ESTABLISHING REFERENCE SITES AND EVALUATING THINNING AND BURNING AS RESTORATION TOOLS FOR THE NORTHERN IDAHO GROUND SQUIRREL (*UROCITELLUS BRUNNEUS*)

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Elise Suronen

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Major Professor: Beth A. Newingham Ph. D.

AUTHORIZATION TO SUBMIT THESIS

This thesis of Elise Suronen, submitted for the degree of Master of Science with a major in Natural Resources and titled "Establishing reference sites and evaluating thinning and burning as restoration tools for the northern Idaho ground squirrel (*Urocitellus brunneus*)," has been reviewed in final form. Permission, as indicated by the signatures and dates given below, is now granted to submit final copies to the College of Graduate Studies for approval.

Major Professor __ Beth A Newingham __________Date__08/02/2012___
Beth A. Newingham

Committee
Members ___Penny Morgan___ Date___07/28/2012___
Penelope Morgan

___Janet Rachlow___Date___07/20/2012___
Janet L. Rachlow

___Eric Yensen___Date___07/31/2012___
Eric Yensen

Department
Administrator ___________________________________Date______________
Jo Ellen Force

Discipline's
College Dean ___________________________________Date_______________
Kurt Pregitzer

Final Approval and Acceptance by the College of Graduate Studies
___________________________________Date_______________

Jie Chen

ABSTRACT

The northern Idaho ground squirrel (NIDGS, *Urocitellus brunneus*) was listed as 'threatened' under the Endangered Species Act in 2000. The federal decision was based on the limited distribution of NIDGS, its endemic status, small population sizes, and evidence of habitat loss from an altered fire regime. Land use within ponderosa pine (*Pinus ponderosa*) forests has led to a decrease in fire frequency and extent, and subsequently tree density and extent has increased. Historically open meadows occupied by NIDGS have become isolated and reduced in size. Managers apply thinning and burning treatments to restore the entire forest as well as specifically restore NIDGS habitat. Restoration efforts will benefit from studies that quantify reference habitat, areas occupied by NIDGS, and that evaluate the treatments themselves. Reference habitat information is necessary to adapt and improve present management plans. I measured habitat and microhabitat attributes that were potentially altered by thinning and burning to characterize NIDGS and provide reference data. I also measured habitat attributes before and after thinning and prescribed burning treatments at three restoration units, which I compared to three reference units to evaluate whether the treatments changed habitat attributes toward reference conditions. Overall, my study provides baseline information to characterize NIDGS habitat and evaluates restoration practices to help managers refine restoration goals.

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ESTABLISHING REFERENCE SITES FOR HABITAT RESTORATION OF A THREATENED ENDEMIC GROUND SQUIRREL, *UROCITELLUS BRUNNEUS*

Abstract

The northern Idaho ground squirrel (NIDGS, *Urocitellus brunneus*) is a threatened species endemic to west-central Idaho. Reductions in habitat are due to fire exclusion and subsequent increases in tree density, in particular tree encroachment into historically open meadows and scablands within ponderosa pine (*Pinus ponderosa*) forest. Management practices for NIDGS recovery include thinning and burning ponderosa pine forest to reduce tree encroachment. Reference site information is useful to create restoration goals, which will help refine present management plans. At seven reference populations, I measured habitat attributes potentially altered by thinning and burning treatments: tree canopy cover, ground cover, litter depth, and soil chemistry. Habitat attributes were described with Bayesian credibility intervals. Reference sites for NIDGS habitat had tree canopy cover of 10-15%, low tree densities of 64-118 trees/ha, and visual obstruction was 20-25 cm. Reference site understory vegetation consisted of 20-30 species of grasses, forbs and shrubs, while across site richness was 153. Fifteen understory species were selected with the highest constancy and cover across the seven sites. Litter depth was 0.7-0.9 cm deep, and the clay loam soils were slightly acidic. My study provides more precise information about NIDGS habitat. Based on my findings, I make recommendations to develop habitat restoration goals: 1) reduction of tree canopy cover by limiting conifer density to 64-118 trees/ha or below mainly consisting of large trees (DBH >20 cm) that may be maintained by thinning and prescribed burns; 2) maintenance of understory visual obstruction height under 25 cm by repeated burning; 3) areas selected for thinning and burning are recommended to

contain a subset of the 13 species commonly associated with NIDGS habitat; 4) repeated fires may be used to keep litter depth below 1 cm; and 5) using our baseline soil data managers can monitor changes in soil chemistry to link to plant responses. Managers may use these results to develop appropriate restoration goals when implementing habitat restoration and increase the efficacy of currently implemented restoration techniques.

Introduction

Anthropogenic activities contribute to habitat loss and fragmentation, which are the leading causes of species decline and extinction (Hoekstra et al. 2005; Anderson and Gutzwiller 2005; Morrison et al. 2006; Fischer and Lindenmayer 2007). Human land use is rapidly changing ecosystems (Foley et al. 2005); therefore, ecological restoration strives to reestablish the structure and function of damaged ecosystems in order to restore habitat and aid species recovery (Society of Ecological Restoration 2004; Palmer 2006). Reference site conditions are often used to justify restoration goals, guide restoration efforts, and evaluate restoration progress (Hobbs and Norton 1996; Block et al. 2001; Bond and Lake 2003; Ruiz-Jaen and Aide 2005; White and Walker 1997). Desired states can reflect pristine conditions that incorporate the historical range of natural variability or a rehabilitated state where function is restored although structure may be different (Higgs 1997; Landres et al. 1999; Whisenant 1999). However, reference sites are difficult to select since ecosystems are naturally variable (Clewell and Rieger 1992; White and Walker 1997).

Identifying reference sites is challenging. Ecosystems have various stable states, historical records might not be available, and climate change is continually shifting abiotic conditions making it difficult to choose an appropriate reference (White and Walker 1997; Harris et al. 2006; Hobbs 2007). Reference site selection becomes increasingly difficult when the restoration goals include conserving a rare or threatened species. Selecting reference sites based on rare species presence or absence can be misleading because existing populations could be occupying sink habitat, and there are typically too few individuals to occupy all available patches (Hall et al. 1997). Assessing habitat quality is time consuming and expensive for it involves studying population dynamics and fecundity (Van Horne 1983; Block et al. 2001; Johnson 2007). Due to economic and social constraints, managers often have to act before they can determine habitat quality.

In 2000 the United States Fish and Wildlife Service listed the northern Idaho ground squirrel (NIDGS, *Urocitellus brunneus;* Yensen 1991; Helgen et al. 2009; Hoisington 2012) as 'threatened' under the Endangered Species Act (Clarke 2000). The NIDGS is endemic to west-central Idaho (Yensen 1991). Listing was approved because of endemic distribution, low population numbers, and the lack of connectivity between NIDGS populations (Gavin et al. 1999; Sherman and Runge 2002; Clarke 1998, 2000). In addition, the NIDGS was considered threatened due to habitat loss, which has been attributed to both interactions with humans and altered environmental conditions (Clarke 1998, 2000; Stoddard 2001). Central Idaho has experienced >100 years of fire exclusion, cattle grazing pressure, and logging practices (Crane and Fischer 1986; Steele et al. 1986), which has altered forest structure.

There is strong circumstantial evidence that the abundance of quality NIDGS habitat (open meadows connected by corridors) has decreased due to an increase in tree density and extent (Truska and Yensen 1990; Yensen and Sherman 1997; Gavin et al. 1999). Changes in forest structure are in part due to human activities altering the historical fire regime of central Idaho: low severity fire occurred every 5-20 years until about 1900 when fires became much less frequent (Crane and Fischer 1986; Steele et al. 1986; Heyerdahl et al. 2008*b*). In order to mimic natural fire regimes, managers now use thinning and prescribed burning to restore NIDGS habitat (USFWS 2003). The Payette National Forest continues to modify prescription treatments in order to meet the objective of enhancing NIDGS habitat. However, little quantitative information exists on the habitat requirements of NIDGS to further define objectives for habitat management (Yensen 2009). Such information will benefit habitat restoration from quantitative baseline information on reference sites.

Since thinning and burning modify habitat and microhabitat features for NIDGS populations, information about habitat features at currently occupied sites is needed to further refine habitat restoration goals. The relationship between habitat attributes and animals can be inferred by habitat use (Gaillard et al. 2010). Therefore, I described habitat and microhabitat attributes to provide baseline data for managers since no published study reports specific details on northern Idaho ground squirrel habitat. Habitat attributes that are potentially altered by thinning and prescribed burns were measured from seven reference sites, while individual microhabitats were determined by observations of habitat use by individual NIDGS at two of the reference sites. Habitat attributes included vegetation and soil characteristics that could impact the NIDGS.

Methods

Species and Site Descriptions

NIDGS are small-bodied rodents with an average body length of 200-260 mm and a hind-foot length of 32-39 mm (Yensen and Sherman 1997). NIDGS hibernate for eight months from late summer to spring (Yensen and Sherman 1997). During the four-month active period in spring and summer, NIDGS adults mate and give birth, and then adults and newly born pups must eat enough to acquire the necessary reserves for subsequent hibernation (Yensen and Sherman 1997).

Study sites are located between the Cuddy and Seven Devils Mountains in Idaho (Yensen 1991). NIDGS typically inhabit meadows and open, park-like, ponderosa pine stands with <30 percent canopy cover, aspects from 90 to 290 degrees, slopes <12 degrees, and elevations of 1,050–2,300 m (Yensen 1991; Nutt 2008; Evans Mack and Bond 2010). The area receives approximately 600 mm of precipitation annually with an average high and low temperature of 13.7°C and -3.7°C, respectively (Western Regional Climate Center 2010).

I selected seven areas occupied by NIDGS to characterize habitat and chose two of the seven sites to describe microhabitat (Figure 1). The seven reference sites included: Cap Gun (45.0150°, -116.7053°), Lost Valley (44.9670°, -116.4515°), Mud Creek (45.0013°, -116.3434°), OX Ranch (45.0159°, -116.6509°), Price Valley (45.0336°, -116.4290°), Slaughter Gulch (44.9810°, -116.4679°), and Summit Gulch (44.9921°, -116.7172°). Elevation of study sites ranged from 1,200-1,500 m. Cap Gun and Summit Gulch were selected for microhabitat attributes because they are open areas surrounded by ponderosa pine and have robust populations of NIDGS. Cap Gun is a historical NIDGS restoration site, while Cap Gun is a relatively new site of interest for restoration work. All seven sites were grazed; four sites were US Forest Service grazing allotments (Cap Gun, Lost Valley, Price Valley and Summit Gulch) and three were privately owned and grazed (Mud Creek, OX Ranch & Slaughter Gulch). The recent fire history was unknown for the private land, but there have been several prescribed fires implemented on public land. In the past 30 years, prescribed fires occurred three times at Cap Gun and Price Valley, five times at Lost Valley, and twice at Summit Gulch.

Habitat Attributes

I randomly distributed six 50-m transects within each reference site to quantify habitat attributes. NIDGS forage up to 100 m away from burrows (Dyni and Yensen 1996); hence, transects were located within 100 m of NIDGS sightings from annual Idaho Fish and Game monitoring surveys. I collected habitat attribute data during the peak of the 2010 and 2011 growing seasons. Due to abnormally wet springs and late snowmelt, the height of the growing season was delayed until mid-July during both years.

Habitat attributes included tree canopy cover and density. I recorded tree characteristics in two 200-m^2 circular plots with an 8-m radius centered at the 0 and 50-m transect marks (12 plots per site). Within each circular plot, I measured 1) the percentage of closed canopy cover using a concave spherical crown densiometer one meter above ground in the four cardinal directions (Lemmon 1956); 2) tree density by counting the number of trees within the circular plot; and 3) tree size using four diameter at breast height (DBH) size classes (DBH <8 cm, 8-12 cm, 13-20 cm, >20 cm). Diameter of all trees within the DBH size class of >20 cm were recorded to the nearest 0.1 cm.

Understory vegetation was characterized by measuring visual obstruction and percent cover. Along each transect, I determined visual obstruction height, the height of the densest part of the vegetation, with a modified Robel pole (Robel et al. 1970; Herrick et al. 2005). I took four measurements in the cardinal directions at three points (0, 25, and 50-m marks) along each transect. Standing 5 m away from the modified Robel pole, I peered through a 1-m tall sighting pole to determine the height of visual obstruction. The modified Robel pole had 2-cm intervals from 0-40 cm; measurements above 40 cm were recorded as >40, >50 or >100 cm. I recorded percent canopy cover with ocular readings of percent cover. Ground cover categories included:

soil (bare ground and rocks < baseball size), litter, woody debris (branches and pieces of wood), rocks (> baseball size), and moss or lichen. I assigned a Daubenmire (1959) cover class (0-5%, 6-25% 26-50%, 51-75%, 76-95%, 96-100%) to each species and ground cover category within ten quadrats (0.5 x 0.5 m) along each transect (60 quadrats per site). I categorized each understory species into a plant functional group: annual forb, perennial forb, annual/perennial forb (species that can take either growth form), annual grass, perennial grass, sedge, rush, and shrub. I created a reference collection of understory species to document species detected at the sites. Taxonomy follows the USDA Plant Database (2012).

Litter and soil parameters were also measured at each reference site. I recorded three depths of litter (top of litter debris to soil surface) every 5 m along each transect (30 depths per transect). I extracted three 10 to 15-cm deep soil cores every 5 m along each transect (30 cores per transect). The 30 cores were homogenized into one composite soil sample per transect (six total composites per reference site). Soil cores were analyzed at North Dakota State University for soil texture, pH, phosphorous and calcium for 2010 and 2011. In 2011 at Cap Gun, Price Valley, and Summit Gulch, I weighed out 10 g of each soil composite and combined with 50 ml of 2M KCl immediately after collection to terminate nutrient cycling within the composite soil samples. Twenty-four hours later, the KCl extract was analyzed for soil nitrate and ammonia at the University of Idaho (Bundy and Meisinger 1994).

Microhabitat Attributes

I documented locations of NIDGS in 20 observation plots at Cap Gun and Summit Gulch (10 plots per site). Understory vegetation obstructed observer visibility; therefore, I limited the plots to 20 x 20 m and restricted the locations of NIDGS activity to sessile behaviors including lookout and food consumption spots. Areas of NIDGS activity were recorded and visited after the observation period to make habitat measurements. Observation periods lasted for 90-minutes in the morning, afternoon, and evening within a one-week period in June, July and August. NIDGS were not active during late-summer afternoons, thus no afternoon observations were conducted in August. Visual obstruction and canopy cover were measured at each activity location. I used a modified Robel pole to quantify microhabitat visual obstruction (see above) and a concave spherical crown densiometer positioned 10 cm above the ground to measure canopy cover. Densiometer readings included mostly understory cover but occasionally incorporated the canopy of a tree.

Statistical Analysis

To describe habitat attributes, I used Bayesian means and 95% credible intervals based on the highest posterior density (95% HPDI). The HPDI is the smallest interval that includes 95% of the data (Carlin and Louis 2000). Statistics were based on a uniform prior in SAS/STAT® version 9.2 software (SAS Institute Inc. 2008).

A group of common plant species across reference sites was created by selecting those species with the highest average canopy cover and constancy values. Understory plant species with the highest percent cover had >1% cover. Reference sites were highly diverse. I defined highest constancy as species that occurred in at least 10 out of 60 quadrats at each site in 2010 and 2011, which had to be true for all or at least the majority of the sites. Before finalizing the common species group, I compared species with the highest cover and constancy to species found in the NIDGS diet. The diet is comprised of 40-50 different plant species and few comprise more than 1% of the diet (Dyni and Yensen 1996; Yensen et al. 2010). Plants identified in the dominant group included plants that were present at most sites with the highest levels of cover and constancy (8 out of 15 species) and were important to the NIDGS diet.

I used PC-ORD version 6 software to calculate understory plant species richness, Shannon's Evenness Index and Simpson's Diversity Index (McCune and Mefford 2011). The calculations included all recorded species for each site and across all reference sites. The Simpson's Diversity Index equation was: $1 - sum (P_i*P_i)$, where "P_i" equals the importance probability in element "i", a plant species, which is relativized by total species cover within a transect (Simpson 1949).

Results

Habitat Attributes

Reference sites had low tree canopy cover and low tree densities. Ponderosa pine was the dominant tree species; 155 of the 162 mature trees (DBH >20 cm) were ponderosa pine (96%). Other tree species included Douglas-fir (*Pseudotsuga menziesii*) and lodgepole pine (*Pinus contorta*). Tree canopy cover averaged ≤20% for each site (Table 1, Figure 2). OX Ranch and Summit Gulch were open meadows with low tree canopy cover; Cap Gun and Lost Valley were open forest stands with the highest tree canopy cover. Trees within NIDGS habitat had an uneven distribution of size classes: smaller trees (<8 cm DBH) were most common, followed by large trees with a DBH >20 cm (Figure 3). Few trees were in the 8-12 and 13-20 cm DBH classes. OX Ranch and Summit Gulch had the lowest forest densities of the reference sites. Tree size varied at both sites; the majority of trees at the OX Ranch had a DBH <8 cm, while Summit Gulch had no trees with a DBH <8 cm. Cap Gun and Lost Valley had the densest forests and the highest number of trees with a DBH >20 cm. The largest tree DBH recorded was 68.9 cm at Lost Valley and second largest was 66.0 cm at Mud Creek, while the largest DBH recordings for the rest of the sites were 40.6-58.4 cm.

Understory vegetation structure had low visual obstruction and was a patchy mosaic of plants intermixed with bare ground. Visual obstruction was 20-24 cm (Table 2, Figure 2). Mud Creek and Slaughter Gulch had the lowest visual obstruction, while the rest of the sites had higher visual obstruction, especially Cap Gun and OX Ranch with visual obstruction over 30 cm. Ground cover within NIDGS habitat was largely bare soil or litter; combined bare soil and litter make up almost 34% of the ground cover (Table 3). Other surface elements, such as rocks and woody materials, made up approximately 8% of the ground cover. Moss and lichen cover was minimal. Overall, vegetation cover was patchy in reference habitat. Perennial forbs and grasses dominated understory species cover, while shrub cover was slightly lower (Table 3). Sedge cover was comparable to annual forb and annual grass cover. Rushes and some annual/perennial forbs had minimal cover of less than 1%.

Plant species diversity and evenness were high across all reference sites (Table 4). I identified 153 total species from the seven sites across two years (Appendix A), while species richness on individual sites varied from 21-34 species. OX Ranch, Summit Gulch, Cap Gun and Lost Valley had the highest species richness (≥30 species). Mud Creek and Price Valley had the lowest richness, evenness, and diversity. Although the sites were highly diverse between and within sites, I selected 15 out of 153 species with the highest understory cover and constancy across the seven sites (Table 5).

Several non-native plants were found at the reference sites (APPENDIX A). Three noxious species listed in Idaho included *Bromus tectorum*, *Linaria vulgaris*, and *Lepidium latifolium* (USDA Plants 2012). Other invasive or weedy plants included *Bromus inermis*, *Bromus arvensis*, *Poa bulbosa, Potentilla recta*, *Rumex acetosella*, *Leucanthemum vulgare*, *Cynoglossum officinale, Ventenata dubia*, *Tragopogon dubius*, and *Dianthus armeria*. Overall, NIDGS habitat was not heavily invaded by one particular species; average cover of non-native species was 1.4 ±0.2 % and eight out of thirteen non-natives were only located at one or two sites.

Reference sites had minimal litter build up on top of clay loam soils with a slightly acidic pH, while soil calcium, phosphorus, nitrate and ammonium concentrations were variable across sites. Litter depths within NIDGS habitat averaged 0.8-0.9 cm (Table 6, Figure 2). The highest average litter depths were at Cap Gun and OX Ranch, while Mud Creek, Price Valley and Slaughter Gulch had the lowest average litter depth. Soil texture at five out of seven sites was a clay loam, while OX Ranch had a silty clay loam and Slaughter Gulch had loamy soil (Table 7). There was slight variation in soil pH (Figure 4): Mud Creek had the most acidic soils (pH = 5.85), while Price Valley had the most alkaline soils (pH = 6.28) (Table 8). OX Ranch had the highest soil calcium levels compared to the rest of the sites. Mud Creek had higher soil phosphorus than the other seven reference sites. Soil nitrate was more abundant than ammonium (Figure 5). Nitrate and ammonium levels at Cap Gun were lower than at Price Valley and Summit Gulch (Table 8).

Microhabitat Attributes

Visual obstruction of used microhabitat was 8-10 cm tall. Microhabitat canopy cover largely represented understory plant canopy and sometimes tree canopy; hence, values ranged from 3-73%. The average canopy cover at used microhabitats was 27%.

Discussion

Detailed information on habitat attributes can assist in developing appropriate habitat restoration goals and practices. However, many habitat restoration projects lack detailed habitat descriptions. Thus, by increasing our knowledge of existing habitat features and requirements for particular species, we can improve present restoration practices and goals. Here, I quantified habitat and microhabitat attributes, which are important to consider in NIDGS habitat restoration. Using these habitat characteristics as reference information, I offer suggestions to refine restoration goals and practices to assist NIDGS recovery.

My findings are consistent with anecdotal descriptions that NIDGS prefer areas with minimal tree density and canopy cover. An open forest structure may increase understory biomass production due to higher solar infiltration, lower levels of litter deposition, and decreased soil nutrient competition between herbaceous species and trees (Pase 1958; Moore and Deiter 1992; Moir 1996; Sabo et al. 2009). Therefore, NIDGS might favor open areas due to increased foraging opportunities. Food consumption is particularly important because small-bodied mammals have high nutritional demands (Demment and Van Soest 1985), including eating enough to accumulate the necessary fat reserves to survive eight months of hibernation. Open areas are also likely to be favored by NIDGS because they can more easily detect predators; tree clusters also provide perching and nesting habitat for birds of prey (Sherman 1989; Yensen 1991). The potential combination of increased foraging and decreased mortality reasonably explains NIDGS use of areas with low canopy cover and tree density.

While little information exists on NIDGS habitat before human settlement, current tree density in NIDGS habitat is likely higher than historical conditions (Gavin et al. 1999), which may be due to high tree recruitment or a lack of disturbance or both. Historically, wildfires were frequent in ponderosa pine forests (Arno 1980; Steele et al. 1986; Barrett 1988; Heyerdahl et al. 2008*a*, *b*). Frequent fires resulted in relatively open stands with large trees and low densities of small trees, but with fewer fires on these sites in the 20th century, stands are likely denser than they were before in 1900 (Agee 1998; Gavin et al. 1999; Moore et al. 1999; Heyerdahl et al.

2008*a*, *b*). Thinning and prescribed burning are commonly used to reduce tree density and accumulated fuels and to restore fire regimes in ponderosa pine forests (Fulé et al. 2012).

Understory characteristics of reference habitat included visual obstruction heights of 20-25 cm created by a diverse understory dominated by forbs and grasses. NIDGS might select for this level of visual obstruction because they can probably see over the vegetation when standing on two legs their eye level is about 15-22 cm off the ground (E. Yensen and D. Evans-Mack *per*. *comm*.2012). This height of vegetation could also help conceal them if they stand on all four legs; eye level is 4.5 cm off the ground (D. Evans-Mack *per*. *comm*.2012). Reference sites were 35-55% bare ground, litter, moss, lichen or rocks. The low vegetation height and relatively sparse cover is characteristic of south-facing slopes in low-elevation mountains of the northern Rockies (Crist et al. 2009). Understory vegetation was highly diverse, many species with low proportional abundance, within and across sites. Although 15 species were identified to have the highest constancy and cover, I suggest that 13 of those species have a strong association with NIDGS habitat: *Achillea millefolium, Allium sp., Anaphalis margaritacea, Epilobium brachycarpum, Eriogonum sp., Fragaria virginiana, Lupinus sp., Polygonum douglassi, Potentilla gracilis, Sedum stenopetalum, Poa pratensis, Poa secunda,* and *Pseudoroegneria spicata*. This group of species was identified to help managers determine NIDGS habitat visually by searching for sites with these species. *Bromus tectorum* and *Poa bulbosa* were in the original group of 15 species with the highest constancy and cover, but they are non-native, invasive species. I have excluded from further consideration because they do not represent historical conditions. The remaining plants are either important to diet or reasons that are to be determined; for example, they could be potentially indicative of favorable abiotic conditions like soil characteristics.

Across reference sites, litter covered 12-17% of the ground and was less than 1 cm deep. Litter can negatively impact the growth and recruitment of understory flora (Pase 1958; Hart et al. 1992, Biswell et al. 1966). Seeds and developing seedlings are impacted by the presence and depth of litter because litter influences abiotic soil conditions, including moisture, temperature, water infiltration, pH, and chemical inputs (Neary et al. 1999, 2005). Generally, reference site soils were clay loam in texture and slightly acidic pH with varying concentrations of phosphorus, calcium and inorganic nitrogen. The only soil parameter I measured that directly impacts ground burrowing animals is soil texture (Laundré and Reynolds 1993); soil texture and the depth of the parent material influence burrow construction (Yensen et al. 1991). Since soil texture was similar across sites, the depth of the parent material and rockiness of soil probably play a larger role in burrow location and type. In addition to soil texture, NIDGS pelage is reddish-brown camouflaging them against the reddish soil (Yensen 1991). This pelage and soil association is one of several characteristics that distinguish NIDGS from its close relative the southern Idaho ground squirrel (Yensen 1991).

Soil chemistry could also indirectly affect NIDGS through effects on plant diversity and composition. Since pH levels were around 6.1, soil nutrient availability within NIDGS habitat was probably not impacted by the pH. Nutrients are most accessible to microbes and plants in slightly acidic soils with a pH around 6.5 (Neary et al. 2005). Soil calcium values ranged from 50,354-57,098 mg/kg in dry soil, while soil phosphorus ranged from 349-437 mg/kg soil. Although calcium is not a limiting nutrient (Barber 1995), calcium weather out of basalt parent material, hence I wanted to measure baseline calcium levels. Phosphorus is a limiting nutrient in all ecosystems (Barber 1995); hence, it was a nutrient of interest along with inorganic nitrogen. Soil nitrate and ammonium concentrations were higher than concentrations reported in neighboring forests (Koyama et al. 2010, 2012). While it has been suggested that soil nutrients are limited in ponderosa pine forest (Covington and Sackett 1986), in a nutrient addition experiment, Jacobsen et al. (1980) found no growth limitations in ponderosa pine growth after adding nitrogen and phosphorus.

NIDGS used microhabitats with understory vegetation heights of 8-9 cm and understory canopy cover of 27%. When NIDGS stand on their hind legs their eye level is about 15-22 cm off the ground (E. Yensen and D. Evans-Mack *per. comm.* 2012). Visual obstruction at the habitat level was about 10 cm higher than values measured at microhabitats selected by individuals; hence, a visual obstruction difference of 10 cm may impact their behavior. Values for microhabitat canopy cover further support that NIDGS use open rather than closed areas.

While density can be a misleading measure of habitat quality, when it comes to rare species, density could be a decent indicator of quality habitat (Van Horne 1983). It is important to note that with rare species quality habitat can be left unoccupied due to limited numbers. Coupled with habitat fragmentation and loss of quality habitat, individuals could be selecting poor-quality habitat. These reference sites could potentially be sink populations, areas of low quality habitat (Pulliam 1988). In addition to population dispersal limitations, habitat fragmentation and loss of quality habitat, NIDGS directly compete and are excluded from habitat by Columbian ground squirrels (*Urocitellus columbianus*) (unpublished data Yensen). Realized habitat quality is the quality experienced when competitors are present (Johnson 2007). The combination of habitat loss, fragmentation and competition decreased my ability to capture the fundamental habitat quality, thus these sites might not represent the ideal reference for NIDGS habitat restoration. I can only infer that these attributes are important to NIDGS survival and reproduction (Lindell 2008).

Further studies are needed to determine NIDGS habitat quality and preference because targeting habitat variables that directly impact species fitness will assist restoration efforts (George and Zack 2001). It would be useful to extend this analysis to a wider number of sites to better encompass the range of variation in habitats occupied by NIDGS. However, the reference sites I measured shared several characteristics, which can serve as preliminary targets for habitat restoration. The following habitat management actions may help achieve conditions that are similar to the currently occupied sites we measured.

1. Thinning and prescription burns can be used to maintain an open forest structure to reduce conifer density below 120 trees/ha, eliminate large trees with a DBH greater than 20cm but also preserving various tree sizes. Reducing tree density and size may also help achieve tree canopy cover levels around 10-15%.
2. Thinning and prescription burns can help achieve understory vegetation height below 25 cm, by either cutting or burning shrubs and small trees. Since thinning and burning practices aim to reduce accumulated fuel, it is likely that restoration treatments already contribute to understory visual obstruction reduction.

3. Future restoration projects should target areas that potentially or already support a subset of the 13 common native plant species listed above. Due to high levels of species diversity, is it unlikely to find a site with all 13 species, so a subset of the group is suggested. Managers presently target forested areas with sagebrush (*Artemisia tridentata*) and antelope bitterbrush (*Purshia tridentata*), since the shrubs likely indicate once open habitat (Rautsaw *per.comm.* 2010). Managers may wish to also include this group of common species associated with NIDGS habitat.

4. Litter cover should remain less than 20% and be less than 1 cm deep. This goal can be achieved using repeated prescription burns, yet seasonality of the burning will affect the impact of heat on soil chemistry and fauna (Busse and DeBano 2008). Repeated burns could successfully lower litter to desirable depths (Scott 2002).

5. Since soil nutrients highly affect plant growth and reproduction, I suggest these parameters are incorporated into future monitoring. In addition, exploring the effects of fire on microbial processes, mycorrhizae, and soil nutrient fluctuations over time would provide insight on plant responses to these restoration treatments.

Conclusion

Habitat is essential for the maintenance of wildlife populations; hence, describing habitat is fundamental for conservation of threatened and endangered species and their habitat (Young 2000; Anderson and Gutzwiller 2005). While habitat degradation or loss is proposed as the primary threat to NIDGS and managers implement habitat restoration based on this assumption, to date no study has quantified habitat characteristics. Restoration ecology emphasizes that ecosystem structure, species diversity, and ecosystem processes are essential to attain restoration goals, but these habitat attributes are often not measured (Ruiz-Jaen and Aide 2005). By measuring these attributes, my study provides managers with quantifiable goals, which they can continually measure to track restoration progress. Investing in descriptive studies to quantify reference sites will help steer restoration goals, improve restoration efforts, and make management actions more efficient. Improving our understanding of wildlife habitat will increase our ability to maintain threatened and endangered species.

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Tables

Table 1. Percent closed canopy cover (%), tree size (number of trees per ha grouped by diameter at breast height size categories) and tree density (number of trees per ha) at reference sites occupied by the northern Idaho ground squirrel (*Urocitellus brunneus*). Individual site means and standard errors for percent canopy cover, tree size categories and total tree density are listed. Across site (n = 7) Bayesian means and 95% highest posterior density intervals (95% HPDI) are provided.

Reference Sites	Canopy Cover	<8 cm	8-12 cm	13-20 cm	>20 cm	Density
Cap Gun	18 ±5	133 ±60	0	4 ±4	21 ±16	187 ±76
Lost Valley	20 ±7	174 ±104	4 ±4	0	21 ±10	139 ±76
Mud Creek	13 ±6	73 ±44	6 ±6	0	19 ±9	97 ±53
OX Ranch	6 ±3	35 ±20	10 ±8	8 ±7	4 ±4	58 ±27
Price Valley	14 ±4	46 ±22	0	0	15 ±8	60 ±26
Slaughter Gulch	12 ±5	41 ±27	4 ±4	12 ±6	4 ±4	73 ±37
Summit Gulch	2 ±2	0	0 ±2	12 ±6	8 ±8	23 ±10
Reference mean	12	66	5	4	17	91
Reference 95% HPI	10 - 15	40 - 91	2 - 8	2 - 6	11 - 22	64 - 118

Table 2. Understory visual obstruction at reference sites occupied by the northern Idaho ground squirrel (*Urocitellus brunneus*). Individual site (n = 6) means and standard errors along with across site (n = 7) Bayesian mean and the 95% highest posterior density interval (95% HDPI) are provided.

Reference Sites	Visual Obstruction (cm)
Cap Gun	30 ±4
Lost Valley	19 ±2
Mud Creek	16 ±4
OX Ranch	31 ±5
Price Valley	21 ±4
Slaughter Gulch	16 ±3
Summit Gulch	24 ±3
Reference sites	22
Reference 95% HPDI	20 - 25

Table 3. Percent ground cover averaged across reference sites (n = 7) occupied by the northern Idaho ground squirrel (*Urocitellus brunneus*). The Bayesian means and the 95% highest posterior density intervals (95% HPDI) are reported for each parameter.

Functional Group	Type/Growth Form	Percent Cover	95% HPDI
Surface	Litter	14.5	11.9 - 17.3
	Soil	19.3	16.3 - 22.3
	Woody Debris	4.2	3.0 - 5.4
	Rock	3.9	2.8 - 4.9
	Moss/Lichen	1.5	0.8 - 2.2
Forb	Annual	1.5	1.3 - 1.7
	Perennial	3.7	3.4 - 4.1
	Annual/Perennial*	0.6	0.3 - 1.0
Grass	Annual	1.4	0.9 - 1.8
	Perennial	3.7	3.2 - 4.3
Rush	Perennial	0.7	0.3 - 1.1
Sedge	Perennial	1.9	1.1 - 2.7
Shrub	Perennial	3.0	2.2 - 3.8

* growth form can either be annual or perennial and isn't determined for this area

Table 4. The average species richness, evenness and diversity of understory plant species at the reference sites (n = 6) occupied by the northern Idaho ground squirrel (*Urocitellus brunneus*) and across sites (n = 7) in 2010 and 2011. Shannon's Evenness Index values are between 0 and 1; communities with even distributions of species have index values near 1. As Simpson's Index of Diversity gets closer to 1 the greater the site diversity, while communities with low diversity have an index value near 0.

Site Name	**Richness**	**Shannon's Evenness**	**Simpson's Diversity Index**
Cap Gun	34	0.83	0.92
Lost Valley	30	0.83	0.91
Mud Creek	21	0.76	0.83
Price Valley	26	0.74	0.85
OX Ranch	33	0.82	0.91
Slaughter Gulch	29	0.83	0.91
Summit Gulch	33	0.81	0.91
Across Sites	29	0.81	0.89

Table 5. Understory forbs, grasses, and shrubs with the highest percent canopy cover and highest constancy for reference sites (n = 7) occupied by the northern Idaho ground squirrel (_Urocitellus brunneus_). The Bayesian means and the 95% highest posterior density intervals (95% HPDI) are reported for each plant.

Type	Species	Common Name	Mean	95% HPDI
Forb	_Achillea millefolium_* [÷†]	common yarrow	5.1	4.1 - 6.1
	_Allium sp._* [÷]	onion	1.1	0.6 - 1.6
	Anaphalis margaritacea [÷†]	western pearly everlasting	3.4	2.6 - 4.4
	Epilobium brachycarpum [÷]	tall annual willowherb	1.6	1.1 - 2.1
	_Eriogonum sp._* [†]	Buckwheat	3.3	1.8 - 4.8
	Fragaria virginiana	Virginia strawberry	1.5	0.9 - 2.1
	_Lupinus sp._*	lupine	2.1	1.3 - 3.0
	Polygonum douglasii [÷]	Douglas' knotweed	1.6	1.3 - 2.0
	_Potentilla gracilis_* [†]	slender cinquefoil	2.8	1.9 - 3.8
	Sedum stenopetalum [÷†]	wormleaf stonecrop	2.2	1.4 - 3.1
Grasses	_Bromus tectorum_* [fl]	cheat grass	1.1	0.6 - 1.7
	_Poa bulbosa_* [†‡]	Bulbous bluegrass	2.2	1.1 - 3.3
	_Poa pratensis_*	Kentucky bluegrass	1.8	0.7 - 2.7
	Poa secunda [†]	Sandberg bluegrass	1.7	1.2 - 2.2
	_Pseudoroegneria spicata_*	bluebunch wheatgrass	2.9	1.6 - 4.2

* species that comprise > 1% of the _U. brunneus_ diet

[÷] > 20 occurence in 84 transects (highest constancy)

[†] > 2% average canopy cover

fl non-native species

Table 6. Litter depth at reference sites occupied by the northern Idaho ground squirrel (*Urocitellus brunneus*). Individual site means (n = 6) and standard errors along with the across site (n = 7) Bayesian mean and the 95% highest posterior density interval (95% HPDI) are provided.

Reference Sites	Litter Depth (cm)
Cap Gun	1.2 ±0.3
Lost Valley	0.9 ±0.2
Mud Creek	0.5 ±0.2
OX Ranch	1.2 ±0.3
Price Valley	0.5 ±0.2
Slaughter Gulch	0.5 ±0.1
Summit Gulch	0.7 ±0.1
Reference sites	0.8
Reference 95% HPDI	0.7 - 0.9

Table 7. Soil physical properties of the northern Idaho ground squirrel (*Urocitellus brunneus*) reference habitat.

Reference Sites	Sand (%)	Silt (%)	Clay (%)	Soil Texture
Cap Gun	25	40	35	Clay Loam
Lost Valley	30	37.5	32.5	Clay Loam
Mud Creek	27.5	35	37.5	Clay Loam
OX Ranch	17.5	45	37.5	Silty Clay Loam
Price Valley	32.5	35	32.5	Clay Loam
Slaughter Gulch	38.8	37.5	23.5	Loam
Summit Gulch	27.5	37.5	35	Clay Loam

Table 8. Soil chemical properties identified at northern Idaho ground squirrel (*Urocitellus brunneus*) reference habitat. Individual site means (n = 6) and standard errors along with the across site (n = 7) Bayesian mean and the 95% highest posterior density interval (95% HPDI) are provided. Soil NO^{3-}-N and NO^{4+}-N were only collected from Cap Gun, Price Valley and Summit Gulch in 2011.

Reference Sites	ph	P (mg/kg soil)	Ca^{2+} (mg/kg soil)	NO^{3-}-N (mg/kg soil)	NH^{4+}-N (mg/kg soil)
Cap Gun	6.17 ±0.07	328 ±26	52,585 ±1450	6.07 ±0.36	0.26 ±0.12
Lost Valley	6.00 ±0.08	360 ±12	48,366 ±5874	*	*
Mud Creek	5.85 ±0.03	630 ±152	53,966 ±5581	*	*
OX Ranch	6.13 ±0.07	460 ±74	72,747 ±826	*	*
Price Valley	6.28 ±0.08	390 ±61	54,026 ±1005	6.73 ±1.50	0.69 ±0.16
Slaughter Gulch	6.15 ±0.08	320 ±30	56,047 ±2651	*	*
Summit Gulch	6.09 ±0.02	367 ±30	45,713 ±2354	6.46 ±0.25	0.70 ±0.09
Reference sites	6.12	394	53,594	6.38	0.53
Reference 95% HPDI	6.07 - 6.17	350-437	50,354 - 57098	5.85 - 06.87	0.37 - 0.70

Figures

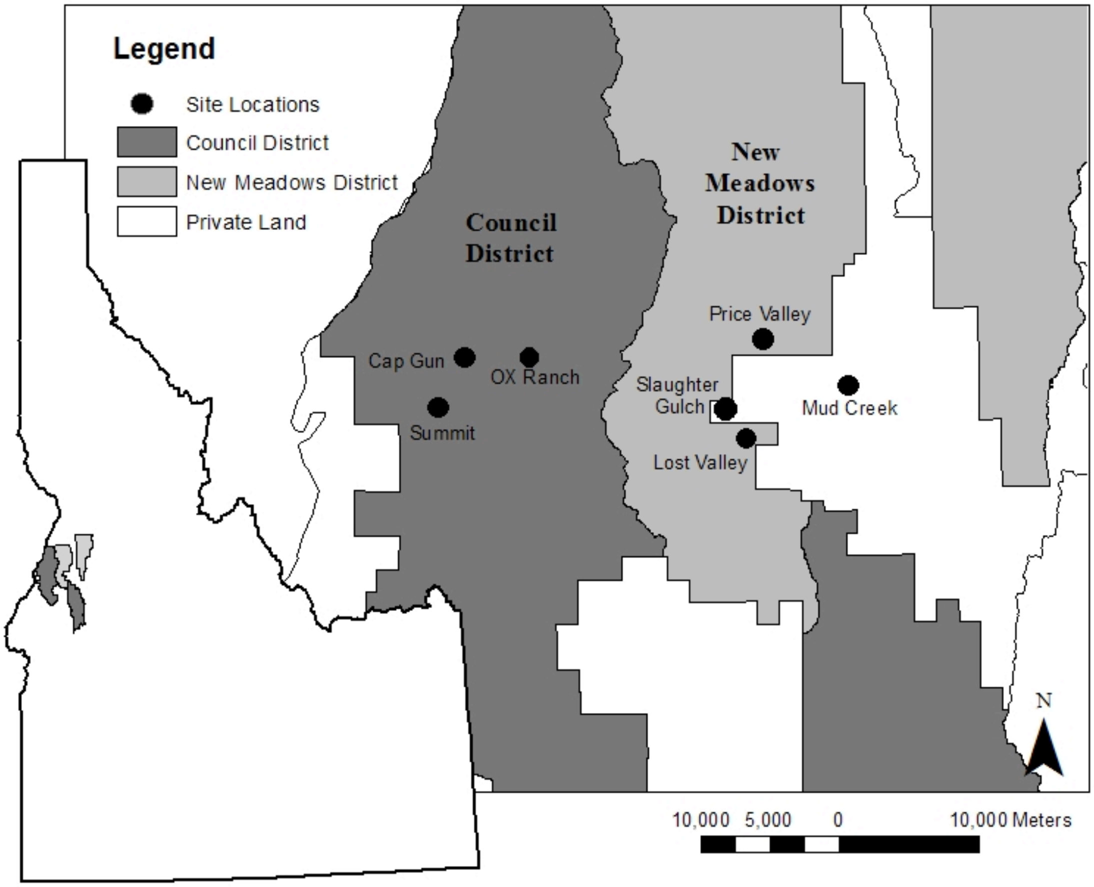

Figure 1. Map of the seven reference sites (black dots) in west-central Idaho. The Cap Gun and Summit reference sites are within the US Forest Service Council District. Lost Valley and Slaughter Gulch reference sites are within the US Forest Service New Meadows District. The OX Ranch (although surrounded by US Forest Service land), Mud Creek and Price Valley reference sites are privately owned.

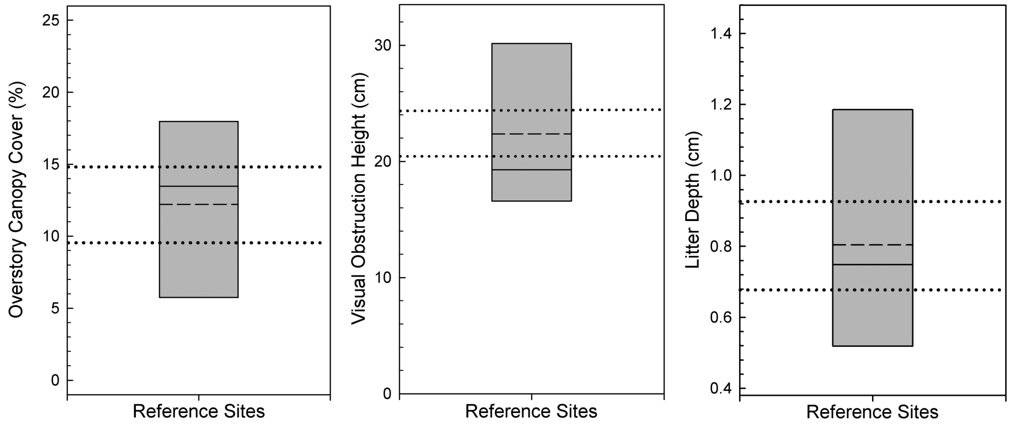

Figure 2. Box plots of closed tree canopy cover, visual obstruction and litter depth across reference sites (n = 7). The bottom of the box is the 25th percentile of the data, while the top of the box is the 75th percentile of the data. The dashed line is the mean and the solid line is the median. The dotted lines are the lower and upper bound of the 95% highest posterior density interval.

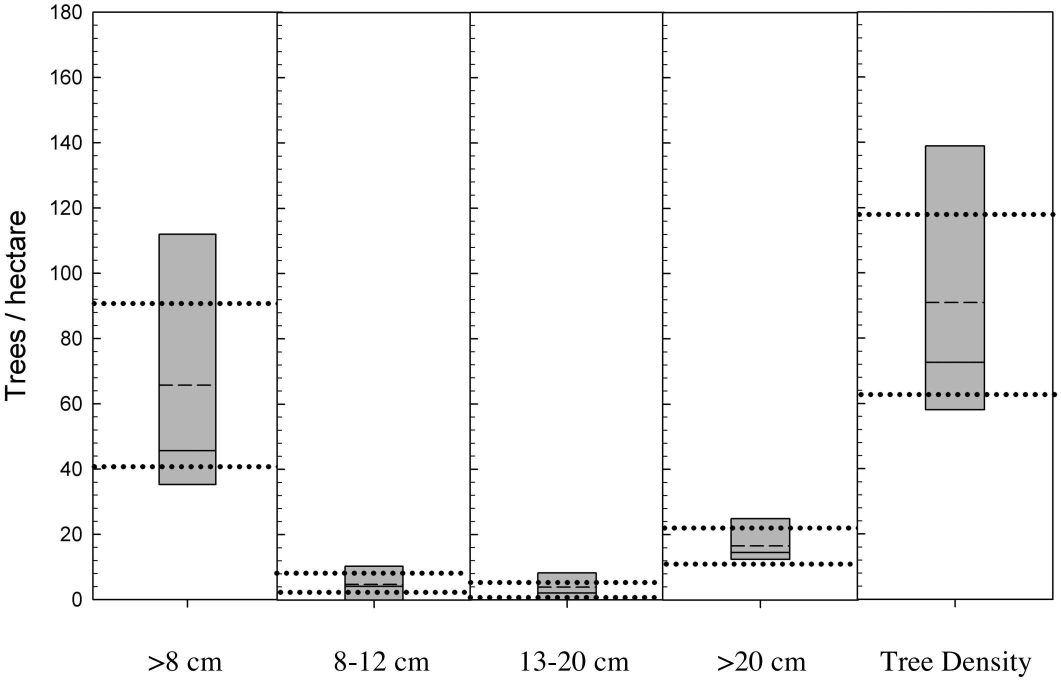

Figure 3. Box plots of tree density (#trees per ha) for the four diameter at breast height size classes and total trees across reference sites (n = 7). The bottom of the box is the 25th percentile of the data, while the top of the box is the 75th percentile of the data. The dashed line is the mean and the solid line is the median. The dotted lines are the lower and upper bound of the 95% highest posterior density interval.

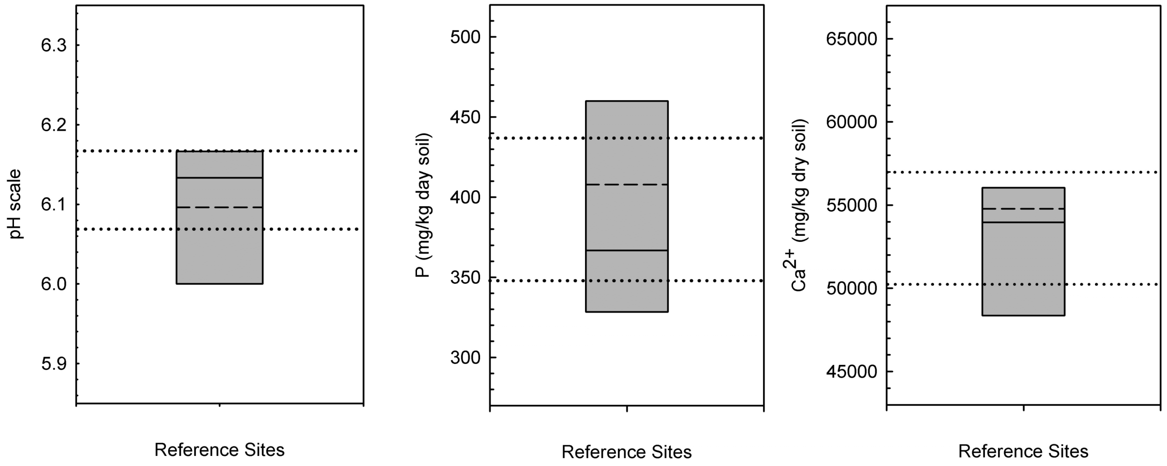

Figure 4. Box plots of soil attributes across reference sites (n = 7), including soil pH, phosphorus and calcium. The bottom of the box is the 25th percentile of the data, while the top of the box is the 75th percentile of the data. The dashed line is the mean and the solid line is the median. The dotted lines are the lower and upper bound of the 95% highest posterior density interval.

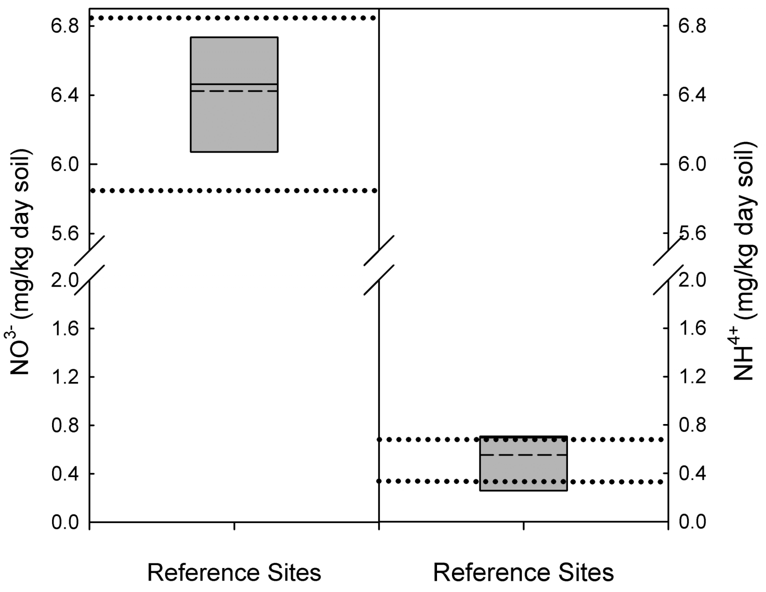

Figure 5. Box plots of inorganic soil nitrogen, nitrate and ammonium, across reference sites (n = 7). The bottom of the box is the 25th percentile of the data, while the top of the box is the 75th percentile of the data. The dashed line is the mean and the solid line is the median. The dotted lines are the lower and upper bound of the 95% highest posterior density interval.

CHAPTER 2

HABITAT RESTORATION FOR NORTHERN IDAHO GROUND SQUIRRELS (*UROCITELLUS BRUNNEUS*): EFFECTS OF PRESCRIBED THINNING AND BURNING ON DWINDLING HABITAT

Abstract

The northern Idaho ground squirrel (NIDGS; *Urocitellus brunneus*) is endemic to Adams and Valley counties in west-central Idaho. In 2000, the NIDGS was listed as threatened primarily due to habitat loss. Open areas within ponderosa pine (*Pinus ponderosa*) forests favored by the NIDGS have decreased due to increased forest extent and density as a result of land use, including wildfire exclusion. Therefore, recovery plans encourage the use of thinning and burning to expand meadows and open corridors. I collected data on habitat attributes altered by prescribed thinning and fall burning at three sites selected for habitat restoration. Each site was divided into two units: a *reference unit* occupied by NIDGS but not treated and a treated *burn unit* not occupied by NIDGS. I sought to assess whether the prescribed thinning and fall burning treatments generated habitat features similar to reference conditions. Burn units had higher densities of pines and tree canopy cover compared to the reference units, which is why they were not occupied and why they are treated. The prescribed thinning and fall burning did not reduce tree density or canopy cover. Understory plant visual obstruction height and litter depth tended to decrease in the burn units to resemble reference conditions although it was not statistically significant. Understory canopy cover was similar between reference and burn units before and one-year post-fire. Plant species composition was highly heterogeneous; however, the prescribed burn shifted species composition toward the reference units. This study captures the effects of thinning and fall

burning one-year post-treatment on NIDGS habitat, which is the first study to evaluate NIDGS restoration practices and help managers to refine their prescriptions.

Introduction

For the past 100 years, anthropogenic land use has changed the historical occurrence and extent of fire in the United States (Steele et al. 1986; Ehle and Baker 2003; Covington 2000; Heyerdahl et al. 2008*a*, *b*). This reduction in fire has important ecosystem consequences considering fire modifies vegetation structure, promotes herbaceous species production (Endstrom et al. 1984; Laughlin et al. 2004), and alters nutrient cycling (Boerner 1982; Neary et al. 1999; Stephens et al. 2009). Fire exclusion has altered ecosystem structure and function (Steele et al. 1986; Barrett 1988; Agee 1998; Bond and Keeley 2005) and can thus affect wildlife habitat (Kennedy and Fontaine 2009, Fontaine and Kennedy 2012).

The primary tools used to restore forests that evolved with frequent fire include thinning, burning, or a combination of the two treatments (Brown et al. 2004; Crist et al. 2009; Schwilk et al. 2009). Ecological restoration is the act of modifying biotic and abiotic elements to improve habitat and promote biodiversity (Young 2000; Miller and Hobbs 2007). Thinning can reduce crown cover by removing large trees to alter solar infiltration and promote herbaceous plant growth (Busse et al. 2000; Agee and Skinner 2005; Allen et al. 2002; Hessburg et al. 2005). Prescribed burning can reduce understory biomass, promote nutrient cycling, and reduce the intensity and severity of subsequent fires (Wagle and Eakle 1979; Crist et al. 2009). Thinning to remove smaller and less fire-tolerant trees followed by prescribed burning can be effective for restoring low-severity fire regimes in dry conifer forests (Brown et al. 2004). Across the western US, managers use these treatments to restore ponderosa pine forests that historically had frequent fires (Stephens et al. 2009). Managers not only apply these practices to restore forest structure and function, but they also use these treatments to restore wildlife habitat (Long and Smith 1983; Gaines et al. 2007; Lyons et al. 2008; Kennedy and Fontaine 2009; Fontaine and Kennedy 2012).

In the absence of fire, ponderosa pine (*Pinus ponderosa*) tree density and expanse has increased in west-central Idaho (Truska and Yensen 1990), and there is strong circumstantial evidence that wildfire suppression has negatively impacted habitat of the northern Idaho ground squirrel (*Urocitellus brunneus*, formerly *Spermophilus brunneus*; NIDGS; Yensen 1991; Helgen et al. 2009; Hoisington 2012; USFWS 2003). Contemporary populations of the NIDGS are small and isolated, which is attributed to habitat loss and fragmentation (Truska and Yensen 1990; Gavin et al. 1999), and thus the NIDGS was listed as threatened under the Endangered Species Act in 2000 (Clark 1998, 2000). Authors examining NIDGS demography and genetics concur that habitat restoration efforts should focus on opening habitat to encourage dispersal among populations (Gavin et al. 1999; Sherman and Runge 2002; Garner et al. 2005).

Prescribed fire and thinning are currently used to restore NIDGS habitat. Restoration efforts began in 1996 (Yensen unpublished data), but early treatments met with mixed results (Doane 2012). Refined prescriptions have reduced tree density and presumably benefited NIDGS by facilitating population dispersal and expansion (W. Rautsaw *pers. comm.* 2010). However, there has been no quantification of changes in habitat after prescribed fire and thinning treatments.

Prescribed fire and thinning treatments were applied in the fall of 2010 to three sites in central Idaho to examine the effects on NIDGS habitat. I established three *reference units* (currently occupied by NIDGS) and three *burn units* (unoccupied by NIDGS, yet adjacent to a site occupied by NIDGS) where various vegetation and soil attributes were measured before and one-year after the restoration treatments were applied. I predicted that there would be differences between the units: 1) the burn units would have more closed canopy cover and higher tree density; 2) the burn unit visual obstruction height would be greater than the reference unit and that litter depths would be deeper in the burn units; 3) soil calcium and phosphorus concentrations would be similar between units; and 4) the nitrate and ammonia levels would be lower in the burn unit due to higher levels of competition due to more trees. also predicted that the restoration burn would change burn unit habitat attributes toward reference conditions: 1) decrease the number of trees in the burn units to a density similar to the reference sites; 2) remove canopy cover because the fire would kill tree and branches; 3) decrease visual obstruction and litter depth toward reference levels; 4) not affect soil calcium and phosphorus levels because the low severity fire would not cause these compounds to volatilize and 5) increase nitrate and ammonium concentration because the combusted biomass would release the inorganic nitrogen reserve from the living biomass.

Methods

Sites and Experimental Design

Experimental sites were meadows and open forest stands of ponderosa pine and Douglas-fir (*Pseudotsuga menziesii*) in Adams County, Idaho. The US Forest Service (USFS) selected three sites for NIDGS habitat restoration in 2010: Cap Gun (45.0150°, -116.7053°), Summit Gulch (44.9921°, -116.7172°) and Price Valley Guard Station (45.0336°, -116.4290°). I divided sites into two units: a *reference unit* and a *burn unit* (Figure

1). The burn units had to be thinned and burned by fall 2010 and had to be adjacent to a reference unit for potential squirrel dispersal after treatments. At each unit, I randomly distributed six 50-m transects within 100 m of actual squirrel sightings from Idaho Fish and Game monitoring surveys (2008). All three sites were at approximately 1,350 m in elevation and were on south- to east-facing slopes. Average annual high temperatures are 13.7°C and lows are -3.7°C in New Meadows (Western Regional Climate Center 2012). The closest weather station to the sites, USFS New Meadows District weather station, reported 701 mm of precipitation in 2010 and 276 mm in 2011, which were about 100 mm above and 315 mm below the average annual precipitation of 591 mm for the last 100 years (Western Regional Climate Center 2012). Data collection occurred during the peak of the growing seasons in mid-July of 2010 and 2011.

Restoration Treatments

Goals of the prescription treatments for NIDGS habitat included: 1) decreasing tree density (trees with a DBH >20 cm), 2) increasing herbaceous community cover by 10-30% one-year post-fire, and 3) retaining an open understory by reducing fuels with DBH <8 cm by 50-90% and decreasing woody debris with DBH >8 cm by 20-65% (Doane 2008; Enna and Cobb 2010). Prescribed thinning and fall burns were conducted by the USFS Council and New Meadows Ranger Districts. Thinning methods varied among sites resulting in varying tree densities (Table 1). Prescribed fire was used to achieve tree thinning rather than mechanical methods at Cap Gun, but rains decreased the intensity of the fire and fewer trees were thinned than planned. Price Valley Guard Station was thinned mechanically from September 2008 to February 2009; trees with a DBH <20 cm were cut to attain 222-296 trees/ha and were left to provide fuel for the prescribed burn. The density of large trees (DBH

>20 cm) was reduced from 210-321 trees/ha to 57-119 trees/ha via thinning and was followed by a fall prescribed burn. Prescribed fire was also used to thin the burn unit at Summit Gulch, but few trees were killed because fire intensity was intentionally low to protect trees in adjacent timber sales. Prescribed burns were conducted in September and October 2010 when NIDGS were in hibernation underground to avoid any direct impacts on the NIDGS (Table 2).

Field Measurements

I established two 200-m^2 circular plots with an 8-m radius centered at the 0 and 50-m marks of the 50-m transect to describe tree forest structure. Tree density was measured by size class and canopy cover; forest structure was characterized by the percentage of closed canopy cover with a concave spherical crown densiometer. Densiometer measurements were taken one meter above the ground in the four cardinal directions around the center of the plot (Lemmon 1956). Every tree within the circular plots was counted and placed into a size category to measure tree density and size distribution. The tree size categories were determined by diameter at breast height (DBH <8 cm, 8-12 cm, 13-20 cm, >20 cm). The diameter of trees with a DBH >20 cm was recorded to the nearest 0.1 cm.

I described understory vegetation height in order to estimate whether or not it was a visual obstruction to the NIDGS. Along each transect, I recorded visual obstruction at three transect points (0, 25, and 50-m marks) in the four cardinal directions with a modified Robel pole (Robel et al. 1970; Herrick et al. 2009). The modified Robel pole had markings every 2 cm from 0-40 cm and measurements were recorded above 40 cm as 41-50 cm, 51-100 cm or >100 cm. Detailed measurements were collected under 40 cm because understory vegetation below that height could obstruct the vision of the NIDGS, which are 15-22 cm tall when standing on their hind legs (Yensen and Evans Mack *per. comm.* 2012). I assessed percent cover of understory species ocularly within a 0.5 x 0.5 m quadrat and used Daubenmire (1959) cover class groups (0 - 5%, 6-25% 26-50%, 51-75%, 76-95%, 96-100%). Voucher specimens were collected for a reference collection of the understory species.

I quantified understory species cover with ten quadrats every five meters along a 50-m transect (60 plots per unit). Plant cover and the percentage of ground that was soil (bare ground and rocks < baseball size), litter, woody debris (branches, pieces of wood, etc.), and rocks (roughly > baseball size; large enough for a squirrel to stand on) were recorded along with the percentage of moss and lichen cover. Some plants were grouped by genus due to similarities between species; therefore, richness is slightly underestimated. I also categorized understory vegetation by functional group: annual forb, perennial forb, annual/perennial forb (species that can take either growth form), annual grass, perennial grass, sedge, rush, and shrub.

Litter depth (top of litter debris to soil surface) was recorded every five meters along each transect. I systematically sampled thirty soil cores (10 to 15-cm deep) randomly along each transect to analyze soil chemistry. Thirty cores were homogenized into one soil composite for each transect (6 composites per unit) and 20g was subsampled to analyze for pH, calcium, phosphorous, nitrate, and ammonium. In 2011, I assessed nitrate and ammonium concentrations by combining 10 g of a soil composite with 50 ml of 2M KCl to terminate nutrient cycling. KCL filtrate solution was analyzed with a FIAlab-2500 system to obtain soil nitrate and ammonium levels (Bundy and Meisinger 1994).

Statistical Analyses

I transformed several dependent variables to improve data normality and homogeneity of variance: canopy cover, visual obstruction, and litter depth were square root transformed; tree density and tree size classes were log+1 transformed; species composition and functional group data were transformed by arcsine-square root. Rare species were excluded from the vegetation analysis. Rare species are classified as those that occurred in four or fewer transects (5% of the 72 transects) (McCune and Grace 2002). Soil pH and phosphorus were transformed with log10+1; calcium, nitrate and ammonium were not transformed because no transformation improved the normality or homogeneity of variance.

Vegetation and soil data were tested for differences across years (n = 2) and units (n = 6) in a non-parametric MANOVA (McArdle and Anderson 2001) using DISTLM software (Anderson 2004). I tested for an interaction between ‘year’ and ‘unit’, which would indicate a restoration treatment effect across years (hereafter referred to as a ‘restoration treatment’ effect). The dependent variables were divided into six groups: 1) vegetation attributes, including litter depth; 2) tree size classes; 3) species composition; 4) plant functional groups; 5) soil attributes; and 6) soil nitrate and ammonium concentrations (ran separately from the other soil attributes due to a small sample size, n = 3, and lack of normality). For each set of dependent variables, the model contained fixed effects of ‘year’ (tested over ‘year’ by ‘site’ interaction), ‘unit’ (tested over the ‘unit’ by ‘site’ interaction), and the interaction between ‘year’ and ‘unit’ (tested over the three-way interaction among ‘year’, ‘unit’, and ‘site’). *P*-values were based on 999 permutations or the Monte Carlo Method (permuted *P*-value). The analysis was performed on normalized data using Euclidean distances for all variables except understory composition and functional group data for which I used Bray-Curtis distances

(Legendre and Legendre 1998). I did not standardize species composition and functional group data by row in the PERMANOVA; therefore, the results depended upon species composition and cover. Significant terms in the multivariate model were further explored with univariate mixed model ANOVAs using SAS/STAT® version 9.2 software (SAS Institute Inc. 2008). I ran t-tests on *a priori* contrasts using Tukey-Kramer corrections for variables with significant differences in the ANOVA output. Statistical significance was set at an alpha of 0.05.

Plant community responses were analyzed using non-parametric multidimensional scaling (NMS) ordination to explore patterns in understory species composition (McCune and Grace 2002). I grouped understory cover data by 'year' (2010 = Before and 2011 =After Burn) and 'unit' (Reference or Burn). Non-parametric multidimensional scaling was run with Sorensen (Bray-Curtis) distance metric in PC-ORD version 6 software (McCune and Mefford 2011). The NMS ordination ran with 250 runs with real data and 250 with randomized data. PC-ORD software was also used to calculate species richness and evenness.

Results

Tree canopy cover and tree density were not affected by the restoration treatment (Table 3). Tree canopy did not change from 2010 to 2011, but canopy cover was significantly greater in the burn units than in reference units (Table 3, Figure 2a), while tree density was not different between years and units (Table 3, Figure 2b).

There was no overall treatment effect on the height of understory visual obstruction (PERMANOVA; Table 3); the lack of a significant treatment effect on understory visual obstruction was likely due to a small sample size (n = 3). Visual obstruction height decreased overall between 2010 and 2011 (Figure 3), but burn units responded differently than reference units based on the *a priori* contrast ($P = 0.16$). Visual obstruction height decreased 43% from 2010 to 2011 in the burn unit [2010, 37.5± (5.2, 5.5) cm [mean± (lower, upper SE)]; 2011, 21.1± (3.8, 4.2) cm; $P < 0.01$]. In comparison, the reference visual obstruction slightly decreased by 25% from 2010 to 2011 [2010, 26.6± (4.3, 4.7) cm; 2011, 20.0± (3.7, 4.1), $P = 0.5$). Visual obstruction in the burn units decreased to levels similar to reference conditions after the burn (Figure 2c).

There was no overall restoration treatment effect on litter depth (PERMANOVA; Table 3). Overall, litter depth was significantly different by year and unit (Univariate, Table 3). Based on *a priori* contrasts, the burn unit responded differently than the reference unit. Litter depth in the burn units was 58% deeper than litter depth at the reference unit in 2010 [burn unit, 2.4± (1.91, 3.40) cm; mean± (lower, upper SE); reference units, 0.99± (0.98, 2.46) cm; $P < 0.0001$]. One year after the restoration treatment, litter depth decreased in the burn unit by 21% [burn unit, 0.86± (0.85, 2.34); reference unit, 0.54± (0.52, 2.00); $P = 0.1$). The restoration treatments reduced litter depth in the burn units toward reference unit levels (Figure 2d).

The restoration treatment had no overall effect on tee density by size categories, understory species cover or functional group cover (Table 3, Table 4 and Figure 3a-f). Understory species richness slightly increased in the burn units (32 species in 2010 to 36 species in 2011); while the number of species at the reference units stayed the same (26 species) (Figure 3g). Evenness for both units was high and did not vary annually or by treatment (Figure 3h).

Patterns in understory community composition were explained by a one-axis solution selected after 14 iterations with a final instability of 0.0047 in the NMS ordination (Figure 4). Understory community patterns of the reference did not change between 2010 and 2011 on Axis 1, while the composition in the burn unit shifted closer to reference units after the restoration treatment (Figure 4). Although not statistically significant, Axis 1 explained 81% ($r^2 = 0.81$) of the variation in species composition between the groups (stress < 0.00001, P = 0.099). Due to minimal annual variation at the reference sites, Axis 1 appeared to be correlated with the effects of the prescribed burn. Due to a small sample size and high levels of heterogeneity, I was unable to detect which plant species were affected by the fire.

Soil attributes overall were not affected by the restoration treatment (PERMANOVA; Table 3). Soil pH did not vary between years, but it was significantly different between units (Univariate; Table 3). The burn unit pH was slightly more acidic than the reference units in 2010 and in 2011 the burn unit acidity decreased (Figure 5a); however, this change was not significant ($P = 0.11$).

The restoration treatment did not affect soil calcium and phosphorus concentrations (Table 3). Soil calcium concentrations at both units mirrored the same annual trend, significantly decreasing in 2011 (Figure 5b). There were no changes in soil phosphorus levels between years and units (Table 3, Figure 5c). In 2011, soil nitrate and ammonium were not significantly different between reference and post-treated burn units (pseudo-$F_{1,2} = 1.64$, permuted $P = 0.29$); however, both tended to be slightly higher in the burn units (Figure 6a and 6b).

Discussion

Habitat restoration treatment goals for NIDGS focused on reducing conifer tree density, increasing herbaceous community cover, and creating an open understory by reducing ladder fuels (Doane 2008; Enna and Cobb 2010). However, these goals were not achieved within one-year post-treatment. Tree canopy cover, tree density, tree size classes, understory species cover, understory functional group cover, and soil chemistry were not affected by the restoration treatment at these sites, likely because prescribed fires without prior thinning did not result in tree mortality, even of trees < 20 cm DBH. Understory visual obstruction height and litter depth were reduced by the restoration treatments toward levels found in reference units. In addition, the restoration treatments changed the understory community toward reference sites. While restoration goals were not met within one-year post-fire, these goals could be attained if the treatments are repeated. The restoration treatments altered a few habitat attributes not included in those goals, which should be incorporated into future restoration goals.

Habitat restoration treatments did not decrease conifer cover or density. Reducing tree cover and tree density in potential NIDGS habitat is important because tree cover shades understory flora, thus influencing which understory plants can grow. Many dead trees were still standing in the burn units and holding onto fire-damaged needles; hence, tree canopy cover in the burn unit remained high one-year post-fire. Tree cover might decrease in subsequent years once needles and dead branches fall.

Restoration treatments were not intense enough to change tree canopy cover and tree density. The forest at Price Valley Guard Station was already thinned before 2010 measurements; therefore, no significant decrease in tree density was expected. Weather and concerns about adjacent forests can interfere with the success of prescribed fires. However, burn-only prescriptions may not be enough to reduce conifer density even three years after fire (Fielder et al. 2010). Thinning by burning is often unpredictable and difficult to control (Harrington and Sackett 1990). Mechanical thinning prior to prescribe burning probably would have achieved thinning goals (Sackett et al. 1996; Fielder et al. 2010). Repeated treatments will likely be needed to obtain desired tree density and also for long-term restoration goals (Crane and Fischer 1986; Keane et al. 1990, Fulé et al. 2001, Stephens et al. 2012).

Visual obstruction was not a specific restoration goal, yet visual obstruction is important to small rodents for predator avoidance (Lima and Dill 1990). I predicted that the burn would significantly reduce visual obstruction by removing understory plants and young trees. While prescribed fire effects were not significant, there was a strong trend for the prescribed fire to reduce visual obstruction levels toward reference conditions. While others have evaluated visual obstruction within ponderosa pine forest for turkeys (*Meleagris gallopavo merriami*) and woodpeckers (*Melanerpes lewis*) (Lutz and Crawford 1987; Linder and Anderson 1988), changes in visual obstruction pre- and post-fire for ponderosa habitat restoration have not been examined. In another study, visual obstruction was significantly reduced one-year post-fire in sand shinnery oak habitat, but visual obstruction recovered to control site levels within five years post-fire (Harrell et al. 2001).

The second restoration goal was to increase herbaceous community cover; however, individual species and functional group cover in the understory was similar between the reference and burn units before and after the burn. Thus, the goal of increasing herbaceous plant cover by 10-30% one-year post-fire was not attained. Since tree canopy cover did not decrease, the intended alteration environmental conditions to benefit understory species did not occur; therefore, an increased growth and reproduction response by the understory species may be delayed (McConnell and Smith 1970; Busse et al. 2000; Sabo et al. 2009). The cover of some understory species may increase in subsequent years after a fire (ex. *Potentilla gracilis*, *Fragaria virginiana, Geum triflorum*, *Elymus elymoides* and *Carex* sp.; Armour et al. 1984; Busse et al. 2000).

Based on community patterns before and after treatment, I suggest that managers should focus on plant community responses to restoration treatments. Patterns in the understory community at the reference unit did not change from 2010 to 2011. Meanwhile, the burn unit community was different from the reference sites before the treatment shifted toward reference conditions after the restoration treatment. Although this shift was not significant, this strong trend is probably important to the NIDGS. Fires might alter vegetation community composition and structure in ways particularly beneficial for NIDGS. Community shifts are difficult to detect especially when the understory is highly heterogeneous and no significant differences are detected in cover. The understory community in the burn units might continue to shift over time or with subsequent burns (Busse et al. 2000; Laughlin et al. 2004), which warrants continued monitoring.

Forest litter depth is important because it influences the recruitment and development of plants. Litter covers bare soil, serves as mulch influencing soil moisture and soil temperature, alters chemical inputs, affects water infiltration, and influences the degree of soil heating during a fire (Neary et al. 1999, 2005). In my study, pre-burn litter was thicker in the burn units compared to the reference units, which was likely due to the higher tree density increasing the needle load. After the prescribed fire, litter depth decreased toward levels similar to reference units. Other prescription fires in ponderosa pine forest decreased litter depth for habitat restoration (Bock and Bock 1983). Litter would likely remain low post-fire if conifer density was decreased. Ponderosa pines drop needles at a higher rate compared to other trees, which often creates a thick layer of litter that is slow to decompose and deters herbaceous species growth (Pase 1958; Hart et al. 1992; Biswell et al. 1966). After litter levels have been lowered, the herbaceous community should respond with increased production (Xiong and Nilsson 1999; Melten and Fielder 2006), which could lead to the desired response of increased understory percent cover.

The prescribed fire slightly shifted soil pH but did not alter soil calcium, phosphorus or inorganic nitrogen. The minimal change in soil pH was likely not biologically significant, since nutrient availability for microbes and plants is highest in soils with a pH around 6.5 (Neary et al. 2005). Fire can increase soil pH in ponderosa pine forests due to the combustion of organic matter; however, others have found no change in pH after fire (Covington and Sackett 1984; DeBano 1991; Neary et al. 2005; Koyama 2010).

Calcium volatilizes at temperatures above 450˚C, which is characteristic of severe fire (Neary et al. 2005); therefore, I did not predict a change in calcium after the low severity understory burn. While the restoration treatment did not affect calcium, I found high levels of calcium in both reference and burn units in 2010 that sharply decreased in 2011 from unknown causes. Soil phosphorus can also volatilize at high temperatures (Neary et al. 2005) characteristic of a medium to high severity fires. I predicted no change in soil phosphorus levels in either unit. Indeed, the prescribed fire did not affect phosphorus. Ponderosa pine ecosystems are often limited in phosphorus, nitrate, and ammonium (Covington and Sackett

1990), and prescribed fires have the potential to increase phosphorus, nitrate, and ammonium availability.

Soil nitrate and ammonium concentrations in the post-fire burn units were not significantly different from the reference units. While inorganic nitrogen is often affected by fire (Covington and Sackett 1990), it is possible I did not capture the spike in nitrogen (if it occurred) because I sampled once approximately ten months after the fire. Gundale et al. (2005) only detected a soil nitrate increase in samples collected in the wet season; thus, it is likely that I could have detected changes in inorganic nitrogen had the cores been collected in the spring or immediately after the burn. However, Koyama et al. (2012) reported increased in nitrate concentrations in burnt soils 1.5-2 years post-fire, pooling wildfire and prescribed fire data, compared to unburned sites. Nitrate concentrations were not significantly different between the reference and burn units, yet these concentrations were higher than those reported in other studies (Covington and Sackett 1986; Covington and Sackett 1990; Koyama et al. 2010, 2012). Without measuring nitrification rates and microbial uptake, I cannot deduce reasons for high levels of nitrate compared to other ponderosa pine forest and other forests in central Idaho. Ammonium levels I measured in the reference and burn units were similar to those in neighboring conifer forests of central Idaho (Koyama et al. 2010).

Restoration treatments are applied to help an ecosystem recover from a degraded state (Society of Ecological Restoration 2004). Treatments should be designed to achieve desired goals, which should be modeled after a reference site (White and Walker 2008). Reference sites typically represent the restoration potential of a restoration project; hence quantitative data is necessary to describe reference sites in order to create restoration goals (Hobbs 2007; Ruiz-Jaen and Aide 2005; White and Walker 2008). Habitat attributes of seven sites occupied by the NIDGS were recently quantified over a two-year period (Chapter 1), which could be used to guide restoration treatment goals. Reference sites were described as areas with tree canopy cover of 10-15%, low tree densities of 63-118 trees/ha and visual obstruction heights of 20-25 cm (Chapter 1). Using reference information and evaluating the effects of thinning and burning treatments on habitat can be used to further refine restoration goals.

While my study only measured changes in habitat one-year post-treatment, it is suggested that investigations continue for 2-5 years post-treatment. Higher intensity burns or mechanical treatments may be advised to reduce mid- to large tree density, and repeated treatments may be necessary to further achieve restoration goals. There is limited information regarding the effects of repeated burns, yet the literature suggests follow-up burning in ponderosa pine forests should occur from anywhere from 2 to 20 years after initial treatments (Harrington and Sackett 1990; Covington et al. 2001; Scott 2002). Repeated burns are often necessary in ponderosa pine ecosystems to maintain desirable forest structure, fuel loads, litter depths and stimulate the nitrogen cycle (Covington and Sackett 1986; Sackett et al. 1994).

Habitat restoration is not an exact science. Evaluating prior restoration actions in an adaptive management framework will assist in refining restoration treatments and goals (Block et al. 2001; Boyd and Svejcar 2009). Restoration treatments and sound management practices are essential for habitat and the persistence of species at risk (Foin et al. 1998; Anderson and Gutzwiller 2005).

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Tables

Table 1. Tree density (mean #trees per ha) for three sites occupied by the northern Idaho ground squirrel after restoration treatments. Tree species include ponderosa pine (PIPO = *Pinus ponderosa*) and Douglas-fir (PSME = *Pseudotsuga menziesii*). Tree densities are divided up into the different diameter at breast height (cm) size classes.

Site	Species	<8	8-13	13 - 20	> 20	Total
Cap Gun	PIPO	54 ±25	25 ±15	25 ±17	46 ±16	158 ±70
	PSME	4 ±4	4 ±4			
Price Valley Guard Station	PIPO	178 ±111	4 ±4	0 ±0	50 ±18	232 ±118
	PSME	4 ±4				
Summit Gulch	PIPO	99 ±90	17 ±8	37 ±19	95 ±36	249 ±111

Table 2. Fire behavior characteristics of the prescribed burns at Cap Gun, Price Valley Guard Station and Summit Gulch.

Measurement	Cap Gun	Price Valley Guard Station	Summit Gulch
Day	Sept. 8, 2010	Oct. 11, 2010	Sept. 2, 2010
Time of day	0940-1200	1400-1600	1200-1855
Area (ha)	68	64	25
Ambient temperature (°C)	16	6	21-26
Relative humidity (%)	47	19	20
Wind speed (km/hr)	1-1.6	1.6-6.4	3.2-9.6
Wind direction	S	SW	SW
Fuel moisture (%)			
1-Hr	11	7	5-6
10-Hr	16	9	8-11
100-Hr	17	10	12-14
1000-Hr	-	33	-
Fuel Consumption (%)			
Litter	-	20-50	-
<2.5 cm	0-20	-	85-90
2.5-7.6 cm	10-30	-	30-60
100-1000 hr fuels	-	20	-
Tree mortalitiy (%)	<10	<15	20
Area burned (%)	40	70	60

Table 3. Results from the non-parametric multivariate analysis of variance (PERMANOVA) analysis for vegetative attributes, tree size classes, species composition, functional groups and soil attributes. Univariate mixed model analysis of variance (ANOVA) results are presented if statistical significance was detected in the PERMANOVA. Canopy cover, tree density, visual obstruction and litter depth were variables within the vegetation attributes PERMANOVA analysis. Soil pH, calcium and phosphorus concentrations were variables in the soil attribute PERMANOVA analysis. The 'restoration treatment' effect is the interaction between year x unit. Statistically significant *P*-values are bold when $\alpha \leq 0.05$.

		Effects					
Analysis	Data	Year		Unit		Restoration Treatment	
		$F_{1,2}$	*P*-value	$F_{1,2}$	*P*-value	$F_{1,2}$	*P*-value
PERMANOVA	Vegetative Attributes	18.20	**0.002**	5.89	**0.009**	3.01	0.08
Univariate	Canopy Cover	4.50	0.17	67.80	**0.014**	0.31	0.64
	Tree Density	0.23	0.68	10.03	0.09	0.42	0.58
	Visual Obstruction	34.38	**0.028**	8.83	0.97	4.83	0.16
	Litter Depth	54.79	**0.018**	38.44	**0.025**	8.61	0.10
PERMANOVA	Tree Size Classes	0.11	0.87	0.82	0.86	2.92	0.21
PERMANOVA	Species Composition	3.32	**0.002**	1.52	0.28	1.69	0.11
PERMANOVA	Fuctional Groups	1.45	0.27	1.33	0.32	0.71	0.69
PERMANOVA	Soil Attributes	24.35	**0.001**	4.75	**0.046**	0.91	0.51
Univariate	pH	4.12	0.18	45.89	**0.02**	7.90	0.11
	Calcium	295.37	**0.003**	0.06	0.83	0.01	0.95
	Phosphorus	0.02	0.90	1.56	0.34	2.12	0.28

Table 4. Mean percent cover for functional groups within reference and burn units in 2010 and 2011. Errors are reported as the standard error of mean.

Functional Group Cover (%)	Reference		Burn	
	2010	2011	2010	2011
Annual Forbs	1.4 ±0.2	0.8 ±0.2	0.6 ±0.1	1.9 ±0.3
Perennial Forbs	3.8 ±0.5	4.6 ±0.5	2.7 ±0.3	3.2 ±0.3
Annual/Perennial Forb	1.9 ±0.6	2.5 ±0.7	2.3 ±0.4	2.3 ±0.4
Annual Grass	1.6 ±0.4	2.7 ±0.5	0.9 ±0.2	0.9 ±0.1
Perennnial Grass	2.4 ±0.3	4.1 ±0.6	2.7 ±0.7	4.7 ±0.8
Shrub	4.7 ±1.1	5.9 ±1.5	5.0 ±0.7	5.0 ±0.9

Figures

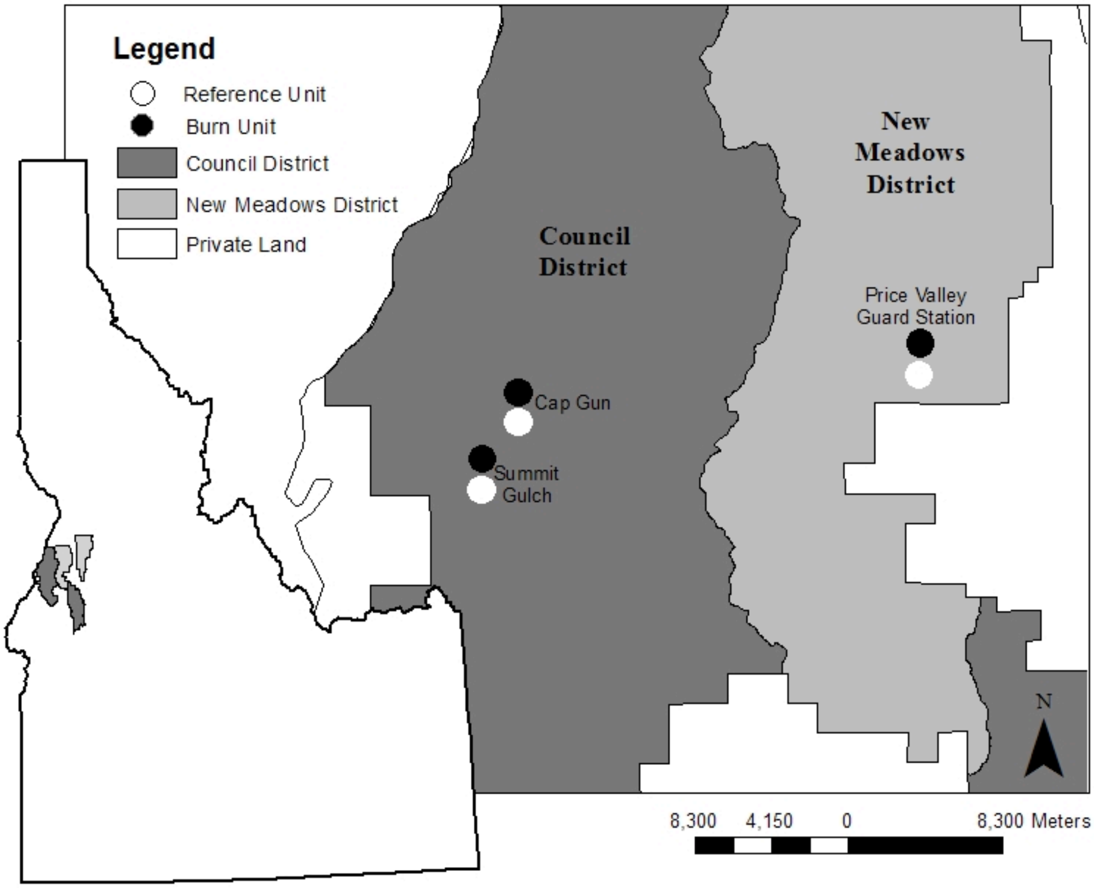

Figure 1. Map of experimental sites in west-central Idaho. Restoration treatments for the northern Idaho ground squirrel (NIDGS, *Urocitellus brunneus*) occurred on the Council and New Meadows Districts of the Payette National Forest. Unfilled circles represent the occupied units that were not treated, while the filled circles represent the unoccupied units that were thinned and burned.

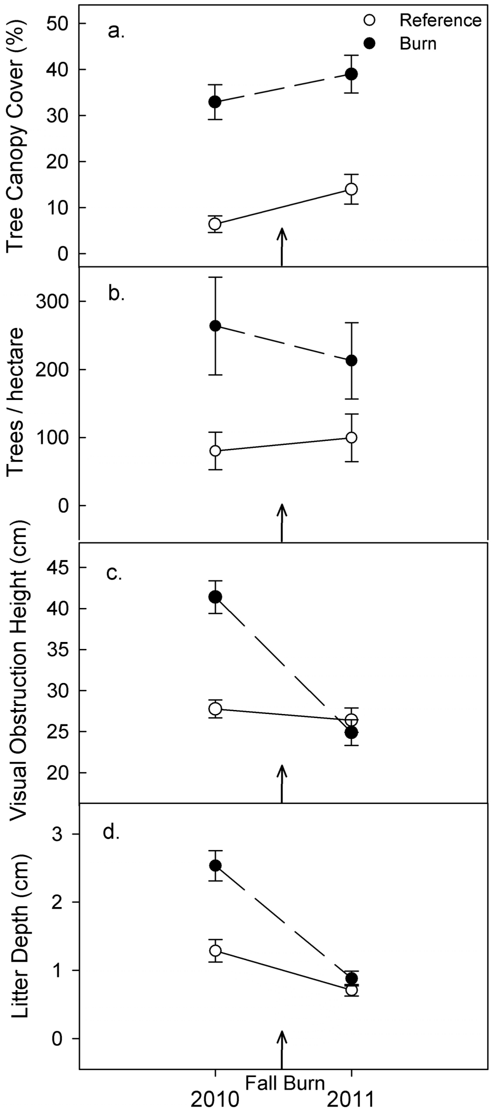

Figure 2. Tree canopy cover, tree density, understory vegetation height, and litter depth at reference (open circles) and burn (filled circles) units in 2010 and 2011. Error bars represent standard error. The arrow indicates when the burn plots were treated.

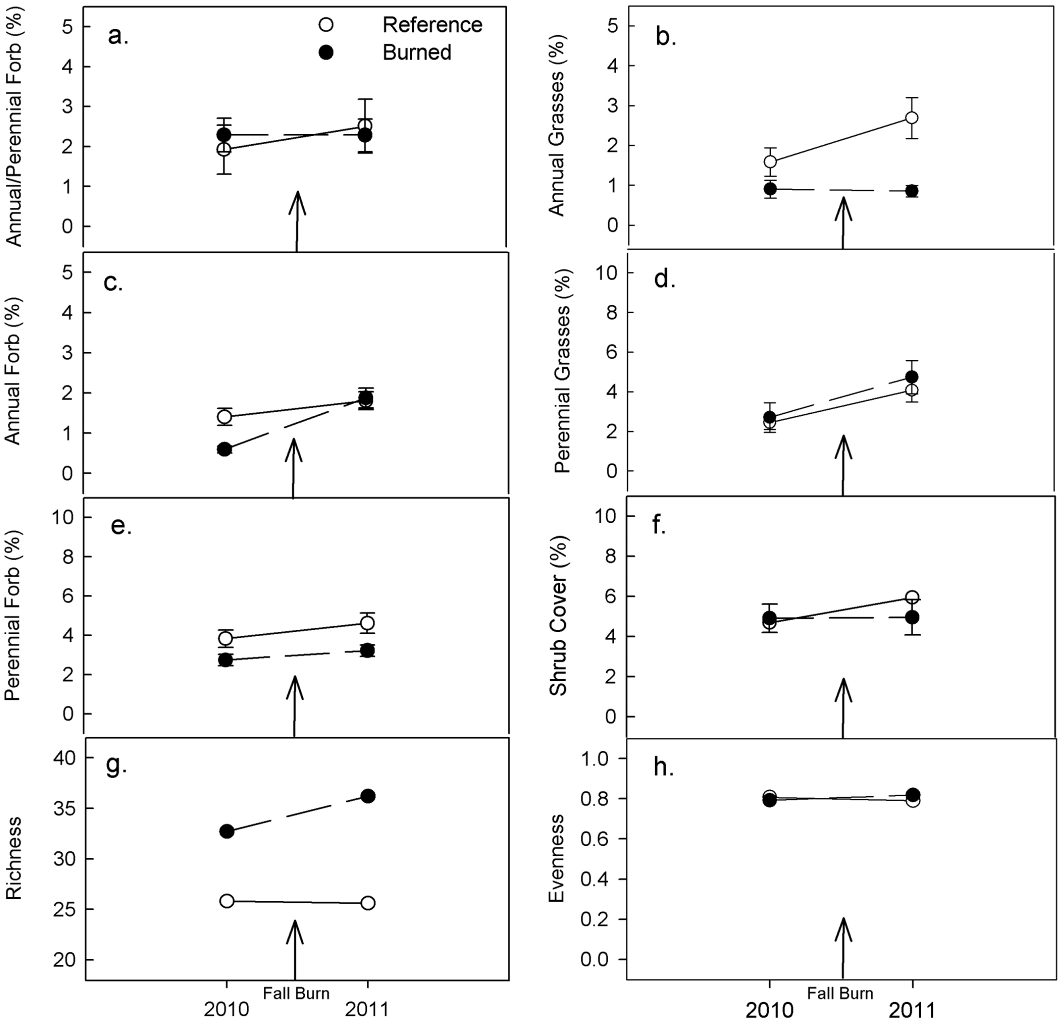

Figure 3. Understory community response between reference (open circles) and burn units (filled circles) in 2010 and 2011. Relative cover of annual forbs, perennial forbs, annual/perennial forbs, annual grasses, perennial grasses and shrubs (a-f), species richness (g), and species evenness (h). Error bars represent standard error. The arrow indicates when the burn plots were treated.

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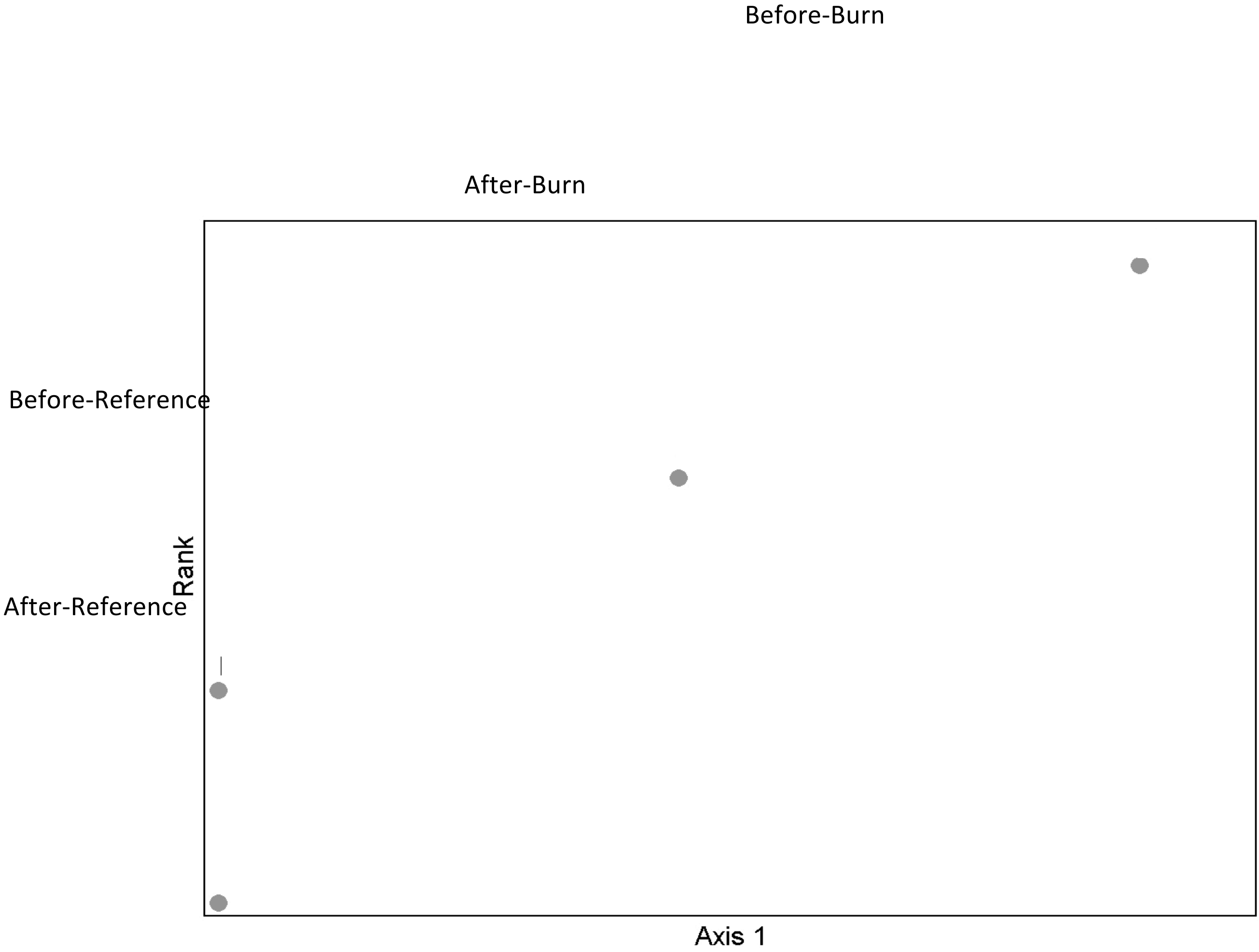

Figure 4. Non-parametric multi-dimensional scaling ordination of species composition for unit groups (Before-After, Reference-Burn). Dissimilarities between groups were captured by one axis ($r^2 = 0.81$).

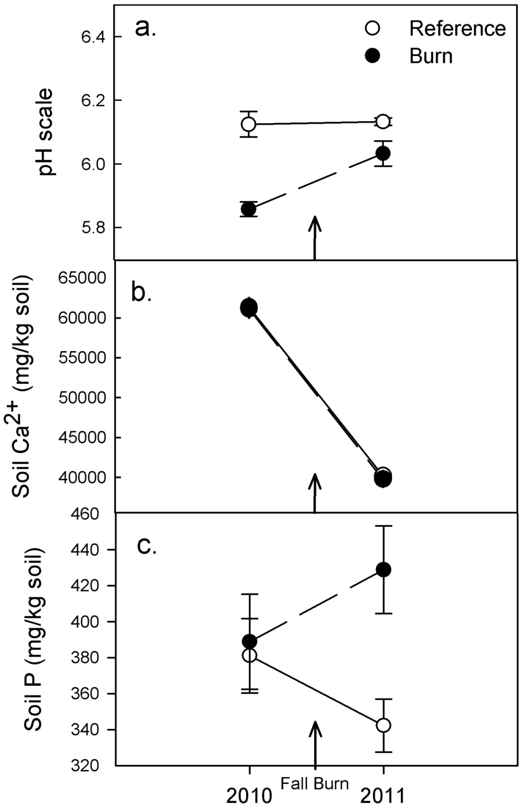

Figure 5. Soil parameters (pH, calcium and phosphorus) at reference (open circle) and burn units (filled circles) in 2010 and 2011. Error bars represent standard error. The arrow indicates when the burn plots were treated.

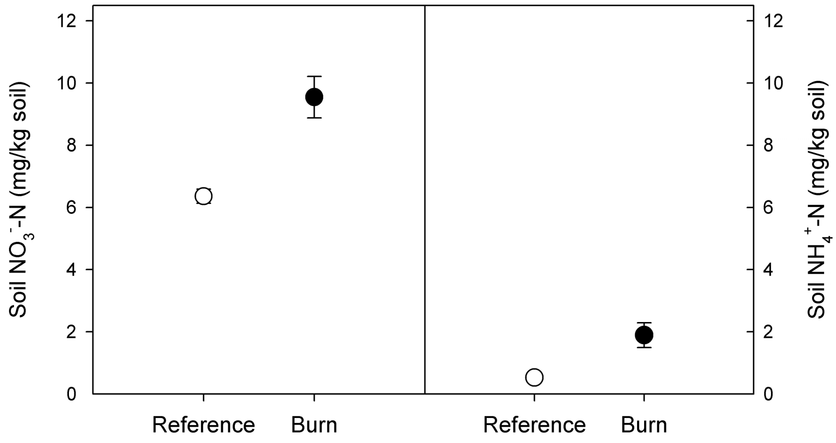

Figure 6. Soil nitrate and ammonium at reference (open circle) and burn (filled circle) units in 2011. Error bars represent standard error.

Appendix A. Understory species found within northern Idaho ground squirrels (*Urocitellus brunneus*) habitat. For each species, site cover averages and standard errors (SE) plus across site averages and SE are provided. Two trees species are included on this list due to their presence within a quadrat. Site abbreviations indicate: Cap Gun = Ca, Lost Valley = LR, Mud Creek = Mu, OX Ranch = OX, Price Valley = Pr, Slaughter Gulch = SG, and Summit Gulch = Su.

Species	Ca	LR	Mu	OX	Pr	SG	Su	Across Sites
Annual Forb								
Agoseris heterophylla var. heterophylla				0.8 ± 0.1	0.4 ± 0.1	0.3 ± 0.0	0.5 ± 0.2	0.6 ± 0.2
Amsinckia lycopsoides			1.8 ± 0.0					1.8 ± 0.0
Castilleja sp.	0.3 ± 0.0	0.3 ± 0.0						0.3 ± 0.0
Castilleja tenuis	0.9 ± 0.3			0.8 ± 0.1			0.3 ± 0.0	0.8 ± 0.2
Cirsium sp.		0.3 ± 0.0	1.3 ± 0.2					1.1 ± 0.3
Clarkia pulchella		0.9 ± 0.3	5.2 ± 1.0			2.1 ± 0.4		2.7 ± 1.0
Clarkia rhomboidea	5.3 ± 1.5				0.7 ± 0.1			1.8 ± 1.1
Collinsia parviflora	1.3 ± 0.3	2.5 ± 0.8	1.3 ± 0.2	1.5 ± 0.5	4.3 ± 0.7	0.8 ± 0.2	2.1 ± 0.5	1.9 ± 0.8
Collomia grandiflora					0.4 ± 0.2			0.4 ± 0.2
Collomia linearis	0.8 ± 0.2	0.4 ± 0.1	0.4 ± 0.0	1.2 ± 0.3	0.7 ± 0.1	0.6 ± 0.1	2.8 ± 0.5	1.1 ± 0.5
Cryptantha affinis		0.3 ± 0.0	0.3 ± 0.0	0.3 ± 0.0	0.9 ± 0.2	0.6 ± 0.1	0.3 ± 0.0	0.5 ± 0.2
Dianthus armeria			0.3 ± 0.0	1.9 ± 0.1	± 0.0			1.3 ± 0.4
Epilobium brachycarpu	2.0 ± 0.3	2.4 ± 0.7	4.5 ± 0.9	1.1 ± 0.5	2.1 ± 0.4	2.7 ± 0.6	4.5 ± 1.0	2.5 ± 0.9
Galium aparine	2.0 ± 0.6	0.3 ± 0.0		2.0 ± 0.5	4.6 ± 1.7	1.9 ± 0.4	0.9 ± 0.3	1.7 ± 0.8
Gayophytum diffusum	0.3 ± 0.0	0.7 ± 0.2	4.0 ± 0.9	0.8 ± 0.0	2.7 ± 1.1	0.7 ± 0.0		2.0 ± 1.1
Lepidium campestre			0.5 ± 0.0					0.5 ± 0.0
Lepidium graminifolium			0.5 ± 0.0					0.5 ± 0.0
Leptosiphon harknessii	0.5 ± 0.2	1.1 ± 0.2	5.2 ± 1.3	0.3 ± 0.0	3.5 ± 1.5	2.7 ± 0.6	0.7 ± 0.2	2.1 ± 1.1
Lotus unifoliolatus var. unifoliolatus	2.0 ± 0.0	0.3 ± 0.0	6.2 ± 1.2	0.9 ± 0.3	± 0.0	0.6 ± 0.1	0.8 ± 0.1	3.5 ± 1.5
Madia glomerata	1.3 ± 0.4	0.5 ± 0.1	1.0 ± 0.3		0.9 ± 0.3	0.5 ± 0.0	6.7 ± 2.0	1.7 ± 1.2
Madia gracilis	1.8 ± 0.0	0.3 ± 0.0	1.5 ± 0.3	1.3 ± 0.4	± 0.0	1.0 ± 0.5	2.6 ± 1.2	1.4 ± 0.7
Madia sp.	1.4 ± 0.7	0.4 ± 0.1	1.0 ± 0.3	1.0 ± 0.2	0.9 ± 0.3	1.5 ± 0.5	6.1 ± 1.4	1.9 ± 1.1
Microsteris gracilis	0.7 ± 0.1	0.8 ± 0.0	0.3 ± 0.0	2.0 ± 0.5	0.5 ± 0.0	0.3 ± 0.0	1.9 ± 0.5	1.4 ± 0.5
Mimulus breweri		0.3 ± 0.0				0.3 ± 0.0		0.3 ± 0.0
Myosotis micrantha				0.9 ± 0.3			0.3 ± 0.0	0.5 ± 0.2
Navarettia sp.	0.7 ± 0.2	1.1 ± 0.4	0.7 ± 0.2	1.1 ± 0.3	0.3 ± 0.0	0.9 ± 0.3	1.3 ± 0.4	1.0 ± 0.4
Phacelia franklinii		0.3 ± 0.0						0.3 ± 0.0
Plagiobothrys leptocladus var.		1.5 ± 0.0		0.3 ± 0.0			1.5 ± 0.0	1.0 ± 0.3
Polygonum douglasii	1.4 ± 0.4	0.5 ± 0.1	2.2 ± 0.4	0.6 ± 0.1	3.1 ± 0.7	1.9 ± 0.4	2.9 ± 0.5	1.8 ± 0.6
Polygonum polygaloides		1.6 ± 0.4	2.9 ± 1.3	1.5 ± 0.3	0.3 ± 0.0	1.0 ± 0.3	0.3 ± 0.0	1.4 ± 0.7
Tragopogon dubius	0.3 ± 0.0			0.7 ± 0.2			0.3 ± 0.0	0.5 ± 0.2
Forb Perennial								
Achillea millefolium	8.7 ± 1.2	3.5 ± 0.6	7.4 ± 1.5	6.1 ± 1.3	4.5 ± 1.3	3.5 ± 1.2	3.5 ± 0.8	5.4 ± 1.6
Agoseris glauca	0.4 ± 0.1	0.5 ± 0.1	± 0.0	0.5 ± 0.1	0.5 ± 0.1	0.3 ± 0.0	0.5 ± 0.0	0.4 ± 0.1
Allium sp.	0.9 ± 0.2	1.8 ± 0.4	0.3 ± 0.0	3.8 ± 0.9	5.3 ± 1.2	2.3 ± 0.5	1.1 ± 0.2	2.8 ± 1.1
Anaphalis margaritacea	5.1 ± 1.2	3.6 ± 0.7	4.5 ± 1.4	1.6 ± 0.4	2.6 ± 0.5	7.6 ± 1.4	6.5 ± 1.4	5.1 ± 1.6
Antennaria rosea	7.6 ± 1.6	1.8 ± 0.0						5.7 ± 1.9

Species	**Ca**	**LR**	**Mu**	**OX**	**Pr**	**SG**	**Su**	**Across Sites**
Arenaria congesta		1.0 ± 0.2				0.4 ± 0.0		0.6 ± 0.2
Arnica cordifolia	0.7 ± 0.2	1.2 ± 0.5	2.0 ± 0.0					1.1 ± 0.4
Arnica sororia	2.7 ± 0.5	6.3 ± 0.0	0.3 ± 0.0	0.9 ± 0.3			1.8 ± 0.6	1.9 ± 0.7
Aster alpigenus	1.3 ± 0.3	1.5 ± 0.0	2.0 ± 0.7	1.3 ± 0.5	1.8 ± 0.3	1.5 ± 0.0	2.4 ± 0.5	1.7 ± 0.5
Balsamorhiza sagittata	3.8 ± 0.6			10.7 ± 2.3	8.1 ± 2.4			7.6 ± 2.7
Barbarea orthoceras			1.5 ± 0.0					1.5 ± 0.0
Besseya rubra	0.3 ± 0.0			0.9 ± 0.3		1.5 ± 0.0		0.7 ± 0.2
Calochortus elegans		0.3 ± 0.0						0.3 ± 0.0
Calochortus eurycarpus	0.5 ± 0.1	0.3 ± 0.0		1.4 ± 0.5	0.6 ± 0.1	0.6 ± 0.1	0.4 ± 0.0	0.7 ± 0.4
Camassia quamash ssp. breviflora		1.5 ± 0.0		1.8 ± 0.8				1.7 ± 0.9
Camissonia subacaulis		0.7 ± 0.2				0.5 ± 0.1	0.3 ± 0.0	0.5 ± 0.2
Castilleja cusickii				0.3 ± 0.0				0.3 ± 0.0
Chamerion angustifolium ssp. circumvagum		5.1 ± 1.1				0.3 ± 0.0		4.5 ± 1.4
Chrysothamnus viscidiflorus		1.5 ± 0.0		3.0 ± 0.0				2.3 ± 0.4
Clematis hirsutissima var. hirsutissima	0.9 ± 0.2	1.3 ± 0.4	0.3 ± 0.0		0.3 ± 0.0	1.5 ± 0.0	3.1 ± 0.4	1.6 ± 0.6
Crepis acuminata ssp. acuminata	0.3 ± 0.0			1.0 ± 0.3	1.0 ± 0.2	1.5 ± 0.0	0.3 ± 0.0	0.8 ± 0.2
Cynoglossum officinale			0.8 ± 0.2	± 0.0				0.8 ± 0.3
Delphinium sp.				0.4 ± 0.1	0.3 ± 0.0		0.4 ± 0.1	0.4 ± 0.1
Dichelostemma congestum	0.3 ± 0.0				0.3 ± 0.0			0.3 ± 0.0
Equisetum sp.			0.8 ± 0.0					0.8 ± 0.0
Erigeron pumilus ssp. intermedius var. intermedius	0.3 ± 0.0			2.4 ± 0.6				2.2 ± 0.8
Erigeron speciosus			1.0 ± 0.2		1.9 ± 0.6			1.7 ± 0.7
Eriogonum sp.	14.4 ± 1.0	9.8 ± 2.8			4.3 ± 0.4	4.6 ± 1.1	19.0 ± 2.3	14.7 ± 4.3
Eriogonum douglasii		10.6 ± 1.6		7.8 ± 1.5		16.4 ± 1.9	23.6 ± 2.7	15.8 ± 3.4
Eriogonum heracleoides	7.9 ± 0.8		3.9 ± 0.7	7.4 ± 1.4	2.1 ± 0.4	0.0 ± 0.0	29.0 ± 0.0	6.4 ± 2.4
Eriophyllum lanatum		0.5 ± 0.2			0.3 ± 0.0	0.3 ± 0.0		0.4 ± 0.2
Fragaria vesca ssp. bracteata					0.3 ± 0.0			0.3 ± 0.0
Fragaria virginiana	3.3 ± 1.1	6.0 ± 1.3	2.1 ± 0.5	1.1 ± 0.2	1.5 ± 0.0	3.8 ± 1.2	2.8 ± 0.7	3.4 ± 1.3
Frasera albicaulis	6.3 ± 1.8	3.0 ± 0.5		2.4 ± 0.6	4.7 ± 1.1	3.9 ± 0.8	1.5 ± 0.0	4.1 ± 1.4
Geum triflorum var. ciliatum	1.1 ± 0.2						4.9 ± 0.9	3.0 ± 1.1
Hieracium scouleri		6.0 ± 1.9			0.3 ± 0.0	3.8 ± 0.0		4.2 ± 2.1
Hydrophyllum capitatum var. capitatum	0.3 ± 0.0	0.3 ± 0.0				0.4 ± 0.1		0.4 ± 0.1
Ipomopsis aggregata ssp. aggregata	0.8 ± 0.3	0.9 ± 0.2	1.9 ± 0.4	2.1 ± 0.3	0.3 ± 0.0	1.0 ± 0.3	1.4 ± 0.2	1.3 ± 0.4
Lathyrus lancifolia					0.3 ± 0.0			0.3 ± 0.0
Lathyrus sp.				0.9 ± 0.2	0.4 ± 0.1		0.3 ± 0.0	0.5 ± 0.1
Lepidium latifolium				0.7 ± 0.2				0.7 ± 0.2

Species	Ca	LR	Mu	OX	Pr	SG	Su	Across Sites
Leucanthemum vulgare			0.3 ± 0.0					0.3 ± 0.0
Linaria vulgaris			1.6 ± 0.6					1.6 ± 0.7
Lithophragma glabrum	0.3 ± 0.0	0.3 ± 0.0			0.3 ± 0.0	0.3 ± 0.0	0.3 ± 0.0	0.3 ± 0.0
Lithospermum ruderale				1.5 ± 0.0				1.5 ± 0.0
Lomatium nudicaule	1.4 ± 0.3			2.2 ± 0.7			0.8 ± 0.3	1.7 ± 0.7
Lomatium sp.		6.4 ± 1.3	1.5 ± 0.0	1.5 ± 0.6	25.9 ± 3.5		0.3 ± 0.0	10.8 ± 5.0
Lupinus sp.	3.5 ± 0.6	5.8 ± 1.7	1.3 ± 0.2	8.4 ± 1.8	1.1 ± 0.2	5.7 ± 1.9	4.0 ± 0.7	5.0 ± 1.8
Mertensia longiflora						3.8 ± 0.0		3.8 ± 0.0
Olsynium douglasii	0.3 ± 0.0	0.5 ± 0.1		1.5 ± 0.0	0.6 ± 0.0	0.6 ± 0.2		0.6 ± 0.1
Paeonia brownii	3.4 ± 0.9	4.0 ± 0.5	3.8 ± 0.0		0.5 ± 0.0	1.1 ± 0.2		2.8 ± 0.8
Penstemon gairdneri var. oreganus	1.9 ± 0.5	3.8 ± 0.0						2.4 ± 0.6
Penstemon globosus			8.0 ± 0.1		3.0 ± 0.0		5.3 ± 1.6	5.8 ± 1.7
Penstemon sp.	0.8 ± 0.3	1.0 ± 0.2		0.3 ± 0.0	0.8 ± 0.2	1.3 ± 0.4	0.3 ± 0.0	0.9 ± 0.4
Perideridia gairdneri ssp. borealis	0.1 ± 0.1	0.3 ± 0.0		2.0 ± 0.3		1.2 ± 0.2	0.6 ± 0.1	1.0 ± 0.4
Phacelia sp.		0.8 ± 0.3			0.3 ± 0.0	0.6 ± 0.2		0.5 ± 0.2
Potentilla glandulosa	4.2 ± 1.2	3.4 ± 0.7				3.8 ± 0.8	1.2 ± 0.5	3.3 ± 1.1
Potentilla gracilis	5.8 ± 1.5	9.1 ± 2.2	7.1 ± 1.4	2.5 ± 0.4		2.6 ± 0.6	7.4 ± 0.9	5.8 ± 1.9
Potentilla recta	0.5 ± 0.0						2.3 ± 0.8	2.0 ± 1.0
Pyrrocoma carthamoides	1.8 ± 0.0	1.5 ± 0.0			1.8 ± 0.0	1.8 ± 0.4		1.7 ± 0.4
Rumex acetosella	0.3 ± 0.0	1.1 ± 0.2	5.0 ± 1.5		1.0 ± 0.3			3.3 ± 1.6
Saxifraga integrifolia		0.3 ± 0.0		0.3 ± 0.0				0.3 ± 0.0
Scutellaria angustifolia	0.6 ± 0.2	1.0 ± 0.3			0.4 ± 0.1	0.3 ± 0.0		0.6 ± 0.2
Sedum stenopetalum	6.2 ± 1.5	3.1 ± 0.7		1.3 ± 0.2	7.0 ± 1.8	6.7 ± 1.0	1.3 ± 0.1	4.8 ± 1.7
Senecio sp.	1.3 ± 0.4	1.8 ± 0.3		7.1 ± 1.5	1.3 ± 0.4	2.6 ± 0.5	0.9 ± 0.3	2.7 ± 1.2
Sidalcea oregana ssp. oregana	0.9 ± 0.3	2.3 ± 0.4	0.4 ± 0.1	3.3 ± 1.2	1.0 ± 0.2	3.0 ± 0.6	2.6 ± 0.5	2.2 ± 0.8
Solidago missouriensis	5.3 ± 1.0			3.0 ± 1.0				4.1 ± 1.3
Trifolium pratense	0.5 ± 0.0							0.5 ± 0.0
Triteleia grandiflora var. grandiflora						0.3 ± 0.0		0.3 ± 0.0
Verbascum thapsus		1.0 ± 0.0	0.5 ± 0.2					0.6 ± 0.2
Viola nuttallii		0.8 ± 0.0	0.3 ± 0.0	0.4 ± 0.1		0.3 ± 0.0	2.2 ± 0.3	0.9 ± 0.4
Viola sp.	0.6 ± 0.1	1.1 ± 0.2	0.7 ± 0.2	1.8 ± 0.5	0.6 ± 0.2	1.1 ± 0.2	1.8 ± 0.4	1.1 ± 0.4
Wyethia helianthoides	3.3 ± 0.6			8.4 ± 3.1			19.3 ± 3.7	12.9 ± 4.7
Zigadenus venenosus	0.3 ± 0.0	0.3 ± 0.0				0.5 ± 0.0	0.3 ± 0.0	0.3 ± 0.0
Forb Annual/Perennial								
Claytonia perfoliata ssp. perfoliata	± 0.0		0.3 ± 0.0	0.3 ± 0.0			2.5 ± 0.0	0.8 ± 0.4
Geranium viscosissimum var. viscosissimum	2.7 ± 0.9	3.5 ± 0.5	2.3 ± 0.8				2.2 ± 0.4	2.6 ± 0.9
Ranunculus uncinatus		0.3 ± 0.0		1.5 ± 0.0				0.9 ± 0.3
Sanguisorba annua			0.9 ± 0.3	0.3 ± 0.0		0.5 ± 0.0	1.8 ± 0.4	1.1 ± 0.4
Taraxacum sp.	1.3 ± 0.5	1.5 ± 0.0	1.8 ± 0.6	2.1 ± 0.2			0.3 ± 0.0	1.4 ± 0.5
Trifolium sp.		± 0.0	12.4 ± 2.5	0.5 ± 0.0				8.4 ± 3.5

Species	Ca	LR	Mu	OX	Pr	SG	Su	Across Sites
Grass Annual								
Bromus arvensis	2.1 ± 0.8		2.4 ± 0.6	0.4 ± 0.1	2.0 ± 0.3	0.9 ± 0.3	2.5 ± 0.6	1.8 ± 0.7
Bromus briziformis	0.3 ± 0.0			3.0 ± 0.0				1.6 ± 0.7
Bromus tectorum	0.3 ± 0.0		3.9 ± 1.2	4.5 ± 1.0	1.0 ± 0.4	0.3 ± 0.0	2.8 ± 0.8	2.8 ± 1.2
Ventenata dubia		0.3 ± 0.0		1.5 ± 0.0				1.1 ± 0.3
Grass Perennial								
Achnatherum sp.	2.4 ± 0.4	1.6 ± 0.2	6.3 ± 1.3	0.0 ± 0.0	0.5 ± 0.1	9.9 ± 0.7	2.7 ± 0.4	3.7 ± 1.9
Bromus inermis			1.6 ± 0.1					1.6 ± 0.1
Bromus marginatus	2.2 ± 0.4	2.0 ± 0.7	0.3 ± 0.0	0.9 ± 0.2	3.2 ± 0.8	0.9 ± 0.2	4.1 ± 1.0	1.8 ± 0.7
Calamagrostis	3.5 ± 0.8	12.5 ± 1.7				3.8 ± 0.0	0.0 ± 0.0	7.8 ± 2.7
Danthonia unispicata	2.8 ± 0.4		0.3 ± 0.0	5.6 ± 1.1	1.9 ± 0.8	3.8 ± 0.0		3.9 ± 1.3
Elymus elymoides	0.9 ± 0.2			1.2 ± 0.3		2.6 ± 0.5	0.9 ± 0.2	1.1 ± 0.4
Festuca idahoensis	3.8 ± 0.8	6.0 ± 1.1	0.3 ± 0.0	2.1 ± 0.2	3.5 ± 0.7	3.4 ± 0.2		3.6 ± 1.0
Festuca sp.	2.6 ± 0.5	5.2 ± 1.3		2.0 ± 0.6	3.9 ± 0.7	4.3 ± 0.8		3.5 ± 1.2
Koeleria macrantha	2.1 ± 0.2	2.0 ± 0.4	1.5 ± 0.0	2.1 ± 0.6	3.0 ± 0.0	3.1 ± 0.7	2.1 ± 0.5	2.3 ± 0.7
Melica bulbosa				1.6 ± 0.1	0.3 ± 0.0	1.0 ± 0.3		1.0 ± 0.3
Melica bulbosa var. intonsa	± 0.0	0.5 ± 0.0				0.3 ± 0.0		0.3 ± 0.1
Pascopyrum smithii	1.0 ± 0.3				0.8 ± 0.3			0.9 ± 0.3
Poa bulbosa	1.9 ± 0.6	0.5 ± 0.0	2.0 ± 0.5	10.2 ± 2.7	4.0 ± 1.0		2.2 ± 0.5	4.5 ± 2.4
Poa pratensis	15.5 ± 3.6	0.3 ± 0.0	5.2 ± 0.8	3.5 ± 1.1	1.0 ± 0.3	0.9 ± 0.2	4.5 ± 1.3	5.1 ± 2.5
Poa secunda	3.5 ± 0.6	2.1 ± 0.4		4.8 ± 0.9	1.8 ± 0.4	0.9 ± 0.2	3.0 ± 0.8	2.9 ± 1.0
Pseudoroegneria spicata	1.4 ± 0.5	11.0 ± 1.7			11.1 ± 2.7	8.3 ± 1.2		9.3 ± 2.9
Grass Annual/Perennial								
Bromus sp.	0.9 ± 0.2	1.0 ± 0.2	8.5 ± 0.0	0.2 ± 0.0	0.7 ± 0.1	0.8 ± 0.3	2.1 ± 0.8	1.2 ± 0.7
Graminoid	5.0 ± 1.2	2.8 ± 0.6	27.1 ± 8.7	6.0 ± 1.0	3.8 ± 0.0	4.2 ± 1.3	1.5 ± 0.0	10.1 ± 6.9
Poa sp.		1.5 ± 0.0	± 0.0	4.1 ± 0.8		2.3 ± 0.3	1.3 ± 0.4	2.7 ± 0.8
Rush Perennial								
Juncus parryi				1.5 ± 0.0				1.5 ± 0.0
Juncus sp.	4.3 ± 0.2			5.3 ± 0.8		3.8 ± 0.0	6.3 ± 0.0	5.0 ± 0.8
Sedge Perennial								
Carex geyeri	9.3 ± 1.5	6.8 ± 1.7	1.5 ± 0.0	1.5 ± 0.0	5.5 ± 1.6	1.8 ± 0.5	1.5 ± 0.0	5.8 ± 2.0
Carex hoodii	2.6 ± 0.5	0.3 ± 0.0				1.5 ± 0.0	2.0 ± 0.7	1.8 ± 0.6
Carex pachystachya							3.8 ± 0.0	3.8 ± 0.0
Carex petasata	1.5 ± 0.0			1.5 ± 0.0				1.5 ± 0.0
Shrub Perennial								
Amelanchier alnifolia	5.0 ± 2.5	4.5 ± 0.0						4.9 ± 2.8
Artemisia sp.		5.3 ± 0.0	3.7 ± 1.0	7.1 ± 2.1		3.8 ± 0.0	2.6 ± 0.5	5.5 ± 2.2
Artemisia tridentata ssp. tridentata	± 0.0					6.9 ± 1.6		6.9 ± 2.1
Ceanothus velutinus	9.8 ± 0.0							9.8 ± 0.0
Purshia tridentata	5.5 ± 1.1	3.8 ± 0.0		5.5 ± 1.4	3.8 ± 0.0	2.3 ± 1.0		4.7 ± 1.5
Ribes cereum		6.3 ± 0.0						6.3 ± 0.0
Symphoricarpos albus	4.6 ± 0.9	4.4 ± 0.7	1.3 ± 0.2	3.3 ± 1.1	12.8 ± 3.3		2.8 ± 0.7	4.3 ± 1.8
Tree Perennial								
Pinus contorta						6.1 ± 1.0		6.1 ± 1.3
Pinus ponderosa	9.8 ± 0.0	0.0	1.5 ± 0.0	1.5 ± 0.0		9.8 ± 0.0		5.6 ± 1.8

*Understory species data and reference collection are archived at the University of Idaho.