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SOIL QUALITY INDICATORS AND VEGETATION RESPONSES FOLLOWING ECOLOGICAL RESTORATION THINNING OF PONDEROSA PINE ON THREE SOIL PARENT MATERIAL TYPES UNDER GRAZING AND NON-GRAZING IN NORTHERN ARIZONA

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By

CHRISTOPHER D. MACDONALD, Master of Science in Forestry

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ABSTRACT

This research was performed on three soil parent material types in a northern Arizona ponderosa pine (*Pinus ponderosa* P. & C., Lawson) forest. The objectives were to a) increase understanding of long-term vegetation responses to ecological restoration treatments on three soils types, with and without grazing, b) evaluate the responses of a suite of select soil physical, chemical, and biological properties to forest ecological restoration treatments, with and without grazing, c) determine the applicability of measured chemical, physical, and biological soil quality indicators to informing risks to sustainable soils resources management in southwestern ponderosa pine forests.

There were no significant differences in soil bulk density by soil type, treatment type, or grazing treatment. No significant differences in saturated infiltration capacity were observed among treatment types under either grazed or non-grazed conditions. Saturated infiltration capacity was affected by soil type under non-grazed conditions with soils derived from basalt and benmoreite having significantly lower saturated infiltration capacity than limestone-derived soils. Grazing resulted in lower soil aggregate stability in comparison to non-grazing. Soil slaking was affected by grazing and forest treatment type. Grazing on basalt-derived soils was less negative to soil slaking potential than limestone-derived soils ($p = 0.008$). All soil slake tests indicated a strong potential for soils to readily slake when rapidly wetted, regardless of forest or grazing treatment type.

Average soil slaking rates rarely exceed a rating of 3, which represents the midpoint of aggregate water dispersion potential.

There were no significant differences in levels of soil organic matter (OM), organic carbon (C), nitrogen (N), calcium (Ca^{2+}), or potassium (K^{+}) (Mg ha^{-1}) in the upper 10 cm of soil based on soil type, forest treatment type, or grazing treatment. There were however statistically significant differences in soil magnesium (Mg^{2+}) content (Mg ha^{-1}) by soil type, with basalt-derived soils having significantly higher mean Mg^{2+} content. This finding was to be expected since eruptive products of the San Francisco Volcanic Field, while of varying petrogenesis, are dominantly basaltic and rhyolitic, which typically have high percentages of Mg^{2+} and Fe^{2+}. Soil sodium (Na) levels (Mg ha^{-1}) were not significantly different by soil type or forest treatment type.

Grazing affected soil Na^{+} levels, with grazing having lower Na^{+} than non-grazing across all soil types and forest treatments. This finding was perplexing since grazing has generally been shown to increase localized soil Na^{+} and Mg^{2+} concentrations since livestock urine and feces can have high concentrations of these elements particularly when mineral supplements are provided to livestock, as is common on this allotment.

Basalt-derived soils had significantly lower soil phosphorus (P) levels than benmoreite-derived soils. Soil sulfur (S) levels (Mg ha^{-1}) were significantly different among soil types with limestone-derived soils having significantly lower soil S than basalt- and benmoreite-derived soils.

Vegetative cover and species richness were affected by forest treatment type, soil parent material type, grazing treatment, and measurement year, depending on life form, longevity, and nativity. As observed by Abella et al. (2015), responses were hierarchically controlled, with soil parent material type often moderating vegetative responses. Limestone-derived soils had the greatest overall total plant cover and benmoreite-derived soils often had the lowest. Grazing affected vegetative cover, depending on measurement year, but rarely affected species richness.

While many of the soil quality indicators were not significant, this study is not without merit. Treatments for this research were implemented using low impact techniques (hand falling using chainsaws and manual removal of slash). Similar studies at operational scales using heavy logging machinery followed by prescribed fire would likely contribute further to the body of knowledge on the utility of the soil quality indicators evaluated in this study for informing sustainable soils management thresholds.

This study highlights the importance of considering both inherent and dynamic soil properties when planning ecological restoration treatments since soil parent material type strongly influences vegetative responses of cover and richness, particularly in relation to livestock and wildlife forage production and sustaining healthy native plant communities following forest restoration treatments. Finally, understanding exotic plant species abundance and richness before forest ecological restoration treatment may prevent undesirable effects to native plant communities by preventing invasions of exotic plants.

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INTRODUCTION

Southwestern ponderosa pine (*Pinus ponderosa* Douglas ex P. Lawson & Lawson) forests in Northern Arizona have experienced significant changes in structure and vegetal composition since Euro-American settlement in the mid- to late-1800s (Cooper 1960; Moore et al., 2004). These forests historically consisted of widely spaced trees with well-developed herbaceous understories which were maintained by frequent, low intensity, naturally ignited wildfires. Over the last 100 years, these forests have developed into much denser stands of ponderosa pine with increasing numbers of fire-sensitive and disease-susceptible species (i.e., *Abies* and *Pseudotsuga* sp.) and progressively decreasing herbaceous understory productivity and diversity. These changes are largely attributed to 1) elimination of the frequent, low-intensity fire regime through active fire suppression, and 2) overgrazing by livestock, which altered forest dynamics by (a) reducing understory density of grasses, forbs, and shrubs which otherwise outcompete tree seedlings, thereby reducing conifer recruitment density, and (b) reducing the amount of fine fuels necessary to carry low-intensity fires across the forest floor.

Ecological restoration treatments are increasingly applied in southwestern ponderosa pine forests to return forest structure, function, and vigor to conditions similar to those found at Euro-American settlement (Covington et al., 1997; Moore et al., 2006). Structural manipulation (forest thinning) and fire reintroduction are the most common ecological restoration techniques applied. While there are numerous studies of the responses of vegetation and soils to ponderosa pine forest restoration treatments and the effects of domestic livestock grazing (Abella et al., 2015; Bakker and Moore, 2007; Moore et al., 2006; Feeney et al., 1998; Skov et al., 2004; Covington et al., 1997; and Harris and Covington, 1983), few studies have examined longer term (i.e., greater than 10 years) vegetative responses to ponderosa pine restoration thinning and the interaction with livestock grazing across different soil types where soil parent materials can affect both above- and below-ground ecological responses. Additionally, there are no known studies of the long-term effects of ponderosa pine ecological restoration treatments and livestock grazing on a suite of soil quality indicators to evaluate the combined effects of both on soil ecosystems.

Abella et al. (2015) studied the response of understory plant communities to ecological restoration treatments in a ponderosa pine bunchgrass ecosystem on soils derived from three different parent materials (basalt, benmoreite, and limestone) in northern Arizona. Ecological restoration treatments included thinning from below, thinning from below plus an aqueous smoke simulation treatment, closed canopy control, and open canopy. The effects of domestic livestock and wildlife ungulate grazing were also evaluated through installation of exclosures that were paired with non-exclosed sites. Pre-treatment analysis of plant community compositions was completed in 2003 and post-treatment analyses were completed in 2006 and 2008.

Using the original study established by Abella et al. in 2003 (Abella et al., 2015), the 12-year post-treatment responses of soils and vegetative communities to forest restoration thinning under grazing and non-grazing were evaluated. Remeasurement of the plant communities included total plant cover and composition (per $1m^2$) and species richness (per $1m^2$ and per $9m^2$) in thinned and non-thinned plots that were grazed or excluded from grazing. Differences in select physical soil quality indicators were evaluated, including bulk density, infiltration, aggregate stability, and slaking potential. In combination with these physical soil quality indicators, measured chemical and biological soil quality indicators (pH, total nitrogen and carbon, extractable phosphorous, potassium, calcium, magnesium, sodium, and sulfur) as reported in the results of Elena Thomas's M.S. thesis (Thomas 2017) were evaluated. The purpose was to understand the effects of forest thinning and grazing treatments to soil quality indicators that can be readily measured on similar soils in southwestern ponderosa pine forests to inform sustainable management of soils during future forest restoration treatments and grazing management practices. Finally, the effects of forest restoration thinning and livestock grazing to plant cover and species richness were analyzed.

OBJECTIVES

The objectives of this research were to: a) evaluate the responses of select soil physical, chemical, and biological properties of three soils derived from different parent materials (basalt, benmoreite, and limestone) to forest restoration treatments under grazed and non-grazed conditions, b) increase understanding of long-term plant vegetative responses to ecological restoration thinning treatments, and c) determine the applicability of measured physical, chemical, and biological soil quality indicators to identifying and evaluating changes to soil quality resulting from management actions that can serve as indicators of risks to sustainable soils resources management in southwestern ponderosa pine forests.

LITERATURE REVIEW

Ecological Restoration Purpose and Approach

Ecological restoration is the ‘process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed’ (SER, 2004). In response to concerns regarding forest ecosystem health and uncharacteristic wildfire behavior in the southwestern United States, land managers and the scientific community have been implementing ecological restoration treatments to restore forest health in degraded ecosystems. In southwestern ponderosa pine forests, ecological restoration treatments such as forest thinning and prescribed fire have been used to convert dense, closed-canopy ponderosa pine stands back to more open stand structures with improved spatial and structural heterogeneity and greater understory diversity. The primary goal of these treatments is to manipulate ecosystem structure to more closely resemble the historic range of variability found in pre-Euro-American settlement ponderosa pine forests. A secondary goal is to decrease the risk of active crown fires at stand and landscape scales through reduction of fuel continuity (Allen et al., 2002).

Understanding of Reference Conditions

Though there is limited historical information regarding understory vegetation in ponderosa pine forests of the Southwest prior to Euro-American settlement, ecologists have made significant advancement in understanding the ecosystems of the Southwest and processes that led to reference conditions. Written historical accounts by explorers such as Vernon Bailey, James Simpson, and E.F. Beale indicate that open and park-like mature ponderosa pine forests with understories composed of abundant grasses and forbs were the norm in the Southwest (Allen, 2002; Pynes, 2000; Beale, 1858).

Dendrochronological studies have advanced understanding of stand densities in presettlement ponderosa pine forests (Covington and Moore, 1994; Fulé et al., 1997; Madany et al., 1982; Baisan and Swetnam, 1990; Abella et al., 2011). Relict stands that are geographically isolated and therefore have not been subjected to logging, grazing, or fire suppression have provided opportunities for comparison of essentially unaltered forests to forests that exhibit anthropogenic change (Fulé et al., 2002).

Long-term forest research plots, known as the ‘Woolsey plots” were established in Arizona and New Mexico during the early twentieth century. These plots were established to study natural regeneration of ponderosa pine in a variety of understory microhabitats that were protected from livestock grazing and fire. Vegetation changes within these plots indicate that stand density and crown cover has increased substantially while understory species richness has declined (Pearson, 1942).

Vegetation Dynamics

Overstory Vegetation Functions and Dynamics

Structure and function of southwestern ponderosa pine forests have been greatly altered since Euro-American settlement. Heavy domesticated livestock grazing, effective fire suppression and exclusion, historic logging practices, a large population of wildlife ungulates (elk), and changing climate patterns have increased the density of younger trees, reduced the number of larger trees, increased forest litter production, and reduced understory herbaceous productivity (Cooper, 1960; Covington and Moore, 1994; Swetnam and Baisans, 1996; Mast et al.,1999; Reynolds et al., 2013). The results include increased susceptibility to large, high severity wildfire; plant communities that are departed from reference conditions; changes to wildlife demographics; loss of biodiversity; and reduced forest and rangeland productivity through impaired ecosystem processes (Reynolds et al., 2013).

Prior to Euro-American settlement, the spatial pattern of most southwestern ponderosa pine forests was strongly influenced by natural environmental factors (e.g., climate, topography, soils, and fire regimes). Native Americans may have also had a role in controlling forest spatial and structural heterogeneity through fire use. The result was forest conditions that were more open than today, with small patches of higher tree densities. Stands often consisted of uneven-aged groups of large trees intermingled with dominantly herbaceous gaps (Beal, 1858; Pearson, 1923; Cooper, 1960; Covington and Moore, 1994; Mast et al., 1999). Where tree aggregation was common, openness was greater, but on areas where less aggregation occurred, openness may have been lower. Spatial arrangement of trees and openings is largely dependent on their sizes and crown widths (Reynolds et al., 2013).

Forest management concerns that have arisen as a result of fire exclusion and other anthropogenic influences include a) increased density of saplings and pole sized trees, b) reduced tree growth rates and increased mortality, particularly in older trees, c) decreased decomposition rates and departed soil nutrient cycles, particularly as influenced by droughts, d) increased fuel loading, e) increased insect infestations and disease outbreaks, e) wildfires of greater areal extent and higher burn severity, and f) reduced understory diversity and productivity (Harrington and Sackett, 1992; Covington and Moore, 1994; Moore et al., 2004; Morehouse et al., 2008; Fulé et al., 2014; Haffey et al., 2018).

Cooper (1960) noted that travelers through northern Arizona forests during the late 19th century recorded conditions of open, park-like stands of ponderosa pine and fir with robust and diverse herbaceous understories which became dominated by ponderosa pine saplings by 1960. Ives (1861) described the foothills at the base of Bill Williams Mountain: “This morning we re-entered the region of pines and have travelled all day in the midst of picturesque and charming scenery. The valleys are covered with a bright green sward, and open groves are dispersed gracefully upon the lowlands and ridges.” In Physical Geology of the Grand Canyon, Dutton (1882) described the Kaibab Plateau: “The trees are large and noble in aspect and stand widely apart, except in the highest part of the plateau where spruces predominate. Instead of dense thickets where we are shut in by impenetrable foliage, we can look far beyond and see the tree trunks vanishing away like an infinite colonnade. The ground is unobstructed and inviting. There is a constant succession of parks and glades".

While most early accounts of conditions in southwestern ponderosa pine forests describe open, park-like stands of large trees, some observers did note areas of young, dense stands. Rixon (1905) observed "phenomenal growth of young pine, approximately 30 feet in height and 6 inches in diameter" along the south side of Spring Creek on the Gila River Forest Reserve. In a report of the Black Mesa Forest Reserve, Plummer (1904) described stands dominated by young trees. Biswell (1973) explained that forestry practices in the early 20th century, including fire suppression, reduced the spread of wildfires, leading to unprecedented fuel loads and stagnation of sapling thickets.

In a review of the literature, Sackett et al. (1996) noted that trees of all size classes in untreated ponderosa pine stands exhibit greater stress, lower vigor, and reduced growth rates. In a study of increment cores from open-grown ponderosa pine, Cooper (1960) found that mean growth per tree in moderately stocked stands remained relatively constant while growth of open-grown trees was directly proportional to their age. Sutherland (1983) performed a dendrochronological study to evaluate the effects of fire exclusion on the radial growth of two age classes of mature ponderosa pine (approximately 150 and 300 years old). The author found that declines in radial growth in both age classes were well correlated with the establishment of a large ponderosa pine seedling crop in 1918.

The greater stand densities in ponderosa pine forests, combined with fire exclusion, has resulted in changes to forest structure and function (Harvey et al., 2000). Tree stress is often greater in dense stands due to limited resources, particularly soil moisture. Increased stand densities have also resulted in a corresponding increase in insect and disease epidemics (Graham and Jain, 2005). Prior to fire exclusion, recently burned stands were frequently infested with western pine beetle (*Dendroctonus brevicomis* LeConte), which killed large, lightning-scarred and fire-weakened trees. Pine engraver (*Ips* spp.), and fir engraver (*Scolytus ventralis* LeConte) beetles attacked young, dense stands or removed trees that were scorched by wildfires and therefore stressed. In both cases, the result is increased numbers and sizes of forest openings and improved gap dynamics that contribute to structural and spatial heterogeneity. In a study of the effects of bark beetle attacks on ponderosa pine following fire in Northern Arizona, McHugh et al. (2003) found that infestations were lowest following a fall prescribed fire, moderate following a summer wildfire, and highest following a spring wildfire.

Edmonds et al. (2000) noted that natural and human-induced disturbances often resulted in increased damage from root diseases. *Armillaria* is most common in younger stands of ponderosa pine, ranging in age from 10 to 25 years. However, Marsden et al. (1993) found that selective logging in the Jemez Mountains of New Mexico resulted in intensification of the severity of *Armillaria* root disease, resulting in extensive mortality of all age classes of ponderosa pine.

Dwarf mistletoe (genus *Arceuthobium*) has been a major damaging agent in southwestern ponderosa pine. Fire was a primary control of dwarf mistletoe by thinning the forest and pruning of diseased limbs (Hawksworth, 1961). In the absence of fire, and the resulting densification of southwestern ponderosa pine stands, the frequency of mistletoe infection has increased (Kauffman et al.; 1992, Hoyt et al., 2017).

Understory Vegetation Functions and Dynamics

The herbaceous understory of southwestern ponderosa pine forests is composed of a diversity of vascular plants including graminoids, forbs, subshrubs, shrubs, ferns, and cacti. This understory includes both aboveground vegetation and underground propagules, such as dormant seeds and roots, which can germinate or sprout following disturbance or other environmental signals (Korb and Springer, 2003). Common native understory species in ponderosa pine forests and rangelands in northern Arizona include native bunchgrasses such as bottlebrush squirreltail (*Elymus elymoides*), Arizona fescue (*Festuca arizonica*), mountain muhly (*Muhlenbergia Montana*), spike muhly (*Muhlenbergia wrightii*), blue grama (*bouteloua gracilis*), mutton grass (*Poa fendleriana*), subshrubs and shrubs such as broom snakeweed (*Gutierrezia sarothrae*), Fendler’s ceanothus (*Ceanothus fendleri*), and Fendler’s meadowrue (*Thalictrum Fendleri*).

Bunchgrasses and other herbaceous vegetation have been shown to provide several ecological functions including soil retention, nutrient cycling, providing food and cover for wildlife, and carrying low-intensity surface fires. They also play a critical role in the life cycle of ponderosa pine forest and rangeland ecosystems by competing with pine seedlings for water and nutrients (Pearson, 1942). However, introduction of domestic livestock grazing, combined with aggressive fire suppression and effective exclusion throughout much of the 20th century, resulted in a variety of negative effects including changes in species composition, accelerated soil erosion, loss of herbaceous vegetative cover and litter, and dense overstories that reduce light penetration to the forest floor (Ffolliott and Clary, 1982). Historic grazing practices and fire suppression tend to favor woody species over herbaceous plant communities (Archer, 1994; Bragg and Hulbert, 1976; Briggs et al., 2002). Additionally, selective grazing by herbivores may also dramatically alter species compositions and slow ecological recovery (Clary, 1975). As livestock graze on palatable grasses and herbaceous understory plants, dominance can shift toward more unpalatable grasses and herbaceous species and favor encroachment of shrubs and trees into forest openings (Manday and West, 1983; Belsky and Blumenthal, 1997; Kerns et al., 2011). The result can be dense, closed-canopy forests that support fewer understory species and therefore exhibit decreased biodiversity in comparison to other ecotones such as areas with low stand densities and meadows (Moore and Dieter, 1992). In a study of five livestock grazing exclosures established in 1912 and measured in 1941 and 2004, Bakker and Moore (2007) found canopy cover of regenerated trees inside exclosures was higher in 1941 than in 2004. Although there was no significant difference between exclosed and grazed plots after overstory effects were considered, species diversity, herbaceous plant density, shrub cover, and total herbaceous cover all declined from 1941 to 2004. Their conclusion was grazing history must be considered when interpreting results among sites since understory vegetation may be more strongly controlled by ponderosa pine overstory than by contemporary grazing practices.

In a study of forest floor fuel loads in Arizona and New Mexico, Sackett (1979) sampled 62 stands. He found fuel loads ranging from 4.8 tons per acre (10.76 Mg ha^{-1}) on the Tonto National Forest to 20 tons per acre (44.83 Mg ha^{-1}) near the north rim of Grand Canyon National Park. Mean forest floor fuel load was 12.5 tons per acre (28.02 Mg ha^{-1}). If woody material greater than 2.5cm in diameter was included in the calculation, the average increased to 21.7 tons per acre (48.64 Mg ha^{-1}). Of the 12.5 tons per acre (28.02 Mg ha^{-1}) average forest floor fuel loading, approximately one ton per acre (2.24 Mg ha^{-1}) consisted of slightly decomposed organic matter, 3.8 tons (8.52 Mg ha^{-1}) were moderately decomposed organic matter, and 6.1 tons (28.02 Mg ha^{-1}) were highly decomposed. The remaining 1.8 tons (4.03 Mg ha^{-1}) were comprised of small diameter woody debris. Of the total fuel load, including forest floor, 42 percent consisted of larger woody debris. There was 1.4 tons per acre (8.97 Mg ha^{-1}) of debris ranging from 2.5 to 7.5cm in diameter, 5.0 tons (11.20 Mg ha^{-1}) of moderately decomposed woody debris greater than 7.5cm in diameter, and 2.8 tons (6.27 Mg ha^{-1}) of sound wood greater than 7.5cm in diameter.

Soil Dynamics

The short-term, localized effects of ecological restoration of southwestern ponderosa pine on vegetation and soils are well studied (Kaye and Hart, 1998a and b; Feeney et al., 1998; Kaye et al., 1999; Casey, 2004; Korb et al., 2004; Boyle et al., 2005; Moore et al., 2006). However, long-term, landscape scale effects of ponderosa pine forest restoration treatments are not as well understood. Additionally, there are no known studies of southwestern ponderosa pine forest restoration treatments under grazing and non-grazing conditions with the intent of increasing understanding of the combined effects of both to select soil quality indicators.

Forest Carbon Pools and Cycles

Carbon (C) sequestration in forest biomass and forest soils is an essential component of healthy forested ecosystems and global carbon cycles, which includes atmospheric C levels. In southwestern ponderosa pine forests, atmospheric CO_2 is converted into organic C through two primary plant photosynthetic pathways (C3 and C4). The C3 plants are those that photosynthesize most efficiently during the cool, wet season while C4 plants photosynthesize more efficiently during the warm, dry season.

Carbon in southwestern ponderosa pine forests is stored in both inorganic and organic forms. Soil inorganic C includes carbonate minerals derived from weathering of limestone parent materials. In the case of the study site for this project, this would be the Kaibab Limestone geologic formation. Organic C is stored in forests in four primary pools: a) aboveground living biomass of both woody and herbaceous plants, including tree boles, branches, foliage, bark, seeds and non-woody plants, b) belowground living biomass of roots and soil organisms, c) surface residues of plants and animals at various stages of decomposition, and d) as a component of soil organic matter (SOM) in the soil profile.

The importance of soil organic carbon storage in forested ecosystems cannot be overemphasized. Forest soil organic carbon pools exist in dynamic equilibrium with C gains and losses. Soil C gains are the result of biomass inputs, including plant and animal residues and deposition of organic and inorganic carbonaceous materials (Lal, 2010). Carbon inputs can then become resistant to microbial decomposition through formation of stable aggregates, transformations of organic compounds into more complex humic substances, translocation of organic C into lower horizons through illuviation, and conversion of labile forms of organic C into more recalcitrant forms. Removal of forest biomass through grazing, harvesting, and fire can result in losses of soil organic C. Soil C is also depleted through accelerated erosion by wind and water, soil disturbances that disrupt soil aggregates, mineralization and leaching of dissolved organic C compounds, and activities that increase microbial responses and decomposition rates.

In a study of C uptake and storage using eddy covariance, Dore et al. (2010) compared the effects of two disturbance types representative of potential future conditions in ponderosa pine forests of northern Arizona to undisturbed sites: (1) high severity wildfire that resulted in a state transition from forest to sparse grassland and (2) forest thinning treatment to reduce the potential for high severity wildfire. They found that high severity wildfire reduced total ecosystem C by 42 percent in relation to the undisturbed site ten years after the fire and that the site had become a net source of atmospheric C for at least 10 years following wildfire. They also observed lower net primary production, evapotranspiration, and water use efficiency in the burned area. Thinning also reduced total ecosystem C and resulted in the treated site becoming a net source of atmospheric C. However, total ecosystem C was only reduced by 18 percent in comparison to the undisturbed site and the site was a source of atmospheric C for only one year following the thinning treatment. Eddy covariance also indicated that both the undisturbed site and the thinned site were atmospheric C sinks before the thinning treatment. Estimates of net ecosystem productivity showed that the undisturbed site constitutes a small annual carbon sink. This could indicate that minor shifts in forest management practices (e.g., frequent fire return interval, greater areal extent of moderate and high vegetation burn severity during prescribed fire, or greater basal area reductions) could increase the areal extent of ponderosa pine forest that serves as a source of atmospheric C for longer periods of time.

Kaye et al. (2005) measured changes in organic C, nitrogen, and phosphorus levels following ponderosa pine restoration treatments in northern Arizona. They evaluated the effects of a) forest thinning, b) thinning with reduction of forest floor organic matter and c) application of prescribed fire. They found total net primary productivity (260g C m^{-2} yr.$^{-1}$) was similar among treatments since reductions in pine foliage and fine roots in restored sites were offset by increased wood, coarse roots and herbaceous vegetation. The herbaceous plant community was a minor component of total plant C, N, and P uptake in untreated (control) plots at less than 20 percent, but in restored plots the herbaceous plant community accounted for 25 to 70 percent of total plant C, N and P uptake. Ecological restoration treatment plots exhibited lower declines in nutrient fluxes than control plots during drought conditions.

Selmants et al. (2008) studied the effects of additions of nitrogen and water on CO_2 efflux in ponderosa pine stands under restored and unrestored conditions over 13 years with a goal of quantifying the effects of these additions on belowground C cycling under dense stand conditions and to determine if restoration thinning would mitigate water and N limitations. They found that forest thinning and prescribed fire treatments increased efflux of both CO_2 and water from the soil surface. Additions of water and N increased soil C efflux at approximately the same levels in both restored and unrestored forest conditions. Plots that received both water and N amendment had greater soil moisture than plots that did not receive amendment. There was no observed difference between plots that received the water plus N amendment and those that received only water.

Nitrogen

Hart et al. (2006) studied C and N cycling processes in both restored (thinning to 80 percent reduction in basal area with 90 percent reduction in tree density and prescribed fire) and unmanaged ponderosa pine stands. They found that both N availability and aboveground net primary production (ANPP) of trees were lower in both restored and unmanaged conditions in relation to other forest cover types. They attributed these differences to the relatively low annual precipitation, prolonged dry periods during the growing season, and high lignin content of litter in southwestern ponderosa pine forests, which contribute low C and N cycling rates. Overall ANPP decreased under restored conditions, but the proportion of ANPP attributable to woody vegetation increased. Additionally, soil temperature, respiration, available water, and net nitrification increased under restored conditions. However, restoration treatments had no effect on N mineralization and microbial N.

In a study of the effects of ponderosa pine forest restoration treatments ((1) partial restoration – thinning to emulate presettlement forest cover conditions, (2) complete restoration – removal of trees and forest floor organic matter (Oi, Oe, and Oa components) followed by addition of native grass litter at a rate of approximately 672 kg ha^{-1}, followed by a prescribed fire the following year, and (3) control), Kaye and Hart (1998a) found that N mineralization and nitrification were similar on an areal basis among the three treatment types, but were higher on a mass basis under both restoration treatments. Furthermore, net N transformation rates in historic herbaceous openings were twice those observed under post settlement forested conditions. They also observed increased soil temperature in restoration treatments in comparison to the control. Thomas (2017) observed similar soil temperature increases in openings created by ponderosa pine forest restoration thinning treatments 12 years following treatments implemented by Abella et al. in 2003.

Feeney et al. (1998) found soil water content had increased under thinning, and thinning with prescribed fire treatments, in comparison to the control. Basal area increment and leaf nitrogen content also increased in the thinning treatments when compared to the control. Resin flow increased under thinning and burning in comparison to the thinning only treatment and control.

Decomposition rate (k) (Jenny et al., 1949) is the ratio of steady state forest floor weight to the annual accumulated weight. Harrington and Sackett (1992) describe decomposition rates of southwestern ponderosa pine as nearly desert-like conditions. They estimated k values for southwestern ponderosa pine forests of 0.074, 0.059, and 0.048, for dense sapling stands, pole stands, and mature old growth stands, respectively. Humid, tropical forests typically have k values approaching 1.0, indicating that decomposition occurs in the same year litter reaches the forest floor.

Cold, wet winters combined with warm, dry summers, which are a common seasonal pattern in southwestern ponderosa pine forests, can limit soil biological activity and therefore decomposition rates (Olson, 1963; Harvey et al., 1979; and Edmunds, 1991). In the absence of fire, the result is an accumulation of plant detritus (needles and woody fuels). Sackett and Haase (1998) noted that fire exclusion may be the cause of soil nitrogen declines due to increases in organic matter in an environment where decomposition rates are low.

Soil Quality and Soil Health

The term soil quality was first introduced in North America primarily to guide judicious allocation of resources toward sustainable agriculture to ensure soil productivity is maintained. The idea has since evolved to encompass not only soil productivity but concepts of ecosystem sustainability, environmental quality, human and animal health, environmental risk assessment, land evaluation, and understanding environmental change (Warkentin and Fletcher, 1977; Larson and Pierce, 1991; Parr et al., 1992).

There are various ideas of what constitutes soil quality. For example, for those in agriculture, soil quality can mean land having a high capability of producing desired crops that are healthy for consumers while maximizing profit and maintaining productive capacities for future generations. To those in ecosystem management, it may mean soils that contribute to optimum nutrient storage and cycling and biodiversity while sustaining water quality and desirable wildlife habitats. Doran and Parkin (1994, 1996) defined soil quality as “the capacity of a soil to function within ecosystem and land-use boundaries to sustain biological productivity, maintain environmental quality, and promote plant and animal health”. The United States Department of Agriculture (USDA), Natural Resources Conservation Service (NRCS) defines soil quality, or soil health, as the continued capacity of soils to function as vital living ecosystems that sustain plants, animals, and humans (USDA, 2019). This definition recognizes soils as a living body, since only living organisms can exhibit healthy or unhealthy conditions. It also recognizes human influences on soils, even in natural ecosystems that receive minimal human use.

There are both inherent and dynamic soil qualities that affect the ability of soils to function naturally while providing important ecosystem services and adjusting to management actions or anthropogenic influences. Dynamic soil quality indicators are typically in the upper soil horizons and respond to both management practices as well as inherent soil properties, while inherent soil properties tend to be more static and often apply to the entire soil profile (Bünemann et al., 2018). While dynamic soil quality indicators are most often the focus of management decisions, understanding inherent soil properties that affect suitabilities and limitations and the behavior of dynamic soil quality indicators is important if practitioners wish to maintain productive capacities, maximize future options, and minimize long term resource damage or adverse effects to soils resources during management activities, such as forest thinning, prescribed fire, and livestock grazing.

Examples of inherent soil qualities include texture, depth to bedrock, drainage class, clay mineralogy, and cation exchange capacity, to name a few. For example, deep, loamy, moderately well drained soils tend to have greater rooting volume and nutrient status than shallow, gravelly soils. Dynamic soil qualities include soil structure, bulk density, aggregate stability, organic matter content, biological activity, infiltration rate, and water holding capacity.

The term soil quality has often been used interchangeably with soil health. However, soil quality often refers to the ability of a soil to function (i.e., to sustain plant and animal productivity, maintain or enhance water and air quality, and support human health and habitation) (Doran and Parkin, 1994; Robinson et al., 2012), while soil health denotes the state of a soil meeting its range of ecosystem functions as appropriate to its environment, or its relationship to reference condition (McBratney et al., 2014). One challenge in discussing the health of a given soil is having adequate understanding of how 'healthy' the soil can be (McBratney et al., 2014).

More recently, the term soil security has been introduced as an overarching concept that encompasses soil quality, soil health, and soil protection (Koch et al., 2013; McBratney et al., 2014; Field et al., 2017). This concept was inspired by recognition of the need to consider soils resources in the context of sustainable development. It has further been recognized that soils have similar existential threats as food and water security, biodiversity, climate stability, delivery of ecosystem services, and energy sustainability (McBratney et al., 2014). Soil security consists of five dimensions including capability, condition, capital, connectivity, and codification of soils managed in an interdisciplinary manner through biophysical, social, and economic sciences. Capability refers to functions that soils perform and their productive capacities. Condition is the current state of a soil and potential change in capability in comparison to reference condition. Capital refers to the value placed on the functions that permit a soil to deliver services. Connectivity is the social component whereby those who manage soils resources are connected to sources of information that assists them with managing soils resources to their full capabilities. Connectivity also recognizes the importance of intergenerational equity and proposes the need for a soil ethic whereby soils are not only valued for the well-being they provide to humans. Codification identifies the need for policies and regulations that contribute to soil security and synergize among other dimensions of soil security such as improved management, increased natural capital, improved education, and societal connectivity (McBratney et al., 2014).

Soil Bulk Density

Soil bulk density is the ratio of oven-dried soil to its total volume, including the volume of both particles and pore space. It is dependent on the arrangement and densities of the soil particles (sand, silt, clay, and organic matter). Mineral particle densities usually range from 2.5 to 2.8g cm^{-3}, while organic particle densities are usually less than 1.0g cm^{-3}. Bulk density is a dynamic soil quality that is easily altered by activities such as machinery traffic, trampling by humans or animals, intensive cultivation, loss of organic matter, additions of organic matter, raindrop impact on bare soil surfaces, etc. It is therefore a useful indicator of soil quality or condition. (Arshad et al., 1996). Compacted soil layers have high bulk densities that can restrict root growth and reduce pore volume, which then inhibit the movement of gasses and water through the soil (Hunt and Gilkes, 1992).

Tarpey et al. (2008) studied the long-term effects of forest thinning and partial cutting on soil compaction in red pine (*Pinus resinosa* Ait.) and northern hardwood stands in the Great Lakes Region. Soils in the red pine stands were Menahga loamy sands, which occur on nearly level glacial outwash. Periodic thinning of red pine stands to 23.0, 27.6, and 32.1m^{2} ha^{-1} residual basal area on these soils over a 57-year period resulted in increased soil compaction as thinning intensity increased. There were no significant differences in bulk densities of the A horizons of the three thinning intensities in the red pine stands. However, they observed bulk densities of the B horizon in the 23 BA and 32.1 BA plots were higher than in the untreated control by 17.0 and 10.3 percent, respectively. The northern hardwood stand occurred on the Argonne sandy loam, which was formed on a glacial till plain with a high rock content and a moderately deep (50-100cm) fragipan. No significant difference in soil bulk densities was found in either the A or B horizons. The authors noted that the high rock-fragment content of the Argonne Series may have protected it from compaction, but it may also have reduced the ability to accurately measure whether soil compaction had occurred as they were only able to sample when their soil coring device could penetrate between rocks. Similar problems occurred when using a penetrometer.

In a study of the effects of two forest thinning treatments (felling only versus felling with skidding) on soil strength in 70- to 80-yr.-old ponderosa pine stands in central Oregon, Parker et al. (2007) found felling with skidding resulted in 44 percent higher soil strength values than felling only. They also found diameter, height, and volume growth of individual trees within plots declined significantly as the average soil strength values within a 9m zone of influence increased from 800 to as much as 2,500kPa. Soils in the study area were characterized as young, poorly developed, and unweathered Cryands with a significant Mt. Mazama ash component and surface textures of loamy sand and sandy loam. These soil characteristics combined with a semiarid climate result in low fertility that is growth-limiting and therefore limits organic matter inputs. Low inherent bulk densities and porosities are also common in these soils. As a result, they are highly susceptible to compaction. The author noted that soil strength was the only soil quality indicator measured in this study. Ponderosa pine growth-limiting increases in soil strength were not partitioned into changes in porosity, infiltration, or root dynamics, so the authors were not able to determine if more specific causal mechanisms may have limited tree growth in the felling with skidding treatment plots.

Korb et al. (2007) studied the effects of different forest restoration treatment levels (control, low, intermediate, and high) and treatment types (machine cut-to-length, hand thinning, and whole-tree harvesting) on soils in a northern Arizona ponderosa pine forest. Soils were classified as a) deep, loamy-skeletal, mixed Mollic Eutroboralfs, b) deep, fine, montmorillonitic Mollic Eutroboralfs, and c) fine, montmorillonitic Typic Argiborolls. The two fine-textured soils were identified as having limitations when wet due to risk of soil compaction and displacement. The authors found that harvest levels did not cause significant differences in soil profile disturbance, soil bulk densities, or changes to profile depths with treatments implemented under dry conditions. However, significant differences in soil disturbance were observed by treatment type. Whole-tree harvesting resulted in no disturbance on significantly less ($p \leq 0.05$) area (41.3 percent) than cut-to-length machine (64.5 percent) or hand thinning (64.9 percent) treatment types. There was also a significantly higher ($p \leq 0.05$) proportion of the study area that exhibited high disturbance under whole-tree harvesting (13 percent) than either the machine cut-to-length (two percent) or hand (2.3 percent) treatment types.

The authors noted that high levels of soil profile disturbance may have been the result of the amount of basal area removed during treatment. The authors further concluded that their experimental design did not stratify based on areas within the study sites that exhibit higher disturbance levels (skid trails and landings). The authors recommended stratified random sampling to quantify areas that would exhibit higher levels of soil disturbance such as landings and skid trails.

Abdel-Magid et al. (1987) evaluated the effects of three grazing systems: a) continuous, b) rotational deferment, and c) short-duration rotation on bulk density and infiltration rates of mixed-grass prairie at the High Plains Grassland Research Station in southeastern Wyoming. Soils were classified as Aridic Argiustolls and had a sandy loam surface texture to 13cm depth overlying sandy clay loam to 28cm depth. Infiltration was measured using a double ring infiltrometer. The research sites had been grazed lightly from 1976 through 1978 and from 1979 to 1982 and for several years prior to 1976 they were not grazed. At the time of establishment of the research plots, soil bulk densities were not significantly different among grazing systems, stocking rates, or season of use. Soil bulk densities were significantly higher after two spring and two fall grazing seasons but were not significantly different among grazing systems or stocking rates. The continuous grazing treatment had an average equilibrium infiltration rate of 9.69cm hr^{-1} in 1983, which was significantly greater than that observed for the rotational deferment treatment, but not significantly different than short-duration grazing. Two spring and two fall seasons after project establishment the short duration grazing treatment had the highest equilibrium water infiltration rate and the continuous grazing treatment had the lowest. Trends in water infiltration rates did not follow those of bulk density, indicating that soil bulk density may not be a causal agent controlling infiltration rates on some soils under different grazing systems and stocking rates. High stocking rates consistently reduced infiltration during the grazing season, but freeze-thaw actions appeared to ameliorate this short-term detrimental condition.

In a paired plot study in northwestern New Mexico designed to evaluate the effects of previous heavy grazing of more than 50 years on soils and vegetation, Orodho et al. (1990) conducted a study on three sites that represented a range of landscape positions (hilltop, hillside, and swale) common to the Four Corners area. Plots 50m x 75m were installed on each landscape position in areas excluded from grazing for 50 years (i.e., within the Chaco Culture National Historical Park) and on adjacent BLM land where heavy grazing was still practiced, although they were protected from grazing for the duration of the study (two years). Twelve soil bulk density samples were collected randomly from the surface five cm within each plot. Significant differences were found between grazed and ungrazed conditions as well as by landscape position. Areas where previous heavy grazing had occurred exhibited soil bulk densities that were eight percent greater than protected areas (1.50g cm^{-3} vs. 1.38g cm^{-3}). The largest increases in bulk density from heavy grazing occurred on the hilltops, while the smallest increases occurred in grazed swales.

Infiltration

Soil infiltration is the ability of water and dissolved solutes to move vertically into soil. The infiltration rate, or infiltration capacity is the maximum rate at which soil can absorb water that is impounded to a shallow depth at the surface with boundary conditions, or fringe effect controlled (Johnson, 1963).

Under natural conditions, water entry into soil is caused by matric and gravitational potentials. Therefore, water entry into soil may occur in the lateral and upward directions as well as downward (Baver et al., 1972; Jury and Horton, 2004). When soils are dry, rainfall tends to infiltrate into the surface soil layers at a decreasing rate over time. As water redistributes through a soil profile, it displaces soil gases and fills pore spaces, causing resistance to flow to increase, thereby decreasing the hydraulic gradient and the capacity for more water to infiltrate. Therefore, soil infiltration capacity at initiation of the process (unsaturated soil) is at its maximum. As saturation increases infiltration decreases asymptotically towards a constant value, or saturated soil condition.

Infiltration capacity is an important soil quality indicator because it determines how much of the incident rainfall will runoff and how much will enter the soil in a specified timeframe. Where the input rate exceeds the infiltration capacity of the soil, rainfall excess (otherwise known as effective rainfall) will be generated and contribute to overland flow.

In a study of the effects of selective timber harvesting on soil physical and hydraulic properties in an oak-hickory forest on moderately well-drained claypan soil (fine, smectitic, mesic Aquertic Chromic Hapludalfs) in Missouri, Langston and Henderson (2016) found that soil bulk density was significantly greater ($p < 0.01$) and saturated hydraulic conductivity (Ksat) was significantly lower ($p < 0.01$) under logging roads and log landing areas to 30cm depth in comparison harvested areas. However, there was no significant difference in these parameters at the 40cm depth. They concluded that logging had minimal effect on soil bulk density and Ksat while logging roads and landings produced significant changes in these soil properties.

Johnson and Beschta (1980) studied the effects of logging on infiltration and surface soil erosion in a Doulas fir (*Pseudotsuga menziesii* [Mirb.]), western hemlock (*Tsuga heterophylla* [Raf.] Sarg.), western redcedar (*Thuja plicata* Donn ex. D. Don), sugar pine (*Pinus lambertiaia* [Dougl.]), and incense cedar (*Libocedrus decurrens* [Torr.]) stands in western Oregon. They investigated four treatment types: 1) shelterwood using ground-based skidding, 2) clearcutting with cable yarding, 3) clearcutting with ground-based skidding and windrowing of slash, and 4) undisturbed forest conditions. Soils were mixed and derived from basalt, red and green breccias, agglomerates, and tuffs; scattered rhyolitic breccia soils were also present in the study area. Infiltration capacity was comparable to undisturbed conditions across all treatment types except ground-based skidding with windrowing of slash. Subsurface clay under this treatment exhibited massive conditions and infiltration capacity was reduced. Harvesting using ground-based machinery also increased soil erodibility in two of the watersheds studied.

Smith (1967) compared infiltration trends under three grazing intensities in relation to exclosures on three cover types of a ponderosa pine-bunchgrass range on the Pike National Forest in Colorado. Infiltration rates were approximately the same from 1941 to 1952. Under moderate grazing infiltration rates remained high, while under heavy grazing, infiltration rates were reduced. At time of exclosure installation in 1941, infiltration rates were similar across both grazed and non-grazed treatments. However, infiltration rates inside exclosures increased six and thirteen years after exclosure installation due to protection from grazing, while infiltration under grazed plots remained approximately the same eleven years after grazing.

In a study of the effects of livestock trampling on watershed condition, runoff, and erosion, in southwestern Idaho, Packer (1953) found disturbance levels up to 40 percent areal extent resulted in removal of ground cover, including litter, which reduced infiltration capacities and caused a corresponding increase in soil erosion rates.

Aggregate Stability

A soil aggregate is ‘‘a group of primary soil particles that cohere to each other more strongly than to other surrounding particles.’’ (SSSA, 1997). Soil aggregation occurs as a result of both attractive and disruptive forces acting on soil particles within the soil matrix to increase cohesion among some soil particles, or groups of particles more effectively than others. Soil aggregation strongly influences root growth and the effectiveness of roots in uptake of water and nutrients.

Aggregate stability refers to the ability of soil aggregates to resist disintegration when disruptive forces are applied (NRCS, 2011). Wet aggregate stability suggests how well a soil can resist dispersion resulting from raindrop impact and runoff, while size distribution of dry aggregates can be used as an indicator of stability against wind erosion (Nimmo, 2004).

Changes in aggregate stability can serve as one measure of either improvement or degradation of soil condition, or quality. Aggregate stability may also serve as an indicator of the amount of organic matter in a soil as well as changes in biological activity and nutrient status. Certain organic compounds such as bacterial polysaccharides can cause attraction between soil particles, while others may influence soil aggregation by affecting surface tension or electrical charges of soil particles (Acton et al., 1962). Plant roots and fungal hyphae adhere to soil particles as part of their natural function. Some chemicals and compounds in soils serve as binding agents that enhance soil aggregation. Common substances that contribute to soil aggregation include humic substances, calcium carbonate, and oxides of iron, aluminum, and silicon. Soil aggregates that are less than 0.25mm in size are often bound by more stable forms of organic matter. Microbial decomposition and enzyme activity tends to release byproducts that are less stable forms of organic matter that bind small aggregates into large aggregates. Larger soil aggregates tend to be more sensitive to management-induced changes to soil organic matter content and types, thereby serving as a better indicator of changes in soil quality.

Increased soil aggregation and aggregate size has been shown to improve both macropore and mesopore volume in soils (Sun and Lu, 2014), which improves soil gas exchange and water entry and movement in soil, which then affect nutrient availability and biological activity. Large pores associated with large, stable aggregates tend to favor high infiltration rates and root aeration. Pore spaces also serve as zones of structural weakness in soil that improves root penetration and growth (USDA, 2011).

When surface soils are weakly aggregated, they are susceptible to dissolution and formation of crusts, which then decrease infiltration, increase erosion, and inhibit vegetation establishment and growth (Fox et al., 2004). Surface soil bulk density increases as pore spaces are filled with fine particles when surface crusts form.

Wind normally detaches and transports only those soil particles that are loosely bound at soil surfaces. However, as windblown soil particles are transported, the process of saltation can occur, whereby blowing soil particles strike bare soil with sufficient energy to dislodge additional particles from soil surfaces, thus propagating the process (Kolk et al., 2012).

In a study of the effects of timber harvesting on fine-textured soils in Germany, Klaes (2016) found that aggregate stability had been compromised by rutting where forwarding of logs had occurred. This determination was made based on percolation rates in rutted and undisturbed areas. They attributed the loss in aggregate stability to topsoil shearing and churning and loss of surface organic matter. As the number of passes increased, aggregate stability and percolation decreased up to the maximum of five passes studied.

Chappell et al. (1999) studied physicochemical properties that affect aggregate stability in tropical Ultisols after forestry operations. They investigated sites that were undisturbed versus those that had been thinned and were eroding. Clay content was not significantly different ($p = 0.10$) between the eroding and noneroding sites; however, a significant but weak positive correlation ($r^2 = 0.371$, $p = 0.027$) between the $\log_{10}$ linearized clay content and aggregate stability was observed at eroding sites. They observed significant variability in aggregate stability between horizons within the same soil profile. They concluded that there was uncertainty in whether forestry activities caused reduced soil aggregate stability or simply exposed unstable horizons.

In a study of the effects of forest thinning in Chinese fir (*Cunninghamia lanceolata*) plantations on water-stable soil aggregates 18 years after treatment, Cheng et al. (2018) found that the distribution of water-stable aggregates was similar among three treatment types – conventional management (single, light thinning (control)) with stand density reduced by 15 percent at year 14, moderate thinning with stand density reduced by 25 percent at year seven, and heavy thinning with stand density reduced by 33 percent at year seven. While only minor changes were observed in the fractions of water-stable aggregates and aggregate–associated carbon in stands that had undergone repeated short-term thinning, bacterial and fungal species and populations were different among treatments. They noted that the rate of recovery of soil properties from disturbance may vary, depending on the degree of soil disturbance and microbial diversity and composition. They further concluded that water-stable aggregates and soil microbial communities are resilient to repeated thinning in Chinese fir plantations.

Warren et al. (1986) studied short duration grazing on the Kavett soil series (clayey, montmorillonitic, thermic, shallow Petrocalcic Calciustolls) at the Texas Agricultural Experiment Station near Sonora, TX. Grazing impacts were studied under wet and dry pasture conditions by fastening pedometers to the forelegs of Brangus heifers to estimate trampling levels and evaluate hoof impacts during grazing. Four grazing levels were studied, with treatments repeated five times at 30-day intervals. Aggregate stability was found to be greater under dry soil conditions than wet when heavy grazing occurred. Otherwise, no significant differences were found in aggregate stability under dry soil conditions at all grazing levels studied within a given treatment interval. However, under moist soil conditions, aggregate stability showed a downward trend over time as stocking rate increased. When compared to untrampled exclosures, moist soil aggregate stability decreased stepwise over time with each successive grazing treatment. They concluded that physical disturbance at levels of intensive rotation grazing was detrimental to soil properties that are strongly correlated to infiltration rates and erosion. Trampling under moist soil conditions destroyed soil aggregates and created flat, impermeable soil surfaces that were dense and unstable. These induced conditions were detrimental to infiltration and resulted in increased sediment production.

Knoll and Hopkins (1959) studied the effects of grazing and tramping on soil physical properties including aggregate stability, moisture content, degree of compaction, and infiltration rates under three grazing intensities. They studied three sites near Hayes, KS. One site carried four animal units per acre for several years (heavily grazed). A second site had one animal unit per acre (moderately grazed), while a third site was excluded from grazing. Soils were derived from loessal materials over limestone hills. Soil aggregates were found to be relatively unstable under all grazing levels. Water-stable aggregate percentages greater than 0.5mm. in diameter were 89.0, 63.6, and 55.6 percent in the ungrazed, moderately grazed, and heavily grazed sites, respectively. Infiltration in a two-hour period was 6.55, 5.28, and 4.01cm at ungrazed, moderately grazed, and heavily grazed sites, respectively.

Available soil moisture did not occur below 0.61m under the heavily grazed pasture in May, but it extended to a depth of 0.91m under the moderately grazed and ungrazed pastures. These conditions persisted well into the dry season when moisture became unavailable altogether.

Wood and Blackburn (1981) studied the effects of grazing on soil parameters that influence water infiltration rates in pastures at the Texas Experimental Ranch, between Throckmorton and Seymour, Texas. They investigated a) moderate and high stocking rates under continuous grazing, b) rested and grazed deferred rotation, c) rested and grazed high intensity, low frequency, and d) two livestock exclosures which had been grazed for 20 years. They studied three vegetation types: shrub canopy, shortgrass interspaces, and midgrass interspaces. Variables investigated that were believed to influence infiltration on grazed areas included ground cover, perennial grass cover, total grass cover, bulk density, aggregate stability, and organic matter content. Aggregate stability was the most influential variable affecting water infiltration in all three vegetation types and in all grazing treatments except the high intensity-low frequency treatment, where aggregate stability was of lesser importance. The less important role of aggregate stability in the high intensity-low frequency treatment was attributed to fewer stable aggregates, believed to be the result of degraded range condition due to higher livestock concentrations.

Soil Slaking

Soil slaking is the disintegration of large, air-dry soil aggregates (greater than 2mm in size) into smaller microaggregates (< 0.25mm) when they are rapidly immersed in water. Slaking occurs when aggregates are too weak to withstand internal stresses caused by rapid water uptake. Internal stresses can result from differential swelling of clay particles, air that is either trapped in soil pores or escapes from them, rapid release of heat during wetting, and the mechanical action of moving water. Organic matter reduces slaking by binding mineral particles and by slowing the rate of wetting (NSW DPIE, 2019). Soil slaking can reduce soil permeability by sealing soil surfaces with microaggregates. This condition can lead to excessive hardness when soils dry.

Tisdall and Oades (1982) and Oades (1984) developed a conceptual model of soil aggregation and the hierarchy of aggregate sizes. Their model describes how particles of primary minerals are bound by bacterial, fungal and decomposing plant materials into microaggregates. The stability of microaggregates is also increased by multivalent cations which improve bonding of organic colloids and clays. Soil microaggregates are then bound into macroaggregates by polysaccharides, bacterial mucilages, organic acids which increase negative charges on clay particles, fungal hyphae, and both living and decomposing plant roots.

There is limited research on the use of soil slake tests for evaluating surface soil slaking and dispersion in forest and rangeland soils management. However, some studies have been undertaken to assess surface soil aggregate size distributions and slaking potential in cropping systems. Six et al. (2000) investigated the effects of cultivation intensity on aggregate size distribution and aggregate C storage in three soils dominated by shrink-swell clay mineralogy (2:1) and a fourth characterized by a mix of 2:1 and 1:1 clays. They investigated sites with native vegetation, no-tillage, and conventional tillage. Slaked (i.e., air dried followed by rapid rewetting) and capillary rewetted soils were separated into aggregate size classes (< 0.053mm, 0.053-0.25mm, 0.25-2.00mm, and > 2.00mm) by wet sieving. There was no significant difference in the proportion of dry soil weight accounted for in macroaggregates (85 percent) among native vegetation and cultivation intensity. However, aggregate distribution in slaked soils shifted away from more macroaggregates toward more microaggregates as cultivation intensity increased.

For soils with 2:1 mineralogy, C content of macroaggregates was 1.65 times greater than microaggregates. These results support the aggregate hierarchy model developed by Tisdall and Oades (1982). Soils with mixed mineralogy did not show a corresponding increase in aggregate C with increasing aggregate size. They concluded that increased cultivation intensity resulted in a reduction in C-rich macroaggregates and a corresponding increase in C-depleted microaggregates that display aggregate hierarchy.

Collins et al. (2015) studied soil aggregate stability as it relates to runoff and sediment yields in southeastern Arizona. The authors used a rainfall simulator to determine runoff and erosion rates. Soil slake tests were used to evaluate aggregate stability, and vegetation cover attributes were measured under three vegetation states for the purpose of informing a state-and-transition model. The states included reference conditions, a site encroached by mesquite, and a site invaded by Lehmann lovegrass (*Eragrostis lehmanniana*). Vegetation was classified by species, and surface cover characteristics below canopy cover and in interspaces were recorded in five categories as either soil, litter, plant crown, gravel, or rock. Aggregate stability was measured using the slake test as described by Herrick et al. (2001) and assigned rankings from one to six based on the stability class. Significant differences in both canopy and ground cover were observed between reference sites and mesquite encroached sites. Mesquite encroached sites had lower grass cover than reference and Lehmann lovegrass sites. As a result, litter cover was also lower in mesquite encroached sites. Total bare soil percentages for the reference and Lehmann lovegrass sites were very low (6 to 13%) due to high amounts of litter cover while the amount of bare soil was significantly greater (42 to 64%) for the mesquite encroached sites. The study revealed a significant negative relationship (r^2 = 0.61) between canopy cover and sediment yield. Significant negative relationships were also found between sediment yield and mean aggregate stability. Significant rilling was not observed on the sites, causing the authors to conclude that the dominant erosion processes were rainsplash and sheet erosion. The authors also concluded that monitoring of aggregate stability combined with vegetative cover appears to be an inexpensive, rapid, repeatable approach to understanding erosion potential in semiarid rangelands.

METHODS

Study Area

This study used existing long-term research plots installed in 2003 in the Northern Arizona University (NAU) Centennial Forest (Abella et al., 2015), located approximately 10 km southwest of Flagstaff, Arizona (Figure 1). The Centennial Forest is a 20,000-ha research, teaching, and demonstration area managed in cooperation with the Arizona State Land Department. The study area is dominated by ponderosa pine and Gambel oak (*Quercus gambelii* Nutt.). The study area occurs within the North central climatological division of Arizona and is generally classified as low sun cold climate class. Precipitation varies from 45cm to 76cm annually and is bimodal. Most of the precipitation falls from November through March, mainly in the form of snow. Winters are cold and the soil temperature regime is classified as frigid, and soils are subject to freezing and thawing. Summer precipitation is irregular, but usually takes place in the form of high-intensity, short duration, isolated thunderstorms during the monsoon season (July through September). Average annual temperatures range from 14.1°C at lower elevations to 11.4°C at higher elevations. For the month of January, mean minimum temperatures range from -12° to -7°C; mean maximum temperatures range from 0° to 10.0°C. For the month of July, mean minimum temperatures range from 7.2° to 11.1°C; mean maximum temperatures range from 21.1° to 37.5°C. Plot elevations throughout the study area range from 2,195m to 2,255m above mean sea level.

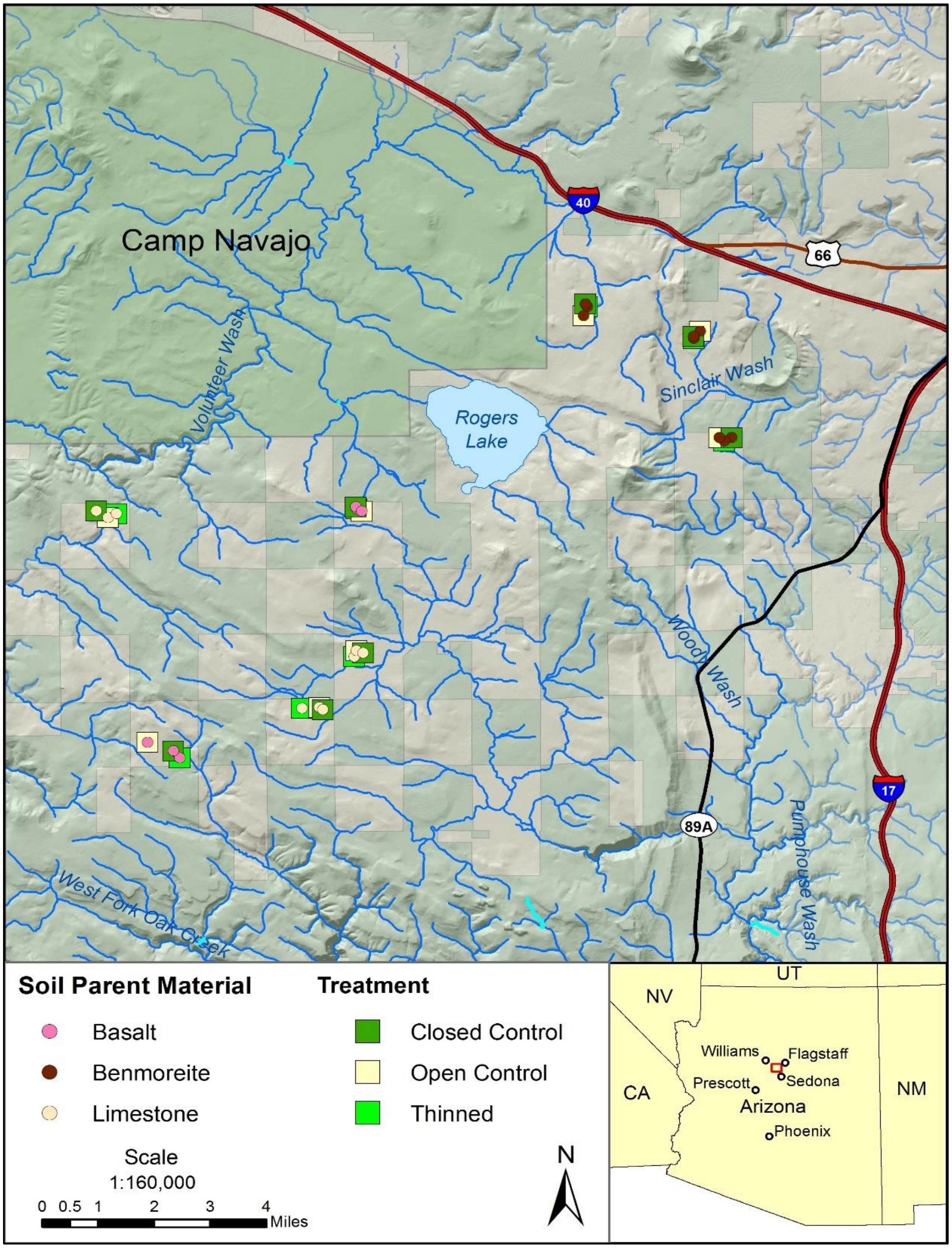

Figure 1. Study site and measurement plot locations in the Northern Arizona University Centennial Forest near Flagstaff, Arizona.

Wildlife common in the study area include mule deer (*Odocoileus hemionus*), pronghorn (*Antilocapra americana*), and Rocky Mountain elk (*Cervus elaphus*), Abert's squirrels (*Sciurus aberti*), and other small mammals, birds, and large predators. Rocky Mountain elk were introduced to Arizona from Yellowstone National Park in 1913 after the Merriam's elk (*Cervus canadensis merriami*) were extirpated (Arizona Game and Fish Department, Phoenix, AZ, U.S.A.).

The study area is used for seasonal livestock grazing of both cattle (*Bos taurus*) and sheep (*Ovis aries*). During the initial study, livestock numbers in animal unit months (AUM) per hectare within the study area ranged from 0.03 to 0.20 AUM ha^{-1} (Bakker and Moore, 2007). Livestock numbers during this study were reportedly 6,780 across the grazing allotments where this research was conducted, which is between 0.002 and 0.010 AUM ha^{-1} (pers. comm. Gary Hase, Rangeland Program Manager, US. Forest Service, Coconino National Forest, Flagstaff Ranger District, Retired); however, significant historic overgrazing did occur throughout much of Arizona between 1850 and 1890 (Loeser et al., 2007), and there were also substantial increases in livestock numbers during both World Wars (USDI, 1992). There are large numbers of wildlife ungulates throughout the project area, primarily large elk herds, but also mule deer and pronghorn.

Soils

The soils of the study area are derived from parent materials consisting of either basalt, benmoreite, or limestone, and were classified as either Typic, Lithic, or Mollic

Eutroboralfs (Table 1). However, since the 'bor' suborder formative element has been eliminated from more recent revisions to soil taxonomy, these soils would most likely be classified as Typic, Lithc, and Udic Haplustalfs, respectively.

Table 1. Soil characterizations as reported by Abella et al. (2015) before implementation of forest restoration treatments.[a]

	Limestone	Benmoreite	Basalt
Elevation (m)	2190 ± 21	2225 ± 40	2214 ± 11
Rock cover (%)	1 ± 1	4 ± 2	5 ± 1
0-15cm soil			
Gravel (%)	28 ± 2	34 ± 3	38 ± 14
Sand (%)	46 ± 11	28 ± 1	30 ± 1
Clay (%)	16 ± 4	18 ± 3	24 ± 3
Organic C (%)	1.5 ± 0.7	1.5 ± 0.1	1.6 ± 0.2
Total N (%)	0.08 ± 0.03	0.09 ± 0.01	0.10 ± 0.01

Values are mean ± standard deviation for the soil parent material types with three replications.

[a] Environmental variables and soil properties obtained in 2003 and described by Abella and Denton (2009). Soil values are percent by weight. Gravel is coarse fragments greater than 2mm in diameter.

The surface texture of the soils derived from limestone parent material are dominantly gravelly loams (Table 1). Soils derived from benmoreite are gravelly to very gravelly silt loams, and soils derived from basalt are gravelly to very gravelly silt loams to loams.

Abella et al. (2015) reported increasing C concentrations with increasing depth in the soils derived from limestone parent materials. This was likely inorganic $CaCO_2$ and not organic C. Thomas (2017) reported surface textures for random samples collected for texture analysis. She found that limestone- and basalt-derived soils had sandy clay loam to loam textures while benmoreite-derived soils had loam textures.

Study Sites

This research uses eight of the original study sites and associated measurement plots installed by Abella et al. (2015). One site (measurement plots CF31-CF35), located on basalt parent materials, was destroyed by a tornado in October 2010 and was therefore excluded from this study. Some exclosure plots were damaged by the tornado and substantial damage to forest vegetation, particularly overstory trees through windthrow and breakage, had occurred. This site was further disturbed by logging machinery associated with a subsequent timber salvage. Domestic livestock and wildlife ungulates may have entered damaged exclosures, grazing and browsing on vegetation.

Study sites were installed across an environmental gradient that encompasses the three surficial geologies (parent materials) previously described. The following criteria were applied by Abella et al. (2015) when selecting sites and installing measurement plots: Each study site was located within 1km of a forest road, had no visual evidence of fire since 1880, and contained ponderosa pine that were greater than 50 years of age, so treatments were representative of typical forest thinning treatments in southwestern ponderosa pine forests. Precipitation patterns, grazing history, forest management practices, and elevations were similar among all study sites, so the primary difference among study sites was the soil parent material. Four treatment plots were installed at each study site. Each treatment plot consisted of two 3.16m × 3.16m (~10m^2) measurement plots delineated within a 20m × 25m (0.05ha) area. These 10m^2 measurement plots had a forest treatment component, grazing control, and grazing exclusion. Each 10m^2 grazing exclosure measurement plot was centered in one half of the 20m x 25m area and had a small buffer 0.08m wide along the outer edge of the plots to prevent microclimate interference caused by snow drift and accumulations due to the presence of exclosure fences. Excluding the buffers, the measurement plots were $9m^2$ in size, and are referred to as the measurement plots for the remainder of this manuscript. Grazing exclosure measurement plots were fenced to approximately 2m height using 1mm gauge welded wire fence with 5cm x 10cm openings. Exclosures were not designed to eliminate all herbivory. Small mammals and avifauna were able to enter grazing exclosures and forage on above-and below-ground biomass. The second (paired) measurement plot at each site, referred to as the grazed plots for the remainder of this manuscript, was originally delineated by metal rods at each corner and was centered in the other half of the 20m x 25m area (Figure 2). In a few cases, rods used to delineate measurement plot boundaries could not be found either visually or by using a metal detector. In these cases, existing rods, plot diagrams and descriptions provided by Abella (2015) were used to reestablish plot boundaries. At the end of this study, all measurement plots were marked with metal rods at each corner.

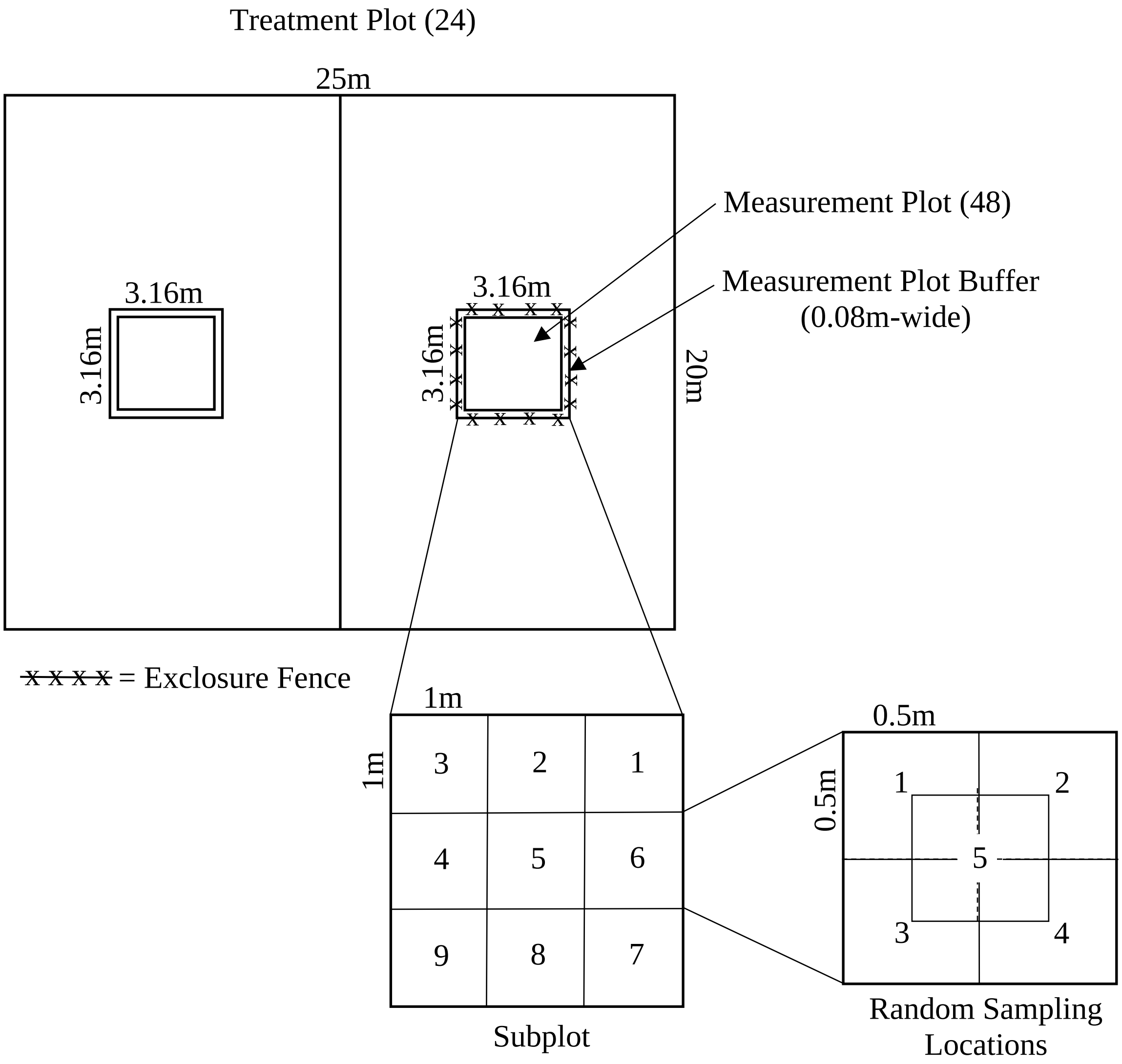

Figure 2. Diagram of plot configurations, including treatment plot, measurement plot, subplot, and sub-subplot used for random sampling.

Study Design

In Abella et al. (2015), plant community responses of species richness, cover, and composition were compared to unrestored control for four types of forest treatments a) ecological restoration thinning, b) open stand conditions, and c) unrestored control (closed canopy conditions), and d) aqueous smoke application simulating implementation of prescribed fire, all of which were analyzed under grazing and non-grazing treatments.

For this study, long term study of aqueous smoke application was not carried forward since Abella et al. (2014) found no significant effect from application of aqueous smoke in their initial study. Treatments were nested within a balanced, four-factor, experimental design, which consisted of three soil types (basalt, benmoreite, or limestone). Nested within each soil type are three treatments (forest thinning, closed canopy conditions, and open canopy conditions). Within each soil type and treatment type were two grazing treatments (grazing or non-grazing). The final factor was time, for which there were four levels: 2003 (pre-treatment), 2006, 2008, and 2015 representing three, five, and, twelve years post-treatment, respectively (Table 2). Soils derived from basalt parent material had one site less than soils derived from benmoreite and limestone parent materials due to the tornado and salvage logging that damaged one of the study sites, so this study was not perfectly balanced.

Table 2. Experimental plot distribution by soil parent material and treatment type.

Basalt			Benmoreite			Limestone		
Thinned	Unthinned	Open	Thinned	Unthinned	Open	Thinned	Unthinned	Open
CF36	CF37	CF40	CF16	CF18	CF20	CF02	CF05	CF04
CF42	CF43	CF44	CF23	CF22	CF24	CF07	CF10	CF08
			CF28	CF30	CF26	CF14	CF15	CF13

Treatments were designed to represent the typical range of forest structural diversity targeted during ponderosa pine forest ecological restoration projects in the southwestern U.S. It is common practice for forest managers to strive to achieve forest spatial and structural heterogeneity by leaving patches where forests are not thinned to increase wildlife habitat and understory vegetative diversity; creating a mosaic of burned and unburned conditions (including occasional fire-induced tree mortality) during implementation of prescribed fire or managing naturally ignited wildfires; leaving remnant openings untreated during forest thinning treatments; and removing a majority of trees when restoring meadows that have been encroached upon by ponderosa pine trees.

Two plots at each study site were randomly selected for thinning treatments by Abella et al. (2015). Ponderosa pine trees were thinned to between 60 and 80 trees ha^{-1} from an original average density of 1,362 trees ha^{-1}. Forest thinning in treatment plots was completed using chainsaws, so there was no ground disturbance from logging machinery. Artificial edge effect was addressed by creating a 5m-wide buffer around each treatment plot that was also thinned. Tree boles and residual woody debris were manually removed from each thinned plot. Each measurement plot was divided into nine subplots measuring $1m^2$ each.

Data Collection

Soil Sampling and Measurements

Samples were collected and measurements taken at each measurement plot under grazing and non-grazing conditions. Five of the nine 1m x 1m subplots were randomly selected for sampling. Each of these five subplots was subdivided into five sub-subplots. Four of these sub-subplots represent one-quarter subplots, each of dimensions 50cm x 50cm. The fifth sub-subplot also represents one quarter of a subplot, but it overlaps the four other sub-subplots with the center being positioned in the centroid of the 1m x 1m subplot (Figure 2). One of these five sub-subplots was selected for sampling. Where large rocks, boulders, trees or previous sampling disturbance by Thomas (2017) precluded sample collection or measurement of soil quality parameters, a different sub-subplot was randomly selected.

Soil Bulk Density

Soil bulk density samples were collected using the extraction method. This method required excavation of a small hole, approximately 10cm deep and 15cm in diameter. All soil, coarse fragments (gravel and rock) were removed from the hole and placed in a plastic bag for laboratory analysis. Roots extending into the hole were clipped flush to the edge of the hole using a flower shear. The hole was then lined with a plastic bag, being sure to extend the bag above the ground surface. Using 1,000ml and 50ml graduated cylinders, the hole was filled with water to the soil surface and the volume of water required to fill the hole was recorded. The amount of water required to fill the hole represents the volume of soil and coarse fragments removed (including the associated pore space). The water filled bag was carefully removed from the hole and checked for leaks to ensure measurement accuracy.

In the laboratory, the excavated material was weighed to determine the moist weight. The excavated material was sieved using an ASTM E-11, Number 10 (2.00mm mesh) sieve. Soil that passed through the sieve was placed in an aluminum container and oven-dried at 105°C until a constant weight was achieved (approximately 24 hrs.). Coarse fragments were weighed to determine the coarse fragment mass. The volume of coarse fragments was determined by placing 300ml or more of water in a 1,000ml or larger beaker, then carefully adding rocks and recording the change in water level. The following equations were then applied:

$$\begin{aligned}&\text{Mass of coarse material (Mg)}\\ &\qquad = \text{coarse material wt. (g) x } \frac{1\text{kg}}{1000\text{g}} \text{ x } \frac{1\text{Mg}}{1000\text{kg}}\end{aligned} \tag{1}$$

$$\begin{aligned}&\text{Vol. of coarse material } (\text{cm}^3)\\ &\qquad = \text{cyl. vol. with coarse frag. (ml)} - \text{initial cyl. vol. (ml)}\end{aligned} \tag{2}$$

$$\text{Volume of soil } (\text{cm}^3) = \text{Total vol. of soil} - \text{Total vol. of coarse fragments} \tag{3}$$

$$\text{Soil water content (g g}^{-1}\text{)} = \frac{\text{wt. moist soil} - \text{wt. oven dry soil (g)}}{\text{wt. oven dry soil (g)}} \tag{4}$$

$$\text{Soil Bulk Denisty (g cm}^{-3}\text{)} = \frac{\text{oven dry weight of soil (g)}}{\text{volume of soil (cm}^{3}\text{)}} \tag{5}$$

$$\text{Volumetric water content (M}^{3}\text{M}^{3}\text{)} = \text{soil water content (M}^{3}\text{) x soil bulk density (Mg M}^{3}\text{)} \tag{6}$$

$$\text{Soil porosity (\%)} = 1 - \frac{\text{soil bulk density (Mg M}^{-3}\text{)}}{\text{2.65 (g cm}^{-3}\text{)}} \tag{7}$$

$$\text{Soil water} - \text{filled pore space (\%)} = \frac{\text{volumetric water content x 100}}{\text{soil porosity}} \tag{8}$$

Infiltration

Infiltration rates were measured using the steady state, ponded infiltration method in each measurement plot by randomly selecting five subplots. Within each subplot, one randomly selected sub-subplot was selected for infiltration measurement using a Turf-Tec International IN7-W Heavy Duty Double Ring Infiltrometer. The Turf-Tec IN7-W has a 15.24cm diameter inner ring and a 30.48cm diameter outer ring. It is 10.16cm tall. The infiltrometer was inserted into the ground to a depth of 6cm by striking with a 1kg rubber mallet a wooden block centrally placed on top of the driving plate, which was placed on top of the infiltrometer to drive it vertically into the ground while taking care to minimize soil disturbance by having the instrument slice uniformly into the soil without fracturing the soil surface and so that it was positioned level to the ground. The inner and outer walls of the outer ring were marked at the 6cm depth level. Pine litter was placed on the ground surface inside the inner and outer rings to dissipate the energy of poured water and to prevent splash or soil dispersion that could adversely affect the infiltration rate. A splash pad was also placed on the ground in the outer chamber and water was poured onto the pad so as not to pour water directly onto the soil surface. The outer chamber was filled to a depth of 3.5cm. The splash pad was then transferred to the inner chamber and the inner chamber was filled with water to a depth of 3.5cm. Once the inner chamber was filled, infiltration was monitored using a stopwatch. Water was added as needed using a 1,000ml graduated cylinder, or smaller to maintain the same water levels in the inner and outer rings and to maintain a constant water elevation in both. Each time water was added, the time interval and amount of water added were recorded. This procedure was continued until a constant infiltration rate was achieved or until 60 minutes had elapsed.

Aggregate Stability

Aggregate stability was determined using the procedure described in the Soil Quality Test Kit Guide (USDA 2001). A 50cm^3 sample of soil was air-dried, sieved though a 2.00mm sieve and then weighed. The sieved soil was homogenized and a 10g subsample taken. The subsample was placed in a pre-weighed 0.25mm sieve and wetted by placing the sieve on a saturated towel for five minutes, allowing the soil to slowly wet. The sample was then wet sieved by placing the sieve containing the soil in a plastic container filled with deionized water and submerging the soil sample. The sample was then gently agitated by moving the sieve up and down approximately 1.5cm in the water at an oscillation rate of approximately 30 times per minute while keeping the sample completely submerged. After agitation (wet sieving) was complete, the sieve was placed in a drying oven and remaining aggregates were dried at 50°C. Once dried, the sieve containing the aggregates was weighed. The sieve with aggregates was then submerged for five minutes in a dispersant solution containing approximately 16g of sodium hexametaphosphate in one liter of water and the sieve containing the sample was gently oscillated to disperse remaining aggregates. This procedure leaves only sand sized particles on the sieve, which were gently washed by running water through the sieve. The sample was again dried at 50°C then weighed. The percentage of water stable aggregates was calculated as follows:

$$\mathrm{H_2O\ stable\ aggregates\ (pct.\ of\ soil > 0.25\ mm)} = \frac{\mathrm{wt. of\ dry\ aggregates - sand}}{\mathrm{wt. of\ dry\ soil - sand}} \times 100 \qquad (9)$$

<u>Slake Test</u>

Soil slake tests were performed using the USDA-ARS Jornada Experimental Range Soil Stability Test Kit (Synergy Resource Solutions, Inc.). Eighteen soil surface aggregates were randomly collected using the spatula provided in the Soil Stability Test Kit. Samples were collected from the same selected sub-subplots within each of the five selected subplots of each measurement plot under grazed and non-grazed conditions.

Each soil surface aggregate was placed in a sieve basket upon collection and allowed to air dry for one hour. The Soil Stability Test Kit case was then filled with water to the top of the cell dividers. Sieve baskets were placed one at a time into a water filled cell and allowed to remain immersed for five minutes. After five minutes had elapsed, the sieve basket was raised until the sample was above the water surface and lowered back to the bottom, taking about one second to complete the oscillation. This was repeated four additional times for a total of five immersions. A soil stability class was then assigned to the sample based on either a) the time required for the aggregate to fully disintegrate upon immersion, or b) the percentage of the soil aggregate remaining on the sieve after five immersion/extraction cycles (Table 3). The process was repeated for all samples.

Table 3. Soil stability classes and associated criteria assigned during soil slake test.

Stability Class	Criteria for assignment to stability class (for "Standard Characterization")
0	Soil too unstable to sample (falls through sieve).
1	50 % of structural integrity lost within five seconds of insertion in water.
2	50 % of structural integrity lost five to 30 seconds after insertion.
3	50 % of structural integrity lost 30 to 300 seconds after insertion or < 10 % of soil remains on the sieve after five dipping cycles.
4	10 – 25% of soil remaining on sieve after five dipping cycles.
5	25 – 75% of soil remaining on sieve after five dipping cycles.
6	75 – 100% of soil remaining on sieve after five dipping cycles.

Vegetation

Terrestrial component surveys, including vegetation were conducted using a 1m x 1m PVC plot frame to visually estimate percent areas of bare ground, rock, and litter cover. Vegetation surveys included complete enumeration and ocular cover estimates for trees, shrubs, forbs, grasses and sedges by species for each of the nine subplots within each measurement plot. Aerial cover of each plant species rooted in each subplot was categorized as either 0.1%, 0.25%, 0.5%, or 1% up to 1% cover, at 1% intervals to 10%, and at 5% intervals above 10% cover. Warm season vegetation data collected in 2015 was compared to vegetation data collected by Abella et al. (2015), who completed pre-treatment (2003) and post-treatment (2006 and 2008) vegetation surveys.

Statistical Analysis

This study captured ecological data that are hierarchically structured based on soil parent material type (three types: limestone, benmoreite, or basalt), ecological restoration treatment type (nested within soil type and having 3 categories: unthinned control, thinned, or existing open condition), grazing treatment (nested within soil and treatment type and with two categories: grazed or excluded from grazing), and time (with four measurement years: 2003 pre-treatment, 2006, 2008, and 2015 representing 3, 5, and 12 years post-treatment). A hierarchical linear model (HLM) was used to analyze treatment effects on measured soil quality indicators, mean percent total understory vegetative cover, mean percent cover of understory vegetation by lifeform (i.e., grasses and forbs), mean percent cover by longevity type (perennials and annuals), and mean percent cover of exotic plant species. Mean total species richness, mean species richness of grasses, forbs, perennials, annuals, biennials, and exotic plant species per subplot and per measurement plot were analyzed based on responses to blocking variables of soil type, forest treatment type, grazing vs. non-grazing, and time. Similar to Abella et al. (2015), 72 combinations (3 soil types x 3 treatment types x 2 grazing treatments x 4 measurement years) were used in the HLM. These combinations were tested, at $\alpha = 0.05$ level, to evaluate differences and to estimate how the effects of the treatment combinations might have affected measured soil quality indicators and vegetative cover dynamics.

Hierarchical linear models use advanced estimation algorithms to measure regression relationships and variance-covariance parameters in hierarchically structured data (McMahon and Diaz, 2007). HLM employs nested regression equations to investigate relationships between variables at different scales. In addition to reducing the potential for Type I errors, another advantage of using HLM is it allows the user to effectively separate within-group effects from between-group effects whenever there is a significant variance in coefficients across groups (Huta, 2014). The hierarchical strategy allows investigation of the incremental increase in variation that each model predictor introduces to the overall model. This is evaluated by assessing the change in R-squared values as each nested model is introduced stepwise. If the R-squared change is greater than zero for any added independent variable, it is inferred that the added element increases the predictive power of the HLM model.

In addition to the HLM approach, a partial, nested mixed model analysis of variance (ANOVA) was used, which included the four factors of soil type, forest treatment type, grazing treatment, and year, and all interactions. To limit the numerous multiple comparisons that result from possible four-way interaction (3 soil types × 3 treatment types × 2 grazing treatments × 4 years) and to identify which treatment combinations had significant effects on the dependent variable, Tukey's Honestly Significant Difference (HSD) for estimated marginal means and Bonferroni adjustment for multiple comparisons were used to conduct post-hoc pairwise tests to compare pre- and post-treatment least-squares means within treatment combinations.

RESULTS

Study Area

In the 20 years preceding this study (1994-2014), Flagstaff and the surrounding area received mean annual precipitation of 47.2cm. Approximately 22cm was as rainfall and 25.2cm was as snow. The mean annual winter low temperature was -3.6ºC and the mean annual winter high temperature was 2.14ºC. The mean annual summer low temperature was 12.76ºC and the mean annual summer high temperature was 18.2ºC. During the year of this study (2015), winter snowfall was approximately 30.22cm, and monsoon precipitation was 8.4cm through September. Average temperatures in the Flagstaff area have been rising since the mid-1980s. Most years since 1985 have had average annual temperatures above the long-term average. Precipitation patterns have remained variable, with no discernable trend. Rising temperatures are expected to increase evapotranspiration rates, leading to drier soils and increasing the frequency, duration, and severity of drought (Meadow et al., 2018). Additionally, rising temperatures have increased the amount of precipitation as rainfall during cold months and this trend is expected to continue. Mean monthly precipitation and temperature data for the Flagstaff Pulliam Airport (Flagstaff, WSO, AP (023010)) are provided in Appendix A.

Restoration and Grazing Treatment Effects to Soil Physical Properties

Soil Bulk Density

There were no statistically significant differences in soil bulk density by soil type, forest treatment type, or grazing treatment (Table 4).

Table 4. Statistical comparisons of soil bulk density measurements ($\alpha = 0.05$).

Fixed effect	Coeff.	Std. Err.	t-ratio	p-value
Intercept 1, $\beta 0$				
Intercept 2, $\gamma 00$	1.257	0.033	38.598	<0.001
Open, $\gamma 01$	0.049	0.037	1.320	0.202
Thinned, $\gamma 02$	0.009	0.037	0.237	0.815
Basalt, $\gamma 03$	-0.017	0.039	-0.434	0.669
Benmoreite, $\gamma 04$	-0.010	0.035	-0.274	0.787
Grazed slope, $\beta 1$				
Intercept 2, $\gamma 10$	0.096	0.060	1.619	0.122
Open, $\gamma 11$	-0.042	0.068	-0.616	0.545
Thinned, $\gamma 12$	0.008	0.068	0.115	0.910
Basalt, $\gamma 13$	0.058	0.071	0.814	0.426
Benmoreite, $\gamma 14$	0.011	0.064	0.174	0.864

Saturated Infiltration Capacity

Saturated infiltration capacity was affected by soil type (Table 5, Figure 3). Soils derived from basalt and benmoreite had significantly lower saturated infiltration capacity than limestone derived soils. Predicted saturated infiltration capacity for basalt and benmoreite-derived soils is 1.96 and 1.84 mm hr^{-1} lower than limestone-derived soils, respectively. No statistically significant differences in saturated infiltration capacity were observed among forest treatment types under either grazing or non-grazing treatments (Table 5).

Table 5. Statistical comparisons of saturated infiltration capacity (mm hr^{-1}) ($\alpha = 0.05$).

Fixed effect	Coeff.	Std. Err.	t-ratio	p-value
Intercept 1, β0				
Intercept 2, γ00	13.39	0.3946	33.43	<0.001
Open, γ01	0.14	0.4474	0.307	0.762
Thinned, γ02	0.25	0.4474	0.558	0.583
Basalt, γ03	-1.95	0.4716	-4.147	<0.001
Benmoreite, γ04	-1.84	0.4218	-4.373	<0.001
Grazed slope, β1				
Intercept 2, γ10	-1.11	0.6023	-1.845	0.081
Open, γ11	0.02	0.6830	0.037	0.971
Thinned, γ12	-0.33	0.6830	-0.476	0.640
Basalt, γ13	-0.96	0.7199	-1.327	0.200
Benmoreite, γ14	-0.26	0.6439	-0.414	0.683

Aggregate Stability

There were no statistically significant differences in aggregate stability among soil types or forest treatments under non-grazing (Table 6). However, grazing had a significant effect on aggregate stability, with predicted aggregate stability being significantly lower under grazing than non-grazing (Figure 4).

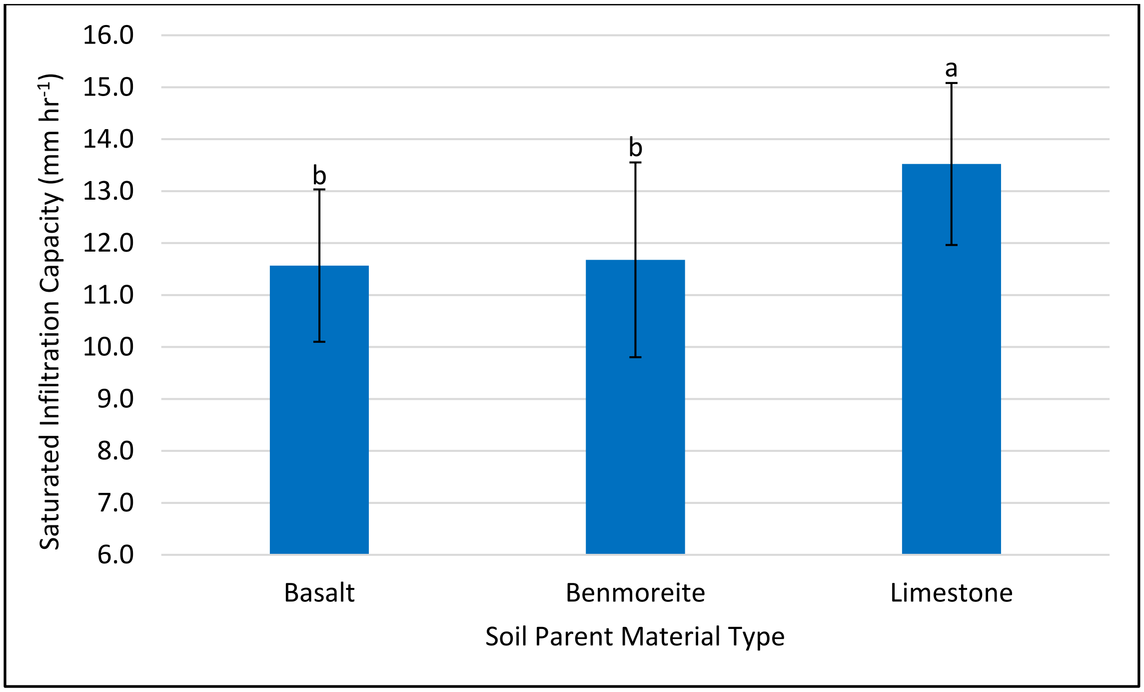

Figure 3. Comparison of means of saturated infiltration capacity (mm hr^{-1}) of soils derived from 3 parent materials. Bars with the same letter are not significantly different ($\alpha = 0.05$). Each bar represents one standard deviation from the mean.

Table 6. Statistical comparisons of aggregate stability ($\alpha = 0.05$).

Fixed effect	Coeff.	Std. Err.	t-ratio	p-value
Intercept 1, $\beta 0$				
Intercept 2, $\gamma 00$	34.12	2.1902	15.577	<0.001
Open, $\gamma 01$	1.59	2.4835	0.639	0.530
Thinned, $\gamma 02$	-2.61	2.4835	-1.052	0.306
Basalt, $\gamma 03$	-4.39	2.6179	-1.677	0.110
Benmoreite, $\gamma 04$	-1.99	2.3415	-0.848	0.407
Grazed slope, $\beta 1$				
Intercept 2, $\gamma 10$	-21.77	2.6216	-8.305	<0.001
Open, $\gamma 11$	5.16	2.9726	1.737	0.099
Thinned, $\gamma 12$	5.71	2.9726	1.921	0.070
Basalt, $\gamma 13$	5.11	3.1334	1.629	0.120
Benmoreite, $\gamma 14$	2.30	2.8026	0.822	0.421

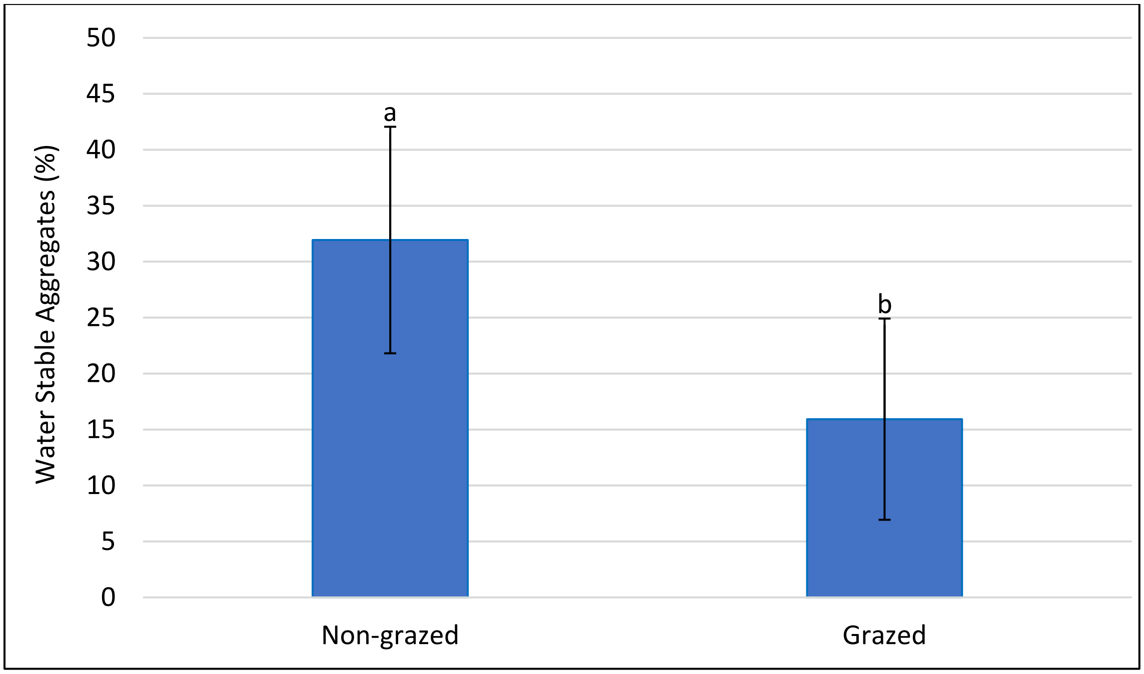

Figure 4. Comparison of means of percentage of water stable aggregates under grazed and non-grazed conditions. Each bar represents one standard deviation from the mean. Bars with different letters are significantly different (α = 0.05).

Soil Slaking

Soil slaking potential was affected by soil type (Table 7, Figure 5). Limestone- and benmoreite-derived soils had the highest average slake ratings but were not significantly different, while basalt-derived soils had the lowest average soil slake rating.

Open canopy conditions that were excluded from grazing had greater resistance to slaking than closed canopy conditions (Figure 6). In the absence of grazing, soil aggregate stability and resistance to slaking increased by 46 percent of one soil stability class under open canopy conditions in comparison to closed control.

Table 7. Statistical comparisons of soil slaking potential (α = 0.05).

Fixed effect	Coeff.	Std. Err.	t-ratio	p-value
Intercept 1, β0				
Intercept 2, γ00	3.09	0.1254	24.644	<0.001
Open, γ01	0.46	0.1422	3.252	0.004
Thinned, γ02	0.07	0.1422	0.457	0.653
Basalt, γ03	-0.56	0.1499	-3.758	0.001
Benmoreite, γ04	-0.24	0.1341	-1.790	0.089
Grazed slope, β1				
Intercept 2, γ10	-0.69	0.1138	-6.133	<0.001
Open, γ11	-0.49	0.1291	-3.816	0.001
Thinned, γ12	-0.08	0.1291	-0.620	0.543
Basalt, γ13	0.41	0.1360	2.981	0.008
Benmoreite, γ14	0.18	0.1217	1.461	0.160

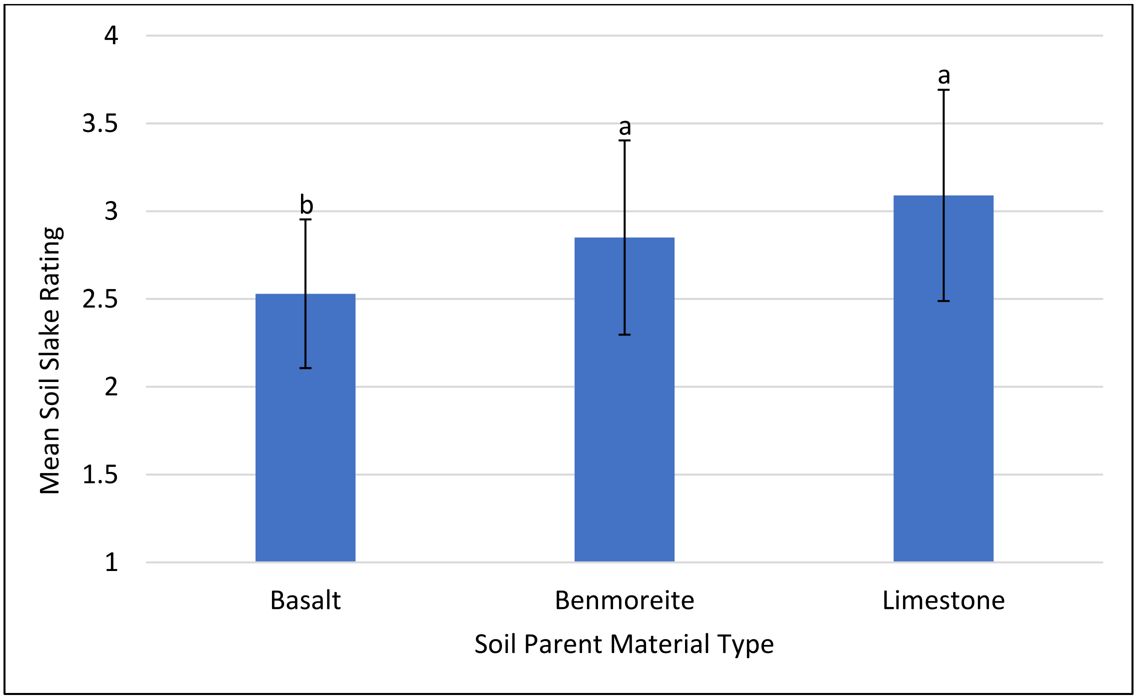

Figure 5. Mean soil slake ratings by soil parent material type. Error bars represent one standard deviation from the mean. Bars with the same letter are not significantly different at the α = 0.05 level.

Grazing had a significant effect on soil slaking, with an average soil slake rating that was almost 70 percent of one soil stability class lower than non-grazing (Table 7, Figure 7). Soil slaking potential under grazing was lower under open canopy conditions than under closed canopy conditions. Soil slaking potential under non-grazing on basalt-derived soils was greater than on limestone-derived soils (Figure 8). All soils tested had strong potential to slake when rapidly wetted, regardless of forest treatment type or grazing treatment. The highest average soil stability class was 3, which indicates that 50 percent of structural integrity of aggregates are lost 30 to 300 seconds after immersion or less than 10 percent of the soil aggregate remains on sieves after 5 immersion cycles.

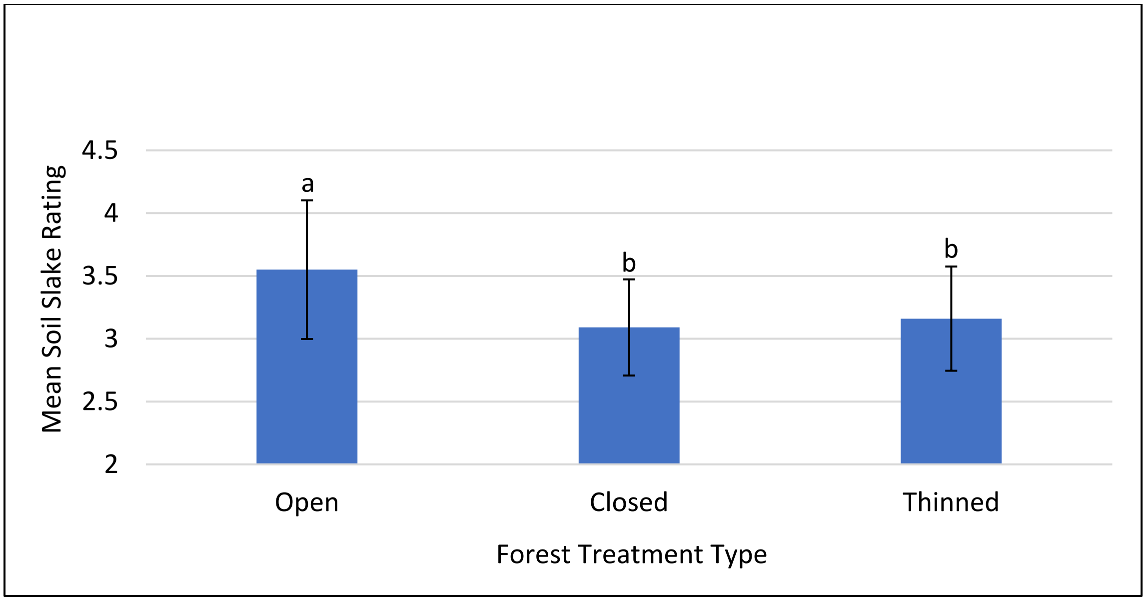

Figure 6. Mean soil slake ratings by forest treatment type under grazing exclusion. Error bars represent one standard deviation from the mean. Bars with the same letter are not significantly different at the α = 0.05 level.

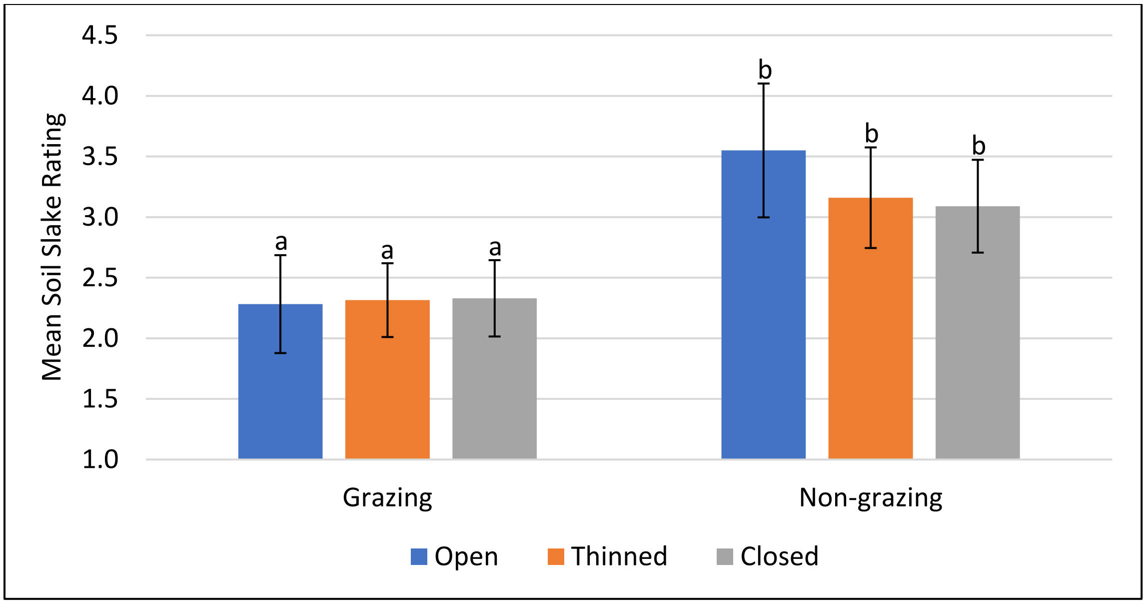

Figure 7. Mean soil slake ratings by forest treatment type under grazing and non-grazing. Errors bars represent one standard deviation from the mean. Bars with the same letter within grazing treatment are not significantly different at the α = 0.05 level.

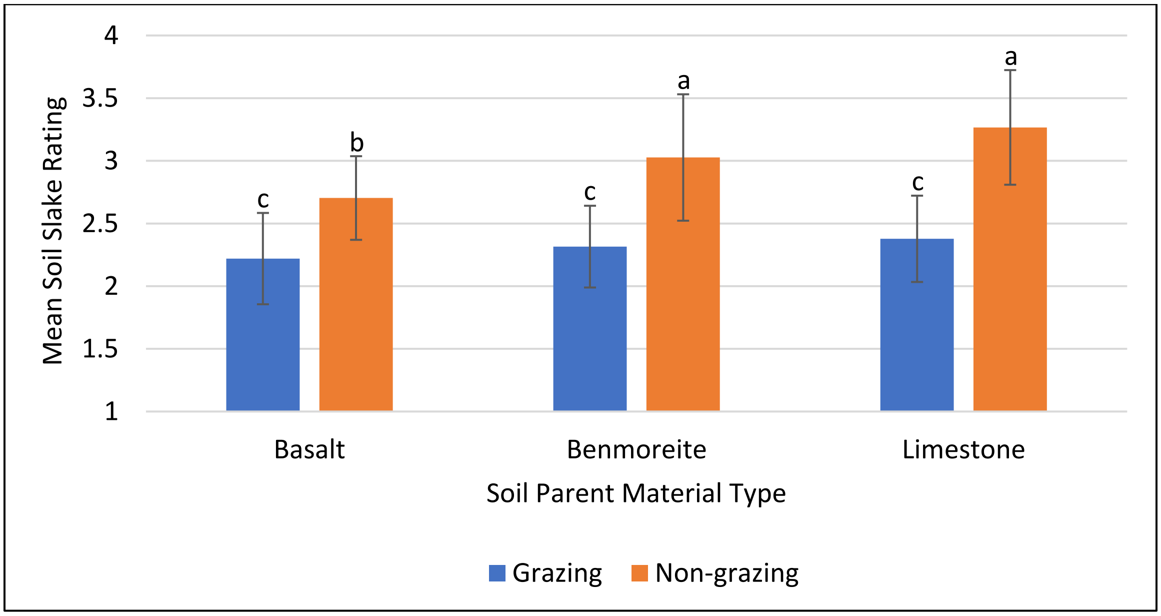

Figure 8. Means soil slake ratings by soil parent material types under grazing and non-grazing. Each bar represents one standard deviation from the mean. Bars with different letters within each soil parent material type are significantly different at the α = 0.05 level.

Restoration and Grazing Treatment Effects to Soil Chemical Properties

Soil Carbon, Nitrogen and Organic Matter

There were no statistically significant differences in soil organic C or N levels (Mg ha^{-1}) in the upper 10cm of soil by soil parent material type, forest treatment type, or grazing treatment. Mean soil organic carbon levels ranged from 43.01Mg ha^{-1} (SD 16.67) in limestone-derived soil under closed canopy conditions with grazing excluded to 142.3 Mg ha^{-1} (SD 140.3) in benmoreite-derived soil under closed canopy conditions with grazing.

There were no statistically significant differences in soil organic matter (OM) levels (Mg ha^{-1}) by soil type, forest treatment type, or grazing treatment. Mean soil OM content ranged from 6.49 Mg ha^{-1} (SD 2.03) in limestone-derived soils under closed canopy conditions with no grazing to 20.99 Mg ha^{-1} (SD 20.82) in benmoreite-derived soils under closed canopy conditions with grazing.

Other Soil Chemical Properties

There were no significant differences in soil calcium (Ca^{2+}) or potassium (K+) levels on an areal basis (Mg ha^{-1}) among soil type, forest treatment type, or grazing treatments (Table B 1). There were significant differences in soil magnesium (Mg^{2+}) content (Mg ha^{-1}) by soil type (Table B 1). Basalt-derived soils had a predicted mean soil Mg^{2+} content of 655.1 Mg ha^{-1} while benmoreite- and limestone-derived soils had predicted mean soil Mg^{2+} contents of 423.4 Mg ha^{-1} and 445.7 Mg ha^{-1}, respectively (Figure 9).

There were no statistically significant differences in mean soil Mg^{2+} content by forest treatment type or grazing treatment (Table B 1).

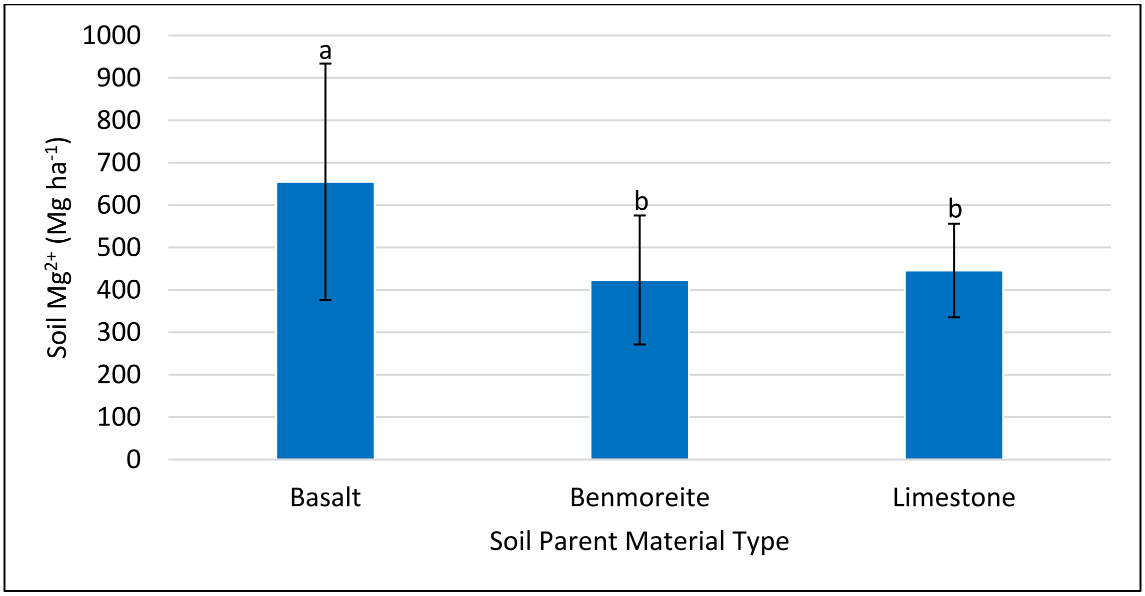

Figure 9. Mean soil Mg^{2+} levels by soil parent material type. Error bars represent one standard deviation from the mean. Bars with the same letter are not significantly different at the $\alpha = 0.05$ level.

Soil sodium (Na^+) levels (Mg ha^{-1}) were not significantly different by soil type or forest treatment type (Figure 10, Table B 1). However, grazing affected soil Na^+, with grazing having an average of 5.04 Mg ha^{-1} less Na^+ than non-grazed conditions across all soil types and ecological restoration treatments.

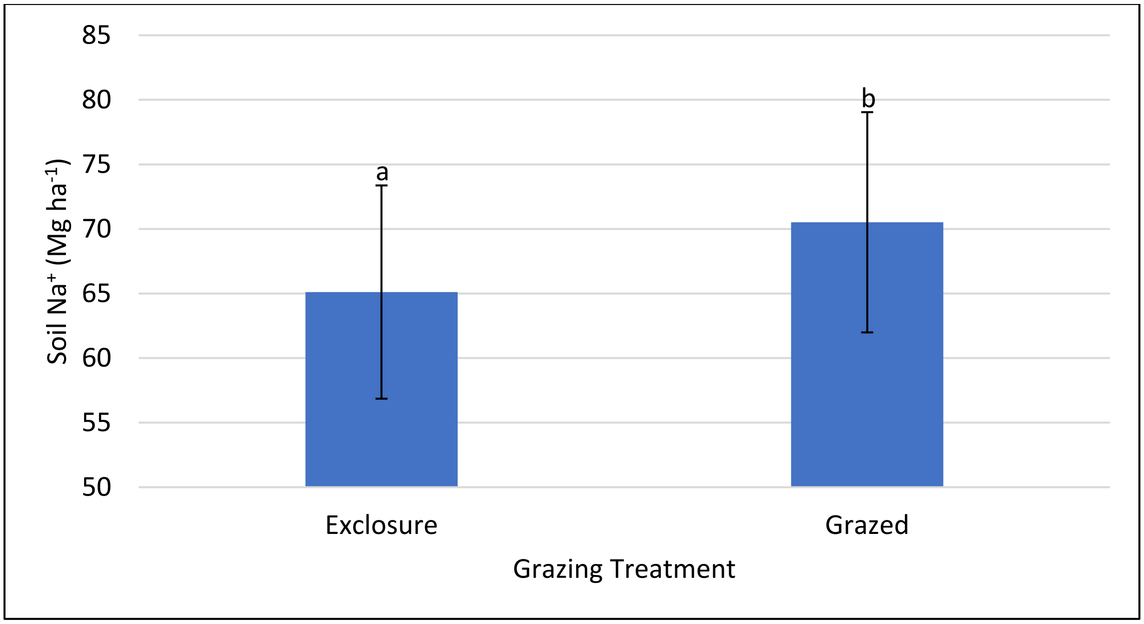

Figure 10. Mean soil Na$^+$ levels (Mg ha^{-1}) by grazing treatment. Error bars represent one standard deviation from the mean. Bars with different letters are statistically significantly different at the $\alpha = 0.05$ level.

Significant differences were found in soil phosphorus (P) levels (Mg ha^{-1}) between basalt and benmoreite-derived soils (Figure 11, Table B 1). Basalt-derived soils had significantly lower soil P levels than benmoreite-derived soils ($p = 0.025$). There were no statistically significant differences in soil P levels among forest treatment types or grazing treatment. Soil P levels in limestone-derived soils were not significantly different from those of benmoreite-derived soils.

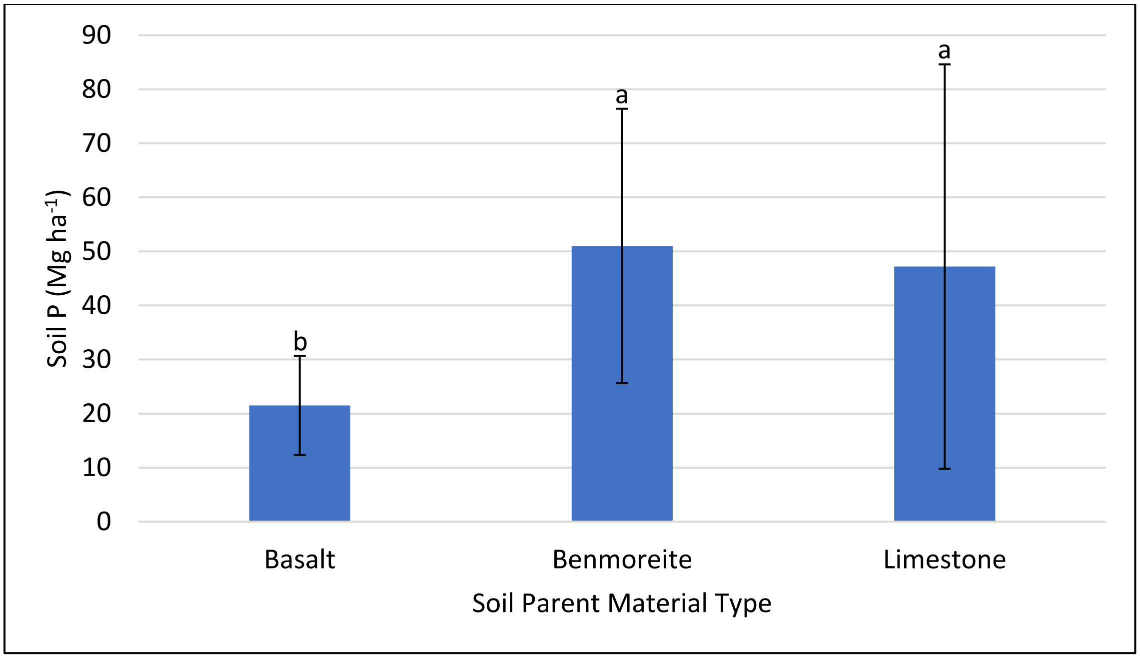

Figure 11. Means soil P levels (Mg ha^{-1}) by soil parent material type. Error bars represent one standard deviation from the mean. Bars with the same letter are not significantly different at the $\alpha = 0.05$ level.

Soil sulfur (S) levels (Mg ha^{-1}) were significantly different among soil types. Limestone- and benmoreite-derived soils had significantly greater soil S than basalt-derived soils ($p = 0.003$, and $p = 0.048$, respectively) (Figure 12, Table B 1).

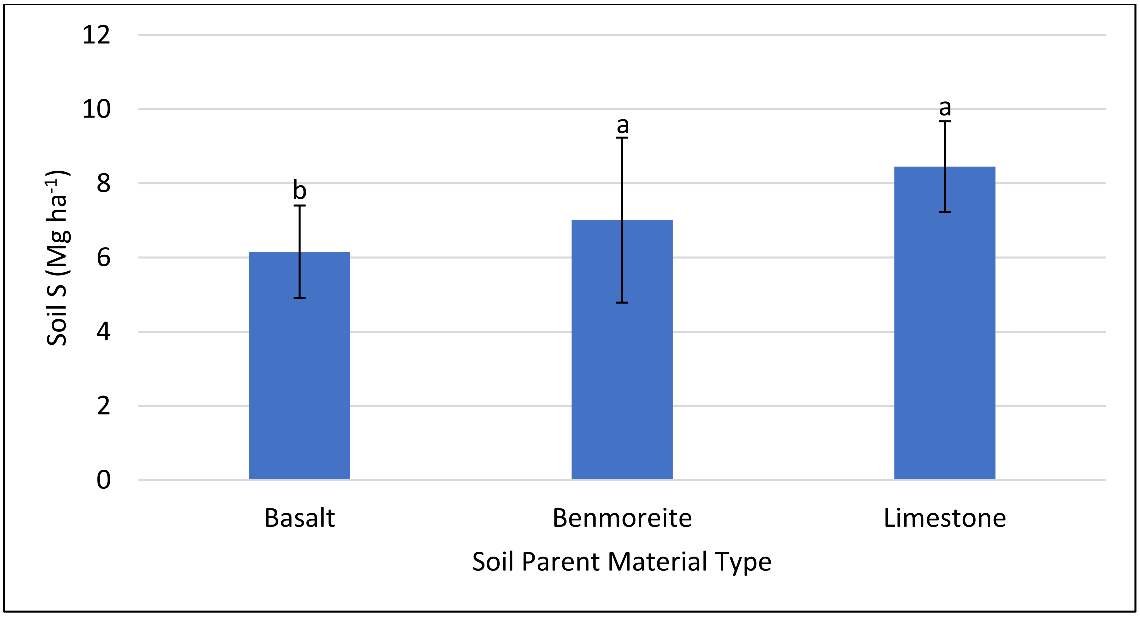

Figure 12. Mean soil S levels (Mg ha^{-1}) by soil parent material type. Error bars represent one standard deviation from the mean. Bars with the same letter are not significantly different at the $\alpha = 0.05$ level.

There were no statistically significant differences in soil S levels between forest treatment types under grazed or non-grazed conditions.

Understory Vegetative Cover

A total of 126 species were recorded across all sites and plots in 2015, with 113 (90 percent) being native and 13 (10 percent) being exotic. Growth form distribution was as follows: 95 forb species (75 percent), 22 grass species (18 percent), 4 sedge species (3 percent), 3 shrub species (2 percent), and 2 tree species (2 percent). This growth form distribution corresponds with the findings of Abella et al. (2015). While the authors recorded a greater number of species (146), percentages of each growth form are similar to this study. Abella et al. (2015) found 90 percent of the species observed were native, 76 percent were forbs, 17 percent were grasses, 3 percent were sedges, 3 percent were shrubs, and 1 percent were trees.

Most plant species exhibit perennial growth duration (86 species, or 68 percent). There were 33 plant species that have annual growth duration (27 percent) and 7 plant species (5 percent) that had other growth durations (biennial, annual-perennial, or annual-biennial). This study found a slightly lower percentage of plants that exhibit perennial growth patterns than Abella et al. (2015), who found 72 percent of the species in their study exhibited perennial growth. This study also found a slightly higher percentage of plants that exhibit annual and other growth durations than Abella et al. (2015).

The most prevalent species recorded during the 2015 warm season sampling period (August through October), occupying at least 50 percent of the 24 plots, included *Elymus elymoides* (Raf.) Swezey (100 percent of plots), *Festuca arizonica* Vasey (83 percent), *Vicia americana* Muhl. Ex Willd. (71 percent), *Poa fendleriana* (Steud.) Vasey (67 percent*), Carex geophila* Mack. (67 percent), *Pinus ponderosa* Lawson & C. Lawson var. scopulorum Engelm. seedlings (63 percent), *Verbascum thapsus* (L.), an invasive species (63 percent), *Muhlenbergia montana* (Nutt.) Hitchc. (58 percent), and *Erigeron flagellaris* A. Gray (54 percent).

Mean Percent Total Vegetative Cover

Table 8 displays the R-squared values and associated changes in R-squared values from each added predictor to the HLM of subplot level mean percent total vegetative cover for the 2015 measurement year.

Table 8. Hierarchical Linear Model (HLM) predictors and their associated R, R-squared, adjusted R-squared, and change in R-squared values for each independent variable used to analyze mean percent total vegetative cover per $1m^2$ for the 2015 measurement year ($\alpha = 0.05$).

					Change Statistics		
Model	R	R-squared	Adjusted R-squared	Std. Error of the Estimate	R-squared Change	F Change	Sig. F Change
1	0.090[a]	0.008	0.006	24.587	0.008	3.453	0.064
2	0.113[b]	0.013	0.008	24.557	0.005	2.029	0.155
3	0.413[c]	0.170	0.165	22.538	0.158	79.802	<0.001

[a] Treatment type

[b] Treatment type plus grazing treatment

[c] Treatment type plus grazing treatment plus soil type

As shown in Table 8, treatment type and grazing treatment did not contribute significantly to the power of the HLM of mean percent total vegetative cover in 2015, although mean percent total vegetative cover by soil parent material type was significant.

While not a significant predictor of mean percent total vegetative cover in the regression model, pairwise comparisons using Bonferroni adjustment for multiple comparisons indicated forest treatment type affected mean percent total vegetative cover in some instances. Mean percent total vegetative cover under forest thinning and open canopy conditions was not significantly different ($p = 0.157$). However, both were significantly greater than the mean percent total vegetative cover under closed canopy conditions ($p < 0.001$ and $p = 0.022$, respectively). Mean percent total vegetative cover under forest thinning and open canopy conditions was 31.23 (SD 28.14) and 26.21 (SD 20.48), respectively, while under closed canopy conditions it was 20.70 (SD 23.80) (Figure 13).

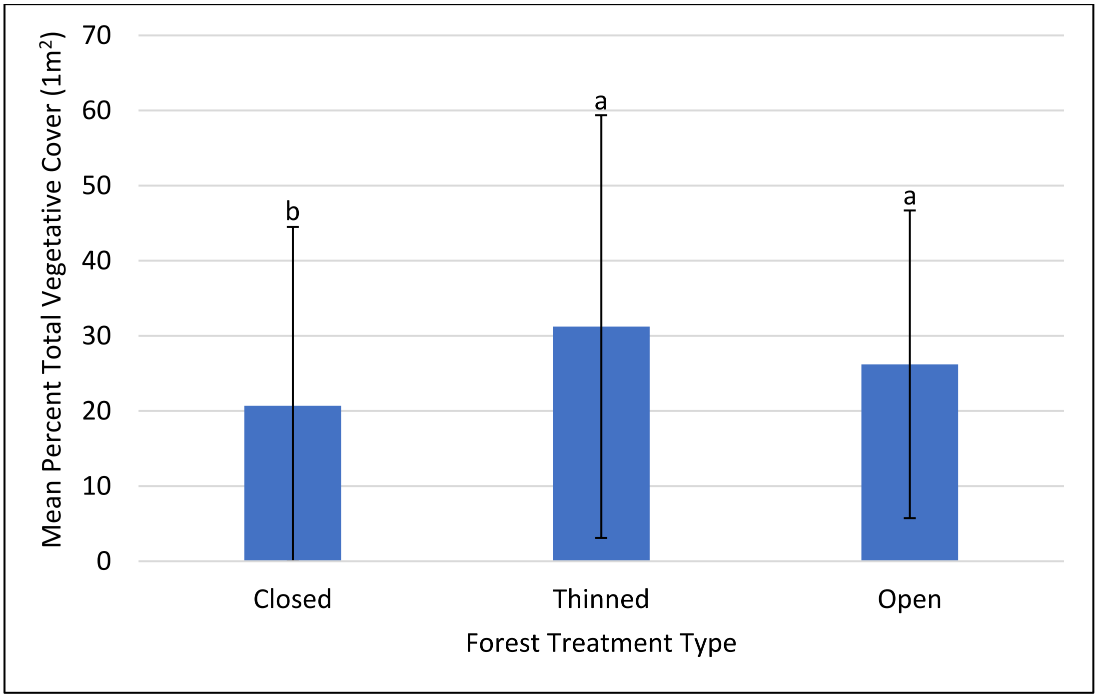

Figure 13. Mean percent total vegetative cover at the subplot level by forest treatment type in 2015. Error bars represent the standard error of the mean. Bars with the same letter are not significantly different at the $\alpha = 0.05$ level.

Thinning on limestone-derived soils had greater mean percent total vegetative cover than closed ($p = 0.029$) and open ($p < 0.001$) canopy conditions (Figure 14). Mean percent total vegetative cover on limestone-derived soils under forest thinning was 54.58 (SD 30.76), while under closed and open canopy conditions it was 34.24 (SD 30.36) and 26.40 (SD 23.80), respectively.

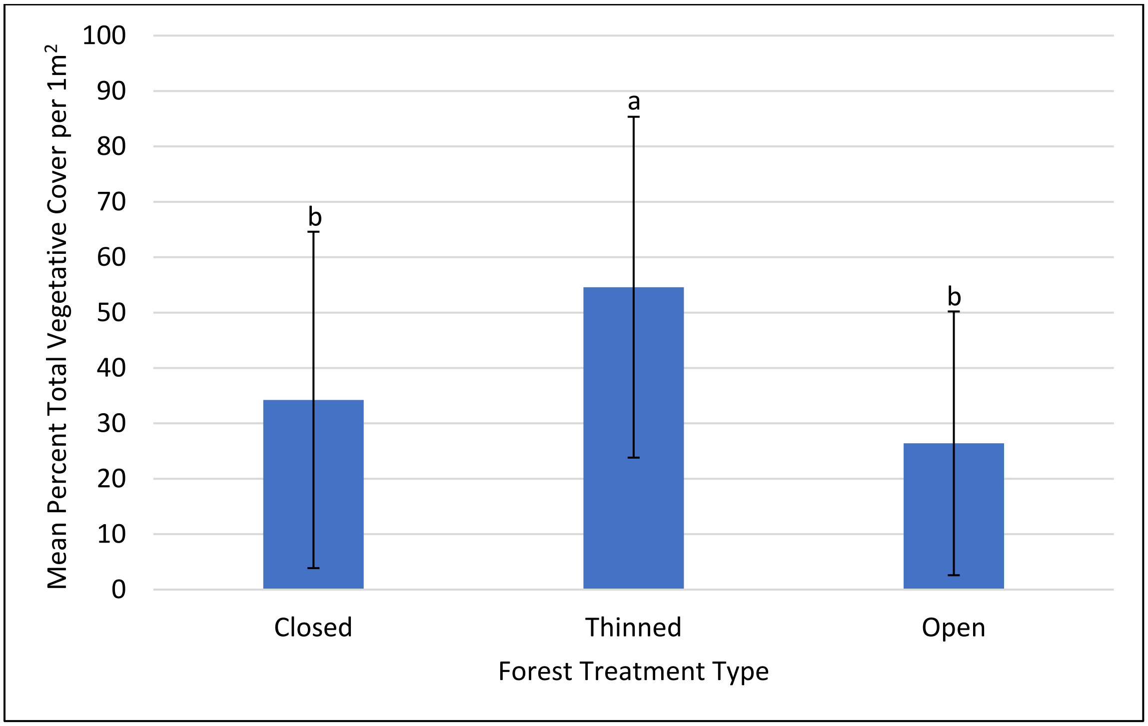

Figure 14. Mean percent total vegetative cover at the subplot level by forest thinning treatments on limestone-derived soils in 2015. Error bars represent one standard deviation from the mean. Bars with the same letter are not significantly different at the α = 0.05 level.

Mean percent total vegetative cover under forest thinning on basalt- and benmoreite-derived soils was not significantly different. However, both were lower than on limestone-derived soils ($p < 0.001$) (Figure 15). Under forest thinning, mean percent total vegetative cover was 12.64 (SD = 7.55) for basalt-derived soils, 19.88 (SD = 14.62) for benmoreite-derived soils, and 54.58 (SD = 30.76) for limestone-derived soils. No significant difference in mean percent total vegetative cover was found between grazed and non-grazed treatments (p = 0.17).

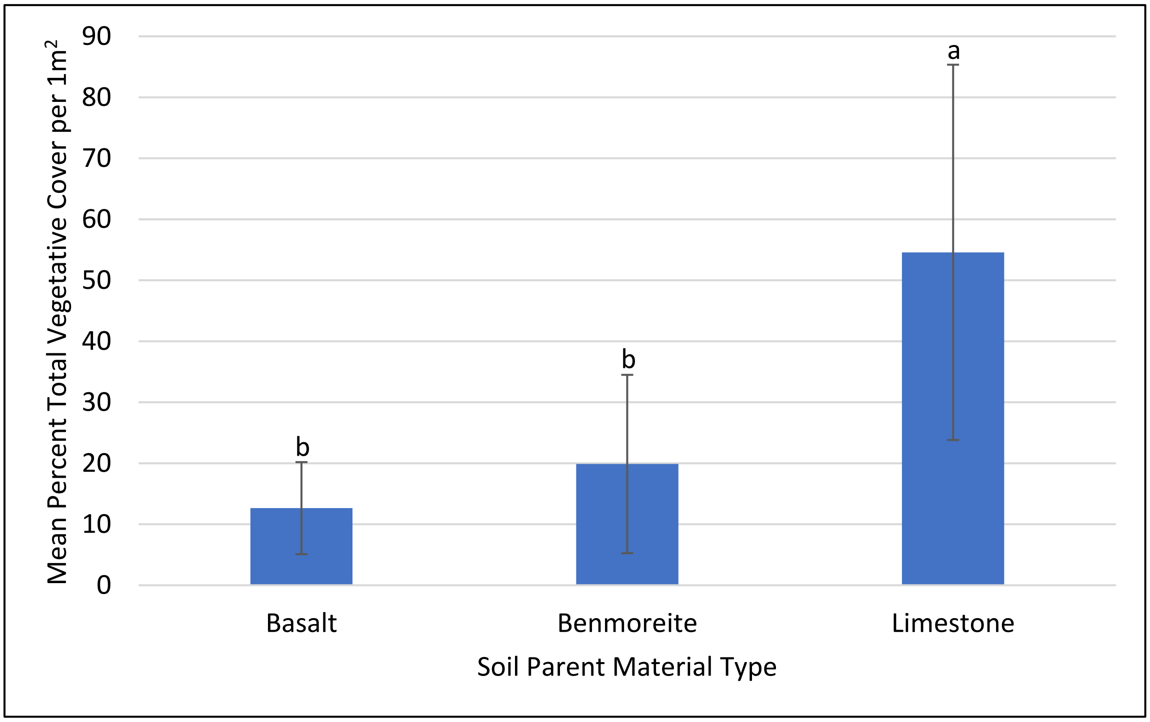

Figure 15. Mean percent total vegetative cover at the subplot level by soil parent material type under forest thinning in 2015. Error bars represent one standard deviation from the mean. Bars with the same letter are not significantly different at the α = 0.05 level.

Mean percent total vegetative cover in 2015 was compared to 2003, 2006, 2008, and 2015 to evaluate long term responses of understory vegetative cover to forest thinning and grazing treatments (Table 9).

Table 9. Hierarchical Linear Model (HLM) predictors and their associated R, R-squared, adjusted R-squared, and change in R-squared values for each independent variables used to analyze mean percent total vegetative cover per 1m^2 for all measurement years (α = 0.05).

					Change Statistics		
Model	R	R-squared	Adjusted R-squared	Std. Error of the Estimate	R-squared Change	F Change	Sig. F Change
1	0.158[a]	0.025	0.025	17.053	0.025	42.612	<0.001
2	0.164[b]	0.027	0.026	17.041	0.002	3.254	0.071
3	0.322[c]	0.104	0.102	16.361	0.077	141.415	<0.001
4	0.510[d]	0.260	0.258	14.872	0.156	348.456	<0.001

[a] Treatment type

[b] Treatment type plus grazing treatment

[c] Treatment type plus grazing treatment plus soil type

[d] Treatment type plus grazing treatment plus soil type plus year

Treatment type, soil parent material type, and measurement year contributed significantly to the power of the HLM of mean percent total vegetative cover. Grazing treatments were again not a significant contributor to the predictive power of the model. Mean percent total vegetative cover under all forest treatment types was significantly greater in 2015 ($p < 0.001$) than under all forest treatment types in all previous measurement years (Table 10).

Table 10. Means and standard deviations of percent vegetative cover by forest treatment type for all measurement years.

	Year			
Treatment	2003	2006	2008	2015
Closed	6.32 (6.93)	7.86 (10.01)	7.70 (11.20)	20.70 (23.80)
Thinned	3.87 (4.62)	14.76 (12.98)	15.14 (13.03)	31.24 (28.14)
Open	12.24 (8.76)	15.68 (12.28)	16.65 (13.30)	26.22 (20.48)

Under closed canopy conditions, mean percent total vegetative cover on limestone-derived soils was significantly greater than on benmoreite- or basalt-derived soils for all measurement years (Figure 16). However, mean percent total vegetative cover on limestone-derived soils under closed canopy conditions in 2006 was not significantly different from pre-treatment (2003) values.

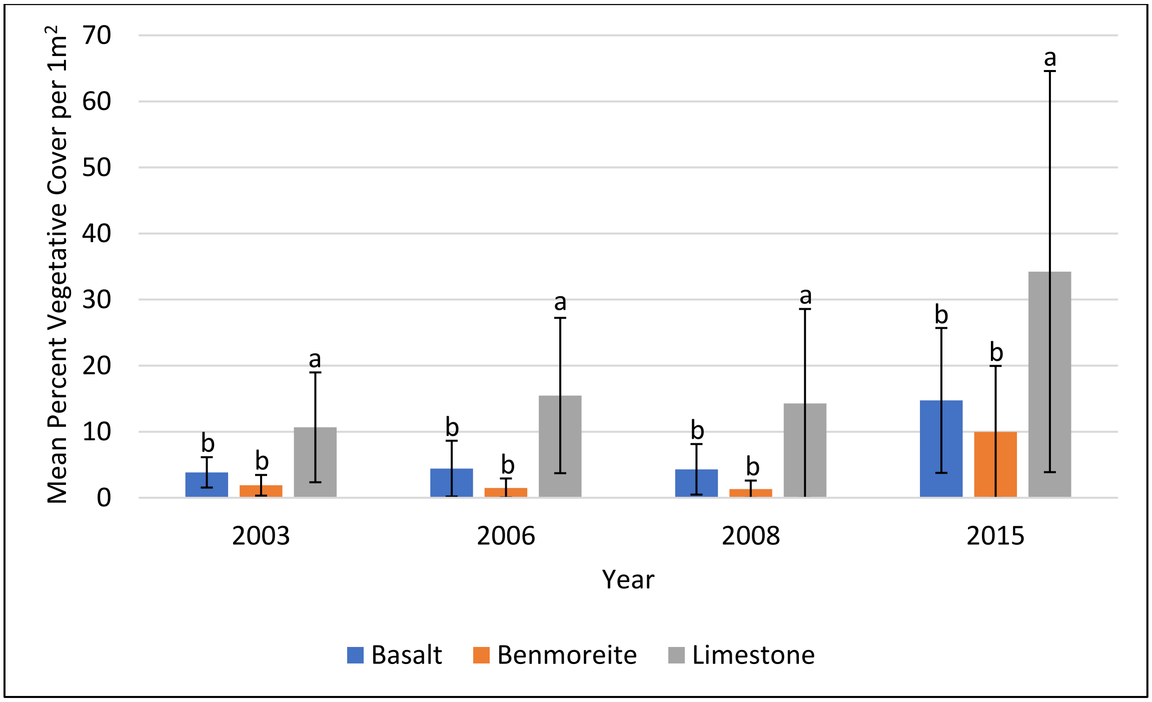

Figure 16. Percent vegetative cover by soil parent material type under closed canopy conditions by measurement year. Error bars represent one standard deviation from the mean. Bars with the same letter within measurement year are not significantly different at the α = 0.05 level.

Under open canopy conditions, mean percent total vegetative cover varied based on soil parent material type. On basalt-derived soils, mean percent total vegetative cover was not significantly different among measurement years ($p = 0.10$). However, in 2015 benmoreite-derived soils had significantly greater mean percent total vegetative cover than all previous measurement years ($p < 0.001$) (Figure 17).

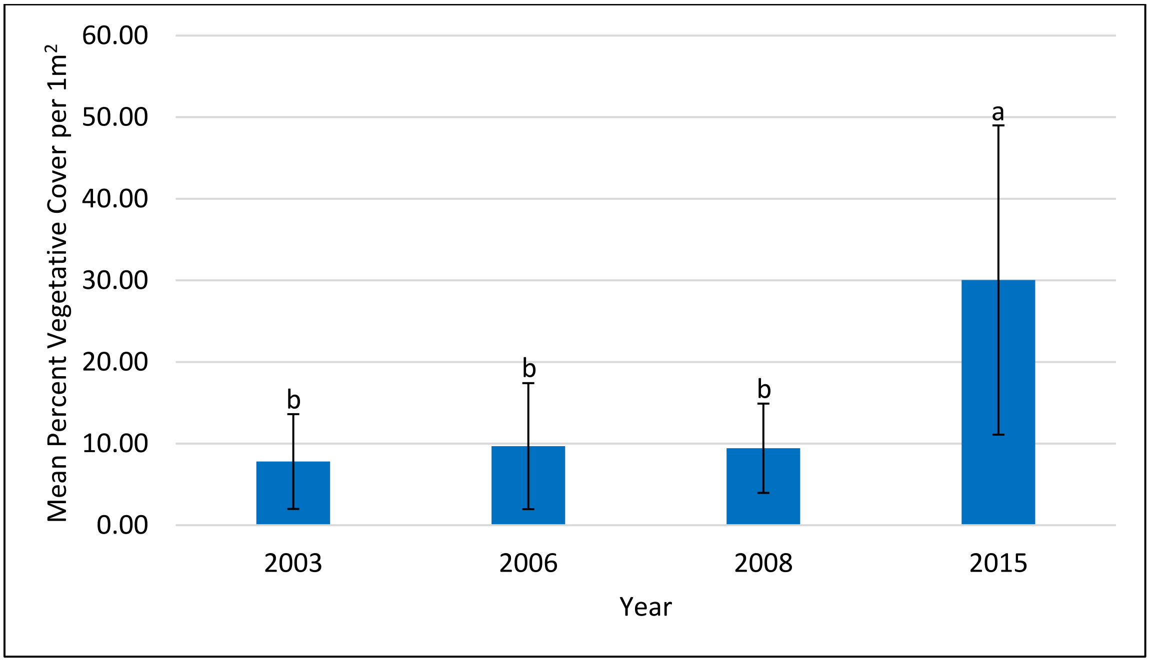

Figure 17. Mean percent total vegetative cover by measurement year under open canopy conditions on soils derived from benmoreite parent material. Error bars represent one standard deviation from the mean. Bars with the same letter are not significantly different at the $\alpha = 0.05$ level.

Mean percent total vegetative cover under open canopy conditions on limestone-derived soils also varied among measurement years. Mean percent total vegetative cover in 2015 was not significantly different from 2008 ($p = 0.118$) but was significantly greater than the 2003 and 2006 measurement years ($p < 0.001$ and $p = 0.041$, respectively).

Under forest thinning, mean percent total vegetative cover was significantly greater in 2015 than all previous measurement years ($p < 0.001$) (Figure 18).

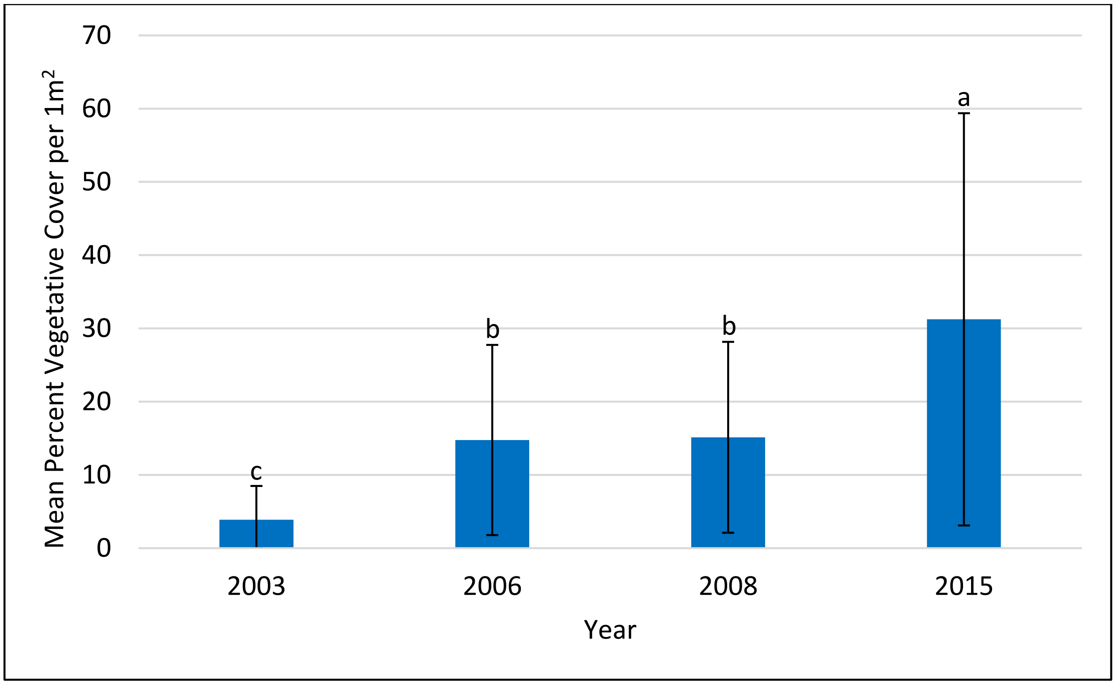

Figure 18. Mean percent total vegetative cover by measurement year under forest thinning. Error bars represent one standard deviation from the mean. Bars with the same letter are not significantly different at the $\alpha = 0.05$ level.

The 2003 measurement year had significantly lower mean percent total vegetative cover than subsequent measurement years ($p < 0.001$). Mean percent total vegetative cover was 3.84 (SD 3.39) in 2003, 14.76 (SD 9.50) in 2006, 15.14 (SD 9.17) in 2008, and 31.01 (SD 25.53) in 2015, indicating a substantial increase in mean percent total vegetative cover since 2008. There was no significant difference in mean percent total vegetative cover between the 2006 and 2008 measurement years ($p = 0.751$).

Forest thinning on limestone-derived soils resulted in significantly greater mean percent total vegetative cover in 2015 ($p < 0.001$) at 54.58 (SD 30.76) in comparison to 2003 at 6.09 (SD 6.14), 2006 at 17.91 (SD 13.48) and 2008 at 15.56 (SD 12.22). There was no significant difference in mean percent total vegetative cover between the 2006 and 2008 measurement years (Figure 19).

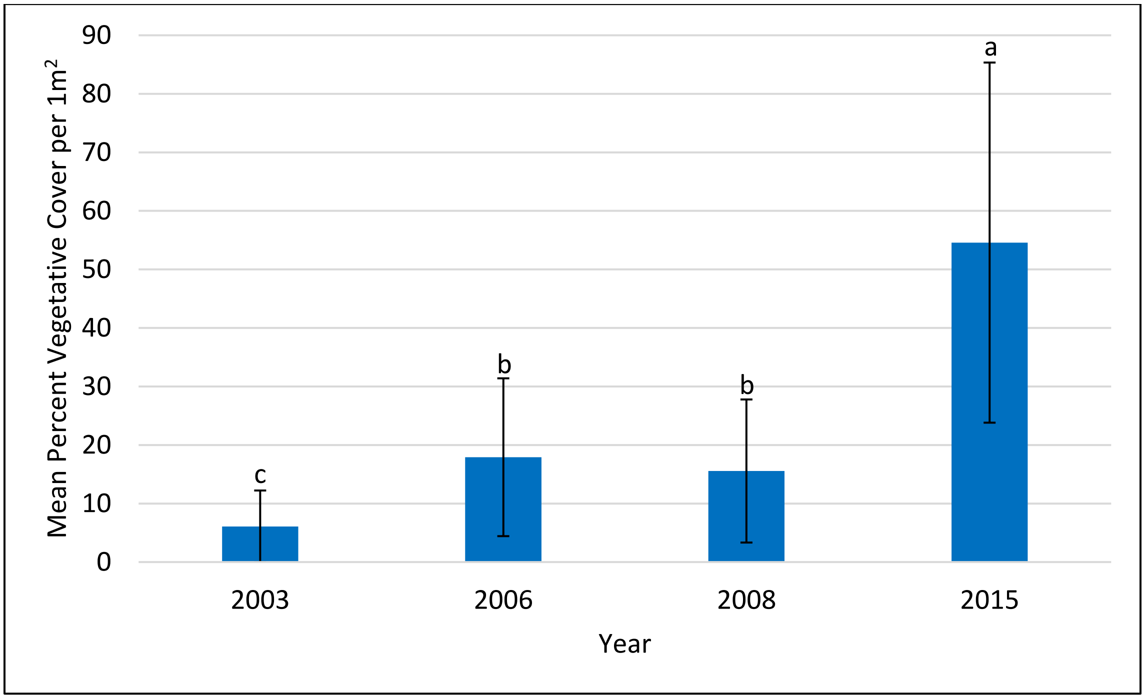

Figure 19. Mean percent total vegetative cover by measurement year under forest thinning on limestone-derived soils. Error bars represent one standard deviation from the mean. Bars with the same letter are not significantly different at the $\alpha = 0.05$ level.

In 2015, mean percent total vegetative cover under forest thinning on benmoreite-derived soils was not significantly different from 2006 ($p = 0.302$) or 2008 ($p = 0.862$), but it was significantly greater than mean percent total vegetative cover in 2003 ($p < 0.001$).

Forest thinning on basalt-derived soils did not result in a statistically significant difference in mean percent total vegetative cover between the 2008 and 2015 measurement years ($p = 0.133$). However, mean percent total vegetative cover on basalt-derived soils in 2015 was significantly greater than in 2006 ($p = 0.043$) and 2003 ($p = 0.004$). Mean percent total vegetative cover under forest thinning on basalt-derived soils was 3.26 (SD 2.57) in 2003, 6.23 (SD 3.90) in 2006, 7.87 (SD 5.68) in 2008, and 12.64 (SD 7.55) in 2015 (Figure 20).

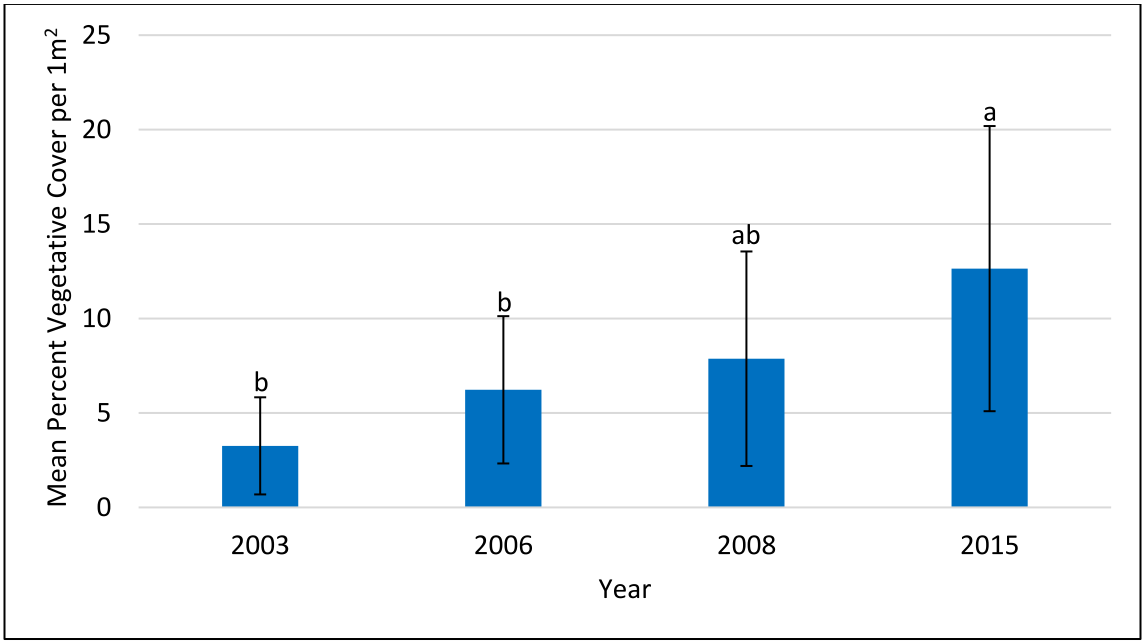

Figure 20. Mean percent total vegetative cover by measurement year under forest thinning on basalt-derived soils. Error bars represent one standard deviation from the mean. Bars with the same letter are not significantly different at the $\alpha = 0.05$ level.

Mean Percent Grass Cover

All independent variables contributed significantly to the power of the HLM of mean percent grass cover in 2015 (Table 11).

Table 11. Hierarchical Linear Model (HLM) predictors and their associated R, R-squared, adjusted R-squared, and changes in R-squared values for each predictor used to analyze mean percent grass cover per 1m^2 for the 2015 measurement year (α = 0.05).

					Change Statistics		
Model	R	R-squared	Adjusted R-squared	Std. Error of the Estimate	R-squared Change	F Change	Sig. F Change
1	.139[a]	0.019	0.017	15.411	0.019	8.331	0.004
2	.248[b]	0.062	0.057	15.094	0.042	18.961	0.000
3	.288[c]	0.083	0.076	14.941	0.021	9.646	0.002

[a] Treatment type

[b] Treatment type plus grazing treatment

[c] Treatment type plus grazing treatment plus soil type

Mean percent grass cover under forest thinning and open canopy treatments was not significantly different (p = 0.838). However, both were significantly greater than the mean percent grass cover under closed canopy conditions (p = 0.016 and p = 0.003, respectively). Mean percent grass cover under forest thinning and open canopy conditions was 15.12 (SD 16.36) and 16.02 (SD 13.79), respectively while mean percent grass cover under closed canopy conditions was 10.70 (SD 15.97).

Thinning on limestone-derived soils yielded significantly greater mean percent grass cover than either closed (p = 0.002) or open (p < 0.001) canopy conditions. Mean percent grass cover on limestone-derived soils under forest thinning was 23.77 (SD 21.87) while under open canopy and closed canopy conditions it was 14.79 (SD 19.82) and 11.40 (SD 11.13), respectively. Open and closed canopy conditions were not significantly different ($p = 0.23$). On basalt-derived soils, forest thinning yielded significantly greater mean percent grass cover than open canopy conditions ($p = 0.015$) but was not significantly different from closed canopy conditions ($p = 0.926$). On benmoreite-derived soils, open canopy conditions had significantly greater mean percent grass cover than either forest thinning or closed canopy conditions ($p = 0.011$ and $p < 0.001$, respectively).

Mean percent grass cover on limestone-derived soils was significantly greater than on basalt-derived soils ($p = 0.006$) but was not significantly different from mean percent grass cover on benmoreite-derived soils ($p = 0.107$). Mean percent grass cover on limestone-derived soils was 16.65 (SD 18.82) while basalt- and benmoreite-derived soils had mean percent grass cover of 10.91 (SD 11.37) and 13.28 (SD 13.77), respectively.

There were no significant differences in mean percent grass cover among soil parent material types under grazing. Under non-grazing, limestone-derived soils had significantly greater mean percent grass cover than basalt- and benmoreite-derived soils ($p < 0.001$ for both comparisons). Mean percent grass cover on limestone-derived soils with no grazing was 23.85 (SD 23.44) while on basalt- and benmoreite-derived soils it was 11.13 (SD12.86) and 14.41 (SD 15.68), respectively. Mean percent grass cover was not significantly different between benmoreite- and basalt derived soils under non-grazing ($p = 0.218$).

Mean percent grass cover on limestone-derived soils under non-grazing was significantly greater than under grazing ($p < 0.001$). Mean percent grass cover under non-grazing was 23.85 (SD 23.44) while under grazing it was 9.45 (SD 7.64).

Mean percent grass cover in 2015 was compared to previous measurement years to evaluate the response of grass cover to forest and grazing treatments over time (Table 12). All model estimators contributed significantly to the power of the HLM of mean percent grass cover.

Table 12. Hierarchical Linear Model (HLM) predictors and their associated R, R-squared, adjusted R-squared, and change in R-squared values for each predictor used to analyze mean percent grass cover per 1m^2 for all measurement years ($\alpha = 0.05$).

Model	R	R-squared	Adjusted R-squared	Std. Error of the Estimate	Change Statistics: R-squared Change	Change Statistics: F Change	Change Statistics: Sig. F Change
1	.159[a]	0.025	0.025	10.365	0.025	42.908	0.000
2	.195[b]	0.038	0.037	10.300	0.013	21.878	0.000
3	.209[c]	0.044	0.042	10.273	0.006	9.667	0.002
4	.438[d]	0.192	0.190	9.446	0.148	302.766	0.000

[a] Treatment type

[b] Treatment type plus grazing treatment

[c] Treatment type plus grazing treatment plus soil type

[d] Treatment type plus grazing treatment plus soil type plus year

Mean percent grass cover was significantly greater in 2015 ($p < 0.001$) than in all previous measurement years for all forest treatments (Table 13).

Table 13. Means and standard deviations of percent grass cover by forest treatment type for all years where measurements occurred.

	Measurement Year			
Treatment	2003	2006	2008	2015
Closed	3.25 (5.01)	3.26 (5.28)	3.63 (8.19)	10.70 (15.97)
Thinned	0.90 (1.11)	4.05 (6.96)	4.78 (7.22)	15.12 (16.36)
Open	6.23 (5.65)	7.38 (5.92)	8.05 (7.52)	16.02 (13.79)

Under closed canopy conditions, mean percent grass cover was not significantly different between measurement years from 2003 through 2006. However, in 2015, mean percent grass cover was significantly greater than all previous measurement years ($p < 0.001$). Mean percent grass cover under closed canopy conditions was 3.25 (SD 5.01) in 2003, 3.26 (SD 5.28) in 2006, 3.63 (SD 8.19) in 2008 and 10.70 (SD 15.97) in 2015.

Under closed canopy conditions, limestone-derived soils had greater mean percent grass cover than basalt- and benmoreite-derived soils ($p = 0.022$ and $p = 0.046$, respectively) (Figure 21). In 2015 mean percent grass cover on benmoreite-derived soils was significantly less than basalt-derived soils ($p = 0.039$), whereas in previous years mean percent grass cover was not statistically significant between benmoreite- and basalt-derived soils.

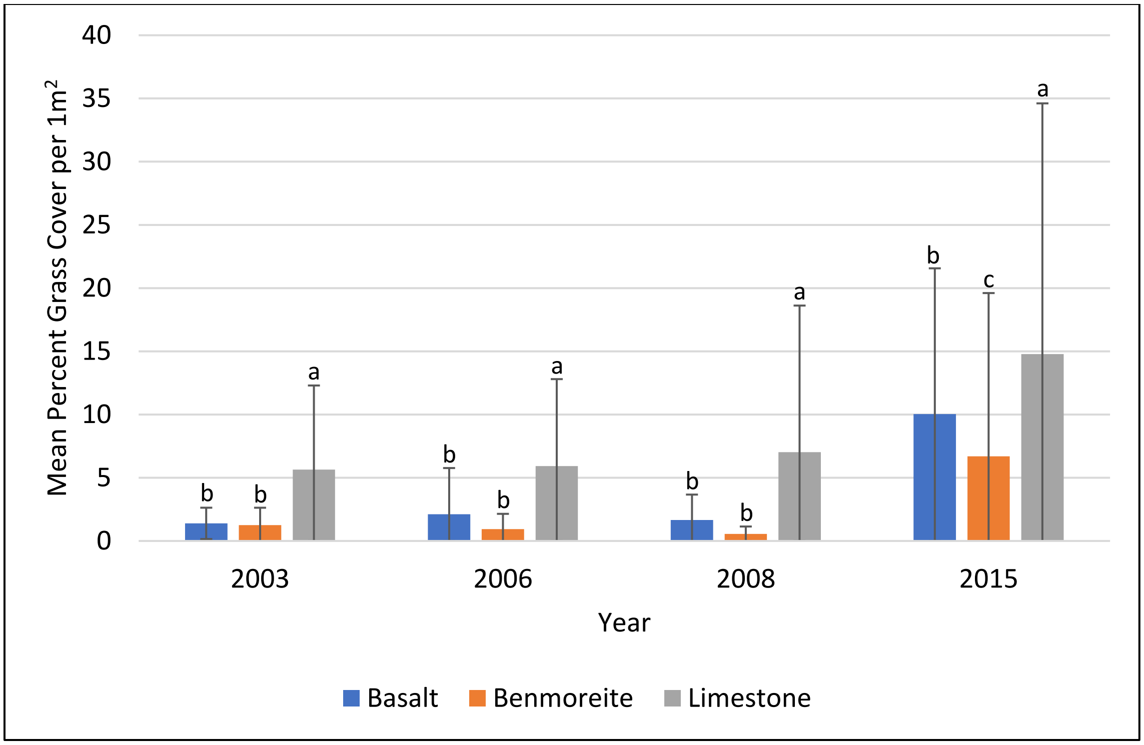

Figure 21. Mean percent grass cover by measurement year and soil parent material type under closed canopy conditions. Error bars represent one standard deviation from the mean. Bars with the same letter within each measurement year are not significantly different at the α = 0.05 level.

Under open canopy conditions, forest treatment outcomes varied among measurement years and soil parent material types. Basalt-derived soils had significantly greater mean percent grass cover than benmoreite- and limestone-derived soils from 2003 through 2008. Mean percent grass cover on benmoreite- and limestone-derived soils was not significantly different for these same measurement years. In 2015, mean percent grass cover was significantly greater on benmoreite-derived soils at 20.43 (SD 15.06), followed by basalt-derived soils with mean percent grass cover of 16.34 (SD 13.50) then limestone-derived soils with mean percent grass cover of 11.40 (SD 11.13) (Figure 22).

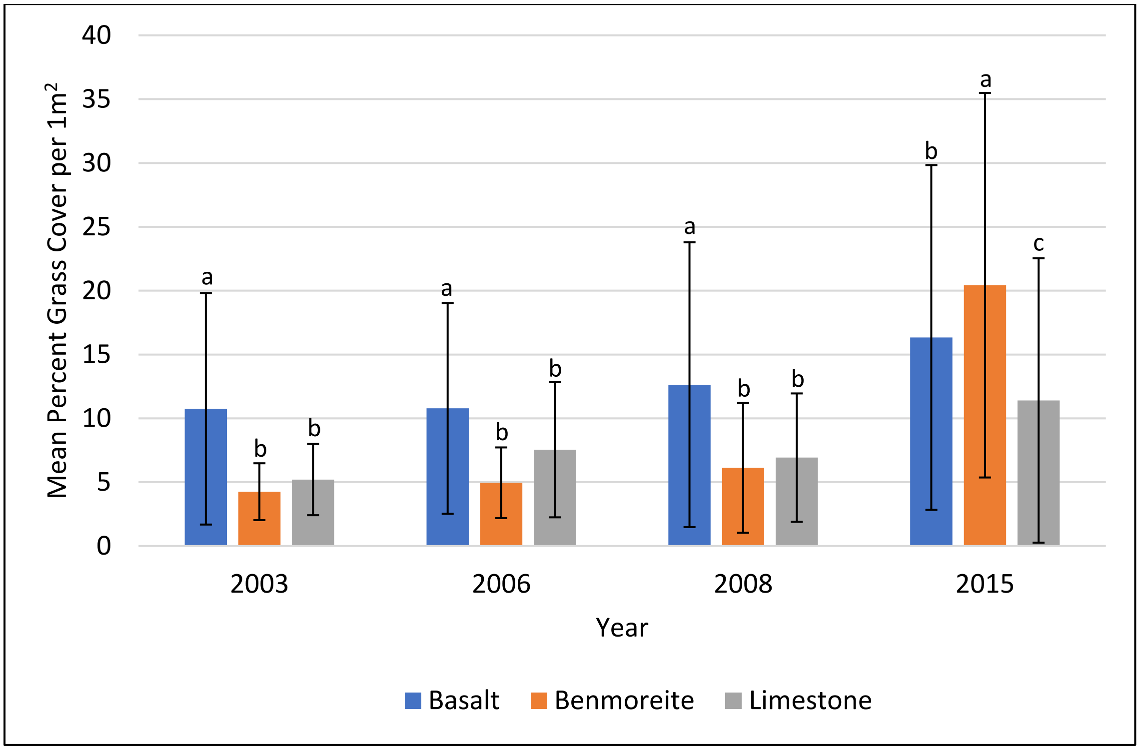

Figure 22. Mean percent grass cover by soil parent material type within measurement year under open canopy conditions. Error bars represent one standard deviation from the mean. Bars with the same letter within each measurement year are not significantly different at the α = 0.05 level.

Under forest thinning, mean percent grass cover in 2006 and 2008 was not significantly different ($p = 0.552$). However, mean percent grass cover for both years was significantly greater than 2003 ($p = 0.009$ and $p = 0.001$, respectively) and significantly lower than 2015 ($p < 0.001$) (Figure 23). Mean percent grass cover was significantly greater in 2015 than all previous years ($p < 0.001$).

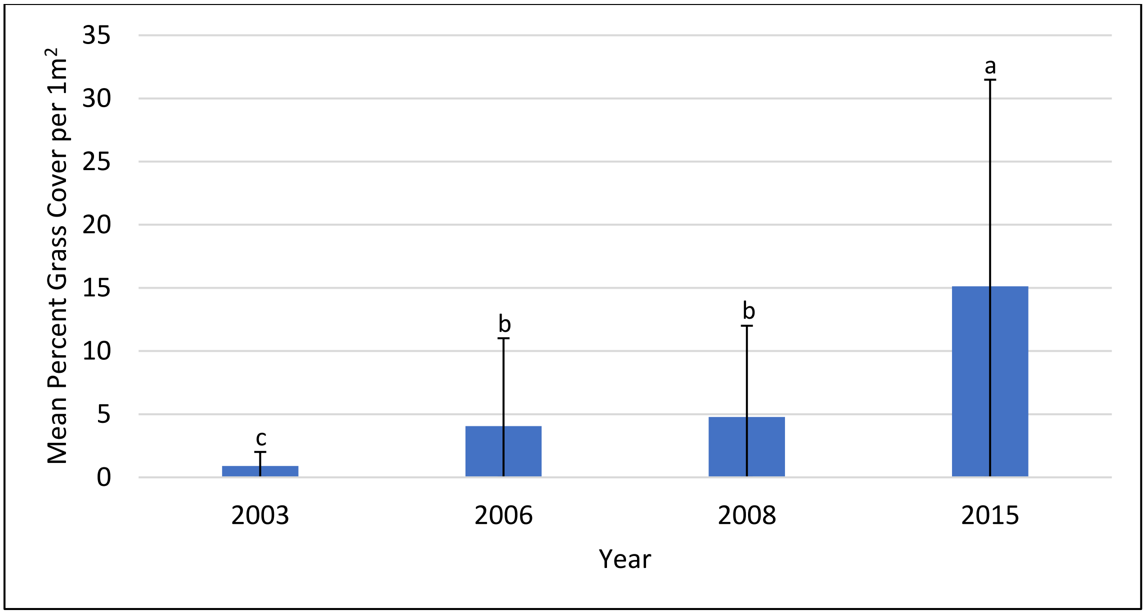

Figure 23. Mean percent grass cover by measurement year under forest thinning. Error bars represent one standard deviation from the mean. Bars with the same letter are not significantly different at the $\alpha = 0.05$ level.

Mean percent grass cover under forest thinning was not significantly different from closed canopy conditions in 2003 (pre-treatment), 2006, and 2008 ($p = 0.104$, $p = 0.480$, and $p = 0.235$, respectively). However, it was significantly lower than under open canopy conditions for these three measurement years ($p < 0.001$) (Figure 24).

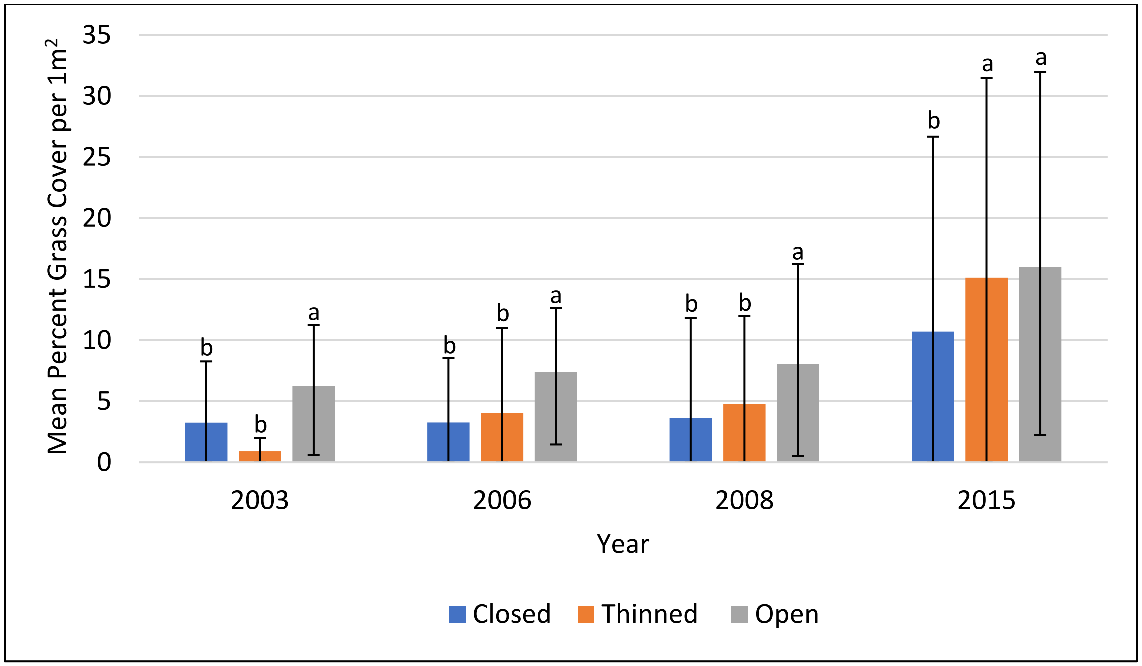

Figure 24. Mean percent grass cover by forest treatment type and measurement year. Error bars represent one standard deviation from the mean. Bars with the same letter within each year are not significantly different at the α = 0.05 level.

In 2003, all soil parent material types in thinned plots had less than 3 percent mean grass cover. In 2015, mean percent grass cover was significantly different among all soil parent material types. Limestone-derived soils had the highest mean percent grass cover at 23.77 (SD 21.82), followed by benmoreite-derived soils at 12.09 (SD 9.11) then basalt-derived soils at 6.52 (SD 5.62). In 2008, mean percent grass cover on basalt-derived soils was significantly lower than on benmoreite-derived soils ($p < 0.001$). Basalt-derived soils had mean percent grass cover of 1.35 (SD 2.18) while benmoreite-derived soils had 8.35 (SD 10.03) (Figure 25).

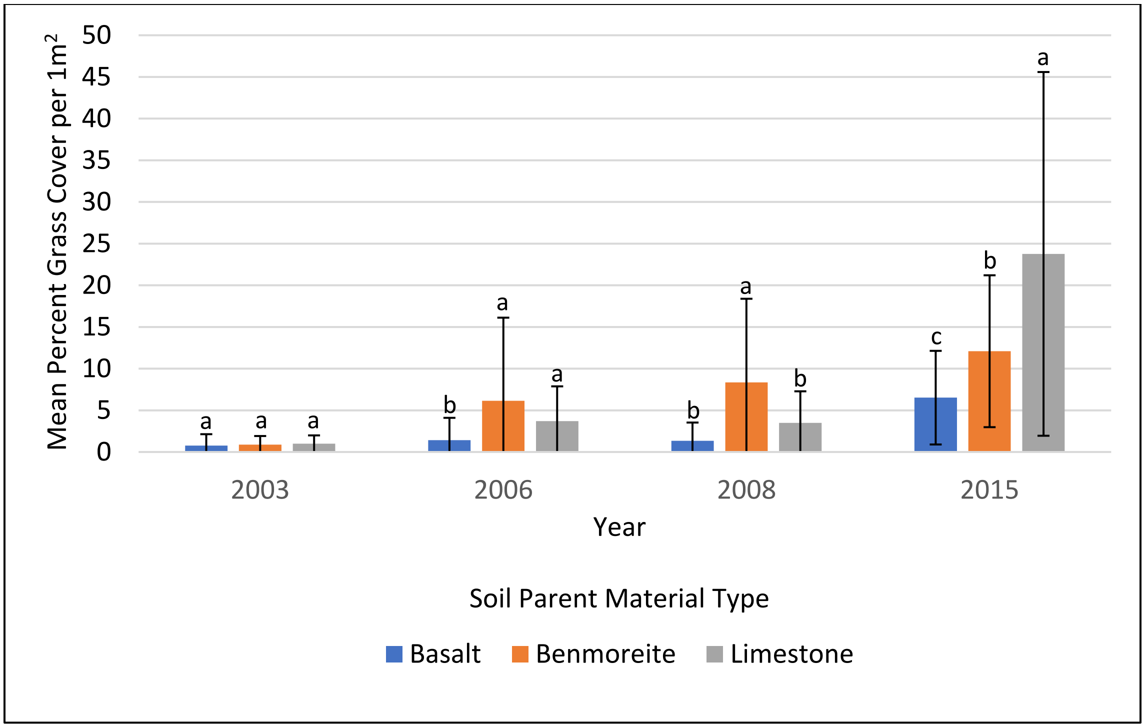

Figure 25. Mean percent grass cover under forest thinning by year and soil parent material type. Error bars represent one standard deviation from the mean. Error bars with the same letter within each soil parent material type are not significantly different at the α = 0.05 level.

Significant differences in mean percent grass cover between benmoreite- and limestone-derived soils under forest thinning was also found for the 2008 and 2015 measurement years. Mean percent grass cover values between 2008 and 2015 indicate a substantial shift on these two soils, although mean percent grass cover increased on both soils. In 2008, mean percent grass cover on benmoreite-derived soils was 8.35 (SD 10.03) vs. 3.5 (SD 3.78) on limestone-derived soils. In 2015, mean percent grass cover on benmoreite-derived soils had increased significantly ($p = 0.031$) to 12.09 (SD 9.11).

Mean percent grass cover on limestone-derived soils indicated an even greater increase ($p < 0.001$) to 23.77 (SD 21.82).

During the 2015 measurement year, non-grazing under forest thinning on limestone-derived soils had significantly greater mean percent grass cover in comparison to grazing ($p < 0.001$). Non-grazing on limestone-derived soils under forest thinning in 2015 had mean percent grass cover of 36.16 (SD 23.60) while under grazing it was 11.37 (SD 9.71).

In 2006 and 2008 under forest thinning on benmoreite-derived soils, grazing exclusion yielded significantly greater mean percent grass cover than grazed plots ($p = 0.01$). Mean percent grass cover on benmoreite-derived soils in 2006 was 2.46 (SD 2.54) under grazing and 9.83 (SD 12.96) under grazing exclusion. In 2008, mean percent grass cover on benmoreite-derived soils under grazing was 5.88 (SD 7.81) while under grazing exclusion it was 10.82 (SD 11.46).

Forb Cover

Forest treatment type alone was not a significant predictor in the HLM of mean percent forb cover in 2015 (Table 14). However, grazing treatment and soil type were significant predictors of variation in mean percent forb cover in the regression model.

Table 14. Hierarchical Linear Model (HLM) predictors and their associated R, R-squared, adjusted R-squared, and change in R-squared values for each independent variable used to analyze mean percent forb cover per $1m^2$ for the 2015 measurement year ($\alpha = 0.05$).

Model	R	R-squared	Adjusted R-squared	Std. Error of the Estimate	Change Statistics		
					R-squared Change	F Change	Sig. F Change
1	0.009[a]	0.000	-0.002	19.54	0.000	0.033	0.856
2	0.098[b]	0.010	0.005	19.47	0.009	4.035	0.045
3	0.438[c]	0.191	0.186	17.61	0.182	94.496	0.000

[a] Treatment type

[b] Treatment type plus grazing treatment

[c] Treatment type plus grazing treatment plus soil type

Mean percent forb cover was greater on limestone-derived soils than on basalt-derived soils for all forest treatment types and was also greater on limestone-derived soils than on benmoreite-derived soils under closed canopy and forest thinning treatments. However, under open canopy conditions there was no significant difference in mean percent forb cover between limestone- and benmoreite-derived soils ($p = 0.106$). There was no significant difference in mean percent forb cover among forest treatments on basalt- and benmoreite-derived soils ($p = 0.922$). Mean percent forb cover on limestone-derived soils was 19.03 (SD 29.30) under closed canopy conditions, 30.01 (SD 30.57) under forest thinning, and 14.53 (SD 19.20) under open canopy conditions. Mean percent forb cover on basalt-derived soils was 2.06 (SD 1.85) under closed canopy conditions, 3.48 (SD 2.72) under forest thinning, and 2.30 (SD 2.53) under open canopy conditions. Mean percent forb cover on benmoreite-derived soils was 1.76 (SD 4.24) under closed canopy conditions, 4.39 (SD 6.26) under forest thinning, and 7.85 (SD 9.72) under open canopy conditions.

A significant difference in mean percent forb cover was found between grazing and non-grazing under closed canopy conditions (p = 0.04). Mean percent forb cover under closed canopy conditions with grazing was 12.45 (SD 26.98) and with non-grazing it was 4.15 (SD 5.51). The grazing treatment effect was particularly pronounced on limestone-derived soils under closed canopy conditions where mean percent forb cover was 31.74 (SD 37.11) under grazing and 6.31 (SD 6.07) under non-grazing.

Limestone-derived soils had significantly greater mean percent forb cover than basalt- and benmoreite-derived soils under closed canopy conditions with grazing. Mean percent forb cover on limestone-derived soils under closed canopy conditions with grazing was 31.74 (SD 37.11), while on basalt- and benmoreite-derived soils it was 2.09 (SD 1.82) and 0.55 (SD 0.89), respectively.

Forest thinning with non-grazing yielded significantly greater mean percent forb cover than closed canopy conditions with no grazing (p = 0.002) but was not significantly different from open canopy conditions. Mean percent forb cover under closed canopy conditions with no grazing was 6.31 (SD 6.07). Under forest thinning with no grazing it was 30.00 (SD 30.57) and under open canopy conditions with no grazing it was 14.53 (SD 19.20).

Mean percent forb cover in 2015 was compared to previous measurement years to evaluate forb cover responses to forest and grazing treatments and soil parent material type over time (Table 15).

Table 15. Hierarchical Linear Model (HLM) predictors and their associated R, R-squared, adjusted R-squared, and changes in R-squared values for each independent variable used to analyze mean percent forb cover per $1m^2$ for all measurement years ($\alpha = 0.05$).

					Change Statistics		
Model	R	R-squared	Adjusted R-squared	Std. Error of the Estimate	R-squared Change	F Change	Sig. F Change
1	.086[a]	0.007	0.007	12.24	0.007	12.48	0.000
2	.097[b]	0.009	0.008	12.23	0.002	3.21	0.073
3	.376[c]	0.141	0.140	11.39	0.132	253.88	0.000
4	.427[d]	0.182	0.180	11.12	0.041	82.78	0.000

[a] Treatment type

[b] Treatment type plus grazing treatment

[c] Treatment type plus grazing treatment plus soil type

[d] Treatment type plus grazing treatment plus soil type plus year

All independent variables except grazing contributed significantly to the power of the HLM of mean percent forb cover.

Closed canopy conditions had significantly lower mean percent forb cover than open canopy conditions and forest thinning. Mean percent forb cover under closed canopy conditions was 4.71 (SD 11.86). Under open canopy conditions and forest thinning, it was 7.39 (SD 10.20) and 8.18 (SD 14.23), respectively.

Limestone-derived soils had significantly greater mean percent forb cover than basalt- and benmoreite-derived soils when all measurement years were analyzed ($p < 0.001$).

Mean percent forb cover was not significantly different between basalt- and benmoreite-derived soils ($p = 0.112$). Mean percent forb cover on limestone-derived soils was 12.35 (SD 17.02), while on basalt- and benmoreite-derived soils it was 2.31 (SD 2.74) and 3.96 (SD 6.68), respectively.

Measurement year affected mean percent forb cover with 2003 having significantly lower mean percent forb cover than all other measurement years ($p = 0.001$), and 2015 having significantly greater mean percent forb cover than all previous measurement years ($p < 0.001$). There was no significant difference in mean percent forb cover between the 2006 and 2008 measurement years ($p = 0.715$). Mean percent forb cover in 2003 was 3.39 (SD 4.89). In 2006 it had increased significantly to 6.74 (SD 9.43), remained static in 2008 at 6.39 (SD 8.68), then increased sharply in 2015 to 10.50 (SD 19.52). Mean percent forb cover was not significantly affected by grazing treatments ($p = 0.101$).

The only significant difference in mean percent forb cover under open canopy conditions was between the 2003 and 2008 measurement years on limestone-derived soils, with 2003 having significantly lower mean percent forb cover than 2008. In 2003, mean percent forb cover on limestone-derived soils under open canopy conditions was 9.50 (SD 6.47). In 2008 it had increased to 15.19 percent (SD 13.19).

Under forest thinning, there was a significant difference in mean percent forb cover between benmoreite- and limestone-derived soils. On benmoreite-derived soils, mean percent forb cover in 2003 was significantly lower than in 2006 and 2008 but was not significantly different than 2015. Mean percent forb cover under forest thinning on benmoreite-derived soils was 0.53 (SD 0.85) in 2003, 9.78 (SD 12.08) in 2006, 8.28 (SD 5.06) in 2008, and 4.39 (SD 6.26) in 2015.

On limestone-derived soils, there were significant differences in mean percent forb cover among measurement years except between 2006 and 2008. Mean percent forb cover on limestone-derived soils under forest thinning in 2003 was 4.33 (SD 6.06). In 2006, it had increased to 11.91 (SD 12.29). In 2008, mean percent forb cover remained static at 10.25 (SD 10.30), then in 2015 it increased significantly to 30.01 (SD 30.57).

Although the grazing treatment was not a significant predictor of mean percent forb cover in the HLM, some significant differences were found in pairwise comparisons. Under closed canopy conditions on limestone-derived soils in 2015, grazing resulted in significantly greater mean percent forb cover than non-grazing ($p < 0.001$). Mean percent forb cover under grazing was 31.74 (SD 37.11) while non-grazing yielded 6.31 (SD 6.07).

Under open canopy conditions on limestone-derived soils, mean percent forb cover was also greater under grazing than non-grazing in 2008. Mean percent forb cover under grazing was 18.95 (SD 15.76) while under non-grazing it was 11.42 (SD 8.77).

Under forest thinning, there were significant differences in mean percent forb cover by year and soil parent material type. In 2006, forest thinning with grazing on benmoreite-derived soils had 9.78 percent greater mean percent forb cover than grazing exclusion ($p < 0.001$). Mean percent forb cover was 14.67 (SD 14.95) under grazing and 4.89 (SD 4.94) under non-grazing. However, no significant difference in mean percent forb cover between grazing and non-grazing on benmoreite-derived soils was found in 2003, 2008, or 2015.

Under forest thinning on limestone-derived soils with grazing mean percent forb cover was significantly greater than under non-grazing ($p < 0.001$). Mean percent forb cover under grazing was 38.55 (SD 36.29) while under non-grazing, mean percent forb cover was 21.47 (SD 20.91).

Exotic Plant Cover

Treatment type did not contribute significantly to the power of the HLM of mean percent exotic plant cover in 2015 (Table 16). However, grazing treatment and soil parent material type are significant predictors of variation in mean percent exotic plant cover. Overall mean percent exotic plant cover across all treatments and soil types was 3.18 (SD 8.93). Average exotic plant cover was 1.02 (SD 1.75) on basalt-derived soils, 1.49 (SD 3.51) on benmoreite-derived soils, and 6.23 (SD 13.44) on limestone-derived soils.

Soils derived from limestone had significantly greater mean percent exotic plant cover than basalt- or benmoreite-derived soils ($p < 0.001$). Mean percent exotic plant cover on limestone-derived soils was 6.23 (SD 13.43) while on basalt-derived soils it was 1.02 (SD 1.75) and on benmoreite-derived soils it was 1.49 (SD 3.51). There was no significant difference in mean percent exotic plant cover between basalt- and benmoreite-derived soils ($p = 0.988$).

Table 16. Hierarchical Linear Model (HLM) predictors and their associated R, R-squared, adjusted R-squared, and changes in R-squared values for each independent variable used to analyze mean percent exotic plant cover per $1m^2$ for the 2015 measurement year (α = 0.05).

					Change Statistics		
Model	R	R-squared	Adjusted R-squared	Std. Error of the Estimate	R-squared Change	F Change	Sig. F Change
1	.053[a]	0.003	0.000	8.928	0.003	1.191	0.276
2	.154[b]	0.024	0.019	8.844	0.021	9.057	0.003
3	.304[c]	0.092	0.086	8.539	0.068	31.681	0.000

[a]Treatment type

[b]Treatment type plus grazing treatment

[c]Treatment type plus grazing treatment plus soil type

Mean percent exotic plant cover in 2015 was compared to previous measurement years to evaluate exotic plant cover responses to forest and grazing treatments over time (Table 17).

Table 17. Hierarchical Linear Model (HLM) predictors and their associated R, R-squared, adjusted R-squared, and changes in R-squared values for each independent variable used to analyze mean percent exotic plant cover at the subplot level ($1m^2$) for all measurement years (α = 0.05).

					Change Statistics		
Model	R	R-squared	Adjusted R-squared	Std. Error of the Estimate	R-squared Change	F Change	Sig. F Change
1	.001[a]	0.000	-0.001	6.336	0.000	0.002	0.969
2	.039[b]	0.002	0.000	6.333	0.002	2.512	0.113
3	.200[c]	0.040	0.038	6.211	0.039	66.546	0.000
4	.232[d]	0.054	0.052	6.168	0.014	24.044	0.000

[a]Treatment type

[b]Treatment type plus grazing treatment

[c]Treatment type plus grazing treatment plus soil type

[d]Treatment type plus grazing treatment plus soil type plus year

Treatment type and grazing treatments did not contribute significantly to the power of the HLM of mean percent exotic plant cover. However, soil parent material type and measurement year were significant predictors.

Although it did not contribute significantly to the power of the HLM, pairwise comparisons of means using Bonferroni adjustment for multiple comparisons revealed treatment type affected mean percent exotic plant cover in some instances. On benmoreite-derived soils, thinning had greater mean exotic plant cover by 2.80 percent ($p < 0.001$) in comparison to closed canopy conditions and 2.55 percent ($p < 0.001$) in comparison to open canopy conditions (Figure 26). Thinning on limestone-derived soils also had greater mean percent exotic plant cover by 0.93 percent in comparison to closed canopy conditions and 1.42 percent in comparison to open canopy conditions (Figure 26).

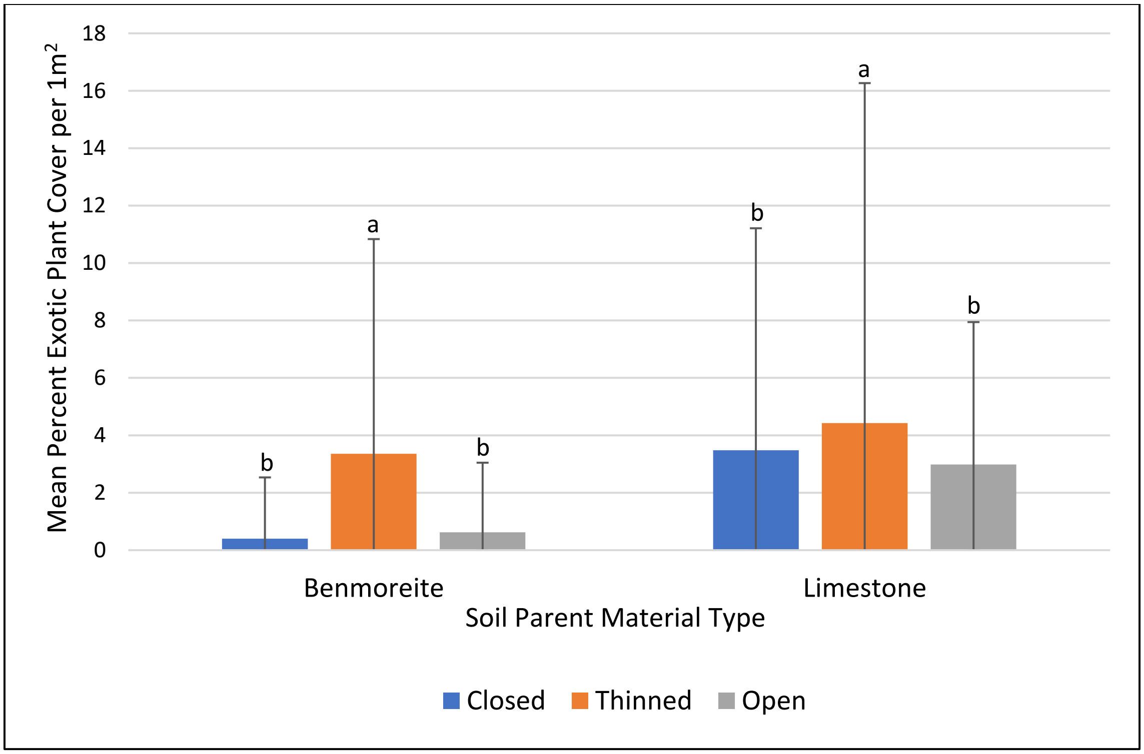

Figure 26. Mean percent exotic plant cover by forest treatment type on benmoreite- and limestone-derived soils. Error bars within soil parent material type represent one standard deviation from the mean. Bars with the same letter are not significantly different at the $\alpha = 0.05$ level.

Soil type was a significant predictor of mean percent exotic plant cover. Limestone-derived soils had significantly greater mean percent exotic plant cover than benmoreite-derived soils ($p < 0.001$) and benmoreite-derived soils had significantly greater mean percent exotic plant cover than basalt-derived soils ($p = 0.045$). Mean percent exotic plant cover on limestone-derived soils was 3.63 (SD 8.66). On benmoreite-derived soils it was 1.51 percent (SD 4.96) and on basalt-derived soils it was 0.68 percent (SD 1.63) (Figure 27).

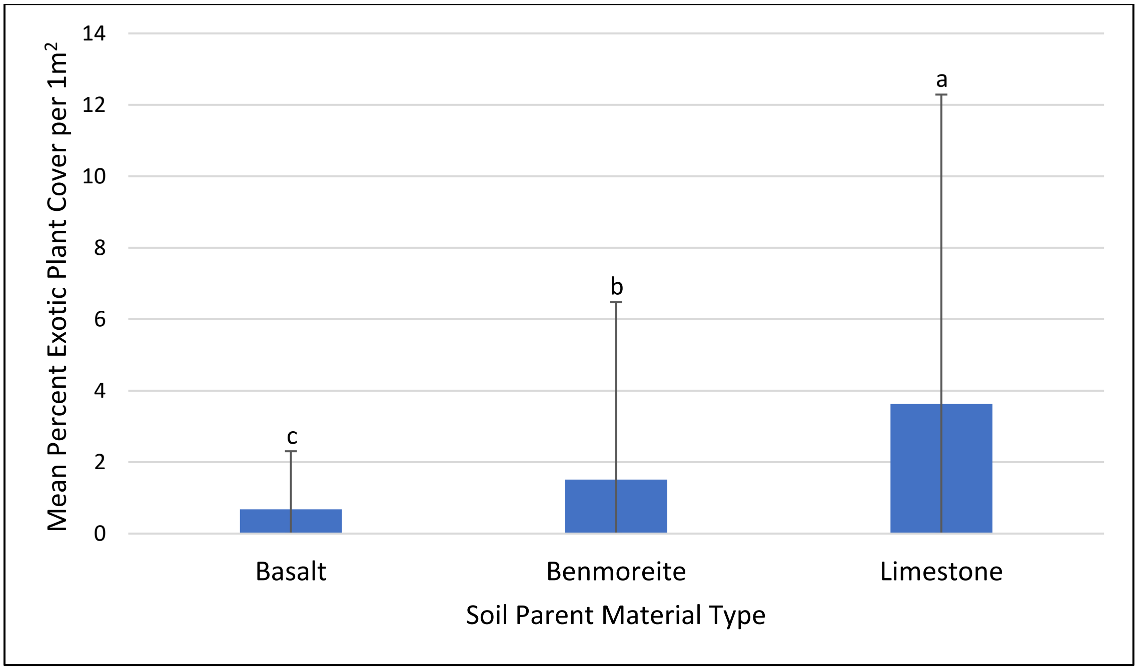

Figure 27. Mean percent exotic plant cover by soil parent material type. Error bars represent one standard deviation from the mean. Bars with the same letter are not significantly different at the $\alpha = 0.05$ level.

Measurement year was a significant predictor of mean percent exotic plant cover. The 2015 measurement year had significantly greater mean percent exotic plant cover in comparison to 2003 and 2008 ($p < 0.001$ and $p = 0.011$, respectively) (Figure 28).

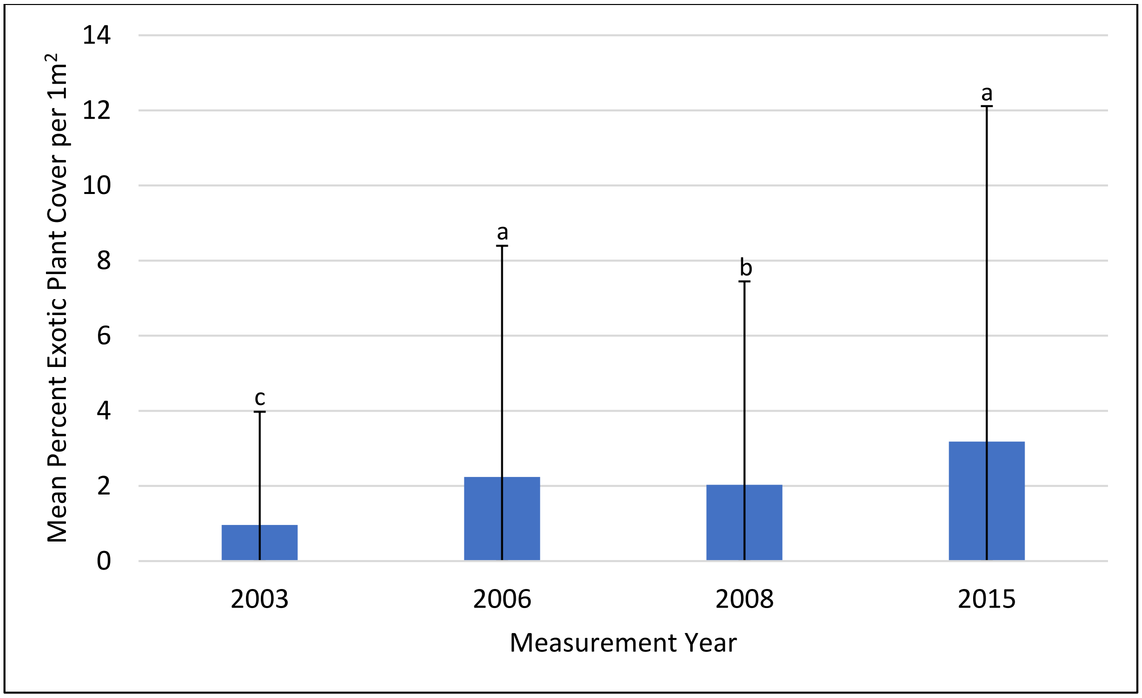

Figure 28. Mean percent exotic plant cover by measurement year. Error bars represent one standard deviation from the mean. Bars with the same letter are not significantly different at the α = 0.05 level.

Grazing treatments only affected mean percent exotic plant cover on limestone-derived soils. Non-grazing had greater mean percent exotic plant cover than grazing ($p < 0.001$). Mean percent exotic plant cover was 2.68 (SD 6.06) under non-grazing and 4.58 (SD 10.56) under grazing. There were no other significant differences in mean percent exotic plant cover by forest treatment type, soil parent material type or measurement year.

Perennial Plant Cover

All independent variables contributed significantly to the power of the HLM of mean percent perennial plant cover in 2015 (Table 18). Closed canopy conditions had significantly lower mean percent perennial plant cover than open conditions ($p < 0.001$) and forest thinning ($p = 0.028$). Open canopy conditions and thinning treatments were not significantly different ($p = 0.430$). Mean percent perennial plant cover under closed canopy conditions was 14.30 (SD 15.73), while under open conditions and forest thinning it was 21.24 (SD 16.88) and 19.00 (17.60), respectively (Figure 29).

Table 18. Hierarchical Linear Model (HLM) predictors and their associated R, R-squared, adjusted R-squared, and changes in R-squared values for each independent variable used to analyze mean percent perennial plant cover per $1m^2$ for the 2015 measurement year ($\alpha = 0.05$).

Model	R	R-squared	Adjusted R-squared	Std. Error of the Estimate	Change Statistics		
					R-squared Change	F Change	Sig. F Change
1	.167[a]	0.028	0.025	16.749	0.028	12.062	0.001
2	.267[b]	0.071	0.067	16.390	0.043	19.677	0.000
3	.314[c]	0.098	0.092	16.167	0.027	12.710	0.000

[a] Treatment type

[b] Treatment type plus grazing treatment

[c] Treatment type plus grazing treatment plus soil type

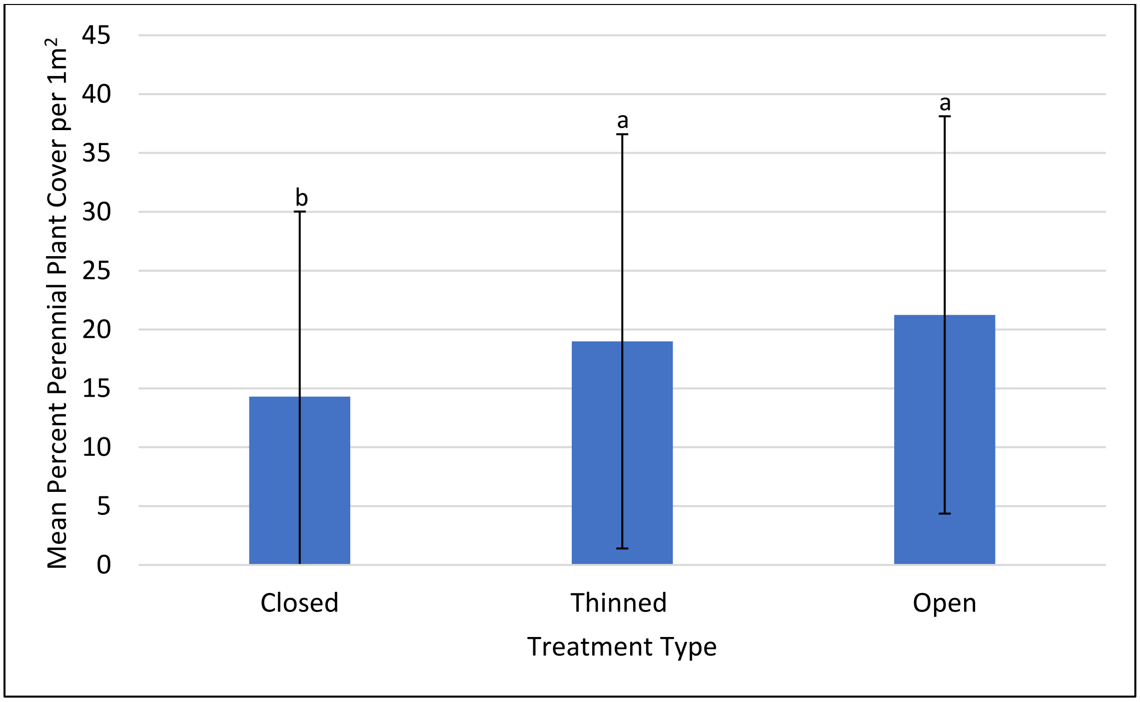

Figure 29. Mean percent perennial plant cover by forest treatment type. Error bars represent one standard deviation from the mean. Bars with the same letter are not significantly different at the α = 0.05 level.

Limestone-derived soils had greater perennial plant cover than basalt- and benmoreite-derived soils (p = 0.001 and p = 0.045, respectively). Mean percent perennial plant cover on limestone-derived soils was 21.52 (SD 19.66), while on basalt-and benmoreite-derived soils it was 14.36 (SD 12.43) and 17.42 (SD 16.06), respectively (Figure 30).

Grazing treatments also affected mean percent perennial plant cover with grazing having 14.72 (SD 12.66) and non-grazing having 21.84 (SD 19.89) percent perennial plant cover.

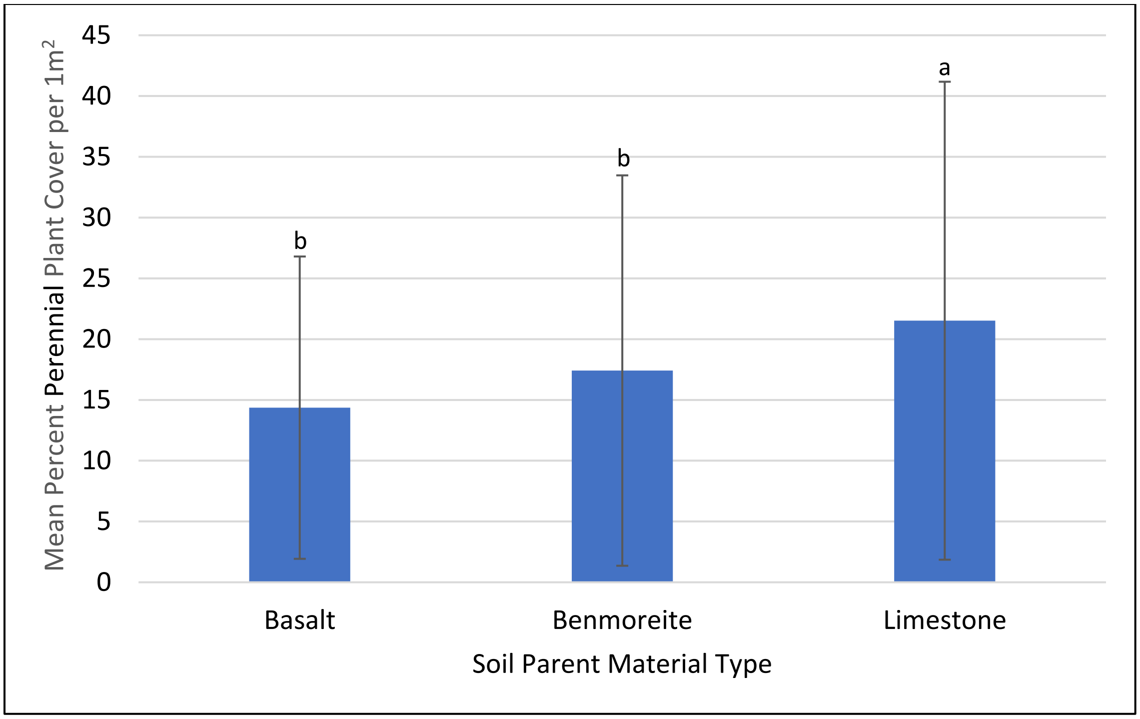

Figure 30. Mean percent perennial plant cover by soil parent material type. Error bars represent one standard deviation from the mean. Bars with the same letter are not significantly different at the $\alpha = 0.05$ level.

Under both closed canopy conditions and forest thinning, limestone-derived soils had significantly greater mean percent perennial plant cover under non-grazing than under grazing ($p > 0.001$), indicating an interaction effect. Mean percent perennial plant cover on limestone-derived soils under closed canopy conditions with grazing was 9.74 (SD 5.52), while under grazing exclusion it was 29.39 (SD 22.76). Under forest thinning with grazing on limestone-derived soils, mean percent perennial cover was 16.53 (SD 12.67) while under grazing exclusion it was 34.8 (SD 25.33) (Figure 31).

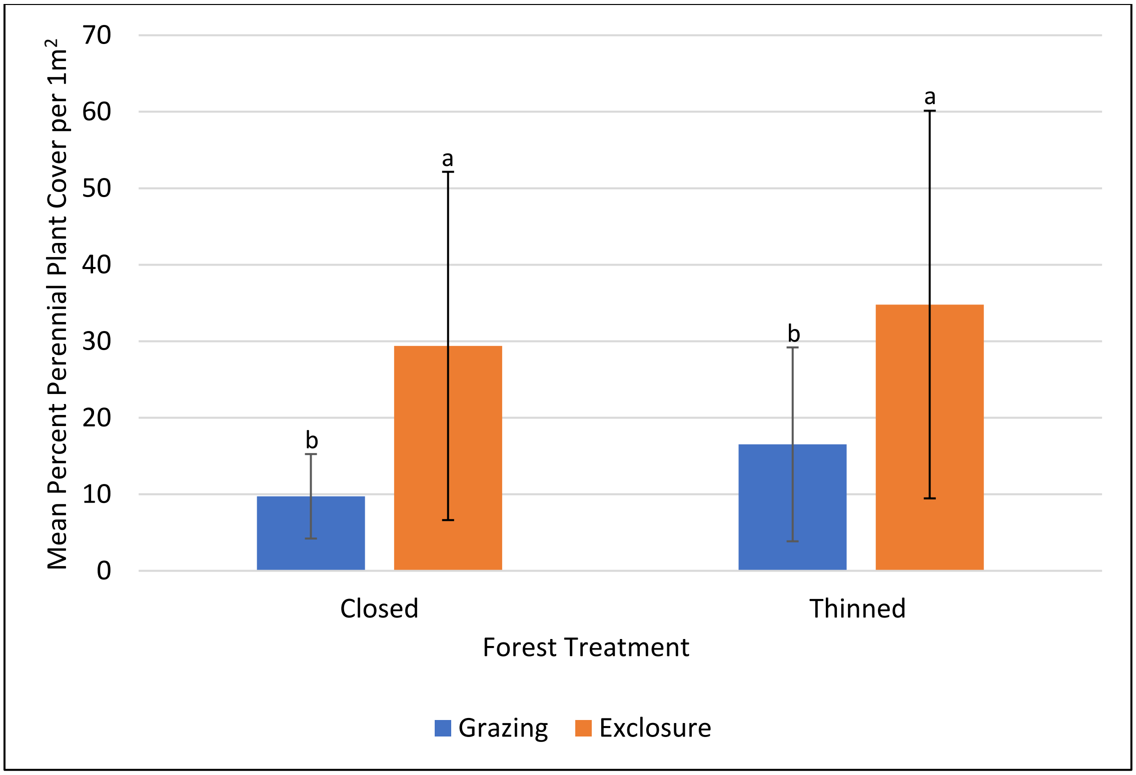

Figure 31. Mean percent perennial plant cover by grazing treatment. Error bars represent one standard deviation from the mean. Bars with the same letter are not significantly different at the $\alpha = 0.05$ level.

Under closed canopy conditions and forest thinning on basalt- and benmoreite-derived soils, mean percent perennial plant cover under grazing and non-grazing was not significantly different. However, under open canopy conditions on soils derived from benmoreite, mean percent perennial plant cover was significantly greater under non-grazing than under grazing ($p < 0.001$). Mean percent perennial plant cover under open canopy conditions with non-grazing was 31.68 (SD 20.09), while under grazing it was 16.61 (SD 8.48).

Mean percent perennial plant cover in 2015 was compared to previous measurement years to evaluate perennial plant cover responses to forest and grazing treatments over time (Table 19).

Table 19. Hierarchical Linear Model (HLM) predictors and their associated R, R-squared, adjusted R-squared values, and change in R-squared values for each independent variable used to analyze mean percent perennial cover at the subplot level (1m^2) for all measurement years ($\alpha = 0.05$).

					Change Statistics		
Model	R	R-squared	Adjusted R-squared	Std. Error of the Estimate	R-squared Change	F Change	Sig. F Change
1	.185[a]	0.034	0.034	12.707	0.034	58.857	0.000
2	.217[b]	0.047	0.046	12.627	0.013	21.977	0.000
3	.311[c]	0.097	0.095	12.295	0.050	91.417	0.000
4	.442[d]	0.196	0.194	11.608	0.099	202.503	0.000

[a] Treatment type

[b] Treatment type plus grazing treatment

[c] Treatment type plus grazing treatment plus soil type

[d] Treatment type plus grazing treatment plus soil type plus year

All model estimators contributed significantly to the power of the HLM of mean percent perennial plant cover. Open canopy conditions had significantly greater mean percent perennial plant cover than closed canopy conditions or forest thinning treatments ($p < 0.001$) (Figure 32). Mean percent perennial plant cover under open canopy conditions was 14.80 (SD 12.65). Forest thinning also yielded significantly greater mean percent perennial plant cover than closed canopy conditions ($p < 0.001$). Under forest thinning mean percent perennial plant cover was 11.85 (SD 13.47), while under closed canopy conditions it was 8.9 (SD 11.91).

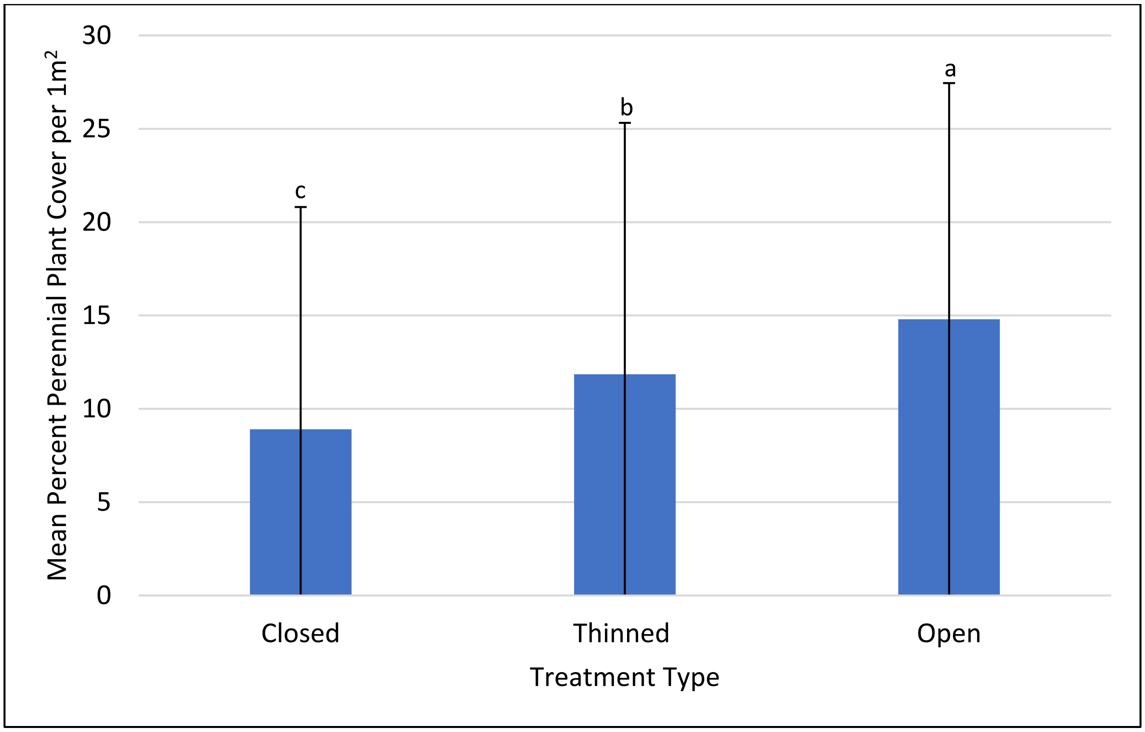

Figure 32. Mean percent perennial plant cover by forest treatment type. Error bars represent one standard deviation from the mean. Bars with the same letter are not significantly different at the α = 0.05 level.

Grazing treatment also affected mean percent perennial cover when all years and soil parent material types are considered (p = 0.012). Mean percent perennial plant cover under non-grazing was 13.41 (SD 14.74) while under grazing it was 10.49 (SD 10.64).

Mean percent perennial plant cover was also affected by soil parent material type. Limestone-derived soils had significantly greater mean percent perennial plant cover than basalt- and benmoreite-derived soils ($p < 0.001$). Mean percent perennial plant cover was not significantly different between basalt- and benmoreite derived soils ($p = 0.619$). Mean percent perennial plant cover on limestone-derived soils was 15.72 (SD 14.00), while on basalt-and benmoreite-derived soils it was 9.77 (SD 10.79) and 9.35 (SD 12.12), respectively (Figure 33).

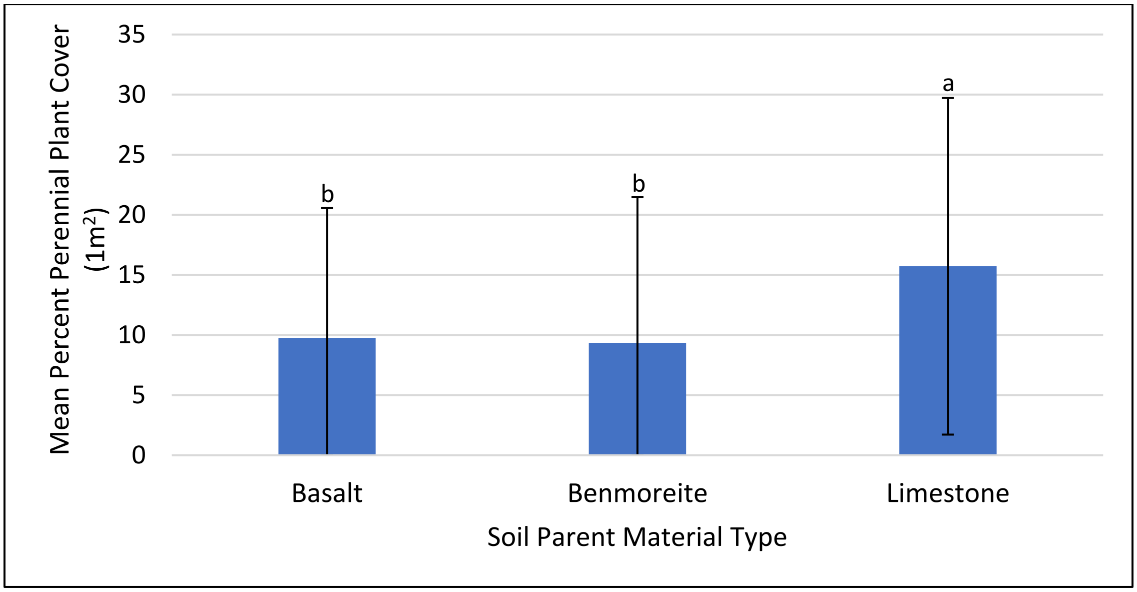

Figure 33. Mean percent perennial plant cover by soil parent material type. Error bars represent one standard deviation from the mean. Bars with the same letter are not significantly different at the $\alpha = 0.05$ level.

Measurement year also significantly affected mean percent perennial plant cover, indicating a trend of increasing perennial plant cover over time for all forest treatment types, grazing treatments, and soil parent material types. Mean percent perennial plant cover in 2003 was 7.14 (SD 7.59). In 2006, it had increased to 10.74 (SD 10.69). It further increased, although not significantly, in 2008 to 11.35 (SD 11.79) and in 2015 mean perennial plant cover increased significantly to 18.21 percent (SD 16.97) (Figure 34).

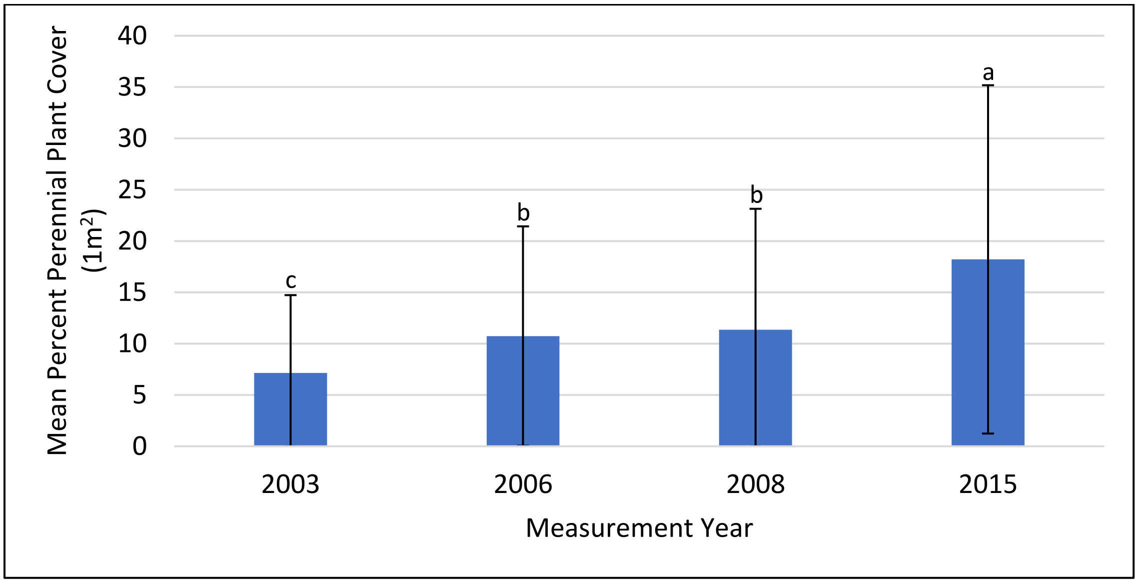

Figure 34. Mean percent perennial plant cover by measurement year. Error bars represent one standard deviation from the mean. Bars with the same letter are not significantly different at the α = 0.05 level.

Under closed canopy conditions and forest thinning, non-grazing had significantly greater mean percent perennial plant cover than grazing ($p = 0.04$ and $p < 0.001$, respectively). No significant difference was found between grazing and non-grazing under open canopy conditions ($p = 0.16$). Mean percent perennial plant cover under closed canopy conditions with grazing exclosure was 10.59 (SD 13.80), while under grazing it was 7.24 (SD 9.43). Under forest thinning with no grazing, mean percent perennial plant cover was 13.86 (SD 16.18), while with grazing it was 9.86 (SD 9.70) (Figure 35).

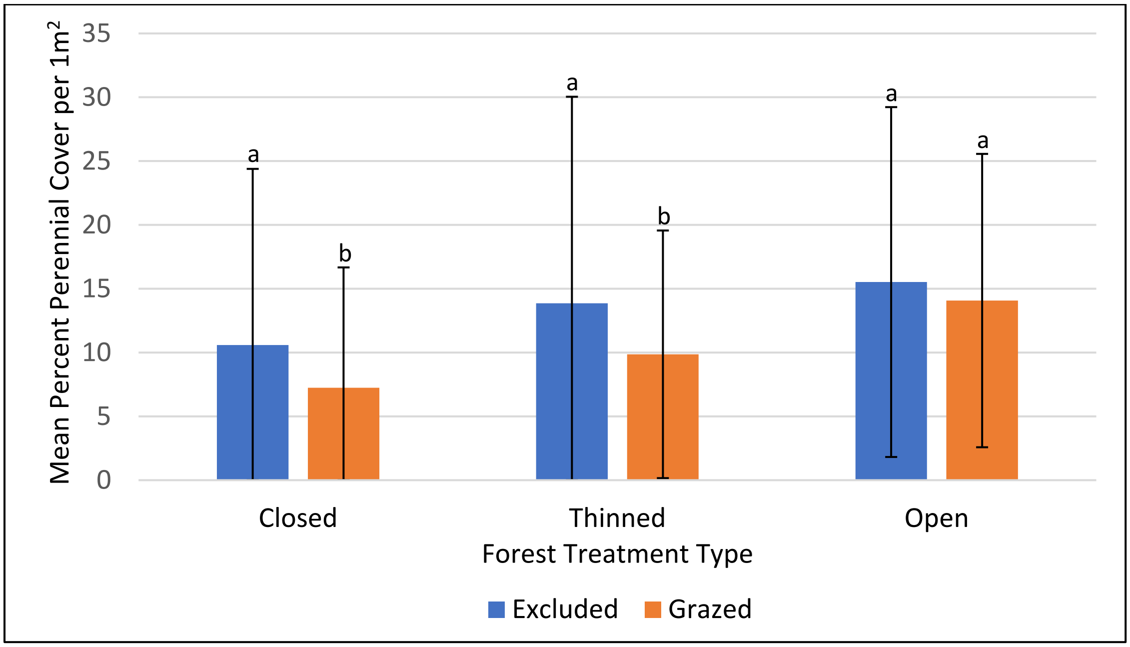

Figure 35. Mean percent perennial plant cover by forest treatment type under grazing and non-grazing. Error bars represent one standard deviation from the mean. Bars with the same letter within forest treatment type are not significantly different at the α = 0.05 level.

On basalt-derived soils, open canopy conditions had greater mean percent perennial plant cover than closed canopy conditions and forest thinning ($p < 0.01$). Closed canopy conditions were not significantly different from forest thinning. Mean percent perennial plant cover on basalt-derived soils under open canopy conditions was 16.90 (SD 13.63). Under forest thinning and closed canopy conditions it was 6.31 (SD 5.64) and 6.03 (SD

7.39), respectively. On benmoreite-derived soils, mean percent perennial plant cover was greater under open canopy conditions and forest thinning than under closed canopy conditions ($p < 0.01$). Mean percent perennial plant cover was not significantly different between open canopy and thinning treatments ($p = 0.83$). Mean percent perennial plant cover on benmoreite-derived soils under open canopy conditions and forest thinning was 11.47 (SD 12.02) and 11.71 (SD 13.61), respectively, while under closed canopy conditions it was 3.67 (SD 7.70). There were no significant differences among forest treatment types on limestone-derived soils ($p = 0.182$).

Interaction between soil parent material type and forest treatment type also affected mean percent perennial plant cover. Under closed canopy conditions, limestone-derived soils had significantly greater mean percent perennial plant cover than basalt- and benmoreite-derived soils ($p < 0.01$). Mean percent perennial plant cover was not significantly different between basalt- and benmoreite-derived soils ($p = 0.089$). Mean percent perennial plant cover under closed canopy conditions on limestone-derived soils was 14.78 (SD 14.21), while on basalt- and benmoreite-derived soils it was 6.03 (SD 7.39) and 3.67 (7.70), respectively.

Under forest thinning, mean percent perennial plant cover was significantly different between all soil parent material types ($p < 0.001$) with limestone-derived soils having the greatest mean percent perennial plant cover and basalt-derived soil having the lowest. Mean percent perennial plant cover on limestone-derived soils was 15.64 (SD 15.59), 11.71 (SD 13.61) on benmoreite-derived soils and 6.31 (SD 5.64) on basalt-derived soils.

Under open canopy conditions, mean percent perennial plant cover was not significantly different between basalt- and limestone-derived soils ($p = 1.00$). However, on benmoreite-derived soils mean percent perennial plant cover was significantly lower than on both basalt- and limestone-derived soils ($p < 0.001$). On basalt-derived and limestone-derived soils under open canopy conditions, mean percent perennial plant cover was 16.90 (SD 13.63) and 16.73 (SD 11.95), respectively, while on benmoreite-derived soils it was 11.47 (SD 12.02).

Mean percent perennial plant cover was affected by a two-way interaction between forest treatment type and measurement year. As would be expected, in 2003 (pre-treatment), mean percent perennial cover was not significantly different between sites with closed canopy conditions (control) and sites subsequently selected for thinning ($p = 0.80$). However, mean percent perennial plant cover under open canopy conditions was significantly greater than closed canopy and thinning treatments ($p < 0.001$) (Figure 36). Mean percent perennial plant cover under open conditions in 2003 was 11.06 (SD 8.52). Under closed canopy conditions and forest thinning mean percent perennial plant cover was 6.04 (SD 6.95) and 3.84 (SD 4.62), respectively. In 2006, mean percent perennial plant cover was significantly different among all forest treatment types (Figure 36).

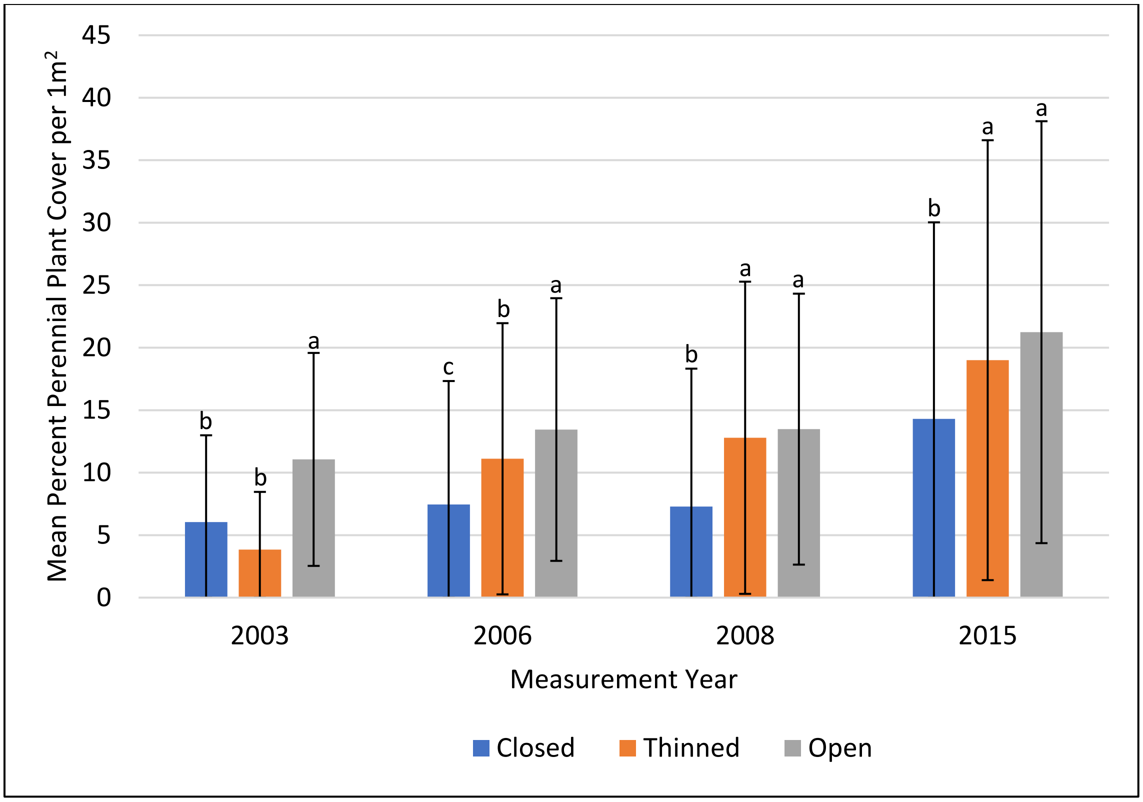

Figure 36. Mean percent perennial plant cover by forest treatment type and year. Error bars represent one standard deviation from the mean. Bars with the same letter within each year are not significant at the α = 0.05 level.

Open canopy conditions had the greatest mean percent perennial plant cover, followed by forest thinning (p = 0.036). Closed canopy conditions had the lowest mean percent perennial cover (p = 0.011). Mean percent perennial plant cover under open canopy conditions in 2006 was 13.45 (SD 10.51). Under closed canopy conditions and forest thinning, it was 7.45 (SD 9.88) and 11.11 (SD 10.85), respectively.

In 2008, mean percent perennial plant cover under open canopy conditions and thinning treatment was not significantly different (p = 0.452). However, closed canopy conditions had significantly lower mean percent perennial plant cover than either open canopy conditions or forest thinning ($p < 0.001$) (Figure 36). Mean percent perennial plant cover under open canopy conditions and forest thinning in 2008 was 13.48 (SD 10.84) and 12.79 (SD 12.49), respectively. Closed canopy conditions had mean percent perennial plant cover of 7.29 (SD 11.03).

In 2015, mean percent perennial plant cover was not significantly different between open canopy conditions and forest thinning although it did approach statistical significance ($p = 0.053$). Mean percent perennial plant cover under open canopy conditions and forest thinning was significantly greater than closed canopy conditions. Open canopy conditions and forest thinning in 2015 had mean percent perennial plant cover of 21.24 (SD 16.88) and 19.00 (SD 17.60), respectively while mean percent perennial plant cover under closed canopy conditions was 14.30 (SD 15.73).

Mean percent perennial plant cover was affected by grazing treatments. Grazing had significantly lower mean percent perennial plant cover than non-grazing ($p < 0.001$). Mean percent perennial plant cover under grazing was 10.49 (SD 10.64) and non-grazing was 13.41 (SD 14.74).

Mean percent perennial plant cover was significantly affected by grazing treatments on soils derived from benmoreite and limestone ($p < 0.001$). However, grazing treatments did not significantly affect mean percent perennial plant cover on basalt-derived soils ($p = 0.953$). Mean percent perennial plant cover under non-grazing on benmoreite-derived soils was 11.25 (SD 14.50), while under grazing it was 7.47 (SD 8.78). Mean percent perennial plant cover under non-grazing on limestone-derived soils was 26.99 (SD 22.50), while under grazing it was 13.73 (SD 12.01). Mean percent perennial plant cover under non-grazing on basalt-derived soils was 9.79 (SD 12.07), while under grazing it was 9.75 (SD 9.40).

Within grazing treatments, mean percent perennial plant cover response was moderated by soil parent material type. Under non-grazing, limestone-derived soils had significantly greater mean percent perennial plant cover than basalt- and benmoreite-derived soils ($p < 0.001$). Mean percent perennial plant cover was not significantly different between basalt- and benmoreite-derived soils ($p = 1.00$). On limestone-derived soils under non-grazing, mean percent perennial plant cover was 17.70 (SD 15.50), while on basalt- and benmoreite-derived soils it was 9.79 (SD 12.07) and 11.25 (SD 14.50), respectively. Under grazing, mean percent perennial plant cover was significantly different among all soil parent material types, with limestone-derived soils having the greatest mean percent perennial plant cover. Benmoreite-derived soils had the second highest mean percent perennial plant cover and basalt-derived soils had the lowest ($p = 0.011$) (Figure 37).

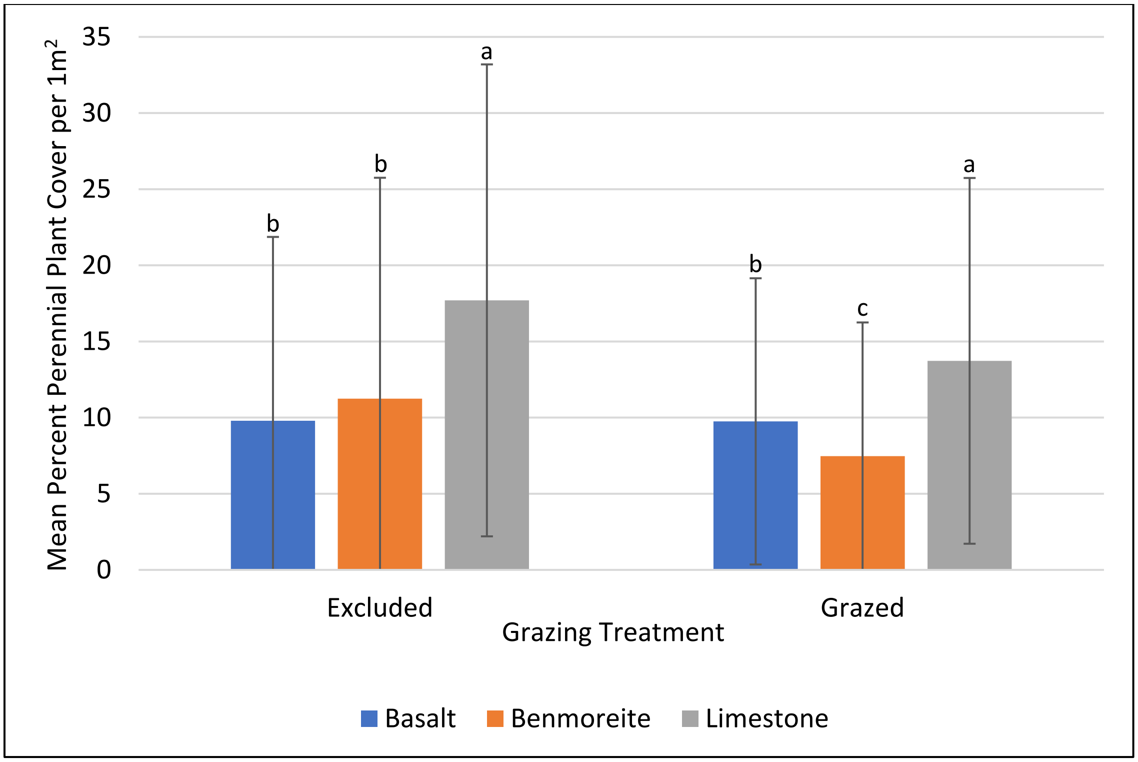

Figure 37. Mean percent perennial plant cover by grazing treatment within soil parent material type. Error bars represent one standard deviation from the mean. Bars with the same letter within each grazing treatment are not significantly different at the $\alpha = 0.05$ level.

A significant two-way interaction between grazing treatment and year affected mean percent perennial plant cover. No significant difference in mean percent perennial plant cover between grazing and non-grazing was found in 2003 ($p = 0.971$) or 2008 ($p = 0.204$). There was a significant difference in mean percent perennial plant cover between the 2006 ($p = 0.035$) and 2015 ($p < 0.001$) measurement years. In 2006, mean percent perennial plant cover under grazing was 9.46 (SD 9.20), while under non-grazing it was

11.98 (SD 11.86). In 2015, mean percent perennial plant cover under grazing and non-grazing was 14.72 (SD 12.66) and 21.84 (SD 19.89) (Figure 38).

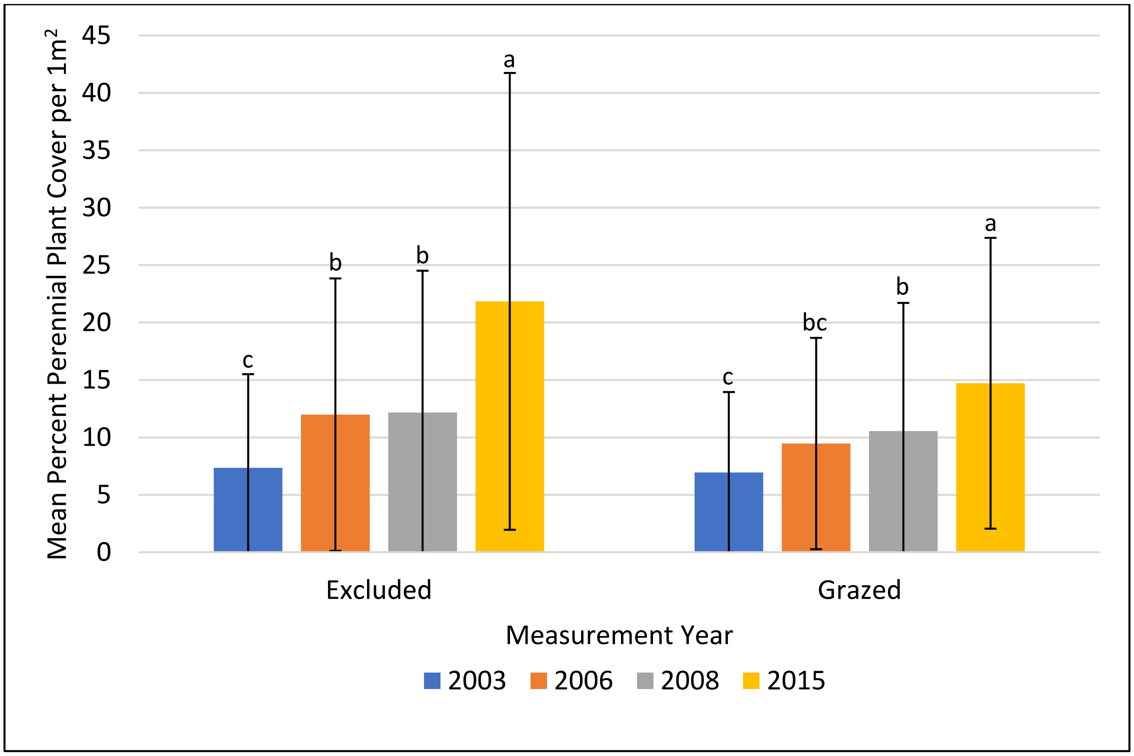

Figure 38. Mean percent perennial plant cover by measurement year within grazing treatment. Error bars represent one standard deviation from the mean. Bars with the same letter within each grazing treatment are not significantly different at the $\alpha = 0.05$ level.

Annual Plant Cover

The only independent variable that contributed significantly to the power of the HLM of mean percent annual plant cover in 2015 was soil parent material type (Table 20).

Table 20. Hierarchical Linear Model (HLM) predictors and their associated R, R-squared, adjusted R-squared, and changes in R-squared values for each independent variable used to analyze mean percent annual plant cover per $1m^2$ for the 2015 measurement year ($\alpha = 0.05$).

					Change Statistics		
Model	R	R-squared	Adjusted R-squared	Std. Error of the Estimate	R-squared Change	F Change	Sig. F Change
1	.002[a]	0.000	-0.002	3.2723	0.000	0.001	0.971
2	.080[b]	0.006	0.002	3.2658	0.006	2.678	0.103
3	.172[c]	0.030	0.023	3.2312	0.023	10.077	0.002

[a] Treatment type

[b] Treatment type plus grazing treatment

[c] Treatment type plus grazing treatment plus soil type

Although annual plant cover was very low overall, limestone-derived soils had significantly greater mean percent annual plant cover than basalt- and benmoreite-derived soils ($p = 0.029$ and $p < 0.001$, respectively). Mean percent annual plant cover on limestone-derived soils was 1.45 (SD 4.98). Basalt- and benmoreite-derived soils had mean percent annual plant cover of 0.54 (SD 1.84) and 0.56 (SD 0.27).

Forest thinning had significantly greater mean percent annual plant cover than open canopy and closed canopy conditions ($p < 0.001$). Mean percent annual plant cover under forest thinning was 1.89 (SD 5.43), while under open and closed conditions, it was 0.12 (SD 0.34) and 0.11 (SD 0.58), respectively.

Mean percent annual plant cover in 2015 was compared to previous measurement years to evaluate annual plant cover responses to forest and grazing treatments over time (Table 21).

Table 21. Hierarchical Linear Model (HLM) predictors and their associated R, R-squared, adjusted R-squared, and changes in R-squared values for each independent variable used to analyze mean percent annual cover at the subplot level (1m^2) for all measurement years ($\alpha = 0.05$).

Model	R	R-squared	Adjusted R-squared	Std. Error of the Estimate	Change Statistics		
					R-squared Change	F Change	Sig. F Change
1	.019[a]	0.000	0.000	2.182	0.000	0.581	0.446
2	.069[b]	0.005	0.004	2.178	0.004	7.329	0.007
3	.102[c]	0.010	0.009	2.172	0.006	9.506	0.002
4	.141[d]	0.020	0.017	2.162	0.009	15.843	0.000

[a] Treatment type

[b] Treatment type plus grazing treatment

[c] Treatment type plus grazing treatment plus soil type

[d] Treatment type plus grazing treatment plus soil type plus year

Grazing, soil parent material type, and measurement year contributed significantly to the HLM of mean percent annual plant cover while forest treatment type did not. However, pairwise comparisons revealed some significant differences in mean percent annual plant cover among forest treatment types. Forest thinning had significantly greater mean percent annual cover than closed canopy and open canopy treatments ($p < 0.001$) while mean percent annual plant cover under closed canopy and open canopy conditions was not significantly different ($p = 0.57$). Mean percent annual plant cover under forest thinning was 0.78 (SD 3.54). Under closed canopy and open canopy conditions, mean percent annual cover was 0.10 (SD 0.94) and 0.23 (SD 0.65), respectively.

Mean percent annual plant cover was also significantly affected by soil parent material types. Limestone-derived soils had significantly greater mean percent annual plant cover than basalt- or benmoreite-derived soils ($p = 0.033$ and $p < 0.001$, respectively). Mean percent annual plant cover on limestone-derived soils was 0.61 (SD 3.23), while on basalt- and benmoreite-derived soils it was 0.30 (SD 1.40) and 0.18 (SD 0.65), respectively. No significant difference in mean percent annual plant cover was found between basalt- and benmoreite-derived soils.

Mean percent annual plant cover was also affected by measurement year. The 2015 measurement year had significantly greater mean percent annual plant cover than all previous measurement years ($p < 0.001$, $p = 0.035$, and $p = 0.018$ for 2003, 2006, and 2008, respectively). Previous measurement years were not significantly different from each other. Mean percent annual plant cover on in 2015 was 0.71 (SD 3.27), while in 2003, 2006, and 2008 it was 0.12 (SD 0.42), 0.34 (SD 1.89), and 0.31 (SD 2.03), respectively (Figure 39).

Mean percent annual vegetation was significantly affected by grazing treatments, with non-grazing having greater mean percent annual plant cover than grazing ($p = 0.013$). Non-grazing had 0.52 (SD 2.92) mean percent annual cover while grazing had 0.27 (SD 1.47).

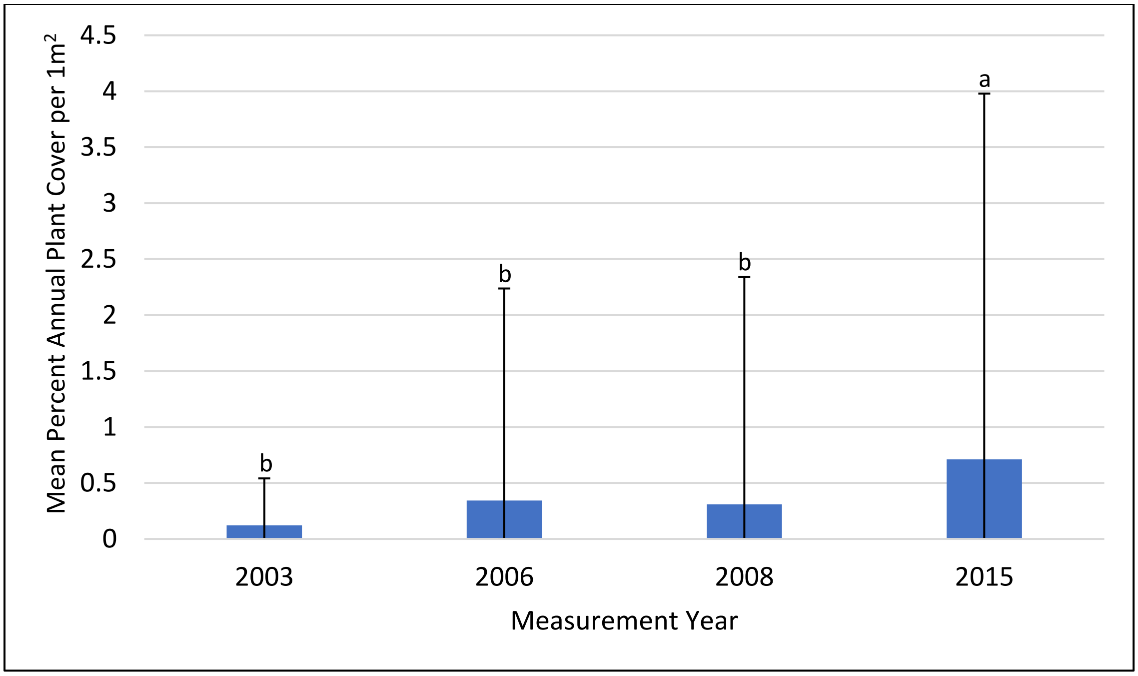

Figure 39. Mean percent annual plant cover by measurement year. Error bars represent one standard deviation from the mean. Bars with the same letter are not significantly different at the α = 0.05 level.

Species Richness

Forest treatment type and soil parent material type contributed significantly to the power of the HLM of mean total species richness in 2015. Grazing treatments were not significant.

Table 22. Hierarchical Linear Model (HLM) predictors and their associated R, R-squared, adjusted R-squared, and change in R-squared values for each independent variable used to analyze mean total species richness per 1m^2 for the 2015 measurement year ($\alpha = 0.05$).

					Change Statistics		
Model	R	R-squared	Adjusted R-squared	Std. Error of the Estimate	R-squared Change	F Change	Sig. F Change
1	.404[a]	0.163	0.161	2.398	0.163	82.289	0.000
2	.412[b]	0.170	0.166	2.392	0.007	3.344	0.068
3	.497[c]	0.247	0.242	2.281	0.077	43.070	0.000

[a] Treatment type

[b] Treatment type plus grazing treatment

[c] Treatment type plus grazing treatment plus soil type

Mean total species richness under open canopy conditions was significantly greater than closed canopy conditions and forest thinning ($p < 0.001$ and $p = 0.008$, respectively). Under open canopy conditions, mean total species richness was 6.87 (SD 3.18), while under closed canopy conditions and forest thinning mean total species richness was 4.29 (SD 2.13) and 5.05 (SD 1.55) (Figure 40).

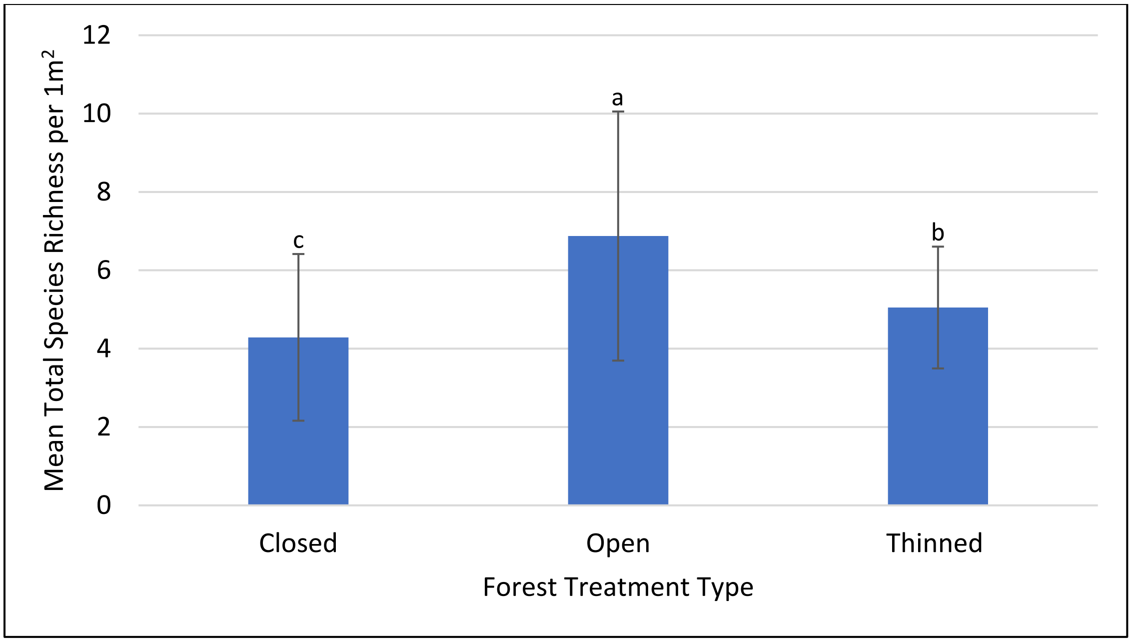

Figure 40. Mean total species richness per $1m^2$ by forest treatment type. Error bars represent one standard deviation from the mean. Bars with the same letter are not significantly different at the $\alpha = 0.05$ level.

Mean total species richness was also affected by soil parent material type at the subplot level. Limestone-derived soils had greater mean total species richness than basalt- and benmoreite-derived soils ($p < 0.001$). Limestone-derived soils had mean total species richness of 6.40 (SD 2.76) while basalt- and benmoreite-derived soils had mean total species richness of 4.88 (SD 1.84) and 4.76 (SD 2.62), respectively.

Finally, mean total species richness per $1m^2$ was affected by three-way interaction between forest treatment type, soil parent material type and grazing treatments in 2015. Under closed canopy conditions, limestone-derived soils with grazing had significantly lower mean total species richness than with non-grazing ($p = 0.002$). Mean total species richness under grazing on limestone-derived soils was 4.67 (SD 1.78), while under non-grazing it was 6.44 (SD 2.10). Under open canopy conditions, basalt-derived soils with grazing also had lower mean total species richness than non-grazing (p = 0.003). Mean total species richness under grazing on basalt-derived soils was 4.22 (SD 0.94), while under non-grazing it was 6.35 (SD 3.16).

Total Species Richness for All Measurement Years

Forest treatment type, soil parent material type and measurement year contributed significantly to the power of the HLM of subplot level mean total species richness, while grazing treatments did not (Table 23). Forest treatment type and soil parent material type contribute more to the predictive power of the HLM than measurement year.

Table 23. Hierarchical Linear Model (HLM) predictors and their associated R, R-squared, adjusted R-squared, and change in R-squared values for each independent variable used to analyze mean total species richness at the subplot level ($1m^2$) for all measurement years ($\alpha = 0.05$).

					Change Statistics		
Model	R	R-squared	Adjusted R-squared	Std. Error of the Estimate	R-squared Change	F Change	Sig. F Change
1	.361[a]	0.130	0.129	3.475	0.130	247.355	0.000
2	.361[b]	0.130	0.129	3.476	0.000	0.360	0.549
3	.486[c]	0.236	0.235	3.258	0.106	229.756	0.000
4	.493[d]	0.243	0.241	3.244	0.007	15.150	0.000

[a] Treatment type

[b] Treatment type plus grazing treatment

[c] Treatment type plus grazing treatment plus soil type

[d] Treatment type plus grazing treatment plus soil type plus year

Forest treatment type significantly affected mean total species richness at the subplot level when all measurement years are considered (p = 0.001). Mean total species richness was greatest under open canopy conditions followed by forest thinning then closed canopy conditions (Figure 41). Mean total species richness under open canopy conditions was 8.07 (SD 4.36). Under forest thinning, mean total species richness was 5.87 (SD 3.01) and under closed canopy conditions it was 4.52 (SD 2.39).

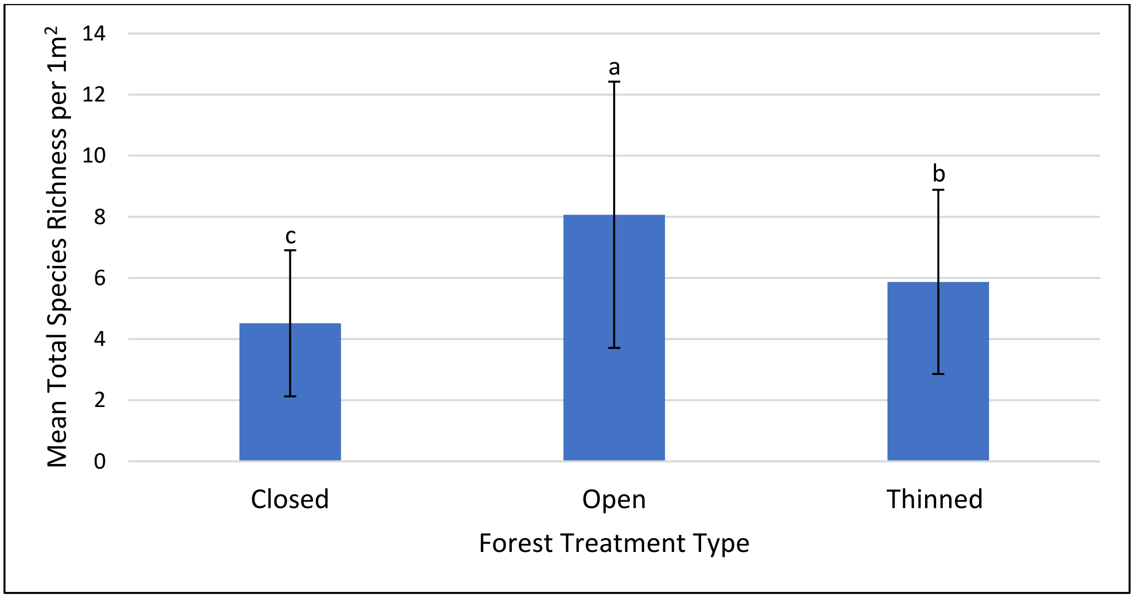

Figure 41. Subplot level mean total species richness by forest treatment type. Error bars represent one standard deviation from the mean. Bars with the same letter are not significantly different at the α = 0.05 level.

Measurement year had a significant effect on mean total species richness at the subplot level when all forest treatment types, grazing treatments, and soil parent material types are considered. Mean total species richness was greatest in 2006, three years after forest treatments were implemented, after which mean total species richness declined. Mean total species richness in 2003 was 5.98 (SD 4.06). In 2006, it had increased significantly to 6.95 (SD 4.16). Mean total species richness declined in 2008 to 6.48 (SD 3.72), then declined further in 2015 to 5.42 (SD 2.62) (Figure 42).

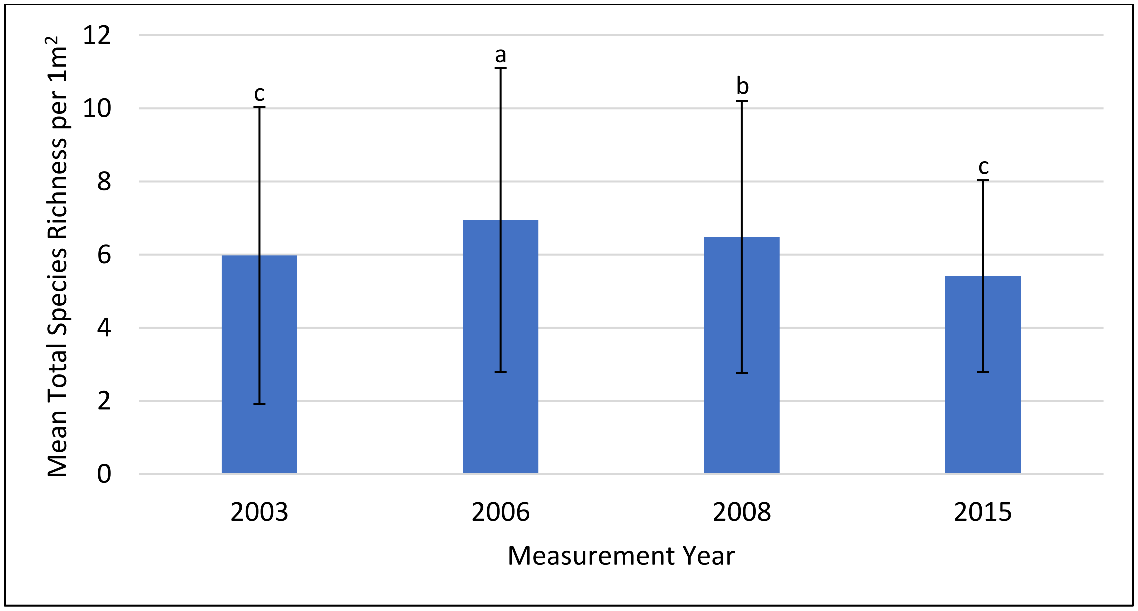

Figure 42. Subplot level mean total species richness by forest treatment type. Error bars represent one standard deviation from the mean. Bars with the same letter are not significantly different at the α = 0.05 level.

Soil parent material type significantly affected mean total species richness at the subplot level when all measurement years, forest treatment types, and grazing treatments were considered. Limestone-derived soils had the greatest mean total species richness followed by basalt-derived soils then those derived from benmoreite. Mean total species richness on limestone-derived soils was 7.82 (SD 4.03), while basalt- and benmoreite-derived soils had mean total species richness values of 5.43 (SD 2.91) and 4.99 (SD 3.23), respectively (Figure 43).

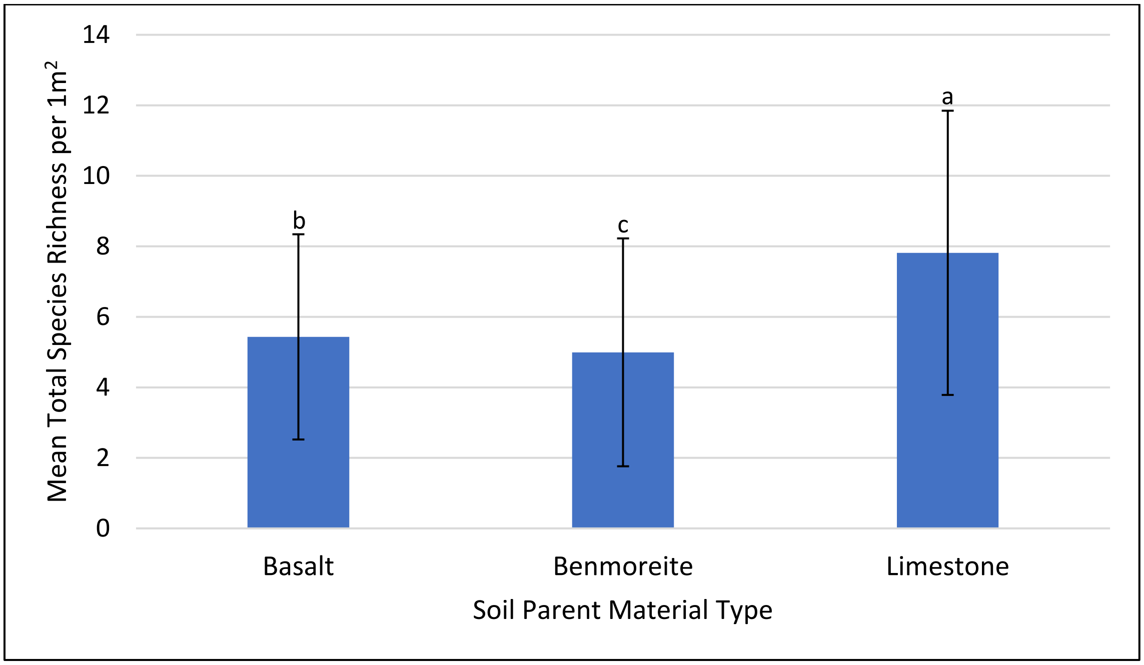

Figure 43. Subplot level mean total species richness by soil parent material type. Error bars represent one standard deviation from the mean. Bars with the same letter are not significantly different at the $\alpha = 0.05$ level.

Mean total species richness was significantly affected by a forest treatment type and measurement year interaction at the subplot level. In 2003 (pre-treatment), mean total species richness under closed canopy conditions was significantly lower than under open canopy conditions ($p < 0.001$), but was not significantly different from mean total species richness under forest thinning ($p = 0.195$). In 2006 and 2008 mean total species richness was greatest under open canopy conditions ($p < 0.001$) and lowest under closed canopy conditions ($p < 0.001$) with thinning being intermediate. In 2015, mean total species richness under open canopy conditions was significantly greater than under closed canopy conditions and forest thinning ($p < 0.001$). Forest thinning was not significantly different from closed canopy conditions ($p = 0.096$) (Table 24).

Table 24. Mean total species richness and standard deviations per $1m^2$ by measurement year and forest treatment type.

		Treatment Type	
Year	Closed	Thinned	Open
2003	5.03 (3.02)	4.17 (2.69)	8.41 (4.65)
2006	5.15 (3.25)	6.63 (2.99)	8.97 (4.99)
2008	4.73 (3.00)	6.49 (3.12)	8.01 (4.15)
2015	4.29 (2.13)	5.05 (1.55)	6.87 (3.18)

No significant difference in mean total species richness was found between measurement years for closed canopy conditions ($p > 0.208$ for all comparisons). Under forest thinning, the 2006 and 2008 measurement years had greater total species richness than was found in the 2003 and 2015 measurement years ($p < 0.001$).

Under open canopy conditions, mean total species richness was significantly lower in 2015 than all previous measurement years ($p = 0.014$ for 2003, and $p < 0.001$ for 2006 and 2008). Mean total species richness under open canopy conditions was greater in 2006 than in 2008 or 2015 ($p = 0.008$ and $p < 0.001$, respectively). Mean total species richness under open canopy conditions in 2003 was not significantly different than in 2006 ($p < 0.001$).

Mean total species richness was affected by forest treatment type within soil parent material types at the subplot level. On basalt-derived soils, open canopy conditions had significantly greater mean total species richness than closed canopy conditions and forest thinning ($p < 0.001$). Mean total species richness was not significantly different between closed canopy conditions and forest thinning ($p = 0.41$) (Table 25).

Table 25. Mean total species richness and standard deviations per $1m^2$ by forest treatment type within soil parent material type.

	Treatment Type		
Soil Type	Closed	Thinned	Open
Basalt	4.58 (2.08)	5.13 (3.06)	6.57 (3.11)
Benmoreite	2.48 (1.54)	5.33 (2.78)	6.60 (3.42)
Limestone	6.69 (2.78)	6.22 (2.66)	10.53 (4.78)

Limestone-derived soils had the greatest overall mean total species richness at the subplot level, with open canopy conditions having the greatest mean total species richness among forest treatments. Forest thinning and closed canopy conditions, while not significantly different from each other, had significantly lower ($p < 0.001$) mean total species richness than open canopy conditions. Mean total species richness on basalt- and benmoreite-derived soils was not significantly different (p = 0.198). This same trend held under forest thinning treatment.

Grazing treatments had no significant effect on mean total species richness at the subplot level (p = 0.266).

Total Species Richness per $9m^2$ in 2015

Forest treatment type and soil parent material type contributed significantly to the predictive power of the HLM at the measurement plot level in 2015, while grazing did not (Table 26).

Table 26. Hierarchical Linear Model (HLM) predictors and their associated R, R-squared, adjusted R-squared, and changes in R-squared values for each independent variable used to analyze mean total species richness at the measurement plot level ($9m^2$) in 2015 (α = 0.05).

					Change Statistics		
Model	R	R-squared	Adjusted R-squared	Std. Error of the Estimate	R-squared Change	F Change	Sig. F Change
1	.418[a]	0.175	0.157	4.868	0.175	9.753	0.003
2	.431[b]	0.186	0.149	4.890	0.011	0.589	0.447
3	.510[c]	0.260	0.210	4.713	0.075	4.439	0.041

[a] Treatment type

[b] Treatment type plus grazing treatment

[c] Treatment type plus grazing treatment plus soil type

Mean total species richness was greater under open canopy conditions than closed canopy conditions (p = 0.021) but was not significantly different from mean total species richness under forest thinning (p = 0.148). Mean total species richness under open canopy conditions was 16.94 (SD 6.27). Under forest thinning and closed canopy conditions, mean total species richness was 13.13 (SD 2.53) and 11.56 (SD 5.11), respectively.

Limestone-derived soils had significantly greater mean total species richness than benmoreite-derived soils (p = 0.031). Mean total species richness on basalt-derived soils was not significantly different from either limestone- or benmoreite-derived soils (p = 0.595 and p = 0.803, respectively). Mean total species richness on limestone-derived soils was 16.11 (SD 5.68). On basalt- and benmoreite-derived soils it was 13.75 (SD 2.63) and 11.72 (SD 5.55), respectively.

Pairwise comparisons revealed significant differences in mean total species richness by forest treatment type within soil parent material type. On benmoreite-derived soils closed canopy conditions had significantly lower mean total species richness than open canopy conditions (p = 0.045). However, mean total species richness was not significantly different between closed canopy conditions and forest thinning treatments (p = 0.117) or open canopy conditions and forest thinning (p = 0.235). Mean total species richness on benmoreite-derived soils under open canopy conditions was 14.50 (SD 7.56). Forest thinning and closed canopy conditions on benmoreite-derived soils had mean total species richness of 13.33 (SD 2.25) and 7.33 (SD 2.66), respectively. Mean total species richness had a different trend on limestone-derived soils. Open canopy conditions had significantly greater mean total species richness than forest thinning (p = 0.022) and mean total species richness was not significantly different between open canopy conditions and closed canopy conditions ($p = 0.152$). There was no significant difference in mean total species richness on basalt-derived soils in pairwise comparisons of forest treatment type within soil parent material type (Figure 44).

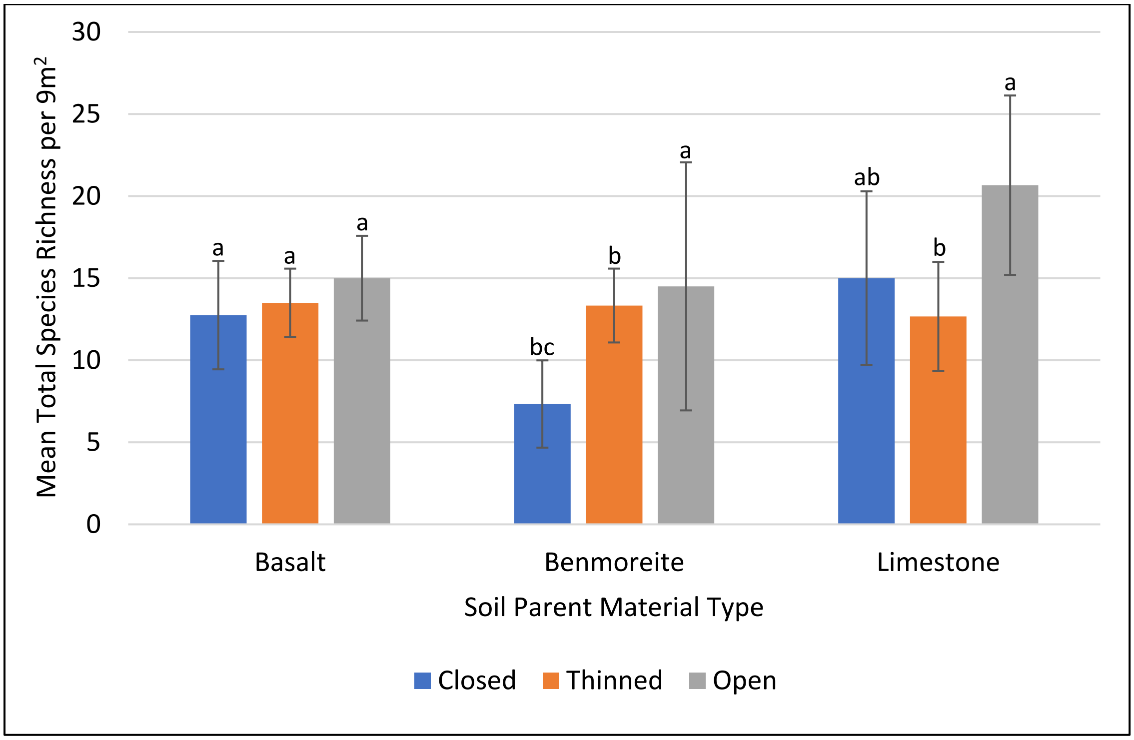

Figure 44. Mean total species richness per 9m^2 by forest treatment type within soil parent material type. Error bars represent one standard deviation from the mean. Bars with the same letter within soil parent material type are not significantly different at the $\alpha = 0.05$ level.

Grazing treatments has no significant effect on mean total species richness at the measurement plot level in 2015.

Total Species Richness per $9m^2$ for All Measurement Years

Forest treatment type and soil parent material type were significant, but weak predictors of total species richness at the measurement plot level while grazing treatments and measurement year do not contribute significantly to the power of the HLM (Table 27).

Table 27. Hierarchical Linear Model (HLM) predictors and their associated R, R-squared, adjusted R-squared, and changes in R-squared values for each independent variable used to analyze mean total species richness at the measurement plot level ($9m^2$) for all measurement years ($\alpha = 0.05$).

					Change Statistics		
Model	R	R-squared	Adjusted R-squared	Std. Error of the Estimate	R-squared Change	F Change	Sig. F Change
1	.399[a]	0.159	0.155	5.772	0.159	35.583	0.000
2	.399[b]	0.159	0.150	5.786	0.000	0.053	0.818
3	.502[c]	0.252	0.240	5.474	0.092	22.967	0.000
4	.505[d]	0.255	0.239	5.478	0.003	0.725	0.396

[a] Treatment type

[b] Treatment type plus grazing treatment

[c] Treatment type plus grazing treatment plus soil type

[d] Treatment type plus grazing treatment plus soil type plus year

Mean total species richness was affected by forest treatment type when all years and grazing treatments are considered. Open canopy conditions had significantly greater mean total species richness than closed canopy conditions ($p < 0.001$) and forest thinning ($p = 0.046$). Mean total species richness under open canopy conditions was 17.22 (SD 7.38). Under forest thinning, mean total species richness was 13.05 (SD 4.95) and under closed canopy conditions it was 11.05 (SD 5.33).

Mean total species richness was affected by soil parent material type. Limestone-derived soils had significantly greater mean total species richness than basalt- and benmoreite-derived soils (p = 0.006 and p = 0.027, respectively). Mean total species richness was not significantly different between basalt- and benmoreite-derived soils (p = 0.069). Limestone-derived soils had mean total species richness of 17.36 (SD 5.86). Basalt- and benmoreite-derived soils had mean total species richness of 12.67 (SD 5.03) and 10.61 (SD 6.75).

There were no statistically significant differences in mean total species richness between measurement years or grazing treatments at the measurement plot level. Additionally, there were no statistically significant interactions among forest treatments, grazing treatments, soil parent material types or measurement year at the measurement plot level.

Species Richness of Grasses per 1m^2 in 2015

Forest treatment type and soil parent material type contributed significantly to the power of the HLM of mean species richness of grasses at the subplot level in 2015. Grazing treatment was not a significant model predictor (Table 28).

Table 28. Hierarchical Linear Model (HLM) predictors and their associated R, R-squared, adjusted R-squared, and change in R-squared values for each independent variable used to analyze mean species richness of grasses per $1m^2$ for the 2015 measurement year ($\alpha = 0.05$).

					Change Statistics		
Model	R	R-squared	Adjusted R-squared	Std. Error of the Estimate	R-squared Change	F Change	Sig. F Change
1	.417[a]	0.174	0.172	1.151	0.174	88.790	0.000
2	.417[b]	0.174	0.170	1.153	0.000	0.064	0.800
3	.452[c]	0.204	0.199	1.133	0.030	16.049	0.000

[a] Treatment type

[b] Treatment type plus grazing treatment

[c] Treatment type plus grazing treatment plus soil type

Open canopy conditions had significantly greater mean species richness of grasses than was found under forest thinning and closed canopy conditions ($p < 0.001$). Forest thinning and closed canopy conditions were not significantly different but did approach statistical significance ($p = 0.059$). Mean species richness of grasses under open canopy conditions was 3.20 (SD 1.34). Under forest thinning and closed canopy conditions it was 2.23 (SD 0.93) and 1.91 (SD 1.13), respectively.

Limestone-derived soils had significantly greater mean species richness of grasses than basalt- and benmoreite-derived soils ($p = 0.02$ and $p < 0.001$, respectively). Mean species richness of grasses was not significantly different between basalt- and benmoreite-derived soils ($p = 0.97$). Mean species richness of grasses on limestone-derived soils was 2.75 (SD 1.30) while on basalt- and benmoreite-derived soils it was 2.28 (SD 1.18) and 2.27 (SD 1.24), respectively.

Grazing treatments did not have a significant effect on mean species richness of grasses, although it did approach significance (p = 0.076). Mean species richness of grasses under grazing was 2.44 (SD 1.11) and under non-grazing it was 2.48 (SD 1.41).

Species Richness of Grasses per 1m^2 for All Measurement Years

All independent variables except grazing treatments contributed significantly to the power of the HLM of mean species richness of grasses at the measurement plot level (Table 29).

Table 29. Hierarchical Linear Model (HLM) predictors and their associated R, R-squared, adjusted R-squared, and changes in R-squared values for each independent variable used to analyze mean species richness of grasses at the subplot level (1m^2) for all measurement years (α = 0.05).

					Change Statistics		
Model	R	R-squared	Adjusted R-squared	Std. Error of the Estimate	R-squared Change	F Change	Sig. F Change
1	.399[a]	0.159	0.159	1.203	0.159	313.958	0.000
2	.401[b]	0.161	0.160	1.202	0.001	2.651	0.104
3	.461[c]	0.213	0.211	1.165	0.052	109.306	0.000
4	.480[d]	0.231	0.229	1.152	0.018	38.020	0.000

[a] Treatment type

[b] Treatment type plus grazing treatment

[c] Treatment type plus grazing treatment plus soil type

[d] Treatment type plus grazing treatment plus soil type plus year

Mean species richness of grasses was affected by forest treatments. Open canopy conditions had significantly greater mean species richness of grasses than forest thinning and closed canopy conditions ($p < 0.001$).

Soil parent material type significantly affected mean species richness of grasses at the subplot level. Limestone-derived soils had significantly greater mean species richness of grasses than basalt- and benmoreite-derived soils ($p < 0.001$). Mean species richness of grasses on basalt- and benmoreite-derived soils was not significantly different ($p = 0.141$). Mean species richness of grasses on limestone-derived soils was 2.54 (SD 1.31). It was 1.85 (SD 1.29) on basalt-derived soils and 2.07 (SD 1.24) on benmoreite-derived soils.

Measurement year significantly affected mean species richness of grasses. The 2015 measurement year had significantly greater mean species richness of grasses than all previous measurement years ($p < 0.001$). Mean species richness of grasses in 2003 was not significantly different from 2006 ($p = 0.401$) but was significantly different from 2008 ($p = 0.042$). Mean species richness of grasses in 2006 was not significantly different from 2008 ($p = 0.076$) (Figure 45).

Mean species richness of grasses in 2003 was 2.02 (SD 1.28). In 2006, it was 2.11 (SD 1.37). In 2008, it was 2.20 (SD 1.29), and in 2015 it had increased significantly to 2.46 (SD 1.27). While the increase was found to be statistically significant numerically in 2015 in comparison to previous sampling years, it is important to note that the values of mean species richness of grasses for all measurement years are quite low, representing fewer than three species per 1m^2 and leading to correspondingly high standard deviations.

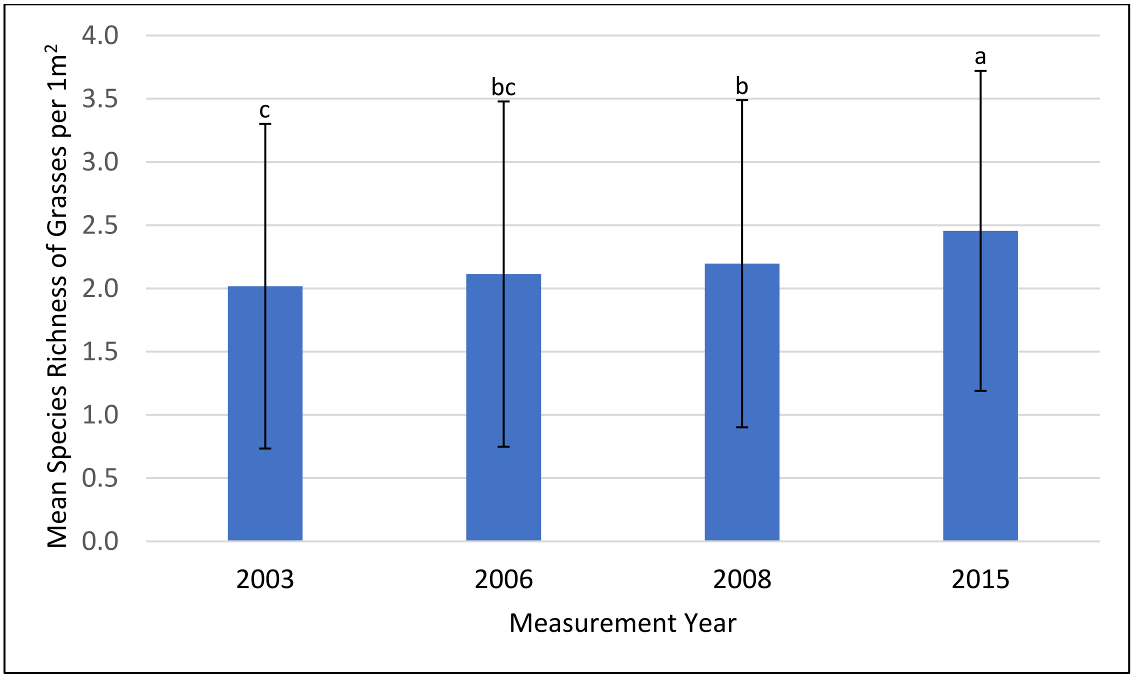

Figure 45. Mean species richness of grasses per 1m^2 by measurement year. Error bars represent one standard deviation from the mean. Bars with the same letter are not significantly different at the $\alpha = 0.05$ level.

Grazing treatment was not a significant predictor of variation in mean species richness of grasses in the HLM. However, pairwise comparisons using Bonferroni correction for multiple comparisons indicated grazing treatments did significantly affected mean species richness of grasses ($p = 0.046$). Mean species richness of grasses under grazing was 2.25 (SD 1.30) and under non-grazing it was 2.15 (SD 1.33).

Mean species richness of grasses was significant affected by a two-way interaction between grazing treatments and forest treatments. Under closed canopy conditions, non-grazing had significantly greater mean species richness of grasses than grazing treatment ($p = 0.049$). Under both thinning and open canopy conditions, non-grazing had significantly lower mean species richness of grasses than grazing ($p = 0.006$ and $p = 0.005$, respectively) (Figure 46).

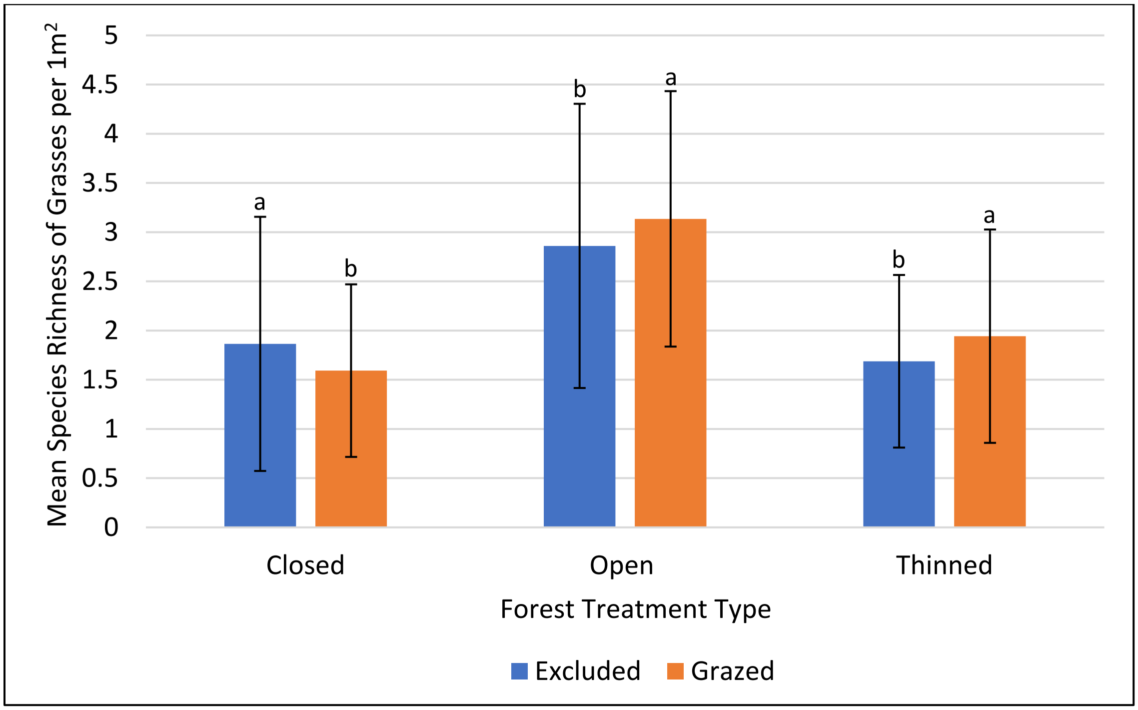

Figure 46. Mean species richness of grasses per $1m^2$ by grazing treatment within forest treatment type. Error bars represent one standard deviation from the mean. Bars with different letters within forest treatment type are significantly different at the $\alpha = 0.05$ level.

Mean species richness of grasses under closed canopy conditions with grazing was 1.59 (SD 0.88) while under non-grazing it was 1.86 (SD 1.29). Under open canopy conditions with non-grazing, mean species richness of grasses was 2.86 (SD 1.44) while under grazing it was 3.14 (SD 1.30). Forest thinning without grazing yielded mean species richness of grasses of 1.69 (SD 0.87), while under grazing it was 1.94 (SD 1.08).

There were no other statistically significant differences in the effects of forest treatments, soil parent material type, grazing treatments, or measurement year on mean species richness of grasses at the subplot (1m^2) level.

Species Richness of Grasses per 9m^2 in 2015

Mean species richness of grasses for the 2015 was also evaluated at the measurement plot level to understand the effects of forest thinning, soil parent material type, and grazing treatments at different scale (Table 30). Like the HLM of subplot level (1m^2) mean species richness of grasses, forest treatment type and soil parent material type contributed significantly to the power of the HLM at the measurement plot level (9m^2) for the 2015 data only. Grazing treatments did not significantly affect mean species richness of grasses.

Table 30. Hierarchical Linear Model (HLM) predictors and their associated R, R-squared, adjusted R-squared, and change in R-squared values for each independent variable used to analyze mean species richness of grasses at the measurement plot level ($9m^2$) in 2015 ($\alpha = 0.05$).

					Change Statistics		
Model	R	R-squared	Adjusted R-squared	Std. Error of the Estimate	R-squared Change	F Change	Sig. F Change
1	.412[a]	0.170	0.152	1.383	0.170	9.411	0.004
2	.418[b]	0.175	0.138	1.394	0.005	0.268	0.607
3	.507[c]	0.257	0.206	1.338	0.082	4.856	0.033

[a] Treatment type

[b] Treatment type plus grazing treatment

[c] Treatment type plus grazing treatment plus soil type

Open canopy conditions had significantly greater mean species richness of grasses than closed canopy conditions (p = 0.025). The effect of closed canopy conditions was not significantly different from thinning (p = 1.00). Mean species richness of grasses under thinning was also not significantly different from open canopy conditions (p = 0.133). Mean species richness of grasses under closed canopy conditions was 3.88 (SD 1.31). Under forest thinning and open canopy conditions it was 4.19 (SD 0.98) and 5.38 (SD 1.75), respectively.

Limestone-derived soils had significantly greater mean species richness of grasses than soils derived from benmoreite (p = 0.032). No significant difference in mean species richness of grasses was found between limestone- and basalt-derived soils (p = 0.346) or basalt- and benmoreite-derived soils (p = 1.00). Mean species richness of grasses on limestone-derived soils was 5.11 (SD 1.61). On basalt- and benmoreite-derived soils it was 4.33 (SD 1.16) and 3.94 (SD 1.43), respectively.

Grazing treatments did not significantly affect mean species richness of grasses at the measurement plot level in 2015.

Species Richness of Grasses per Measurement Plot ($9m^2$) for All Measurement Years

Mean species richness of grasses at the measurement plot level was compared to previous measurement years to evaluate responses to forest and grazing treatments and soil parent material types over time (Table 31).

Table 31. Hierarchical Linear Model (HLM) predictors and their associated R, R-squared, adjusted R-squared, and changes in R-squared values for each independent variable used to analyze mean species richness of grasses at the measurement, plot level ($9m^2$) for all measurement years (α = 0.05).

Model	R	R-squared	Adjusted R-squared	Std. Error of the Estimate	Change Statistics		
					R-squared Change	F Change	Sig. F Change
1	.401[a]	0.161	0.157	1.545	0.161	36.094	0.000
2	.408[b]	0.166	0.157	1.545	0.005	1.181	0.279
3	.485[c]	0.236	0.223	1.483	0.069	16.831	0.000
4	.532[d]	0.284	0.268	1.440	0.048	12.396	0.001

[a] Treatment type

[b] Treatment type plus grazing treatment

[c] Treatment type plus grazing treatment plus soil type

[d] Treatment type plus grazing treatment plus soil type plus year

All model estimators except grazing treatments contributed significantly to the power of the HLM of mean species richness of grasses at the measurement plot level. Open canopy conditions had greater mean species richness of grasses than thinning or closed canopy conditions ($p < 0.001$). There was no significant difference in mean species richness of grasses between forest thinning and closed canopy conditions ($p = 0.94$). Mean species richness of grasses under open canopy conditions was 4.89 (SD 1.94) while forest thinning, and closed canopy conditions had values of 3.55 (SD 1.33) and 3.24 (SD 1.21).

Limestone-derived soils had greater mean species richness of grasses than basalt- and benmoreite-derived soils ($p=0.016$, and $p < 0.001$, respectively). There was no significant difference between the effects of basalt- and benmoreite-derived soils on mean species richness of grasses ($p = 0.952$). Mean species richness of grasses on limestone-derived soils was 4.53 (SD 1.63). On basalt- and benmoreite-derived soils it was 3.69 (SD 1.56) and 3.40 (SD 1.64), respectively.

The 2015 measurement year had significantly greater mean species richness of grasses than the 2003 measurement year ($p = 0.011$). However, 2006, 2008, And 2015 were not significantly different in their effect on mean species richness of grasses. Mean species richness of grasses in 2003 was 3.47 (SD 1.61). In 2006, it had increased significantly to 3.81 (SD 1.76) and remained static in 2008 at 3.83 (SD 1.74) and 2015 at 4.48 (SD 1.50) which was not significantly different from 2006 or 2008 values.

Open canopy conditions had greater mean species richness of grasses than closed canopy conditions and forest thinning. Mean species richness of grasses under closed canopy conditions and forest thinning was not significantly different.

Mean species richness of grasses was only affected by grazing through a two-way interaction with forest treatment type when all measurement years were considered. Non-grazing under closed canopy conditions yielded greater mean species richness of grasses than grazing. However, under open canopy conditions and forest thinning, this trend was reversed with grazing resulting in greater mean species richness of grasses than non-grazing.

Forb Species Richness per 1m^2 in 2015

All model predictors contributed significantly to the power of the HLM of mean forb species richness in 2015 (Table 32).

Table 32. Hierarchical Linear Model (HLM) predictors and their associated R, R-squared, adjusted R-squared, and changes in R-squared values for each independent variable used to analyze mean forb species richness per 1m^2 for the 2015 measurement year ($\alpha = 0.05$).

					Change Statistics		
Model	R	R-squared	Adjusted R-squared	Std. Error of the Estimate	R-squared Change	F Change	Sig. F Change
1	.248[a]	0.062	0.059	1.747	0.062	27.754	0.000
2	.276[b]	0.076	0.072	1.736	0.014	6.572	0.011
3	.452[c]	0.204	0.198	1.613	0.128	67.415	0.000

[a] Treatment type

[b] Treatment type plus grazing treatment

[c] Treatment type plus grazing treatment plus soil type

Mean forb species richness at the subplot level was affected by forest treatment type in 2015. Mean forb species richness under open canopy conditions was significantly greater than closed canopy conditions and forest thinning ($p < 0.001$ and $p = 0.004$, respectively). There was no significant difference in mean forb species richness from closed canopy conditions and forest thinning ($p = 0.60$). Mean forb species richness in 2015 under open canopy conditions was 2.95 (SD 2.14). Under closed canopy conditions and forest thinning it was 1.86 (SD 1.64) and 2.25 (SD 1.36), respectively.

Limestone-derived soils had greater mean forb species richness than basalt- or benmoreite-derived soils ($p < 0.001$). Mean forb species richness on basalt- and benmoreite-derived soils was not significantly different ($p = 0.19$). Limestone-derived soils had mean forb species richness of 3.20 (SD 1.98) while basalt- and benmoreite-derived soils had mean forb species richness of 1.80 (SD 1.22) and 1.85 (SD 1.60), respectively.

Grazing treatments significantly affected mean forb species richness at the subplot level in 2015. Non-grazing had significantly greater mean forb species richness than grazing ($p = 0.017$). Mean forb species richness under non-grazing was 2.58 (SD 2.03) and under grazing it was 2.14 (SD 1.52).

Forb Species Richness per 1m^2 for All Measurement Years

Forest treatment type, soil parent material type and measurement year contributed significantly to the power of the HLM of subplot level mean forb species richness (Table 33).

Table 33. Hierarchical Linear Model (HLM) predictors and their associated R, R-squared, adjusted R-squared, and changes in R-squared values for each independent variable used to analyze mean forb species richness at the subplot level (1m^2) for all measurement years ($\alpha = 0.05$).

					Change Statistics		
Model	R	R-squared	Adjusted R-squared	Std. Error of the Estimate	R-squared Change	F Change	Sig. F Change
1	.278[a]	0.077	0.076	2.678	0.077	138.107	0.000
2	.278[b]	0.077	0.076	2.678	0.000	0.750	0.387
3	.460[c]	0.211	0.210	2.478	0.134	280.205	0.000
4	.480[d]	0.231	0.229	2.447	0.020	42.031	0.000

[a] Treatment type

[b] Treatment type plus grazing treatment

[c] Treatment type plus grazing treatment plus soil type

[d] Treatment type plus grazing treatment plus soil type plus year

Forest treatment type and soil parent material type contribute more to the predictive power of the HLM than measurement year. The low R-squared value and adjusted R-squared values with each additional predictor indicates substantial variability in mean total species richness that the independent variables do not explain.

Mean forb species richness was greatest under open canopy conditions ($p < 0.001$) followed by forest thinning ($p < 0.001$) then closed canopy conditions ($p = 0.001$). Mean forb species richness under open canopy conditions was 4.26 (SD 3.27). Under forest thinning, it was 2.92 (SD 2.36) and under closed canopy conditions it was 2.37 (SD 2.22) (Figure 47).

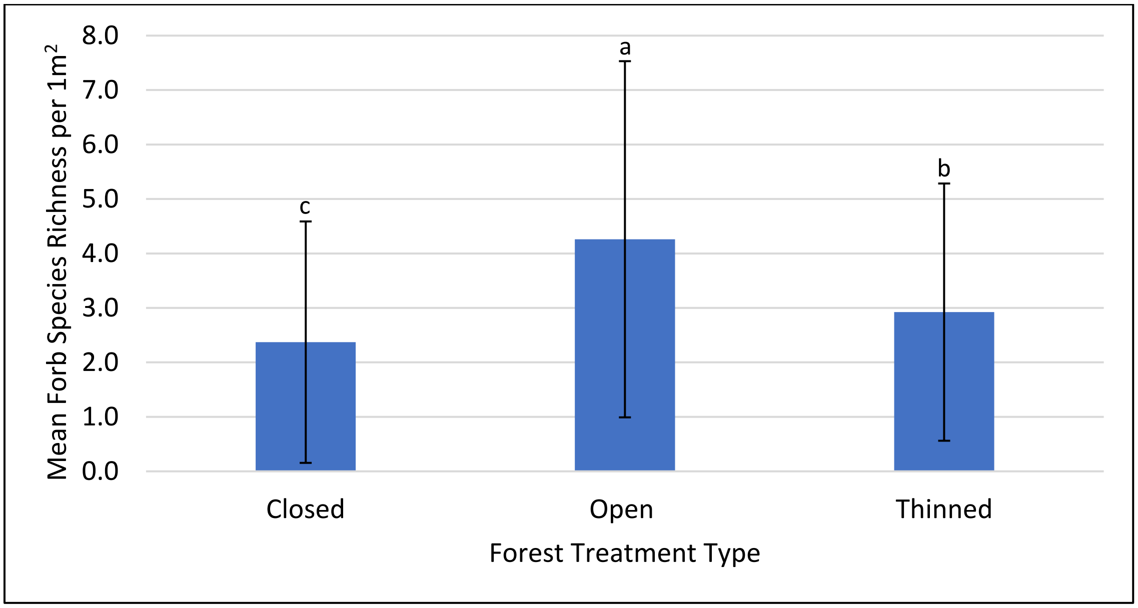

Figure 47. Mean forb species richness per 1m^2 by forest treatment type when all measurement years are considered. Error bars represent one standard deviation from the mean. Bars with different letter are significantly different at the $\alpha = 0.05$ level.

Limestone-derived soils had the greatest mean forb species richness ($p = 0.006$). Mean forb species richness on basalt-derived soils was intermediate ($p < 0.001$) and soils derived from benmoreite had the lowest ($p < 0.001$). Mean forb species richness on limestone-derived soils was 4.56 (SD 3.14), while basalt- and benmoreite-derived soils had mean forb species richness values of 2.52 (SD 2.14) and 2.22 (SD 2.11), respectively (Figure 48).

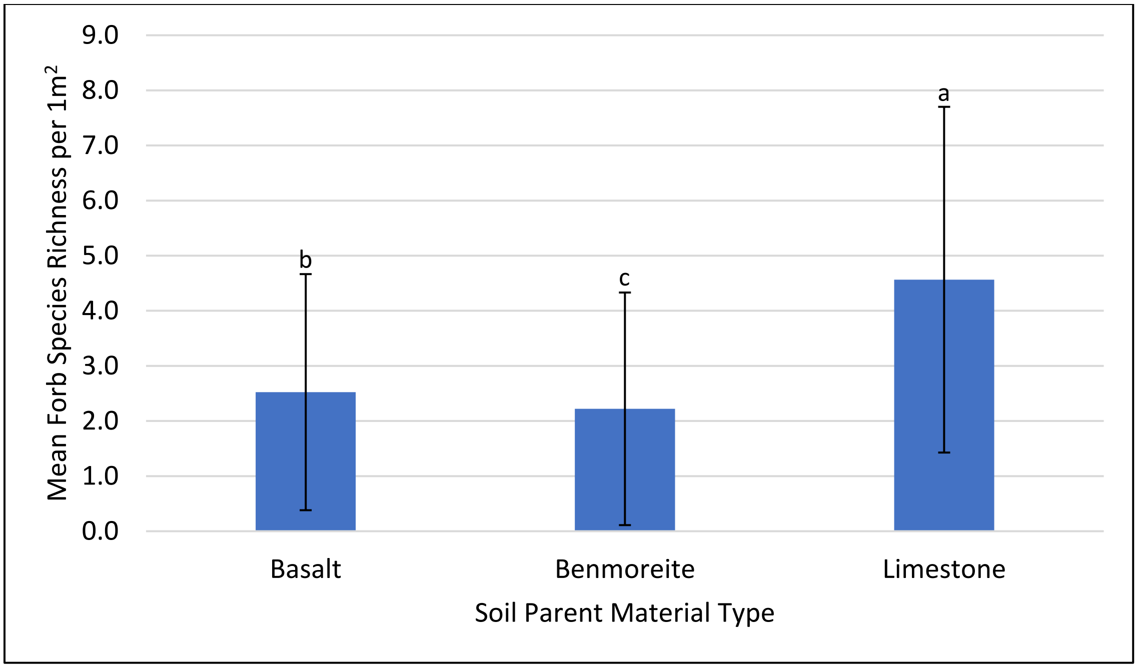

Figure 48. Mean forb species richness per $1m^2$ by soil parent material type across all measurement years. Error bars represent one standard deviation from the mean. Bars with different letters are significantly different at the α = 0.05 level.

The 2006 and 2008 measurement years were not significantly different (p = 1.00) but both had greater mean forb species richness than the 2003 and 2015 measurement years (p = 0.029 and p = 0.023, respectively). Mean forb species richness in 2003 was intermediate and the 2015 measurement year had the lowest mean forb species richness. Mean forb species richness in 2006 and 2008 was 3.57 (SD 3.06) and 3.64 (SD 2.83), respectively. The 2003 measurement year had mean forb species richness of 3.30 (SD 3.09) and 2015 had 2.36 (SD 1.80) (Figure 49).

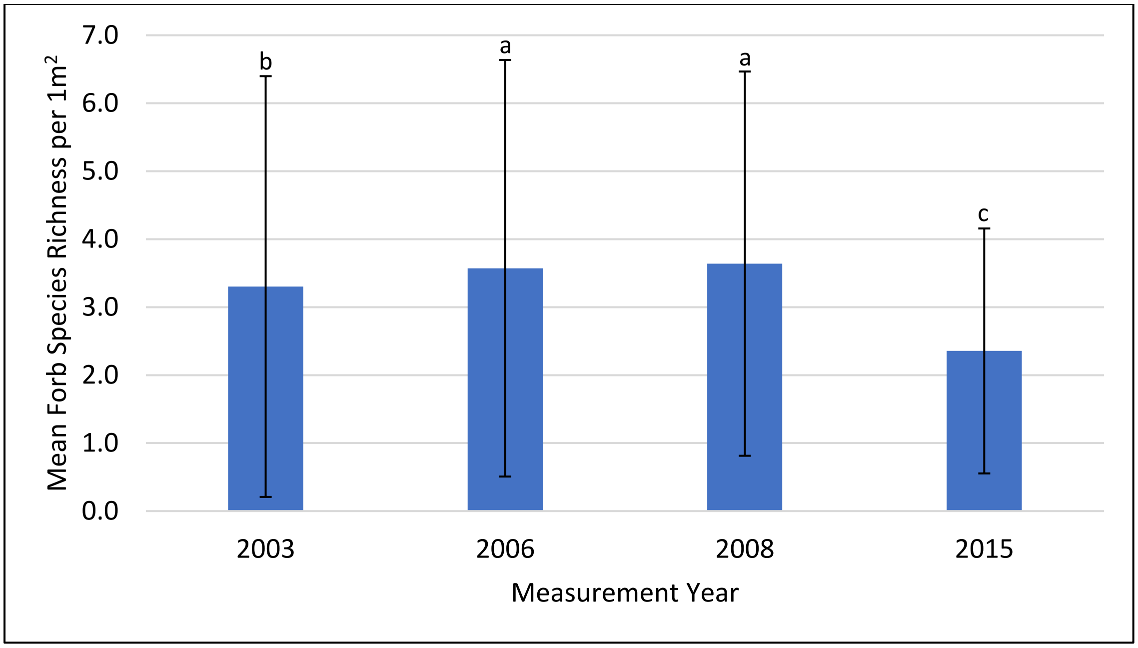

Figure 49. Mean forb species richness per 1m^2 by measurement year. Error bars represent one standard deviation from the mean. Bars with the same letter are not significantly different at the α = 0.05 level.

Forb Species Richness per 9m^2 in 2015

Forest treatment type and soil parent material type contributed significantly to the power of the HLM of mean forb species richness at the measurement plot level in 2015, while grazing treatment was not significant (Table 34).

Table 34. Hierarchical Linear Model (HLM) predictors and their associated R, R-squared, adjusted R-squared, and changes in R-squared values for each independent variable used to analyze mean forb species richness at the measurement plot level (9m^2) in 2015 (α = 0.05).

					Change Statistics		
Model	R	R-squared	Adjusted R-squared	Std. Error of the Estimate	R-squared Change	F Change	Sig. F Change
1	.316[a]	0.100	0.081	3.908	0.100	5.117	0.028
2	.338[b]	0.114	0.075	3.920	0.014	0.717	0.401
3	.484[c]	0.234	0.182	3.685	0.120	6.900	0.012

[a] Treatment type

[b] Treatment type plus grazing treatment

[c] Treatment type plus grazing treatment plus soil type

There were no significant differences in mean forb species richness among forest treatment types or grazing treatments at the measurement plot level in 2015. However, soil parent material type did significantly affect mean forb species richness. Limestone-derived soils had significantly greater mean forb species richness than benmoreite-derived soils. Limestone- and basalt-derived soils were not significantly different (p = 0.239) nor were basalt- and benmoreite derived soils (p = 1.00) (Figure 50).

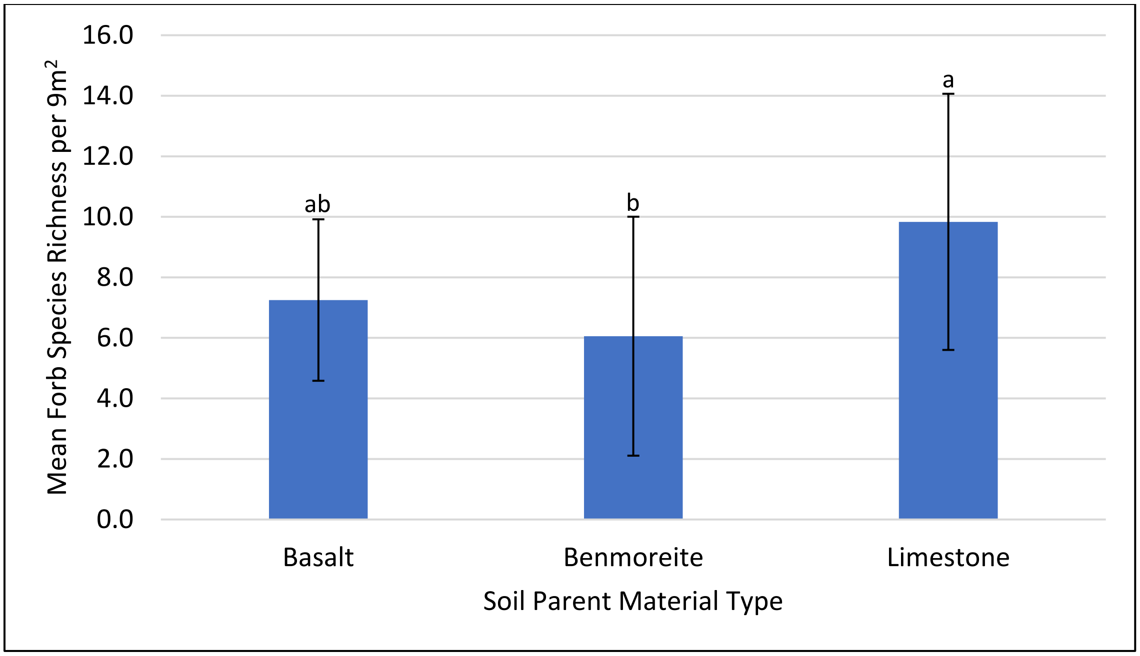

Figure 50. Mean forb species richness per $9m^2$ by soil parent material type. Error bars represent one standard deviation from the mean. Bars with the same letter are not significantly different at the $\alpha = 0.05$ level.

Mean forb species richness on limestone-derived soils was 9.83 (SD 4.23) while mean forb species richness on basalt- and benmoreite-derived soils was 7.25 (SD 2.67) and 6.06 (SD 3.95), respectively.

Forb Species Richness per $9m^2$ for All Measurement Years

Forest treatment type and soil parent material type are significant predictors of mean forb species richness at the measurement plot level while grazing treatments and measurement year are not (Table 35).

Table 35. Hierarchical Linear Model (HLM) predictors and their associated R, R-squared, adjusted R-squared, and changes in R-squared values for each independent variable used to analyze mean forb species richness at the measurement plot level ($9m^2$) for all measurement years ($\alpha = 0.05$).

					Change Statistics		
Model	R	R-squared	Adjusted R-squared	Std. Error of the Estimate	R-squared Change	F Change	Sig. F Change
1	.330[a]	0.109	0.104	4.585	0.109	22.949	0.000
2	.331[b]	0.109	0.100	4.596	0.001	0.126	0.723
3	.474[c]	0.224	0.212	4.301	0.115	27.545	0.000
4	.474[d]	0.224	0.208	4.312	0.000	0.006	0.937

[a] Treatment type

[b] Treatment type plus grazing treatment

[c] Treatment type plus grazing treatment plus soil type

[d] Treatment type plus grazing treatment plus soil type plus year

Open canopy conditions had significantly greater mean forb species richness than closed canopy conditions or forest thinning ($p < 0.001$ and $p = 0.003$, respectively). The effects of closed canopy conditions and forest thinning on mean forb species richness were not significantly different ($p = 0.408$). Mean forb species richness under open canopy conditions was 10.44 (SD 5.28). Under forest thinning and closed canopy conditions it was 6.53 (SD 4.04) and 7.66 (SD 4.32), respectively.

Limestone-derived soils had significantly greater mean forb species richness in comparison to basalt- and benmoreite-derived soils ($p = 0.001$ and $p < 0.001$, respectively). Mean forb species richness was not significantly different between basalt- and benmoreite-derived soils ($p = 0.136$). Mean forb species richness on limestone-derived soils was 10.64 (SD 4.99). On basalt- and benmoreite-derived soils mean forb richness was 7.71 (SD 3.81) and 6.10 (SD 4.24).

Forest treatment type also had a significant effect on mean forb species richness in a two-way interaction with soil parent material type. Under closed canopy conditions, limestone-derived soils had greater mean forb species richness than soils derived from benmoreite ($p < 0.001$) but mean forb species richness was not significantly different between limestone- and basalt-derived soils ($p = 0.091$). Under open canopy conditions, limestone-derived soils had greater mean forb species richness than either basalt- or benmoreite-derived soils ($p = 0.001$ and $p < 0.001$, respectively). Mean forb species richness was not significantly different between basalt- and benmoreite-derived soils ($p = 0.837$).

Grazing treatments did not significantly affect mean forb species richness when all measurement years were analyzed ($p = 0.823$). Also, measurement year did not significantly affect mean forb species richness at the measurement plot level.

Exotic Plant Species Richness per 1m^2 in 2015

A total of 311 individual exotic plants were found during the 2015 measurement year, which constituted 9.1 percent of all plants recorded. The total population of exotic plant species was therefore low in comparison to native plant species, rendering statistical analysis of exotic plant species richness tenuous since many subplots had only one exotic plant present and many measurement plots had fewer than 10 exotic plants present.

However, to understand the effects of forest treatment types, soil parent material type, and grazing treatments on mean exotic plant species richness, statistical analyses were performed.

Forest treatment type and soil parent material type contributed significantly to the power of the HLM of mean exotic plant species richness in 2015. Grazing treatments did not contribute significantly to the predictive power of the HML of mean exotic plant species richness (Table 36).

Table 36. Hierarchical Linear Model (HLM) predictors and their associated R, R-squared, adjusted R-squared, and changes in R-squared values for each independent variable used to analyze mean exotic plant species richness per $1m^2$ for the 2015 measurement year ($\alpha = 0.05$).

					Change Statistics		
Model	R	R-squared	Adjusted R-squared	Std. Error of the Estimate	R-squared Change	F Change	Sig. F Change
1	.209[a]	0.044	0.042	0.786	0.044	19.340	0.000
2	.212[b]	0.045	0.041	0.787	0.001	0.546	0.460
3	.255[c]	0.065	0.059	0.779	0.020	9.075	0.003

[a] Treatment type

[b] Treatment type plus grazing treatment

[c] Treatment type plus grazing treatment plus soil type

Mean exotic plant species richness under forest thinning was significantly greater than open canopy and closed canopy conditions ($p = 0.004$ and $p < 0.001$, respectively). Open canopy conditions had greater mean percent exotic plant species richness than closed canopy conditions ($p < 0.001$). Mean exotic plant species richness under forest thinning in 2015 was 1.06 (SD 0.95). Under open canopy conditions it was 0.78 (SD 0.72) and under closed canopy conditions it was 0.36 (SD 0.54).

Soil parent material affected mean exotic plant species richness at the subplot level in 2015. Mean exotic plant species richness on limestone-derived soils was significantly greater than on benmoreite-derived soils ($p < 0.001$) but was not significantly different from mean exotic plant species richness on basalt-derived soils ($p = 0.080$). Mean exotic plant species richness on limestone-derived soils was 0.90 (SD 0.88). On basalt-derived soils it was 0.70 (SD 0.84), and on benmoreite-derived soils it was 0.58 (SD 0.65).

Grazing had greater mean exotic plant species richness than non-grazing on basalt-derived soils ($p = 0.007$). Mean exotic plant species richness under non-grazing on basalt-derived soils was 0.51 (SD 0.67) while under grazing it was 0.89 (SD 0.95). No significant difference in grazing versus non-grazing was found on soils derived from benmoreite or limestone ($p = 0.200$ and $p = 0.660$, respectively).

Exotic Plant Species Richness per $1m^2$ for All Measurement Years

All independent variables contributed significantly to the power of the HLM of subplot level mean exotic plant species richness when all measurement years are analyzed. Forest treatment type and soil parent material type contributed more to the predictive power of the HLM than grazing treatment or measurement year (Table 35).

Table 37. Hierarchical Linear Model (HLM) predictors and their associated R, R-squared, adjusted R-squared, and changes in R-squared values for each independent variable used to analyze mean exotic plant species richness at the subplot level (1m^2) for all measurement years (α = 0.05).

					Change Statistics		
Model	R	R-squared	Adjusted R-squared	Std. Error of the Estimate	R-squared Change	F Change	Sig. F Change
1	.151[a]	0.023	0.022	0.850	0.023	38.835	0.000
2	.164[b]	0.027	0.026	0.848	0.004	6.974	0.008
3	.271[c]	0.074	0.072	0.828	0.047	83.256	0.000
4	.291[d]	0.085	0.083	0.823	0.011	20.319	0.000

[a] Treatment type

[b] Treatment type plus grazing treatment

[c] Treatment type plus grazing treatment plus soil type

[d] Treatment type plus grazing treatment plus soil type plus year

Forest treatment type significantly affected mean exotic plant species richness at the subplot level ($p < 0.001$). Mean exotic plant species richness was greatest under open canopy conditions and forest thinning than closed canopy conditions ($p < 0.001$). Mean exotic plant species richness under open canopy conditions was not significantly different from forest thinning. Mean exotic plant species richness under open canopy conditions was 0.71 (SD 0.88). Under forest thinning, mean total species richness was 0.75 (SD 0.90) and under closed canopy conditions it was 0.38 (SD 0.73).

Soil parent material type significantly affected mean exotic plant species richness at the subplot level when all measurement years are considered. Limestone-derived soils had significantly greater mean exotic plant species richness than basalt- and benmoreite-derived soils ($p < 0.001$). Mean exotic plant species richness was not significantly different between basalt- and benmoreite-derived soils (p = 1.00). Mean exotic plant species richness on limestone-derived soils was 0.85 (SD 0.97). Basalt- and benmoreite-derived soils had mean exotic plant species richness of 0.44 (SD 0.73) and 0.49 (SD 0.76), respectively.

Grazing treatments significantly affected mean exotic plant species richness at the subplot level when all years were analyzed. Grazing had greater mean exotic plant species richness than non-grazing (p = 0.001). Mean exotic plant species richness under grazing was 0.67 (SD 0.89), while under non-grazing it was 0.56 (SD 0.83).

Measurement year affected mean exotic plant species richness at the subplot level. The 2006 measurement year had significantly greater mean exotic plant species richness than the 2003 and 2008 measurement years (p < 0.001 and p = 0.003, respectively). The 2015 measurement year also had significantly greater mean exotic plant species richness than the 2003 and 2008 sampling years (p < 0.001 and p = 0.020, respectively). Mean exotic plant species richness in 2003 was 0.45 (SD 0.83). In 2006, mean exotic plant species richness had increased significantly to 0.65 (SD 0.90). Mean exotic plant species richness in 2008 remained static at 0.62 (SD 0.88). In 2015 mean exotic plant species richness significantly increased again to 0.73 (SD 0.80) (Figure 51).

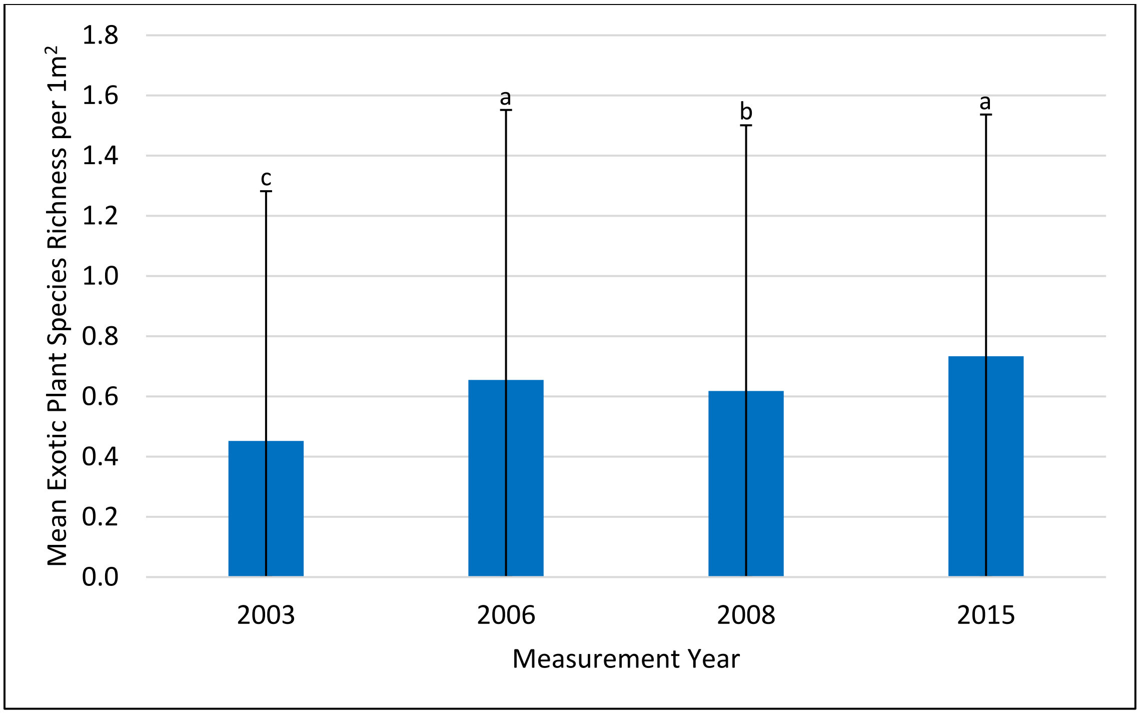

Figure 51. Mean exotic plant species richness per 1m^2 by measurement year. Error bars represent one standard deviation from the mean. Bars with the same letter are not significantly different at the α = 0.05 level.

Exotic Plant Species Richness per 9m^2 in 2015

Only forest treatment type affected mean exotic plant species richness at the measurement plot level in 2015 (Table 38). Grazing treatment and soil parent material type did not contribute significantly to the predictive power of the HLM of mean exotic plant species richness.

Table 38. Hierarchical Linear Model (HLM) predictors and their associated R, R-squared, adjusted R-squared, and changes in R-squared values for each independent variable used to analyze mean exotic plant species richness at the measurement plot level ($9m^2$) in 2015 ($\alpha = 0.05$).

					Change Statistics		
Model	R	R-squared	Adjusted R-squared	Std. Error of the Estimate	R-squared Change	F Change	Sig. F Change
1	.347[a]	0.120	0.101	1.197	0.120	6.299	0.016
2	.351[b]	0.123	0.084	1.209	0.003	0.128	0.722
3	.397[c]	0.157	0.100	1.198	0.034	1.797	0.187

[a] Treatment type

[b] Treatment type plus grazing treatment

[c] Treatment type plus grazing treatment plus soil type

Forest thinning and open canopy conditions had significantly greater mean exotic plant species richness than closed canopy conditions (p = 0.009 and p = 0.033, respectively). Mean exotic plant species richness under forest thinning and open canopy conditions were not significantly different (p = 1.00).

Soil parent material type did not significantly affect mean exotic plant species richness at the measurement plot level in 2015.

Exotic Plant Species Richness per $9m^2$ for All Measurement Years

Forest treatment type, soil parent material type, and measurement year were significant predictors of mean exotic plant species richness at the measurement plot level (Table 39). Grazing treatments were not significant predictors of mean exotic plant species richness (p = 0.052). Soil parent material type provided the greatest contribution to the value of R-squared and R-squared change in comparison to other independent variables.

Table 39. Hierarchical Linear Model (HLM) predictors and their associated R, R-squared, adjusted R-squared, and changes in R-squared values for each independent variable used to analyze mean exotic plant species richness at the measurement plot level ($9m^2$) for all measurement years ($\alpha = 0.05$).

					Change Statistics		
Model	R	R-squared	Adjusted R-squared	Std. Error of the Estimate	R-squared Change	F Change	Sig. F Change
1	.219[a]	0.048	0.043	1.278	0.048	9.483	0.002
2	.259[b]	0.067	0.057	1.268	0.019	3.815	0.052
3	.389[c]	0.151	0.137	1.213	0.084	18.386	0.000
4	.501[d]	0.251	0.234	1.143	0.100	24.604	0.000

[a] Treatment type

[b] Treatment type plus grazing treatment

[c] Treatment type plus grazing treatment plus soil type

[d] Treatment type plus grazing treatment plus soil type plus year

Forest treatment type significantly affected mean exotic plant species richness at the measurement plot level when all measurement years are included in the analysis. Open canopy conditions and forest thinning had significantly greater mean exotic plant species richness than closed canopy conditions ($p < 0.001$) but mean exotic plant species richness was not significantly different between open canopy conditions and forest thinning ($p = 1.00$). Mean exotic plant species richness under closed canopy conditions was 0.92 (SD 1.11). Under open canopy conditions and forest thinning, mean exotic plant species richness was 1.63 (SD 1.29).

Limestone-derived soils had significantly greater mean exotic plant species richness than basalt- and benmoreite-derived soils at the measurement plot level ($p < 0.001$). Mean exotic plant species richness was not significantly different between basalt- and benmoreite-derived soils ($p = 1.00$). Mean exotic plant species richness on limestone-derived soils was 1.90 (SD 1.24). Basalt- and benmoreite-derived soils had mean exotic plant species richness of 1.04 (SD 1.27) and 1.19 (SD 1.27), respectively.

The 2015 measurement year had significantly greater mean exotic plant species richness than all previous measurement years ($p < 0.001$ for 2003, $p = 0.027$ for 2006, and $p = 0.015$ for 2008). The 2003 measurement year had significantly lower mean exotic plant species richness than all subsequent measurement years. ($p = 0.025$ for 2006, $p = 0.049$ for 2008 and $p < 0.001$ for 2015). Mean exotic plant species richness during the 2006 and 2008 measurement years were not significantly different ($p = 1.00$). Mean exotic plant species richness in 2002 was 0.81 (SD1.12). In 2006, it increased significantly to 1.44 (SD 1.29), remained static in 2008 at 1.40 (SD 1.30), then increased significantly again in 2015 to 2.02 (SD 1.26).

Perennial Plant Species Richness per 1m^2 in 2015

All modeled predictors contributed significantly to the power of the HLM of mean perennial plant species richness in 2015 (Table 40).

Table 40. Hierarchical Linear Model (HLM) predictors and their associated R, R-squared, adjusted R-squared, and change in R-squared values for each independent variable used to analyze mean perennial plant species richness per $1m^2$ for the 2015 measurement year ($\alpha = 0.05$).

Model	R	R-squared	Adjusted R-squared	Std. Error of the Estimate	Change Statistics		
					R-squared Change	F Change	Sig. F Change
1	.379[a]	0.143	0.141	2.214	0.143	70.587	0.000
2	.391[b]	0.153	0.149	2.204	0.010	4.850	0.028
3	.466[c]	0.217	0.211	2.121	0.064	34.271	0.000

[a] Treatment type

[b] Treatment type plus grazing treatment

[c] Treatment type plus grazing treatment plus soil type

Open canopy conditions had significantly greater mean perennial plant species richness at the subplot level than forest thinning treatment or closed canopy conditions ($p < 0.001$). Mean perennial plant species richness was not significantly different between closed canopy conditions and forest thinning ($p = 1.00$). Mean perennial plant species richness under open canopy conditions was 5.85 (SD 2.91). Under forest thinning mean perennial plant species richness was 3.61 (SD 1.38) and under closed canopy conditions it was 3.65 (SD 1.97).

Limestone-derived soils had significantly greater mean perennial plant species richness than basalt- and benmoreite-derived soils ($p < 0.001$). Mean perennial plant species richness on limestone-derived soils was 5.24 (SD 2.73). On basalt- and benmoreite-derived soils, mean perennial plant species richness was 3.98 (SD 1.74) and 3.88 (SD 2.16), respectively.

Grazing had significantly lower mean perennial plant species richness than non-grazing ($p = 0.027$). Mean perennial plant species richness under non-grazing was 4.68 (SD 2.71) and under grazing it was 4.19 (SD 2.00).

Perennial Plant Species Richness per 1m^2 for All Measurement Years

Forest treatment type, soil parent material type and measurement year contributed significantly to the power of the HLM of mean perennial plant species richness at the subplot level when all years were analyzed (Table 41).

Table 41. Hierarchical Linear Model (HLM) predictors and their associated R, R-squared, adjusted R squared, and change in R-squared values for each independent variable used to analyze mean perennial plant species richness at the subplot level (1m^2) for all sampling years ($\alpha = 0.05$).

					Change Statistics		
Model	R	R Square	Adjusted R Square	Std. Error of the Estimate	R Square Change	F Change	Sig. F Change
1	.299[a]	0.089	0.089	3.180	0.089	162.087	0.000
2	.299[b]	0.089	0.088	3.180	0.000	0.512	0.475
3	.480[c]	0.230	0.229	2.925	0.141	302.883	0.000
4	.488[d]	0.238	0.236	2.911	0.007	16.194	0.000

[a] Treatment type
[b] Treatment type plus grazing treatment
[c] Treatment type plus grazing treatment plus soil type
[d] Treatment type plus grazing treatment plus soil type plus year

Open canopy conditions had significantly greater mean perennial plant species richness than closed canopy conditions or forest thinning treatments ($p < 0.001$). Thinning also yielded significantly greater mean perennial plant species richness than closed canopy conditions ($p = 0.010$). Mean perennial plant species richness under open canopy conditions was 6.54 (SD 4.03). Under forest thinning it was 4.48 (SD 2.41) and under closed canopy conditions it was 4.12 (SD 2.73) (Figure 52).

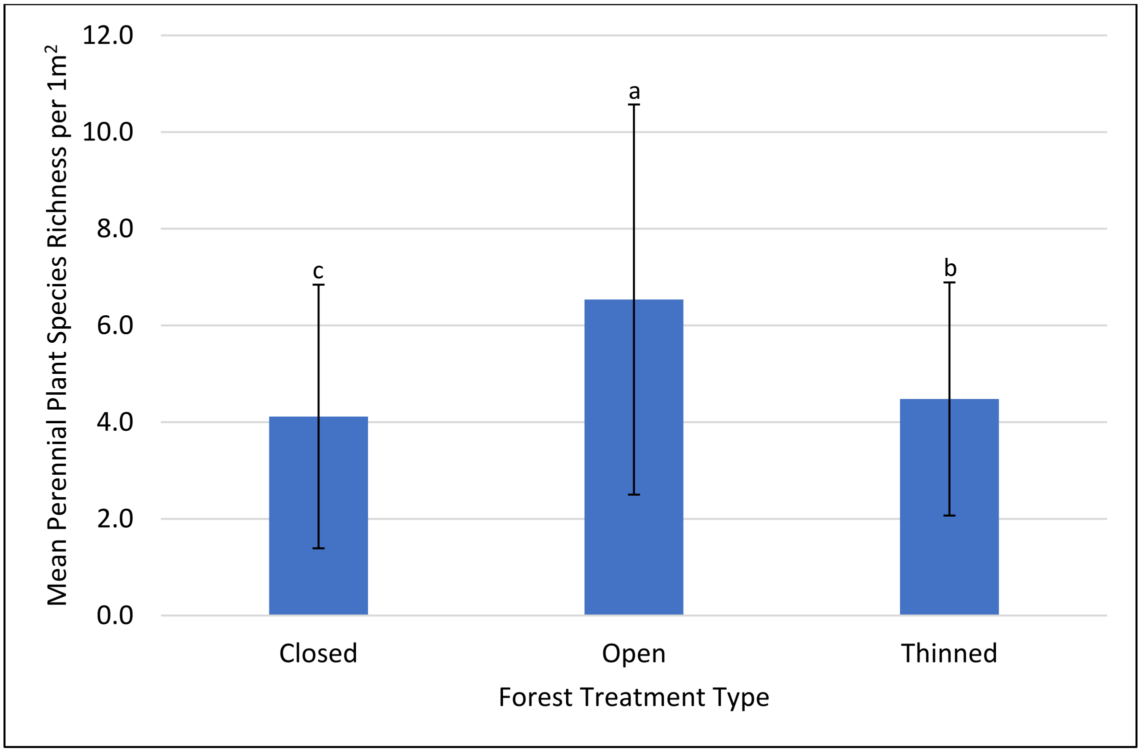

Figure 52. Mean perennial plant species richness per $1m^2$ by forest treatment type across all measurement years. Error bars represent one standard deviation from the mean. Bars with different letter are significantly different at the $\alpha = 0.05$ level.

Limestone-derived soils had significantly greater mean perennial plant species richness than basalt- and benmoreite-derived soils ($p < 0.001$). Mean perennial plant species richness on basalt-derived soils was significantly greater than on benmoreite-derived soils (p = 0.003). Mean perennial plant species richness on limestone-derived soils was 6.74 (SD 3.73). Basalt- and benmoreite-derived soils had mean perennial plant species richness of 4.27 (SD 2.41) and 3.82 (SD 2.60), respectively (Figure 53).

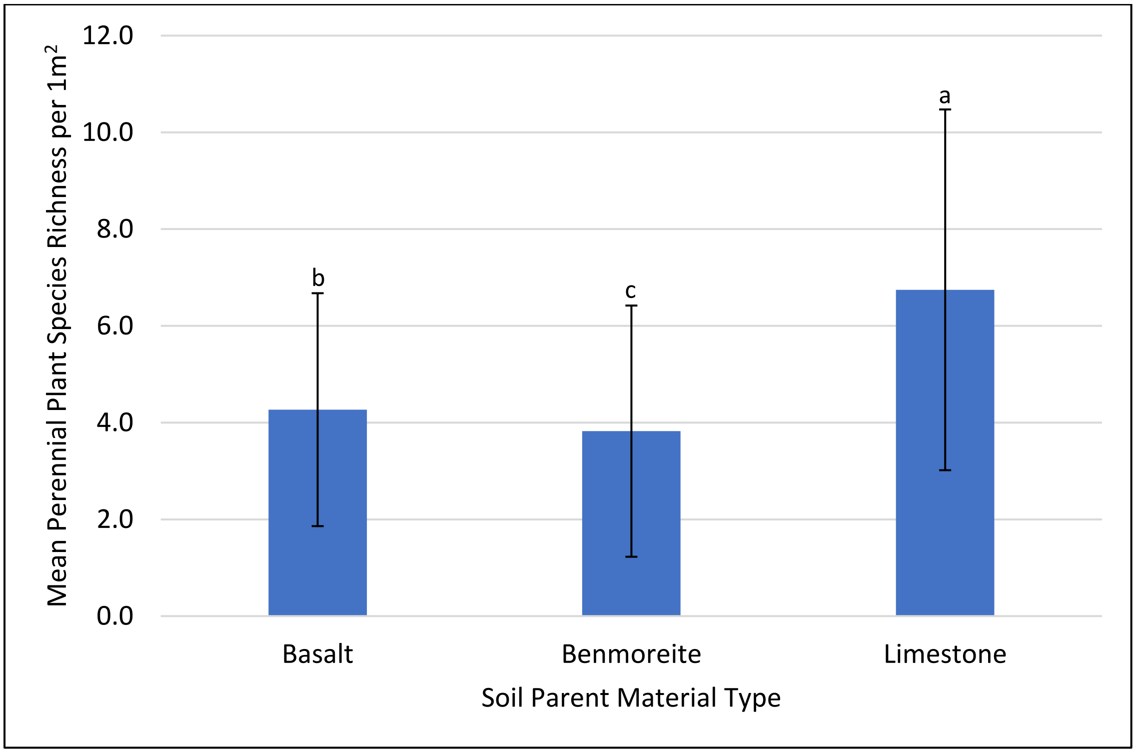

Figure 53. Mean perennial plant species richness per 1m^2 by soil parent material type across all measurement years. Error bars represent one standard deviation from the mean. Bars with different letter are significantly different at the α = 0.05 level.

The 2006 measurement year had the greatest mean perennial plant species richness of all measurement years (p = 0.001). The 2008 measurement year had significantly lower mean perennial plant species richness that 2006 (p = 0.001). Mean perennial plant species richness in 2003 was not significantly different from 2008 or 2015 (p = 0.849 and p =

0.845, respectively). In 2015, mean perennial plant species richness was significantly lower than in 2006 and 2008 ($p < 0.001$ and $p = 0.017$, respectively), but was not significantly different from 2003 ($p = 0.845$) (Figure 54). Grazing treatments did not significantly affect mean perennial plant species richness at the subplot level.

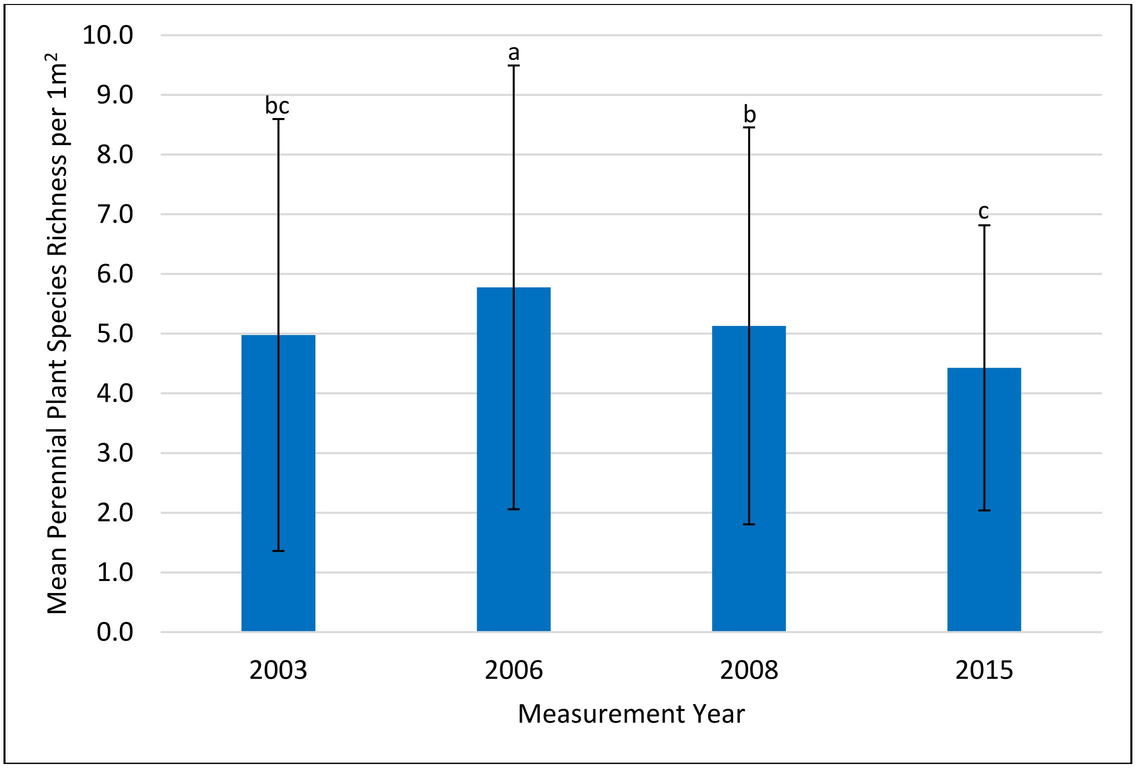

Figure 54. Mean perennial plant species richness per $1m^2$ by measurement year. Error bars represent one standard deviation from the mean. Bars with the same letter are not significantly different at the $\alpha = 0.05$ level.

Perennial Plant Species Richness per $9m^2$ in 2015

Forest treatment type and soil parent material type contributed significantly to the power of the HLM of measurement plot level mean perennial plant species richness in 2015, while grazing treatments did not (Table 40).

Table 42. Hierarchical Linear Model (HLM) predictors and their associated R, R-squared, adjusted R squared, and changes in R-squared values for each independent variable used to analyze mean perennial plant species richness at the measurement plot level ($9m^2$) for all sampling years ($\alpha = 0.05$).

					Change Statistics		
Model	R	R Square	Adjusted R Square	Std. Error of the Estimate	R Square Change	F Change	Sig. F Change
1	.354[a]	0.125	0.106	4.475	0.125	6.593	0.014
2	.365[b]	0.133	0.095	4.504	0.008	0.411	0.525
3	.508[c]	0.258	0.207	4.214	0.125	7.402	0.009

[a] Treatment type

[b] Treatment type plus grazing treatment

[c] Treatment type plus grazing treatment plus soil type

Differences in mean perennial plant species richness among forest treatment types were not statistically significant. Mean perennial plant species richness under open canopy conditions was 13.44 (SD 5.61). Closed canopy conditions had mean perennial plant species richness of 9.38 (SD 5.02) and forest thinning had mean perennial plant species richness of 10.88 (SD 4.73).

Limestone-derived soils had greater mean perennial plant species richness than benmoreite-derived soils (p = 0.006). Mean perennial plant species richness was not significantly different between limestone-derived soils and basalt-derived soils (p = 0.234) nor basalt-derived soils and benmoreite-derived soils (p = 0.727). Mean perennial plant species richness on soils derived from limestone was 13.39 (SD 5.65). Basalt- and benmoreite-derived soils had mean perennial plant species richness of 10.50 (SD 1.73) and 8.61 (SD 3.99), respectively (figure 55). Grazing treatments did not significantly affect mean perennial plant species richness at the measurement plot level in 2015 (p = 0.58).

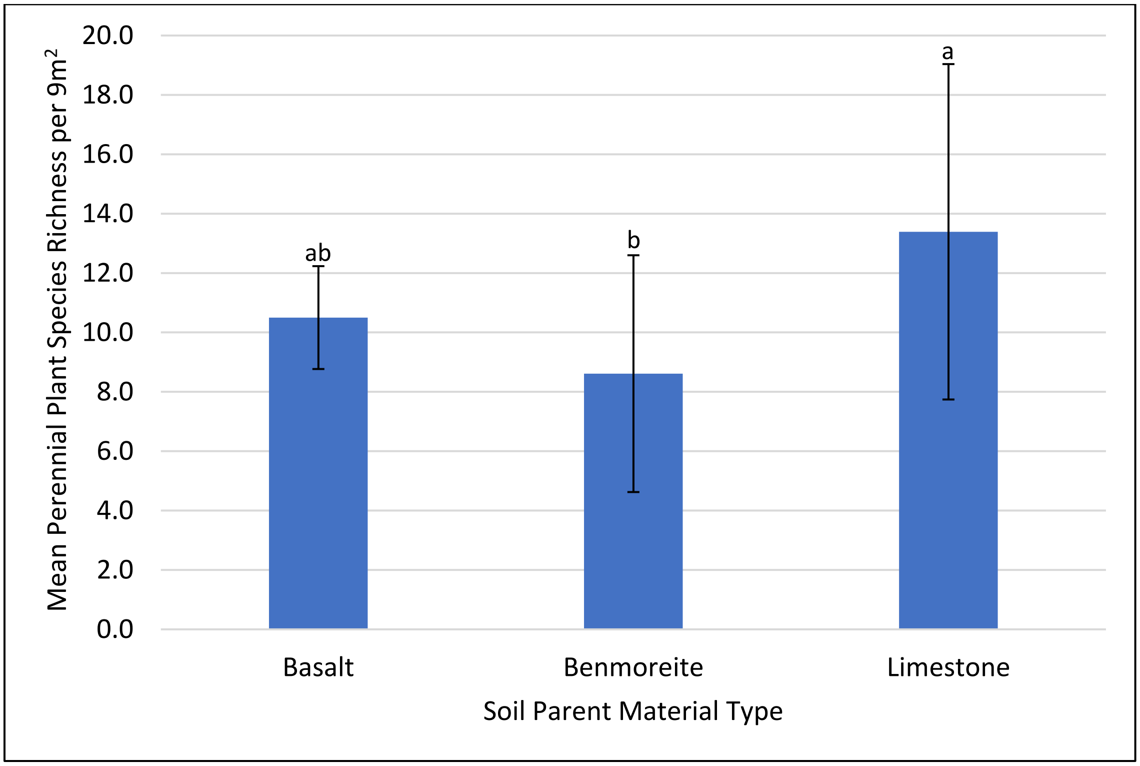

Figure 55. Mean perennial plant species richness per $9m^2$ by soil parent material type. Error bars represent one standard deviation from the mean. Bars with the same letter are not significantly different at the α = 0.05 level.

Perennial Plant Species Richness per $9m^2$ for All Measurement Years

Forest treatment type and soil parent material type are significant predictors of mean perennial plant species richness at the measurement plot level when all measurement years are analyzed. Grazing treatments and measurement year were not significant. (Table 43).

Table 43. Hierarchical Linear Model (HLM) predictors and their associated R, R-squared, adjusted R-squared, and changes in R-squared values for each independent variable used to analyze mean perennial plant species richness at the measurement plot level ($9m^2$) for all measurement years (α = 0.05).

					Change Statistics		
Model	R	R-squared	Adjusted R-squared	Std. Error of the Estimate	R-squared Change	F Change	Sig. F Change
1	.328[a]	0.107	0.103	5.187	0.107	22.633	0.000
2	.328[b]	0.107	0.098	5.201	0.000	0.000	0.984
3	.509[c]	0.259	0.247	4.750	0.152	38.151	0.000
4	.509[d]	0.260	0.244	4.763	0.000	0.047	0.829

[a] Treatment type

[b] Treatment type plus grazing treatment

[c] Treatment type plus grazing treatment plus soil type

[d] Treatment type plus grazing treatment plus soil type plus year

Open canopy conditions had significantly greater mean perennial plant species richness than forest thinning ($p = 0.001$) or closed canopy conditions ($p < 0.001$). There was no significant difference in mean perennial plant species richness between forest thinning and closed canopy conditions ($p = 0.091$). Mean perennial plant species richness under open canopy conditions was 13.63 (SD 6.41). Under forest thinning and closed canopy conditions it was 10.13 (SD 3.77) and 9.24 (SD 4.95), respectively.

Limestone-derived soils had greater mean perennial plant species richness that basalt- and benmoreite-derived soils ($p < 0.001$). Mean perennial plant species richness was not significantly different between basalt- and benmoreite-derived soils ($p = 0.105$). Limestone-derived soils had mean perennial plant species richness of 13.39 (SD 5.65). Basalt- and benmoreite-derived soils had mean perennial plant species richness of 10.50 (SD 1.73) and 8.61 (SD 3.99), respectively.

Grazing treatments and measurement years did not significantly affect mean perennial plant species richness at the measurement plot level.

Annual Plant Species Richness per 1m^2 in 2015

Only soil parent material type contributed significantly to the power of the HLM at predicting mean annual plant species richness at the subplot level in 2015. Forest treatment type and grazing treatment did not contribute significantly to the power of the HLM (Table 42).

Table 44. Hierarchical Linear Model (HLM) predictors and their associated R, R-squared, adjusted R-squared, and change in R-squared values for each independent variables used to analyze mean annual plant species richness per 1m^2 for the 2015 measurement year ($\alpha = 0.05$).

Model	R	R-squared	Adjusted R-squared	Std. Error of the Estimate	Change Statistics		
					R-squared Change	F Change	Sig. F Change
1	.062[a]	0.004	0.001	0.431	0.004	1.621	0.204
2	.069[b]	0.005	0.000	0.431	0.001	0.393	0.531
3	.190[c]	0.036	0.029	0.425	0.031	13.609	0.000

[a] Treatment type

[b] Treatment type plus grazing treatment

[c] Treatment type plus grazing treatment plus soil type

While not a significant predictor in the HLM, forest treatment type was significant in pairwise comparisons of mean annual plant species richness using Bonferroni adjustment for multiple comparisons. Forest thinning had greater mean annual plant species richness than open canopy ($p = 0.007$) and closed canopy conditions ($p < 0.001$). There was no significant difference in mean annual plant species richness between open canopy and closed canopy conditions ($p = 0.329$). Mean annual plant species richness under forest thinning was 0.30 (SD 0.49). Under open canopy conditions mean annual plant species richness was 0.17 (SD 0.43) and under closed canopy conditions it was 0.11 (SD 0.33).

Limestone-derived soils had greater mean annual plant species richness than basalt- and benmoreite-derived soils ($p = 0.045$ and $p = 0.001$, respectively). Basalt-derived soils had greater mean annual plant species richness than benmoreite-derived soils ($p = 0.036$). Mean annual plant species richness on limestone-derived soils was 0.31 (SD 0.53). On basalt- and benmoreite-derived soils mean annual plant species richness was 0.20 (SD 0.42) and 0.07 (SD 0.26), respectively.

Grazing treatments did not significantly affect mean annual plant species richness in 2015 (p = 0.242).

Annual Plant Species Richness per 1m^2 for All Measurement Year

Only forest treatment type contributed significantly to the power of the HLM of subplot level mean annual plant species richness. Grazing treatments, soil parent material type, and measurement year were not strong predictors of mean annual plant species richness (Table 45).

Table 45. Hierarchical Linear Model (HLM) predictors and their associated R, R-squared, adjusted R-squared, and change in R-squared values for each independent variable used to analyze mean annual plant species richness at the subplot level (1m^2) for all measurement years (α = 0.05).

					Change Statistics		
Model	R	R-squared	Adjusted R-squared	Std. Error of the Estimate	R-squared Change	F Change	Sig. F Change
1	.147[a]	0.022	0.021	0.538	0.022	36.557	0.000
2	.147[b]	0.022	0.021	0.539	0.000	0.143	0.705
3	.149[c]	0.022	0.021	0.539	0.001	1.036	0.309
4	.155[d]	0.024	0.022	0.538	0.002	3.166	0.075

[a] Treatment type

[b] Treatment type plus grazing treatment

[c] Treatment type plus grazing treatment plus soil type

[d] Treatment type plus grazing treatment plus soil type plus year

Open canopy conditions had significantly greater mean annual plant species richness than forest thinning ($p = 0.003$) and closed canopy conditions ($p < 0.001$). Forest thinning had significantly greater mean annual plant species richness than closed canopy conditions ($p = 0.044$). Mean annual plant species richness under open canopy conditions was 0.33 (SD 0.65). Under forest thinning mean annual plant species richness was 0.22 (SD 0.54), and under closed canopy conditions it was 0.13 (SD 0.39).

Soil parent material type, grazing treatments and measurement year did not significantly affect mean annual plant species richness at the subplot level.

Annual Plant Species Richness per 9m^2 in 2015

None of the independent variables contributed to the power of the HLM of predicted mean annual plant species richness at the measurement plot level in 2015 (Table 46).

Table 46. Hierarchical Linear Model (HLM) predictors and their associated R, R-squared, adjusted R-squared, and changes in R-squared values for each independent variable used to analyze mean annual plant species richness at the measurement plot level (9m^2) in 2015 ($\alpha = 0.05$).

					Change Statistics		
Model	R	R-squared	Adjusted R-squared	Std. Error of the Estimate	R-squared Change	F Change	Sig. F Change
1	.213[a]	0.046	0.025	0.955	0.046	2.195	0.145
2	.231[b]	0.053	0.011	0.961	0.008	0.361	0.551
3	.283[c]	0.080	0.017	0.958	0.027	1.282	0.264

[a] Treatment type

[b] Treatment type plus grazing treatment

[c] Treatment type plus grazing treatment plus soil type

Pairwise comparisons using Bonferroni correction for multiple comparisons further revealed that no significant differences in mean annual plant species richness resulted from forest treatment types, grazing treatments, or differences in soil parent material type at the measurement plot level in 2015. There were no statistically significant interactions among the independent variables.

Annual Plant Species Richness per $9m^2$ for All Measurement Years

The only independent variable that affected mean annual plant species richness was forest treatment type (Table 47).

Table 47. Hierarchical Linear Model (HLM) predictors and their associated R, R-squared, adjusted R-squared, and changes in R-squared values for each independent variable used to analyze mean annual plant species richness at the measurement plot level ($9m^2$) for all measurement years ($\alpha = 0.05$).

					Change Statistics		
Model	R	R-squared	Adjusted R-squared	Std. Error of the Estimate	R-squared Change	F Change	Sig. F Change
1	.245[a]	0.060	0.055	1.092	0.060	11.970	0.001
2	.245[b]	0.060	0.050	1.095	0.000	0.037	0.847
3	.247[c]	0.061	0.046	1.097	0.001	0.210	0.648
4	.248[d]	0.062	0.041	1.100	0.000	0.084	0.772

[a] Treatment type

[b] Treatment type plus grazing treatment

[c] Treatment type plus grazing treatment plus soil type

[d] Treatment type plus grazing treatment plus soil type plus year

Pairwise comparisons using Bonferroni correction for multiple comparisons revealed mean annual plant species richness under open canopy conditions was significantly greater than under closed canopy conditions (p = 0.003).

Mean annual plant species richness under open canopy conditions was 1.14 (SD 1.31) and under closed canopy conditions it was 0.47 (SD 0.67). Mean annual plant species under forest thinning was 0.78 (SD 1.26) which was not significantly different from mean annual plant species under open canopy conditions (p = 0.530) or closed canopy conditions (p = 0.364). No significant differences in mean annual plant species richness resulted from grazing treatments, differences in soil parent material types, or measurement year at the measurement plot level. There were no statistically significant interactions among any of the independent variables.

Biennial Plant Species Richness per 1m^2 in 2015

Forest treatment type and soil parent material type contributed significantly to the predictive power of the HLM of mean biennial plant species richness in 2015. Grazing treatments did not contribute to the power of the HLM in predicting mean biennial plant species richness (Table 48).

Table 48. Hierarchical Linear Model (HLM) predictors and their associated R, R-squared, adjusted R-squared, and change in R-squared values for each independent variable used to analyze mean biennial plant species richness per 1m^2 for the 2015 measurement year ($\alpha = 0.05$).

					Change Statistics		
Model	R	R-squared	Adjusted R-squared	Std. Error of the Estimate	R-squared Change	F Change	Sig. F Change
1	.118[a]	0.014	0.012	0.625	0.014	5.814	0.016
2	.119[b]	0.014	0.009	0.626	0.000	0.084	0.772
3	.214[c]	0.046	0.039	0.617	0.031	13.413	0.000

[a] Treatment type

[b] Treatment type plus grazing treatment

[c] Treatment type plus grazing treatment plus soil type

Mean biennial plant species richness under closed canopy conditions was significantly lower than forest thinning and open canopy conditions ($p < 0.01$ and $p = 0.023$, respectively). There was no significant difference in mean perennial plant species richness between open canopy conditions and forest thinning ($p = 0.134$). Mean forb species richness in 2015 under closed canopy conditions was 0.28 (SD 0.47). Under forest thinning and open canopy conditions it was 0.65 (SD 0.75) and 0.47 (SD 0.59), respectively.

Limestone-derived soils had greater mean biennial plant species richness per 1m^2 than basalt- or benmoreite-derived soils in 2015 ($p = 0.011$ and $p < 0.001$, respectively). There was no significant difference in mean biennial plant species richness between basalt- and benmoreite-derived soils ($p = 0.73$). Mean biennial plant species richness on limestone-derived soils was 0.62 (SD 0.70) and on basalt- and benmoreite-derived soils it was 0.40 (SD 0.61) and 0.34 (SD 0.52), respectively.

Mean biennial plant species richness was not significantly affected by grazing treatments (p = 0.64) and there were no statistically significant interactions among forest treatment types, grazing treatments or soil parent material types.

Biennial Plant Species Richness per 1m^2 for All Measurement Years

Forest treatment type, soil parent material type and measurement year contributed significantly to the power of the HLM of subplot level mean biennial plant species richness (Table 49).

Table 49. Hierarchical Linear Model (HLM) predictors and their associated R, R-squared, adjusted R-squared, and changes in R-squared values for each independent variable used to analyze mean biennial plant species richness at the subplot level (1m^2) for all measurement years (α = 0.05).

Model	R	R-squared	Adjusted R-squared	Std. Error of the Estimate	Change Statistics		
					R-squared Change	F Change	Sig. F Change
1	.196[a]	0.039	0.038	0.621	0.039	65.925	0.000
2	.197[b]	0.039	0.038	0.621	0.000	0.194	0.659
3	.203[c]	0.041	0.039	0.620	0.002	4.140	0.042
4	.226[d]	0.051	0.049	0.617	0.010	17.367	0.000

[a] Treatment type

[b] Treatment type plus grazing treatment

[c] Treatment type plus grazing treatment plus soil type

[d] Treatment type plus grazing treatment plus soil type plus year

Mean biennial plant species richness was greatest under open canopy conditions and forest thinning ($p < 0.001$). Closed canopy conditions had the lowest mean biennial plant species richness ($p = 0.032$). Mean biennial plant species richness under open canopy conditions and forest thinning was not significantly different ($p = 0.275$). Mean biennial plant species richness under open canopy conditions was 0.56 (SD 0.65). Under forest thinning, mean biennial plant species richness was 0.51 (SD 0.72) and under closed canopy conditions it was 0.25 (SD 0.44).

Limestone- and benmoreite-derived soils had the greatest mean biennial plant species richness ($p = 0.003$ and $p = 0.020$, respectively). Mean biennial plant species richness on basalt-derived soils was lowest ($p = 0.020$ for benmoreite and $p = 0.003$ for limestone). Mean biennial plant species richness on limestone- and benmoreite-derived soils was not significantly different ($p = 1.00$). Limestone- and benmoreite-derived soils had mean biennial plant species richness values of 0.46 (SD 0.61) and 0.49 (SD 0.67), respectively, while basalt-derived soils had mean biennial plant species richness of 0.35 (SD 0.61).

The 2006 and 2008 measurement years were not significantly different ($p = 0.484$), and both had greater mean biennial plant species richness than the 2003 and 2015 measurement years ($p < 0.001$). Mean biennial plant species richness was lowest in 2003 and intermediate in 2015. Mean biennial plant species richness in 2006 and 2008 was 0.51 (SD 0.68) and 0.57 (SD 0.68), respectively. In 2003 mean biennial plant species richness was 0.22 (SD 0.46) and in 2015 it was 0.46 (SD 0.63) (Figure 56).

Mean biennial plant species richness was not significantly affected at the subplot level by grazing treatments (p = 0.812) and there were no statistically significant interactions among forest treatment types, grazing treatments or soil parent material types.

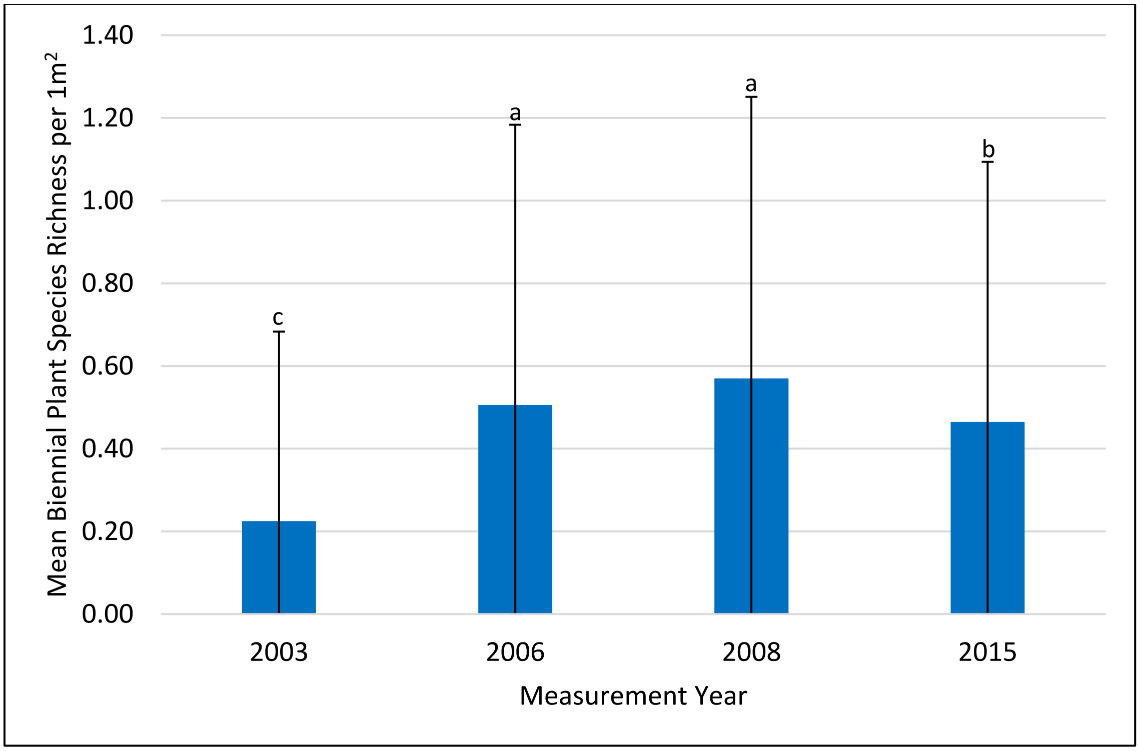

Figure 56. Mean biennial plant species richness per $1m^2$ by measurement year. Error bars represent one standard deviation from the mean. Bars with the same letter are not significantly different at the $\alpha = 0.05$ level.

Biennial Plant Species Richness per $9m^2$ in 2015

None of the independent variables contributed significantly to the power of the HLM of mean biennial plant species richness at the measurement plot level in 201 (Table 50).

Table 50. Hierarchical Linear Model (HLM) predictors and their associated R, R-squared, adjusted R-squared, and changes in R-squared values for each independent variable used to analyze mean biennial plant species richness at the measurement plot level ($9m^2$) in 2015 ($\alpha = 0.05$).

Model	R	R-squared	Adjusted R-squared	Std. Error of the Estimate	Change Statistics R-squared Change	F Change	Sig. F Change
1	.275[a]	0.076	0.056	0.911	0.076	3.764	0.059
2	.276[b]	0.076	0.035	0.921	0.001	0.025	0.876
3	.291[c]	0.085	0.022	0.927	0.009	0.419	0.521

[a] Treatment type

[b] Treatment type plus grazing treatment

[c] Treatment type plus grazing treatment plus soil type

No significant difference in mean biennial plant species richness was found between forest treatment types, soil parent material types, or grazing treatments in 2015 using pairwise comparisons with Bonferroni correction for multiple comparisons.

Biennial Plant Species Richness per $9m^2$ for All Measurement Years

Forest treatment type and measurement year were significant predictors of mean biennial plant species richness at the measurement plot level while grazing treatment and soil parent material type were no (Table 51).

Table 51. Hierarchical Linear Model (HLM) predictors and their associated R, R-squared, adjusted R-squared, and changes in R-squared values for each independent variable used to analyze mean biennial plant species richness at the measurement plot level (9m^2) for all measurement years (α = 0.05).

Model	R	R-squared	Adjusted R-squared	Std. Error of the Estimate	Change Statistics		
					R-squared Change	F Change	Sig. F Change
1	.262[a]	0.069	0.064	0.863	0.069	13.903	0.000
2	.263[b]	0.069	0.059	0.865	0.000	0.020	0.887
3	.265[c]	0.070	0.055	0.867	0.001	0.204	0.652
4	.367[d]	0.135	0.116	0.838	0.065	13.914	0.000

[a] Treatment type

[b] Treatment type plus grazing treatment

[c] Treatment type plus grazing treatment plus soil type

[d] Treatment type plus grazing treatment plus soil type plus year

Open canopy conditions and forest thinning treatments had significantly greater mean biennial plant species richness than closed canopy conditions (p = 0.001 and p = 0.003, respectively). The effects of open canopy conditions and forest thinning on mean biennial plant species richness were not significantly different (p = 0.613).

Mean biennial plant species richness under open canopy conditions and forest thinning was 1.17 (SD 0.92) and 1.05 (SD 0.98), respectively. Mean biennial plant species richness under closed canopy conditions was 0.60 (SD 0.64).

The 2006, 2008, and 2015 measurement years were not significantly different (p = 1.00 and p = 0.406) and all had greater mean biennial plant species richness than before treatments were implemented in 2003 (p = 0.016). Mean biennial plant species richness in 2003 was 0.49 (SD 0.72). It had increased significantly in 2006 to 0.94 (SD 0.89). In

2008 and 2015, mean biennial plant species richness was 1.15 (SD 0.86) and 1.19 (SD 0.94), respectively.

There were no significant differences in mean biennial plant species richness at the measurement plot level between soil parent material types or grazing treatments when all measurement years were included in the analysis and there were no significant interactions among the independent variables.

Open canopy conditions and forest thinning treatments were not significantly different (0.613), but both treatments had significantly greater mean biennial plant species richness than closed canopy conditions ($p < 0.001$ and $p = 0.003$, respectively).

DISCUSSION

Soil Bulk Density

Forest thinning treatments consisted of tree felling using chainsaws, bucking, and hand removal of thinning debris (slash) from treated research plots. This approach to forest restoration thinning substantially limited soil mechanical disturbance and potential for soil compaction in comparison to typical heavy logging machinery commonly used at operational scales. Additionally, forest thinning treatments were implemented twelve years prior to this study. Given that minimal impact methods were used for forest thinning treatments and sampling was 12 years after treatment, the finding of no significant difference in soil bulk density among forest treatment types is to be expected.

The similar particle size distributions and low variability in surface textures could partially explain why there were no significant differences in soil bulk density by treatment type, soil parent material type or grazing treatment. Abiotic factors such as freeze-thaw and repeated wetting and drying of surface horizons in these soils could further explain similarities in bulk densities observed. Freeze-thaw and wetting-drying cycles can reduce soil bulk densities and influence bulk density measurements.

Soils within the project area have expansive clay mineralogy (2:1 phyllosilicate, or smectite). These soils are prone to shrinking when dry and swelling when wet. When these soils dry to the point where desiccation cracks form, soil bulk density measurement can be affected due to increased void space per unit volume of soil sampled. Berndt and Coughlan (1976) found water loss during the normal shrinkage phase of saturated soils with expansive clays is accompanied by a corresponding loss of soil volume, and the shrinkage is multi-dimensional.

The pastures where this research was conducted are usually grazed from June 1st through October 15th, based on range readiness. The maximum allowable level of combined utilization of herbaceous and non-riparian woody vegetation by both livestock and wildlife is 35%. The total AUMs authorized was approximately 0.25 AUM ha^{-1}. This stocking level represents conservative grazing capacity. It is therefore possible that no significant differences in soil bulk densities were detected due to livestock and wildlife ungulate populations being well dispersed over such a large land area, limiting the potential for detectable soil compaction to occur.

Saturated Infiltration Capacity

Abella et al. (2015) found soil textures in this study did not vary greatly by soil parent material type. Their surface texture determinations were loam for limestone and basalt-derived soils and silt loam for soils derived from benmoreite. Thomas, et al. (2017) conducted texture analysis on a few randomly chosen samples and found limestone- and basalt-derived soils had sandy clay loam and loam textures, while benmoreite-derived soils had loam textures. Given these broad similarities in soil textures, saturated infiltration capacity does not appear to have been influenced by surface soil texture. This would agree with hydrologic soil properties classified by soil texture (Brooks and Cory, 1964; Rawls et al., 1982). A high degree of variation in surface horizon characteristics such as the spatial arrangement of soil particles and aggregates, presence of macropores and coarse fragments (stones, rocks, and channers) as well as shrinkage cracks, earthworm channels and root channels that lead to secondary porosity appear to have influenced infiltration rates through induced preferential flow, particularly under low antecedent soil moisture conditions. Rock soil interfaces provide conduits for rapid movement of the soil solution via preferential flow paths. This is particularly the case in soils that are dry or water repellant (hydrophobic) as is common in these soils. As soils shrink, rock soil interfaces separate, creating conditions conducive to unstable wetting fronts.

Antecedent soil moisture also appears to have influenced infiltration rates, but not infiltration capacities. Under dry condition, macropore flow in the A horizons and bypass flow in shrinkage cracks in the shallow Bt horizons was likely as evidenced by some of the infiltration curves plotted from saturated infiltration capacity tests. Antecedent soil moisture can exert significant influence on soil infiltration (Ma et al., 2020; Ruggenthaler et al., 2016; Song and Wang, 2019; Hardie et al., 2011). This study was conducted during the monsoon season with high spatiotemporal variability in precipitation patterns and average storm cell diameters being less than 8km. It is likely that some soils across the project area had high antecedent soil moisture content when studied while others had low antecedent soil moisture.

Aggregate Stability

Grazing treatments had a significant effect on aggregate stability with predicted water stable aggregates averaging 21.77 percent fewer than under non-grazing. This finding is supported by those of Knoll and Hopkins (1959), who studied the effects of grazing on aggregate stability in pastures near Hays, Kansas. Wen et al. (2016) also support the findings of this study. They found SOM stability increased after 30 years of grazing exclusion due to the greater physical protection of SOM by stable soil aggregates.

Soil Slaking

Thomas (2017) found no statistically significant differences in pH, electrical conductivity (EC), carbon (C), nitrogen (N), sulfur (S), phosphorous (P), calcium (Ca^{2+}), magnesium (Mg2+), or organic matter (OM) among soil types. It is therefore unlikely that soil pH, electrical conductivity, macronutrient concentrations, or OM content contributed to observed differences is soil slaking potential by soil type or forest treatment type under grazing or non-grazing. Soils throughout the study area are highly variable. In a study of weathering, genesis, and classification of selected basaltic soils of the San Francisco Volcanic Field (SFVF), Cheevers (1982) determined that eolian transport and deposition of non-basaltic minerals readily occurs. Deposition of quartz, chert, K-feldspars and mica from local formations such as the Coconino Sandstone, Kaibab and Moenkopi Formations and silicic volcanic centers is common. This may partially explain the lack of statistical significance in many of the soil macronutrients measured as well as soil textures. Trampling and biomass removal by livestock, wildlife ungulates, and small mammals may also be factors contributing to soil slaking potential in grazed versus non-grazed treatments.

Soil Carbon, Nitrogen and Organic Matter

No significant differences in levels of soil organic C and N (Mg ha^{-1}) and organic matter (OM) in the upper 10cm of soil by forest treatment type, soil parent material type, or grazing treatment agrees with the results of Thomas (2017). Soil bulk densities did not influence soil C or N values. This may be partially explained by the study area having no visual evidence of fire in the post-settlement period since 1880 (Abella et al., 2015), leading to a somewhat steady state and thrifty relationship between rates of soil detrital input and decomposition. The findings of this study are supported by those of Kaye and Hart (1998b), who found no statistically significant differences in mineral soil N, P, OM, or C/N ratios among forest restoration treatments that included partial restoration, complete restoration, and control.

Other Soil Chemical Properties

No statistically significant differences were found in soil calcium (Ca^{2+}) levels (Mg ha^{-1}) among soil type, forest treatment type, or grazing treatments (Table B 1), in contrast to Thomas (2017), who found a significant interaction between forest restoration treatment type and grazing management ($p = 0.0482$). The authors found closed canopy conditions with grazing had lower concentrations of Ca^{2+} than open canopy conditions with grazing. Additionally, the authors found soils derived from benmoreite had a significant three-way interaction ($p = 0.0519$) where closed canopy conditions under grazing had lower Ca^{2+} concentrations than either basalt or limestone-derived soils. In this study, which included consideration of soil bulk density, no significant differences were observed. Surface soils vary considerably throughout the study area due, in part, to eloian influences as previously described. This has likely played a role in soil evolution throughout the region through burial of residual soil profiles. Wells et al. (1990) described the effects of eolian transport across northern Arizona, which resulted in late Pleistocene eolian deposits across portions of the southeastern Colorado Plateau and further described the effects of atmospheric dust (windblown silt and clay) on soil formation and Ca^{2+} dynamics, including secondary carbonate formation and authigenic factors of carbonate leaching and translocation to depths as much as 2.5 m in highly permeable, well-sorted sandy parent materials. Reynolds et al. (2001) provided evidence that the Sonoran and Mojave deserts are both historic and contemporary sources of dust across the Colorado Plateau, which extends as far northeast as Canyonlands National Park in southeastern Utah. It is therefore likely that eolian erosion, transport, deposition, and translocation of Ca^{2+} and other soil nutrients in the upper part of soil profiles has influenced the findings of this study and the findings of the companion study conducted by Thomas (2017).

Soil potassium (K^+) was not significantly different by soil type, forest treatment type, or grazing treatment. It is possible that natural spatial variation in K^+ is high in soils throughout the study area. It is also likely that K^+ levels are influenced by feldspars, such as orthoclase, and micas, which are common throughout the study area and release K^+ through weathering processes.

Soil parent material type significantly affected soil Mg^{2+} content (Mg ha^{-1}), with basalt-derived soils having significantly greater mean soil Mg^{2+} content than benmoreite- and limestone-derived soils. Forest treatment type and grazing treatment did not significantly affect soil Mg^{2+} content (Mg ha^{-1}). These findings do not agree with Thomas (2017), who found Mg^{2+} concentrations (mg kg^{-1}) were significantly different in a forest treatment type and grazing treatment interaction. However, the findings of this study were expected. Eruptive products of the SFVF, while of varying petrogenesis, are dominantly basaltic and rhyolitic (Hanson et al., 2008). Basalt typically has high percentages of Mg^{2+} and Fe^{2+}. Basalts of the SFVF tend to be Fe^{2+} rich, yet low in MnO and are silica (SiO_2) undersaturated (Arculus and Gust, 1995) with high percentages of Fe, Al_2O_3, MnO, Na_2O, K_2O. This could partially explain the significant differences Mg^{2+} found between soil parent material types.

Soil sodium (Na) levels (Mg ha^{-1}) were not significantly different by soil type or forest treatment type. However, grazing affected soil Na^+ levels, with grazing having significantly less Na^+ than non-grazed conditions across all soil types and ecological restoration treatments. These findings do not agree with Thomas (2017), who found Na^+

concentrations were only significant in a three-way interaction among soil type, restoration treatment type, and grazing treatment. In this study, lower Na^+ levels in grazed plots versus non-grazed on all soil types and restoration treatments is perplexing. Grazing has generally been shown to increase localized soil Na^+ and Mg^{2+} concentrations since livestock urine and feces can have high concentrations of these elements (Sainju et al., 2011; Hamamoto and Uchida, 2015), particularly when mineral supplements are provided to livestock, which is a common practice on USFS grazing allotments in Northern Arizona.

Vegetative Cover

Forest treatment type affected mean percent total vegetative cover, with forest thinning and open canopy conditions having greater mean percent total vegetative cover than closed canopy conditions. This finding generally agrees with the finding of Abella et al. (2015), who found that forest thinning treatments significantly increased plant cover on limestone-derived soils. However, they also noted that open canopy plots had significant increases in plant cover 5 years after initiating the study. They also noted that there was no significant change in mean total plant cover on basalt-derived soils following forest thinning treatments. Thomas (2017) also found open canopy conditions and forest thinning had significantly higher mean percent total vegetative cover.

It is well documented that there is a negative relationship between overstory density and understory vegetative cover in ponderosa pine ecosystems (Bakker, 2005; Moore and

Deiter, 1992; Laughlin et al., 2006). Forest thinning and existing open canopy conditions would be expected to support greater total percent understory cover due to increased light interception, reduced overstory competition and evapotranspiration by trees, more favorable soil moisture conditions, and less pine litter than is found under closed canopy conditions. However, as noted by Abella et al. (2015), understory vegetative cover responses to forest thinning may be moderated by soil type. Highly productive soils that support robust understory cover in the absence of treatments may not respond as strongly as less productive soils that support sparse understory cover before restoration treatments are implemented.

In this study, thinning on limestone-derived soils had greater percent vegetative cover than either closed or open canopy conditions, which is supported by Abella et al. (2015) who also noted a significant increase in understory cover on limestone-derived soils following forest thinning. In this study, mean percent total vegetative cover under forest thinning on basalt- and benmoreite-derived soils was statistically similar yet were significantly lower than on limestone-derived soils, which corresponds with the findings of Abella et al. (2015).

Mean percent total vegetative cover was not significantly different between grazed and non-grazed treatments. This finding agrees with that of Thomas (2017), who found grazing treatment did not affect litter, total vegetative cover, or exotic species cover. Abella et al. (2015) also generally support this finding since they observed grazing exclosures only had significant effects in an interaction with forest thinning. The findings of Bakker and Moore (2007) support the conclusion that ponderosa pine stand density more strongly influences understory vegetative cover than grazing.

Findings of mean percent grass cover in this study agree with both Thomas (2017) and Abella et al. (2015). Mean percent grass cover under forest thinning was not significantly different from open canopy conditions at the measurement plot level ($9m^2$), yet both were significantly greater than under closed canopy conditions. Although Abella et al. (2015) did not discuss the response of grasses specifically to forest treatments, given their high importance value in relation to other understory lifeform functional groups, grasses had a major role in responses of total vegetative cover in their study. Furthermore, Abella et al. (2015) found that vegetative cover increased substantially on both basalt- and benmoreite-derived soils. In a study of a range of forest restoration treatments on ponderosa pine forests in northern Arizona (untreated, thinned, thinned and burned, and burned by wildfire), Griffis et al. (2001) found graminoid abundance increased significantly with thinning and burning treatments, but decreased significantly in stands impacted by wildfire. Ffolliott and Gottfried (1989) studied the effects of small patch clearcutting on herbaceous understory production in mixed conifer stands in eastern Arizona that included ponderosa pine and found patch clearcuts produced significantly greater herbage than stands that received partial cutting treatments.

Thomas (2017) found the forest thinning treatment resulted in greater prevalence of forb species than closed and open canopy conditions, which supports the finding of this study, and Griffis et al. (2001) found an increase in exotic forb species with increased forest treatment intensity, with the greatest abundance and prevalence of exotic forbs occurring in wildfires.

Grazing by both domestic and wildlife herbivores may have contributed to an increase in exotic forb cover under closed canopy conditions since several of the forb species observed in grazed plots are exotic (e.g., common mullein (*Verbascum Thapsus*, L.), Dalmatian toadflax (*Linaria Dalmatica*, L. Mill.), and common dandelion (*Taraxacum officinale* F.H. Wigg.). It is likely, however, that some of these exotic forbs are more effective at competing for scarce soil resources (nutrients and soil moisture) and persisting under conditions of sunlight deprivation and thick needle cast than native grasses and forbs. The multi-year analysis indicates that forest treatment type affects mean percent forb cover over time. While it was not a significant predictor in 2015 alone, forest treatment type did have a significant effect on mean percent forb cover in the multi-year model, as both an individual predictor as well as through interactions with measurement year and grazing. High variability in forb cover was particularly evident in the multi-year model, with standard deviations often exceeding means, indicating significant dispersion in mean forb cover values at the subplot level. There were no observed outliers in the mean forb cover data, distributions were normal, and kurtosis was only slightly positive (+ 3), which did not warrant removal of any data or transformation of these data to meet linear regression analysis objectives.

Limestone-derived soils had the highest overall levels of mean percent forb cover under closed canopy conditions in 2015 and in the multi-year model. This finding may display the importance of considering soil parent material type when implementing forest restoration treatments, particularly as it relates non-native (exotic) plants that could also be invasive and potentially allelopathic.

Thomas (2017) also observed greater forb abundance at the subplot level under forest thinning in comparison to open and closed canopy conditions. Abella et al. (2015) found forb colonization was the primary driver of increased species richness under grazing, and suggested seed dispersal by herbivores or soil disturbance from thinning on the rocky basalt-derived soils may have contributed to this effect.

Forest thinning was found to increase mean percent exotic plant cover on all soil parent material types in this study. Thomas (2017) also observed an increase in mean exotic plant cover in their companion study. Mean percent exotic plant cover was significantly lower in this study than was found by Thomas (2017) for all forest treatment types. Limestone-derived soils had the most pronounced increase in mean percent exotic plant cover, but these soils also tend to have the greatest overall total vegetative cover. Mean exotic plant cover on limestone-derived soils represented 26 percent of the mean total vegetative cover on these soils after forest thinning. This high percentage of exotic plant cover may pose a risk to ecosystem processes on these highly productive soils. What is perplexing about this finding is this high exotic plant cover was strongly corelated to grazing exclusion. It is possible that the low AUMs and conservative grazing are preventing exotic plant invasion on limestone-derived soils. Griffis et al. (2001) did not evaluate grazing impacts in their study of understory responses to forest management treatments but observed a stronger response of exotic species to forest treatments than native plant species. In a study of the effects of grazing and soil quality on native and exotic plant species diversity on Rocky Mountain grasslands in Colorado, Wyoming, and Montana, Stohlgren et al. (1999) found no significant difference in cover of lifeforms between grazing and non-grazing, and concluded that grazing appears to have minimal effect on the rate of spread of most exotic plant species at landscape scales. In a study of the encroachment of cheatgrass (*Bromus tectorum*) in ponderosa pine stands that had been thinned and burned, Gottfried et al. (2008) determined that interactions of prescribed fire, forest thinning, livestock grazing, and drought contributed to the large increase of this invasive species into treated stands.

Mean percent perennial plant cover findings do not fully agree with those of Thomas (2017), who reported that forest restoration treatments and soil parent material had no significant effect on mean percent perennial plant cover. Differences in the findings of this study and those of Thomas (2017) could be due in part to the timing when measurements occurred. Thomas (2017) sampled late in the cool growing season and early in the warm growing season (June and July) for northern Arizona whereas vegetation measurements for this study were completed near the end of the monsoon and warm growing season (August and September). It is likely that vegetative cover and floristic composition had increased between measurements conducted by Thomas (2017) and measurements taken for this study. The findings of this study are supported, in part by those of Abella et al. (2015), who found that limestone-derived soils had greater perennial plant cover than basalt- and benmoreite-derived soils prior to forest restoration treatments and perennial plant cover increased after forest thinning treatments were implemented. The authors further found the response of total plant cover to forest thinning was weakest on basalt-derived soils, which was also a finding in this study. However, in this study, mean perennial plant cover under open canopy conditions was not significantly different between basalt- and limestone-derived soils. Additionally, Abella et al. (2015) reported the magnitude of increase in native perennial graminoid cover was greatest in the absence of grazing. This study found that perennial plant cover displayed an increasing trend over time following forest thinning and the trend was more pronounced under non-grazing. Abella et al. (2015) also found that grazing exclusion had a significant positive effect on mean perennial plant cover under forest thinning. The findings of this study partially agree. However, this study also found a significant effect from grazing exclusion under closed canopy conditions. Grazing exclusion had no significant effect on mean perennial plant cover under open canopy conditions.

Species Richness

This study found no significant differences in mean total species richness at the measurement plot level between measurement years or grazing treatments. Additionally, no significant interactions were found at the measurement plot level among forest treatments, grazing treatments, soil parent material types or measurement years. Thomas (2017) reported grazing had minimal effect on subplot level species richness. The authors reported that species richness was only affected by grazing in a two-way interaction between forest treatment type and grazing treatment. They also found there was a four-way interaction among all independent variables. This study partially supports the findings of Thomas (2017), since an effect from grazing on mean total species richness was only found in a three-way interaction between forest treatment type, soil parent material type, and grazing treatment. This study found closed canopy conditions on limestone-derived soils that were grazed had significantly lower mean total species richness that non-grazed ($p = 0.002$).

Mean total plant species richness was greatest under open canopy conditions followed by forest thinning. Closed canopy condition had the lowest mean total species richness. This finding is partially supported by Abella et al. (2015) who found forest thinning significantly increased mean total species richness on both benmoreite- and basalt-derived soils, although their study found mean total species richness on limestone-derived soils did not increase. In a study of forest mosaics consisting of areas with tree cover intermingled with non-treed areas in the Front Range and the Uncompahgre Plateau of Colorado, Matonis and Binkley (2018) found understory species richness increased up to 3 percent per meter with distance from overstory trees after forest thinning. Abella and Covington (2004) found total mean species richness was not significantly different between control, low- and moderate-intensity forest thinning, but species richness in a high intensity forest thinning that reduced stand density by 85 percent was twice as high as the other treatments.

Forest treatment type affected mean species richness of grasses at both subplot and measurement plot levels with open canopy conditions having greater mean species richness of grasses than closed canopy conditions and forest thinning. Mean species richness of grasses under closed canopy conditions and forest thinning was not significantly different.

Soil parent material type also affected mean species richness of grasses at the subplot and measurement plot levels. Limestone-derived soils had significantly greater mean species richness of grasses that basalt- and benmoreite-derived soils. Mean species richness of grasses was not significantly different between basalt- and benmoreite-derived soils. This trend was found in the analysis of the 2015 data only as well as in the multi-year model.

Measurement year affected mean species richness of grasses at the subplot level. There was a general trend of increasing species richness of grasses from the time treatments were implemented in 2003 through the 2015 measurement year. However, this trend did not hold when mean species richness of grasses was analyzed at the measurement plot level in which there was no significant difference in mean species richness of grasses from 2006 through 2015.

Mean species richness of grasses at the subplot level was only affected by grazing through a two-way interaction with forest treatment type when all measurement years were considered. Non-grazing under closed canopy conditions yielded greater mean species richness of grasses than grazing. However, under open canopy conditions and forest thinning, this trend was reversed with grazing resulting in greater mean species richness of grasses than non-grazing.

Abella et al. (2015) noted that species richness on limestone-derived soils did not increase although percent cover increased significantly. The authors concluded that the highly productive status of limestone-derived soils was already supporting an abundance of grass species, which constrained increases in species richness. In contrast, Abella noted that basalt-derived soils were not supporting a high degree of vegetative cover, which allowed for a greater response to forest thinning.

The findings of this study are partially supported by those of Thomas (2017). This study found open canopy conditions alone had the greatest mean species richness of grasses and closed canopy conditions and forest thinning were not significant. Thomas (2017) found open canopy conditions and forest thinning had greater mean species richness of grasses than closed canopy conditions and that open canopy conditions and forest thinning were not significantly different.

In a study of ponderosa pine understory response to forest treatments, Griffis et al. (2001) found no significant difference in mean species richness of graminoids (native or exotic) from forest thinning or burning treatments 8 and 14 years after thinning treatments were implemented.

Matonis and Binkley (2018) observed increasing species richness with increasing distance from tree cover in forest-meadow mosaics in northern Arizona (i.e., large herbaceous openings and interspaces intermingled among higher density forest cover) which supports the hypothesis that forest openings of sufficient size to minimize competition from the overstory for light, nutrients, and soil moisture support greater understory species abundance and richness.

Mean forb species richness had a similar trend as mean species richness of grasses. Open canopy conditions had greater mean forb species richness at the subplot level than closed canopy conditions and forest thinning in 2015 and in the multi-year model. Mean forb species richness also had a similar trend at the measurement plot level when all measurement years were analyzed, with open canopy conditions having greater mean forb species richness than closed canopy conditions and forest thinning. However, in 2015 there were no significant differences in mean forb species richness by forest treatment type at the measurement plot level. Thomas (2017) found a slightly different trend than this study. The author found greater mean forb species richness under forest thinning than under open and closed canopy conditions, and found mean forb species richness was not significantly different between open and closed canopy conditions. Abella et al. (2015) did not report forb species richness values in relation to forest treatments. However, the authors did note that increased total species richness on grazed plots was a result of forb colonization.

In their study of forest-meadow mosaics in Colorado, Matonis and Binkley (2018) found forb species richness increased with distance from trees, which would support the finding of greater forb species richness under open canopy conditions. Griffis et al. (2001) also evaluated forb species richness in response to forest management in northern

Arizona ponderosa pine. The authors found no significant difference in either native or exotic forb species richness between unmanaged stands, forest thinning, or thinning plus burning. However, they did observe a sharp increase in exotic forb species richness following a wildfire.

Laughlin et al. (2006) studied understory responses to ponderosa pine forest restoration treatments near Flagstaff, Arizona. Although they did not study species richness, they did study standing crops of perennial and annual forbs following forest restoration treatments. Legume standing crop in the remnant grass patches remained high and did not increase significantly in treatment areas. They found perennial forb standing crop did not change significantly throughout the study. However, the authors recommended future studies on species composition of perennial forbs among patches. The authors found annual forb standing crop was not significantly different among treatments at the beginning of the study but increased significantly in treatments where post-settlement trees were removed. They attributed this change to soil disturbance.

In this study, soils derived from limestone had significantly greater mean forb species richness than basalt and benmoreite-derived soils for all treatment types. Abella et al. (2015) noted high species richness on limestone-derived soils both before and after forest restoration treatments and grazing treatments were implemented.

Exotic plant species richness was affected by forest treatment type at the subplot and measurement plot level. In 2015 at the subplot level mean exotic plant species richness was greatest under forest thinning followed by open canopy conditions, then closed canopy conditions. When all years were analyzed at the subplot level, mean exotic plant species richness under thinning and open canopy conditions were not significantly different, but both were greater than under closed canopy conditions. At the measurement plot level, exotic plant species richness under forest thinning and open canopy conditions were the same in 2015 as when all measurement years were considered. Mean exotic plant species richness under forest thinning and open canopy conditions were not significantly different. Closed canopy conditions had significantly lower mean exotic plant species richness than open canopy conditions and forest thinning. Thomas (2017) reported mean exotic plant species richness increased three and five years following restoration treatments at both subplot and measurement plot levels. At the measurement plot level, Thomas (2017) reported the greatest mean exotic plant species richness was found under forest thinning. Abella et al. (2015) did not report exotic plant species richness in their study on these sites. In their study of understory responses to ponderosa pine forest restoration treatments and wildfires in northern Arizona, Griffis et al. (2001) reported species richness and abundance of exotic forbs and graminoids did not significantly increase under forest thinning in comparison to unmanaged stands. Thinning plus prescribed fire treatments did increase mean exotic forbs species richness, but not exotic graminoid richness.

This study supports the hypothesis that forest treatments that disturb soil surfaces have potential to increase exotic species abundance and richness. However, given the low level of impact to soils from forest thinning treatments in this study (hand felling with manual removal of residual woody debris), it is likely that other factors have contributed to expansions of exotic species populations, such as livestock and wildlife ungulates serving as vectors of seed dispersal, drought that inhibit maintenance and establish of native plant communities, and recreationists increasing ruderal areas where exotic species can establish and compete with native plant communities. Sieg et al. (2003) noted that some exotic plant species that are opportunistic invaders in the Southwest respond to disturbance related to timber harvesting activities and forest restoration such as skidding, log landing, slash pile burning, and fire, with severe disturbances that open forest canopies and expose mineral soils providing opportunities for invasion.

Limestone-derived soils had greater mean exotic plant species richness than basalt- and benmoreite-derived soils at both the subplot and measurement plot levels in 2015 and in the multi-year analysis. Thomas (2017) reported exotic species richness at the subplot level was significantly affected by soil parent material type and grazing treatment interaction. Non-grazing on basalt-derived soils had significantly lower mean exotic species richness than soils derived from limestone. Thomas (2017) also reported grazing treatments and several interactions among independent variables affected mean exotic plant species richness when all measurement years were analyzed. These included a two-way interaction between soil type and grazing treatment, a two-way interaction between treatment type and grazing treatment, two three-way interactions (soil type by treatment type by grazing and soil type by treatment type by measurement year), and a four-way interaction (soil type by treatment type by grazing by measurement year) when all measurement years were analyzed. Thomas (2017) also found closed canopy conditions and grazing treatments had greater mean exotic plant species richness than other treatments. Finally, the authors noted grazing under forest thinning treatments had the greatest exotic plant species richness at the measurement plot level.

In this study, at the subplot level, non-grazing had greater exotic plant species richness than grazing treatments in 2015 only. Grazing treatment did not significantly affect mean exotic plant species richness at the measurement plot level when all measurement years were included in the analysis.

This study and the findings of Abella et al. (2015) and Thomas (2017) strongly support the importance of pre-treatment inventories of understory plant communities prior to implementing forest restoration treatments as well as post-treatment monitoring of treatment outcomes. Inventories can be used to inform treatment priorities. Where populations of exotic species are identified, particularly problematic invasive species, forest restoration treatments should be delayed until exotic species control efforts have been implemented and found to be successful at containing or controlling exotic plant species infestations and reducing soil seed banks (Sieg et al., 2003). It would be beneficial to conduct vegetation inventories to identify populations of exotic plant infestations as a component of range readiness evaluations before stocking pastures. This practice would provide resource managers opportunities to treat exotic plant infestations and reduce the possibility of domestic livestock serving as vectors of exotic plant species dispersal.

Mean perennial plant species richness was affected by forest treatment type. Open canopy conditions had greater mean perennial plant species richness than closed canopy or forest thinning at both the subplot level and measurement plot level in 2015. However, no significant differences in mean perennial species richness were found among forest treatments at the subplot or measurement plot level in the multi-year analysis. At the subplot level in 2015, mean perennial species richness was greatest under open canopy conditions, intermediate under forest thinning, and lowest under closed canopy conditions. At the measurement plot level in 2015, open canopy conditions had the greatest mean perennial plant species richness, but forest thinning and closed canopy conditions were not significantly different.

This study also found soil parent material type affected mean perennial plant species richness. Limestone-derived soils had greater perennial plant species richness than basalt- or benmoreite derived soils at the subplot level. However, at the measurement plot level, mean perennial plant species richness was not significantly different between limestone- and basalt-derived soils. Both limestone- and basalt-derived soils had significantly greater mean perennial plant species richness than benmoreite-derived soils. This finding partially agrees with those of Abella et al. (2015) since the authors found limestone-derived soils had greater species richness than basalt- and benmoreite-derived soils. They also noted that soils derived from benmoreite supported low species richness. In this study, a general trend was observed in mean perennial plant species richness over time. Under open canopy conditions and forest thinning mean perennial plant species richness declined from 2006 to 2015. No significant changes occurred over time under closed canopy conditions.

The findings of this study do not agree with those of Thomas (2017) since the author reported that the only significant effect to mean perennial plant species richness was through grazing treatment at the measurement plot level, with grazing having greater perennial plant species richness than non-grazing. Since grazing treatments did not have significant effects on mean perennial species richness in this study, this declining trend in perennial species richness under open canopy conditions and forest thinning is perplexing and supports the hypothesis that confounding factors such as wildlife herbivory, drought, and spatial temporal variation in precipitation patterns across the study area are likely influencing variability in perennial plant species richness. Another factor that could be contributing to the decline in perennial plant species richness could be the length of time since fire has affected the study area. Prescribed fire and low burn severity wildfire have been shown to improve soil nutrient status and cycling through rapid incorporation of organic matter and carbon which, over time increases available nitrogen and other nutrients.

Forest treatment type was the only independent variable that affected mean annual plant species richness. Mean annual plant species richness under forest thinning was greater than open and closed canopy conditions. In the multi-year analysis, open canopy conditions had greater mean annual plant species richness followed by forest thinning then closed canopy conditions. This finding agrees with those of Abella and Covington

(2007) who studied the effects of a forest floor treatment of scarification in the absence of fire. The authors observed short lived native annuals such as *Chenopodium graveolen*, *Muhlenbergia ramulosa*, and *Nama dichotomum,* were more frequent in restoration prescriptions and exhibited overall increases through time. However, over the full duration of the study, no significant differences in vegetative composition or richness were found. Abella et al. (2015) did not report annual plant species richness as a component of their study. Thomas (2017) also did not report statistical findings of mean annual plant species richness.

No significant differences in mean biennial plant species richness at the measurement plot level were found among soil parent material types or grazing treatments when all measurement years were included and no statistically significant interactions among independent variables were found.

Mean biennial plant species richness was affected by forest treatment type in 2015 and when all measurement years were analyzed. Open canopy conditions and forest thinning treatments were not significantly different, but both treatments had significantly greater mean biennial plant species richness than closed canopy conditions. At the measurement plot level, forest treatment did not have a significant effect on mean biennial plant species richness in 2015. However, when all measurement years were analyzed, the same trend was found for subplot data in 2015 and when all years were analyzed. Both open canopy and forest thinning treatments had significantly greater mean biennial plant species richness than closed canopy conditions.

Soil parent material type affected mean biennial plant species richness at the subplot level. However, no significant differences in the effects of soil parent material were found at the measurement plot level in 2015 or when all years were included in the analysis. At the subplot level in 2015, limestone-derived soils had greater biennial plant species richness than basalt- and benmoreite-derived soils. However, when all measurement years were analyzed, limestone- and benmoreite-derived soils were not significantly difference in their effects on mean biennial plant species richness. Basalt-derived soils had lower mean biennial plant species richness that both limestone- and benmoreite-derived soils. Thomas (2017) found biennial species richness per measurement plot was influenced by treatment type with thinning treatments having higher biennial species richness. Abella et al. (2015) did not report findings of biennial plant species richness.

CONCLUSIONS

Hierarchical Linear Modeling (HLM) was used to evaluate responses of soil quality indicators and understory vegetation to three forest treat types with and without grazing. The HLM approach is an effective method for analyzing nested data. Users can add or remove independent variables in a stepwise manner to evaluate the incremental effect of each independent variable on the response, or dependent variable. Another benefit of HLM is fewer assumptions need to be met than required for ANOVA. For example, HLM allows time of measurements to be treated as either a fixed or random effect. In the case of this study, time was treated as a random effect since measurement intervals were not equally spaced. While HLM was an effective analytical approach in this study, it revealed that soil parent material type, forest treatment type, and grazing treatment type were not strong predictors of understory vegetation responses of mean percent vegetative cover and species richness. As evidenced by the low R-squared and adjusted R-squared values for each independent variable, confounding factors such as tree spatial distribution within and among treatments, timing of field measurements in relation to livestock grazing patterns including rest-rotation schedules, timing and levels of wildlife herbivory, drought, and spatial temporal variation in precipitation patterns across the study area likely have a major influence on variability in vegetative cover and species richness and these parameters were not analyzed in this study.

Forest ecological restoration treatments did not have significant effects on most of the select soil quality indicators analyzed in this study. There were no statistically significant differences in mean soil bulk density, or saturated infiltration capacity. However grazing treatments did significantly affect some soil quality indicators, both individually and through interactions with other independent variables that were analyzed in this study.

Saturated infiltration capacity was affected by soil parent material type under non-grazing with basalt-derived soils having significantly lower saturated infiltration capacity than limestone-derived. Spatial variation in horizonal soil textures and the presence of vertical tubular macropores and coarse fragments (stones, rocks, and channers) as well as shrinkage cracks, earthworm channels and root channels that lead to secondary porosity and macropore flow appear to have influenced infiltration rates through induced preferential flow, particularly under low antecedent soil moisture conditions. Antecedent soil moisture also appears to have influenced infiltration rates, but not infiltration capacities. Under dry condition, macropore flow in the A horizons and bypass flow in shrinkage cracks in the shallow Bt horizons was possible as evidenced by some of the infiltration curves plotted from saturated infiltration capacity tests.

Grazing affected mean soil aggregate stability with grazing having approximately 22 percent fewer water stable aggregates than non-grazing. This finding could indicate risk to soil stability and productivity since livestock trampling has potential to pulverize aggregates, reducing average aggregate size, which could subject them to accelerated loss of organic matter and other binding agents, and therefore productivity. Additionally, as aggregates disperse, risk of surface soil sealing could lead to water ponding, loss of infiltration, and accelerated runoff and soil erosion.

Soil slaking was affected by soil parent material type, grazing and forest treatment type. Grazing had lower soil slake ratings for all soil parent material types. Limestone-derived soils had the highest soil stability rating. All soil slake tests indicated a strong potential for soils to readily slake when wetted, regardless of forest treatment type or grazing vs. non-grazing. Average soil slake tests rarely exceed a rating of 3, which represents the midpoint of aggregate water dispersion potential. Given the clayey texture of soils analyzed in this study and the 2:1 smectitic clay mineralogy, soil propensity to slake would be high. Soil slaking is an indicator of soil structural integrity and could serve as an indicator of risk of soil crusting and risk of accelerated soil erosion through particle detachment and entrainment in runoff or aeolian transport. This is a useful soil quality indicator for assessing rangeland health and disturbed soils.

There were no statistically significant differences in levels of soil organic matter (OM), organic carbon (C), nitrogen (N), calcium (Ca^{2+}), potassium (K^{+}) (Mg ha^{-1}) in the upper 10 cm of soil on an areal basis by soil type, ecological restoration treatment type, or grazing management. There were statistically significant differences in soil magnesium (Mg^{2+}) content (Mg ha^{-1}) by soil parent material type. Basalt-derived soils had significantly greater mean Mg^{2+} content. This is likely due to the eruptive products of the San Francisco Volcanic Field, which are dominantly basaltic and rhyolitic and basalt typically has high percentages of Mg^{2+} and Fe^{2+}. Soil sodium (Na) levels (Mg ha^{-1}) were not significantly different by soil type or ecological restoration treatment type. However, grazing affected soil Na^+ levels, with grazing having an average of 5.04 Mg ha^{-1} less Na^+ than non-grazing conditions across all soil types and ecological restoration treatments. This finding was perplexing since grazing has generally been shown to increase localized soil Na^+ and Mg^{2+} concentrations since livestock urine and feces can have high concentrations of these elements particularly when mineral supplements are provided to livestock, as is common on USFS grazing allotments in Northern Arizona.

Basalt-derived soils had significantly lower soil phosphorus (P) levels than benmoreite-derived soils. Soil sulfur (S) levels (Mg ha^{-1}) were significantly different among soil types with limestone-derived soils having significantly lower soil S than basalt- and benmoreite-derived soils.

Vegetative cover and species richness were affected by forest treatment type, soil parent material type, grazing treatment, and measurement year, depending on life form, longevity, and nativity and were variable. As observed by Abella et al. (2015), responses were hierarchically controlled, with soil parent material type often moderating vegetative responses to treatments. Limestone-derived soils had the greatest overall total plant cover and benmoreite-derived soils often had the lowest. Grazing affected vegetative cover, depending on measurement year, but had minimal effect on species richness.

The analyses of vegetative responses to forest restoration and grazing treatments broadly support those of Thomas, et al. (2017) and Abella et al. (2015), with some minor differences that can mostly be attributed to the natural range of variation in ponderosa pine forested ecosystems and their associate climatic and edaphic ecosystem components. Vegetation responses were indeed diverse across soil parent material types and treatments. While grazing treatments seemed to have minimal effect on the findings of this study, it is important to note that grazing was not a well-controlled component of this research. It is quite possible that the pastures upon which this study was conducted were being rested from livestock grazing during the year that measurements were taken. It is also possible, given the low intensity of grazing on the allotment where this study was conducted that no livestock grazing occurred within grazable research plots.

While many of the soil quality indicators were not significant, this study is not without merit. Treatments for this research were implement using very low impact techniques (hand felling using chainsaws and manual removal of slash). Similar studies at operational scale using heavy logging machinery and followed by prescribed fire would likely contribute further to the body of knowledge on the utility of the soil quality indicators evaluated in this study and could lead to identification of applicable soil quality management thresholds.

This study highlights the importance of considering both inherent and dynamic soil properties when planning ecological restoration treatments since soil parent material type strongly influences vegetative responses of cover and richness, particularly in relation to livestock and wildlife forage production and sustaining healthy native plant communities following forest restoration treatments. Finally, understanding exotic plant species abundance and richness before forest ecological restoration treatment may prevent undesirable effects to native plant communities by preventing expansion of exotic plants.

Spatial autocorrelation may exist among subplots , and this could have affected statistical findings. The boundary of each subplot intersected two or more boundaries of adjacent subplots with no buffer to separate them. This may have resulted in some subplots having similar values to adjacent subplots since some plants were rooted in multiple subplots.

Grazing exclosure fences were 3.16m x 3.16m, so there was a 0.08m buffer on at least one outside boundary of each subplot, except the one located in measurement plot center. There is the possibility that exclosure fences intercepted snow and rain, which may have affected soil moisture levels along outer boundaries of subplots.

Ponderosa pine forested ecosystems are fire dependent. Studies which include operational scale forest thinning treatments followed by low intensity and low burn severity prescribed or managed wildland fire would further contribute to the body of knowledge of the effects of the full suite of forest ecological restoration treatments commonly used in southwestern ponderosa pine ecosystems.

A more detailed study of the O horizon, such as presence or absence, depths of Oi, Oe, and Oa components, detrital characteristics, and biota, including macroorganisms, fungal hyphae, and fine root biomass, might have better informed soil factors affecting aggregate stability, slaking potential and other physicochemical responses.

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APPENDIX A

Table A 1. Mean monthly minimum temperature for Flagstaff AZ from 1994 through 2014.

Year	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Annual
1994	-16.7	-20.0	-8.9	-6.1	-3.9	-0.6	2.8	8.9	-0.6	-8.9	-13.9	-11.7	-20.0
1995	-22.2	-10.0	-11.1	-10.6	-2.8	-4.4	2.8	7.8	-1.7	-6.7	-10.6	-17.8	-22.2
1996	-17.2	-19.4	-16.1	-7.2	-2.2	-0.6	7.8	3.9	-1.7	-10.6	-15.0	-14.4	-19.4
1997	-21.7	-16.1	-23.3	-8.9	-5.6	-2.2	1.1	6.7	0.0	-8.9	-9.4	-22.2	-23.3
1998	-18.9	-21.1	-18.3	-12.2	-4.4	-2.2	4.4	6.1	-1.1	-7.8	-12.2	-22.8	-22.8
1999	-12.2	-13.3	-9.4	-13.3	-3.3	-2.8	7.8	2.2	-1.1	-8.3	-16.1	-15.6	-16.1
2000	-17.8	-11.7	-12.2	-6.1	-4.4	0.0	4.4	5.6	0.0	-6.1	-19.4	-12.8	-19.4
2001	-19.4	-20.0	-12.2	-6.7	-7.2	-3.9	2.8	4.4	-0.6	-4.4	-10.6	-22.2	-22.2
2002	-20.0	-15.6	-19.4	-7.2	-4.4	2.2	8.3	2.2	0.6	-3.3	-8.9	-20.6	-20.6
2003	-12.8	-13.9	-14.4	-11.1	-5.0	0.6	7.2	7.8	0.6	-2.2	-13.3	-17.2	-17.2
2004	-19.4	-20.0	-13.3	-5.6	-3.9	-1.1	0.6	4.4	-2.8	-9.4	-17.8	-14.4	-20.0
2005	-20.0	-13.9	-15.6	-7.8	-4.4	-0.6	3.3	5.6	-1.7	-3.9	-15.6	-16.7	-20.0
2006	-18.3	-14.4	-16.1	-5.0	-2.8	0.6	7.8	3.9	-2.8	-6.7	-11.7	-16.1	-18.3
2007	-26.1	-20.0	-14.4	-10.0	-6.1	-1.1	8.9	8.3	-2.2	-7.2	-11.1	-22.2	-26.1
2008	-21.7	-22.2	-11.7	-7.8	-5.6	-0.6	7.2	7.8	1.7	-11.7	-11.1	-21.1	-22.2
2009	-13.9	-19.4	-11.1	-9.4	0.6	0.6	7.8	1.7	1.1	-11.1	-10.0	-21.1	-21.1
2010	-16.1	-16.7	-13.9	-8.9	-6.7	0.6	3.9	5.6	1.1	-3.3	-20.0	-25.0	-25.0
2011	-28.3	-20.0	-12.2	-13.9	-7.2	-2.8	2.8	8.3	2.2	-4.4	-13.9	-17.8	-28.3
2012	-10.6	-13.3	-11.1	-6.7	-3.9	1.1	8.9	8.9	1.1	-7.2	-12.8	-20.0	-20.0
2013	-22.8	-22.2	-12.2	-10.6	-4.4	4.4	8.9	5.6	-3.9	-6.7	-11.1	-23.3	-23.3
2014	-11.7	-14.4	-11.7	-8.3	-3.9	0.6	7.8	5.0	0.0	-3.3	-11.7	-15.0	-15.0
Mean	-18.3	-17.2	-13.9	-8.9	-4.4	-0.6	5.6	5.6	-0.6	-6.6	-13.3	-18.3	-21.1

Table A 2. Mean monthly maximum temperature for Flagstaff AZ from 1994 through 2014.

0.56	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Annual
1994	15.6	14.4	20.0	25.0	27.2	27.2	31.1	32.8	27.8	20.6	18.3	13.3	32.8
1995	14.4	15.6	17.8	18.9	22.8	29.4	34.4	32.2	30.0	23.3	19.4	16.7	34.4
1996	16.7	18.9	18.3	26.1	30.0	31.1	33.3	31.1	27.2	27.2	18.9	15.0	33.3
1997	13.3	12.2	21.1	20.6	28.3	28.3	32.8	30.0	27.2	23.9	20.0	10.6	32.8
1998	12.2	12.2	18.9	20.6	21.7	31.7	32.8	30.0	25.0	22.8	18.3	15.6	32.8
1999	14.4	18.3	17.8	21.1	23.3	31.7	31.1	28.9	27.8	23.9	21.1	13.3	31.7
2000	16.1	15.0	17.8	25.6	30.0	32.8	32.2	32.2	31.1	26.1	15.0	14.4	32.8
2001	13.3	13.9	18.9	23.3	28.9	32.2	32.2	30.6	28.3	23.9	18.3	11.1	32.2
2002	16.7	17.8	20.6	23.3	31.7	31.7	34.4	32.8	28.9	23.3	17.2	11.1	34.4
2003	16.7	15.6	17.2	18.3	30.0	30.6	34.4	31.1	27.8	26.7	17.2	15.0	34.4
2004	10.6	13.3	22.2	21.7	27.2	29.4	30.6	29.4	27.2	22.2	17.2	16.1	30.6
2005	12.8	10.0	17.2	19.4	28.9	31.1	33.9	28.9	26.1	22.2	18.9	15.6	33.9
2006	14.4	15.6	15.0	21.7	27.2	32.8	32.2	27.8	26.1	22.8	20.0	13.9	32.8
2007	12.8	12.8	22.8	24.4	27.2	31.1	35.6	30.0	27.8	22.2	20.0	17.8	35.6
2008	8.3	15.0	17.8	21.7	29.4	31.7	31.7	31.7	28.9	24.4	21.1	18.9	31.7
2009	13.9	15.0	19.4	22.8	27.8	31.1	33.9	32.2	28.3	24.4	22.8	11.1	33.9
2010	12.2	10.6	16.7	20.6	23.9	32.2	32.2	30.6	28.9	26.7	21.1	19.4	32.2
2011	15.0	16.7	21.7	23.3	23.9	31.7	31.1	30.0	28.9	23.9	18.9	16.1	31.7
2012	15.0	13.9	18.9	25.6	28.9	31.7	31.7	31.1	26.7	25.6	20.6	16.7	31.7
2013	11.1	13.3	19.4	25.0	26.7	35.6	33.3	28.9	27.2	21.1	17.8	15.6	35.6
2014	15.6	18.9	16.7	21.7	27.8	30.6	31.1	28.9	26.7	23.9	20.6	12.8	31.1
Mean	13.9	14.4	18.9	22.2	27.2	31.1	32.8	30.6	27.8	23.9	19.4	15.0	32.8

Table A 3. Mean monthly precipitation for Flagstaff AZ from 1994 through 2014.

Year	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Annual
1994	0.38	2.47	3.03	2.48	1.01	M	1.7	3.61	2.75	1.12	1.91	1.43	M
1995	2.39	3.77	3.99	1.49	0.88	0.06	0.61	2.29	M	0.01	0.31	0.35	M
1996	0.19	1.36	0.54	0.07	T	T	1.79	0.92	3.73	0.93	M	0.64	M
1997	3.21	0.99	0.03	0.60	0.08	0.18	0.21	2.83	3.68	1.15	0.8	1.85	15.61
1998	1.30	2.15	3.74	1.52	1.31	0	4.72	2.82	4.45	3.11	1.76	0.42	27.3
1999	0.28	0.48	0.53	2.80	0.42	0.95	3.27	2.45	4.54	T	0	T	15.72
2000	0.62	1.61	3.12	0.19	0.12	1.11	0.29	2.83	0.36	3.85	1.07	0.21	15.38
2001	2.60	1.68	1.28	1.40	0.82	0.03	2.80	3.46	0.68	1.21	0.43	1.16	17.55
2002	0.02	0.07	0.62	0.51	0	0	2.60	1.00	4.01	1.88	1.48	0.69	12.88
2003	0.14	2.75	1.13	0.44	0.73	0.04	3.40	3.03	2.62	0.14	2.51	0.92	17.85
2004	0.76	1.06	0.74	1.81	0	0.02	1.47	4.71	1.76	3.51	3.10	4.67	23.61
2005	6.58	4.19	2.43	2.15	0.08	0.40	2.51	3.41	0.46	1.59	0.2	0.01	24.01
2006	0.23	0.09	2.16	0.99	0.08	0.65	4.07	2.83	1.24	2.55	0.06	0.61	15.56
2007	1.20	0.81	0.51	0.38	0.14	0.01	2.86	2.56	2.90	0.38	1.38	4.33	17.46
2008	3.95	2.56	0.04	T	1.17	0.02	2.35	2.40	0.69	0.21	1.29	4.17	18.85
2009	0.73	1.48	0.22	0.33	2.08	0.36	1.00	0.74	0.77	0.21	0.88	2.85	11.65
2010	5.60	1.81	1.31	0.57	0.10	0.09	5.94	3.56	0.79	2.93	1.79	3.40	27.89
2011	0.06	3.25	0.95	1.14	0.87	0	2.31	2.76	3.36	2.05	1.76	2.16	20.67
2012	0.42	0.99	1.81	1.24	0	0	3.45	2.59	0.26	1.03	1.02	2.08	14.89
2013	2.64	0.86	1.38	0.12	0.23	0.03	7.57	4.85	3.25	0.29	2.22	1.35	24.79
2014	0.17	0.43	1.24	1.25	0.12	0	4.32	5.31	3.10	0.69	0.60	3.44	20.67
Mean	1.59	1.66	1.47	1.02	0.49	0.20	2.82	2.90	2.27	1.37	1.23	1.75	19.02

APPENDIX B

Table B 1. Means and standard deviations for soil macronutrients levels (Mg ha^{-1}) across soil parent material types, forest ecological restoration treatments and grazing treatments. Concentrations (mg kg^{-1}) reported by Thomas (2015) using Mehlich III extractant solution.

Nutrient (Mg ha^{-1}) Soil Type	Restoration Treatment	Grazed	Excluded
N			
Basalt			
	Closed	5.854 (2.580)	5.015 (3.131)
	Open	3.649 (0.561)	4.340 (0.019)
	Thinned	13.694 (12.950)	4.619 (0.480)
Benmoreite			
	Closed	6.221 (4.502)	3.783 (0.361)
	Open	5.369 (1.456)	7.064 (6.924)
	Thinned	5.788 (2.036)	5.687 (2.689)
Limestone			
	Closed	4.371 (1.073)	3.423 (1.287)
	Open	4.791 (1.537)	3.761 (0.844)
	Thinned	4.264 (0.235)	3.953 (0.778)
P			
Basalt			
	Closed	17.207 (1.350)	13.811 (0.197)
	Open	21.756 (2.376)	27.622 (6.017)
	Thinned	26.221 (23.926)	22.415 (5.884)
Benmoreite			
	Closed	60.481 (34.301)	65.399 (14.536)
	Open	51.692 (14.584)	40.630 (8.955)
	Thinned	52.390 (48.454)	35.407 (21.829)
Limestone			
	Closed	30.849 (29.457)	20.415 (12.889)
	Open	56.406 (27.091)	81.335 (56.344)
	Thinned	43.963 (34.437)	50.192 (48.892)
K^+			
Basalt			
	Closed	192.528 (30.349)	172.862 (67.037)
	Open	260.444 (6.279)	262.966 (15.978)
	Thinned	238.112 (68.054)	190.132 (70.338)

Continued

Nutrient (Mg ha^{-1}) Soil Type	Restoration Treatment	Grazed	Excluded
Benmoreite	Closed	216.108 (35.274)	284.172 (27.301)
	Open	254.471 (36.402)	210.151 (18.742)
	Thinned	177.131 (115.253)	165.173 (67.753)
Limestone			
	Closed	166.217 (47.967)	174.217 (19.041)
	Open	223.138 (93.922)	277.589 (100.742)
	Thinned	178.228 (32.491)	222.285 (97.743)
Ca^{2+}			
Basalt			
	Closed	2872.410 (323.534)	2809.092 (497.214)
	Open	2600.782 (1577.214)	2492.721 (1334.942)
	Thinned	3202.361 (404.347)	2219.095 (273.835)
Benmoreite			
	Closed	1778.792 (1028.667)	2006.303 (581.559)
	Open	2983.166 (597.662)	2637.264 (505.106)
	Thinned	2421.127 (278.491)	2116.633 (187.191)
Limestone			
	Closed	2367.381 (348.490)	2356.090 (421.013)
	Open	2541.016 (166.163)	2260.463 (323.078)
	Thinned	2087.966 (678.489)	2119.440 (182.389)
Mg^{2+}			
Basalt			
	Closed	671.250 (63.207)	736.832 (98.951)
	Open	670.392 (543.204)	590.662 (465.025)
	Thinned	826.566 (386.413)	434.590 (22.339)
Benmoreite			
	Closed	329.587 (243.547)	349.006 (137.350)
	Open	526.050 (147.202)	471.225 (177.527)
	Thinned	471.905 (122.429)	392.555 (67.302)
Limestone			
	Closed	548.215 (35.170)	585.376 (80.062)
	Open	392.419 (64.745)	371.110 (81.380)
	Thinned	380.178 (126.903)	396.869 (38.755)
S			

Continued

Nutrient (Mg ha^{-1}) Soil Type	Restoration Treatment	Grazed	Excluded
Basalt	Closed	5.322 (0.997)	5.343 (0.284)
	Open	6.160 (1.863)	7.168 (2.486)
	Thinned	6.964 (0.281)	5.983 (0.363)
Benmoreite			
	Closed	6.556 (0.417)	8.169 (1.750)
	Open	7.088 (3.062)	7.225 (3.543)
	Thinned	7.186 (2.865)	5.824 (2.032)
Limestone			
	Closed	7.936 (0.840)	7.978 (1.509)
	Open	8.424 (0.915)	7.789 (1.275)
	Thinned	8.784 (0.598)	9.783 (1.645)

APPENDIX C

United States Department of Agriculture **Forest Service** **Flagstaff Ranger District**

5075 North Highway 89
Flagstaff, AZ 86004-2852
Phone (928) 526-0866
FAX (928) 527-8288

File Code: 2210/2230
Date: April 2, 2015

This document serves as your 2015 Annual Operating Instructions (AOI) for the Mooney Mountain and Woody Mountain Allotments and Beaverhead-Grief Hill Driveway.

This AOI is part of your Term Grazing Permit as provided for in Part 2, section 8(a).

I. AUTHORIZED LIVESTOCK NUMBERS/PERIOD OF USE

Your 2015 Authorized Use Request has been approved and you are authorized the following Livestock Numbers and Periods of Use:

2670	Sheep	May 1, 2015 to May 10, 2015	Beaverhead-Grief Hill Driveway
14	Burros	May 1, 2015 to May 10, 2015	Beaverhead-Grief Hill Driveway
2670	Sheep	May 15, 2015 to May 31, 2015	Beaverhead-Grief Hill Driveway
14	Burros	May 15, 2015 to May 31, 2015	Beaverhead-Grief Hill Driveway
830	Sheep	June 1, 2015 to October 15, 2015	Woody Mountain
1840	Sheep	June 1, 2015 to August 31, 2015	Mooney Mountain

The 2015 authorized livestock numbers and the authorized period of use were determined based on referencing the Western Regional Climate Center 12-month Standardized Precipitation Index (Northeast Divisio*n, Arizona; Climate Division 2) and allotment inspections conducted prior to the grazing season.

Any increase in the 2015 authorized livestock numbers will require that you file an additional Authorized Use Request. Be advised that the Forest Service will need a minimum of 45 days to assess resource conditions and process a supplemental grazing bill for additional livestock. Please contact your Rangeland Management Specialist prior to making any decisions regarding increased livestock numbers.

II. GRAZING SCHEDULE

The following is the planned grazing sequence for the 2015 grazing period. The pasture move dates shown are an estimate and may need to be amended based on actual conditions.

Beaverhead-Grief Hill Driveway

Pasture	Planned Graze Period	Livestock Numbers
Beverhead-Grief Hill Driveway; Prescott National Forest	5/1/2015-5/10/2015	2670 Sheep, 14 Burros
Beaverhead-Grief Hill Driveway; Coconino National Forest	5/15/2015-5/31/2015	2670 Sheep, 14 Burros

***Use of the Beaverhead-Grief Hill driveway includes crossing the Verde River through occupied and proposed critical habitat (PCH) for the federally-listed Mexican gartersnake and narrow-headed gatersnake. Due to pending Endangered Species Act (ESA) Section 7 consultation for these recently listed species, sheep are not authorized to cross the Verde River on the Beaverhead-Grief Hill driveway for the 2015 grazing season. To avoid using the Verde River crossing, sheep will use the Prescott NF portion of the driveway to the Hayfield Draw OHV area (approx. 0.75 miles west of the Verde River) where they will be loaded onto trucks and hauled to State/Private land grazing allotments near Williams. Sheep use of the Beaverhead-Grief Hill Driveway on the Coconino will begin near the Rocky Park exit on I-17 (approx. 22 miles west/northwest of the Verde River). Sheep will be trucked to this location, off-loaded and continue use of the driveway north and west to the Woody Mountain allotment on the Coconino NF and the Garland Prairie allotment on the Kaibab NF.**

Mooney Mountain

Pasture	Planned Graze Period	Livestock Numbers
Mooney Mountain West	7/15/2015-8/31/2015	1840 Sheep
Mooney Mountain East	9/1/2015-10/15/2015	1840 Sheep

Woody Mountain

Unit	Planned Graze Period	Livestock Numbers
Fry	6/1/2015-6/8/2015	830 Sheep; 815 Sheep (AZ st)
Black Tank	6/9/2015-6/16/2015	830 Sheep; 815 Sheep (AZ st)
Aspen Springs	6/17/2015-6/24/2015	830 Sheep; 815 Sheep (AZ st)
Garden Springs	6/25/2015-7/3/2015	830 Sheep; 815 Sheep (AZ st)
Tule Tank	7/4/2015-7/11/2015	830 Sheep; 815 Sheep (AZ st)
Budweiser Tank	7/12/2015-8/2/2015	830 Sheep; 815 Sheep (AZ st)
Black Springs	8/3/2015-8/18/2015	830 Sheep; 815 Sheep (AZ st)
Fry	8/19/2015-8/27/2015	830 Sheep; 815 Sheep (AZ st)
Black Tank	8/28/2015-9/4/2015	830 Sheep; 815 Sheep (AZ st)
Aspen Springs	9/5/2015-9/13/2015	830 Sheep; 815 Sheep (AZ st)
Garden Springs	9/14/2015-9/22/2015	830 Sheep; 815 Sheep (AZ st)

Tule Tank	9/23/2015-10/2/2015	830 Sheep; 815 Sheep (AZ st)
Budweiser	10/3/2015-10/15/2015	830 Sheep; 815 Sheep (AZ st)

Changes from the grazing schedule will be made only with Forest Service approval. The planned use period in a pasture may be shortened or lengthened depending on forage availability and utilization in key areas. Moves from one grazing unit to the next grazing unit will be completed within a two (2) day period; sheep will be actively herded from one grazing unit to another. It is imperative that sheep remain within the designated grazing unit. For various reasons (i.e. pasture gates left open, cattleguards not maintained, cut fences, etc.), sheep may stray from the designated grazing unit. Preventing sheep from grazing adjacent allotments, your allotment's rested areas, or regrazing previously used units is the permittee's responsibility.

III. DROUGHT MANAGEMENT

Drought is an inevitable occurrence in the southwestern United States. Both the Forest Service and grazing permittees must plan for drought as a normal part of management. During periods of drought, the following management actions may be necessary:

- Stocking levels may need to be reduced. Reductions may be necessary prior to the permitted season of use and/or during the permitted season of use.
- Season of use may need to be shortened. Depending on the severity of the drought and the stocking level, a reduced grazing season may be necessary.
- Pasture use periods may need to be shortened. Pastures will not be regrazed during the same grazing season and this may ultimately result in an early exit from the allotment.
- Utilization and/or seasonal utilization (grazing intensity) standards may need to be reduced. Depending on the severity of the drought and the stocking level, reduced utilization standards will likely result in shortened pasture use periods and may ultimately result in an early exit from the allotment.
- Lack of livestock water, or poor distribution of livestock water, may result in reduced pasture/allotment use periods. Depending on forage conditions, this effect may be mitigated by hauling water to temporary livestock watering facilities.
- Allotment Management Plan direction for pastures requiring rest or deferment will be followed. Livestock use of planned rested pastures due to drought will not be authorized.

IV. GRAZING STRATEGY

Beaverhead-Grief Hill Driveway

Use of the Prescott National Forest portion of the driveway will occur from May 1 to May 10. Please contact Kelli Spleiss at 928-567-1126 to schedule when she should meet you in the field.

Use of the Coconino National Forest portion of the driveway will occur from May 15 to May 31.

Mooney Mountain

The Mooney Mountain allotment is used in conjunction with the Garland Prairie and Pomeroy allotments on the Kaibab National Forest. The 90 day use period of the Mooney Mountain allotment will be deferred annually; occurring either from June 1 to August 31 or from July 16 to October 15. Sheep will be day herded and a deferred rotation grazing system will be used on the two grazing units of the allotment. The management of sheep movement on this allotment will be based upon seasonal utilization guidelines and allowable forage utilization standards.

Woody Mountain
Sheep will be day herded and a deferred rotation grazing system will be used on the six grazing units of the allotment and one Arizona State grazing unit. The management of sheep movement on this allotment will be based upon seasonal utilization guidelines and allowable forage utilization standards.

V. ALLOWABLE UTILIZATION STANDARDS

Upland and Non-Riparian Areas
The maximum allowable level of utilization on herbaceous and non-riparian woody vegetation is 35%. This allowable use level is the utilization level permitted by both livestock and wildlife. Utilization is defined as the proportion or degree of current year's forage production by weight that is consumed or destroyed by animals. It is a comparison of the amount of herbage left with the amount of herbage produced during the year. Utilization will be measured at the end of growing season when the total annual production can be accounted for and the effects of grazing in the whole management unit can be assessed.

Aspen
Utilization of aspen will be limited to 20% of current year's growth.

Riparian Areas
Utilization of woody vegetation in riparian areas will not exceed 20%.

VI. SEASONAL UTILIZATION

A light to moderate seasonal utilization guideline of up to 50% will be used within grazed pastures on the allotments. Seasonal utilization is defined as the amount of herbage removed through grazing or trampling during the grazing period. Seasonal utilization will be measured at the end of each pasture's livestock grazing period.

The seasonal utilization guideline will be the primary factor in determining when livestock need to move to the next grazing unit; but other factors such as climatic conditions, opportunity for plant growth, and previous year's utilization level will also be considered. Annual reductions in the seasonal utilization guideline may be made based on resource conditions.

VII. RESOURCE MONITORING

Forage Utilization
Forage utilization within each pasture will be monitored and the allowable utilization standards (section IV) will be enforced. This may result in earlier pasture moves than indicated in the Grazing Schedule (section II).

Numerous key areas for forage utilization monitoring have been identified for the Mooney Mountain and Woody Mountain allotments. Key areas are defined as "…a portion of range, which because of its location, grazing or browsing value, and/or use, serve as an indicative sample of range conditions, trend, or degree of use seasonally. A key area guides the general management of the entire area of which it is a part." Key areas will generally have the following characteristics: they will be approximately 20 to 500 acres in size; they will be located on productive soils with slopes less than 15%; they will have a plant community that is representative of the larger area; and they will generally be located 0.25 to 1.0 miles away from livestock water sources.

The location of key areas for forage utilization monitoring, key vegetation species, and the specific application to your allotment should be discussed with your Rangeland Management Specialist. Permittee participation in utilization monitoring is encouraged.

Permittee Monitoring
The permittee will monitor and document the following items on the enclosed Actual Use and Forage Utilization Monitoring Form: 1) Actual Grazing Period; 2) Actual Livestock Numbers; and 3) Average Utilization of all forage species at two different time periods: Just prior to livestock entering the pasture, and immediately after livestock leave the pasture. **It is very important that this monitoring be completed during the time frames specified, and that the Forage Utilization Monitoring Form is submitted to your Rangeland Management Specialist at the end of the 2015 grazing period.**

VIII. MITIGATION MEASURES FOR SPECIAL STATUS SPECIES

The following mitigation measures are required for threatened and endangered species on the Beaverhead-Grief Hill Driveway and the Mooney Mountain and Woody Mountain Allotments.
Mexican Spotted Owl

- No human disturbance or construction activities associated with livestock grazing operations will occur within protected activity centers (PACs) during the breeding season (March 1 through August 31).
- Bedding of sheep is not allowed within PACs.
- Continue to monitor grazing use by livestock and wildlife in the ponderosa pine/gamble oak vegetation type and specifically within PACs.
- In addition to the Salting and Protein Block practices identified in Section XI, the following additional guidelines are required within Mexican spotted owl habitat:
  1) Do not place salt, mineral block, or supplements within riparian areas, mountain meadows, or non-riparian drainages in ponderosa pine.
  2) Do not place salt, mineral block, or supplements within spotted owl PAC's.

IX. RANGE IMPROVEMENTS

Construction of New Structural Range Improvements

There are no new structural range improvements scheduled for your allotment this year.

Removal of Structural Range Improvements

There are no structural range improvements scheduled for removal this year.

Existing Structural Range Improvements

Structural range improvements assigned to you for maintenance are listed in your Term Grazing Permit and are also identified on the allotment map. These improvements are to be fully maintained annually to comply with permit requirements (Part 2, Section 8i). Any maintenance you perform must conform to the standards specified by your District Rangeland Management Specialist. The grazing permittee is responsible for all maintenance materials, supplies and equipment necessary to properly maintain all range structural improvements. The Forest will replace range structural improvement materials and/or supplies at the end of the improvement's life; when maintenance and repair is no longer feasible to keep the improvement properly maintained and functioning. **Please note that you must notify the District Rangeland Management Specialist at least 60 days prior to the beginning of any maintenance work that requires the use of heavy equipment.**

X. SALTING AND PROTEIN BLOCK PRACTICES

Proper salt and protein block location can be a good tool to aid in livestock distribution and they will be used in a manner to spread livestock utilization throughout the pasture. The following guidelines will be used when placing salt and protein blocks:

1. In general, salt and protein blocks should not be placed within ¼ mile of water.
2. Salt and protein blocks will not be placed in areas of depleted range, erosive soils, or sensitive plant or animal species. Areas of sensitive plant or animal species will be identified by your Rangeland Management Specialist and included in each year's AOI.
3. No more than three blocks (50 lb. blocks) of salt/protein will be placed at any location at any one time.
4. Salt/protein will not be placed at the same location twice during the same grazing season.
5. The use of portable salt/protein block containers is encouraged but not mandatory.

XI. PORTABLE WATER HAULING

Temporary sites for portable haul water may be needed and should be used as necessary to assist in livestock distribution. The following requirements will apply to portable haul water locations:

1. Coordinate with the District Rangeland Management Specialist to identify portable water haul locations for individual pastures prior to the grazing period.
2. To aid in livestock distribution, the portable water haul locations should generally be in areas of light forage utilization.

3. Generally, portable water haul locations will not be located at sites used in previous years.
4. Portable water hauls will not be located in areas of depleted range, erosive soils, or sensitive plant or animal species.
5. Portable water haul locations will be moved when the desired forage utilization levels have been reached.
6. Portable haul water storage tanks and troughs will be removed when livestock leave the pasture.

XII. FIRE

Please use caution during all ranch activities that could potentially start a fire. All ranch vehicles must be equipped with an axe, shovel, and water bucket. All open camp and branding fires require a ten foot fire ring cleared down to mineral soil.
During periods of critical fire danger the Forest Supervisor may declare a Red Flag Alert or issue other special orders. Under these conditions, the building or use of fires, the use of power saws, heave equipment, ATVs and other motorized equipment may be prohibited on Forest Service land. All Federal, State and Local fire restrictions and regulations must be observed.

Report all fires to the Flagstaff Ranger District Office at: (928) 526-0866.

Cooperation in the reporting, prevention, and suppression of fires will be expected as specified in Part 2, section 10 of your Term Grazing Permit.

XIII. MOTOR VEHICLE USE/RESTRICTIONS

On May 1, 2012, the Coconino National Forest began implementation of the Travel Management Rule. The Travel Management Rule requires the designation of roads, trails, and areas open to motor vehicle use and these routes are identified on the Motor Vehicle Use Map. Implementation of the Travel Management Rule requires active management of all motorized use, including use related to permitted grazing activities. Motorized travel off the designated road system, including off-road access, by grazing permit holders will be based on the need to carry out required management practices necessary to comply with the terms and conditions of the Term Grazing Permit. Examples of required management practices include, but are not limited to: the repair and maintenance of structural range improvements; transport and placement of mineral and protein supplement; herding of livestock; and tending to sick or injured animals. Legitimate motorized use off the designated road system, including off-road access, for conducting activities required under the Term Grazing Permit will be authorized as follows:

1. **For motorized use with ATVs, UTVs, motorcycles, and full-size pickup or smaller vehicles (curb weight less than 8,000 pounds):** Grazing permittee must obtain a grazing allotment specific Restricted Road, Trail and Off-Road Administrative Use Permit. Motorized vehicle use off the designated road system will be limited to only that use which is necessary for compliance with the terms and conditions of the Term Grazing Permit.

2. **For motorized use with larger vehicles (curb weight greater than 8,000 pounds or larger than a full-size pickup) or any heavy equipment:** Grazing permittee must obtain a site

specific Restricted Road, Trail and Off-Road Administrative Use Permit. Motorized vehicle use off the designated road system will be limited to only that use which is necessary for compliance with the terms and conditions of the Term Grazing Permit.

To request site specific Restricted Road, Trail and Off-Road Administrative Use Permit, the grazing permittee must provide their Rangeland Management Specialist with the project details a minimum of 30 days prior to the proposed starting date for the permit.

3. A copy of the approved Restricted Road, Trail and Off-Road Administrative Use Permit must be with the vehicle operator while operating the vehicle off the designated road system.

4. A Permit Placard must be attached to all permitted vehicles while operating off the designated road system.

5. Off-Highway vehicles (ATVs, UTVs, and motorcycles) used for conducting activities required under the Term Grazing Permit must be licensed and permitted in accordance with Arizona State law.

6. The Motor Vehicle Use Map for the Coconino National Forest identifies certain roads as "Roads Open to Highway Legal Vehicles Only". These roads are open only to motor vehicles licensed under State law for general operation on all public roads within the state. Grazing permittees operating motor vehicles on these roads must comply with the legal requirements. Operating a motor vehicle off the designated road system to avoid the legal requirements of "Roads Open to Highway Legal Vehicles Only" is not permitted.

7. Grazing permittees will utilize motorized vehicles off the designated road system in a manner that will not result in damage to soil, watershed, vegetation, wildlife, heritage, or other forest resources.

8. Failure to comply with the conditions of the Restricted Road, Trail and Off-Road Administrative Use Permit may result in the immediate cancellation of the Restricted Road, Trail and Off-Road Administrative Use Permit and may result in the issuance of a Notice of Non-Compliance for violation of the terms and conditions of your Term Grazing Permit.

Grazing permittees that do not obtain a Restricted Road, Trail and Off-Road Administrative Use Permit are required to operate motorized vehicles only on the designated road system. Operating a motorized vehicle off the designated road system without a valid Restricted Road, Trail and Off-Road Administrative Use Permit will result in the immediate issuance of a Notice of Non-Compliance for violating the terms and conditions of your Term Grazing Permit (Part 2, Section 8a). A Notice of Non-Compliance may result in Term Grazing Permit suspension or cancellation proceedings.

XIV. PERMITTEE REVIEW

On April 2nd, 2015, [redacted] met with the permittee to discuss the 2015 Annual Operating Instructions. The permittee reviewed and agreed with these instructions.

If you have any questions or wish to discuss anything further, please feel free to contact [redacted]

Sincerely,

/s/ Debra Mollet
DEBRA L MOLLET
Deputy District Ranger

USDA United States Department of Agriculture | Forest Service | Flagstaff Ranger District

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File Code: 2210/2230
Date: May 13, 2015

This document serves as your summer 2015 Annual Operating Instructions (AOI) for the Windmill West Allotment.

I. AUTHORIZED LIVESTOCK NUMBERS/PERIOD OF USE

Your 2015 Authorized Use Request has been approved and you are authorized the following Livestock Numbers and Periods of Use:

374 Cattle (Adult)	March 1, 2015 to February 28, 2016
105 Cattle (Adult)	April 20, 2015 to February 28, 2016

You have requested non-use, resource protection, for:

86 Cattle (Adult)	April 20, 2015 to February 28, 2016

The 2015 authorized livestock numbers and the authorized period of use were determined based on referencing the Western Regional Climate Center 12-month Standardized Precipitation Index (Northeast Division, Arizona; Climate Division 2) and allotment inspections conducted prior to the grazing season.

Any increases in the 2015 authorized livestock numbers will require Forest Service approval.

II. GRAZING SCHEDULE

The following is the planned grazing sequence for the 2015 summer grazing period (June 1, 2015 to October 31, 2015). The pasture move dates shown are an estimate and may need to be amended based on actual conditions.

Main Cow Herd

Pasture	Planned Graze Period	Livestock Numbers
Winter Cabin	5/28-6/1	479
West Barney	6/2-6/29	479
East Barney	6/30-7/11	479
Lockwood	7/12-7/26	479
Fry Park East	7/27-8/20	479
Mill Park* 32 bulls moved to Horse Pasture	8/21-9/21	447
Roger's Lake	9/22-10/18	447
Metz	10/19-10/27	447

Bull Herd

Pasture	Planned Graze Period	Livestock Numbers
Horse Pasture	8/21-10/30	32

Rested Pastures for the Summer 2015 grazing season:

Fry Park West, Harding Point, Mexican Pocket

Changes from the grazing schedule will be made only with Forest Service approval. The planned use period in a pasture may be shortened or lengthened depending on forage availability and utilization in key areas. Livestock pasture moves will be completed within a ten (10) day period and livestock will be actively herded from one pasture to another. It is imperative that once a pasture move occurs, the livestock remain within the designated pasture. For various reasons (i.e. pasture gates left open, cattleguards not maintained, cut fences, etc.), cattle may stray from the designated pasture. Preventing livestock from grazing adjacent allotments or your allotment's rested pastures is the permittee's responsibility.

III. DROUGHT MANAGEMENT

Drought is an inevitable occurrence in the southwestern United States. Both the Forest Service and grazing permittees must plan for drought as a normal part of management. During periods of drought, the following management actions may be necessary:

- Stocking levels may need to be reduced. Reductions may be necessary prior to the permitted season of use and/or during the permitted season of use.
- Season of use may need to be shortened. Depending on the severity of the drought and the stocking level, a reduced grazing season may be necessary.
- Pasture use periods may need to be shortened. Pastures will not be regrazed during the same grazing season and this may ultimately result in an early exit from the allotment.
- Utilization and/or seasonal utilization (grazing intensity) standards may need to be reduced. Depending on the severity of the drought and the stocking level, reduced utilization standards will likely result in shortened pasture use periods and may ultimately result in an early exit from the allotment.
- Lack of livestock water, or poor distribution of livestock water, may result in reduced pasture/allotment use periods. Depending on forage conditions, this effect may be mitigated by hauling water to temporary livestock watering facilities.
- Allotment Management Plan direction for pastures requiring rest or deferment will be followed. Livestock use of planned rested pastures due to drought will not be authorized.

IV. GRAZING STRATEGY

Graze livestock in a deferred, rest-rotation grazing system for the six month summer grazing period. The management of livestock pasture moves in this allotment will continue to be based upon seasonal utilization guidelines and allowable forage utilization standards.

V. ALLOWABLE UTILIZATION STANDARDS

Upland and Non-Riparian Areas
The maximum allowable level of utilization on herbaceous and non-riparian woody vegetation is 35%. This allowable use level is the utilization level permitted by both livestock and wildlife. Utilization is defined as the proportion or degree of current year's forage production by weight that is consumed or destroyed by animals. It is a comparison of the amount of herbage left with the amount of herbage produced during the year. Utilization will be measured at the end of growing season when the total annual production can be accounted for and the effects of grazing in the whole management unit can be assessed.

Riparian Areas
Livestock utilization of woody vegetation in riparian areas may not exceed 20%.

VI. SEASONAL UTILIZATION

A light to moderate seasonal utilization guideline of up to 50% will be used within grazed pastures on the allotment. Seasonal utilization is defined as the amount of herbage removed through grazing or trampling during the grazing period. Seasonal utilization will be measured at the end of each pasture's livestock grazing period. The seasonal utilization guideline will be the primary factor in determining when livestock need to move to the next pasture; but other factors such as climatic conditions, opportunity for plant growth, and previous year's utilization level will also be considered. Annual reductions in the seasonal utilization guideline may be made based on resource conditions.

VII. RESOURCE MONITORING

Forage Utilization
Forage utilization within each pasture will be monitored and the allowable utilization standards (section IV) and the seasonal utilization guidelines (section V) will be enforced. This may result in earlier pasture moves than indicated in the Grazing Schedule (section II).

Numerous key areas for forage utilization monitoring have been identified for the Windmill West allotment. Key areas are defined as "…a portion of range, which because of its location, grazing or browsing value, and/or use, serve as an indicative sample of range conditions, trend, or degree of use seasonally. A key area guides the general management of the entire area of which it is a part." Key areas will generally have the following characteristics: they will be approximately 20 to 500 acres in size; they will be located on productive soils with slopes less than 15%; they will have a plant community that is representative of the larger area; and they will generally be located 0.25 to 1.0 miles away from livestock water sources.

The location of key areas for forage utilization monitoring, key vegetation species, and the specific application to your allotment should be discussed with your Rangeland Management Specialist. Permittee participation in utilization monitoring is encouraged.

Permittee Monitoring
The permittee will monitor and document the following items on the enclosed Actual Use and Forage Utilization Monitoring Form: 1) Actual Grazing Period; 2) Actual Livestock Numbers; and 3) Average Utilization of all forage species at two different time periods: Just prior to livestock

entering the pasture, and immediately after livestock leave the pasture. **It is very important that this monitoring be completed during the time frames specified, and that the Forage Utilization Monitoring Form is submitted to your Rangeland Management Specialist at the end of the 2015 grazing period.**

VIII. MITIGATION MEASURES FOR SPECIAL STATUS SPECIES

The following mitigation measures are required for threatened and endangered species on the summer portion of the Windmill West allotment.

Mexican Spotted Owl

Livestock grazing or livestock management activities will occur within Protected Activity Centers (PACs), but no human disturbance or construction activities associated with the livestock grazing operation will occur in PACs during the breeding season (March 1 through August 31).

Continue to monitor grazing use by cattle and wildlife in the ponderosa pine/gamble oak vegetation type and specifically within PACs.

In addition to the Salting and Protein Block practices identified in Section IX, the following additional guidelines are required within Mexican spotted owl habitat:
1) Do not place salt, mineral block, or supplements within riparian areas, mountain meadows, or non-riparian drainages in ponderosa pine.
2) Do not place salt, mineral block, or supplements within spotted owl PACs.

Follow best management practices associated with watershed protection.

Livestock distribution techniques such as salting and herding should be used.

IX. RANGE IMPROVEMENTS

Construction of New Structural Range Improvements
There are no new structural range improvements scheduled for your allotment this year.

Removal of Structural Range Improvements
There are no structural range improvements scheduled for removal this year.

Existing Structural Range Improvements
Structural range improvements assigned to you for maintenance are listed in your Term Grazing Permit and are also identified on the allotment map. These improvements are to be fully maintained annually to comply with permit requirements (Part 2, Section 8i). Any maintenance you perform must conform to the standards specified by your District Rangeland Management Specialist. The grazing permittee is responsible for all maintenance materials, supplies and equipment necessary to properly maintain all range structural improvements. The Forest will replace range structural improvement materials and/or supplies at the end of the improvement's life; when maintenance and repair is no longer feasible to keep the improvement properly maintained and functioning. **Please note that you must notify the District Rangeland Management Specialist at least 60 days prior to the beginning of any maintenance work that requires the use of heavy equipment.**

X. SALTING AND PROTEIN BLOCK PRACTICES

Proper salt and protein block location can be a good tool to aid in livestock distribution and they will be used in a manner to spread livestock utilization throughout the pasture. The following guidelines will be used when placing salt and protein blocks:

1. In general, salt and protein blocks should not be placed within ¼ mile of water.
2. Salt and protein blocks will not be placed in areas of depleted range, erosive soils, or sensitive plant or animal species. Areas of sensitive plant or animal species will be identified by your Rangeland Management Specialist and included in each year's AOI.
3. No more than three blocks (50 lb. blocks) of salt/protein will be placed at any location at any one time.
4. Salt/protein will not be placed at the same location twice during the same grazing season.
5. The use of portable salt/protein block containers is encouraged but not mandatory.

XI. PORTABLE WATER HAULING

Temporary sites for portable haul water may be needed and should be used as necessary to assist in livestock distribution. The following requirements will apply to portable haul water locations:

1. Coordinate with the District Rangeland Management Specialist to identify portable water haul locations for individual pastures prior to the grazing period.
2. To aid in livestock distribution, the portable water haul locations should generally be in areas of light forage utilization.
3. Portable water hauls will not be located in areas of depleted range, erosive soils, or sensitive plant or animal species.
4. Portable water haul locations will be moved when the desired forage utilization levels have been

reached.

5. Portable haul water storage tanks and troughs will be removed when livestock leave the pasture.

XII. FIRE

Please use caution during all ranch activities that could potentially start a fire. All ranch vehicles must be equipped with an axe, shovel, and water bucket. All open camp and branding fires require a ten foot fire ring cleared down to mineral soil.

During periods of critical fire danger the Forest Supervisor may declare a Red Flag Alert or issue other special orders. Under these conditions, the building or use of fires, the use of power saws, heave equipment, ATVs and other motorized equipment may be prohibited on Forest Service land. All Federal, State and Local fire restrictions and regulations must be observed.

Report all fires to the Flagstaff Ranger District Office at (928) 526-0866 or to the Coconino National Forest Supervisors Office at (928) 526-0600.

Cooperation in the reporting, prevention, and suppression of fires will be expected as specified in Part 2, section 10 of your Term Grazing Permit.

XIII. MOTOR VEHICLE USE/RESTRICTIONS

On May 1, 2012, the Coconino National Forest began implementation of the Travel Management Rule. The Travel Management Rule requires the designation of roads, trails, and areas open to motor vehicle use and these routes are identified on the Motor Vehicle Use Map. Implementation of the Travel Management Rule requires active management of all motorized use, including use related to permitted grazing activities. Motorized travel off the designated road system, including off-road access, by grazing permit holders will be based on the need to carry out required management practices necessary to comply with the terms and conditions of the Term Grazing Permit. Examples of required management practices include, but are not limited to: the repair and maintenance of structural range improvements; transport and placement of mineral and protein supplement; herding of livestock; and tending to sick or injured animals. Legitimate motorized use off the designated road system, including off-road access, for conducting activities required under the Term Grazing Permit will be authorized as follows:

1. **For motorized use with ATVs, UTVs, motorcycles, and full-size pickup or smaller vehicles (curb weight less than 8,000 pounds):** Grazing permittee must obtain a grazing allotment specific Restricted Road, Trail and Off-Road Administrative Use Permit. Motorized vehicle use off the designated road system will be limited to only that use which is necessary for compliance with the terms and conditions of the Term Grazing Permit.

2. **For motorized use with larger vehicles (curb weight greater than 8,000 pounds or larger than a full-size pickup) or any heavy equipment:** Grazing permittee must obtain a site specific Restricted Road, Trail and Off-Road Administrative Use Permit. Motorized vehicle use off the designated road system will be limited to only that use which is necessary for compliance with the terms and conditions of the Term Grazing Permit.

To request site specific Restricted Road, Trail and Off-Road Administrative Use Permit, the grazing permittee must provide their Rangeland Management Specialist with the project details a minimum of 30 days prior to the proposed starting date for the permit.

3. A copy of the approved Restricted Road, Trail and Off-Road Administrative Use Permit must be with the vehicle operator while operating the vehicle off the designated road system.

4. A Permit Placard must be attached to all permitted vehicles while operating off the designated road system.

5. Off-Highway vehicles (ATVs, UTVs, and motorcycles) used for conducting activities required under the Term Grazing Permit must be licensed and permitted in accordance with Arizona State law.

6. The Motor Vehicle Use Map for the Coconino National Forest identifies certain roads as "Roads Open to Highway Legal Vehicles Only". These roads are open only to motor vehicles licensed under State law for general operation on all public roads within the state. Grazing permittees operating motor vehicles on these roads must comply with the legal requirements. Operating a motor vehicle off the designated road system to avoid the legal requirements of "Roads Open to Highway Legal Vehicles Only" is not permitted.

7. Grazing permittees will utilize motorized vehicles off the designated road system in a manner that will not result in damage to soil, watershed, vegetation, wildlife, heritage, or other forest resources.

8. Failure to comply with the conditions of the Restricted Road, Trail and Off-Road Administrative Use Permit may result in the immediate cancellation of the Restricted Road, Trail and Off-Road Administrative Use Permit and may result in the issuance of a Notice of Non-Compliance for violation of the terms and conditions of your Term Grazing Permit.

> Grazing permittees that do not obtain a Restricted Road, Trail and Off-Road Administrative Use Permit are required to operate motorized vehicles only on the designated road system. Operating a motorized vehicle off the designated road system without a valid Restricted Road, Trail and Off-Road Administrative Use Permit will result in the immediate issuance of a Notice of Non-Compliance for violating the terms and conditions of your Term Grazing Permit (Part 2, Section 8a). A Notice of Non-Compliance may result in Term Grazing Permit suspension or cancellation proceedings.

XIV. PERMITTEE REVIEW

On May 12th, 2015 Mandy Roesch met with the ranch manager to develop and discuss the 2015 Summer Annual Operating Instructions for the Windmill West allotment. The manager has reviewed and agreed with these instructions.

If you have any questions or wish to discuss anything further, please feel free to contact [redacted] [redacted].

Sincerely,

/s/ Debra Mollet
DEBRA L MOLLET
Deputy District Ranger

VITA

Christopher “Kit” MacDonald was born in Dallas, Texas in 1963, son of John Robert MacDonald and Linda Evans. After graduating from Lake Highlands High School in Richardson, Texas in 1982, he worked as a machinist and computer technician while attending the University of Texas at Dallas. He completed his Bachelor of Science in Business Administration degree in 1995. Shortly thereafter, he entered the Graduate School of Stephen F. Austin State University where he received his Master of Science in Forestry in 1999. He has since worked as a Soil Scientist, mapping industrial forestlands throughout eastern Texas and southwestern Louisiana; as an Environmental Scientist with a large forestland owner and forest products manufacturer; and as a soils and watershed program manager with the U.S. Forest Service, Coconino and Kaibab National Forests. Mr. MacDonald is currently employed with the U.S. Forest Service as the Biophysical Resources and Planning Staff Officer of the George Washington and Jefferson National Forests in Roanoke, Virginia.

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This thesis was typed by Christopher MacDonald