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Effects of Ponderosa Pine Ecological Restoration on Forest Soils and Understory Vegetation in Northern Arizona

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By

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Effects of Ponderosa Pine Ecological Restoration in Forest Soils and Understory

Vegetation in Northern Arizona

By

ELENA MARIE THOMAS, Bachelor of Science in Environmental Science

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ABSTRACT

The human exclusion of wildfire and overgrazing by livestock since settlement have caused dramatic changes in ponderosa pine (*Pinus ponderosa* Dougl ex Laws) forest ecosystems. These changes include increased numbers of tree stems, reduced understory cover and diversity, and the introduction of invasive, non-native understory species. This study evaluated the coverage and species composition of understory vegetation present in the "cool-season" (late spring and early summer) in a ponderosa pine forest on grazed and ungrazed plots that had undergone restoration treatments on three different soil/geologic parent material types near Flagstaff, Arizona, twelve years after tree thinning and grazing exclosure treatments were applied. Several measured soil properties, such as soil respiration and temperature, were also evaluated in this study. Species richness of "cool-season" vegetation was influenced more by grazing practices than restoration treatments. Differences could be less or greater when vegetation that is active later in the season is measured. Vegetative cover was significantly influenced by restoration treatments (9.3% cover under open canopies and 6.5% under dense canopies), probably due to differences in competition for light and other resources (i.e. soil moisture and nutrients). Unlike finding by Abella et al. (2015), who studied "warm-season" vegetation, "cool-season" understory cover was not influenced by soil parent material type in this study, which might suggest that differences in understory cover due to soil properties are only seen shortly after restoration treatments are applied, or the time of year vegetation is evaluated may play a role in the differences seen. Soil respiration was highest on limestone soil parent material type (3.3 g C-CO$_2$ m^{-2} day^{-1}), and soil temperature was lowest under closed canopy treatments (15°C).

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INTRODUCTION

The purpose of ecological restoration is to enhance degraded ecosystems that have undergone human caused disturbances, such as heavily grazed grasslands, mine lands, agricultural conversion, or tree invasion of grasslands due to fire exclusion, and to remove invasive species. Restoration is the ability to establish or enhance an ecological system that will adapt to abiotic and biotic factors, as well as maintain and support a highly diverse and functional ecosystem (DellaSala, et al., 2003). Ecological restoration is greatly influenced by plant-soil interactions, for soils affect vegetative communities, while vegetation can alter soil properties (Eviner and Hawkes, 2008). Management practices used in ecological restoration result in increases in biodiversity, and can eventually lead to the rehabilitation and enhancement of key ecological functions (Eviner and Hawkes, 2008), such as soil erosion control, improved wildlife habitat, increases in nutrient cycling and energy pathways, and improved regulation of the water cycle (Sekercioglu, 2010).

Creating diverse vegetative communities is often a key goal in ecological restoration. Ecosystems with high species diversity are known to have higher ecological cognitive functions, such as increased water quality and availability, resilience to natural disturbances and decreased recovery time from these disturbances, higher rates of gas exchange between the atmosphere and soil, greater plant productivity, and higher soil fertility. With greater plant productivity on high diversity sites, more plant biomass will accumulate, and an organic matter horizon may be formed, thus protecting the mineral soil from climatic stresses (Cain, et al., 2014).

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There are many types of ecological restoration, such as forest, prairie, and wetland. Each type has different goals, and sites often vary greatly. A common problem for restoration efforts is being able to conduct restoration on a site-specific basis. What is known about ecological restoration is limited, and what might work on one site may not work on another.

Lockwood and Pimm (1999) conducted a meta-analysis on 87 restoration sites, and found that 19.5% of the sites were wholly unsuccessful in their restoration efforts, 48% only met some goals, and 13% met all of the goals with continuous management. However, Eviner and Hawkes (2008) attributed the restoration failures to a lack of knowledge on a site specific basis. Without the knowledge of soil conditions on a site, efforts at restoration of plant communities will founder. Understanding soil processes can lead to improved project success and selection of appropriate techniques and management. Soil conditions can limit or enhance performance and composition of plant communities, and in turn plant communities can have an impact on nearly all soil functions and structure aspects. Soil properties, with the exception of climate, are the single most important factor that governs ecological responses to restoration efforts (Selmants, et al., 2003).

Plants and microorganisms play a large role in the carbon (C) cycle, and to understand the C cycle at both a regional and local scale, the functions of terrestrial ecosystems and their role as C sinks and sources must be considered. It is understood that terrestrial ecosystems do in fact act as a large C sinks. There is three times the amount of C present in soils than in vegetation, and two times as much C than the atmosphere (Smith, 2004). To control net C losses from these C sinks, control processes that include the regenerative stage and topographical differences of forest landscapes, management practices, and the condition of forest nutrition must be implemented (Chen, et al., 2004). Restoration or management practices that simulate natural forest

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disturbances and preserve a continuous canopy cover will likely attain a high C storage (Lal, 2005). The intensity and frequency of a disturbance can greatly vary cumulative net ecosystem production, but the net ecosystem production of C can be enhanced with management practices (Chen, et al., 2004). Converting forest land to agricultural land can decrease the soil organic carbon (SOC) by about 20 to 50%. Agriculture, and other practices, reduces the biomass C and soil C, which are the main components of C storage. With the reduction of these components the rate of decomposition can increase, due to a change in temperature and moisture regimes. Afforestation of past agricultural land can lead to an increase in SOC stock. Trees with a cover crop, like *Fescue* spp., have a definable increase in SOC stock. SOC concentrations were found to be higher on northeastern facing slopes than on southwestern slopes in subalpine forests of the Olympic Mountains. This implies that landscape position, and in turn soil water regime, cation exchange capacity (CEC), texture, and aggregation affect the SOC stock (Lal, 2005). The duration of soils acting as a C sink depends on the maintenance of the management practices (Smith, 2004).

Through studies conducted on the effect soil moisture and soil temperature have on soil respiration rates, and the resulting CO_2 emissions to the atmosphere, Liu et al. (2014) concluded that with more frequent drying-wetting cycles greater CO_2 emissions to the atmosphere arise. As a result, forest ecosystems can act as a net producer of CO_2, instead of a C sink. By accounting for the potential losses, altering the canopy structure and the subsequent soil microclimate effects may allow for a more productive forest C budget. As C is closely tied to many ecosystem processes and is influenced by climate variation and human caused disturbances, determining the amount of ecosystem C present, in the form of CO_2, presence of litter, and SOC, an ecosystem's health can be assessed. Management practices will influence vegetative cover, soil chemistry,

forest and microclimate structures, litter quality and quantity, and root biomass, which can also influence the amount of C present in an ecosystem, as well as other nutrients that drive biogeochemical processes (Concilio, et al., 2006).

Ecological restoration practices of forest ecosystems, such as tree thinning and prescribed burning, are carried out to lower the intensity of surface fires and reduce the likelihood of stand replacing fires. Excessive accumulation of ground fuels, due largely in part to fire exclusion after Euro-American settlement, increases the potential occurrence of stand replacing fires. As a result, the southwestern United States (US) has had increased acreage burned by wildfires over the last three decades (Grady and Hart, 2006) compared to the late 1800s following Euro-American settlement. With fire exclusion, specifically in the western US, there has been an increase in tree encroachment on grass-dominated areas. As tree encroachment occurs, carbon has been accumulating within these terrestrial ecosystems (Kaye and Hart, 1998). Fire exclusion has also led to the establishment of less fire-tolerant species, which serve as fuel, increasing the probability of stand replacing fires (Gundale, et al., 2005). However, accumulation of ground fuels, such as leaf litter and coarse woody debris, create an insulating effect that can keep the temperature of soils cooler during warmer periods, providing improved soil moisture during the summer periods, and providing organic substrates for microorganisms such as fungi.

Before Euro-American settlement, ponderosa pine (*Pinus ponderosa* Dougl. Ex Laws) forests were heterogenic in nature, and historically maintained by natural or Native American caused, frequent, low-intensity surface fires that occurred every 2-20 years (Hart, et al., 2006). However, with Euro-American settlement, practices such as fire exclusion, grazing, logging, large predator hunting, and introduction of exotic species have led to the current dense, young-pine dominated, and largely homogenous, low biodiversity ponderosa pine forests of today (Allen, et al., 2002).

The lack of fire may change the composition and productivity of the ecosystem; for example, fire provides mineralized nitrogen in otherwise nitrogen limited ecosystems, such as ponderosa pine forests (Gundale, et al., 2005). Fire exclusion has also led to the establishment of less fire-tolerant species, such as white fir (*Abies concolor* Lindl. Ex Hildebr), juniper (*Juniperus* L. spp.), and Douglas-fir (*Pseudotsuga menziesii* Mirb. Franco), in ponderosa pine communities across the western US. The establishment of these species has changed understory conditions, producing thick layers of slowly decomposing conifer needles on the forest floor, lowering percent cover of grasses and forbs, and decreasing species diversity (Allen, et al., 2002). Decreases in understory vegetation may alter hydrologic processes, such as base flows and peak flows, which could be detrimental to forested ecosystems during periods of drought. Thus, ecological restoration practices for ponderosa pine forests typically involve thinning trees, re-introducing fire, or a combination of the two to enhance the ability of the ecosystem to withstand natural disturbances, such as drought, wildfires, and insects.

Grazing is a controversial management practice, especially in the western US, and is considered by some to adversely affect the environment and plant-soil relations by altering functions of ecosystems (Piñero, et al., 2010). Plant and soil relations are an important aspect of restoration practices and must be considered. There are positive and negative aspects to plant-soil interactions, such as planting specific plants that can augment poor soil conditions, or an invasive exotic plant entering an ecosystem and altering soil conditions (Eviner and Hawkes, 2008). Grazing is thought to be a major contributor for introducing invasive exotic species to an area, as well as decreasing native grass populations due to competition and selective grazing. Studies that include a grazing exclosure are useful in determining the effects of grazing on vegetation communities (Bakker and Moore, 2007).

Few studies have fully elucidated what the active target of restoration is, in terms of plant communities and densities, what restoration management techniques are needed to achieve the desired outcomes, and the time-frame needed to achieve the restoration goals, especially on different sites. To determine variation among sites, such as the effect of microclimate, different soil parent materials, and other ecological factors, long-term studies are necessary (Bakker and Moore, 2007). This study will examine the effects that restoration treatments have on forest soils within ponderosa pine forest ecosystems. Soil chemical properties, soil respiration, and the amount of carbon accumulating on the forest floor will be determined to attempt to evaluate the effects of ecological restoration on soils of sites undergoing restoration. In addition to soil response, the response of understory forest vegetation to restoration treatments will be evaluated. Both soil and vegetation response will be measured on a study of ponderosa pine ecosystem restoration started in 2003 (Abella, et al., 2015).

Abella et al. (2015) analyzed understory vegetative response to ecological restoration of a ponderosa pine ecosystem near Flagstaff, Arizona across three different soil parent material types, limestone, basalt, and benmoreite. Ecological restoration treatments included thinning from below, thinning from below plus a simulated smoke treatment, closed canopy, and open canopy. In addition, a grazing aspect was analyzed, in which exclosures were erected to keep out ungulate and livestock grazers, to determine the effect grazing has on herbaceous plant communities and the underlying soils. Vegetation analysis was done pre-treatment in 2003 and post-treatment in 2006 and 2008.

This study repeated understory vegetation measurements and a few select soil property measurements, which were obtained 12 years after treatments were applied in the original study. By conducting vegetation surveys on these in-place plots and analyzing soil properties, long-term

responses to restoration practices in these forested ecosystems can be assessed, increasing

knowledge about ecological restoration to help develop better management practices.

OBJECTIVES

The principle goals of this project were to evaluate the response of select soil chemical and biological properties to common ecological forest restoration treatments, and to analyze the herbaceous vegetation response under different forest ecosystem restoration practices over a 12 year time frame. More specifically, the objectives of this study were to determine the effect of restoration treatments on:

1. the select soil properties of pH, total nitrogen and carbon, and extractable phosphorous, potassium, calcium, magnesium, sodium, sulfur, and boron;

2. soil respiration, temperature, and moisture regimes;

3. herbaceous plant communities.

LITERATURE REVIEW

Carbon Sinks in Forest Ecosystems

Overstory Tree Biomass, Above- and Belowground

There are many pathways through which carbon (C) in a forest ecosystem can pass. It can be

utilized by heterotrophic decomposer organisms, provide fuel for fire, or can be removed from

tree harvesting or deforestation. C is fixed during photosynthesis, and is assimilated in various

plant tissues, a portion of which is stored in plant structures which serve as a C sink. Fuel loads

have increased over time due to fire exclusion, and as a result large amounts of C are stored in

understory vegetation, organic soil horizons, and in soil organic matter (Selmants, et al., 2003).

Primary production rates govern the amount of chemical energy that is produced from the

conversion of solar energy. Primary production is carried out by photoautotrophs, such as plants,

and is utilized by heterotrophs, such as decomposer organisms. Gross primary production (GPP)

in most terrestrial ecosystems is the amount of C fixed during photosynthesis, and is influenced

by climate and leaf area index. Plant respiration releases large amounts of the fixed C. Net

primary production (NPP) is the amount of C left over after C is lost through plant respiration,

from the amount of C produced during photosynthesis (GPP). NPP can also be described as the

total net input of C to an ecosystem. After plant respiration has been conducted, the remaining C

can be used for growth, defense against predators and climate stresses, and internal storage. C

may also be allocated to different areas of the plant, such as roots, leaves, or stems. The amount

of nutrients and water available affects where C is allocated in the plant. Ponderosa pine allocate

more available C to leaf growth in response to increased competition for light in dense stands, which are commonly found due to fire exclusion. However, on nutrient poor sites ponderosa pine can allocate over 50% C to roots to achieve greater surface area for nutrient uptake.

Mid- and Understory Vegetation Biomass, Above- and Belowground

Davis et al. (2010) found root biomass for all vegetation within an ecosystem was a small component of the total ecosystem C stock, with six percent in mineral soils and one percent in organic soils, and found the above- and belowground vegetation C pools were at about 38 to 42% of C distribution through the ecosystem. According to Birdsey (1992), who considered all key forest ecosystem components in his models, above- and belowground vegetation account for about 32% of the total C stock in an average US forest system.

Organic Surface Soil Horizons

The amount of soil organic matter (SOM) present in the forest floor (O horizons) will vary by ecosystem, but the amount of soil organic carbon (SOC) present, the utilization of SOM and the ensuing release of CO_2 by microorganisms through respiration, and C cycling rates depend on temperature, moisture, litter type, and canopy cover. Southwestern ponderosa pine forests have characteristically low rates of C-cycling, due to low quality litter (high lignin to N ratio), low annual precipitation, and persistent dry periods during the summer. Hart et al. (2006) found in an unmanaged ponderosa pine stand, the mean residence time, which includes forest floor content and litter-fall input, of C on the forest floor was between 26 and 50 years, which are some of the lowest decomposition rates reported for forest ecosystems worldwide. Factors, such as poor litter

quality, low annual precipitation, and persistent dry periods, limit the rate of decomposition of plant litter by decomposer organisms.

Kerns et al. (2003) measured differences in soil properties under various ponderosa pine-bunchgrass patches: grassy openings, old-growth stands, and young, dense stands. Old-growth stands had measureable O horizons compared to grassy openings, and had significantly thicker O horizons compared to young, dense pine stands. However, there was no significant difference for old-growth stands in A horizon thickness compared to grassland and young, dense pine plots. C concentrations were similar among patches, but, generally, were lower for grassland plots compared to old-growth plots, which may be attributed to higher organic C cycling rates under grassland vegetation. In addition, C concentrations, in both O and A horizons, for old-growth plots were higher than younger pine plots. It was also found that A horizon C concentrations were positively correlated with O horizon thickness. Kerns et al. (2003) hypothesized that parent material and soil texture may have had an effect on accumulation of vegetation derived organic matter.

Soil Organic Carbon in Mineral Soil

Total organic carbon (TOC) is the amount of C found in soil organic matter, while soil organic carbon (SOC) is the C component of SOM that is utilized by soil microorganisms. The amount of C in SOM is about 58%, by weight (Edwards, et al., 1999). SOM substrates promote soil aggregates and soil structure, enhancing gas and water exchange through the soil, and making soil less susceptible to erosion.

The amount of C stored within an ecosystem can vary by type of organic matter, vegetation, various soil properties, and site history. Davis et al. (2010) found total ecosystem C content tended to be highest in very poorly drained sites compared to excessively drained sites by a factor of three in New England temperate hardwood forests. It was theorized that variations in soil C pools are related to organic substrate quality, litter decomposition rates, and soil drainage within an ecosystem, and not by respiration, biomass, or litter inputs. Typically, grassland soils, compared to coniferous and deciduous forest soils, have thicker A horizons, lower carbon nitrogen ratios (C:N), and higher accumulation of mineral soil organic matter. These differences can be attributed to higher organic matter deposition rates and higher quality substrates that are characteristic of grass ecosystems. In comparison, forest organic substrates, particularly including those of conifer dominated ecosystems, including ponderosa pine, have higher lignin:N and C:N ratios compared to grassland systems (Kerns, et al., 2003).

Ecological Restoration Effects on Soil Carbon

Grady and Hart (2006) did not find any significant differences among restoration treatments for total soil C in a ponderosa pine-bunchgrass ecosystem, nor did the net C mineralization rates differ between unthinned and thinned treatments. Net C mineralization rates are determined by the quality of organic C substrates available to heterotrophic microorganisms (Grady and Hart, 2006), and coniferous litter is of low quality (Kaye and Hart, 1998). Gundale et al. (2005) found no significant difference for the C:N ratio in the mineral soil, but they did find that the C:N of organic horizons was significantly different between treatment years. Total C in the mineral soil and O horizons was not statistically different among treatments, which included thinning,

burning, and thinning + burning. SOC was also found not to be significantly different following prescribed fire in ponderosa pine forests of central Oregon (Busse and Riegel, 2005.)

Carbon pools for unmanaged stands in a ponderosa pine forest were found to be 19% higher than restored stands. In addition, litterfall C inputs for restored stands were 45% of the unmanaged stands litterfall C input, possibly due to the thinning and burning carried out in the restored treatment (Hart, et al., 2006).

Grazing Effects on Soil Carbon

Grazing can also affect the amount of NPP present throughout an ecosystem. In areas of heavy grazing, plants may allocate increased amounts of C to produce tannins and terpenes, compounds that discourage grazing (Cain, et al., 2014). Large amounts of C are stored in aboveground biomass, which may cause the NPP to decrease under higher levels of grazing conditions. Grazing also can alter the species composition of aboveground vegetation, thus indirectly changing NPP (Piñeiro, et al., 2010).

Schultz et al. (2011) studied the effects of grazing exclusion on phytomass (live and dead plant tissue) accumulation in Australia. Excluding grazers resulted in a greater accumulation of phytomass, thus a greater amount of C to be incorporated into the soil. In addition, the microclimate and water, light, and nutrient availability are affected by grazing, and will impact cycling and storage rates of C and other essential nutrients. Gao et al. (2007) compared these variations for nine years under different grazing intensities, and found that grazing affected the amount and quality of stored soil C. They also found that high intensity grazing increased SOC significantly in the top 10 cm of the soil profile, when compared to the light or moderate grazing

treatments. The higher SOC under heavy grazing may be attributed to an increase in root biomass as plants allocate less C towards aboveground biomass, which provide increased inputs of organic matter to the soil when roots die and decay. However, the aboveground C decreased under heavy grazing, which is not desirable for livestock production, and over time the total soil C stored declined.

Chen et al. (2015) evaluated grazing effects on soil C pools in China, and found soil C to be highest under moderate grazing, due to high production and turnover of roots. Areas that underwent a resting phase had less accumulated soil C compared to moderate grazing, but root biomass in these areas was larger. Heavy impact grazing treatments yielded unfavorable results in above- and belowground C allocation. Grasslands dominated by C3-grasses and exposed to high intensity grazing were low in soil C. However, C4-dominated grasslands were found to have higher soil C stocks. Negative effects resulted from moderate grazing intensities in mixed C3-C4 grasslands, but positive effects resulted from light and heavy grazing intensities in these mixed grasslands.

Research has led to mixed results regarding the effect of grazing on SOC. For example, Smoliak et al. (1972) found a significant increase in SOC under heavy grazing pressures compared to no grazing. Piñeiro et al. (2010) conducted a meta-analysis and found SOC to decrease under heavy grazing over a 50-year period (Fuhlendorf, et al., 2002). In addition, the meta-analysis included a 75-year study that found no significant difference in SOC under grazing (Frank, et al., 1995), a 12-year study that found SOC to increase under grazing (Schuman, et al., 1999), and a 56 year study that found SOC to increase under grazing (Reeder, et al., 2004), when grazing exclosures were utilized. Finally, an 18-year study examining grazing intensity found SOC to decrease (Xie and Wittig, 2004). Grazing, which reduces litter inputs, can indirectly

affect soil temperature and evaporation rates; with an increase in these variables decomposition rates increase as well (Piñeiro, et al., 2010). However, the effects soil temperature and soil evaporation rates have on decomposition rates depends on the initial soil moisture content. Reductions in SOC can result from a decrease in litter presence, which triggers microclimate changes, and in turn increases soil organic matter decomposition. Changes in SOC stocks also results in changes in soil organic nitrogen content and cycling.

Effects of Ecological Restoration on Soil Nitrogen

Nitrogen (N) is a key element in terrestrial ecosystems, as it allows organisms to produce organic compounds that contain N, such as amino acids and proteins, and metabolize these compounds for energy and growth. Because of its necessity for living organisms, particularly the chlorophyll in plants that carry out photosynthesis, it is commonly a limiting factor in primary production. The amount of N present in soil and vegetative sinks and the rate of N cycling varies among ecosystem types (Butterbach-Bahl, et al., 2011). The majority of total N in forest soils exists as organic N in organic matter, which is an unusable form for plants. Total N content increases with an increase in organic matter content, therefore total N and C content are strongly correlated.

Biological Nitrogen Fixation

Biological nitrogen fixation, the conversion of atmospheric nitrogen (N_2) to ammonia (NH_3) or ammonium (NH_4), is carried out by a specialized group of prokaryotes on or within the soil,

and includes organisms such as cyanobacteria, free-living bacteria (*Azotobacter* and *Clostridium*), associative bacteria that have relationships with plants (*Azospirillium*), and bacteria that form symbiotic relationships with legumes (*Rhizobium*) and other plants (*Frankia*). With these organisms reducing atmospheric nitrogen and mineralizing organic nitrogen compounds, plants can use the resulting inorganic nitrogen to create amino acids, nucleic acids, and chlorophyll (Wagner, 2011). N fixation is carried out by many legumes, in conjunction with Rhizobia bacteria, including a forest-grassland ecosystem like that of this project where *Lupinus* spp. occur.

Nitrogen Mineralization

In order for plants to utilize N in organic matter, it must be converted through mineralization by decomposer organisms to plant available inorganic N, such as ammonium (NH_4) or nitrate (NO_3). Heterotrophic microorganisms can carry out the process of N mineralization, or ammonification, of organic N (Butterbach-Bahl, et al., 2011), which releases N as ammonia (NH_3). If hydrogen ions (H^+) are present, NH_4 will form, which is a stable plant available form of N that can be held on soil cation exchange sites. The gross ammonification rates and available C in the soil, will influence the amount of NH_4 that is assimilated by soil microbes. Mineralization of N is affected by soil moisture and climate, and will typically be higher during periods of high moisture and warm temperatures. Soils that are saturated do not follow this same trend. Mineralization can also be influenced by the amount of other plant available nutrients (Vesterdal, et al., 1995), the soil microbial communities, and the amount of fine fuels consumed by fire (Gundale, et al., 2005). Mineralization rates increase due to reduced competition for nutrients and water between roots and saprophytic microorganisms (Vesterdal, et al., 1995). Additionally,

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mineralization of N has been known to increase with increased temperatures up to 30°C, and will vary among eco-regions and topography. Shallower A-horizons and variable soil moisture may make the potential mineralization N present inconsistent among sites (Zhang, et al., 2002). Mineralization rates can also be affected by soil physical properties, such as bulk density, texture, and porosity, which will influence gas exchange and water movement, and in turn will affect the amount of nutrients available (Selmants, et al., 2003). Plants can bypass utilization of microbial N mineralization in N-poor and cold ecosystems by taking up amino acids and organic monomers present in the soil, and the uptake of N by plants may be influenced by soil pH, light, temperature, and ammonium, nitrate, and organic N concentrations within the soil (Butterbach-Bahl, et al., 2011).

<u>Nitrification</u>

The oxidation of NH_4 to nitrite (NO_2), and subsequently to NO_3 is nitrification, and can be performed by either autotrophic or heterotrophic organisms. *Nitrosomonas* is an example of a chemoautotrophic ammonia oxidizer bacterium that converts NH_4 to NO_2, and *Nitrobacter*, also a chemoautotrophic bacterium, is a nitrite-oxidizer that converts NO_2 to NO_3. The chemolithotrophic bacterium *Nitrosococcus* is another organism that can oxidize ammonia. Nitrification is influenced by multiple soil factors, such as soil temperature, soil water content, thus soil texture, and soil pH. The optimum pH range for nitrifying organisms is between 5.5 and 6.5. Due to the energy required for bacteria and fungi to utilize NO_3, NH_4 is more widely used by microorganisms in the soil. Nitrification is governed by the same factors as N mineralization, in

that the rate will increase with an increase in temperature until 30°C, but will be limited by the amount of NH_4 in the soil (Butterbach-Bahl, et al., 2011).

Denitrification

Denitrification is the subsequent loss of gaseous N in the forms of nitrous oxide (NO_x) and elemental nitrogen (N_2) from the reduction of NO_3 to NO_2, and then NO_2 to nitric oxide (NO) by microbes (Butterbach-Bahl, et al., 2011). Bacteria and some fungi and archaea are able to conduct denitrification, and are usually anaerobic organisms. However, there are organisms that can perform aerobic denitrification. Generally, denitrification will occur in water-logged soils and soils with high C content. Periods of high soil temperatures and saturated soil conditions can intensify denitrification occurring in the soil.

Carbon and Nitrogen Dynamics

Environmental conditions, such as climate, vegetation, management practices, and soil properties can have a pronounced influence on soil N dynamics. Temperature and moisture are climate factors that can influence N cycling; for example mineralization and nitrification will increase if the soil moisture increases, as long as saturation is not reached. The structure of the forest canopy, the form of leaf litter, root distribution, and litter quality will play a role in N mineralization rates within a forested ecosystem. These parameters can affect the amount of gaseous losses of N_2, in the forms of NO or N_2O, that is released due to conditions of soil aeration, soil moisture, and the nature of organic substrates. Management activities such as livestock grazing or timber harvesting and thinning operations, can alter various ecosystem

factors such as litter quality, aeration, soil moisture content, soil temperature, and microbial communities present within soils. These ecosystem factors will subsequently influence ecosystem C and N dynamics.

Soil properties such as texture, which influences moisture retention, can play a fundamental role in N cycling. Fine-textured soils tend to hold more water for longer periods of time than coarse-textured soils. Therefore, large precipitation events can cause anaerobic conditions to more readily develop in fine textured soils. Anaerobic conditions reduce N availability through reduced mineralization and N losses due to denitrification. The high porosity and low cation exchange capacity of coarse-textured soils can increase nitrate leaching, reducing plant available N in soils. Gaseous losses of N from denitrification and leaching tend to be higher with low C:N ratios (less than 20:25) in the mineral soil or forest floor. The C:N ratio represents the relative amounts of C and N present within organic residues. Due to the effects of the C:N ratio, management practices, such as thinning and grazing, can influence the amount and quality of organic matter present, thus affecting the amount of C to N.

The ecosystem factors that play a role in N cycling also influence one another. Traits specific to different species of plants, such as biomass production, litter persistence, litter chemistry and structure, can influence the C:N ratio and soil organic matter content (Eviner and Hawkes, 2008). Soil organic matter C:N is influenced by appropriation of nutrients by plant roots, the quantity and quality of plant exuded substrates within the soil, and the effect a specific plant species has on the soil microclimate and soil microbes. While a plant can enhance one parameter in the soil, it can also decrease another. Eviner and Hawkes (2008) provide the example of a plant species subject to rapid decomposition that cycles N and other nutrients quickly, but with enhanced decomposition rates there is a decrease in the litter layer present, which will decrease soil

temperature during net cooling periods, due to more rapid reradiation of heat energy from the soil, and increased soil temperature during periods of net warming. The amount of litter a plant produces, and its persistence in the ecosystem, can also be altered by numerous coinciding environmental and soil conditions, such as soil moisture, temperature, season, plant age, and microenvironment. For example, the N cycling rate will decrease with a decrease in soil moisture.

Under southwestern ponderosa pine forests, it has been found that grassy openings contained most of the N in the top 15 cm of the soil profile, mainly tied up in soil organic matter and roots. However, under dense tree canopies most of the N present was in unavailable, organic forms in the O horizons, such as substrates like needles and branches (Selmants, et al., 2003). Grady and Hart (2006) found increases in N cycling rates following thinning only and thinning plus prescribed burning, due to increased soil temperature. However, they found that microbial N decreased in thinning treatments, and was dependent upon rate of recovery of vegetative inputs. Gundale et al. (2005) found no significant differences for each treatment in total C, N, and C:N ratio in the mineral soil for any of the years tested, strengthening the hypothesis that time is a factor in N cycling changes. Kaye and Hart (1998) also found gross and net N transformations, ammonification, nitrification, and mineralization, increased with thinning and complete restoration (prescribed burning and tree removal) treatments. This suggests that N cycling rates declined with pine encroachment in grassy openings. Due to higher soil N mineralization, nitrification, and respiration rates found under grassy openings by Kaye and Hart (1998), and the fact that grassy openings have higher solar radiation, soil temperature, and soil moisture during the growing season, compared to dense, younger pine and old-growth pine stands, Kerns et al.

(2003) determined low N accumulations in this situation could be a result of high organic carbon cycling rates.

Smoliak et al. (1972) found significantly higher C:N ratios under different grazing intensity treatments compared to a no grazing treatment over a 19 year period. Johnson and Matchett (2001) found grazing to increase N cycling rates and N availability in grassland soils. Clapperton et al. (2002) were able to see the effects of disturbances and assess the response of litter abundance and nutrient cycling over a longer-term period (50 years), and discovered an increase in percent total N and a decrease in mineralizable N and soil organic matter with an increase in grazing. They found the abundance of microarthropods that assist in decomposition, and thus nutrient cycling, increased under no- to little grazing, as vegetation was given time to fully respond to management practices.

Effects of Ecological Restoration on Microbial Respiration

There are many types of soil microflora, and each have specific roles within the soil. Actinomycetes break down organic compounds that are more resistant to decomposition, N-fixing bacteria fix N to become plant available, and mycorrhizal fungi assist plants in water and phosphorous (P) uptake through fungi-root mycorrhizal associations. Soil microflora activity and colony size generally decrease with a decrease in C inputs caused by management practices, such as grazing. As a result, microbial activity is closely related to C inputs from primary production in southwestern ponderosa pine forests (Grady and Hart, 2006).

Soil respiration in temperate forests seasonally changes with soil temperature and moisture. With a shallow soil wetting event, the stored CO_2 can be displaced and can alter the CO_2 efflux of the soil (Liu, et al., 2014). CO_2 efflux is the amount of CO_2 produced from root respiration, and the amount released during decomposition of litter (Epron, et al., 2001). It has been hypothesized that soils at or near field capacity are at the optimal point for high soil respiration rates. At field capacity the macropores will be filled mostly with air, which will facilitate O_2 diffusion, and the micropores will be filled with water, which will promote solubilization of C substrates. This hypothesis is based on the principle that in a very wet soil O_2 diffusion is limited through the pore spaces, and in very dry soils organic-C substrates are not as soluble. Heterotrophic organisms need both C substrates to produce energy and oxygen for respiration (Davidson, et al., 2000). At about 60% or higher of the water holding capacity, the rate of soil respiration will start to decline due to reduced gas exchange (Liu, et al., 2014).

Soil water retention, which is effected by soil texture, can vary greatly. When approximately 80 to 90% of the soil pore space is filled with water, gas exchange through the soil profile is retarded, and it is believed that microbial respiration will cease when only 20% of the soil pores are filled with air (Brady and Weil, 2002). Soil texture also has an influence on soil porosity, which can strongly affect the diffusion of O_2 and CO_2 through the soil. For example, large, interconnecting pores will allow water and air to move through the soil profile, while smaller pores will retain water that restricts O_2 and CO_2 diffusion rates (Bouma and Bryla, 2000). An increase in soil bulk density (i.e. compaction) also reduces gas exchange through soil pores, and may result in a decrease in soil respiration rates. However, the pore volume within a soil, as well as litter, can each hold up to one day's worth of CO_2 production.

Bouma and Bryla (2000) used clean, autoclaved soil at constant temperatures to test the effect soil moisture and texture have on root and soil respiration rates, and to determine the extent to which roots contribute to total soil CO_2 efflux. Lemon seeds were germinated and placed into the autoclaved soil, with daily watering and weekly fertilizing. Respiration rates between the soil texture samples were similar before a wetting cycle, and all treatments rapidly declined soon after wetting, for the soil was saturated. However, respiration rates returned to pre-wetting levels for the sandy soils two times faster than the finer-textured soils. With continuous drying and wetting cycles there were significant differences between water content of the soils, and subsequently CO_2 concentrations within the soil. Soil type influences the rate at which CO_2 concentrations within the soil will recover after a wetting cycle, as well as the rate of water use by plants, thus soil CO_2 concentrations vary with time. Overall, it was found that finer-textured soils had higher soil CO_2 concentrations due to a higher water holding capacity. These results were found in a laboratory setting, and do not take into account the role biopores (earthworm burrows or root channels) play in altering the soil structure and soil porosity. Biopores allow for gas diffusion, and could have a great impact on soil CO_2 efflux.

Davis et al. (2010) hypothesized that a poorly drained soils will have higher accumulated organic matter, and therefore, higher respiration rates during a drought period, due to higher O_2 concentrations and decomposition rates that would not be inhibited by water saturation. In contrast, well-drained soils, that would normally have higher respiration rates than poorly-drained soils, would have decreased rates due to lack of soil moisture during a drought. It was found that soil type had a slightly significant effect on soil respiration, but by taking into account the soil type and sampling date interaction there was a larger significant difference for soil respiration rates. However, respiration was more variable over time than across sites. During the course of

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the study, a summer drought occurred during the bulk of the growing season, and following the first fall rain event CO_2 efflux was greater in the excessively drained and well-drained sites. Davidson et al. (2000) also found that during the beginning of the rainy season soil respiration rates were high, possibly caused by "wet-up effects", which caused pulses of microbial activity and respiration.

Restoration treatments that increase temperature, raise soil evaporation rates, cause soil compaction and loss of vegetation patches, and alter the soil microclimate, may affect soil respiration (Ma et al., 2005). In turn, microbial communities can influence soil structure, plant growth and competition, and most biogeochemical transformations within the soil, but also N cycling, CO_2 production, the formation and decomposition of soil organic matter, and plant success (Eviner and Hawkes, 2008). Finally, drainage class may play an important role during periods of stress resulting from climate changes, and with increased frequency and duration of climate changes poorly drained soils may, in-time, have higher soil respiration rates.

Tree Thinning Effects on Soil Respiration

In a ponderosa pine forest, Gundale et al. (2005) determined that tree thinning resulted in greater soil respiration rates. Grady and Hart (2006) found that soil net CO_2 efflux decreased following thinning only and thinning plus prescribed burning treatments, due to an increase in soil temperature. They also found that biological activity, and subsequently microbial N, decreased in thinning treatments, and concluded that moderately reducing vegetation density within stands provided this result and additionally decreased plant available N.

Kaye and Hart (1998) measured growing season soil respiration rates under complete treatment type, which included a prescribed burn along with tree removal, partial treatment type, which included removal of aboveground tree biomass that was post-settlement, and a control treatment type, which had no tree removal. They found that the partial restoration treatment had the lowest soil respiration rates the first year after treatment. However, in the second year the control treatment was lower in growing season soil respiration than the other treatments. The complete restoration treatment was significantly higher than the other treatments throughout the study, and exhibited a higher increase in soil respiration at the start of the growing season. The complete restoration treatment did not have the lowest respiration rates in the study, but did have the lowest concentrations of forest floor N, P, and organic matter. Because of this, forest floor N, P, and organic matter are not major factors in controlling respiration rates. Net soil CO_2 efflux increased in the short- and long-term thinning- and complete restoration treatments, and the gross and net N transformations increased. Due to the higher respiration rates under the grassy opening (complete restoration treatment), it appears that microbial activity is higher under bunchgrass ecosystems than forested ecosystems. With the increase in these microbial processes, it was concluded that temperature and substrate quality were the driving forces. Higher substrate quality included a reduction of low quality pine litter and an increase in high quality herbaceous litter. The most significant factors that influenced soil respiration in the study were soil temperature, with seasonal patterns acting as the driver for variation, and soil moisture affecting soil respiration interannualy. There has not been conclusive evidence that tree removal has any significant, specific effect on soil respiration rates, for it can either increase, decrease, or stay the same. The variation in respiration rates, due to soil water content and temperature, may depend on a site's rate of successional change. If succession is slow, C inputs will be low; thus, root

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respiration and microbial respiration will be low. Soil respiration differences among the canopy or treatment types were attributed to changes in soil microclimate, soil microbial communities, and soil organic substrates that were influenced by plant and soil interactions. Grassy openings are known to produce higher net N transformation rates, larger microbial biomass production, a greater bacteria to fungi ratio, and greater soil respiration rates (Kaye and Hart, 1998).

Within mixed conifer forests, soil respiration rates have been found to vary among patch type, soil moisture content, and soil temperature (Concilio, et al., 2006; Ma, et al., 2005). Concilio et al. (2006) found closed canopy patches, which underwent an unburned-overstory thinned treatment, had an increase in soil respiration, and open canopy patches, that had undergone the same treatment, decreased in soil respiration rates each year. There were significant interannual differences among patch types for the unburned-unthinned treatment. It was found that temperature and litter depth were drivers for variation in soil respiration in post-treatment years. Soil respiration fluctuated by year, patch type, and treatment type, and the intensity of the treatment was related to the soil respiration rate response. The speed at which soil respiration rates recovered was dependent on the vegetation, type of disturbance, and patch type. Soil respiration under the thinned-open canopies was significantly less than under the thinned-closed canopies.

In a study conducted in the same area by Ma et al. (2005), it was also found that soil respiration rates varied among patch types, with the lowest soil respiration rates occurring in the open canopy treatments. Closed patches generated more CO_2 than open patches, possibly due in part to more fine roots present in the closed system. They also found that litter and coarse woody debris could play a role in the soil respiration rates. It was found that respiration rates decreased after a rain event, regardless of soil temperature, and were highest from noon to late afternoon in

a mesic system. After a rain event soil moisture is increased, so soil aeration and oxygen (O_2) availability will decrease, effectively reducing CO_2 efflux. However, this effect will depend on the duration and intensity of the storm, as well as the initial soil moisture content before the storm, and soil texture and porosity. Generally, CO_2 efflux concentrations increase from high root and microorganism respiration when there is higher soil moisture, effectively increasing the amount of soil CO_2 present, but it has been suggested by some authors that soil CO_2 might not be immediately released into the atmosphere.

The nature of leaf litter biomass may play a role in soil respiration rates (Liu, et al., 2014). Large and small canopy coniferous and deciduous forests were studied for variations in soil water content and soil respiration. Coniferous litter is more difficult to decompose and seldom incorporates into the mineral soil, compared to deciduous tree species litter. Therefore, the slow to decompose coniferous litter creates a cover over the soil that intercepts more water, reducing the amount of water reaching the mineral soil. With soil moisture lower under a coniferous canopy, fine root respiration within the soil may comprise a larger portion of total soil respiration. Liu et al. (2014) also found that soil respiration was significantly different among canopy size classes in dry conditions, but not in wet conditions. The small canopies produced higher soil temperatures during the dry season, but lower soil temperatures during the wet season. This was likely due to the shading effect that the larger canopies produced. The mean soil volumetric water content (VWC) of the smaller canopy was 17-28% lower than that of the large canopy. When there was a drying-wetting transition present, changes in soil moisture were significantly greater with the smaller canopy than the larger canopy. There was a positive correlation between soil respiration rates and soil moisture under dry conditions, yet a negative correlation was present under wet conditions between soil respiration and soil moisture, once the soil reached

saturation. They concluded that canopy morphology (canopy width, leaf shape, canopy depth, and size class), and the subsequent direct and indirect effects on soil microclimate, are influenced by tree species and wetting-drying transitions present. They also concluded that leaf litter and fine root biomass do not play as great a role in altering soil respiration rates as soil temperature and moisture.

Grazing Effects on Soil Respiration

Davidson et al. (2000) found that during wet seasons, primary forests (which have not been cleared) and active pasture (cleared, planted with grasses, and grazed) had the highest soil CO_2 emissions. Although this study takes place in the Amazon, there are few studies that examine the effects of grazing on soil respiration. In addition, this study looks at the effect of soil moisture content on soil respiration rates following soil wetting periods similar to that of the monsoon seasons in drier areas. The degraded pasture had the lowest soil CO_2 emissions in both the dry and wet seasons due to its sparse vegetation. Secondary- (cleared, planted with grasses, grazed, and then left to succession) and primary forests had higher soil CO_2 emissions in the dry seasons than the active and degraded pasture lands, possibly due to more extensive and larger root systems. Although the secondary forest, once a degraded pasture, had more vegetation in the form of small trees, it had yet to reach the CO_2 production rate of the primary forest soils. However, respiration rates across all sites greatly decreased during the dry season, thus respiration rates are primarily tied to soil water content. Soils with low moisture content typically do not have large respiration rates, but can increase when soil water content increases, until it reaches field capacity, above which soil respiration starts to decrease.

Clapperton et al. (2002) found an increase in the amount of litter layer inside ungrazed areas, and concluded that there was an insulating effect that created the optimal, stable environment within the soil for microarthropods to exist. With an increased litter layer inside the exclosures, there were more mites present than under the light and very heavy grazing treatments. However, the abundance of microarthropods varied with grazing treatment, seasonality, and nutrient cycling rates. Many microarthropods that were present in the summer months of the heavy grazing treatment were few in abundance or absent in the fall, suggesting respiration rates change with temperature and abundance of litter. A positive correlation between the abundance and diversity of microarthropods in the soil, the reduction of soil disturbance, and the increase in productivity of grasslands under light grazing were also found.

Chen et al. (2016) found that grazing exclusion decreased soil respiration rates. During the growing season soil respiration rates decreased by 23.6% under grazing exclusion, and over the course of the entire year a 21.4% reduction was found. The slight difference between growing season and annual respiration rates showed that respiration was occurring primarily during the growing season. Soil moisture and aboveground biomass were increased with grazing exclusion, while soil temperature was significantly decreased. The increased soil moisture and decreased soil temperature by excluding livestock, were attributed to the increase in vegetative cover and the resulting decrease in soil warming and evaporation. However, Fu et al. (2014) found no significant differences in respiration between grazing exclusion and nonexclusion. It was found that soil respiration was positively correlated to soil organic C, microbial biomass, belowground biomass, and N cycling within the soil, and the lack of response of soil respiration to grazing exclusion may have been due to poor substrate quality (soil organic C and N cycling).

Neff et al. (2005) found no significant difference in soil respiration between historically grazed sites, with 30 years recovery time, and never grazed sites. Johnson and Matchett (2001) found soil respiration rates to be lower under grazing treatments. On grazed sites C may be allocated more into the shoots of vegetation instead of the roots, which could possibly alter biochemical properties in the rhizosphere; thus, decreasing the soil CO_2 efflux. In addition, surface litter accumulation and canopy photosynthesis will alter C concentrations within an ecosystem (Bremer, et al., 1998). Recently grazed areas may also have significantly lower respiration rates compared to ungrazed sites. Bremer et al. (1998) found soil CO_2 to be 22-47% lower in grazing treatments that had vegetation cut six days earlier than the unclipped treatments. Furthermore, soil respiration rates for simulated grazing, or clipping, treatments was negatively correlated to soil temperature during the growing season.

Ecological Restoration Effects on Select Soil Properties

Zahawi et al. (2015) found reference forests to have significantly greater differences in soil parameters, such as pH, CEC, percent base saturation, and type of cations and their concentrations, than the active restoration treatments, which included planting seedlings. The passive restoration treatment, which had no seedlings planted, was either intermediate in variable differences or comparable to the active restoration treatments. These differences in treatments indicate that restoration treatments may create soil conditions that are more favorable for grassland ecosystems than forest ecosystems. Soil quality is an important aspect in ecological restoration and function in an ecosystem. It is a measure of how well a soil can grow and

maintain plant biomass, cycle nutrients, store C, distribute water, and provide food and habitat for many different organisms (Schoenholtz, et al., 2000).

Ecological Restoration Effects on Soil pH

The acidity of a soil is expressed as the soil pH, and is the quantification of hydrogen ions (H^+) present in solution as the negative log of the hydrogen ion concentration (Brady and Weil, 2002). Soil pH affects the chemical, physical, and biological properties of a soil. Soil pH influences nutrient cycling rates, specifically N through nitrification and denitrification, and can be increased or decreased by these processes (Čuhel, et al., 2010). For example, denitrification tends to increase pH levels as more N_2O is produced by denitrification. Soil pH also affects soil physical properties such as aggregate stability. Soils with high pH, or alkaline soils, are often structureless, and may have higher levels of soluble salts and exchangeable sodium (Na) (Brady and Weil, 2002). Soil pH is affected by the cation exchange between plants and soils, the form in which N is absorbed, and organic acids present in plant litters. Traits specific to different plant species can influence soil pH, and in turn pH can affect plant uptake of nutrients (Eviner and Hawkes, 2008). The accumulation of organic matter, which is made up of various acid functional groups, will lower soil pH as it is a source of H^+ ions. Organic matter will also cause leaching of basic cations, such as calcium (Ca) and magnesium (Mg), when it forms soluble complexes with these cations (Brady and Weil, 2002).

In a short-term study, Gundale et al. (2005) found that pH was not significantly different among restoration treatments. Vesterdal et al. (1995) found, in a 30 year study, pH was highest in the heaviest thinned plots. Where the C:N and C:P ratios were highest, pH tended to be lowest;

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these significant differences between pH and C:N and C:P ratios may indicate there was a difference in available nutrients among sites, which could have been from change in mineralization rates. The pH was also found to not have many significant differences among treatments, but did have some site variation. Grady and Hart (2006) found no significant difference in pH between the unmanaged and thinned forest restoration treatments. Johnson et al. (1991) found a significant decrease in pH of mineral soil horizons E and B_h following whole-tree harvest. Zahawi et al. (2015) found a significant difference between the reference forest and active restoration treatment for soil pH. Boerner et al. (2007) found no significant difference in soil pH levels between an unmanaged stand and a mechanically thinned stand, but significant differences were found in stands that received fire and thinning + fire, compared to unmanaged stands.

Grassland soils will typically have a higher soil pH, due to higher organic matter deposition rates and higher quality substrates, which are characteristic of grasses. Kerns et al. (2003) found soil pH in the A horizon to be higher under grassy openings compared to old-growth and dense, younger ponderosa pine plots.

Grazing Effects on Soil pH

Johnston et al. (1970) found a significant difference in pH levels among grazing treatments, with the light, moderate, and heavy grazing treatments, and the heavy and very heavy grazing treatments similar. Smoliak et al. (1972) found a significant decrease in soil pH in the moderate and heavy grazing treatments compared to the ungrazed treatments. Johnston et al. (1970) suggested that under heavy grazing the soil microclimate was becoming drier as indicated by pH,

color, temperature, and moisture changes. Under predominantly aspen cover, pH was found to significantly increase under grazed conditions by elk. The pH determined for a meadow vegetation type had no significant decrease in pH under grazing (Binkley, et al., 2003). Yimer et al. (2008) found pH to increase for a grassland under grazed conditions, when compared to a native forest.

Ecological Restoration Effects on Plant Nutrients

The accumulation of common plant macronutrients, like N and P, and micronutrients, such as manganese (Mn) and copper (Cu), in soils are dependent on the type of litter, how much litter is produced and decomposed, and the type of tree species present. Perala and Alban (1982) compared the amount of nutrients found in the mineral soil and annual litterfall under different vegetation types. Jack pine (*Pinus banksiana* Lamb.) had higher mineral soil concentrations of N, Ca, and Mg, but lower mineral soil concentrations of P and potassium (K), compared to aspen in a loamy soil system. Within a sandy soil system, jack pine had higher mineral soil concentrations for all nutrients than aspen. Aspen litterfall had higher concentrations of P, K, Ca, and Mg than jack pine, but lower N concentrations, on both soil types. The amount of nutrients taken up by trees and where the nutrients are stored varies with species; for example, nutrients are contained mostly in foliage of conifer trees. The rate at which these nutrients cycle through an ecosystem is dependent on both natural and anthropogenic sources. Nutrients can be introduced into an ecosystem by weathering, atmospheric deposition, and fertilization, but can be removed by whole-tree harvesting, erosion, and leaching. Prescribed burning can leave plant available nutrients, such as K, Mg, Ca, and P, in ash on the forest floor, which can be leached into the soil

to be immobilized, taken up by plants, or lost through volatilization or in surface runoff (WDNR, 2011).

Vesterdal et al. (1995) conducted a forest restoration study with different thinning intensities over a 30 year time period, and found the soil C:N and C:P ratios tended to be lowest in heavily thinned plots. There was a higher accumulation of C, N, and P within the site that contained the highest percent sand, while the site with the smallest percent sand had the lowest accumulation. However, there were significantly greater litter accumulation differences among sites than among treatments. It was found as thinning intensities increased the amount of accumulated nutrients on the forest floor decreased. Easily altered C may increase soil organic C within low nutrient conditions and incite decomposition rates of more resistant litter (Eviner and Hawkes, 2008). When the production of litter exceeds the rate of decomposition in the forest floor, mineral nutrients accumulate in plant unavailable forms. In forest ecosystems nutrient biogeochemical cycling is imperative, as it allows nutrients that are unavailable in organic forms to be mineralized and become plant available.

Organic matter accretion and nutrient cycling rates can be affected by intensity of tree stand thinning. With an increase in thinning intensity, nutrient accumulation on the forest floor decreases, and the microclimate changes by increasing evaporation and decreasing transpiration rates. From these changes more favorable moisture regimes come about, resulting in a more favorable environment for microorganisms. Mineralization rates of C, as well as N, will increase due to less competition for nutrients and water between roots and saprophytic microorganisms. Whole tree harvesting can have effects on soil biochemistry by altering the chemical composition of plant litter and removing portions of the mineral nutrient capital within the tree (Piñeiro, et al., 2010; Hassett and Zak, 2005). Immediately following harvest the quantity of litter introduced

into the forest floor system, normally composed of fine roots, leaves, and their dead tissues, decreases. This decrease in substrate quantity leads to a shift in microbial communities and the inherent metabolic breakdown processes associated with them; thus, a long-term change in forest productivity in ecosystem functions (Hassett and Zak, 2005). Tree foliage, wood, and roots are all products of C assimilation, and play an essential role in the C-cycle. When trees shed litter in the forms of leaves and wood, there is an imbalance in growth and maintenance of the associated assimilation. When a plant becomes stressed or goes into senescence, tissues die and shed from the plant, and the result is C-filled litter (Schulze, 2000).

Johnson et al. (1991) found a significant change in exchangeable soil cation concentrations between pre-harvest and whole-tree harvest treatments. The exchangeable cations Ca, Mg, and K were significantly lower after harvest in the O_a horizon, and Mg and K were significantly lower in the E horizon. They also found the formation of an A_p horizon during post-harvest sampling; this A_p horizon formation reflects soil physical and chemical changes that occurred within the A horizon during the study due to mechanized equipment. Gundale et al. (2005) found no significant difference for exchangeable ions, such as Ca, Mg, K, Na, and extractable P.

Zahawi et al. (2015) found Ca and Mg concentrations for a reference forest to be significantly higher than both the passive and active restoration treatments. However, the reference forest was lowest in Cu concentrations. Concentrations of K were significantly higher for both the reference forest and passive restoration treatment forest than the forest receiving active restoration; while concentrations of iron (Fe) in the active restoration treatment were significantly different from the reference forest. However, Boerner et al. (2007) conducted a study in a mixed oak forest, and found no significant differences between the unmanaged and mechanical thinning forest restoration treatments for concentrations of exchangeable cations K and Ca. It was also found

that available P, when compared to the thinning treatment, was 46% lower in plots that received a prescribed burn, and 54% lower in plots that received both thinning and burning. All active restoration treatments had significantly lower aluminum (Al) concentrations than the control.

Grazing Effects on Plant Nutrients

Smoliak et al. (1972) found Ca and Na concentrations to be significantly higher under no grazing than with grazing, but exchangeable K, available P, and total P had no significant differences. Johnston et al. (1970) also found total P to be similar across grazing treatments. Beebe and Hoffman (1968) found P levels to be higher in the less grazed and reference sites than moderately and heavily grazed sites. Mg concentrations were also found to increase slightly with an increase in grazing intensity.

Binkley et al., (2003) found exchangeable Mg, Ca, K, and P were reduced under elk grazing treatments of different vegetation types. Under aspen (*Populus tremuloides* Michx.) vegetation extractable K was significantly higher in the grazed treatment. Yimer et al. (2008) found no significant differences in Na, K, Ca, or Mg concentrations when comparing grazed versus native forest. Neff et al. (2005) found historical grazing sites to have significantly lower concentrations of total elemental soil Mg, Na, and Mn than never grazed sites, but Ca concentrations in the soil were significantly lower for never grazed sites. K concentrations for the never grazed sites were higher than the historically grazed sites of similar soil taxonomic class, but P concentrations were not significantly different.

Grazing Effects on Soil Color

Soil color has been found to change under grazing intensity, and result in an increase in chroma and soil temperature. Johnston et al. (1970) found soil chroma became lighter as grazing pressures increased, and can be attributed to an increase in vegetation use and loss of organic matter. The soils in this study are classified in the Chernozemic order, the Canadian equivalent of Mollisols order, which have characteristically low chroma values due to melanization (Buol, et al., 2011). In addition, the higher chroma found could have been a factor in increased soil erosion rates. High chroma values are indicative of low amounts of organic matter present, and may be a reflection of the amount of water present in the soil pores. As grazing intensity increases water infiltration decreases, which could alter soil processes. Contradictorily, Smoliak et al. (1972) found no change in dry soil chroma among the ungrazed, lightly grazed, moderately grazed, and heavily grazed treatments on soils classified as Solonetzic, which are similar to Mollisols but with a lower Na content.

Ecological Restoration Effects on Soil Temperature and Moisture

Many plant traits influence soil temperature and moisture. Soil temperature and moisture are influenced by canopy coverage, litter persistence, chemistry, structure, and shoot morphology. Soil temperature is also influenced by soil color, while soil moisture is influenced by water uptake by plants, root and leaf area, and the water holding capacity and structure of the soil (Eviner and Hawkes, 2008). Dense canopies create a cooler soil microclimate than open canopies, creating a shift towards decreasing soil evaporation rates and increasing transpiration rates. Persistent high soil moisture content can also increase the amount of organic matter content

present and the amount of total N in the soils. Forest soils tend to have higher saturated hydraulic conductivity rates due to the presence of large macropores, created by roots or burrowing animals called biopores (Brady and Weil, 2002). However, forests that have been thinned by mechanized equipment are likely to have lower hydraulic saturated conductivity resulting from soil compaction.

Another factor that plays a role in soil temperature is the amount of direct sunlight reaching the soil. Open canopies tend to have higher soil temperatures than closed canopies, followed by thinned canopies during net warming periods. Moisture content will also decrease due to higher evaporation from the soil under warmer conditions from being exposed to a larger amount of direct sunlight. Higher moisture levels can accelerate organic matter decomposition, while increases in solar radiation will affects rate of microbial reactions (Hassett and Zak, 2005). Significant temperature differences were found by Ma et al. (2005) between open and closed canopies. The open canopy temperature stayed in the 15-25°C range fairly consistently throughout the year, while the closed canopy had a temperature range for most of the year between 10 and 20°C, but the soil moisture for both the open and closed canopy were similar throughout the year. Liu et al. (2014) found significant differences in soil temperature and VWC under small and large conifer canopies. Kaye and Hart (1998) found no significant difference in soil temperature among treatments.

Grazing Effects on Soil Temperature and Moisture

Grazing can affect soil physical properties, such as bulk density, which in turn will affect soil moisture. Grazing may cause soil compaction, especially when large amounts of herbivores are

present for long periods of time. Fine-textured soils can easily be compacted due to greater total pore space and their ability to retain water more readily than coarse textured soils. Soils that contain coarse fragments and large amounts of organic matter are more resistant to compaction (Selmants, et al., 2003). Soils high in organic matter will have higher porosity and lower bulk density, which allow for increased water and root movement through the soil. Soils that contain large amounts of coarse fragments will also allow for greater infiltration capacities, due to the "canopy" protection rocks provide to the soil from raindrops.

Johnston (1961) found soil moisture to be significantly lower at a depth of 0-5 cm, under the lightly grazed treatment compared to the ungrazed. Johnston et al. (1970) found the light grazing treatment had the lowest soil temperature, and the very heavy grazing treatment was significantly higher than all other treatments. Soil moisture under light grazing was similar to the moderate grazing treatment, but was greater than the heavy and very heavy grazing treatments. The moderate grazing treatment was similar in soil moisture as the heavy grazing treatment, but was significantly greater than the very heavy grazing treatment, which had the lowest soil moisture.

Clapperton et al. (2002) compared soil temperature and soil moisture at different depths within the soil profile, 0-4 cm and 4-8 cm, at different times of the year, June and October, and under a light grazing, heavy grazing, and exclosure treatments. It was found that soil moisture was significantly different among all treatments in the month of June at both depths, but there was only a significant difference between the very heavy grazing and exclosure treatment during the month of October at a sample depth of 4-8 cm. There was also a significant difference at a soil depth of 0-4 cm for both months among all treatments studied. Soil moisture was significantly lower under the very heavy grazing treatment at both sample depths and months.

However, soil moisture was similar under the light grazing and exclosure treatments, and significantly lower at a soil depth of 4-8 cm, among all treatments.

Vegetation Dynamics

Vegetation and soil interact in many ways, so when conducting ecological restoration it is important to look at the effect vegetation has on soil properties, and vice versa. Vegetation can alter soil structure by creating pores in soils, enhance wetting and drying cycles that facilitate soil shrinking and swelling, and decrease bulk density. With a decrease in bulk density, gas and water exchange between plant roots and the atmosphere and root penetration will be facilitated (Weidenhamer and Callaway, 2010). As plant roots die and decay they promote organic matter addition, which will influence many soil chemical and biological processes. With fire exclusion in the southwestern US, vegetation communities have shifted and pine encroachment has resulted. Before fire exclusion an open forest structure was predominantly maintained pre-settlement by lightning-caused fires that occurred on average about every 10 years (Laughlin, et al., 2007), with some Native American influence, who burned frequently with low-intensity. In fact, lightning-caused fires started to mimic Native American set fires after tree canopies became open (Kaye, 1998). Decreased understory richness and cover, the domination of shorter-lived species, and the reduction of nutrient cycling rates may result from fire exclusion and woody plant encroachment, such as ponderosa pine increasing in density and volume within an otherwise low stocked forest or prairie (Laughlin, et al., 2004; reviewed by: Metlen and Fiedler, 2006). Intensively grazed areas coincide with pine encroachment and high tree densities in conjunction with fire exclusion

in the southwestern US (Bakker and Moore, 2007). Drier ponderosa pine forests are well-adapted

to low- and mixed-intensity fire regimes, so stand replacing fires, although a natural and healthy

component of many forest environments, may have detrimental effects on some ponderosa pine

ecosystems (Metlen and Fiedler, 2006). The probability of stand replacing fires, increased

resource competition, and potential risk of insect and disease outbreaks are possible results from

increased stocking rates in ponderosa pine forests (Fajardo, et al., 2007). Native understory

plants of these ecosystems are not adapted to intense, stand replacing fires; therefore such events

may result in the acceleration of exotic species establishment (Metlen and Fiedler, 2006).

Understory species are also affected by potentially limiting factors, such as low available N and

water.

Effects of Ecological Restoration on Vegetation

Through fire exclusion, ponderosa pine stands have become dense, resulting in a decrease of

understory vegetation cover and richness. With restoration treatments, such as thinning and

burning, understory richness and cover may be increased. Stoddard et al. (2011) implemented

restoration treatments (thinning + prescribed burning) at different thinning intensity rates. Over

an eight-year period vegetation was monitored using indicator species, defined as the most

profuse and most common species. The first two post-treatment years for medium- and high-

intensity treatments, yielded non-native dominated areas at 50% and 45% of the indicator species

found. The final post-treatment year had Dalmatian toadflax (*Linaria genistifolia ssp. dalmatica*

L.) as the only non-native, indicator species in the medium-intensity treatment, but for other

treatments non-native species were not a major component. Non-significant trends in species

richness as a result of reductions in basal area and canopy cover were seen, which differs from other ponderosa pine restoration studies (Metlen and Fiedler, 2006; Moore, et al., 2006; Laughlin and Fulé, 2008; reviewed by: Stoddard, et al., 2011). Stoddard et al. (2011) found herbaceous plant cover was inversely related to thinning intensity. Graminoid cover increased by more than 470% in high-intensity treatments by the final post-treatment year, but had only a 53% increase for the control treatment. They found that when different thinning intensities were applied, the understory community under ponderosa pine was reshaped, and hypothesized that under post-fire conditions in ponderosa pine forests, encroachment of non-native, invasive species is higher in the drier and warmer low-elevation ecosystems. In addition, they concluded that to achieve a significant increase in herbaceous cover the tree basal area must be 10 m^2 ha^{-1} or less.

Moore et al. (2006) conducted an ecological restoration study that included a control, a thinning treatment (thinning from below), and a composite treatment (thinning from below, forest floor manipulation, and periodic prescribed burning). While graminoid cover for all treatments was not significant, C_3 graminoids, such as bottlebrush squirreltail (*Elymus elymoides* Raf. Swezey) and various sedges (*Carex* L. spp.), responded more than shade-intolerant, C_4 graminoids, like mountain muhly (*Muhlenbergia Montana* Nutt. Hitchc). Graminoids decreased one year after the second prescribed burn, but five years later C_3 graminoids comprised over half the understory percent cover in all treatments. The increase in C_3 graminoids could be due to bottlebrush squirreltail's typically swift colonization of disturbed areas and proliferation after low-intensity burns (Jones, 1998; Young and Miller, 1985; reviewed by: Moore, et al., 2006). C_4 graminoids could have been affected adversely by prescribed burns, because the functional group never increased in the composite treatment. This result coincides with another study that found a 10% decrease in mountain muhly one year after prescribed burning in a ponderosa pine forest

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(Gaines, et al., 1958; reviewed by: Moore, et al., 2006). Forb communities, which never differed between the thinning and composite treatments, were mostly comprised of perennial forbs. Perennial forb cover did not differ by treatment or year a few years into the study, but 10 years later perennial forbs cover decreased by 7-13% in all treatments, respectively. Annual-biennial forbs, such as common mullein (*Verbascum Thapsus* L.), were negligible in the restoration treatments at the beginning of the study, but were greater in the composite treatments than the other treatments for the last five years of the study. One year after a second prescribed burn the annual-biennial forbs were 20% of the total standing plant biomass in the composite treatment. However, after a severe drought three years later the annual-biennial forbs were only 7% of the total standing plant biomass for the composite treatment. Legume forbs had a greater change response to treatments than non- leguminous perennial forbs, but did not peak until one year after the second prescribed burn, with a decline thereafter. C_3 graminoids showed an immediate response, but forbs did not respond until five to six years after the initial thinning and burning (Moore, et al., 2006).

Kerns et al. (2003) analyzed understory response under grassy openings, dense, young ponderosa pine stands, and old-growth ponderosa pine stands. Transects comparing old-growth and grassland plots found a significantly smaller percentage of grass cover and more forest floor cover under old-growth stands. In comparing old-growth and dense, younger pine transects, there was no difference in percentage of grass, forb, shrub, or forest floor cover.

In an ecological restoration study by Busse et al. (2009), treatments included a thinned and unthinned treatment, each of which received an additional no burn or burn treatment. Tree mortality was greater in unthinned plots than thinned plots, but tree mortality between burned and unburned for unthinned plots was the same. However, live-tree volume was reduced by burning

in unthinned plots only. Height to green crown for both thinned and unthinned plots were affected by burning. Shrub cover and persistence varied with treatment. In the short-term, thinning alone reduced shrub cover by 28% from pretreatment values, but over the course of the study shrub cover increased. Thinning + burning dramatically reduced shrub cover, but cover quickly increased post-fire. On untreated plots shrub cover steadily declined throughout the experiment, but shrub cover was not essentially eliminated unless repeated prescribed burns occurred. Following a single burn there was no significant difference between the burned and unburned treatments in shrub cover five years later. Total cover for shrubs was significantly correlated to ponderosa pine cover, but shrub composition did not change under the assorted treatments conducted. Three years after burning was the only time in which herbaceous species richness was reduced under thinned plots, but no significant effect was found for species richness under unthinned + burned plots. Herbaceous species richness was also not affected by thinning or slash removal, but Bakker and Moore (2007) relate that logging slash can increase understory vegetation due to overstory reduction, but when too much logging slash accumulates understory vegetation will decrease.

Busse et al. (2009) found second-growth ponderosa pine ecological restoration characteristics desired to be mostly met by thinning only treatments, while still maintaining soil and biological properties. There are four main principles in establishing fire resistance in dry forests: reduce surface fuels, increase height to live crown, decrease crown density, and keep large trees of resistant species (Agee and Skinner, 2005). Basal area and stand density were reduced by approximately 50% to levels designed to reduce crown fire, constrain insect infestation, and improve tree growth (Busse, et al., 2009). However, unthinned stands showed tree mortality from insect infestation (bark beetle), had decreased habitat and cover for wildlife and livestock with the

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reduction of N-fixing shrubs, and tree vigor was two times less than in thinned plots. With an increase in tree vigor, older trees are less likely to be stricken by insects and diseases (Fajardo, et al., 2007). Busse et al. (2009) suggested that nutrient poor sites with little understory population may not need a prescribed burn following thinning restoration treatments. Also, retaining thinning residues after harvest is not needed, and is not a preferred method in reducing fire risk. Ritchie et al. (2007) observed that crown fire spread and tree mortality abruptly stopped when fire reached areas that had undergone thinning and understory burning, and tree mortality and scorching occurred only in small patches and the fire was reduced to a surface fire in areas that received thinning only (reviewed by: Busse, et al., 2009).

Metlen and Fiedler (2006) found a significant decrease in understory species richness under the control treatment, compared to all other treatments, thin, burn, and thin + burn. Immediately following the first burning, the burn-only treatment had the lowest total species richness, and the thin-only treatment had the highest total species richness. In the final year of the study, the thin-burn treatment had the greatest total species richness, followed in decreasing order by the thin-only, burn-only, and control treatments. The thin-only treatment also had the greatest species-richness of forbs following the first burn. Within the study period the burn-only treatment only gained three species, while the thin-burn treatment had the largest gain of 25 species. Usually, understory cover increases with a decrease in overstory cover (Wienk, et al., 2004), but Metlen and Fiedler (2006) found understory cover decreased with a decrease in overstory cover. However, slash was not removed offsite, unlike the study conducted by Wienk et al. (2004). Metlen and Fiedler (2006) suggest that the rapid recovery and increase in species richness and cover under the thin-burn treatment following fire showed the understory community's elasticity to thinning and burning treatments.

Fajardo et al. (2007) performed cut-only and cut-burn treatments. Any felled trees or logging slash resulting from the cutting treatment were left to provide fuel for the prescribed burn, but this would have created a higher intensity fire than what was typical of pre-settlement fires. Tree vigor was analyzed by growth efficiency (GE), which is the ratio of stemwood production per unit leaf area. It is also a representation of a tree's ability to assimilate C in its crown, and can be affected by stand structure and light availability. Tree vigor was greatest in the cut-only treatment, followed in order by the cut-burn and control treatments, which were not significantly different. Mature trees (50-100 years old) had the greatest basal area reaction to restoration treatments. The growth in basal area since treatments were applied almost doubled after 10-years in the cut-only treatment, compared to the control. Mature tree basal area growth for the cut-burn treatment was 40% more than the basal area growth for the control treatment. The youngest trees (less than 50 years) had no significant response in mean basal area to restoration treatments, and compared to the older tree class (greater than 100 years) was significantly lower in growth rate. Mature and young tree growth and vigor were more negatively affected by burning.

The results of Fajardo et al. (2007) suggest prescribed burning after cutting in ponderosa pine forests may create a negative response in tree growth, compared to thinning only. These results are similar to other studies conducted in ponderosa pine forests that reported cut-burn treatments are not always superior to cut-only treatments, when looking at success variables such as residual tree growth and vigor (Kaye and Hart, 1998; Kaye, et al., 2005; Hart, et al., 2005; reviewed by: Fajardo, et al., 2007). Historical fires in ponderosa pine forests typically occurred during late summer and early fall, but, due to safety concerns in these dry ponderosa pine forests, prescribed burning usually takes place in early spring when moisture conditions are high. Burning in the spring could have a detrimental effect on tree crowns and roots. Leaf surface area that enables

high transpiration rates may be reduced by fire, which may have a negative effect on photosynthetic properties (Wyant, et al., 1983; reviewed by Fajardo, et al., 2007). In addition, it has been found that physiological activity of roots is at its highest during the spring, and burning could greatly reduce fine root biomass by about 60% (Grier, 1989; reviewed by: Fajardo, et al., 2007).

Effects of Grazing on Vegetation

Soil quality and health, biota diversity and abundance, and structure of plant communities are all influenced by livestock grazing. There are many contradicting reports concerning grazing in the western US. Grasslands that have been subjected to continual grazing by large herbivores historically are more resilient to grazing pressure, and will undergo less of an ecological change than grasslands that have not undergone grazing (Schultz, et al., 2011). Piñeiro et al. (2010) also suggested historical grazing patterns may influence ecosystem composition change, but can also be influenced by particular processes, such as selective grazing, resource availability changes, and the breakdown of non-resilient species. Loeser et al. (2007) proposed that grazing intensity impacts on vegetation vary by climatic conditions, such as precipitation, soil quality, and landform type. They also found a 10% decrease in plant cover under cattle removal and moderate grazing treatments, and a greater than 30% decrease in plant cover under a heavy-impact grazing treatment, two years after a severe drought that forced ranchers to remove cattle from the land, suggesting that past grazing practices highly influence the plant community under climate changes. However, as pointed out by Piñeiro et al. (2010), study sites comparing grazed and ungrazed plots will have similar species composition if a history of heavy grazing has occurred,

but sites with short grazing histories will have a different species composition between grazed and ungrazed plots.

Schultz et al. (2011) found in Australia that excluding grazers significantly increased the amount of phytomass, or live and dead plant biomass. Higher accumulations of phytomass led to a significant decrease in species richness, while lower amounts of phytomass led to a significant increase in species richness. Grazing exclusion increased vegetative cover, but species evenness decreased significantly. However, species evenness decreased at all sites. Phytomass accumulation can be attributed to rainfall, soil parameters, such as plant available N, and tree cover. It was also found that the greater the tree cover, the less phytomass accumulation, thus demonstrating that grass productivity is suppressed by trees.

Johnston (1961) found there was a more diverse vegetative community on grazed land than on undisturbed land in a fescue grassland. There was an increase in forb and shrub species from 13 in 1949 to 22 in 1960 under light grazing and a decrease to 12 species under no grazing. Grasses decreased with both treatments, from nine species in 1949 to eight species in 1960 under light grazing and six species under no grazing. It was also found that conditions were cooler and more moist in the top 12 inches of the soil profile on the ungrazed site than the lightly grazed site. Johnston attributed the decreased diversity of vegetation in the ungrazed treatment to an accumulation of surface organic matter.

Gao et al. (2007) found vegetative cover to be greatest under moderate grazing, and lowest under heavy grazing on the eastern Tibetan plateau. Above- and belowground biomass, both living and dead, was significantly lower in the heavy grazing treatment compared to the light and moderate grazing treatments. The intensity of grazing will influence the magnitude of response the vegetative community undergoes, for grasses decline and sedges proliferate under heavy

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grazing. Light to moderate grazing will create a stable soil and vegetative environment, where aboveground biomass is high with predominately forage grasses, and the plant community is diverse and stable.

Restoration and Grazing Effects on Vegetation

Bakker and Moore (2007) conducted a study on ponderosa pine from 1941 to 2004 in areas where grazing exclosures had been maintained since 1912. It was found that total canopy cover within exclosures were almost two times higher than grazed areas, and total tree density was more than three times greater inside than outside the exclosures in 2004. However, in 1941 both variables did not differ between grazing treatments. Historical livestock grazing effects on overstory vegetation were noticeable in 1941, but by 2004 they were increasingly more obvious. These differences in tree canopy cover and tree density between grazing treatments can also be seen in Grand Canyon National Park (GCNP) and the surrounding area. Where GCNP, from the late 1800s, was heavily grazed by sheep and cattle, and then excluded since the late 1930s, tree densities have become higher than the surrounding, continually grazed areas (Fulé, et al., 2002; reviewed by: Bakker and Moore, 2007). Bakker and Moore (2007) also found that grazing might have an impact on pine regeneration. Meta-analyses were conducted with both non-adjustments and adjustments for overstory effect, and concluded that overstory effects were larger than grazing effects. When adjusted for overstory effects, graminoid cover was the only variable that differed between grazing treatments. Bakker and Moore (2007) also stated that sheep and cattle preferentially graze in openings rather than dense tree stands, and will affect the understory response found among vegetation canopy types. It was also suggested that any study determining

the change in understory vegetation over time, either with grazing or not, should account for overstory effects, for the ponderosa pine canopy has a significant effect on understory communities, perhaps more than grazing or temporal dynamics.

Vandvik et al. (2005) concluded that vegetation responses to fire and grazing were habitat-specific, because site-scale richness and fine-scale species density were higher under moist heathlands than dry heathlands, during a five-year period. One year following a fire disturbance, all treatments sampled, except for a moist, north-facing, ungrazed site, had lower average plot species densities than the control. By the second year after fire disturbance, all treatments, except for a dry, south-facing, ungrazed site, were above the control level for average plot species densities. Throughout the experiment the moist, north-facing, ungrazed and the dry, south-facing, ungrazed sites had the highest and lowest site species richness and average plot species density throughout the study period. Grazing also produced more random variation among replicates, opportunities for new species to colonize, successional dynamics that were more complex, and composition trends that were different among the three sites tested. Vandvik et al. (2005) concluded that grazing is a heterogenizing factor, which contradicts other studies (Olff and Ritchie, 1998, Stohlgren, et al., 1999, and Harrison, et al., 2003; reviewed by: Vandvik, et al, 2005), for grazers will not evenly graze landscapes. For example, it was found in Norway that sheep graze burnt heath instead of un-burnt, prefer ridges with open areas instead of low-lying or steep areas, and graze in dry areas instead of moist. Thus, grazing intensity will depend on topography, moisture conditions, as supported by Loeser et al. (2007), and the presence of fire (Vandvik, et al., 2005). Species richness may decrease with preferred grazing, but on a larger, landscape scale more heterogenic species variations may occur.

<u>Invasive Plant Species</u>

Exotic or invasive species, which are native to other regions, but can survive and reproduce in another, are very important factors in assessing plant-soil relations within an ecosystem. Invasive species have the ability to alter nutrient cycles, outcompete native vegetation, and release secondary compounds, which can alter the composition of the soil in which plants grow, through root exudates, leaf litter, leaf leachates, and leaf and root volatiles. With a shift in the type of secondary metabolites that are exuded into the soil profile, an ecosystem's soil biogeochemical processes may be altered (Weidenhamer and Callaway, 2010). Common invasive species that can be found in this study area include cheatgrass (*Bromus tectorum* L.), spotted knapweed (*Centaurea maculosa* Lam.), and Dalmatian toadflax, all of which proliferate readily (Northum and Meyer, 2009). Nutrient cycling is directly affected by invasive plants, and it has been found that decomposition and nutrient cycling rates can increase under invasive plant cover due to their higher growth rates, specific leaf areas, and leaf nutrient concentrations, compared to native vegetation (Weidenhamer and Callaway, 2010). For example, N, P, Mg, Fe, K, and Ca availability have been found to increase under cheatgrass invasions, and could affect vertical nutrient distribution through the soil profile (Blank, 2008: reviewed by: Weidenhamer and Callaway, 2010). The effects invasive plants have on nutrient pools within soils are site dependent, so nutrient-poor sites are more likely to have stronger increases in nutrient concentrations than nutrient-rich sites (Weidenhamer and Callaway, 2010). The invasive forb garlic mustard (*Alliaria petiolate* M. Bieb. Cavara & Grande) has been found to increase soil pH and N, Ca, Mg, and P availability. Secondary compounds released by garlic mustard, root exudates or leaf volatiles, were not found to affect nutrient cycling in these soils. However,

nutrient cycling was altered by decomposition of garlic mustard leaf litter, and the resulting

increase in decomposition rates of litter of native tree species (Rodgers, et al., 2008; reviewed by:

Weidenhamer and Callaway, 2010).

Higher P concentrations within soils are often associated with the presence of invasive plants,

but whether they are present due to high P concentrations or are creating high P concentrations in

the soil is still not understood. For example, spotted knapweed was found to have two times

more available P than three abundant native species measured in a field setting. In addition, in

field conditions where P availability was low, P uptake was six times higher in spotted knapweed

than the native legume *Lupinus argenteus* (Thorpe, et al., 2006; reviewed by: Weidenhamer and

Callaway, 2010). It appears that spotted knapweed exudes phosphatases and chelating

compounds that increase the concentration of available phosphorous in the rhizosphere, thus high

P levels found in a soil invaded with spotted knapweed may be a direct result of the plant

(Weidenhamer and Callaway, 2010).

Controlling invasive plants is usually done with the use of herbicide. A common herbicide

used is glyphosate, but the extent to which glyphosate affects soil microbiology or chemistry is

not well known. The extent to which the microbial population is affected is difficult to determine

because less than one percent of soil microbes can be cultivated in the lab (Weidenhamer and

Callaway, 2010). Glyphosate was applied 9-13 years before microbial biomass, respiration, and

metabolic diversity were measured in three ponderosa pine forests, and the effects on microbial

communities were minimal (Busse, et al., 2001; reviewed by: Weidenhamer and Callaway, 2010).

In a different experiment microbial biomass was found to increase in response to glyphosate

application, but, despite the response being minimal, this small change may alter soil processes

and food webs (Lupwayi, et al., 2009; reviewed by: Weidenhamer and Callaway, 2010). Non-

target plants can also be affected by herbicide application. For example, in controlling spotted knapweed the herbicide picloram is used. After one experimental application of picloram to the native arrowleaf balsamroot (*Balsamorhiza sagittata*), flowering and seed set was reduced for four years (Crone, et al., 2009; reviewed by: Weidenhamer and Callaway, 2010).

Invasive exotic species can be unintentionally favored under some ecological restoration treatments, and it was found by Metlen and Fiedler (2006) that forests that underwent thin + burn treatments had the highest abundance of exotic species. Thinning-related treatments did produce an increase in exotic, as well as native, species, yet despite this increase in exotics, untreated forests may be more susceptible to severe invasion of exotic species. High intensity fires can occur in areas with high amounts of accumulated fuels, and invasion of exotics may increase. Haskins and Gehring (2004) also found species richness to be lower, biomass 19% greater, and populations of exotic species to be four times as abundant on burn sites compared to other areas (reviewed by: Metlen and Fiedler, 2006).

In a study conducted by Loeser et al. (2007), there was variation of climatic conditions, most notably a drought in the year 2002. After the drought period they found that native plants decreased under heavy grazing and exotic plants flourished. Plant cover returned the following year, but had more exotic species, especially annual plants such as cheatgrass. One year after the drought, cheatgrass dominated the high-impact grazing site, and 80% of subplots were covered, while less than 50% of the subplots were covered in the ungrazed and moderately grazed treatments. A year later cheatgrass almost completely dominated the high-impact grazing treatments. Cattle grazing, which has been known to increase the presence of cheatgrass and other exotics, can alter the plant community and displace native vegetation. However, cheatgrass is also known to be present in locations where no grazing activity is present. It was hypothesized

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that watering holes, corals, and trails, or areas that have a large amount of ungulates present, are more likely to have an increase in exotic species after a severe drought. At the end of the study, exotic plants increased 13% in the high-impact grazing treatment compared to the moderate grazing treatment. In addition to an increase in exotic species, high-impact grazing homogenized the plant community, increased annual grass cover, and steadily decreased perennial forb cover. Perennial grass and annual forb fluctuations were too inconsistent to be considered significant. Sites that have an abundance of exotic species and low productivity tend to be the sites with the greatest species richness (Eviner and Hawkes, 2008).

With climate change more droughts are likely to occur in some cases, which can severely diminish the native plant cover if heavy-impact grazing is practiced. The effects of invasive, non-native plant species on microbial communities, nutrient cycles, and other soil characteristics can linger years after removal (Eviner and Hawkes, 2008), and can have negative chemical and biological consequences (Weidenhamer and Callaway, 2010). Understanding this concept and using it to create better management practices is the key to maintaining a viable plant community, which will help forest managers and ranchers, as well as the environment.

Effects of Soil Properties on Vegetation

Compared to forest soils, soils with a large grass component have substrate additions with higher pH, lower lignin:N and C:N ratios. These influences on soils will increase over time as the vegetation communities persist. Kerns et al. (2003) compared old-growth, grass, and young-pine sites in a ponderosa pine-bunchgrass ecosystem, and found the old-growth plots had measurable O horizons and a significantly lower pH for the A horizon compared to the grass plots. C and N

concentrations were similar among plots, but generally lower for grass plots compared to old-growth plots, with C having the greatest difference. The old-growth plots had significantly thicker O horizons and higher A horizon C and N concentrations than the young-pine plots. It was found that A horizon C concentration was positively correlated with O horizon thickness for all plots. Lower C concentrations may be attributed to high cycling rates of organic matter under grass vegetation, when compared to old-growth plots.

Laughlin et al. (2007) used structural equation modeling (SEM) to determine how understory species richness and cover, forest canopy cover, and properties of the mineral soil are correlated in ponderosa pine ecosystems. Silt %, which represented soil texture in the model, was positively correlated to understory species richness, but gravel content was negatively correlated. Organic matter content and understory plant cover were positively correlated to understory species richness, yet litter depth, pine basal area, and total N had no correlation with species richness. Litter depth and pine basal area had a negative relationship, but litter depth and total N had a positive association. Gravel content had no significant effect on understory plant cover. It was suggested that the second-order, positive relationship between plant cover and richness may be a result of reduced competition and success in plant colonization under open stands with low abundance of pine. They found that litter did not cause variation in plant cover or richness under presence of pine, which suggests pine density has a greater effect on plant abundance than litter accumulation. They implied that prescribed fire, as a litter reduction strategy, may not achieve ecological restoration goals, as supported by Fajardo et al. (2007). Soils that had higher total N content had lower species richness levels, indicating reduced ability of soils to maintain production of herbaceous plants, due to nutrient depletion. Soils that had high gravel content also had lower species richness, which can be related to the reduction of soil available water capacity,

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high soil drainage rates, and root impediment. In addition, soils that were derived from black, cinder parent materials had lower species richness with high gravel content, but had higher species richness with low gravel content. It was concluded that species diversity, on a small, local scale, is directly and indirectly affected by soil properties.

Abella et al. (2015) conducted ecological restoration treatments across three different soil parent material types, limestone, benmoreite, and basalt, and under four different restoration treatments, each of which had a grazing and no grazing component. One treatment, the thinned + simulated smoke treatment, will not be discussed due to inconsistent data. The three remaining treatments included an open, thinned, and closed treatment. Among all treatments, species richness was affected by thinning, with basalt and benmoreite parent materials having significantly higher richness. However, when factoring in a grazing variable, grazed and ungrazed treatments under a thinning treatment had a significant increase in species richness from pretreatment to posttreatment on benmoreite parent material soils. On limestone and basalt soils only a thinning and grazing interaction was found to increase species richness. However, cover under the thinning treatment was significantly higher under no-grazing, and was the most pronounced on limestone soils. Cover was also significantly higher after a thinning treatment under no-grazing for the benmoreite parent material. No significant change in cover between pretreatment and post-treatment was found on the basalt soils. Of the 145 species recorded, 90% were native (130 species) for all plots measured, and 104 of the recorded species found were perennial. Interestingly, three years after treatment (2006) the vegetative response in species richness, composition, and cover had occurred, and little change was found five years after treatment (2008).

Justification

To understand the effects of ponderosa pine ecological restoration treatments, repeated measurements of both vegetation and soil parameters must be carried out over the long-term. This study was conducted to determine if management practices on sites of specified soil parent materials have experienced changes in understory plant species composition and structure, as well as, potential changes in select soil chemical and biological properties. Few studies elucidating the influence soil parent material type has on ecological restoration effects have been carried out, so by adding data to the original study longer term effects of soil parent material type were analyzed. Additionally, early season vegetation data was analyzed, so species that flourish early in the growing season could also be distinguished. By utilizing the information collected, more effective, site specific restoration approaches can be developed for these sites.

MATERIALS AND METHODS

Study Area

This research was conducted within the Northern Arizona University Centennial State Forest, located 10 kilometers (km) southwest of Flagstaff, Arizona, and used most of the study sites used by Abella et al. (2015). Sampling was completed during the months of June and July 2015 for soil parameters and vegetation surveys. The soils of the study area have parent materials of either basalt, benmoreite, or limestone, and are classified as either Typic, Lithic, or Mollic Eutroboralfs. The study area is dominated by ponderosa pine and Gambel oak (*Quercus gambelii* Nutt.). The climate of the area has a mean precipitation of 57 cm year^{-1}, with an average of 25 cm year^{-1} water equivalent as snow, a January daily low temperature average of -12°C, and a July daily high temperature average of 27°C (2,137m elevation; 1950-2015 records; Western Regional Climate Center, Reno, NV, U.S.A.). The study area receives monsoonal moisture typically between June and September (Hutchings, et al., 2009).

Wildlife present within the study area include mule deer (*Odocoileus hemionus*), a native species, and Rocky Mountain elk (*Cervus elaphus*), which were introduced to Arizona in 1913 (Arizona Game and Fish Department, Phoenix, AZ, U.S.A.). The study area also supports grazing livestock seasonally, such as cattle (*Bos primigenius*) and sheep (*Ovis aries*). When the original study was being conducted the density of livestock (animal unit months per hectare) within the study area ranged from 0.03 to 0.20 AUM ha^{-1} (Bakker and Moore, 2007; U.S. Forest Service, Coconino National Forest, Flagstaff, AZ, U.S.A.). The density of livestock while this study was taking place was between 0.002 and 0.010 AUM ha^{-1} (US. Forest Service, Coconino

National Forest, Flagstaff, AZ, U.S.A.); however, significant overgrazing historically occurred throughout much of Arizona between 1870 and 1890 (Loeser, et al., 2007).

Soil Types

The sites selected by Abella et al. (2015) in 2003, and also used in this study, were similar in many properties, but were distinguished by one or more different characteristics, such as depth to the argillic horizon. Clay concentration among the sites varied from 16 to 24% in the surface soil (0-15 cm), and the organic C concentration within the subsoil was twice as much for limestone than basalt parent material sites (Table 1). The limestone soils had coarser textures, a higher vegetative species richness and cover, few coarse fragments relative to the other two parent material types, and supported a mixture of understory perennial grasses and forbs. The benmoreite locations had silt loam textures, with clay texture within the first 30 cm, not as much vegetative diversity and abundance as the limestone soils, and were intermediate in coarse fragment content. Benmoreite is a rare intermediate volcanic rock containing clinopyroxene, feldspar, and plagioclase minerals (Ronga et al., 2009). The basalt soils were dominated by forbs, grasses such as mountain muhly, and graminoids such as sand dropseed (*Sporobolus interruptus* Torr. A. Gray) and White Mountain sedge (*Carex geophila* Mack.), had many coarse fragments, and were composed of finer textured materials. Basalt is a basic volcanic rock containing olivine, plagioclase, and clinopyroxene minerals (Ronga, et al., 2009). A few random samples were chosen to conduct texture analysis, using the Bouyoucos method, for the three soil types.

Limestone and basalt soils were sandy clay loam and loam textures. Benmoreite soils were found

to have a loam texture.

Table 1. Characterization of soils and vegetation before restoration treatments were implemented
in the Centennial State Forest in Flagstaff, Arizona.

	Limestone	Benmoreite	Basalt
Elevation (m)[a]	2190 ± 21	2225 ± 40	2214 ± 11
Rock cover (%)	1 ± 1	4 ± 2	5 ± 1
0-15 cm 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
Dominant species	*Poa pratensis*	*Festuca arizonica*	*Carex geophila*
	Erigeron formosissimus	*Elymus elymoides*	*Muhlenbergia montana*
Dense tree canopy[b]			
Species per subplot	6 ± 2	2 ± 1	3 ± 1
Species per measurement plot	13 ± 3	5 ± 3	9 ± 2
Plant cover (%)	8 ± 6	1 ± 1	3 ± 1
Open tree canopy			
Species per subplot	12 ± 5	6 ± 4	6 ± 2
Species per measurement plot	23 ± 7	14 ± 7	14 ± 4
Plant cover (%)	15 ± 8	8 ± 7	12 ± 11

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 2 mm in diameter.
[b]Vegetation variables for both tree canopy types are pre-treatment in 2003 and were obtained through data collection by Abella et al. (2015).

Study Sites

There were a total of eight study sites, which were located on three different parent material

types, limestone, basalt, and benmoreite. This follow-up study used sites and measurement plots

chosen and put into place by Abella et al. (2015), with the exception of one site. This site, on

basalt parent material, was excluded due to a tornado that had passed through the area, resulting

in some exclosures being destroyed and considerable tree damage, and it could not be determined

if ungulates or grazers had entered the fallen exclosure. Each study site exhibited the following

characteristics: located within 1 km of a forest road, had no visual evidence of fire since 1880,

and contained ponderosa pine that were greater than 50 years of age to mimic forests that are

undergoing thinning treatments. The precipitation, historical grazing and forest management

practices, and mean elevation were similar among all study sites, so the primary difference among

study sites was the soil geologic parent material. On each study site, there were four

experimental treatment plots (Figure 1) in which each treatment plot consisted of two 3.16 m $\times$

3.16 m (10 m^2) measurement plots delineated within a 20 m $\times$ 25 m (0.05 ha) area. These 10m^2

measurement plots had a grazing treatment component, grazing control and grazing exclusion.

For these 10m^2 measurement plots there was a small, hypothetical buffer zone on the outer edge

of the 10m^2 plots to compensate for a change in microclimate caused by snow drift and

accumulations due to the presence of exclosures for the grazing treatment. With the buffer, the

plots became 9m^2 in size, and from this point on will be referred to as the measurement plots.

Grazing exclosures were 2 m tall, had four metal fence posts, 1mm thick wire with 5 cm x 10 cm

openings, were located in the center of half of each delineated area. The second measurement

plot at each site, from this point on known as the grazed area, was originally delineated by metal

rods at each corner, and was located in the center of the other half of the 20 m x 25 m area

(Figure 2). If some of the rods delineating measurement plots could not be found at a site,

schematics drawn of the measurement plots were used to determine the location in which to

collect data. If only a few of the rods could be found, measurements were made to make the best

estimation of the approximate location, and were delineated with a pin flag for future reference,

until rods could be reinstalled. Coordinates for sites in this study are given in Table 2.
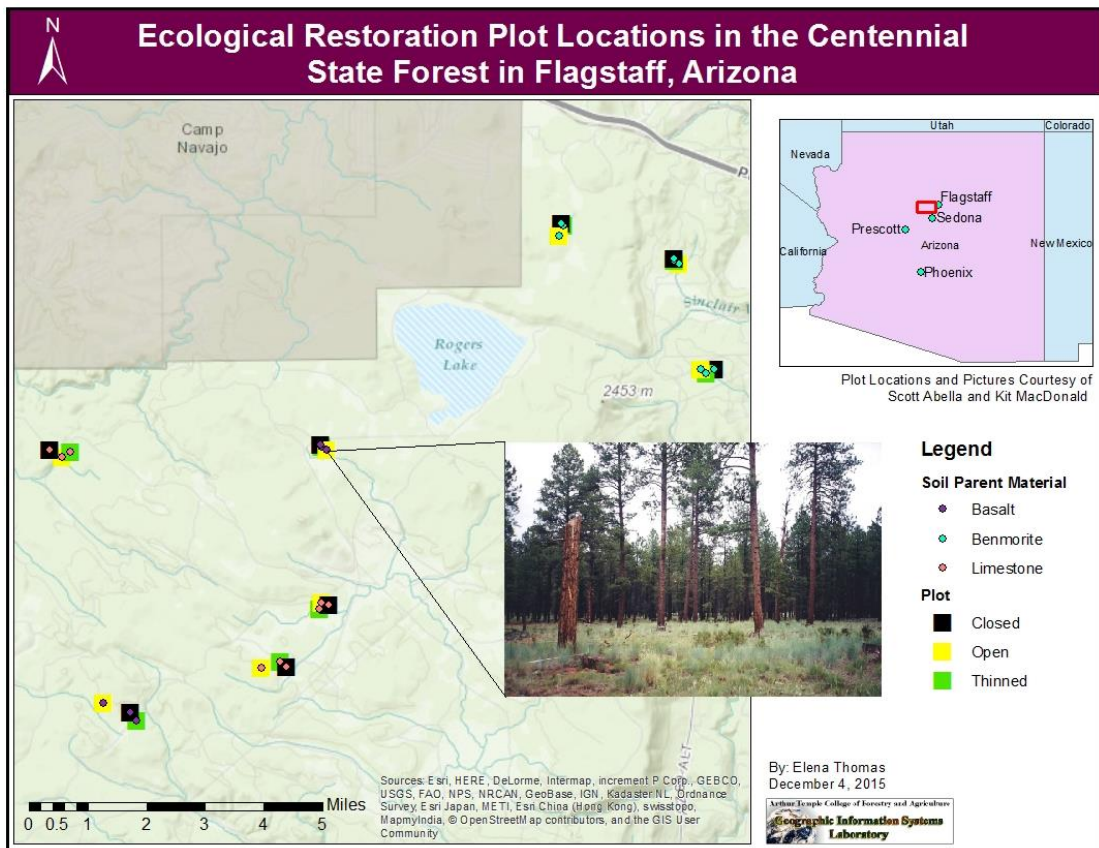
Figure 1. Site map of plot locations by parent material and treatment type in the Centennial State Forest in Flagstaff, Arizona.
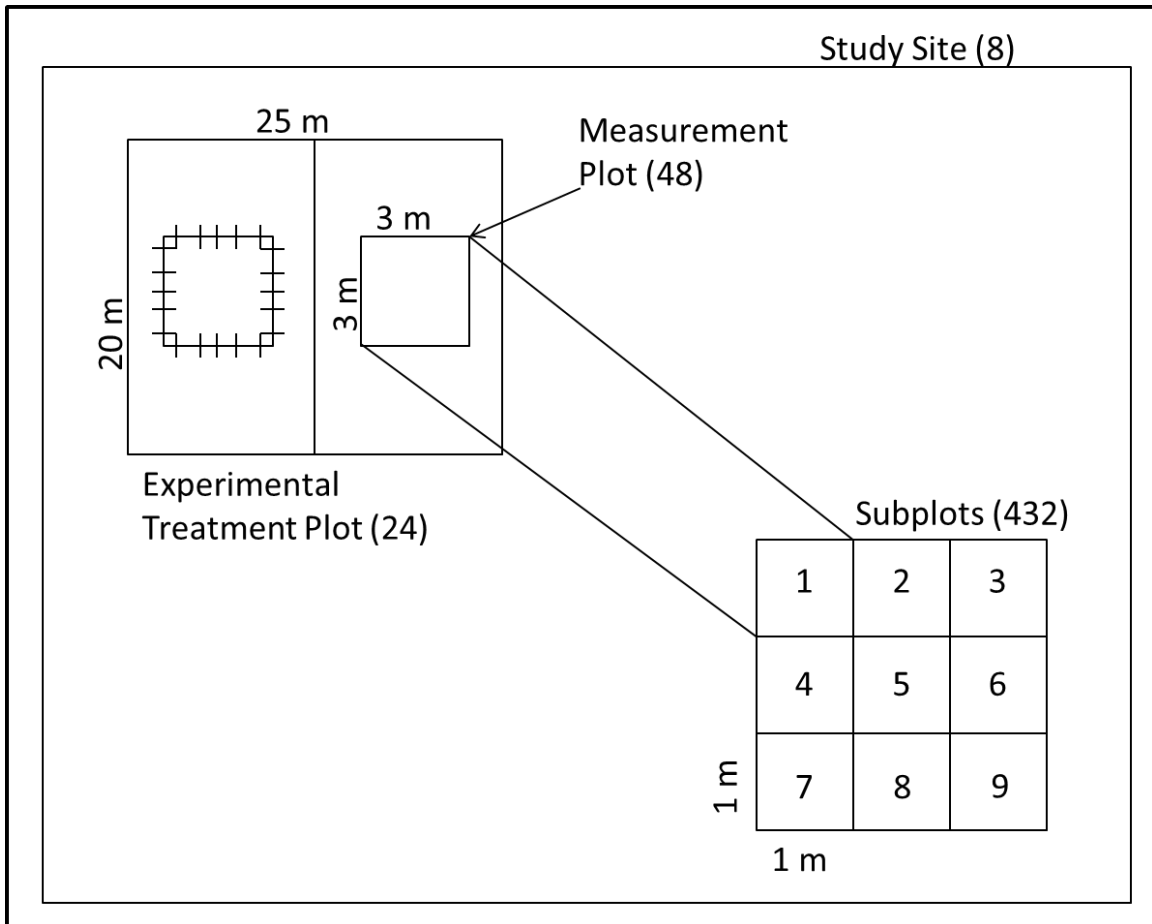
Figure 2. Diagram of study sites, each of which includes three experimental treatment plots; each experimental treatment plot has two measurement plots, which are broken into subplots. Numbers in parenthesis represent the total amount of each.

Table 2. UTM coordinates of study sites near Flagstaff, Arizona in zone 12S.

Site	UTM (X,Y)	
CF02	425095	3884455
CF04	425149	3884598
CF05	425306	3884541
CF07	424216	3883280
CF08	423789	3883149
CF10	424344	3883160
CF13	419329	3887911
CF14	419525	3888013
CF15	419051	3888079
CF16	430645	3892992
CF18	430602	3893052
CF20	430559	3892769
CF22	434020	3889750
CF23	433843	3889676
CF24	433724	3889756
CF26	433245	3892120
CF28	433144	3892192
CF30	433138	3892242
CF36	425171	3888083
CF37	425156	3888127
CF40	425297	3888021
CF42	420965	3881999
CF43	420816	3882180
CF44	420211	3882392

Experimental Treatments

Of the four treatment types in the original study, unthinned control, thinned, open, and thinned+smoke, only the first three treatment types were examined in this study. The simulated smoke treatment type was excluded in this study due to the fact that smoke had little to no effect on soil chemical properties in the first study. Each of these treatments were nested within a four-factor, balanced experimental design. Each soil parent material type had three treatments or patch types: closed (control), open, and thinned (Table 3). A grazing variable, grazed (control) and excluded, was also applied to each treatment type. Time was the final factor; with the experiment conducted there are four levels: 2003 pre-treatment, 2006 and 2008 representing three and five years post-treatment, and this study: 2015 representing 12 years post-treatment. Basalt soil parent material sites had three sites less than both benmoreite and limestone soils, due to the tornado damage.

Table 3. Experimental study site numbers and their specific soil parent material and experimental treatment type within the Centennial State Forest in Flagstaff, Arizona; each study site has a grazed and grazing exclusion measurement plot.

Basalt			Benmoreite			Limestone		
Thinned	Closed	Open	Thinned	Closed	Open	Thinned	Closed	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

Patch type of this experiment was premeditated to realistically simulate ecological restoration treatments where the natural ecological structures were maintained or re-created. This involved

allowing for wildlife diversity by leaving unthinned patches and remnant patches of open areas to remain open. Thinning was carried out as a treatment to mimic common fuel reduction strategies carried out by land managers. Tree thinning was completed using hand-operated chainsaws in September 2003, and slash was removed from plots. No mechanized equipment was driven on plots to remove or cut trees for the thinning process. The number of trees thinned resulted in 60 or 80 trees ha^{-1} remaining, which was approximately three or four trees per plot. The beginning density of these plots was about 1,362 trees ha^{-1}, and all trees that were thinned were ponderosa pine. Edge effect was also reduced by creating a buffer of 5 m around each plot. The US Forest Service, the State of Arizona, and many private land owners have grazing allotments in the area, so the experimental sites were placed in these allotments to determine the affect grazing had on the vegetative community, specifically those communities that are undergoing restoration. Many of the experimental sites were near watering holes or "tanks", so there could have been a somewhat higher grazing intensity at these locations. Generally, these higher grazed areas where less than 0.40 to 1.6 kilometers away from water sources (US. Forest Service, Coconino National Forest, Flagstaff, AZ, U.S.A.).

Experimental Plots

Soil and respiration samples were taken at each measurement plot, grazed and grazing exclusion, within the 20 m x 25 m area of each experimental treatment plot, providing five samples from the grazed area and five samples from the exclosure. Each measurement plot was divided into nine subplots (Figure 2), and the location in which each sample was taken from within the subplot was selected by a random number generator, ranging from 1 to 9.
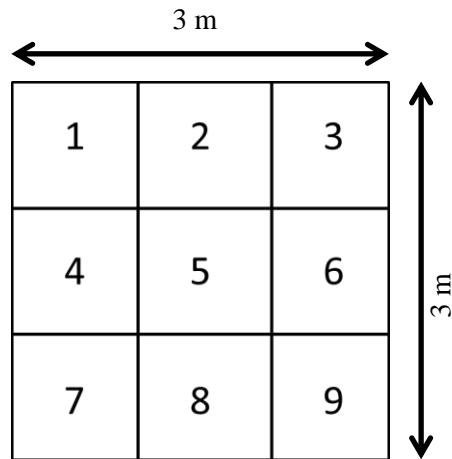
Figure 3. Experimental plot division layout used to conduct soil and vegetation sampling.

The five selected subplots for taking soil samples were then divided into five different sampling locations: the center of the subplot and the four cardinal directions. North was assigned as one, east as two, south as three, west as four, and center as five. This was done to randomly select the location within the subplot, in which the samples were taken.

Measured Soil Parameters

Parameters included soil respiration, pH, electrical conductivity (EC), moisture, temperature, nitrate and nitrite, dry and moist soil color, total carbon, total nitrogen, organic matter content (%), and soil nutrients. The procedure for soil respiration (Natural Resources Conservation Service, 1998) included the use of a 15.24 cm diameter ring that was pressed into the soil, at the randomly selected location (i.e. Subplot 7, North direction), with a rubber mallet. If the randomly selected location was covered with large coarse fragments that impeded the ring from being

driven into the ground, then a new direction was randomly generated until a suitable location was selected. Once the ring was firmly set in the soil, the heights from the soil surface to the top of the ring were measured, as the depth in which the ring penetrated varied. The ring was then covered for 30 minutes with a plastic, tight fitting lid that had installed rubber septa. The septa allowed gas flow when the two required hypodermic needles were inserted into the rubber septa, one of which was attached to the respiration apparatus. The apparatus included a 140cc syringe attached to a plastic tubing, which was attached to one end of a Dräger tube. Attached to the other end of the Dräger tube, which determined CO_2 efflux, was another plastic tubing attached to the second hypodermic needle. Once both needles were inserted into the septa, measurements were taken. If the first measurement, where the syringe was pulled from 0 to 100cc at a consistent rate of 15 seconds, read less than 0.5% on the Dräger tube, then four additional readings were taken in the same manner immediately after the first 15 second reading. Temperature (°C) was taken alongside the respiration ring with a digital soil thermometer inserted in the top 10 cm of the soil, to correct for differences from standard temperature (25°C). The soil respiration values given by the Dräger tubes was measured in ppm, but differences in atmospheric pressure and ring height in the ground needed to be corrected for. Equation 1 converts the Dräger tube output in parts per million (ppm) to grams (g) of CO_2 m^{-2} day^{-1}, and Equation 4 converts g of CO_2 m^{-2} day^{-1} to g of C-CO_2 m^{-2} day^{-1}. In Equation 3, the value 0.9866 is the pressure correction factor, adjusting for the variance from standard pressure (1013 mbar). In addition, the variable h is the average measured height from the ground to the top of the ring of the apparatus, and is measured in cm.

Equation (1):

$$ppm \times \frac{44.01 \, g \, mol^{-1}}{24.1 \, L \, mol^{-1}} \times 0.9866 \times h \times \frac{1 \, m}{100 \, cm} \times \frac{1 \, g}{1000 \, mg} \times 30 \, min \times \frac{60 \, min}{1 \, hr} \times \frac{24 \, hr}{1 \, day}$$

Equation (2):

$$\frac{g \, CO_2}{m^2 \, day} \times \frac{1 \, mol \, CO_2}{44.01 \, g \, CO_2} \times \frac{1 \, mol \, C}{1 \, mol \, CO_2} \times \frac{12.01 \, g \, C}{1 \, mol \, C} = \frac{g \, C}{m^2 \, day}$$

Soil moisture, in the top 10 cm of the soil, was determined using the gravimetric method, with samples dried at 105°C to constant weight. The field CO_2 flux calculation was then estimated with Equation 3, in which percent gravimetric water content (w) and temperature (t) are directly related to predicting soil CO_2 efflux (Wildung, et al., 1975). These two methods for determining field soil respiration were then compared.

$$CO_2 = 0.88 \pm 0.013(w)(t) \qquad\qquad (3)$$

Two different field moist soil samples were obtained with a spade to a depth of 10 cm, and placed in polyethylene lined soil sample bags. One was used in the EC, pH, and nitrate/nitrite process, and the other sample was used to calculate the mass water content (MWC), as previously mentioned. EC, pH, nitrate/nitrite, and moisture were all measured at the same time after sieving (2 mm) the soil at field moisture, for the same soil sample was used to measure each parameter. These measurements were made in the soils lab of Northern Arizona University, in Flagstaff, Arizona, within one to three days of sampling. EC, pH, and nitrate/nitrite were measured using the USDA-National Resources Conservation Service's (1998) method. Using an EC pocket

meter, 30 mL of soil was mixed with 30 mL of distilled water (a 1:1 soil to water ratio), and a measurement of electrical conductivity was taken after mixing. After 10 minutes the same sample used to measure EC was used to measure pH with a portable pH meter. Grainger 12.5 cm diameter, 25 micrometer (μm) qualitative filter paper was folded into a funnel shape and then used to filter the soil/water mixture. Once the solution had filtered through the paper, nitrate and nitrite concentrations were determined by putting a drop of the filtered water onto a nitrate/nitrite test strip. The soil wet weight was measured at Northern Arizona University in Flagstaff, Arizona, and soil samples used for the moisture corrections were placed into sealed Ziploc bags to be air dried and transported back to the Stephen F. Austin State University forest soils laboratory, where the samples were oven-dried at 105°C and dry weights determined. The soil was quantitatively transferred from the bag to a beaker to ensure all soil particles were removed from the Ziploc bag. The soil mass water content (MWC) was determined using the MWC equation (Gardiner and Miller, 2008), Equation 4.

$$MWC = \frac{wet\ weight - dry\ weight}{dry\ weight} \qquad (4)$$

Both air-dried, dry-soil and moist-soil color were determined for each sample with a Munsell Soil Color Book in a natural light setting. The total C and N, and subsequently the C:N ratio and OM content, were obtained by sending air-dried soil samples to the Stephen F. Austin State University Soil, Plant, and Water Testing Laboratory to be analyzed on a LECO C/N analyzer. In addition, other soil nutrients (K^+, Na^+, Mg^{2+}, Ca^{2+}, S, P, and B) were measured using a Meilich III extraction solution, and pH and EC were obtained with laboratory, calibrated probes. Soil pH in

the lab was found with a 1:2 soil to distilled water ratio, with 30 minutes on a reciprocal shaker and 10 minutes of rest.

Measured Vegetation Parameters

Vegetation analysis was conducted by using a 1m x 1m PVC plot frame to visually estimate percent areas of bare ground, rock and litter cover, and tree, forb, and grass cover by species for each of the nine subplots. The vegetation collected in the summer of 2015 was compared to the vegetation data collected by Abella et al. (2015) in the previous study. They conducted vegetation measurements pre-treatment (September 2003) and post-treatment (2006 and 2008). A companion study was started to measure "warm-season" vegetation, as well as soil physical properties, but was not completed at the time of this writing.

Statistical Analysis

A partial nested mixed model analysis of variance (ANOVA) model was used to analyze total species richness per measurement plot, total species richness per subplot, total percent cover of understory vegetation, as well as lifeforms (i.e. grasses and forbs), exotic species, and longevity types, responses to applied restoration treatments, and included the four factors: soil type, treatment type, grazing, and time. Similarly to Abella et al. (2015), there were 72 combinations plus interactions generated (3 soil types x 3 patch type x 2 grazing treatments x 4 measurement years) to reduce the copious multiple comparisons that were generated from a 4 factor interaction. These combinations were tested, at $\alpha = 0.10$, and focused on how effects of the treatment combinations changed throughout time.

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A three-factor experimental design mixed model ANOVA was used to analyze the soil parent material, treatment, and grazing treatment type effects on vegetation and select soil properties measured for this study. Type III fixed effects that were significant for both vegetation and soil data, at a 0.10 significance level, were analyzed further to compare means. Unadjusted comparisons of differences of least square means were used to determine significantly different means between significant fixed effects. In addition, soil respiration was analyzed using a regression model in SAS, where soil moisture and coarse fragment content acted as influencing variables (SAS Institute, 2009) to determine if these parameters acted as covariates in influencing the obtained soil respiration.

RESULTS AND DISCUSSION

Study Area

In 2014, the year before this study was conducted, the Flagstaff area received approximately 52.9 cm of precipitation, and by the beginning of this study (June 4, 2015) the area had already received about 27.8 cm of precipitation for the year. During the study months (June and July 2015) there was an accumulation of 11.4 cm of precipitation, which is 3.8 cm above the normal accumulation levels for those two months. The January low for 2015 was recorded at -6.1°C, and the July high was 25.3°C (NOAA, 2014). Daily weather data for the Flagstaff area in June and July are displayed in the Appendix in Table A.1 and Table A.2.

Restoration Treatment Effects

Soil Respiration

Soil respiration rates were affected by soil type (p= 0.0846). Limestone soils had significantly higher respiration rates than benmoreite soils (Table 4), while basalt derived soils were not significantly different than either other soil parent material type (Figure 4). The soil respiration rates were plotted per soil type, with restoration treatments acting as the independent variable, to assess the validity of the significant three-way interaction (p = 0.0447), (Figure A.1). Grazing had the least influence on basalt soils among all restoration treatments examined. Litter might not have been a driving force in soil respiration differences in this study, as litter percent

cover was only significantly different among treatment types (p= 0.0231), with open and thinned canopies being significantly higher than closed canopies. It is likely due to the small canopy sizes of the open treatments (0.01 to 0.1 ha), that both the thinned and open treatments are receiving similar amounts of litter inputs. However, litter coverage could be indirectly influencing other soil properties that will eventually influence soil respiration rates, such as soil temperature or moisture. Soil texture has been found to influence soil respiration rates (Bouma and Bryla, 2000), but as the texture determinations for limestone and basalt derived soils were loam, and benmoreite silt loam, it does not seem that texture influenced soil respiration rates at these study sites.

Soil respiration is a combination of soil microorganisms and soil macroorganisms conducting respiration and plant root aerobic respiration. As neither species richness nor plant cover was significantly different among soil parent material types, and soil respiration was affected by soil type, it can be assumed root respiration alone was not a factor in influencing measured respiration rates. Instead, soil respiration was most likely being influenced by soil macro- and microorganisms and environmental factors, or any combination thereof.

Open treatment canopies were only 0.01 (100 m^2) to 0.1 ha in size (Abella, et al., 2015), so pine needle inputs were probably still occurring. With slow to decompose pine needles still present under all treatment type canopies, it is possible that treatment type did not influence soil respiration because of similar organic substrate inputs. It could also be possible that since soil respiration rates were taken early in the growing season, fine root production had not yet reached its peak, so soil respiration rates measured in this study period were probably lower than if taken later in the growing season (Laughlin, et al., 2011). Additionally, Laughlin et al. (2011) found as ponderosa pine basal area increased there was a shift towards herbaceous understory vegetation

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that exhibited lower amounts of fine roots and smaller leaf surface area. This shift could alter the soil respiration rates, especially if measurements were taken early in the growing season, as aforementioned.

Soil respiration rates reported by other authors are similar to the rates reported in this study, but other reported respiration rates were generally measured annually or in the summer and fall. As this study was only conducted in June and July it is possible the measured soil respiration rates are higher than if reported on an annual basis. In a similar study near Flagstaff, Arizona, a ponderosa pine-bunchgrass ecosystem reportedly had soil respiration rates between 1 to 3 g C-CO_2 m^{-2} day^{-1} (Kaye and Hart, 1998). Additionally, from regression analysis they also concluded that soil moisture content was not influencing soil respiration rates, as soil water content was only accounting for 20% of the variability, and that it could be soil temperature that was driving soil respiration rates found. Raich and Schlesinger (1992) also reported mean annual soil respiration rates to be 681 ± 95 g of C m^{-2} yr^{-1} in temperate coniferous forests.

With soil moisture not significantly different between treatment types, decomposition rates may be similar, thus the latency period for these organic substrates is likely the same. The time frame in which the soil respiration rates were measured could also influence the amount of CO_2 released. Fine roots may not yet be respiring at optimal level since readings were taken early in the growing season. Additionally, it is possible that the full extent of the vegetation present on these sites was not quite seen during the sampling time. Having sampling measurements throughout the early and late growing season would be more beneficial to fully elucidate the underlying soil community.

When MWC and soil temperature were put into the CO_2 efflux equation described by Wildung et al. (1975), there were no significant differences among soil type, treatment type, or

grazing treatment. The differences ranged from absolute values of 0.0016 to 8.0850 from

calculated values from Dräger tubes. This may suggest there are more factors influencing soil

respiration rates at these study sites than soil moisture and temperature. It may also suggest some

sites are more reliant on soil moisture and temperature, which can be altered by soil texture and

bulk density, parameters that were not measured in this study, but will be investigated in a

companion study.

Table 4. Means and standard deviations for select soil properties across soil parent material types and restoration treatments. Respiration samples were taken to a depth of 4 to 12 cm, and soil samples were taken to a depth of 10 cm.

	Limestone	Benmoreite	Basalt
Respiration (g C m^{-2} day^{-1})			
Open	3.32 (0.813)	2.51 (0.963)	2.95 (1.121)
Thinned	3.44 (0.785)	2.65 (0.27)	2.90 (0.753)
Closed	3.19 (1.024)	2.31 (0.781)	2.31 (0.615)
Mass Water Content (%)			
Open	19.11 (17.233)	19.68 (14.443)	18.96 (8.493)
Thinned	22.16 (13.899)	22.85 (21.146)	36.28 (17.535)
Closed	24.37 (11.676)	22.53 (15.054)	27.5 (18.539)
Temperature (^{o}C)			
Open	18.62 (1.919)	20.78 (4.652)	19.12 (3.363)
Thinned	15.13 (1.992)	18.22 (1.800)	16.99 (1.520)
Closed	13.11 (1.830)	14.07 (2.324)	19.26 (3.760)
Coarse Fragments (g)			
Open	208.09 (109.794)	367.95 (250.019)	466.97 (236.593)
Thinned	135.23 (93.183)	473.31 (378.539)	436.59 (271.858)
Closed	144.08 (183.133)	282.23 (265.051)	486.55 (448.881)
pH- lab			
Open	5.915 (0.25)	6.154 (0.37)	6.113 (0.23)
Thinned	5.740 (0.30)	5.952 (0.33)	5.981 (0.27)
Closed	6.031 (0.27)	5.679 (0.60)	6.393 (0.18)
Electrical Conductivity			
Open	140.728 (41.58)	155.269 (63.16)	127.395 (49.22)
Thinned	194.013 (58.71)	165.973 (68.92)	155.140 (49.59)
Closed	221.457 (106.30)	175.753 (70.28)	135.510 (47.22)
C (%)			
Open	5.14 (2.203)	8.17 (7.194)	4.12 (1.278)
Thinned	4.31 (1.934)	8.02 (5.812)	6.82 (5.844)
Closed	4.21 (2.783)	7.50 (7.504)	8.45 (8.459)
N (%)			
Open	0.33 (0.095)	0.46 (0.314)	0.29 (0.067)
Thinned	0.30 (0.062)	0.45 (0.251)	0.67 (1.456)
Closed	0.29 (0.118)	0.38 (0.237)	0.40 (0.184)
C:N			
Open	6.93 (1.418)	6.66 (1.678)	7.29 (0.650)
Thinned	7.46 (1.127)	6.45 (1.506)	6.46 (1.355)
Closed	7.44 (1.016)	6.22 (1.521)	6.12 (1.778)
OM (%)			
Open	10.27 (4.407)	16.34 (14.389)	8.24 (2.555)
Thinned	8.61 (3.868)	16.04 (11.625)	11.13 (4.887)
Closed	8.42 (5.566)	15.01 (15.008)	16.89 (16.918)
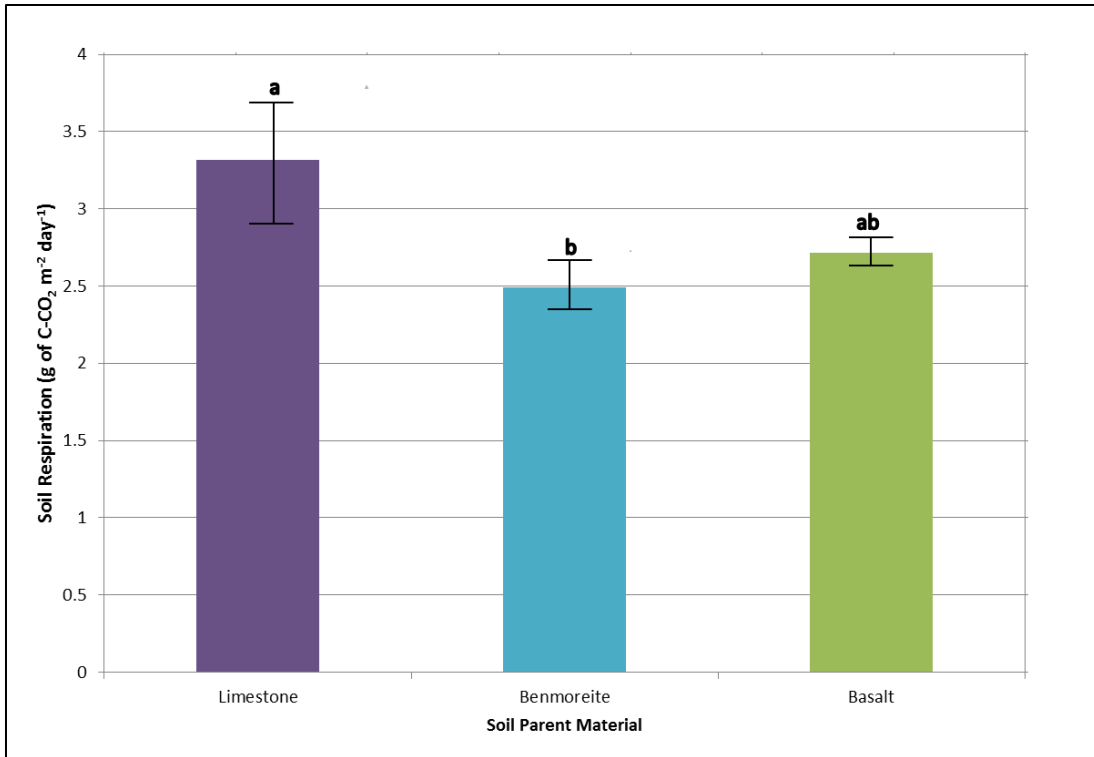
Figure 4. Soil respiration rates by soil parent material, letters that are the same are not statistically different ($\alpha = 0.10$). Each bar represents one standard deviation away from the mean of soil respiration.

Soil Temperature

Soil temperature was significantly different among treatment types. Soil temperature was significantly higher in the open canopy treatments (19.6°C) than in thinned (16.8°C) and closed canopy treatments (15°C). The ANOVA table for significant effects on soil temperature and other significantly affected soil properties are shown in Table A.3. As Ma et al. (2005) found, there were correlations between canopy gap size and soil temperature. Small canopies at the study sites, or open canopies, ranged from 15 to 30°C, while closed, dense canopies ranged from 10 to 26°C. The general trend between soil moisture and temperature was as soil temperature

decreased, soil moisture increased (Figure 5); however, the soil moisture differences were not significant. The trend could be due to the shading effect the canopies provide as they get denser from open to closed canopies, which reduces temperature and therefore evaporation. It is surprising that the significantly higher soil temperatures in the open canopy did not cause any significant differences for MWC, but this could be because the decreased evaporation rates under the closed canopies were offset by increased transpiration rates from the tree crowns.

Mass Water Content

When MWC and soil respiration were plotted together, it was determined that soil moisture did not affect the respiration rates per treatment type. For example, the thinned and closed canopies had similar MWCs, but the soil respiration under the closed canopy was significantly less than the thinned canopies. So, decomposition of organic matter due to increased soil moisture content is also likely not a contributing factor to soil respiration rates at these study sites. There were no significant differences in MWC values for any effect tested. MWC was used to express soil moisture content, since bulk density work was part of an uncompleted companion study on soil physical properties.

The significance of soil moisture (MWC) and coarse fragment content in influencing soil respiration was analyzed using regression. Forward selections were made to determine if one or more parameters acted as covariates. For the limestone and benmoreite soil parent material types no model had significant results (Table 5). However, for basalt soil parent material types, when both MWC and coarse fragment content were included in the regression, a significant model and covariate was found. The model had a significance level of p= 0.0409, and the significant

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covariate was MWC (p= 0.0327), at α= 0.10. When viewed in a model by itself, MWC was also

a significant covariate (p= 0.0531) for respiration rates on basalt soil parent material types.

Table 5. P values found using regression procedure in SAS (2009) for the calculated soil
respiration model and each parameter given tested at a significance level of α= 0.10.

	Basalt	Limestone	Benmoreite
Model	*0.0409	0.3701	0.9295
Intercept	*< 0.0001	*< 0.0001	*< 0.0001
MWC (%)	*0.0327	0.3102	0.8804
Coarse Fragments (g)	0.1026	0.2664	0.7464

* indicates a significant value at α= 0.10 level
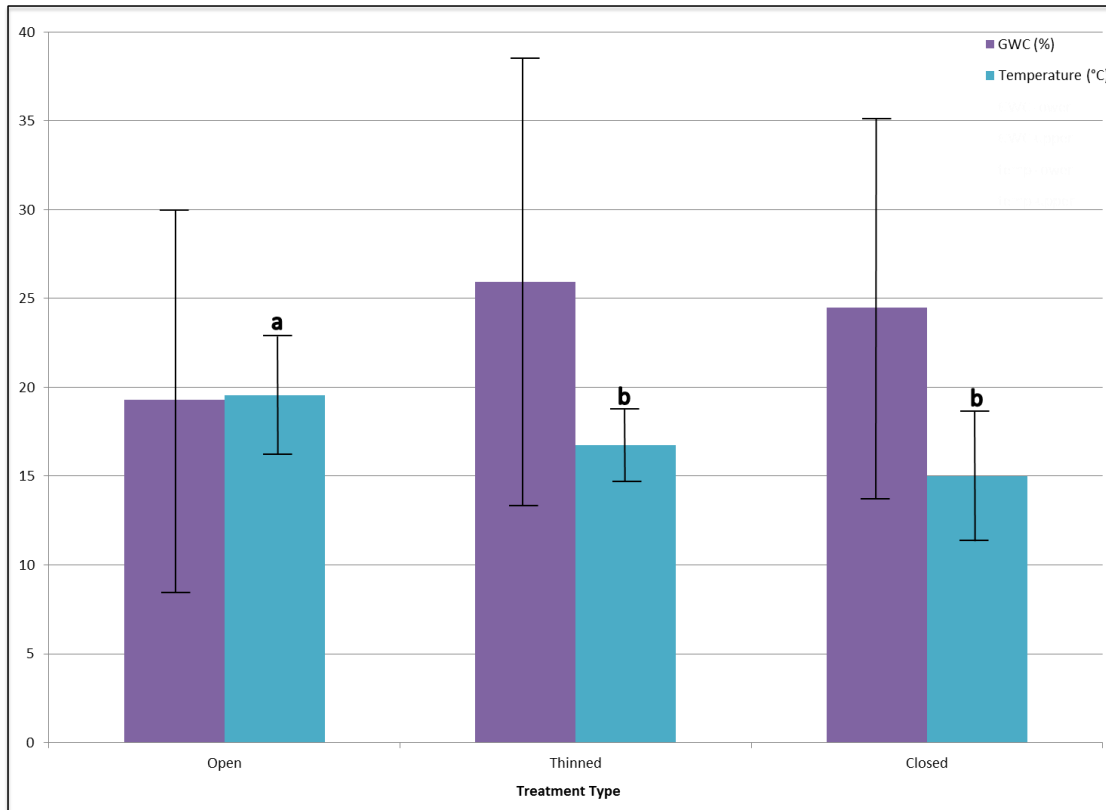
Figure 5. Comparison of soil temperature and soil moisture as canopy cover increased. Bars with the same letter are not significantly different ($\alpha= 0.10$). Each bar represents one standard deviation away from the mean.

Coarse Fragment Content

The soil coarse fragment content may have played an indirect role in soil respiration rates. Coarse fragment content significantly varied between soil types, with basalt soils having significantly higher amounts of coarse fragments than limestone soils (Figure 6). Limestone soils had the greatest soil respiration rates, which could be due to lower coarse fragment content in the soil, allowing for higher soil moisture retention and greater gas exchange pathways during dry

periods (Brady and Weil, 2002). Also, with less coarse fragment content there is greater soil

volume for microbes and roots to occupy.
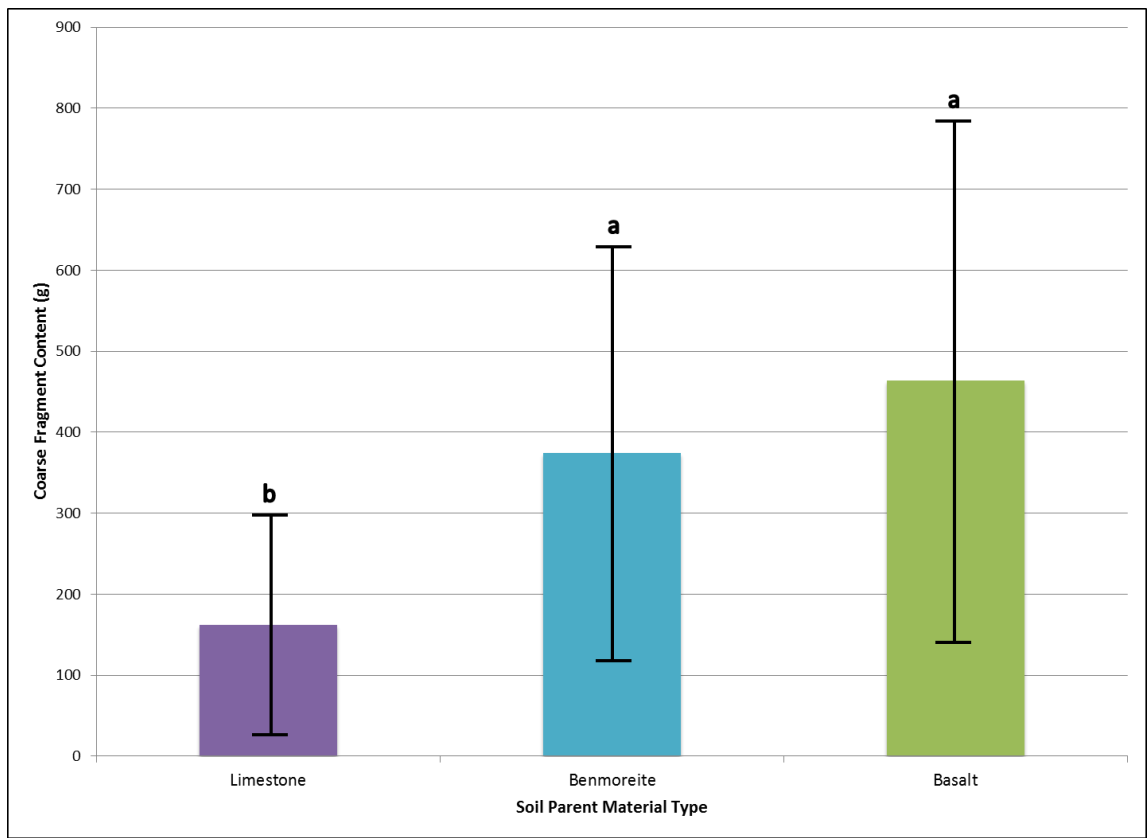
Figure 6. Coarse fragment content by soil parent material type, letters that are the same are not statistically significant ($\alpha=0.10$). Each bar represents one standard deviation away from the mean. Whole soil volumes were approximately the same.

Soil pH

Soil pH, when tested in the lab with a pH electrode, was found to be more acidic than field

measured values. Soil pH found in the lab was similar to that of Abella et al. (2015) and more

reproducible than soil pH's found with the field probe, so pH values used were those found in the

lab. Soil pH was expressed as hydrogen ion concentration for statistical analysis. There were no significant effects found on both field and lab measured pH levels, which coincides with findings from multiple studies (Grady and Hart, 2006; Gundale, et al., 2005; Boerner, et al., 2007; Johsnon, et al., 1991). Although pH was not significantly different among soil parent material types, pH values were more acidic in nature then one would expect for soils derived from limestone and basalt. Benmoreite, however, is a source for lateritic soils and aluminum oxides (Arculus and Gust, 1995), which might explain the low pH values found on measurement plots of benmoreite soil parent material types.

Measurement year was also included in a separate statistical procedure, and it was found that average hydrogen ion concentrations were not significantly different between treatment years at the 0.10 significance level. Data for soil pH at the beginning of this long-term study was obtained by Abella et al. (2015) in 2003. The results of this study differs from a ponderosa pine-bunchgrass ecosystem studied by Kerns et al. (2003), who found transition plots (similar to thinned treatment plots) to be significantly higher in soil pH than the old growth plots (similar to the closed treatment plots).

Soil Macronutrients

Soil calcium (Ca^{2+}) was significantly affected by an interaction between treatment type and grazing treatment (p= 0.0482), in which closed treatments under grazed conditions exhibited lower concentrations than open treatments under grazed conditions (Figure 7). There is not a clear reason why this is. There also was a significant three-way interaction (p= 0.0519), where closed canopies under grazed conditions on benmoreite soils exhibited lower calcium

concentrations than the other two soil types (Figure A.2). Grazed treatments under thinned canopies on basalt soil parent material types displayed higher calcium concentrations than any other basalt treatment and grazing combination.

Ca^{2+} and Mg^{2+}, the main elemental components of dolomite (Brady and Weil, 2002), are similar in concentration when comparing soil types (Table 6). Magnesium (Mg^{2+}) concentrations in the soil were also significantly affected by a treatment type and grazing treatment interaction, as well as a three-way interaction. It is surprising that basalt soils did not have significantly higher concentrations of Mg^{2+} than both limestone and benmoreite soil parent material types. Basalt is largely composed of Mg^{2+} and iron rich minerals such as olivine (Encyclopedia Britannica eds, 2011), which can act as a large source of Mg^{2+} in soils, while calcareous parent materials, such as limestone, are intrinsically low in Mg^{2+} concentrations (Havlin, et al., 2014).

As Ca^{2+} is present on most exchange sites of calcareous soils (limestone), it is interesting that the limestone derived soils did not exhibit significantly higher concentrations of Ca^{2+}. However, it is possible that the limestone parent material soils were more a derivative of dolomitic limestone, due to the higher magnesium concentrations. However, it is likely that the soils in this study are highly weathered, as evidenced by the relatively low pH, and it is possible that the parent material is now at a great depth and is only playing a diminished role in influencing the above soil and various soil properties.

Table 6. Means and standard deviations for soil macronutrients and elements extracted with Meilich III solution across soil parent material types and restoration treatments. Soil samples taken to a depth of 10 cm.

	Limestone	Benmoreite	Basalt
Ca^{2+} (mg kg^{-1})			
Open	1877.35 (355.244)	2096.2 (519.744)	1827.36 (678.491)
Thinned	1564.63 (404.329)	1764.22 (359.655)	2037.03 (539.255)
Closed	1800.21 (385.263)	1462.69 (620.556)	2073.21 (464.982)
Mg^{2+} (mg kg^{-1})			
Open	299.40 (71.792)	373.06 (124.107)	447.03 (244.253)
Thinned	289.09 (76.429)	337.09 (92.945)	468.88 (275.319)
Closed	432.07 (82.061)	262.99 (141.597)	516.58 (148.908)
K^+ (mg kg^{-1})			
Open	196.94 (79.630)	171.31 (30.404)	192.40 (37.873)
Thinned	149.66 (65.044)	131.52 (71.356)	162.30 (49.943)
Closed	129.41 (28.222)	192.73 (55.373)	132.68 (36.025)
P (mg kg^{-1})			
Open	54.51 (38.872)	33.94 (11.663)	18.12 (5.000)
Thinned	35.28 (28.626)	33.17 (23.028)	18.61 (12.673)
Closed	18.97 (14.958)	48.02 (18.565)	11.24 (3.840)
Na^+ (mg kg^{-1})			
Open	50.67 (5.487)	56.19 (9.085)	49.13 (5.143)
Thinned	49.16 (9.055)	53.09 (7.633)	49.23 (8.844)
Closed	50.34 (8.689)	53.03 (10.253)	47.35 (5.018)
S (mg kg^{-1})			
Open	6.34 (1.640)	5.38 (2.337)	4.85 (1.220)
Thinned	6.90 (1.503)	5.01 (1.884)	4.89 (1.106)
Closed	6.03 (2.094)	5.66 (1.755)	3.90 (1.467)
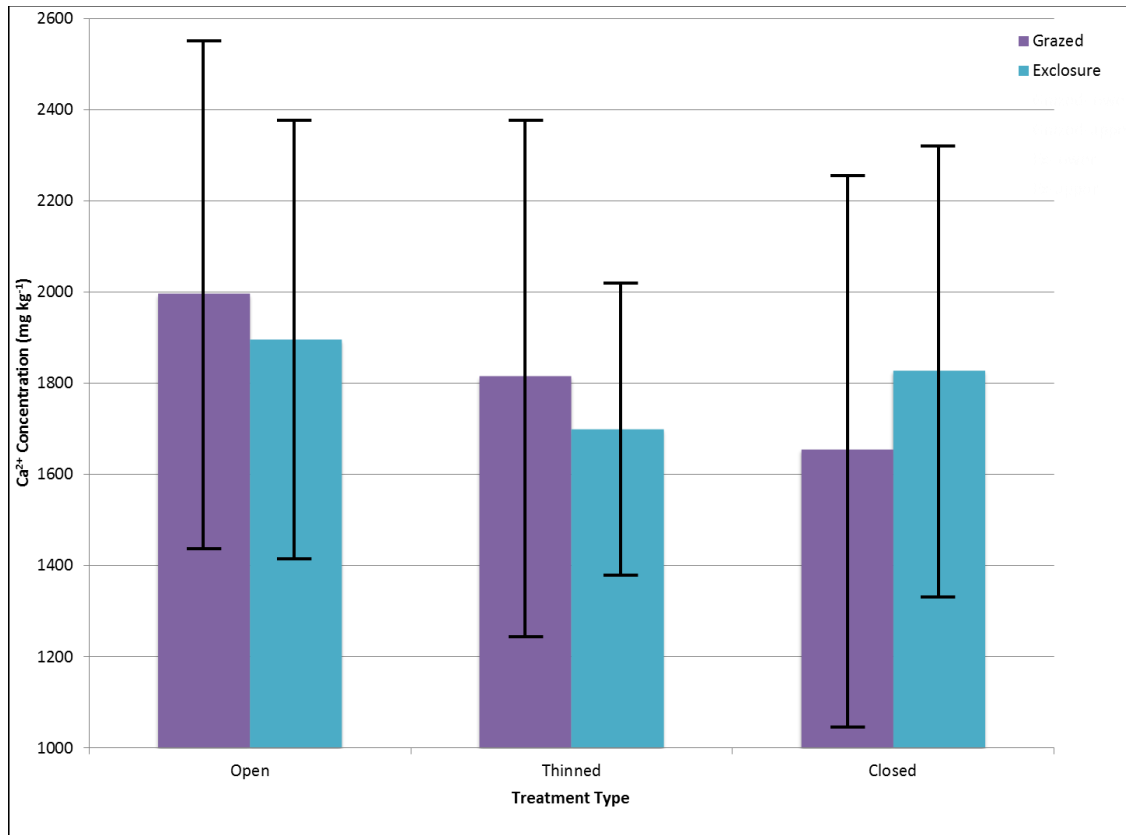
Figure 7. Interaction effect between treatment type and grazing treatment on calcium (p=0.0482), tested at α=0.10 significance. Each bar represents one standard deviation away from the mean.

Mg^{2+} soil concentrations were highly influenced by an interaction between treatment type and grazing treatment (p= 0.0084), and were also significantly affected by a three-way interaction (p= 0.0165). Under a thinned treatment type and grazing exclusion Mg^{2+} concentrations were lowest, but were highest under closed treatment types with grazing exclusion. Open canopies had the least derivation between grazing treatments regarding Mg^{2+} concentrations (Figure 8). Magnesium concentrations showed the same general trend on limestone soils with increasing canopy cover under both grazing treatments (Figure A.3). Contrastingly, Mg^{2+} concentrations on

benmoreite soils generally decreased as canopy cover increased; while there was no general trend in Mg^{2+} concentrations on basalt soils. Thinned canopies on basalt soils, however, displayed lower Mg^{2+} concentrations when grazing exclusion was applied compared to grazing. Sodium (Na^+) was only significant under a three-way interaction between soil type, treatment type, and grazing treatment, but that interaction was too significant to ignore (p= 0.0374) and cannot be explained.
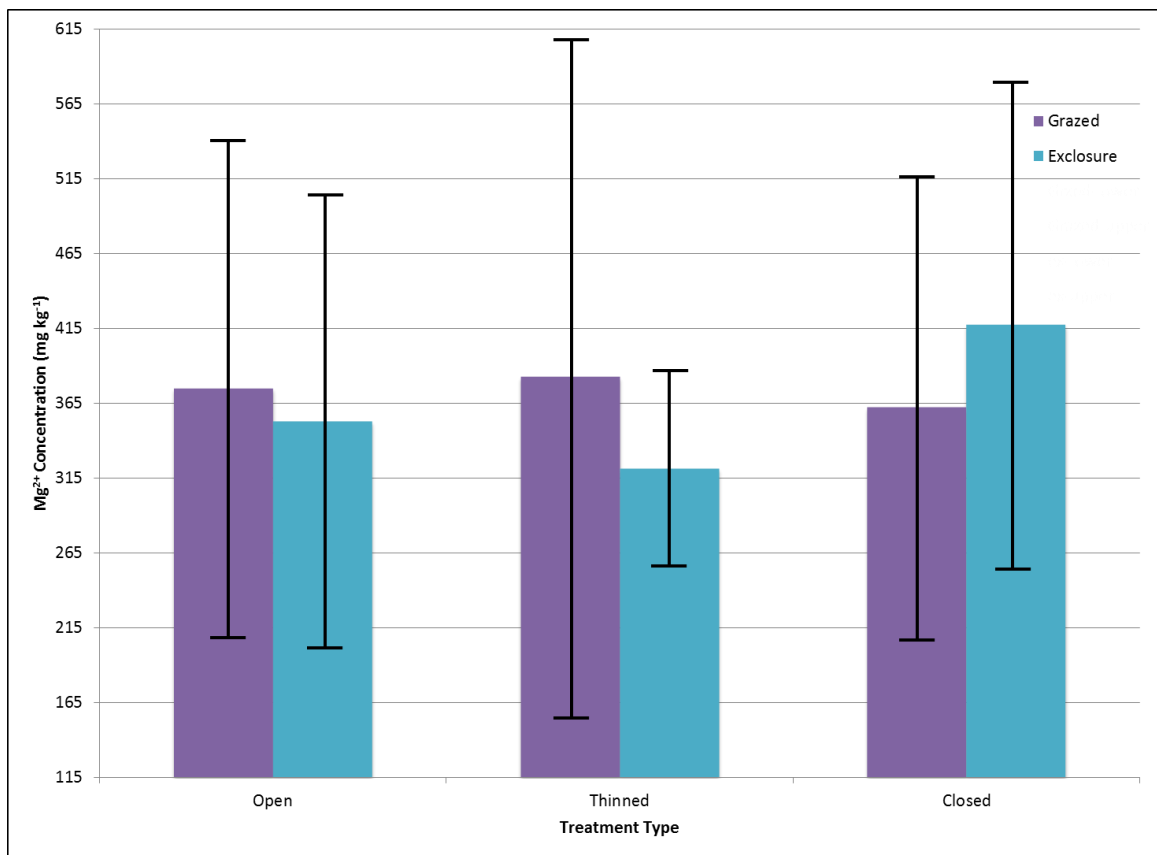
Figure 8. Interaction effect of treatment type and grazing treatment on magnesium concentrations (p= 0.0084), tested at α= 0.10 significance. Each bar represents one standard deviation away from the mean.

Boerner et al. (2007) found soil Ca^{2+}, P, and K^+ to not be significantly affected by restoration treatments, Gundale et al. (2005) found that restoration treatments had no significant effect on exchangeable cations, such as Ca^{2+}, Na^+, K^+, Mg^{2+}, or soil P, and Johnson et al. (1991) found exchangeable cations (Ca^{2+}, Mg^{2+}, and K^+) to have no significant change in the A horizon after whole-tree harvesting. This study's findings concur with those three studies, but Boerner et al. (2007), Gundale et al. (2005), and Johnson et al. (1991) did not include a grazing component, which was found to interact with restoration treatments to significantly affect Ca^{2+} and Mg^{2+} concentrations. In addition, Gundale et al. (2005) did not mention the soil parent material, and the soils had differing taxonomic classifications. As there were many soil type, plot type, and grazing treatment interactions for almost all of the above mentioned elements, it is important to note the similarities and differences between soil types.

Soil Carbon, Nitrogen, and Organic Matter

Percent soil carbon (%C) and total nitrogen (%N) were not significantly different in this study. Gundale et al. (2005) claimed that ponderosa pine ecosystems are N limited, and the low concentrations could be due to microorganisms quickly scavenging any available N that has been released into the soil profile in such an ecosystem. In addition, available N concentrations, which were not measured in this study, could be lower from lack of recent fire in these ecosystems, as fire rapidly increases available N in an ecosystem. As Vesterdal et al. (1995) indicated, soil properties such as C, N, and P are related more to site-dependent properties and not restoration treatments, which the data from this study supports. C:N ratios for the study sites ranged from three to nine, but were not significant among treatment types. If C:N from the O horizon had

been measured, we would have a better understanding of the nature of the organic/litter substrates present. Nitrate and nitrite were measured with test strips, but were undetectable in the top 10 cm of the mineral soil. It would be desirable to have fire effects examined in a similar study, suitable conditions allowing, so the true extent of soil chemical and biological processes can be examined.

The concentration of soil organic matter (OM) was also found to be unaffected by soil parent material type, restoration treatment type, or grazing treatment. Similarly, to Gundale et al. (2005) and this study, Zahawi et al. (2015) found no significant differences among the control and restoration treatments carried out for either %N, %C, P, or OM concentrations.

Other Soil Parameters Not Significantly Affected

Soil electrical conductivity (EC) was not significantly affected by soil parent material type, treatment type, grazing treatment, or any interaction including the aforementioned parameters. Sulfur (S) and phosphorous (P) soil concentrations were also not significantly affected by any of the effects tested, and soil boron (B) was non-detectable in the soil samples.

Grazing Treatment Effects

Soil Respiration

Soil respiration rates were only influenced by grazing in a three-way interaction (p= 0.0447). The basalt soil parent material did not appear to vary much in soil respiration rates under grazing and grazing exclusion across treatment types (Table 7), but under the grazing treatments, on both

limestone and benmoreite parent materials, it appeared that soil respiration rates varied under different canopy cover types (Figure A.1). For example, as canopy cover increased on benmoreite soil parent material undergoing grazing exclusion, soil respiration rates increased from an open to intermediate (or thinned) canopy cover, but then slightly decreased under an increased canopy cover (closed control). However, on limestone parent material soils undergoing grazing exclusion, the soil respiration rates decreased from an open to an intermediate canopy cover and slightly increased under a high canopy cover.

Soil Temperature and Mass Water Content

Soil temperature was significantly reduced by grazing ($p= 0.0347$), with temperatures averaging 17.8°C under grazing exclusion and 16.4°C under grazed treatments. The higher temperatures found in the exclosures could be from an increase in the observed litter layer that creates an insulating effect (Clapperton, et al., 2002), reducing re-radiation of heat energy during net-cooling seasons. However, soil MWC was not significantly different for grazing treatments. Indirectly, grazing could be affecting soil respiration rates due to its effect on soil temperatures.

Coarse Fragment Content

Coarse fragment content was also significantly different by grazing treatment, but this significance is most likely happenstance.

Table 7. Means and standard deviations for select soil properties across restoration and grazing treatments. Respiration samples were taken to a depth of 4 to 12 cm, and soil samples were taken to a depth of 10 cm.

	Grazed	Excluded
Respiration (g C m^{-2} day^{-1})		
Open	2.93 (0.774)	2.92 (0.831)
Thinned	3.11 (0.711)	2.91 (0.576)
Closed	2.60 (0.759)	2.68 (0.730)
Mass Water Content (%)		
Open	19.02 (15.792)	19.55 (12.717)
Thinned	25.11 (17.688)	26.79 (19.617)
Closed	24.34 (13.708)	24.59 (16.031)
Temperature (°C)		
Open	18.69 (3.405)	20.42 (3.622)
Thinned	16.08 (1.660)	17.43 (2.550)
Closed	14.54 (2.941)	15.48 (4.126)
Coarse Fragments (g)		
Open	208.09 (109.794)	367.95 (250.019)
Thinned	135.23 (93.183)	473.31 (378.539)
Closed	144.08 (183.133)	282.23 (265.051)
pH- lab		
Open	6.065 (0.31)	6.041 (0.32)
Thinned	5.894 (0.37)	5.866 (0.26)
Closed	5.909 (0.58)	6.070 (0.38)
Electrical Conductivity (μS/cm)		
Open	144.936 (49.46)	140.501 (56.37)
Thinned	173.300 (56.30)	174.260 (68.29)
Closed	179.668 (83.04)	185.995 (92.62)
C (%)		
Open	5.54 (3.343)	6.44 (6.013)
Thinned	6.49 (5.470)	6.16 (4.476)
Closed	7.70 (6.957)	5.31 (6.158)
N (%)		
Open	0.35 (0.140)	0.39 (0.263)
Thinned	0.52 (1.038)	0.38 (0.194)
Closed	0.39 (0.228)	0.31 (0.132)
C:N		
Open	6.78 (1.044)	7.05 (1.657)
Thinned	6.75 (1.481)	6.91 (1.345)
Closed	6.15 (1.456)	7.15 (1.466)
OM (%)		
Open	11.08 (6.703)	12.87 (12.026)
Thinned	11.73 (8.043)	12.33 (8.953)
Closed	15.40 (13.913)	10.62 (12.316)

Soil Macronutrients

Under grazing exclusion conditions soil K^+ was significantly higher than under grazed conditions (Table 8). This does not coincide with findings by Smoliak et al. (1972), who found no significant differences in soil K^+ concentrations under different grazing conditions. However, Smoliak et al. (1972) found that heavy grazing conditions will cause herbaceous plants to allocate more biomass to belowground roots, when compared to ungrazed or light grazed areas. This could explain why there are higher concentrations of soil K^+ in grazing exclosures of this study. Fendler's buckbrush (*Ceanothus fendleri* Gray), which was observed more often in grazing exclosures, could also be facilitating the increase in soil K^+ through scavenging K^+ from deeper in the soil with its extensive and deep root systems. The higher concentrations of K^+ in the soil could also be attributed to greater amounts of herbaceous plants aboveground; as the plants die they release stored K^+ back into the soil. Soil texture plays a role in presence of K^+ in the soil (Havlin, et al., 2014), but as the textures varied little between soil types, texture is likely not influencing soil K^+ concentrations. Soil moisture was not significantly affected by grazing, but soil temperature was significantly higher under grazing exclusion conditions. Thus it is more likely that vegetation was allocating more biomass aboveground than belowground, creating higher K^+ concentrations present in the soil, and it has been found that K^+ concentrations in the soil are directly related to biomass productivity in eastern Arizona (Naumburg and DeWald, 1999). Soil K^+ was also significantly influenced by a soil parent material type, treatment type, and grazing treatment interaction (p= 0.0533).

Table 8. Means and standard deviations for soil nutrients extracted with Meilich III solution across restoration and grazing treatments.

	Grazed	Excluded
Ca^{2+} (mg kg^{-1})		
Open	1996.04 (558.947)	1895.30 (479.11)
Thinned	1815.53 (564.468)	1699.62 (320.352)
Closed	1655.79 (601.134)	1828.00 (494.321)
Mg^{2+} (mg kg^{-1})		
Open	375.00 (166.632)	352.91 (153.169)
Thinned	382.62 (227.465)	321.45 (65.246)
Closed	362.16 (155.697)	417.42 (164.003)
K^+ (mg kg^{-1})		
Open	178.46 (49.586)	194.12 (61.894)
Thinned	139.19 (61.571)	152.85 (67.589)
Closed	139.27 (39.252)	168.67 (58.072)
P (mg kg^{-1})		
Open	33.68 (18.269)	41.72 (36.057)
Thinned	30.50 (23.602)	30.14 (24.906)
Closed	27.67 (22.079)	28.18 (21.384)
Na^+ (mg kg^{-1})		
Open	52.76 (7.749)	51.87 (7.351)
Thinned	51.47 (9.160)	49.83 (8.027)
Closed	49.90 (10.012)	51.31 (7.436)
S (mg kg^{-1})		
Open	5.49 (1.927)	5.72 (1.935)
Thinned	5.59 (1.610)	5.79 (2.027)
Closed	4.95 (1.938)	5.77 (2.002)

Soil Parameters Not Significantly Affected

Soil pH was not significantly affected by grazing treatments, which differs from findings by Johnston et al. (1970), Somliak et al. (1972), and Yimer et al. (2008). Soil EC, %C, %N, OM, and C:N ratio were not significantly affected by grazing treatment, or any interaction including grazing, as well as soil P and S concentrations. Johnston et al. (1970), like this study, found soil

P to not be significantly affected when grazing treatment only was analyzed. They also found that soil N and OM content were not significantly different among grazing treatments, which corresponds to this study. However, Neff et al. (2005) found soil C and N to be significantly higher in never grazed treatments compared to historically grazed treatments, and Smoliak et al. (1972) found higher C:N ratios under grazing than no grazing treatments.

<div align="center">Restoration Treatment Effects on Vegetation</div>

<u>Cover</u>

Percent cover per measurement plot ($9m^2$) was significantly affected by treatment type (p= 0.0418), with open canopies having significantly (p= 0.0158) higher percentages than closed canopies (Figure 9). The ANOVA table for significant vegetation effects can be found in the Appendix (Table A.4). In addition, treatment type significantly affected the percent grass cover (p= 0.0412). Thinned and open treatments were statistically similar and significantly greater (Table 9) in percent grass cover than the closed treatment (Figure 10). The percent cover of exotic species was significantly affected by treatment type (p= 0.0495), where thinned treatments yielded larger cover percentages of exotic species than both open and closed treatments (Figure 11). Exotic species cover accounted for about 13% of the total understory cover. Restoration treatments and soil parent material type had no significant effect on percent cover of forbs or vegetation when categorized by longevity type (i.e. annual, annual-perennial, biennial, and perennial).
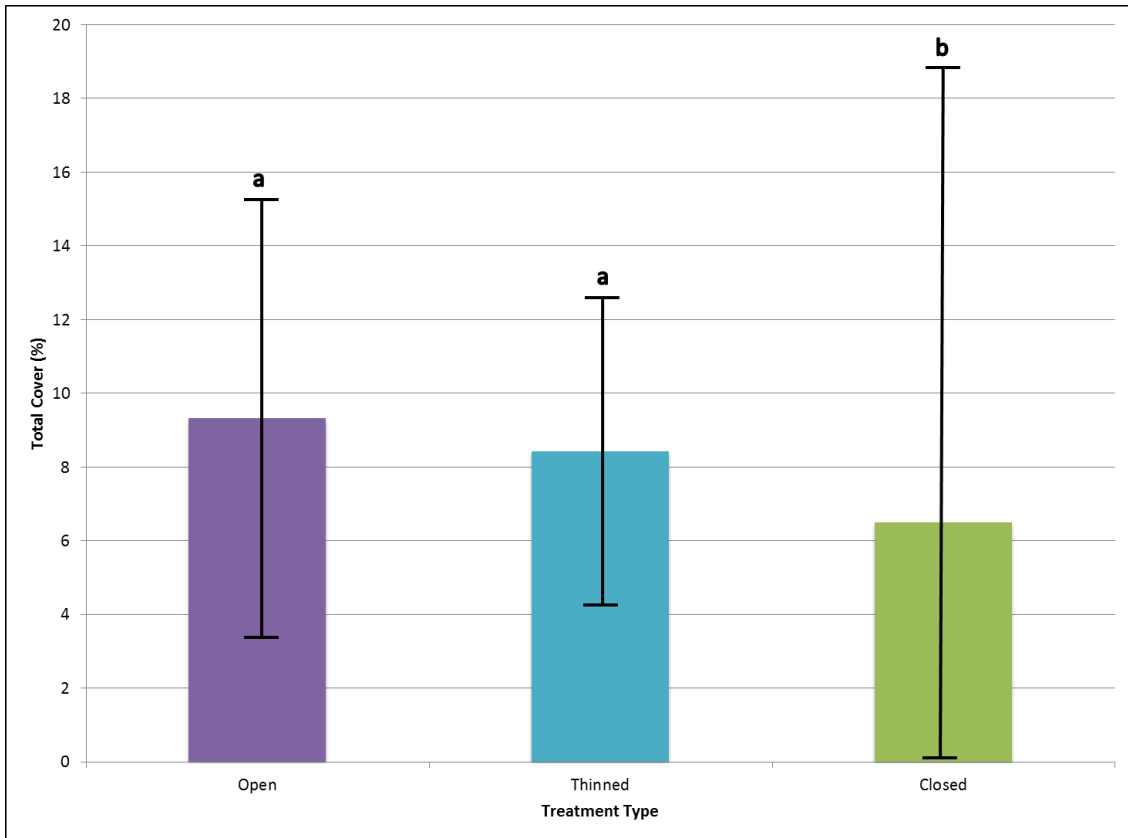
Figure 9. Mean percent cover of total vegetation among treatment types, bars with similar letters are not significantly different (α=0.10). Each bar represents one standard deviation away from the mean.

Table 9. Means and standard deviations for select vegetation measurements across soil parent material types that across restoration treatments.

	Limestone	Benmoreite	Basalt
Cover (%)			
Open	11.2 (15.79)	7.4 (8.82)	8.9 (14.56)
Thinned	10.8 (12.22)	8.2 (11.80)	5.0 (6.77)
Closed	5.6 (5.21)	8.8 (15.11)	5.8 (5.87)
Exotic Species Cover (%)			
Open	3.4 (2.56)	2.3 (1.97)	0.5 (0.58)
Thinned	10.2 (11.42)	3.0 (1.55)	2.6 (1.72)
Closed	1.6 (1.74)	0.0 (0.00)	1.3 (1.50)
Grass Cover (%)			
Open	13.6 (10.44)	10.2 (3.08)	7.7 (1.94)
Thinned	17.5 (9.10)	11.5 (3.33)	4.9 (2.16)
Closed	7.4 (3.56)	5.5 (2.02)	6.6 (3.41)
Forb Cover (%)			
Open	8.6 (8.10)	3.2 (1.04)	4.2 (2.52)
Thinned	6.2 (4.16)	5.9 (7.17)	5.6 (5.27)
Closed	4.1 (1.15)	4.3 (1.49)	2.7 (0.91)
Species Richness per 1m^2			
Open	36.0 (9.49)	31.0 (9.32)	34.5 (6.14)
Thinned	43.2 (10.25)	43.5 (11.88)	40.0 (14.70)
Closed	46.7 (14.85)	23.0 (12.57)	25.8 (6.85)
Species Richness per 9m^2			
Open	9.0 (2.00)	7.0 (1.10)	9.0 (3.83)
Thinned	9.3 (2.07)	10.0 (2.37)	9.8 (2.87)
Closed	9.5 (2.59)	6.2 (3.49)	6.3 (1.26)
Exotic Species Richness per 1m^2			
Open	6.0 (5.40)	6.2 (5.53)	0.5 (0.58)
Thinned	8.2 (5.81)	9.2 (4.83)	3.8 (3.20)
Closed	1.7 (2.34)	1.5 (3.67)	1.5 (2.38)
Exotic Species Richness per 9m^2			
Open	6.0 (5.40)	6.3 (5.68)	0.5 (0.58)
Thinned	7.8 (5.74)	8.7 (4.97)	3.8 (3.20)
Closed	1.7 (2.34)	0.0 (0.00)	1.5 (2.38)
Forbs Species Richness per 1m^2			
Open	17.8 (10.53)	12.3 (6.02)	12.0 (5.29)
Thinned	26.0 (12.25)	23.0 (11.24)	22.3 (14.03)
Closed	29.8 (14.52)	9.2 (10.21)	7.5 (3.51)
Forb Species Richness per 9m^2			
Open	5.8 (2.48)	3.7 (1.03)	4.8 (3.30)
Thinned	6.5 (1.64)	6.2 (2.56)	5.5 (3.00)
Closed	7.0 (2.45)	3.3 (2.66)	3.0 (1.41)
Grass Species Richness per 1m^2			
Open	18.2 (4.17)	15.2 (3.25)	18.5 (2.65)
Thinned	16.0 (2.76)	16.2 (2.32)	14.8 (3.50)
Closed	16.7 (3.08)	8.5 (6.92)	13.5 (8.66)
Grass Species Richness per 9m^2			
Open	3.3 (0.82)	2.7 (0.52)	3.0 (0.00)
Thinned	2.7 (0.82)	2.5 (0.55)	3.0 (0.82)
Closed	2.7 (0.82)	2.2 (1.33)	2.5 (0.58)
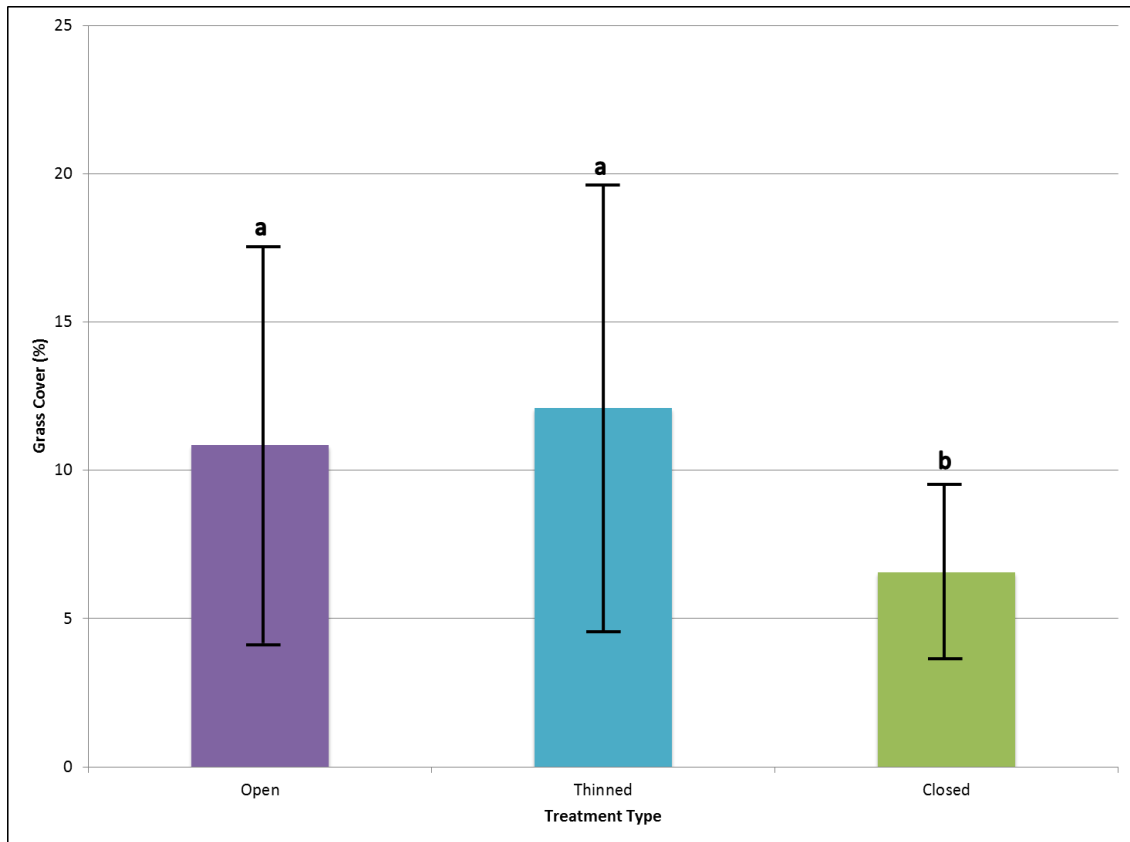
Figure 10. Percent cover of grass species among treatment types, bars with similar letters are not significantly different (α= 0.10). Each bar represents one standard deviation away from the mean.
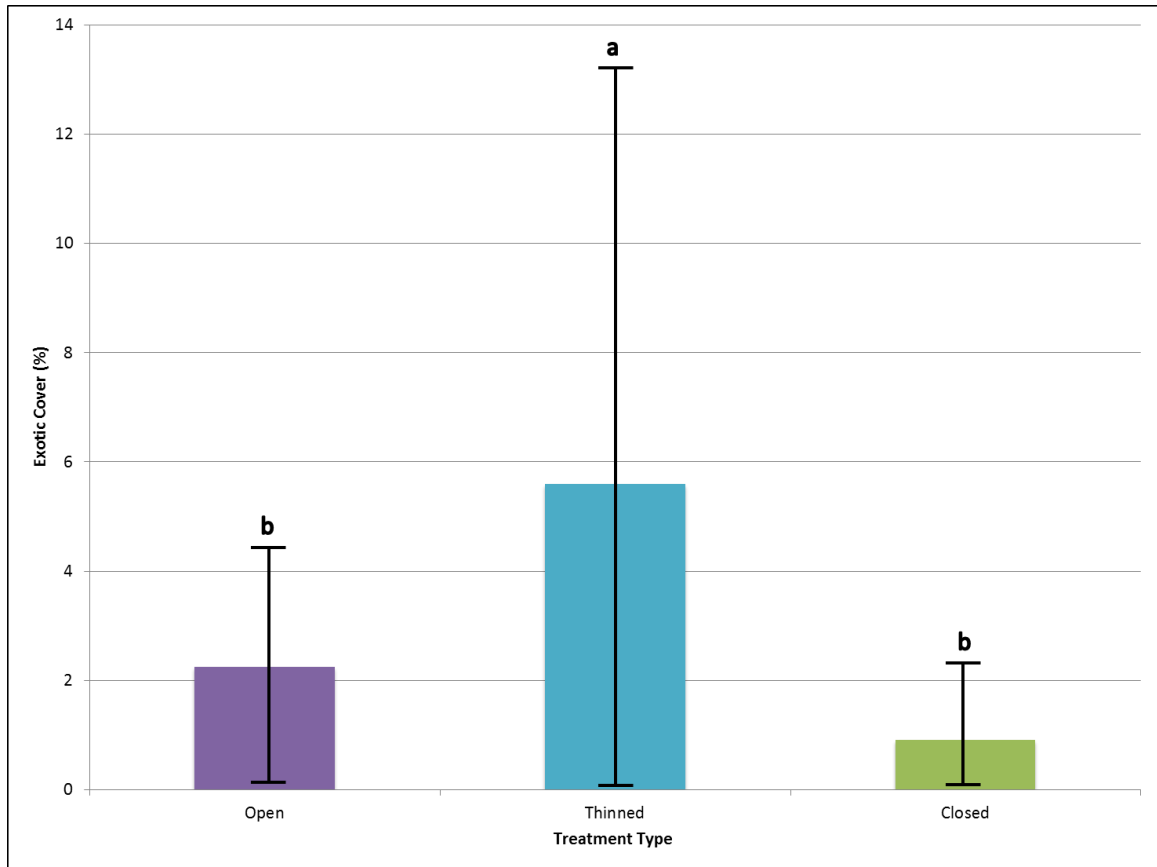
Figure 11. Exotic species percent cover among treatment types (α= 0.10), bars with the same letter are not statistically different. Each bar represents one standard deviation away from the mean.

Species Richness per Subplot

Species richness per subplot was significantly affected by treatment type (p= 0.0421) and an

interaction between treatment type and grazing treatment. Thinned treatments had significantly

higher species richness per subplot than both closed and open treatments (Figure 12). Species

richness per subplot was significantly affected by a soil type and treatment type interaction, and

many measurement year interactions, when measurement year was statistically factored in (Table

A.5). Species richness per subplot was the highest three years following restoration treatments. In addition, species richness across the three soil parent material types appeared to have a greater variation pre-treatment and three years following treatment, with benmoreite soils exhibiting the lowest. Limestone soils had significantly greater species richness levels pre-treatment and three-years after treatment. Though not significant, this study showed a trend toward greater species richness than five years following treatment application on limestone sites. As this study was conducted early in the growing season, it is possible that by late-growing season the species richness levels on limestone study sites, were closer to pre-treatment levels than was measured. Measurements conducted three years following restoration treatments, exhibited significantly less species under the closed control canopy than both open and thinned canopies. However, twelve years following treatment application showed no significant difference between open and closed canopies.

Exotic species richness per subplot was significantly affected by treatment type (p= 0.0392), with the thinned treatment significantly higher than the closed treatment (p= 0.0129). Exotic species richness per subplot for all measurement years was significantly affected by treatment type (p= 0.0391). Thinned plots exhibited significantly higher species richness for all measurement years after restoration treatments were performed. Closed and open plots were similar throughout the measurement years following restoration treatments. Exotic species richness per subplot was significantly affected by a soil type and measurement year interaction (p= 0.0019) and treatment type and measurement year interaction (p < 0.0001), (Table A.5).

Lifeform type was also influenced by treatment type at the subplot level, with thinned treatments (p= 0.0236) yielding more forb species than both open and closed canopies (Figure 13). Of the seven exotic species identified across all sites five were forbs; closed canopies among

all sites exhibited all five of those forb species. However, grass species per subplot was not

significantly affected by restoration treatment or soil type. Perennial species richness per subplot

was significantly affected by a soil type and treatment type interaction (p= 0.0882). Perennials

present included Arizona fescue (*Festuca arizonica* Vasey), trailing fleabane (*Erigeron*

flagellaris Gray), and Wright's deervetch (*Lotus wrightii* Gray Greene). Biennials were also

influenced at the subplot level, with an interaction between soil type and grazing treatment and a

three-way interaction between soil type, treatment type, and grazing treatment.
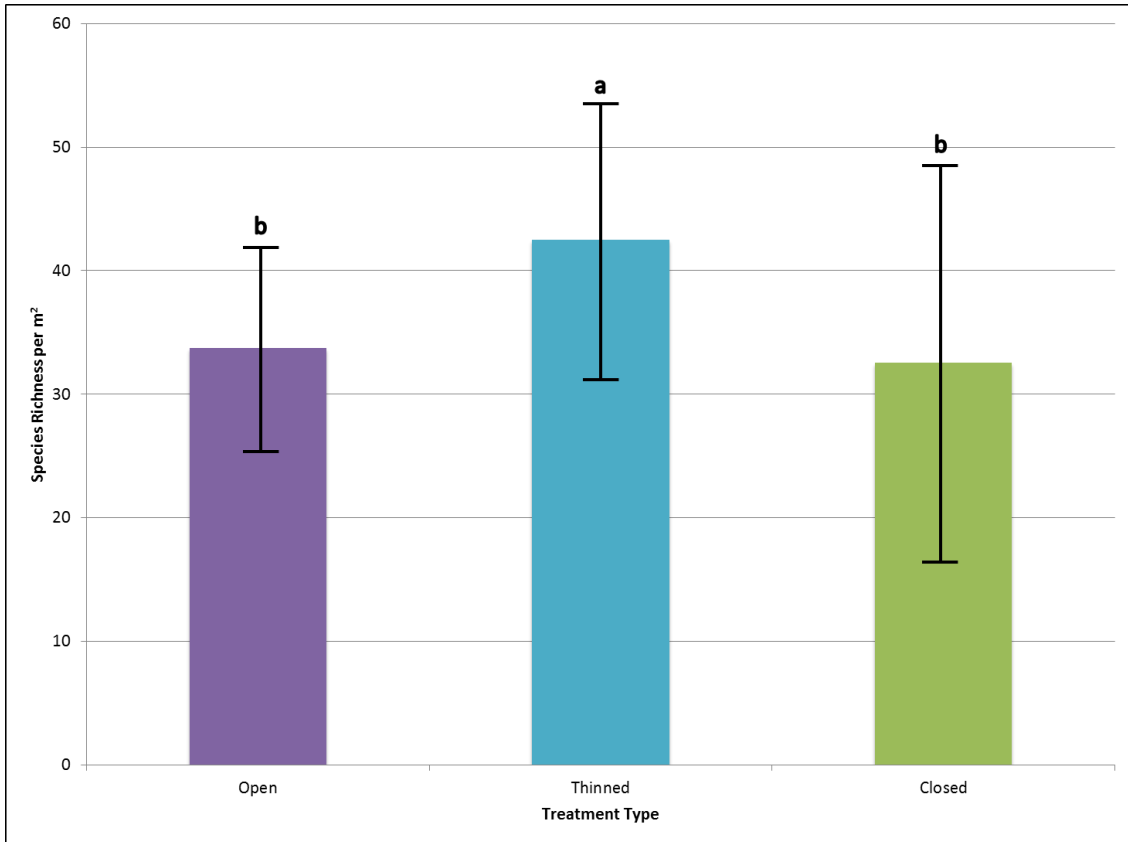
Figure 12. Mean species richness per subplot for total vegetation among treatment types, bars with similar letters are not significantly different (α=0.10). Each bar represents one standard deviation away from the mean.
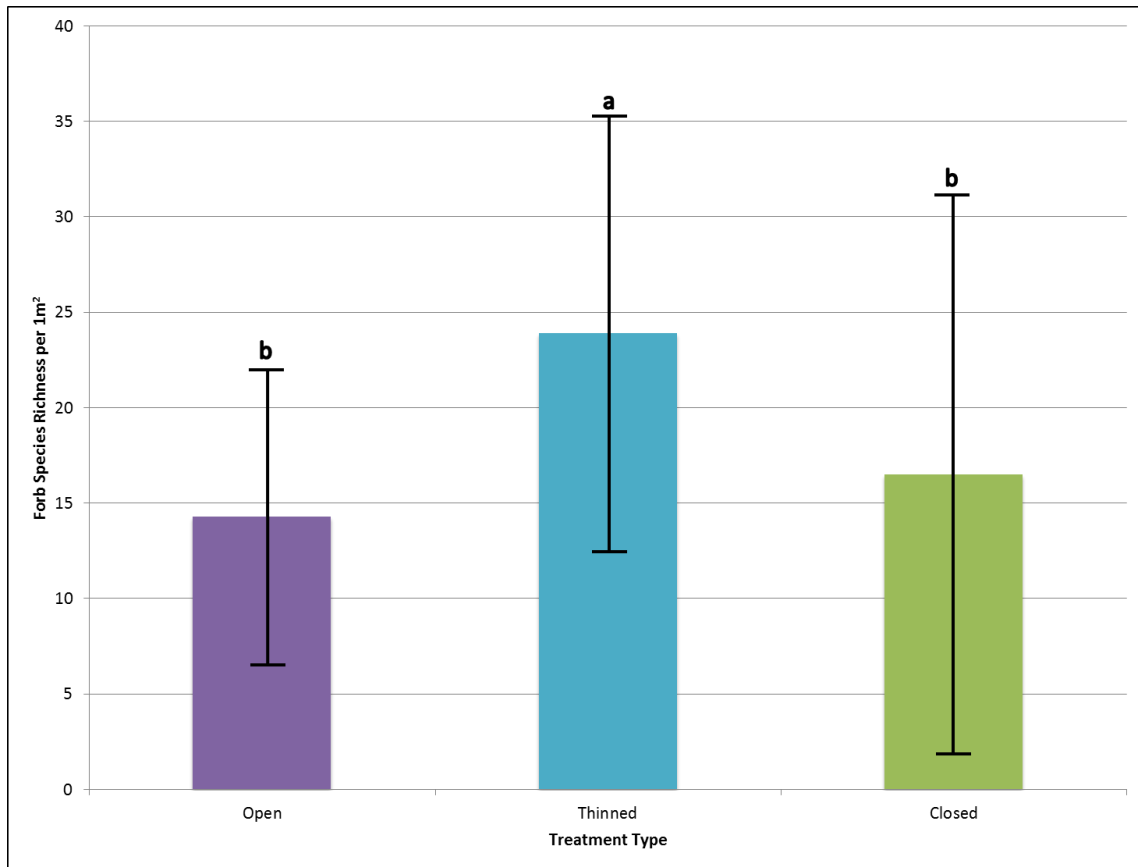
Figure 13. Forb species per subplot among treatment types, bars with similar letters are not significantly different ($\alpha = 0.10$). Each bar represents one standard deviation away from the mean.

Species Richness per Measurement Plot

Species richness per measurement plot was not significantly different among treatment types or soil parent material types, which contradicts what was found by Laughlin et al. (2011) on the Coconino National Forest near Flagstaff, Arizona, where species richness was found to decline when treatments were applied. When examining species richness per measurement plot for all measurement years, highly significant effects included: measurement year, a soil type and

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measurement year interaction, a treatment type and measurement year interaction, and a soil type, treatment type, and measurement year interaction (Table A.5). Three years following treatment application the species richness per measurement plot was the highest, but this study exhibited significantly lower species richness (12 years following treatment application). Between 2006 and 2008, three and five years after treatment application, there was no significant difference in species richness, but by 2015, this study, species richness had dropped. However, this study was conducted in early- and mid-summer, when some species were still dormant, compared to measurement years 2006 and 2008 when more species were abundant and active, like many perennials.

Species richness for vegetation at the measurement plot level for this study was influenced more by grazing, with the exception of exotic species and biennial and annual-perennial longevity types. The exotic species richness per measurement plot was significantly affected by treatment type (p= 0.0101), with closed canopies being significantly smaller in richness of exotic species than both open and thinned canopies (Figure 14). Exotic species included cheatgrass (*Bromus tectorum* L.), bull thistle (*Cirsium vulgare* (Savi) Ten.), prickly lettuce (*Lactuca serriola* L.), dalmation toadflax (*Linaria dalmatica* (L.) P. Mill.), Kentucky bluegrass (*Poa pratensis* L.), common dandelion (*Taraxacum officinale* G.H. Weber ex Wiggers), and common mullein (*Verbascum thapsus* L.). Cheatgrass was the dominant exotic species across all sites (8.0%), but was mostly found on limestone soil parent material types. Common mullein was the second most common exotic species across all study sties at 6.6% the total cover. McGlone et al. (2011) also found cheatgrass and common mullein to be the two most predominant non-native species on basaltic soils in northwestern Arizona.

When analyzing the change of exotic species richness from the beginning of the original study by Abella et al. (2015) in 2003 to this current study, it was found that exotic species richness per measurement plot was significantly influenced by almost all statistical effects analyzed (Table A.5). Most notably were the measurement year, soil type and measurement year interaction, treatment type and measurement year interaction, treatment type, and grazing treatment effects. For all measurement years the closed canopy treatment exhibited lower exotic species richness, as well as the exclosure grazing treatment. When examining interaction effects, such as soil parent material type and grazing, the basalt-derived soils exhibited considerably less exotic species per measurement plot than both limestone and benmoreite soils. Additionally, pre-treatment levels exhibited significantly less exotic species per measurement plot, than any other measurement year following restoration treatments. Limestone soils in 2006 exhibited the highest species richness of exotics. However, as this study was conducted in the summer, where some annuals were at their peak (i.e. cheatgrass) and some biennials and perennials had not yet emerged, there could be significantly greater or less differentiation between the measurement years. Measurements that were conducted in the fall (i.e. September) had more biennials and perennials, as well as annual-biennials, that were not present in the summer (i.e. June and July) survey. There were four more forb species and one more grass species listed in the fall measurements compared to the summer measurement for observed exotic species.

Biennial species richness per measurement plot was influenced by treatment type (p= 0.0143), with thinned treatments exhibiting higher biennial species richness, and a treatment type and grazing treatment interaction, but annual-perennial prevalence was influenced by soil parent material type only (p= 0.0652). Benmoreite soil parent material types had significantly greater annual-perennial species richness than basalt soil parent material types (p= 0.0360), which

contained no annual-perennial species. The most prevalent biennial was trailing fleabane, and

annual-perennials included forbs such as lobe-leaf groundsel (*Packera multilobata* Torr. & Gray

ex Gray) and Norwegian cinquefoil (*Potentilla norvegica* L.).
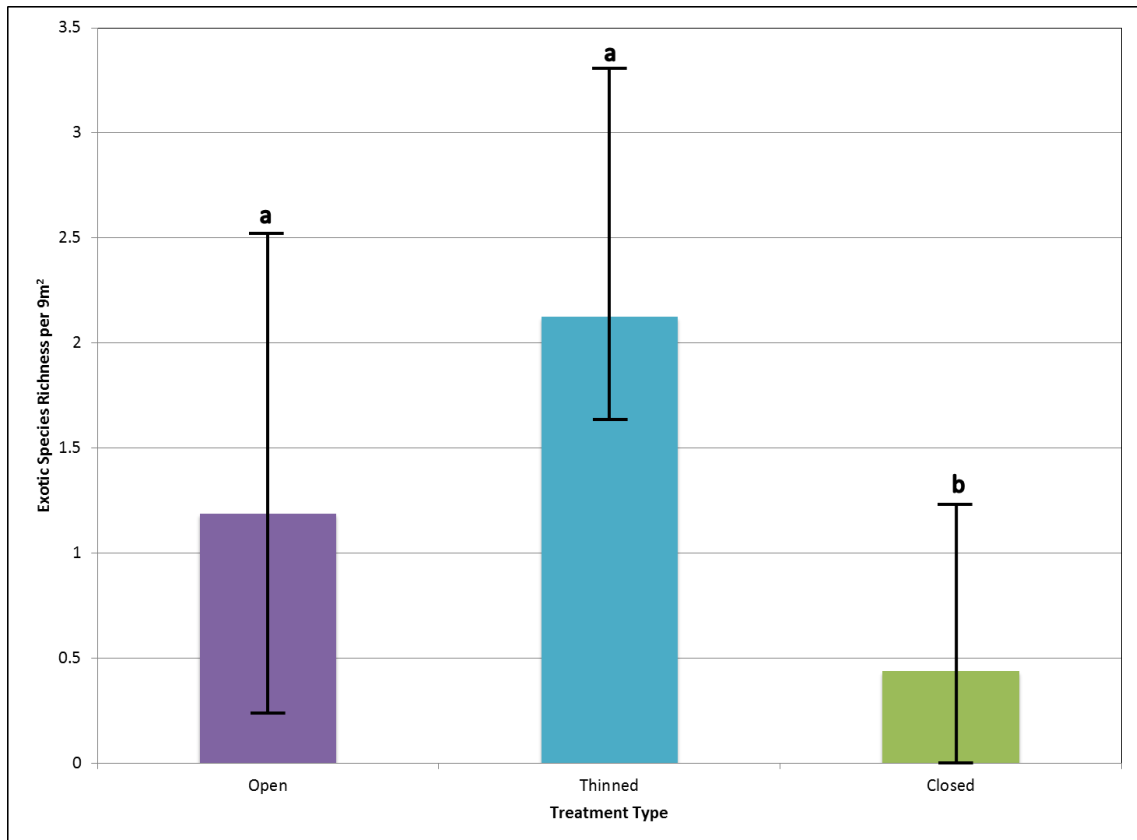
Figure 14. Species richness per measurement plot for exotic species, bars with letters that are different across treatment type for each parameter are significantly different (α= 0.10). Each bar represents one standard deviation away from the mean.

The dominant species found on basalt soils were Arizona fescue and Wright's deervetch, both

native perennials, while benmoreite soils exhibited dominant species of the native perennials

Fendler's buckbrush and western wheat grass (*Pascopyrum smithii* Rydb. A. Löve). Arizona

fescue was also the dominant species on limestone soils, with bulb panic grass (*Pancium bulbosum* Kunth), another native perennial, as the second dominant species. Ponderosa pine seedlings were more evident on basalt and benmoreite soil parent material types than limestone soils. This contradicts Naumburg and DeWald (1999), who found that ponderosa pine seedling regeneration occurs rarely on basalt derived soils, so any pine seedlings evident on these soils may have been on exposed mineral soils and had little competition with grasses.

Additionally, open restoration treatments had dominant species of trailing fleabane, a native biennial forb, and bulb panic grass; while closed treatments had mutton grass (*Poa fendleriana* Steud.) and Arizona fescue as dominant species, both perennial natives. Finally, thinned treatments exhibited dominant species of mutton grass and Fendler's buckbrush. Ponderosa pine saplings and seedlings were more abundant under the closed and open canopies.

Many believe that with an increase in ponderosa pine canopy the understory species richness and percent cover declines (Laughlin, et al., 2007). The increase in percent cover for total vegetation under thinned and open canopies verifies this hypothesis, but the only instance in which the study sites exhibited an increase in desired species richness with the creation of large spaces between trees was with forbs at the subplot level. As other studies have shown, the percent understory cover increased with a decrease in ponderosa pine canopy cover; this could be due to an increased light availability, which is what Moore et al. (2006) concluded from their results.

At the measurement plot level exotic species richness increased with a decrease in canopy cover. Coupled with this, on all sites there was a relatively low N status, with total N averaging 0.39 mg/kg across all sites, and high plant-available P, which averaged 32.03 mg/kg across all sites. With limited N and high P resources it is possible that native perennial vegetation will be

allowed to establish and dominate in an ecosystem more than non-native annuals, which rely more on plant-available nutrients for successful growth and dominance (McGlone, et al., 2011). McGlone et al. (2011) found that bottlebrush squirreltail can outcompete cheatgrass when mature plants have been established, as they are physiologically active during the same time of year as cheatgrass. This is evidenced on sites in this study. Most experimental treatment plots that contained cheatgrass had no presence of bottlebrush squirrel tail, or there were very little mature plants present. In addition, all experimental treatment plots that had high percentages of bottlebrush squirreltail cover had little to no amount of cheatgrass, which was also found in the study conducted by McGlone et al. (2011).

It has also been shown that after clear-cutting of ponderosa pine, there will be smaller increases in biomass of cool-season species compared to warm-season species. However, after several years following thinning of ponderosa pine cool-season grasses have been found to have greater biomass under larger densities of ponderosa pine than smaller density stands (Naumburg and DeWald, 1999). Laughlin et al. (2011) also found a shift in species composition where cool-season vegetation dominated. The findings from this study correlate to what Naumburg and DeWald (1999) and Laughlin et al. (2011) concluded, as can be seen by the two cool-season grasses that dominate the closed canopy (mutton grass and Arizona fescue), compared to the open and thinned canopies, which are dominated by two grasses, a forb, and a shrub.

Grazing Treatment Effects on Vegetation

<u>Cover</u>

While litter coverage, percent cover, and exotic species percent cover were significantly affected by restoration treatments, grazing played no role in influencing these parameters. It can be surmised that grazing starts to have little effect on total plant response to restoration treatments after a long period of time following application. The only instance in which there was a restoration treatment and grazing interaction was for total species richness per subplot (p= 0.0878), with thinned canopies undergoing grazing having the greatest species richness (Figure 15). However, when factoring in nativity, lifeform, and longevity, grazing effects can be seen.
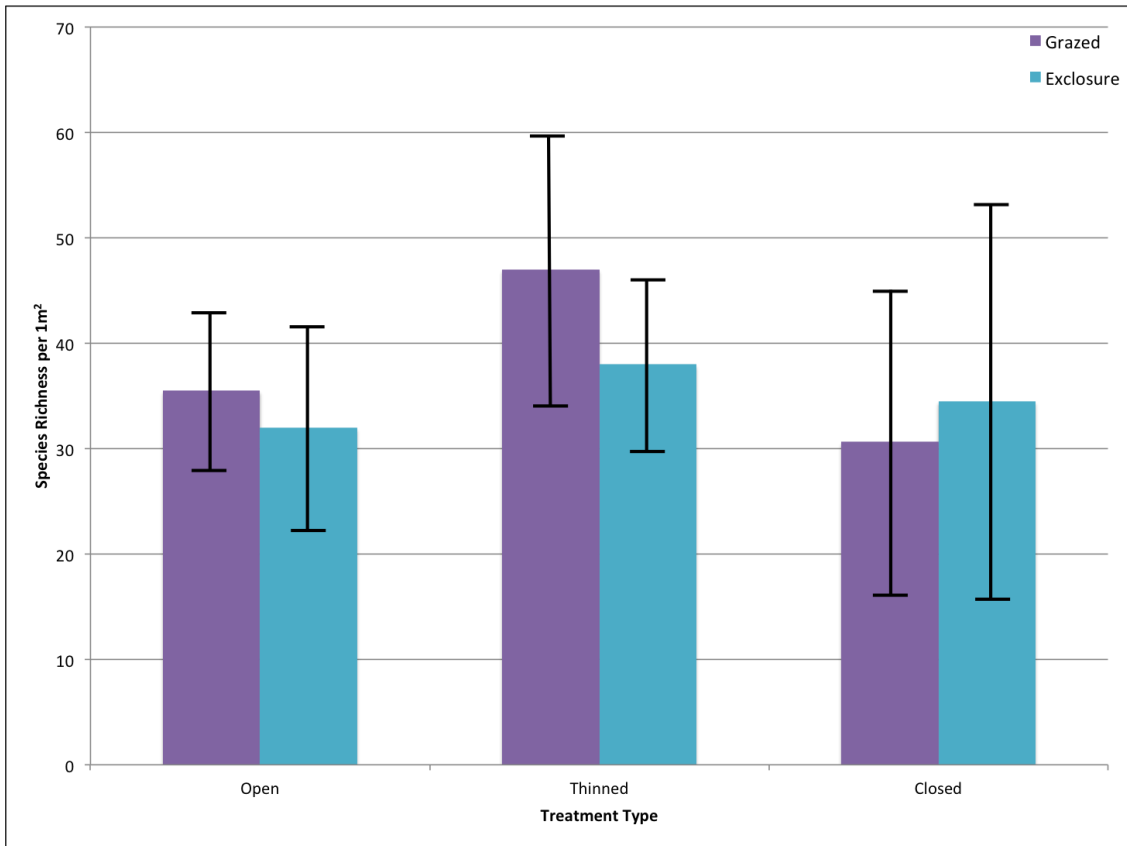
Figure 15. Total species richness per subplot for the significant treatment type and grazing treatment interaction (0.0878), $\alpha = 0.10$. Each bar represents one standard deviation away from the mean.

Table 10. Means and standard deviations for various vegetation measurements between grazing treatments undergoing restoration treatments.

	Grazed	Excluded
Cover (%)		
Open	8.8 (13.31)	9.9 (13.75)
Thinned	6.5 (7.81)	10.9 (13.99)
Closed	7.1 (12.09)	6.0 (5.24)
Exotic Species Cover (%)		
Open	2.5 (2.58)	2.0 (1.88)
Thinned	2.8 (1.32)	8.4 (10.30)
Closed	0.6 (1.19)	1.2 (1.65)
Grass Cover (%)		
Open	8.1 (2.15)	13.6 (8.76)
Thinned	10.8 (7.81)	13.4 (7.69)
Closed	6.7 (3.16)	6.4 (3.02)
Forb Cover (%)		
Open	5.8 (7.78)	5.1 (1.82)
Thinned	4.1 (2.80)	7.8 (6.76)
Closed	3.3 (1.25)	4.3 (1.23)
Species Richness per 1m^2		
Open	35.5 (7.35)	32.0 (9.62)
Thinned	47.0 (12.69)	38.0 (8.11)
Closed	30.6 (14.34)	34.5 (18.75)
Species Richness per 9m^2		
Open	9.1 (2.03)	7.4 (2.50)
Thinned	10.8 (2.38)	8.6 (1.60)
Closed	7.4 (3.07)	7.5 (3.25)
Exotic Species Richness per 1m^2		
Open	4.0 (4.24)	5.4 (6.09)
Thinned	7.3 (3.45)	7.6 (6.63)
Closed	0.8 (1.75)	2.4 (3.38)
Exotic Species Richness per 9m^2		
Open	4.1 (4.45)	5.4 (6.09)
Thinned	7.0 (3.30)	7.3 (6.61)
Closed	0.8 (1.75)	1.3 (2.12)
Forb Species Richness per 1m^2		
Open	14.8 (6.80)	13.9 (9.36)
Thinned	26.8 (13.76)	21.1 (8.97)
Closed	14.9 (12.19)	18.1 (17.90)
Forb Species Richness per 9m^2		
Open	5.6 (2.20)	3.9 (2.30)
Thinned	6.9 (2.47)	5.4 (1.85)
Closed	4.6 (2.72)	4.6 (3.25)
Grass Species Richness per 1m^2		
Open	18.0 (2.73)	16.3 (4.37)
Thinned	16.3 (3.58)	15.3 (1.39)
Closed	12.8 (7.21)	12.9 (7.08)
Grass Species Richness per 9m^2		
Open	3.0 (0.53)	3.0 (0.76)
Thinned	2.8 (0.71)	2.6 (0.74)
Closed	2.3 (1.04)	2.6 (0.92)

<u>Species Richness per Subplot</u>

Total species richness per subplot was significantly affected by a treatment type and grazing treatment interaction (Table 10), which was discussed in the previous section. Grazing had little effect on species richness per subplot when factoring in different measurement years; it only significantly affected species richness per subplot in a treatment type and grazing treatment interaction (p= 0.0932) and a four-way interaction (0.0966) between soil type, treatment type, grazing treatment, and measurement year (Table A.5).

Exotic species richness per subplot was significantly affected by a soil type and grazing treatment interaction (p= 0.0679), where the grazing exclosure on basalt parent material was significantly lower than the grazing exclosure on limestone parent material (p= 0.0703). When factoring in the measurement year, exotic species richness per subplot was significantly affected by a grazing treatment and measurement year interaction. Pre-treatment measurements showed the lowest exotic species richness among years measured per subplot, but following restoration treatments, three years and five years later, there was significantly higher exotic species per subplot.

Grasses and forbs did not exhibit differences between grazing treatments for species richness per subplot. However, perennials and biennials showed a response in species richness per subplot to grazing effects. Biennials were influenced by a soil type and grazing treatment interaction (p= 0.0540) and a three-way interaction (p= 0.0700), and perennials were influenced by a treatment type and grazing treatment interaction (p= 0.0882) at the subplot level. Limestone soils with grazing exclosures exhibited higher biennial species richness at the subplot level than basalt soils

with grazing exclosures, and benmoreite soils undergoing grazing had significantly lower biennial

species richness than limestone soils undergoing grazing (Figure 16).
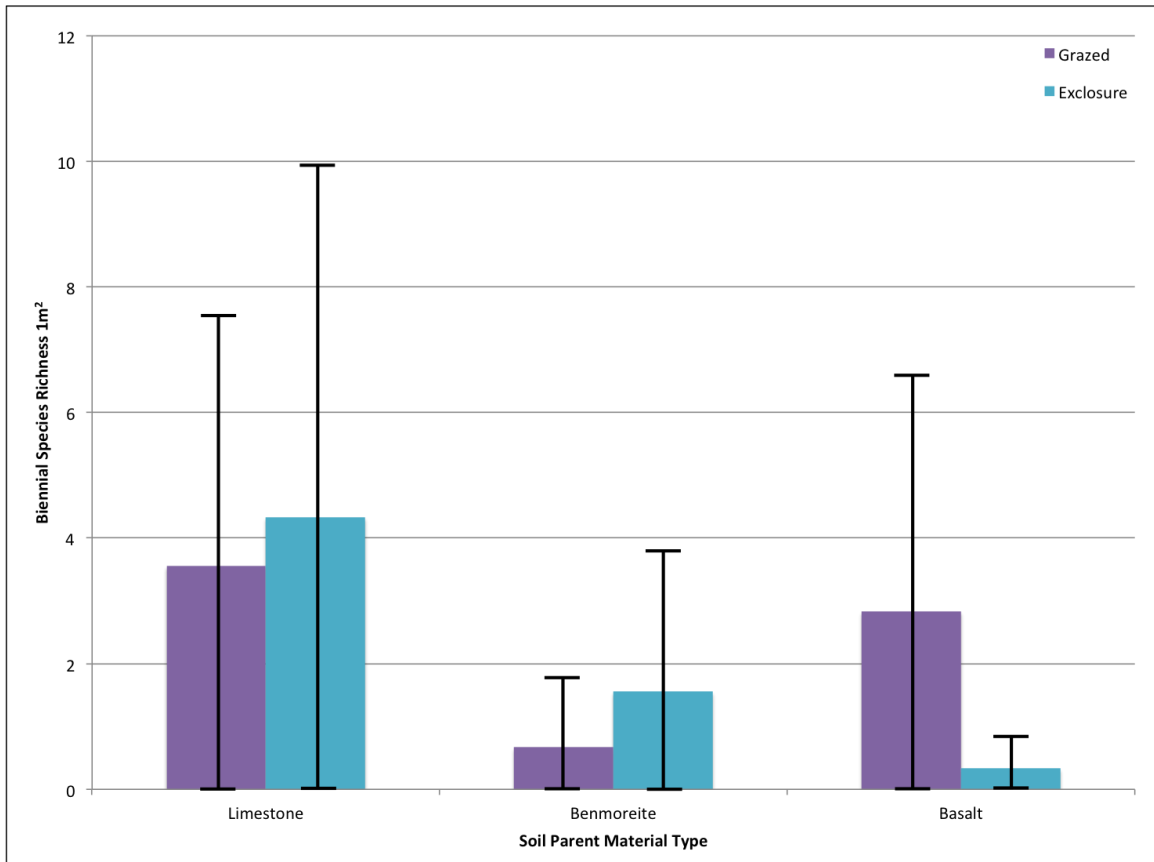
Figure 16. Biennial species richness per subplot significantly affected by an interaction between soil type and treatment type (p= 0.0540), α= 0.10. Each bar represents one standard deviation away from the mean.

Species Richness per Measurement Plot

When examining species richness at the measurement plot level, forbs were significantly

influenced by grazing (p= 0.0269). The increase in forb species under grazed conditions could be

from introduction of seeds through grazing vectors, or a more preferential grazing for grasses

compared to forbs. Increased richness of forbs could also be due to soil disturbance and an increase in exposed mineral soil. However, species richness for grasses remained unaffected by grazing treatment.

Perennial species richness was the only longevity category that was significantly influenced by grazing alone (p= 0.0388) at the measurement plot level, with grazed plots having higher perennial species richness than the exclosure plots. It has been found that areas with higher perennial cover are more resistant to non-native invasion (McGlone, et al., 2011); although exotic species richness is not significantly affected by grazing alone, as a whole there are fewer exotic species under grazed conditions. Biennial species richness per measurement plot was influenced by a treatment type and grazing treatment interaction (p= 0.0840). Thinned treatments undergoing grazing had significantly higher biennial species richness per measurement plot than both closed and open treatments undergoing grazing (Figure 17). Thinned treatments also had higher biennial species richness than closed treatments in the grazing exclosures. Exotic species richness per measurement plot was significantly affected by a soil type and grazing interaction (p= 0.0332), with exotic species richness the lowest under a basalt soil parent material type undergoing grazing exclusion (Figure 18).

Exotic species richness per measurement plot was significantly affected by grazing treatment, a soil type and grazing treatment interaction, a treatment type and grazing treatment interaction, two three-way interactions, and a four-way interaction when all measurement years were statistically analyzed (Table A.5). When examining all measurement years, the control or grazing treatment was significantly higher in species richness of exotics per measurement plot (p= 0.0120). Measurement plot undergoing grazing on basalt parent materials had the lowest exotic species richness (p=0.0048), and exclosure measurement plots on basalt soils had significantly

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less exotic species than exclosure measurement plots on limestone soils. Additionally, grazed measurement plots under thinned treatment canopies had the highest exotic species richness per measurement plot, while grazed measurement plots under closed control canopies had the lowest.
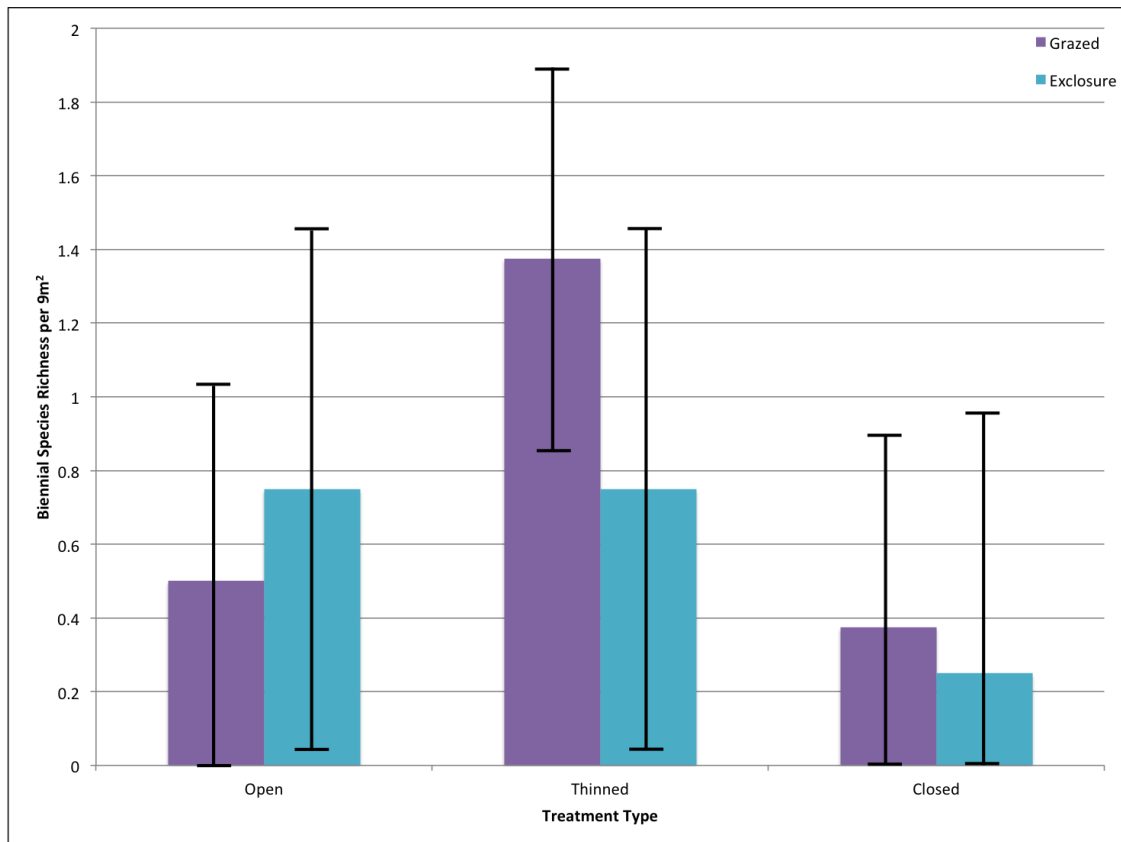
Figure 17. Treatment type and grazing treatment interaction effect for biennial species richness per measurement plot (p= 0.0840), α= 0.10. Each bar represents one standard deviation away from the mean.
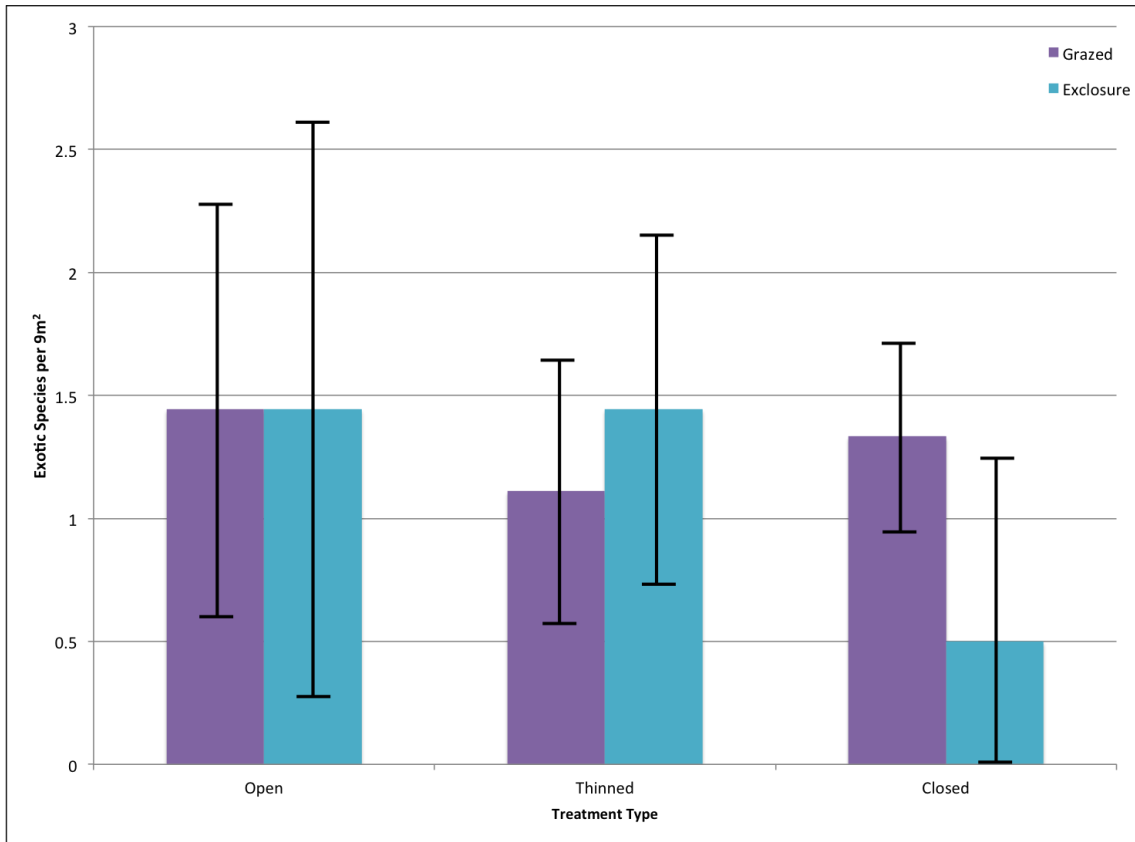
Figure 18. Soil type and grazing treatment interaction (p= 0.0647) for exotic species richness per measurement plot, bars with similar letters are not significantly different (α= 0.10). Each bar represents one standard deviation away from the mean.

The greatest plant response to grazing in this study was for total species richness per measurement plot (p= 0.0226). However, species richness per measurement plot was not affected by grazing when all measurement years were analyzed. Grazed measurement plots exhibited a higher species richness than grazing exclosure measurement plots, which coincides with Schultz et al. (2011), who found grazing exclusion to decrease species richness due to increased accumulations of phytomass, both live and dead plant matter. However, litter coverage rates per

measurement plot were not significantly affected by grazing treatment, so litter accumulations are most likely not driving species richness differences at these study sites. Similarly to Johnston (1961), there was greater plant diversity under grazed conditions than ungrazed conditions at these study sites, and unlike many other studies, percent cover was not significantly affected by grazing treatments in the long-term. Following treatment application at these study sites almost 10 years prior to this study (2006), Abella et al. (2015) found grazing had a significant influence on vegetative cover (p= 0.024), but now grazing effects have become insignificant (p= 0.2886). However, the differences in grazing intensity have changed, so influences of grazing on vegetation could have also changed. In addition, the vegetation surveys taken for this study were completed during June and July, which evaluated the "cool season" vegetation, while Abella et al. (2015) conducted vegetation surveys in September on the "warm season" vegetation. With these variations in season, the vegetation differences could be greater.

The dominant species in grazed areas were trailing fleabane, a biennial forb, and mutton grass, both native species. Ponderosa pine saplings and seedlings were commonly found in grazed areas in addition to the dominant species mentioned. Grazing exclosures had dominant species of Fendler's buckbrush, a shrub, and bulb panic grass, both native perennials.

CONCLUSIONS

Restoration treatments did not appear to significantly influence the select soil properties, which were instead influenced partly by grazing and soil parent material type. However, grazing appeared to influence the soil parameters measured more than soil parent material type, which might suggest that the soils were so weathered that the underlying geologic substrates played a minimal role in influencing the select soil properties. Instead, management practices contributed measured differences in select properties of the soil, most evident is soil temperature and soil nutrient response to grazing and restoration treatments.

Vegetative cover was significantly influenced by restoration treatments, possibly due to differences in competition for light and other resources (i.e. soil moisture and nutrients). Unlike Abella et al. (2015), understory cover was not influenced by soil parent material type in this study, which might suggest that differences in understory cover due to soil properties are only seen shortly after restoration treatments are applied. On the other hand, the differences could be attributed to the time frame in which the vegetation was measured in this study (i.e. earlier in the growing season).

Species richness was influenced more by grazing practices than restoration treatments, with current grazing practices increasing species richness. Soil parent material type also did not play a role in species richness, except when soil type was included in an interaction, so it appears that soil type may have had little influence on vegetative communities in the years since the restoration treatments were implemented at these study sites.

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It also appears that restoration treatments and grazing might create an environment conducive to non-native, exotic species establishment, (Table A.5). Establishment of exotic species can be limited by a strong presence of competitive native vegetation, such as bottlebrush squirreltail, and low plant nutrient regimes, so when establishing a management plan the soil type, as well as other soil properties, and existing vegetation should be considered.

This study exhibited lower species richness than in previous measurement years, but this could be from the type of vegetation that was measured and the fact that many perennials were still dormant early in the growing season. Abella et al. (2015) concluded that the plant communities had become established by three years post-treatment (2006), but it could be that the "warm-season" vegetation had become established in that time frame, while the "cool-season" vegetation had not, or possibly will never become established. However, differences in species richness could be less or greater when vegetation that is active later in the season is measured in addition to the vegetation measured in this study. To fully determine the extent that restoration treatments, soil parent material type, and grazing treatments have on vegetation, both "cool-season" vegetation, such as early annuals that emerge in May like cheatgrass, and "warm-season" vegetation, such as late perennials like showy goldeneye (*Heliomeris multiflora* Nutt.), which blooms from August to September (Kershaw, et al., 1998), should be taken into account.

The completion of the companion study that examines soil physical properties and later season vegetation will provide a more complete picture of current conditions on these sites. Another limitation to this study is the lack of a fire treatment, which due to forest conditions was not possible at the time of the other treatment applications. Introduction of fire to this ecosystem could greatly affect the status of soil properties and vegetation. It is possible that a much longer

term study (i.e. 50 years or more) is needed to fully elucidate the biological and chemical

processes that are occurring in response to the restoration treatments.

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APPENDIX

Table A. 1. June 2015 daily weather data for the Flagstaff, Arizona area (NOAA, 2014).

Date	Temperature				HDD	CDD	Precipitation	New Snow	Snow Depth
	Maximum	Minimum	Average	Departure					
6/1/2015	77	42	59.5	3.9	5	0	0	0	0
6/2/2015	75	40	57.5	1.6	7	0	0	0	0
6/3/2015	73	31	52	-4.2	13	0	0	0	0
6/4/2015	73	30	51.5	-4.9	13	0	0	0	0
6/5/2015	65	48	56.5	-0.2	8	0	1	0	0
6/6/2015	65	40	52.5	-4.5	12	0	0.07	0	0
6/7/2015	73	33	53	-4.3	12	0	0	0	0
6/8/2015	79	37	58	0.4	7	0	0	0	0
6/9/2015	68	55	61.5	3.6	3	0	0.01	0	0
6/10/2015	70	55	62.5	4.3	2	0	0.04	0	0
6/11/2015	73	47	60	1.5	5	0	0	0	0
6/12/2015	73	41	57	-1.8	8	0	0	0	0
6/13/2015	75	47	61	1.9	4	0	0.02	0	0
6/14/2015	78	42	60	0.6	5	0	T	0	0
6/15/2015	83	44	63.5	3.8	1	0	0	0	0
6/16/2015	84	49	66.5	6.4	0	2	0	0	0
6/17/2015	86	50	68	7.6	0	3	0	0	0
6/18/2015	86	50	68	7.3	0	3	0	0	0
6/19/2015	85	45	65	4	0	0	0	0	0
6/20/2015	88	44	66	4.7	0	1	0	0	0
6/21/2015	85	46	65.5	3.9	0	1	0	0	0
6/22/2015	86	48	67	5.1	0	2	0	0	0
6/23/2015	86	52	69	6.8	0	4	0	0	0
6/24/2015	86	50	68	5.5	0	3	0	0	0
6/25/2015	87	53	70	7.2	0	5	0	0	0
6/26/2015	92	50	71	7.9	0	6	0	0	0
6/27/2015	86	53	69.5	6.1	0	5	0.09	0	0
6/28/2015	82	53	67.5	3.9	0	3	0.06	0	0
6/29/2015	85	54	69.5	5.6	0	5	0.41	0	0
6/30/2015	86	58	72	7.9	0	7	0	0	0
Sum	2390	1387	-	-	105	50	1.7	0	-
Average	79.7	46.2	63	3.1	-	-	-	-	0
Normal	77.9	41.9	59.9	-	170	17	0.36	0	-

Temperature is measured in °F
HDD: Heating degree days (base of 65)
CDD: Cooling degree days (base of 65)
Precipitation, new snow, and snow depth is measured in inches
T= Trace amounts of precipitation that is < 0.01 inch measuring limit

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Table A. 2. July 2015 daily weather data for the Flagstaff, Arizona area (NOAA, 2014).

| Date	Temperature				HDD	CDD	Precipitation	New Snow	Snow Depth
	Maximum	Minimum	Average	Departure					
7/1/2015	83	58	70.5	6.1	0	6	0.06	0	0
7/2/2015	83	55	69	4.4	0	4	0.06	0	0
7/3/2015	84	58	71	6.2	0	6	0.53	0	0
7/4/2015	77	56	66.5	1.5	0	2	0.19	0	0
7/5/2015	75	56	65.5	0.3	0	1	0.53	0	0
7/6/2015	75	53	64	-1.4	1	0	0.07	0	0
7/7/2015	76	47	61.5	-4.1	3	0	0.25	0	0
7/8/2015	75	47	61	-4.7	4	0	0	0	0
7/9/2015	71	43	57	-8.9	8	0	0	0	0
7/10/2015	72	37	54.5	-11.5	10	0	0	0	0
7/11/2015	74	46	60	-6.1	5	0	0.01	0	0
7/12/2015	78	44	61	-5.2	4	0	0	0	0
7/13/2015	80	48	64	-2.3	1	0	T	0	0
7/14/2015	77	50	63.5	-2.9	1	0	0.01	0	0
7/15/2015	78	46	62	-4.5	3	0	0	0	0
7/16/2015	80	49	64.5	-2	0	0	0	0	0
7/17/2015	79	50	64.5	-2.1	0	0	0	0	0
7/18/2015	76	50	63	-3.6	2	0	0.79	0	0
7/19/2015	71	52	61.5	-5.1	3	0	0.09	0	0
7/20/2015	73	52	62.5	-4.1	2	0	T	0	0
7/21/2015	76	48	62	-4.6	3	0	0	0	0
7/22/2015	77	49	63	-3.6	2	0	0	0	0
7/23/2015	78	47	62.5	-4.1	2	0	0	0	0
7/24/2015	79	52	65.5	-1	0	1	0	0	0
7/25/2015	81	51	66	-0.5	0	1	0	0	0
7/26/2015	81	52	66.5	0.1	0	2	0	0	0
7/27/2015	80	46	63	-3.4	2	0	0	0	0
7/28/2015	82	49	65.5	-0.8	0	1	0	0	0
7/29/2015	81	53	67	0.7	0	2	0	0	0
7/30/2015	75	51	63	-3.2	2	0	0.02	0	0
7/31/2015	78	50	64	-2.1	1	0	0.16	0	0
Sum	2405	1545	-	-	59	26	2.77	0	-
Average	77.6	49.8	63.7	-2.4	-	-	-	-	0
Normal	81.2	50.9	66.1	-	32	64	2.61	0	-

Temperature is measured in °F
HDD: Heating degree days (base of 65)
CDD: Cooling degree days (base of 65)
Precipitation, new snow, and snow depth is measured in inches
T= Trace amounts of precipitation that is < 0.01 inch measuring limit
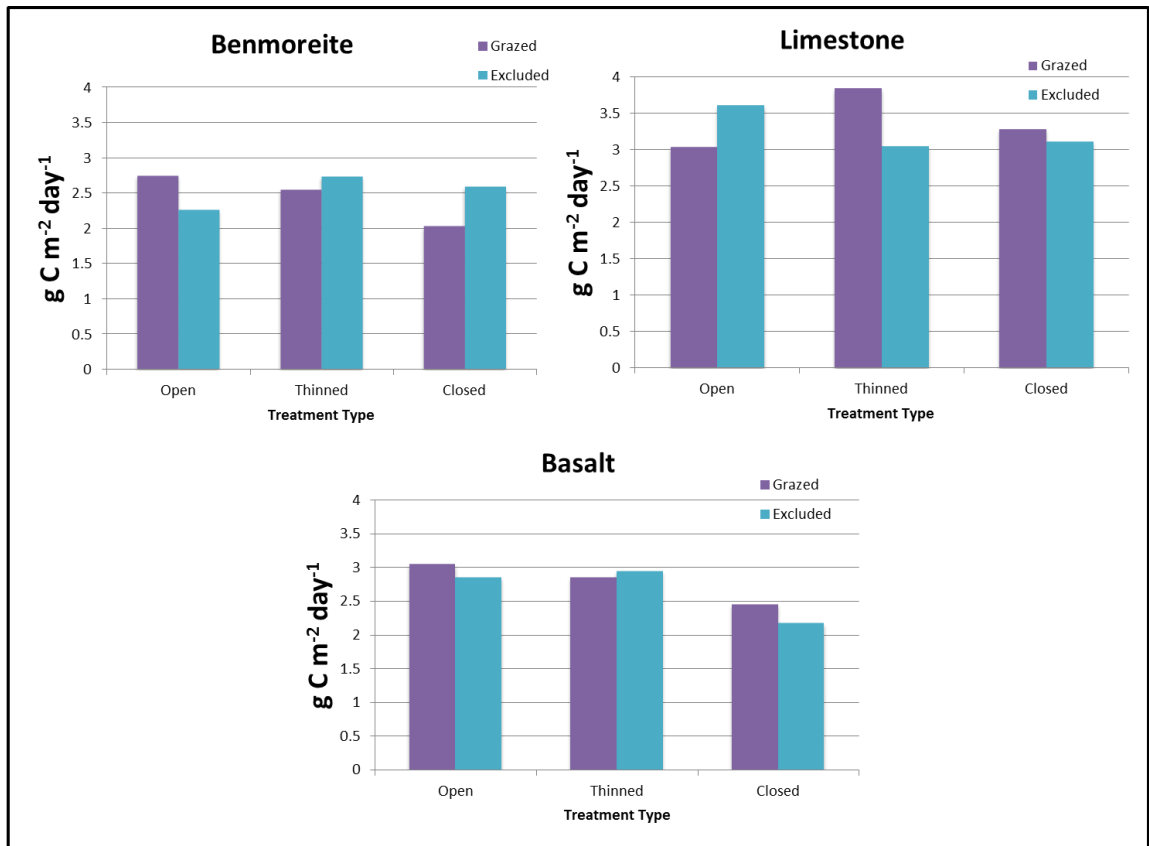
Figure A. 1. Three way interaction graphs for soil respiration, in which the effects of soil parent material type, restoration treatment type, and grazing treatment are examined (α= 0.10).

Table A. 3. ANOVA F statistics and p values for significant soil parameters and their effects, α= 0.10.

	Soil Type	Plot Type	Grazing	Soil Type x Plot Type	Soil Type x Grazing	Plot Type x Grazing	Soil Type x Plot Type x Grazing
				F Statistic (P value)			
Coarse Fragments	8.28 (0.0259)	-	4.86 (0.0786)	-	-	-	-
Respiration (g of CO_2 m^{-2} day^{-1})	4.21 (0.0846)	-	-	-	-	-	3.63 (0.0447)
Soil Temperature (°C)	-	4.79 (0.0347)	6.24 (0.0546)	-	-	-	-
Soil Na^+	-	-	-	-	-	-	3.88 (0.0374)
Soil Mg^{2+}	-	-	-	-	-	8.01 (0.0084)	5.13 (0.0165)
Soil Ca^{2+}	-	-	-	-	-	4.17 (0.0482)	3.43 (0.0519)
Soil K^+	-	-	4.74 (0.0814)	-	-	-	3.39 (0.0533)

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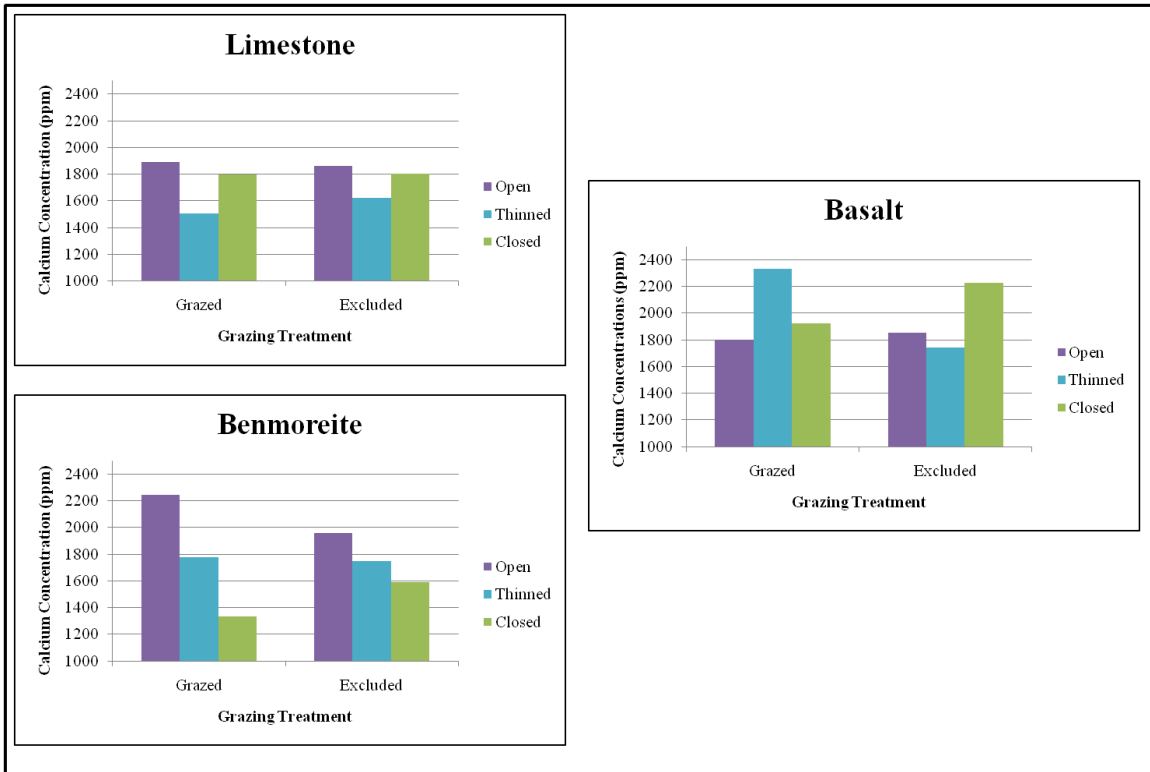
Figure A. 2. Three way interaction graphs for soil calcium concentrations, in which the effects of soil parent material type, restoration treatment type, and grazing treatment are examined (α= 0.10).
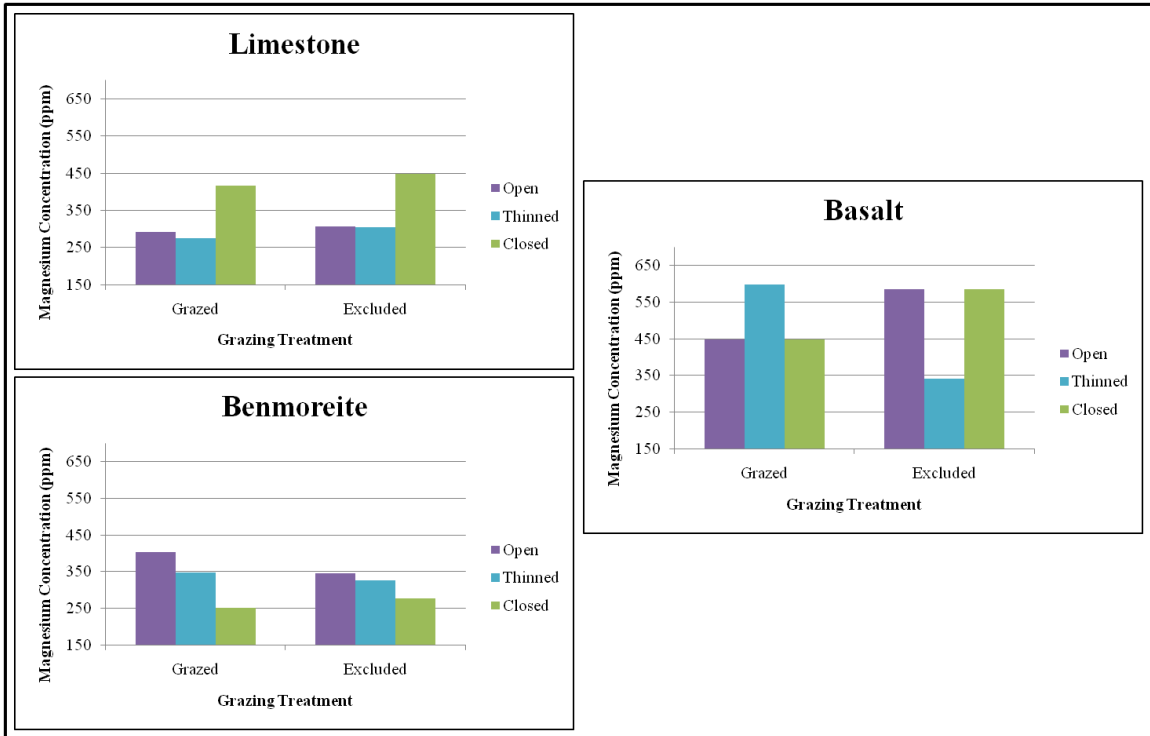
Figure A. 3. Three way interaction graphs for soil magnesium concentrations, in which the effects of soil parent material type, restoration treatment type, and grazing treatment are examined (α= 0.10).

Table A. 4. ANOVA F statistics and p values for significant vegetation parameters and their effects, $\alpha= 0.10$.

	Soil Type	Plot Type	Grazing	Soil Type x Plot Type	Soil Type x Grazing	Plot Type x Grazing	Soil Type x Plot Type x Grazing
				F Statistic (P value)			
Plant Cover (%)	-	4.44 (0.0418)	-	-	-	-	-
Litter Cover (%)	-	5.62 (0.0231)	-	-	-	-	-
Species per Subplot	-	4.42 (0.0421)	-	-	-	3.13 (0.0878)	-
Species per Measurement Plot	-	-	10.58 (0.0226)	-	-	-	-
Exotic Species per Subplot	-	4.55 (0.0392)	-	-	4.83 (0.0679)	-	-
Exotic Species per Measurement Plot	-	16.69 (0.0007)	-	-	7.26 (0.0332)	7.53 (0.0101)	4.42 (0.0259)
Exotic Species Cover (%)	-	4.12 (0.0495)	-	-	-	-	-
Grass Cover (%)	-	4.64 (0.0412)	-	-	-	-	-
Forbs per Subplot	-	5.58 (0.0236)	-	-	-	-	-
Forbs per Measurement Plot	-	-	9.61 (0.0269)	-	-	-	-
Annual-perennial per Measurement Plot	4.95 (0.0652)	-	-	-	-	-	-
Biennial per Subplot	-	-	-	-	5.53 (0.0540)	-	-
Biennial per Measurement Plot	-	6.69 (0.0143)	-	-	-	3.21 (0.0840)	3.04 (0.0700)
Perennial per Subplot	-	-	-	2.76 (0.0882)	-	3.04 (0.0928)	-
Perennial per Measurement Plot	-	-	7.75 (0.0388)	-	-	-	-

Table A. 5. ANOVA table for comparisons of species richness among all measurement years, bold indicates significance at α= 0.10.

Effect	Species Richness per Subplot	Species Richness per Measurement Plot	Exotic Species Richness per Subplot	Exotic Species Richness per Measurement Plot
Soil Type	1.43 (0.3234)	1.02 (0.4240)	1.65 (0.2812)	1.60 (0.2909)
Plot Type	2.38 (0.1424)	2.30 (0.1502)	**4.56 (0.0391)**	**5.93 (0.0201)**
Grazing	1.87 (0.2294)	3.67 (0.1136)	1.02 (0.3586)	**14.82 (0.0120)**
Year	**19.60 (< 0.0001)**	**16.82 (< 0.0001)**	1.53 (0.2202)	**4.82 (0.0054)**
Soil Type x Plot Type	**4.02 (0.0338)**	2.46 (0.1129)	1.08 (0.4182)	1.57 (0.2567)
Soil Type x Grazing	1.68 (0.2762)	1.44 (0.3200)	2.92 (0.1447)	**8.57 (0.0242)**
Soil Type x Year	**8.39 (< 0.0001)**	**5.43 (0.0003)**	**4.22 (0.0019)**	**6.59 (< 0.0001)**
Plot Type x Grazing	**3.04 (0.0932)**	1.59 (0.2514)	1.89 (0.2007)	**4.26 (0.0460)**
Plot Type x Year	**13.95 (< 0.0001)**	**10.72 (< 0.0001)**	**6.94 (< 0.0001)**	**10.95 (< 0.0001)**
Soil Type x Plot Type x Grazing	0.91 (0.4938)	0.58 (0.6850)	**3.18 (0.0630)**	**6.54 (0.0075)**
Soil Type x Plot Type x Year	**3.64 (0.0008)**	**2.89 (0.0049)**	1.38 (0.2118)	**2.22 (0.0270)**
Grazing x Year	0.51 (0.6790)	0.81 (0.4973)	**2.43 (0.0779)**	0.51 (0.6757)
Soil Type x Grazing x Year	0.89 (0.5068)	0.97 (0.4559)	1.76 (0.1291)	1.61 (0.1662)
Plot Type x Grazing x Year	1.26 (0.2950)	0.64 (0.6940)	1.78 (0.1246)	**2.59 (0.0305)**
Soil x Plot x Grazing x Year	**1.71 (0.0966)**	1.31 (0.2451)	0.98 (0.4832)	**2.19 (0.0286)**

VITA

After graduating from Waxahachie High School, Waxahachie, Texas, in 2011, Elena Thomas

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