumulate in aboveground pools (Norris et al. 2001). By the time woodlands reach an ecosystem recovery threshold near 50 % tree cover, aboveground biomass may account for as much as 40 % of ecosystem C, but still only 10 % of ecosystem N.

Prescribed burning below ecosystem recovery thresholds may restore native perennial herbaceous species, and inhibit exotic grass invasion. Prescribed fire will release portions of aboveground C and N to the atmosphere, but some C and N should be incorporated into soil pools. We have demonstrated that burning increases C and N near the soil surface immediately following fire, but that over longer periods it is difficult to detect these changes.

Our empirically derived spreadsheet model predicts that once tree cover exceeds 70 % aboveground C surpasses soil pools and that fire occurring at this stage could release 40 % of ecosystem C, but still less than 10 % of ecosystem N to the atmosphere.

Additionally burning in closed canopy woodlands has the potential to release C and N from soil pools, further aggravating ecosystem losses.

In our attempt to quantify ecosystem C and N we have encountered several setbacks which will need to be resolved. Of critical importance are our estimates of soil C and N increases following fire. We observed that our estimates of soil C and N gain after fire exceed the possible inputs from aboveground biomass. We ultimately believe that this is due to sampling and transformation errors. Because our sample size is relatively low we may not have accurately quantified the average soil C and N concentrations before and after burning. Additionally, because these concentrations are then transformed to kg ha^{-1} using estimates of coarse fragment, bulk density, and vegetation cover we propose we have over estimated high concentrations following fire across the landscape. Inevitably the high concentrations measured near the center of a vegetation microsite decreases by some factor as distance from the center of the microsite increases. This could possibly explain some of the discrepancy in our estimates. Sampling protocols will certainly need to be addressed that will take into account the extreme heterogeneity of these sagebrush and woodland systems.

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