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Abstract

The Intermountain West is a critical region for ecosystem services, providing habitat for diverse wildlife and pollinator species, maintaining vegetation biodiversity, and sequestering carbon in forests and rangelands. However, historical land management practices have led to challenges in sustaining these ecosystems. In the face of declining global pollinator populations and fragmented sagebrush (*Artemisia* spp.) ecosystems, preserving and restoring these areas is crucial. My study aimed to assess the impact of common land management practices, such as prescribed burning, tree and brush removal, and seeding, while evaluating soil carbon and nitrogen, plant biodiversity, carbon sequestration, and pollinator abundance and diversity in the Intermountain West. I conducted a comprehensive meta-analysis of peer-reviewed studies spanning a century, which addresses key questions about the overall effect of these practices on ecosystem services. My results revealed that mechanical treatments positively impacted pollinator abundance, while cut and leave and burn treatments had no significant effect. However, shrub abundance decreased under disturbance, and annual exotic forb and grass abundance responded positively to disturbance caused by treatments. Despite varying responses to treatments, none of them substantially affected most measured variables. My study underscores the importance of understanding mechanisms and long-term impacts of these treatments on ecological communities. As the region is predominantly comprised of public land, this research can be used to inform decision-making and resource management for all stakeholders. Future research is crucial to delve deeper into impacts and ensure the continued protection of this ecological region.

In Chapter 1, I conducted a comprehensive literature review that looked at the concepts of ecosystem services within rangeland ecosystems. This chapter highlights the crucial role ecosystem services have in sustaining human life and wellbeing. It discusses the various factors influencing ecosystem services, such as land management practices, disturbance regimes, invasive species, and climate change. Chapter 1 focused on the importance of understanding and managing ecosystem services for biodiversity, resource sustainability, and human welfare. This chapter also evaluated the complexities behind valuing ecosystem services and the challenges associated with their decline. It examines the impacts on different ecosystem services such as carbon sequestration, pollinator abundance, and plant biodiversity.

In chapter 2, I evaluated the long-term impacts on various land management practices on ecosystem services within a sagebrush steppe ecosystem. The study focused on empirical data collected in Glade Park, Colorado. The areas that were evaluated had common land management practices implemented such as tree and brush removal, prescribed burning, and seeding. To assess the soil carbon and nitrogen levels, perennial grass and shrub heights, plant cover, species richness, and diversity measurements were taken at 12 different transects. The prescribed burning sites resulted in the lowest woody cover but the highest total soil carbon. The seeding site resulted in the highest annual grasses found within any transect. The findings from Chapter 2 underscore the many effects of these land management practices on ecosystem services and discovered both positive and negative impacts.

In Chapter 3, I conducted a meta-analysis of peer-reviewed literature to further investigate the relationship between land management practices and ecosystem services. The analysis focused on brush and tree removal, seeding, and prescribed burning. I examined these treatment impacts on plant biodiversity, pollinator habitat, and carbon sequestration. Results from this chapter indicate that disturbance alone can increase annual exotic plants and have a negative impact on ecosystem services. When I evaluated disturbance and seeding treatments, I found that they do not substantially alter the effects. When considering pollinator abundance, I found that mechanical treatments had a positive correlation with pollinator abundance, while the cut and leave or prescribed burning treatments did not have any noticeable effects. Chapter 3 highlights the complex dynamic involved in ecosystem service response to common management practices in the Intermountain West.

ASSESSING RANGELAND MANAGEMENT PRACTICES AND ECOSYSTEM SERVICES WITHIN THE INTERMOUNTAIN WEST

By

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CHAPTER ONE: SUSTAINING ECOSYSTEM SERVICES IN RANGELANDS OF THE US INTERMOUNTAIN WEST: IMPACTS AND CHALLENGES OF LAND MANAGEMENT PRACTICES

ABSTRACT

Ecosystem services play a critical role in sustaining human life and are increasingly recognized as essential for human wellbeing. This study evaluated the challenges associated with valuing ecosystem services and the consequences of their decline, while emphasizing the need for integrated approaches to land management that consider the collective impacts on multiple ecosystem services. It is important moving forward in land management that stakeholders have an understanding how different land management practices affect ecosystem services, such as carbon sequestration, pollinator abundance, and plant biodiversity. This study underscores the necessity of incorporating such knowledge into the decision-making processes for sustainable rangeland management.

INTRODUCTION

Ecosystem services are the natural function within ecosystems that help sustain human life (Robertson, 2012). Ecosystem services encompass both tangible material products and intangible services that are vital for human survival, health, and overall wellbeing (Hasan et al. 2020). The concept of ecosystem services has been referred to as a bridge between human fulfillment and the natural environment (Grêt-Regamey et al. 2012). Understanding and managing these ecosystem services are crucial for maintaining biodiversity, sustaining resources, and ensuring a healthy environment for present and future generations. The management of ecosystem services is critical for maintaining biodiversity and the production of ecosystem goods, such as food and timber, forage production, natural fiber, pharmaceuticals, and other industrial products (Millennium Ecosystem Assessment, 2005). Examples of ecosystem services include pollination, water quality, microbial decomposition, clean air, wildlife habitat, aesthetic qualities, and soil water infiltration (Robertson, 2012). These ecosystem services are often binned into four major categories: provisioning services, cultural services, regulation services, and supporting services (Shaad et al. 2022, Millennium Ecosystem Assessment, 2005). Rangelands produce a variety of important ecosystem services, including the provisioning of food and fiber, carbon sequestration, maintenance of biodiversity (conservation), and recreation (Sala and Paruelo 1997). Humans directly rely on ecosystem services for survival and in turn, that impacts the way in which we manage and maintain ecosystem services (Abson and Termansen 2011). With the increase in human populations there is a growing need to prevent further degradation of ecosystem services and land (Turner et al. 2016). Understanding the effects of various activities, such as land management practices, on rangeland ecosystem services is an important part of the stakeholder decision making process (Yahdjian et al. 2015). It is important to understand the challenges associated with valuing ecosystem services and the consequences of their decline.

Rangeland ecosystems have the potential to increase or decrease human satisfaction and in turn this creates a value (Maczko et al. 2011). Yet, how to sustainably and simultaneously manage all ecosystem services that are contributing to the wellbeing of humans is an open question (Costanza et al. 2014). Ecosystem services can be valued in different ways including willingness to pay (Maczko et al. 2011), projected land use, (Kubiszewski et al. 2017), and benefit transfer (Costanza et al. 2014). Willingness to pay assessments typically requires a survey that assesses the dollar amount respondents are willing to pay for an ecosystem service

(Maczko et al. 2011). Valuing ecosystem services can be challenging; some services have no direct or indirect material benefit, such as cultural services (Small et al. 2017), while other ecosystem services are necessary for human survival such as food (Small et al. 2017). A study conducted by Kubiszewski et al. (2017) mapped 16 different biomes for land use at a 1-km^2 resolution, which enabled the authors to create an estimated value for the global ecosystem. Kubiszewski et al. (2017) found that the global value for ecosystem services could decline by $51 trillion per year or be increased by $30 trillion per year depending on land management techniques. The range of these estimates highlights the possibilities for management to affect ecosystem service provision. Benefit transfer occurs when management decisions trade one resource for another (Costanza et al. 2014). A benefit transfer study performed by Costanza et al. (2014), determined there was a global loss between $4.3 and $20.2 trillion per year in ecosystem services from 1997 to 2011. These values are lower than what Kubiszewski et al. (2017) estimated, because Kubiszewki et al. (2017) evaluated a spectrum of possibilities to determine global loss while Costanza et al. (2014) conducted a benefit transfer estimate. Although different, these two assessments create a starting point for assessing the value of ecosystem services across a large area. However, the "benefits transfer" method assumes every hectare of habitat is of equal value, and this approach does not allow for proper assessment if the area is altered by agriculture or through a land management practice (Nelson et al. 2009). A study conducted by Díaz et al. (2018) analyzed the state of global biodiversity and ecosystem services and found that human activities have caused a significant decline in both. Similarly, another study by (Groot et al. 2010) demonstrated the economic value of ecosystem services and the costs of their degradation. Fletcher et al. (2024) emphasized the importance of ecosystem services within rangeland ecosystems, and the opportunities available to integrate ecosystem service valuation into land management planning. They found an increase in the non-market value of ecosystem services when looking at brush management, herbaceous weed treatment, and prescribed grazing (Fletcher et al. 2024). It is important to recognize the ecosystem benefits of conservation practices on grazing lands, and how people rely on these ecosystems for recreation, and food production (Fletcher et al. 2024). Regardless of the valuation method, however, understanding how land management practices affect ecosystem services is critical for planning future management in rangelands.

Sagebrush (*Artemisia* spp.)-steppe ecosystems and shrub/grasslands comprise 29% of land cover in the US and are an important provider of many rangeland ecosystem services (Reed et al. 2018). Sagebrush ecosystems are among the largest ecosystems in North America to experience negative impacts after Euro American settlement (Shinneman et al. 2023). Sagebrush ecosystems originally covered > 500,000 km^2, but agriculture, human development, and altered fire regimes have caused ~45% of the historical ecosystem to be lost (Miller et al. 2011). Since the mid-1800's pinyon (*Pinus*)-juniper (*Juniperus*) woodlands have been encroaching into sagebrush (*Artemisia* spp.)-steppe shrublands and grasslands and they now comprise 40% of the total forest and woodland area of the United States Intermountain West (Filippelli et al. 2020). Sagebrush vegetation communities occupy 56% of their historical range and are often highly fragmented (Schroeder et al. 2004). Pinyon-juniper ecosystems now cover over 40 million hectares, making them the third largest vegetation type in the United States (Filippelli et al. 2020). Recently, sagebrush ecosystems have been decreasing across Colorado and Wyoming due to conifer and invasive plant species encroachment (Reinhardt et al. 2020). This change in vegetation type has caused cascading effects to many ecosystem services such as habitat quality for wildlife (Woods et al. 2013, Donovan et al. 2024). Much of this woodland expansion has been driven by anthropogenic activity, which affects both top-down processes, such as grazing and altered fire regimes (Reinhardt et al. 2020), and bottom-up processes, such as altered resource availability (Kormos et al. 2017). A particularly important impact of the encroachment of native conifers (*e.g.*, pinyon-juniper) into western sagebrush-steppe rangelands is the negative effect on sagebrush-dependent wildlife (USDA-NRCS 2019).

Alteration to the vegetation component within sagebrush-steppe ecosystems is changing the habitat available for wildlife, specifically the greater sage-grouse (*Centrocercus urophasianus*) and the Gunnison sage-grouse (*C. minimus*), hereafter collectively referred to as 'sage-grouse.' Sage-grouse declines have become an indicator for changes in ecological biodiversity (Reinhardt et al. 2020). Baruch-Mordo et al. (2013) indicated that sage-grouse often abandon their breeding grounds when pinyon-juniper canopy cover reaches 4% in sagebrush-dominated landscapes in eastern Oregon. This has instigated many land management practices within sagebrush communities to reduce conifer abundance and limit non-native species from establishing. Some of the treatments that have been used within sagebrush ecosystems include brush and tree removal, seeding, and prescribed burning. These land management practices have been found to increase native plant establishment and in turn increase wildlife habitat (Reinhardt et al. 2023). Recent studies have shown that after encroached conifers are removed, sage-grouse occupancy and nest survival is improved (Severson et al. 2017). Studies also show that populations of sagebrush songbirds increase following the removal of conifer that has en-croached into sagebrush (Crow et al.2010). Quantifying the impacts of ongoing conifer removal efforts on other ecosystem services, such as biodiversity and carbon sequestration, is one of the challenges in pinyon-juniper management and sagebrush habitat restoration (Reinhardt et al. 2020).

Land management practices have caused many alterations to plant biodiversity within rangeland ecosystems (Plantureux et al. 2005). Livestock grazing (Starns et al. 2019), fire (Uys et al. 2014), introduction of invasive plant species (Bell et al. 2020), urbanization (Deng et al. 2021), and seeding (Davies et al. 2015) can all impact ecosystem services. With the recent changes in land use and management, many rangelands and grasslands have seen an increase in exotic annual grasses. These annual grasses can be a threat to biodiversity, wildlife habitat, ecosystem function, and livestock production (Davies et al. 2013). The introduction and spread of invasive species can have negative impacts on ecosystem services (Jeschke et al. 2014). When land managers combine fire and grazing, fire regimes have been shown to increase the occurrence of invasive species (Condon and Pyke 2018). This has led to an increase in invasive plants around the world (Rohde et al. 2019). To reduce invasive plant species after a fire, land managers often reseed the disturbed area. If reseeding efforts are successful in re-establishing the desired plant community, this can reduce establishment and survival of invasive plants (Chen et al. 2012) and increase ecosystem services. Seeding postfire not only helps reduce the chance of invasive plants but it can quickly stabilize the soil and increase water infiltration (Chen et al. 2012). Sheley and Bates (2008) found that after western juniper (*J. occidentalis*) was removed from an area that was predominately sagebrush/bunchgrass and snowberry (*Symphoricarpos* spp.)/fescue (*Festuca* spp.), native species establishment was high (bluebunch wheatgrass (*Pseudoroegneria spicata*), Idaho fescue (*Festuca idahoensis*), Sandberg bluegrass (*Poa secunda*), and western yarrow (*Achillea millefolium*)). However, Trowbridge et al. (2017) found that it can be challenging to properly distribute seeds and maintain exotic grass encroachment. Results from a study conducted by Stanley et al. (2010) showed that supplemental seeding was necessary after the initial seeding to increase native species. Additionally, the encroachment of non-native species into natural habitats, driven by anthropogenic activity (Manier et al. 2014), has been shown to negatively impact ecosystem services (Kumar and Singh 2020). These findings emphasize the importance of proper land management practices to re-establish desired plant communities and diversity post disturbance.

One particularly important ecosystem service provided by diverse rangelands is pollinator abundance and diversity (Orford et al. 2016). Disturbance and ecological change have led to a decline in ecosystem services, specifically pollinators (Mathiasson and Rehan 2020). Many plant populations are dependent on a healthy pollinator population because pollinators assist with the seed and fruit formation necessary for plant reproduction (Kearns and Inouye 1997, Rohde et al. 2019). As such, the loss of pollinator species has created a large concern for biodiversity and the success of natural and agricultural systems (Kearns and Inouye 1997, Biesmeijer et al. 2006). Black et al. (2011) found that 70% of bee species nest in the ground and 30% nest within plants or old trees. If this nesting habitat experiences mowing or burning, it could lead to a severe decline of pollinator species in an area (Black et al. 2011). Rohde et al (2019) found that burned areas that were seeded supported different insect assemblages compared with areas that were burned and left to recover with no supplemental seeding. However, seeding post fire did not always create a more stable environment for insect communities to reestablish (Rohde et al. 2019). Short fire intervals have also been shown to threaten some pollinators, especially lepidopterans (Carbone et al. 2019). Pollinator species have different habitat requirements for success, and this makes understanding the relationship between all pollinator species and land management practices of high importance for maintaining ecosystem service provision in rangelands.

Rising levels of carbon dioxide (CO2) within the atmosphere has become a growing concern and has led to the interest in carbon sequestration (Boerner et al. 2008). Although carbon sequestration is often less per capita in grassland versus forest ecosystems, the large proportion of the terrestrial land surface covered by grasslands makes them an important contributor to global carbon sequestration (Dass et al. 2018). Changes within our climate are affecting the capacity of ecosystems to sequester carbon, which is essential for mitigating greenhouse gas emissions. Climate change is altering the rates of carbon storage in terrestrial ecosystems, leading to a net loss of carbon (Canadell and Raupach, 2008). A primary mechanism for climate change effects on carbon sequestration is through effects on plant growth and composition (De Deyn et al. 2008, Fekete et al. 2017).

Plant community dynamics can play an important role in altering carbon sequestration in rangelands in the Intermountain West (Fernandez et al. 2013). Carbon sequestration has been found to increase when forest cover increases within a grassland (Fernandez et al. 2013). When areas of woody plant encroachment are disturbed from land management treatments such as brush management and prescribed burning, they can significantly reduce the amount of aboveground carbon that is stored (Abson and Termansen 2011). A study conducted by Fernandez et al. (2013) found that if woodland stands experience death from wildfire, more than 25% of the carbon stabilized in the past century could return to the atmosphere. Invasive plant species have also been found to alter carbon storage within an ecosystem. For example, invasion of cheatgrass (*Bromus tectorum*) within a sagebrush steppe ecosystem has been found to reduce belowground organic carbon due to a change from deep-rooted perennial vegetation to shallow-rooted annual grasses (Rau et al. 2011). If invasive annual plant species increase in abundance in the Intermountain West, re-seeding after disturbances may lead to more carbon sequestration from native perennials. Boerner et al. (2008), found that western coniferous forests responded differently to fire and fire surrogates (mechanical treatments designed to simulate prescribed fire) than mixed coniferous forests and eastern deciduous forests. These results suggest that the effects of land management practices are complex and that there may be no single treatment that will be beneficial for all ecosystem services. Therefore, information about the collective effects of land management practices on various ecosystem services should be matched up with site-specific management goals.

Recent focus has been paid to simultaneous development of multiple ecosystem services (Bennett et al. 2009, Gamfeldt et al. 2013, Tamburini et al. 2020) rather than focusing on a single service alone. Indeed, undesirable declines in some ecosystem services may occur when there are tradeoffs among services and when management focuses on only one service at a time (Millennium Ecosystem Assessment 2005). Focusing on one ecosystem service at a time has not only been a downfall within land management but also within research. Many studies ignore the possibility that ecosystem services covary with one another (Bennett 2009). To develop a better understanding of the current state of ecosystem services and the collective impacts of land management practices, we must simultaneously evaluate multiple ecosystem services to determine tradeoffs among different services. Below, I outline my methods that provide insight into an applicable assessment of land management practices to represent how they are altering various ecosystem services.

PROBLEM STATEMENT & OBJECTIVES

Rangelands provide many ecosystem goods and services, and quantification of long-term impacts of common management practices in the US Intermountain West is needed. I evaluated the impacts of various land management practices on multiple ecosystem services within US Intermountain West rangeland ecosystems. I began by synthesizing current scientific knowledge and highlighting the importance of sustainable management practices to ensure the provision of critical ecosystem services. I focused on sagebrush steppe ecosystems that have experienced one or multiple management practices (e.g., tree and brush removal, prescribed burns, and seeding).

To establish a better understanding of how land management practices are impacting ecosystem services within this region, I quantified the potential effects of land management practices on ecosystem services by completing the following two objectives. First, I collected and analyzed empirical data about long-term impacts of various management practices in a sagebrush ecosystem in western Colorado (Chapter 2). Second, I used results from existing peer-reviewed literature to synthesize understanding about the effects of land management practices on ecosystem services throughout the Intermountain West (Chapter 3). These approaches assessed both the positive and negative effects on rangeland ecosystem services associated with various land management practices.

To accomplish Objective 1, I evaluated the impacts of the following commonly used land management practices in a sagebrush ecosystem near Glade Park, Colorado: tree and brush removal, prescribed burning, and seeding. Within areas that had these management practices applied 2-12 years ago, I measured the following ecosystem services: soil carbon and nitrogen, perennial grass and shrub heights, plant cover, plant species richness, and plant species diversity.

To accomplish Objective 2, I conducted a meta-analysis of the scientific peer-reviewed literature to assess impacts of various land management practices on multiple ecosystem services. The land management practices I focused on included brush and tree removal, seeding, and prescribed burning. The ecosystem services I included were plant biodiversity, pollinator habitat, and carbon sequestration.

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CHAPTER TWO: EFFECTS OF LAND MANAGEMENT PRACTICES ON ECOSYSTEM SERVICES IN WESTERN COLORADO

ABSTRACT

This study evaluates the impacts of common land management practices on plant cover, biodiversity, soil carbon, and soil nitrogen in a sagebrush ecosystem in western Colorado. I selected 12 locations based on similar elevation, precipitation, and vegetation type with three transects in each of the treatments: prescribed burning, tree and brush removal, seeding, and a control. I collected measurements to analyze the percent cover of vegetation, bare ground, and litter using the line point intercept (LPI) method. Additionally, I recorded shrub and perennial grass heights, as well as soil carbon and nitrogen levels. Statistical analysis revealed significant differences among treatments for shrub height, soil carbon levels, annual grass cover, woody species cover, species diversity, and richness. Prescribed burn sites exhibited the shortest shrub height and highest soil carbon levels, while seeding treatments had the greatest annual grass cover. These results indicate that seeding treatments have the potential to bring in unwanted invasives. My study highlights the complexity of interactions between land management practices and ecosystem dynamics in a sagebrush (*Artemisia* spp.) ecosystem and provides insight for sustainable land management strategies moving forward.

INTRODUCTION

It is estimated that 2 billion people worldwide rely on products from rangelands (Sayre et al. 2013, Briske et al. 2015, Davis et al. 2020). Rangelands within western Colorado provide many ecosystem services to the area. These ecosystem services include, but are not limited to, wildlife habitat, carbon sequestration, cultural importance, plant biodiversity, water filtration, and livestock grazing (Kandziora et al. 2013). Rangelands within western Colorado provide wildlife with land continuity for winter and summer migration, and suitable habitat for survival (Schroeder et al. 2004, Stephens et al. 2016). Rangelands occupy around half of the world's land area, and they have the potential to globally store more than 10% of above ground carbon and 30% of below ground carbon (Derner and Schuma 2007). In turn, rangelands can reduce the levels of atmospheric carbon. Rangeland ecosystems can also provide a place to preserve cultural resources for present and future generations. Plant biodiversity within rangeland ecosystems can increase biomass yields (Schaub et al. 2020) while conifer encroachment has been found to have a negative impact on wildlife habitat (Woods et al. 2013). Understanding the drivers of these ecosystem services within rangelands will be critical for maintaining these benefits to society throughout the 21st century.

With changes in public land management priorities, land managers have been implementing different vegetation management techniques. From the 1950s through the 1970s tree-reduction treatments involving chaining, bulldozing, or cabling were most prevalent, with over 163,000 ha of land treated with these methods in the western United States (Redmond et al. 2014). Prescribed burning became increasingly prevalent in the 1980s, with over 43,000 ha burned throughout the western United States (Redmond et al. 2014). In more recent years, hydro axe treatments have become common (4,400 ha treated) within the western United States, but to a much lesser extent than prescribed burns between 1950-2003. Over 60% of these tree-reduction treatments were done in conjunction with revegetation or seeding treatments (Redmond et al. 2014). A study by Johnson et al. (2010) found that long-term responses of grasses and forbs to low-severity fire varied depending on the initial vegetation cover, and ensuing management practices have been found to significantly affect post-fire vegetation recovery. Management techniques such as tree and brush removal, prescribed burning, and seeding can have varied impacts on the ecosystem services present within rangelands. Prescribed burning has been shown to reduce conifer encroachment but can also lead to more non-native and invasive plant species (Davies et al. 2019). Prescribed burning has the potential to initially reduce some wildlife presence but not all (Russell et al. 2009) prescribed fire can have a positive, negative, or neutral impact on wildlife (Block et al. 2016). Burning has been shown to have a negative effect on some butterfly species, and a neutral or positive effect on bee populations (Glenny et al. 2022). Tree and brush removal can reduce conifer encroachment, increase suitable wildlife habitat, and increase herbaceous vegetation (Severson et al. 2017). If the appropriate seed mix is used, seeding can increase perennial vegetation after disturbance, and it can reduce the establishment of exotic or non-native plant species (Davies et al. 2018). Seeding can also increase plant cover and forbs available for wildlife and livestock (Summers and Roundy 2018). Rangeland ecosystems within the United States have been altered by fire suppression, over grazing, and the introduction of nonnative plant species (Mclver et al. 2014). To help restore rangeland ecosystems that have been altered, public land managers have implemented multiple techniques including tree and brush removal, prescribed burning and seeding. It is estimated that over 247,000 hectares of land were treated with tree reduction treatments (chaining, hydro axing, prescribed burning), corresponding to 6.6% of the pinion–juniper vegetation type within Bureau of Land Management (BLM)-managed lands between 1950-2003 (Redmond et al. 2014). The Natural Resource Conservation Service (NRCS) has provided agricultural producers with conservation measures and funding assistance on approximately 409 million acres of non-federal rangelands. Data from the CEAP-Grazing Lands shows that the NRCS invests an average of $71 million each year for federal land conservation assistance (Fletcher et al. 2024).

Studies on rangeland restoration have explored the impacts of management practices on post-fire vegetation recovery and the response of grasses and forbs, highlighting the significance of management techniques in influencing recovery (Davies et al., 2015, Boyd et al., 2019). The implementation of management practices such as tree and brush removal, prescribed burning, and seeding can have diverse effects on ecosystem services provided by rangelands (McIver et al., 2015, Davies et al., 2018). Prescribed burning, for example, has the potential to reduce conifer encroachment but may initially decrease wildlife and pollinator habitat due to the lack of suitable vegetation. However, as vegetation regrows, wildlife and pollinators are likely to return (Davies et al., 2018, Stambaugh et al., 2020). Tree and brush removal can mitigate conifer encroachment, enhance wildlife habitat, and promote plant species diversity (Davies et al., 2015, McIver et al., 2015). Seeding practices, when appropriately executed, can either increase or decrease plant species diversity and minimize the establishment of non-native or invasive species. Moreover, seeding can enhance plant cover, benefiting both wildlife and livestock (Boyd et al., 2019, James et al., 2021).

Land management agencies are interested in the effectiveness of these practices at achieving desired outcomes because they spend significant funds on reclamation activities. An example of these government programs is the Environmental Quality Incentive Program (EQIP), which was designed to address natural resource concerns and deliver environmental benefits such as improved water and air quality, conserved ground and surface water, increased soil health and reduced soil erosion and sedimentation, improved or created wildlife habitat, and mitigation against increasing weather volatility. This is a voluntary program for land managers that encourages conservation. Using EQIP agriculture, producers can help to increase clean water and air, healthy soils, and increase wildlife habitat, while improving agriculture operations (USDA-NRCS, 2020).

ASSESSMENTS AND PREDICTIONS

In this chapter I examined the effects of tree and brush removal, prescribed burning, and seeding treatments on ecosystem services in a big sagebrush (*Artemisia tridentata*) ecosystem in western Colorado. Specifically, I measured how these management practices have altered diversity and average cover (%) in woody species, forbs, C3-perennials, C4-perennials, annual grass, grass and shrub heights, and carbon and nitrogen levels within the soil. This assisted me to address the overarching objective of this chapter: <u>to determine how various common land management practices are altering vegetation cover, plant species diversity, and carbon and nitrogen levels within the soil in a Colorado sagebrush ecosystem.</u> To address this overall objective, I tested the following predictions:

I predicted that sites undergoing land management practices will have a greater plant diversity than reference sites (i.e., those that did not undergo land management practices).

1) Seeded areas will have the greatest plant diversity.
2) Prescribed burned plots will have the lowest amount of woody cover and the greatest diversity of forbs present.
3) Tree and brush removal sites will have the greatest cover of C3 perennial plants.
4) Control sites will have the greatest soil nitrogen and carbon.

METHODOLOGY

Quantifying Investments in Ecosystem Service Provisions

To demonstrate the dollar amount agencies are contributing to land treatments within Colorado, I compared county-based land management data available through the NRCS. I acquired data that were collected by the NRCS that includes the different treatments performed within each county in Colorado between 2005 and 2019. The data provided were summarized as part of (EQUIP) and then separated by county. The data included multiple treatment types, brush management, conservation cover, prescribed burning, rangeland planting, upland wildlife habitat management, and early successional habitat development/management within counties in Colorado. These data did not have vegetation types listed, so I grouped data into three categories: tree and brush removal (practice code 314), seeding (practice code 550 and 327), and prescribed burning (practice code 338). When assessing the data provided, I only evaluated treatments that were “certified” or “partially certified” within the practice status. The data collected included the total monetary amount used to implement each practice, and the resource concerns treated.

Site Description

The field-based portion of this chapter was conducted in western Colorado near the town of Glade Park in Mesa County (Fig. 2.4). The study sites were located at an elevation of 2134 m (7,000 ft) on BLM grazing allotments with little to no slope. The sites were surrounded by private land, the Colorado National Monument to the northeast, the Grand Mesa National Forest to the south, and McInnis Canyons National Conservation Area north of Glade Park. Glade Park is historically a ranching and farming community that receives, on average, 28.1 cm (11.6 in) of precipitation (Grant-Hoffman and Plank 2021). The site locations were dominated by big sagebrush. Prevalent woody plants within the area included two-needle piñon pine (*Pinus edulis*), Utah juniper (*Juniperus osteosperma*), and Gambel's oak (*Quercus gambelii*). The primary soils in the area were Alfisols, Aridisols, Entisols, and Mollisols (Kettler et al. 1996).

In previous decades, the BLM has performed various land management practices on Glade Park rangelands, including seeding, brush and tree removal, and prescribed burns (Table 2.1). In the past the BLM has seeded plant species such as crested wheatgrass (*Agropyron cristatum*) to increase forage production levels and reduce erosion.

Experimental design

I selected the 12 site locations based on similar elevation, yearly average precipitation, and vegetation type (pre-treatment). Once the locations were established, I randomly selected the exact sampling sites within the location. I selected three replicate locations for each of the following treatments: prescribed burning, tree and brush removal, seeding, and no practice (control; Figure 2.4; $n = 3$). Two prescribed burn treatments were completed in September 2009, and one was completed in September 2010 by fire personnel from the Grand Junction BLM Field Office. Tree and brush removal were completed at one site in 2008 and two sites in 2018. This treatment included handsaw, axe, Pulaski, and hand clippers. The power tools used included chain saws and power brush saws. One site was seeded in 2009 and two sites were seeded in 2013. Seeding was done by a seed drill. The seed drill opened a furrow in the seedbed, calibrated the density of seeds put into the furrow at a specified depth, and then the furrow was closed to cover the seeds. From the available data, there was no evidence that the seeded sites had been cut first, but I assumed this was the case. The seed mix included bluebunch wheatgrass (*Pseudoroegneria spicata*), thickspike wheatgrass (*Elymus lanceolatus*), slender wheatgrass

(*Elymus trachycaulus*), western wheatgrass (*Pascopyrum smithii*), Indian ricegrass (A*chnatherum hymenoides*), blue grama (*Bouteloua gracilis*), small burnet (*Sanguisorba minor*), blue flax (*Linum perenne*), milkvetch (*Astragalus* spp.), and shadscale saltbush (*Atriplex confertifolia*; Table 2.1).

Within each treatment area, I established one 50-meter transect within the treatment boundaries. Each transect was treated as an independent 'site' for the remainder of this chapter. At each site, I measured line point intercept (LPI), shrub/tree and perennial grass heights, and below ground carbon and nitrogen levels.

DATA COLLECTION

Vegetation

I conducted vegetation sampling in late-May and early-August 2019. Along each 50-m transect, I measured the total percent coverage of vegetation, bare ground, and litter using LPI. I obtained LPI measurements by dropping a pin at every meter along the 50-m transect (Fig. 2.5). I recorded all vascular plant species and ground cover (e.g., moss, lichen, bare ground, rock) touching the pin. If present, I recorded any overhead vegetation (i.e., tree or tall shrub cover, >1.5 m) that was directly above the pin within the "Top Canopy Cover." If no overhead vegetation was present, I recorded the first leaf, stem, or plant base intercepted as the "Top Canopy Cover." If no vegetation, leaf, stem, or plant base was intercepted, I recorded "NONE" in the "Top Canopy Cover." If there was vegetation within the "Top Canopy Cover" I recorded all other vegetation intercepted within the "Lower Canopy Cover." The "Lower Canopy Cover" was allowed to have multiple plant species occurring at a single point, but each species was only recorded once at each pin (*i.e.*, plant species were only recorded once even if the same plant crossed the pin flag several times, Figure. 2.5, B). I then converted these pin hits into percent cover by species. To calculate the species-level percent vegetation cover, I summed the total number of hits for each plant species within the “Top Canopy Cover” of the transect. I then divided the number of “Top Canopy Cover” intercepts by the number of points and multiplied by 100, providing percent vegetation cover by species. I summed species-level percent cover estimates to obtain total vascular vegetative cover. To calculate the percent bare ground within a transect, I counted the total amount of bare ground hits, divided it by the number of points and multiply by 100. Bare ground was accounted for when no vascular vegetation or ground cover is intercepted by the pin flag. To calculate the litter within a transect, I counted the total amount of litter hits, divided it by the number of points and multiplied by 100. Litter was only documented as present within the “Lower Canopy Cover” of the pin flag drop, it was not accounted within “Soil Surface” estimates. These methods were adapted from the *Monitoring Manual for Grassland, Shrubland, and Savanna Ecosystems* (Herrick et al. 2009), and the BLM-*Sage-Grouse Habitat Assessment Framework* (Stiver et al. 2015).

To determine shrub/tree and perennial grass heights within each transect, I measured the tallest perennial grass species and shrub or tree within a 1-m radius of the line point at every fifth meter within the transect, resulting in 30 height measurements per treatment type. I measured each plant species as they stood naturally, and I only included the vegetation, no flowering stalks were included. Heights were then averaged by functional group to get one mean perennial grass and shrub/tree height measurement per transect.

Soil Carbon and Nitrogen

I conducted soil sampling in late August. I collected soil samples at each of the established plots. Along each 50-m transect, I sampled 0-10 cm of soil using a 1.91-cm internal diameter soil core at every fifth meter along the 50 m transect, totaling 10 soil samples. I then combined soil samples and homogenized them for each transect. I then removed plant matter within each soil sample using a 2-mm soil sieve, ground the soil with a ball grinder for 3 minutes, and measured carbon and nitrogen levels using a LECO CN analyzer (TrueSpec CN Carbon Nitrogen Determinator model number: PI2250, serial number: 21809271). I subsampled soil (ca. 0.15 g) from soil samples and wrapped them in aluminum foil. Finally, I ran the samples in the LECO CN analyzer. I ran a duplicate every 10 samples to ensure the machine was working properly. I ran a blank every six samples to clear the chamber within the instrument and insure proper calibration.

Statistical Analysis

I compared vegetative cover, litter, bare ground, plant species cover, soil carbon, and soil nitrogen among land management and control treatments using one-way type 3 analysis of variance (ANOVA). I assessed Tukey-adjusted differences among treatments using multiple comparisons testing with the ‘emmeans’ package (Lenth et al. 2023). I examined model residuals to assess ANOVA assumptions of normality and homogeneity of variance, and all models were found to satisfy these assumptions. I conducted all analyses in R (R core team, 2018) and considered results significant when p-values exceeded alpha = 0.05.

RESULTS

Land Management Practices

Figures (2.1, 2.2, and 2.3) illustrate potential impacts on land management practices and the average dollar amount spent on treatments (seeding, tree and brush removal, and prescribed burn) within Colorado. These figures are aligned with the BLM land management sites.

Height

There were no differences in grass height among the treatment sites ($F_{3,8} = 1.54$, $P = 0.279$). Mean grass height was tallest in the control sites (17 cm [SE = 0.36]). The shortest recorded mean grass height was 12 cm (SE = 3.07) within the prescribed burn sites. Additionally, the seeding site displayed a mean grass height of 13 cm (SE = 1.36), while the tree and brush removal sites exhibited a mean grass height of 14 cm (SE = 0.41).

Shrub height differed among treatment sites ($F_{3,8} = 11.33$, $P = 0.003$; Fig. 2.7). The prescribed burn sites had the shortest mean shrub height at 1 cm (SE = 0.40), which was significantly different from control ($P < 0.01$), seeding ($P = 0.04$), and tree and brush removal ($P = 0.04$). Control sites featured the tallest shrub height, with a mean of 21 cm (SE = 4.56) yet did not differ from shrub height in seeding and tree and brush removal treatments (Fig. 2.7).

Soil Carbon and Nitrogen

There were no significant differences among treatment types for mean percent soil nitrogen at the sites (Fig. 2.8, $F_{3,8} = 0.696$, $P = 0.580$). Soil nitrogen levels (Fig. 2.8) yielded mean percent averages of 0.20% (SE = 0.04) at control sites, 0.23% (SE = 0.01) within prescribed burn sites, 0.19% (SE = 0.05) in seeding sites, and 0.17% (SE = 0.001) in tree and brush removal sites. However, there was a difference in soil carbon levels among treatments ($F_{3,8} = 4.45$, $P = 0.041$; Fig. 2.8). Soil carbon averaged 1.42% (SE = 0.66) in the control treatment, 3.64% (SE = 0.41) in the prescribed burn treatment, 1.53% (SE = 0.76) in the seeding treatment, and 1.13% (SE = 0.01) in the tree and brush removal treatment (Fig. 2.8). Specifically, soil carbon was higher at prescribed burn sites compared to tree and brush removal treatments (Fig. 2.8, P = 0.047). No other treatment types differed when compared to one another.

Average Species Cover

There were significant differences in annual grass cover among treatments ($F_{3,8}$ = 12.84, P = 0.002). The highest annual grass cover was found within the seeding treatment at 32% (SE = 0.08). The prescribed burn (P = 0.004) and tree and brush removal treatments (P = 0.004; Fig. 2.9) did not have any annual grasses present within the sampling sites. Similarly, C4-perennial grasses were only found within the seeding treatment, accounting for 2.5% (SE = 0.02) of total cover ($F_{3,8}$ = 1.00, P = 0.441). However, there were no significant differences among treatments for C3-perennial grass cover ($F_{3,8}$ = 1.63, P = 0.257) or forb cover ($F_{3,8}$ = 0.91, P = 0.475). Although, there was a trend toward higher cover of C3-perenial grasses and forbs in the prescribed burn treatment (P = 0.004).

Woody species cover exhibited notable variation among treatments ($F_{3,8}$ = 4.97, P = 0.030), with the highest average cover observed in the control plots at 52% (SE = 0.12; Fig. 2.9). In comparison, average woody cover at seeding treatment sites was 28% (SE = 0.04), and the tree and brush removal 26% (SE = 0.06) sites. Importantly, woody cover was lowest at the prescribed burn sites (mean = 10% [SE = 0.03]) compared to the control and prescribed burn treatment sites (Fig. 6, P = 0.021).

Species Diversity and Richness

The overall diversity among treatment types was significantly different ($F_{3,8}$ = 5.21, P = 0.027). The prescribed burn treatments yielded a Shannon diversity of 1.13, lower than the control sites at 1.71 (Prescribed Burning vs Control, P = 0.039), and the seeding sites at 1.72 (Seeding vs Prescribed Burn, P = 0.038).

There were no differences in C3-perennial diversity ($F_{3,8}$ = 0.813, P = 0.521). The seeded sites were highest in C3-perennial diversity (Shannon's index = 0.71). However, there were no differences among treatment types for forb diversity ($F_{3,8}$ = 0.883, P = 0.489), with the tree and brush removal sites exhibiting the highest forb diversity at 1.11. Woody species diversity did not exhibit any significance among treatments ($F_{3,8}$ = 0.67, P = 0.593) and was highest in the control sites at 0.82.

For species richness, the only treatment that was significantly different was the Annual Grass ($F_{3,8}$ = 4.3, P = 0.031), compared to Seeding (P < 0.01), Control (P < 0.01), and Prescribed Burn (P < 0.01). Overall richness was not different ($F_{3,8}$ = 0.846, P = 0.506), with the seeding and control treatments having the highest species richness at 8.33. C3-perennial richness did not differ among treatments ($F_{3,8}$ = 0.178, P = 0.908), and C3-perennials displayed similar richness within all treatment types, with the highest recorded within the seeding treatment at 2.67. C4-perennial richness did not differ among treatments ($F_{3,8}$ = 1, P = 0.441), and C4-perennials were only found in seeding sites. Woody vegetation richness did not differ among treatment types ($F_{3,8}$ = 1.02, P = 0.432), with the highest woody richness found within the control sites at 3.33. Forb richness did not differ among treatments ($F_{3,8}$ = 0.42, P = 0.743), with the tree and brush removal sites exhibiting the greatest forb richness at 3.3.

DISCUSSION

In this chapter I aimed to assess the impact of common land management practices on vegetation cover, plant species diversity, and carbon and nitrogen levels in a sagebrush ecosystem in western Colorado. I predicted that sites undergoing land management practices would have greater plant diversity than reference sites (i.e., those that did not undergo land management practices). This was not supported by my findings when comparing all functional groups, the seeding and control sites had the same species diversity, and the other land management treatment resulted in less species diversity. I predicted that prescribed burning sites would have the lowest amount of woody cover, which was supported by my findings. However, I predicted that the burned plots would also have the greatest diversity of forb species, but this was not supported by my results. I also predicted that the tree and brush removal sites would have the greatest cover of C3-perennial plants and my results suggest that C3-perennial plants had the greatest cover within the prescribed burned sites. Finally, I predicted that my control sites would have the greatest soil nitrogen and carbon, but my results revealed that the prescribed burn sites had the highest levels of soil carbon and nitrogen.

Plant Height and Cover

I found that shrub height was markedly lower in areas that were burned (Fig. 2.7), which coincides with findings by Ellsworth et al. (2016) who observed shorter shrub heights 17 years after a prescribed burn treatment in a sagebrush steppe ecosystem. The typical fire return intervals for Wyoming big sagebrush (*A. t.* ssp. *wyomingensis*) communities are every 50-120 or more years, and 15-25 years within a mountain big sagebrush (*A. t.* ssp *vaseyana*) (Wright and Bailey 1982) dominated ecosystem. By comparison, Baker (2006), reported even longer fire return intervals of 325-450 years in low sagebrush (*A. arbuscula*), 70-200 years in mountain big sagebrush, and 100-240 years in Wyoming big sagebrush. The prescribed burn sites exhibited the shortest shrub heights compared to any treatment (Fig. 2.7). Interestingly, Ellsworth et al. (2016) found that sagebrush started filling in interspaces 17 years after the fire in their study, which indicates it will take more time for woody cover measurements to recover at the prescribed burned sites in my study (Fig. 2.9). Woody plants including juniper have been expanding into sagebrush steppe ecosystems and tend to restrict necessary resources for herbaceous plant communities in the understory (Mclver et al. 2022). In my study, the prescribed burn plots were lowest in woody cover. The most common woody species recorded in study was big sagebrush. Tree and brush removal sites did not have the greatest cover of C3-perennial. These results did not align with my original predictions that C3-perennials would have the greatest cover within the tree and brush removal sites. I predicted C3-perennials would have the greatest cover within the tree and brush removal sites because they would have less disturbance compared to the prescribed burning and seeding. Sites that had been prescribed burned had the greatest cover of C3-perennials, this may have been due to the seed bank that was previously in the soil before the burn.

Prescribed burning and thinning within a sagebrush ecosystem can often lead to more invasion and dominance by exotic annual grasses (Owen et al. 2009, Ross et al. 2012, Davies et al. 2021), which coincides with reductions in woody cover. I found that woody cover decreased with burning; however, I did not find corresponding increases in annual grasses. This may be due to a general lack of annual grasses present in most of these sites prior to treatment, effectively limiting the spread of annual grasses under local disturbance. However, there was much greater cover of annual grasses in the seeding treatments, which was entirely composed of cheatgrass. Because cheatgrass was not found within any of the other treatment types it may have been brought in by the seeding equipment used, which highlights the importance of limiting spread of invasive species during management efforts. It remains to be seen whether annual grass will spread to nearby areas that were burned or mechanically treated.

Prescribed burn sites had greater cover of forbs compared to the other sites, which coincides with a previous study that found annual forbs to increase post burn (Huffman et al. 2014). The increase of forb species may be a direct impact of fewer shrubs and trees present within the site opening space for establishment. The most common forb found within the prescribed burn sites was an introduced annual, desert madwort (*Alyssum desertorum*). Notably, the seeding site was occupied by blue grama, the only C4-perennial grass present within the sampled area.

Plant Species Diversity and Richness

Species diversity patterns across treatments did not align with my original predictions that the sites undergoing land management practices will have the greatest plant diversity compared to the reference sites (i.e., those that did not undergo land management practices). I found that when looking at all species diversity (Fig. 2.10), the disturbed sites, inclusive of prescribed burn and tree and brush removal, had the least amount of species diversity. This finding does not fit with the abundance of literature showing that plant species diversity increases with disturbance (Ye et al. 2022, Petraitis et al. 1989, Collins et al 1995, Dial et al. 1988). This may be because these are sensitive ecosystems and many of the species present rely on nurse plants. Although not statistically significant, the seeded and control sites were also found to have a high plant diversity compared to the tree and brush removal and the prescribed burn sites, which suggests that a certain level of seedling success is occurring from seed mixtures. The seeded sights were found to have the greatest diversity of C3-perennials and the least diversity of woody vegetation. My study revealed that the seeded sites had the highest C3-

perennial richness of any treatment. A study conducted by Munson and Lauenroth (2011), found that the number of C3-perennials, C4-perennials, and overall recovery of their sites depended on the seed mix along with climatic variability. Furthermore, the tree and brush removal and prescribed burn sites had the highest number of native C3-perennial grass species, indicating a potentially positive impact on wildlife and grazing potential (Barbehenn et al. 2004).

My findings indicate that species richness was highest in the control and seeding treatments, consistent with Carter and Blair (2012) who reported increased species richness in seeded sites. Furthermore, woody species richness was found to be highest in the control sites, aligning with the observed percent cover results (Fig. 2.11). The lack of disturbance in the control sites allowed tree and shrub populations to expand and increase in richness. These results suggest that there was not a strong effect between diversity and richness of any treatment when splitting among functional groups.

Soil Carbon and Nitrogen

Although I did not find any differences among treatment types for soil nitrogen levels, there were strong differences among treatments for soil carbon. Prescribed burn and mechanical removal are often used interchangeably to remove aboveground biomass, it seems as if their effects on soil carbon were quite different. The prescribed burn sites were highest in mean percent soil carbon (3.64%), which does not correlate with findings from Nicholas et al. (2021) that revealed fire reduced the amount of carbon within a sagebrush ecosystem. However, soil carbon levels have been shown to increase up to seven years post burn with the increase of graminoid root growth (February et al. 2013). Alternately, tree and brush removal had lower soil carbon, which may be due to loss of aboveground carbon inputs without the same stimulation of root growth as fire. These results do not align with my original predictions that the control sites would have the highest amounts of below ground nitrogen and carbon. The highest percentage of below ground nitrogen and carbon were found within the prescribed burn site.

CONCLUSION

Rangelands in western Colorado play a crucial role in providing ecosystem services to the area, benefiting both human and wildlife populations. These services include wildlife habitat, carbon sequestration, cultural importance, plant biodiversity, and water filtration. However, the management practices employed in these rangelands can have a significant impact on the ecosystem services they provide. My study demonstrated that a substantial amount of money is being used to support various land management practices, and that these practices can influence a variety of ecosystem services in sagebrush ecosystems, often in contrasting ways. By examining the vegetation and soil responses to different treatments, we can gain insights into the potential trade-offs and synergies between management actions and ecosystem services. This knowledge is crucial for making informed decisions about sustainable land management practices that balance the needs of grazing, conservation, and other societal interests. It is also important to consider the specific goals and desired outcomes when selecting and implementing different management techniques in a sagebrush ecosystem.

My study supports my initial hypotheses that land management practices have an important effect on plant species diversity but not in the direction predicted based on diversity-disturbance theory. Specifically, sites that underwent prescribed burning showed lower plant diversity compared to reference sites that did not undergo any management. However, my results suggest that seeding may alleviate these diversity losses and as such, should be a priority to include with other management practices to maximize positive effects of land management practices while minimizing negative effects.

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Management events	*Vegetation Treatment Type*	Description
2008 and 2018	*Mechanical Treatment-Brush and Tree Removal*	Brush and tree removal includes techniques such as hand tools and hand-operated power tools to cut, clear, or prune herbaceous and woody species. The hand tools used included handsaw, axe, shovel, rake, machete, Pulaski, and hand clippers. The power tools used included chain saws and power brush saws.
2009 and 2010	*Prescribed Burn*	Boundary lines were assessed, and prescribed burning occurred. (early spring, winter, or fall)
2009 and 2013	*Seeding*	Seeding was done by a seed drill. The seed drill opens a furrow in the seedbed, a measured number of seeds are put into the furrow, and then the furrow is closed to cover the seeds. Seed mix included: bluebunch wheatgrass (*Pseudoroegneria spicata*), thickspike wheatgrass (*Elymus lanceolatus*), slender wheatgrass (*Elymus trachycaulus*), western wheatgrass (*Pascopyrum smithii*), Indian ricegrass (*Achnatherum hymenoides*), blue grama (*Bouteloua gracilis*), small burnet (*Sanguisorba minor*), blue flax (*Linum perenne*), milkvetch (*Astragalus* spp.), shadscale saltbush (*Atriplex confertifolia*)

TABLES

Table 2.1. Summary of land management practices applied to proposed research areas by the Bureau of Land Management. The information here is summarized from Integrated Vegetation Management Handbook (2008).

FIGURES

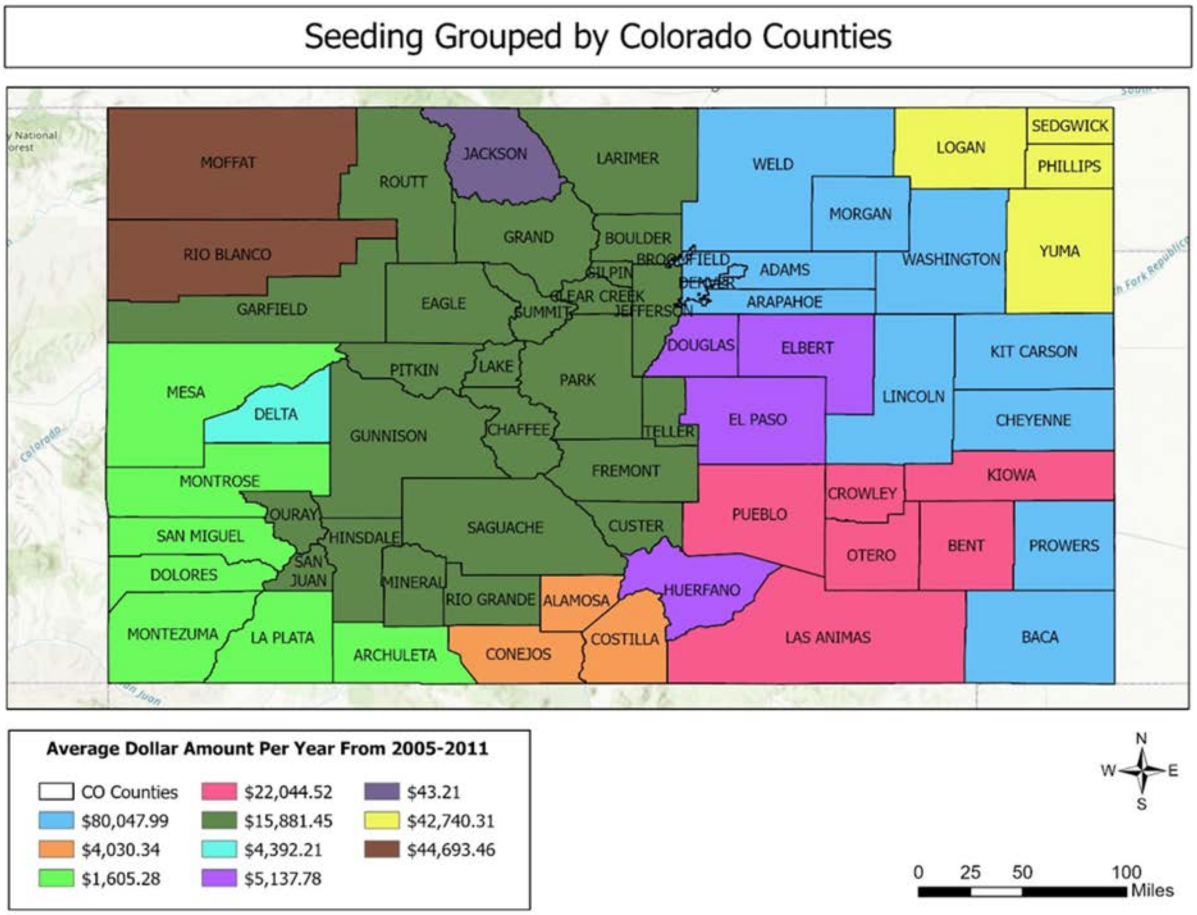

Figure 2.1. This map reports the average dollar amount spent per year from 2005-2011 on seeding treatments throughout Colorado counties. Each color represents the different dollar amount for the group of counties shown.

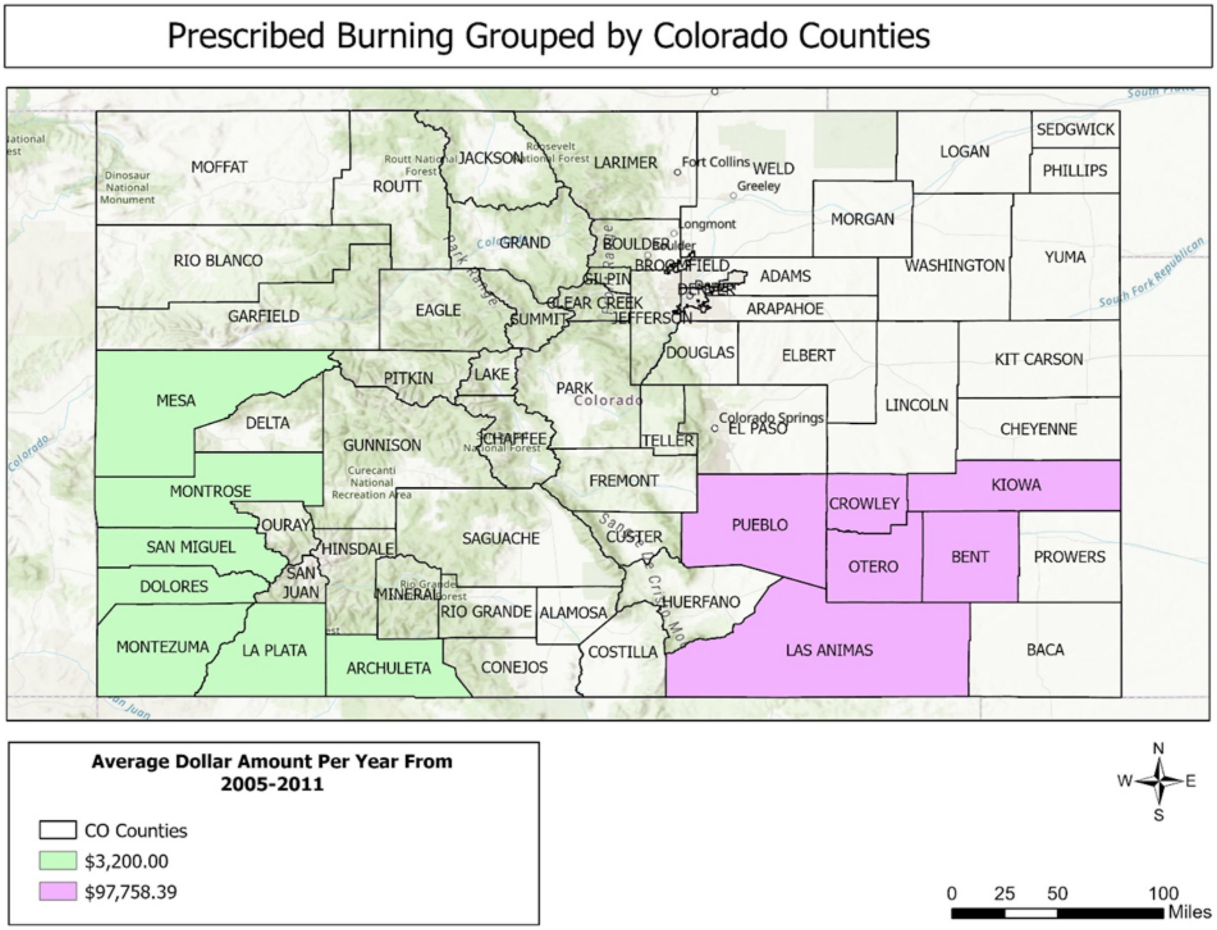

Figure 2.2. This map reports the average dollar amount spent per year from 2005-2011 on prescribed burning treatments throughout Colorado counties. Each color represents the different dollar amount for the group of counties shown.

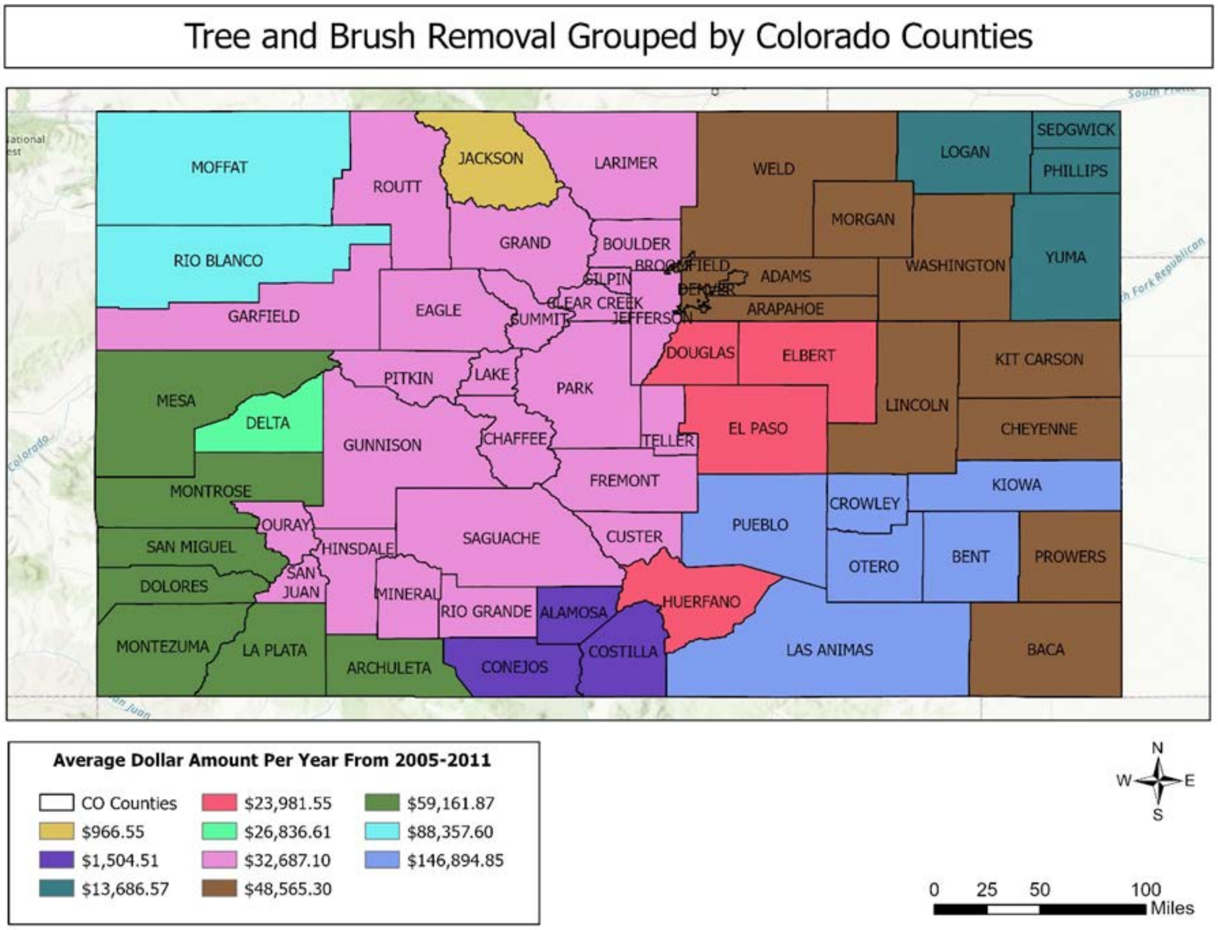

Figure 2.3. This map reports the average dollar amount spent per year from 2005-2011 on tree and brush removal treatments throughout Colorado counties. Each color represents the different dollar amount for the group of counties shown.

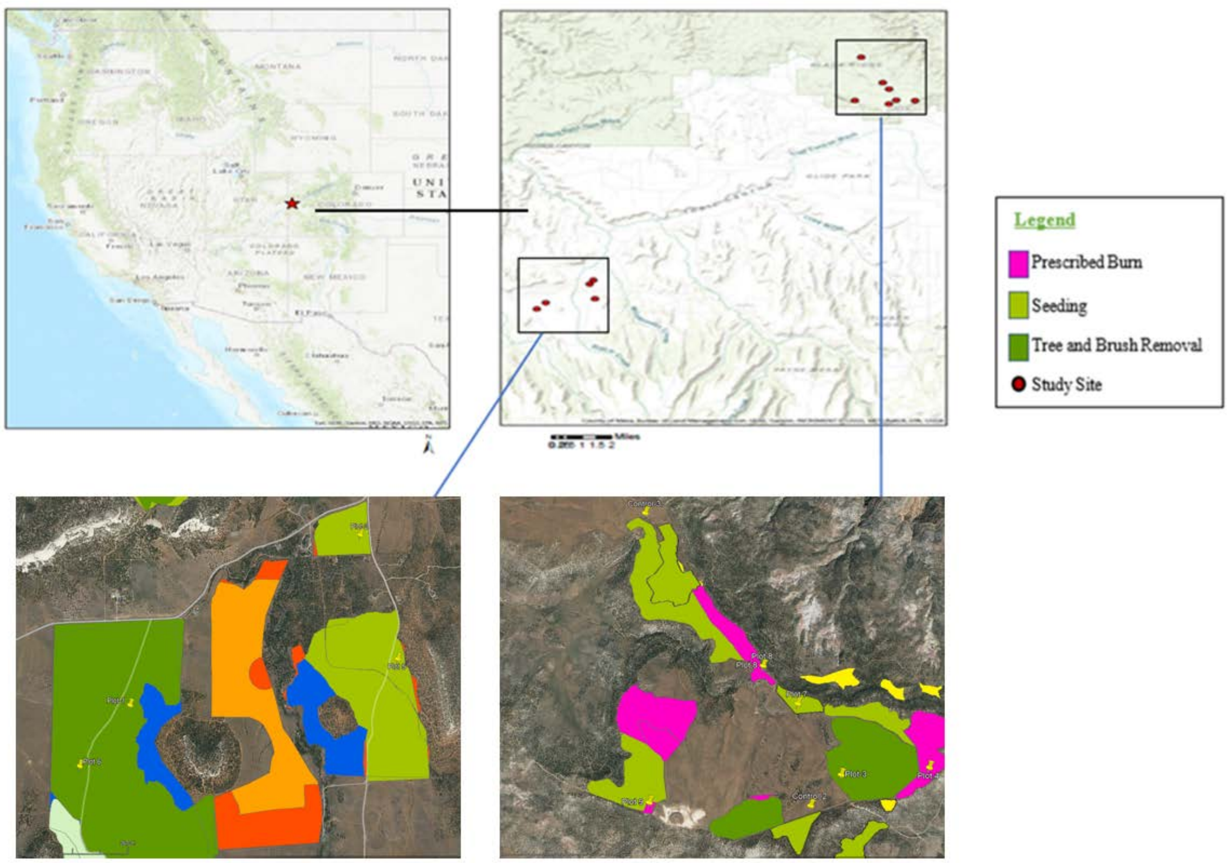

Figure 2.4. Location of land management practice locations across Glade Park, Colorado. Research sites were located on public land managed by the Bureau of Land Management. The blue, orange, and light orange colors represent different land management practices not included within my study.

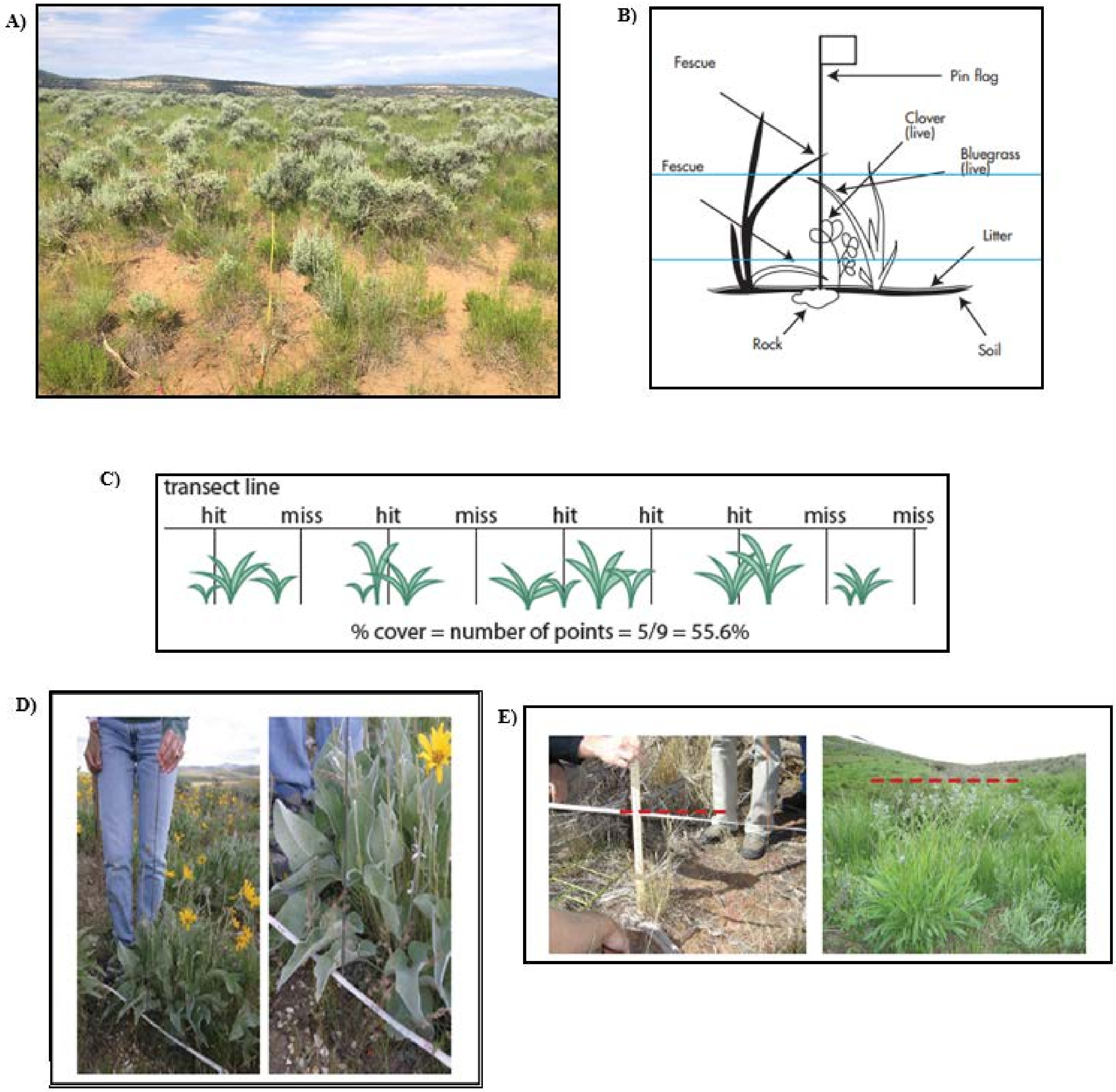

Figure 2.5. A) Represents a transect line at a sampling site. B) An example of a pin flag drop for LPI (Herrick et al. 2009). The “Top Layer is touching Fescue. “Lower Layer” is touching bluegrass and clover. The “Soil Surface” is touching rock. C) Example of a pin drop along a transect line (Stiver et al. 2015). D) An example of Line Point Intercept vegetation within the transect (Stiver et al. 2015). E) An example of measuring vegetation within the transect (Stiver et al. 2015)

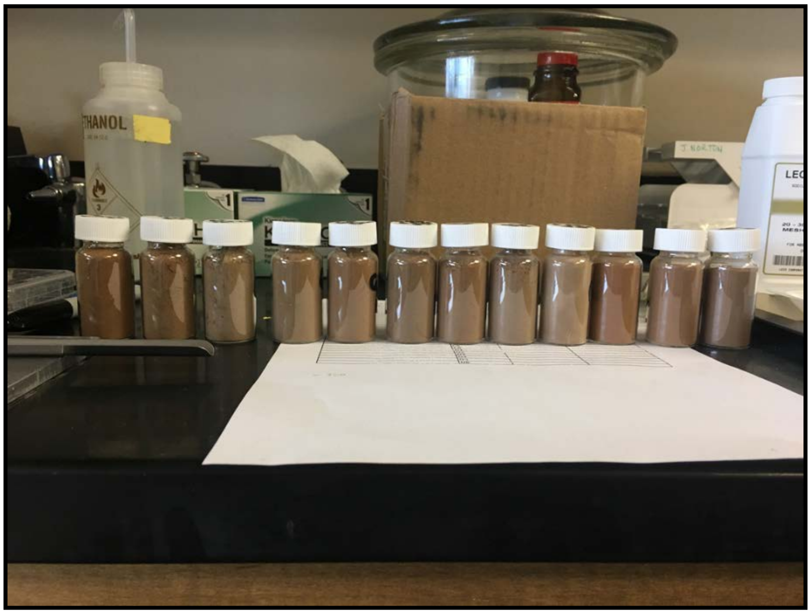

Figure 2.6. Photo of ground soil samples.

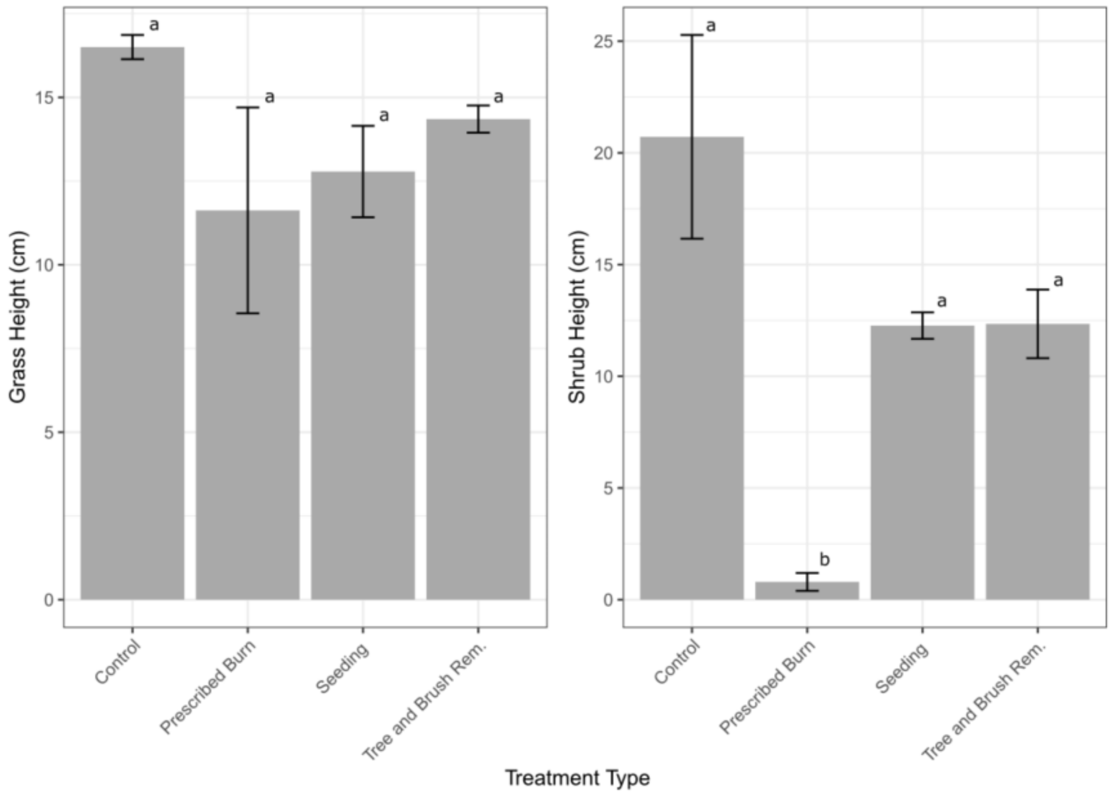

Figure 2.7. Mean grass and shrub height (cm). The error bars represent one standard error from the mean. The subscripts represent the significant difference between treatment types.

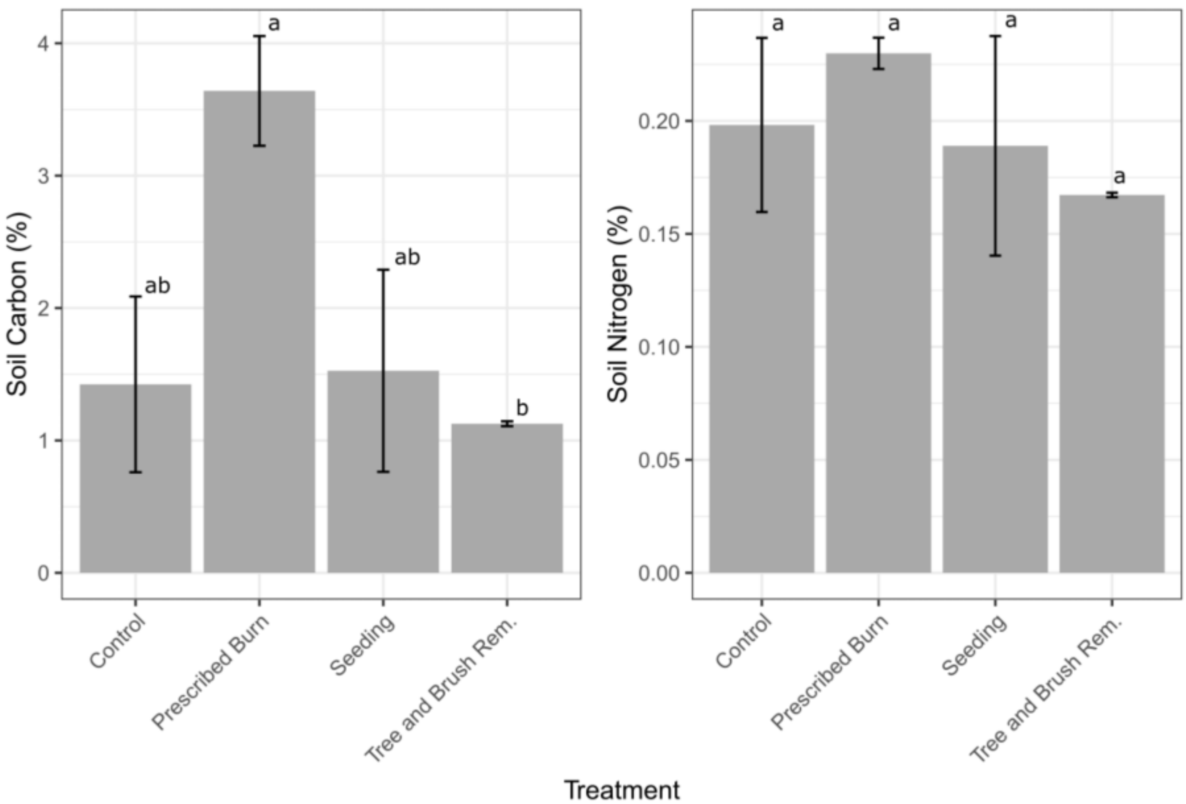

Figure 2.8. Mean (%) soil carbon and nitrogen for all treatments. The error bars represent one standard error from the mean. The subscripts represent the significant difference between treatment types.

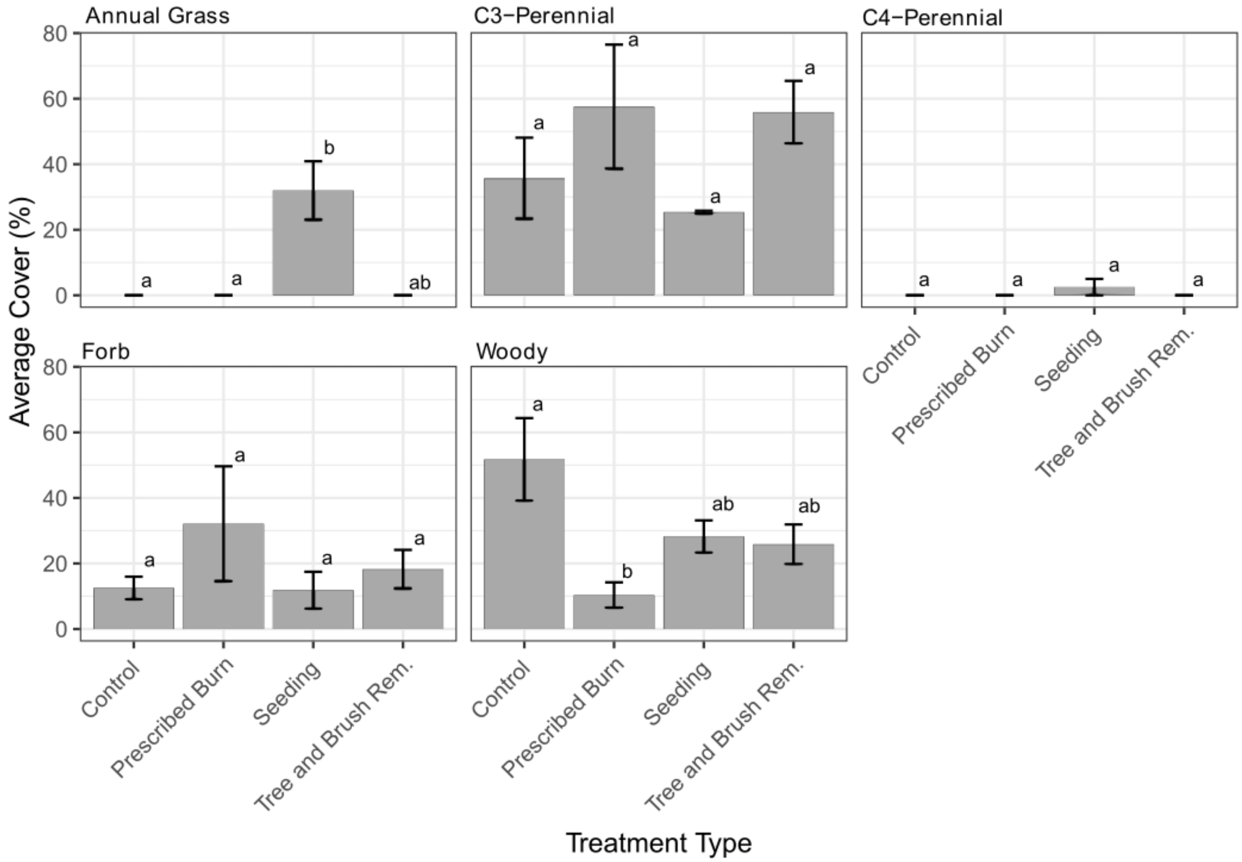

Figure 2.9. Mean cover (%) for all functional groups. The error bars represent standard error. The subscripts represent the significant difference between treatment types.

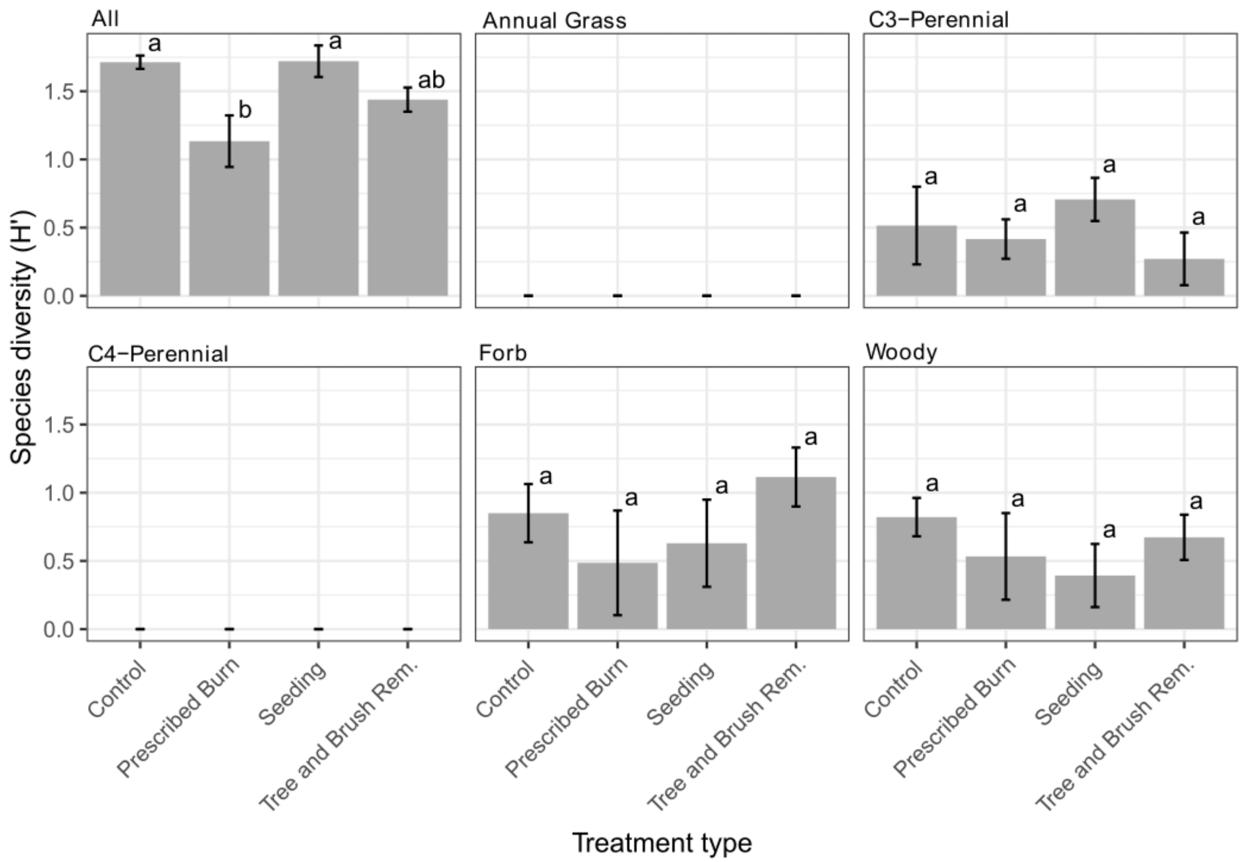

Figure 2.10. Mean species diversity as measured with the Shannon's diversity index (H') for all functional groups. The error bars represent standard error. The subscripts represent the significant difference between treatment types.

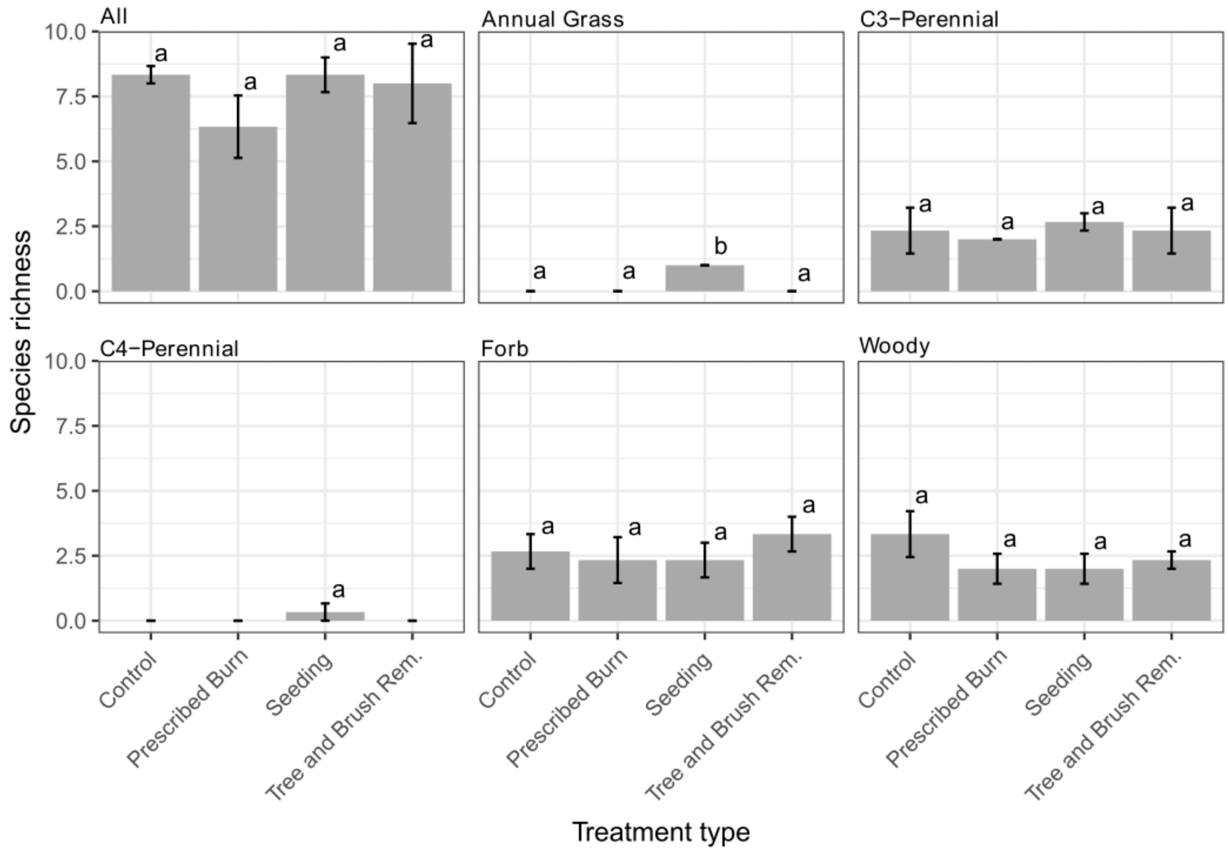

Figure 2.11. Mean species richness for all functional groups. The error bars represent standard error. The subscripts represent the significant difference between treatment types.

CHAPTER THREE: EFFECTS OF LAND MANAGEMENT PRACTICES ON ECOSYSTEM SERVICES IN THE INTERMOUNTAIN WEST

In the form for manuscripts submitted to *Rangeland Ecology and Management*

ABSTRACT

The Intermountain West spans from Canada to the southwestern United States and has significant ecological importance. This landscape is characterized by a diverse rangeland plant community that includes sagebrush (*Artemisia* spp.) ecosystems, pinyon (*Pinus* spp.)-juniper (*Juniperus* spp.) woodlands, and grasslands. This study examines the relationship between common land management practices and ecosystem services by using a meta-analysis. I compiled peer-reviewed literature from 1900-2020 to better understand the effects of disturbance from prescribed burning, tree and brush removal, and seeding within a meta-analysis. This analysis evaluated the effects of treatment types on soil carbon and nitrogen, plant abundance and richness, and pollinator abundance and richness. Through this meta-analysis I found that disturbance alone increased annual exotic plant abundance, while shrub abundance was lower within areas that had been disturbed and disturbed and seeded. I found that perennial forb abundance decreased with disturbance and increased with seeding. Through this research, I discovered that perennial grass richness decreased with seedling, while mechanical treatments can increase pollinator abundance. Overall, this study contributes valuable insight into improving and sustaining land management strategies for the Intermountain West and provides insight into these common land management treatments for stakeholders and land managers. This study highlights the importance of continued research and informed decision making to conserve and enhance the ecological integrity of rangeland ecosystems in the Intermountain West.

Keywords: Intermountain West, land management practices, ecosystem services, sagebrush ecosystems, pollinators, meta-analysis, biodiversity, carbon sequestration.

INTRODUCTION

The Intermountain West and Western Great Plains hereafter grouped and referred to as 'Intermountain West' span from Canada, northern Washington, and Montana south through Colorado, Utah, southern Idaho, Wyoming, California, Nevada, Arizona, and New Mexico (Fig. 3.1). This region has one of the highest proportions of federal land within the United States (USDA-NRCS 2018). The Intermountain West consists of sagebrush (*Artemisia* spp.) ecosystems, forests, and a wide variety of other rangeland plant communities. These ecosystems provide a variety of services such as habitat for wildlife and pollinator species, vegetation biodiversity, and carbon sequestration (Meinke et al. 2009, Fernandez et al. 2013). In the past, the Intermountain West has experienced various land management practices and disturbances including heavy grazing and changes in fire patterns and frequency (Perryman et al. 2021). Land managers in the Intermountain West are continually trying to determine the best land management practices to improve and sustain forage quality, plant biodiversity, wildlife and pollinator habitat, carbon balance, and ecosystem service stability.

Sagebrush ecosystems are one of the largest ecosystems in North America (Finch et al. 2016). Big sagebrush (*A. tridentata*) is one of the dominant shrubs in the Intermountain West. Although big sagebrush does not require pollinators for reproduction, many other shrub and forb species found in sagebrush ecosystems attract and provide habitat for several native species of bumble bees (*Bombus* spp.; Cook et al. 2011) and other insects. A study conducted by Cook et el. (2011) captured 12 different species of bumble bees in two locations within a sagebrush ecosystem, highlighting the high diversity of bumble bees present in sagebrush ecosystems. Today, sagebrush ecosystems are highly fragmented across the Intermountain West, and these ecosystems have been found to only occupy 56% of their historical range (Schroeder et al. 2004). With pollinator populations across the globe decreasing (Potts et al. 2010), intact sagebrush ecosystems within the Intermountain West could provide critical habitat for pollinator survival. If pollinator populations continue to decline, there could be a loss of plant biodiversity (Biesmeijer et al. 2006) and large economic losses due to reduced plant pollination (Watanabe 2014). Wildlife and pollinator population declines often indicate a negative impact on sagebrush steppe ecosystems and have strong effects on multiple ecosystem services. Pollinator species within sagebrush ecosystems may experience positive and negative effects from different drivers. For example, prescribed burning and drill seeding may initially kill larva and adults of pollinators. Also, prescribed burning reduces conifer encroachment. Yet, it may initially reduce the amount of sagebrush, grasses, and forbs present in an area, and thus reduce pollinator abundance and diversity, carbon sequestration, and habitat for wildlife species. Once sites recover, increases in plant diversity may increase pollinator species (Sexton and Emery 2020) and ecosystem function (Tilman 1997). Seeding may enhance diversity of grasses or forbs to a site that could be beneficial for wildlife, livestock, carbon sequestration, and pollinator species. Combining prescribed burning and seeding may result in the most benefit for wildlife, livestock, carbon sequestration, and pollinator species abundance. Seeding after fire or mechanical tree removal could reduce the number of invasive species that establish within a site after the treatment. The success of many of these treatments can be dependent on ideal weather conditions.

Many land management practices within sagebrush communities in the Intermountain West have been conducted to reduce conifer and non-native species from establishing. These land management practices include tree and brush removal, seeding, and prescribed burning. These land management practices have been used to increase land health and wildlife habitat. Recent studies show that after encroaching conifers are removed, greater sage-grouse (*Centrocercus urophasianus*) occupancy, nest survival, and brood success are greatly improved (Olsen et al. 2021). For example, tree and brush removal through mechanical treatment has the potential to increase the amount of shrub, forb, and grass species present within an area (Boyd et al. 2017). This increase in plant diversity may have positive cascading effects on the survival and habitat for pollinators and wildlife.

To manage public and private lands for ecosystem service provisioning, the effects of various land management practices must be understood. Additionally, it is critical to understand how co-occurring land management practices affect multiple ecosystem services simultaneously. To this end, I conducted a meta-analysis of existing peer-reviewed studies to assess effects of land management practices on ecosystem services in the Intermountain West. Specifically, I assessed the effects of tree and brush removal, seeding, and prescribed burning on soil carbon, plant biodiversity, wildlife habitat, and pollinator abundance and diversity. This provided an opportunity to accomplish the following overarching objective to determine how common land management practices alter ecosystem services within the Intermountain West.

QUESTIONS AND PREDICTIONS

To achieve this objective, I addressed the following questions:

Q1) How do disturbance and seeding treatments alone affect plant abundance, plant diversity, and soil carbon and nitrogen?

Q2) How does the combination of disturbance and seeding alter these effects?

Q3) How do mechanical, cut and leave (lop and scatter), and prescribed burning treatments impact pollinator abundance and richness?

To help answer these questions, I used meta-analysis techniques to test the following predictions:

Q1 Predictions

1. I predicted that disturbance and seeding will have the greatest effect on perennial grass abundance and richness.

B. I predicted that disturbance alone will result in the greatest abundance of annual-exotic plant species found within a site.

C. I predicted that soil carbon and nitrogen will be highest in areas that have been disturbed.

Q2 Predictions

A. Disturbance and seeding will result in greater provision of ecosystem services than singly occurring practices. For example:
    a. Disturbance and seeding will increase the amount of perennial plant species richness and reduce exotic annual plant species richness.

Q3 Predictions

A. Pollinator abundance and richness will decrease with prescribed burning.

METHODOLOGY

Site Description

The Intermountain West includes northern Washington and Montana south through Colorado, Utah, southern Idaho, Wyoming, California, Arizona, New Mexico, and Nevada (Fig.3.1). The Intermountain West consists of sagebrush ecosystems, forest, and a wide variety of other rangeland plant communities. According to the (USDA-NRCS 2018), the Intermountain West was historically dominated by bunchgrasses and shrubs. The dominant woody species in rangeland communities within this region include sagebrush, greasewood (*Sarcobatus spp*.), salt desert scrub (*Atriplex spp*.), mountain mahogany (*Cercocarpus spp*.), and woodlands of pinyon pine (*Pinus spp.*) and juniper (*Juniperus spp*.). Native perennial grasses found within these woody communities include bluebunch wheatgrass (*Pseudoroegneria spicata*), blue grama (*Bouteloua gracilis*), dropseed (*Sporobolus spp*.), Idaho fescue (*Festuca idahoensis*, needlegrass (*Stipa spp.*), prairie junegrass (*Koeleria macrantha*), sandberg’s bluegrass (*Poa secunda*), and western wheatgrass (*Pascopyrum smithii*; USDA-NRCS 2018). The Intermountain West has a large variation in elevation, mean annual temperature, and mean annual precipitation.

Throughout this chapter, I focused on locations that include sagebrush steppe, conifer (pinyon and juniper), shortgrass prairie, and northern mixed grass prairie. I evaluated public and private land within the Intermountain West known to have at least one of the following land treatments: tree and brush removal, prescribed burns, and seeding.

Data Compilation

To determine how current land management practices (tree and brush removal, prescribed burning, and seeding) are altering ecosystem services, I performed a meta-analysis using all relevant literature including peer reviewed literature, books, and book chapters from 1900-2020. Preliminary literature searches show that studies describing effects on pollinator abundance and diversity often use different terminology than studies focused on plant biodiversity, wildlife habitat, and carbon sequestration. For this reason, I conducted two literature searches using Web of Science during December 2020. The first focused on publications that report on plant biodiversity, sage-grouse habitat, and carbon sequestration associated with land management practice techniques in the Intermountain West. The second search focused on pollinator species associated with land management practice techniques in the Intermountain West.

Criteria that I used for the first search included the following search terms: (“Carbon Sequestration” OR “Biodiversity” OR “Sage-Grouse”) AND (“Prescribed Burn” OR “Cut” OR “Fire” OR “Seed”) AND (“Sagebrush” OR “Juniper” OR “Pinyon” OR “Woody-Plant” OR “Rangeland” OR “Shortgrass” OR “Northern mixed”), where AND OR statements represent Boolean operators and terms are grouped by parentheses. These search criteria identified 471 peer-reviewed journal articles, books, book chapters, and government publications.

The second search criteria included these search terms: (“Bee" OR Pollinat* OR ”Arthropod” OR “Insect”) AND (“Prescribed burn” OR “fire” OR “Cut” OR “Seed”) AND (“Sagebrush” OR “Juniper” OR “Pinyon” OR “Woody-Plant” OR “Rangeland” OR “Shortgrass” OR “Northern mixed”). This search criteria came up with 211 peer-reviewed journal articles, books, book chapters, and government publications.

I included studies from the list generated from the searches above based on the following criteria:

1. Cropland studies were excluded.
2. Study was performed in intact ecosystems within, sagebrush steppe, Pinyon-Juniper, shortgrass prairie, or northern mixed grass prairie ecosystems in the U.S. and Canada.
3. Land management practices – prescribed burns, seeding, or tree and brush removal – were applied to treatment plots, and there were comparable control plots where land management practices were not applied.
4. Response variables were reported from the following lists: soil carbon, plant biodiversity, or wildlife habitat for studies from the first search; total cover, and pollinator abundance or diversity for studies from the second search.
5. I limited the study to only focus on sampling plots that have experienced one or more of the treatment types (prescribed burns, seeding, or tree and brush removal).
6. Studies must report the average or median response variable across plots for both treatment and control. Additionally, standard error or the standard deviation and sample size must be reported.

I also included publications that were cited within the studies found but were not present in the original Web of Science search that were also conducted within the Intermountain West. The additional literature that fit into these criteria were also added into my meta-analysis.

From each publication fitting the criteria above, I extracted the following information. To calculate response ratios, I extracted means and standard deviation of the response variables in 'treatment' (areas where land management practices were applied) and 'control' (areas where land management practices were not applied), as well as the number of replicates in each and the land management practice applied. I combined cover and density measures into "abundance" to create a larger sample size for comparison. After studies were eliminated based upon my criteria, there were 21 relevant publications for the first search criteria and 6 for the second search. Values were gathered from each study as text, from tables, or from figures using WebPlotDigitizer (Rohatgi, A., 2018).

Calculating meta-analysis metrics and statistical methods

I conducted a meta-analysis using all studies that met the criteria above. To calculate effect sizes for each dependent variable within each study, I used the mean of the dependent variable within the experiment and the control (Xa, Xc), standard error of the experiment and control (Se, Sa), and the sample size (Ne, Na). Effect sizes were aggregated from multiple studies to produce a weighted mean effect size (Hedges et al. 1999) using random effects models. I used equations from Borenstein (2009), Curtis and Wang (1998), and Groenigen (2014), to perform the meta-analysis.

To answer both my questions, I calculated weighted response ratios and 95% confidence intervals to see if responses to various land management practices overlapped zero. The treatment types considered were prescribed burning, tree and brush removal, and seeding. The response variables included pollinator richness and abundance, total herbaceous abundance, shrub abundance, bare ground abundance, perennial forb abundance, annual exotic (forb and grass) abundance, perennial grass abundance, perennial forb richness, perennial grass richness, annual exotic richness, total herbaceous richness, soil carbon and soil nitrogen. I calculated the natural logarithm of the response ratio for each response variable in each study as:

$$ln(r) = ln\,(E/C) \qquad (1)$$

where *E* represents the mean of the experimental group and *C* represents the mean of the control group. I used the sample variance and 95% confidence intervals to assess whether aggregate effects of land management practices on ecosystem services differed from 0. The variance (V) was calculated as:

$$V = \frac{Se^2}{Ne \times Xe^2} + \frac{Sa^2}{Na \times Xa^2} \qquad (2)$$

where *Xe*, *Xa* represent the response means of the experiment and control, *Se*, *Sa* represent standard deviations of the experiment and control, and sample sizes were represented by (*Ne*, *Na*) for the experiment and control. 95% confidence intervals were calculated as:

$$95\,CI = ln(r) - 1.96\sqrt{V} \;\; to \;\; ln(r) + 1.96\sqrt{V} \qquad (3)$$

I used the weighted mean log ratio to combine results from multiple studies and give a greater weight to the experiments that have a smaller standard error and in turn increase the accuracy of the combined estimate (Wang 1998): (4)

$$M = \frac{\sum_{i=1}^{k} W_i ln(r)_i}{\sum_{i=1}^{k} W_i}$$

where *W* is the weight assigned to each study, calculated as: (5)

$$W_i = \frac{1}{V ln(r)_i}$$

Then, I calculated the variance of the weighted mean as: (6)

$$V_M = \frac{1}{\sum_{i=1}^{k} W_i}$$

where k is the number of studies in each group. The standard error of the weighted mean was then calculated as: (7)

$$SE_M = \sqrt{V_M}$$

Finally, the 95% lower and upper limits for the weighted mean were calculated as: (8)

$$LL_M = M - 1.96 \times SE_M$$

To answer my first question, I assessed the effects of management and seeding practices when applied singly on each response variable. To answer my second question, I compared effect sizes from co-occurring treatments with treatments applied singly. I used R (R Core Team, 2018) and the *metafor* package (Viechtbauer 2010) to conduct all analyses.

Results

Total herbaceous abundance was not significantly impacted by disturbance, seeding, or disturbance and seeding treatments (Fig 3.2a). Mean shrub abundance was -0.68 (SE = 0.25) and was negatively impacted by disturbance (z = -2.71, $P < 0.01$; CI [-1.16, -0.19]; RR = -0.77; Table 3.1; Fig. 3.2b) and disturbance and seeding (mean = -0.75, SE= 0.29, z = -2.51, P = 0.012; CI [-1.33, -0.16]; RR = -0.86; Table 3.1; Fig. 3.2b), but not affected by seeding alone. Perennial forb abundance was significantly impacted by disturbance (mean = -0.22, SE = 0.11, z = -1.94, P = 0.051; CI [-0.45, 0.00]; RR = -0.27; Table 3.1; Fig. 3.2b), but not affected by disturbance and seeding or seeding. Bare ground, and perennial grass abundances were all not altered under disturbance, seeding, or disturbance and seeding treatments (Table 3.1; Figs. 3.2 c, d, f). Annual exotic abundance increased overall in disturbance treatments (mean = 0.48, SE = 0.16, z = 2.9, P < 0.01; CI [-0.45, 0.00]; RR = 0.53; Table 3.1; Fig. 3.2), but not seeding or disturbance and seeding treatments (Table 3.1).

Perennial forb richness did not have a significant response to any of the treatments. Annual exotic richness was not significantly impacted by any of the treatments. Perennial grass richness was lower in the seeding treatments (mean = -0.37, SE = 0.17, z = -2.08, P = 0.037; CI [-0.72, -0.02]; RR = 0.21; Table 3.2; Fig. 3.3). However, perennial grass richness was not impacted by disturbance and disturbance and seeding. Similarly, I found no significant relationship between the annual exotic richness and the response variables measured. In addition, I found no significant response ratio when looking at the effects on herbaceous richness with disturbance, disturbance and seeding, and seeding.

Soil carbon and soil nitrogen (Fig. 3.4) were only found to be present in the studies that had experienced a disturbance (burning and tree and brush removal). The soil carbon response was not significant. Similarly, the response ratio for the soil nitrogen was not significant. Pollinator abundance (Fig. 3.5) increased in the mechanical treatments, but pollinator abundance was not increased in the cut and leave or burn treatments. There was no evidence of treatment effects on pollinator richness.

DISCUSSION

My study aimed to determine how various land management practices influence ecosystem services in the Intermountain West. My results shed light on the impact of disturbance, disturbance and seeding, and seeding treatments on plant abundance, plant richness, and soil carbon and nitrogen levels. Additionally, my study explored how different types of disturbance affect pollinator abundance and diversity within the Intermountain West.

I began by asking how disturbance and seeding treatments alone affect plant abundance, plant diversity, and soil C and N. I predicted that disturbance and seeding would lead to the greatest perennial grass abundance and richness.The findings revealed that total herbaceous abundance, and perennial grass abundance did not differ by disturbance, seeding, or disturbance and seeding. Shrub abundance was negatively impacted by both disturbance alone and disturbance combined with seeding, but not by seeding. However, I predicted that annual exotic abundance would increase within the disturbance and this was supported by results. Annual exotic abundance increased under disturbance treatments, but not under seeding or disturbance and seeding treatments. Perennial forb abundance was negatively impacted by distrubance. I also predicted that soil carbon and nitorgen would increase in the distrubed sites and these results were found to not be significant.

To answer my second question, how does the combination of disturbance and seeding alter these effects? I predicted that disturbance and seeding would increase perennial grass species richness and reduce exotic annual plant species richness. Perennial grass richness was found to decrease with seeding, but the other treatment types were insignificant. The data indicated that disturbance and seeding had no significant impact on soil carbon and nitrogen levels.

To answer my third question, how do mechanical, cut and leave, and prescribed burning treatments affect pollinator abundance and richness? I predicted that pollinator abundance and richness would decrease with prescribed burning. I found that mechanical disturbance treatments increased pollinator abundance, while cut and leave or prescribed burn treatments did not have any noticeable effects on pollinator abundance. Pollinator richness did not show any significant results when looking at the diferent treatment types.

Vegetation Abundance

Shrub abundance was negatively impacted by all types of disturbance that I evaluated (prescribed burn, cut and leave, and mechanical vegetation treatments; Fig. 3.2). As expected, Davies et al. (2020) found that sagebrush cover was greatest within control sites compared to treated areas that had mechanical vegetation treatments. Annual exotic abundance was slightly higher within disturbed treatment areas, these results are in line with a study conducted by Williams et al. (2017) who found that cheatgrass (*Bromus tectorum*) increased on prescribed burned and cut and leave treatments. However, Kerns et al (2020) found a reduction in annual abundance when seeding occurred after disturbance. This result indicates that a disturbance followed by seeding may be important if you are trying to reduce or prevent annual exotics. Williams et al. (2017) also found that bare ground was slightly less within their control versus prescribed burning and cut and leave treatments. This correlates with the results from the meta-analysis. A study done by Davies et al. (2012) did not find any strong correlation between forb cover and perennial bunchgrass cover. These results were similar to my findings that perennial forb and perennial grass abundance does not produce a strong relationship when comparing vegetation treatments.

One study I included in my meta-analysis, Davies et al. (2012) found that disturbance (mowing) increased the amount of perennial grass abundance while another study that I included in my analysis conducted by Kerns et al. (2020) found a negative response when comparing disturbance (cut/mechanical and burn) and perennial vegetation abundance. These differences among studies could be the reason behind the large error bars. The bare ground abundance results also have large error bars. This appears to be because Davies et al. (2020) found a strong positive correlation with disturbance (mowing) and less abundance of bare ground, while another study included within my meta-analysis by Davies et al. (2014) found a negative correlation with disturbance (mowing) and bare ground. These opposite results likely led to large error bars within bare ground abundance.

Vegetation Richness

Kerns et al. (2020) reported that seeding had a minimal impact on total species richness, which aligns with the findings from my current study (Fig. 3.3). The limited effect of seeding on species richness observed in both studies could potentially be attributed to challenges in seedling establishment and the lack of diversity within the seed mix used. This aspect is particularly relevant to my research, as I also did not find any significant results when investigating perennial forb richness, perennial grass richness, annual exotic richness, and total herbaceous richness. Interestingly, the results of another study conducted by McCain et al. (2010) contradicted these findings. They observed that removing dominant species from a site led to increased species richness. These contrasting results suggest that the relationship between disturbance, species richness, and vegetation dynamics in sagebrush steppe ecosystems is complex and may be influenced by numerous factors over time.

Overall, these findings collectively indicate that increasing vegetation richness within a sagebrush steppe ecosystem poses a significant challenge. Seeding efforts may not lead to substantial gains in species richness, and the response to disturbances can vary depending on specific conditions and temporal dynamics. Understanding the underlying mechanisms that drive these outcomes is crucial for effective land management and conservation efforts in these ecologically important regions.

Soil Carbon and Nitrogen

My meta-analysis found no effects of disturbance treatments on soil carbon and nitrogen (Fig. 3.4). It is important to note that the sample sizes for these results were limited to three, and to get a more informative idea of the significance these treatments have, more studies need to be included that fall within the scope of this study. I found no publications that fit the criteria for this study that evaluated soil carbon and nitrogen levels within a seeded or a disturbed and seeded site.

Pollinator Abundance and Richness

Results from this meta-analysis suggest that pollinator abundance showed a positive correlation with mechanical treatments (Fig. 3.5). A study I included in my meta-analysis, conducted by Kleintjes et al. (2004) found that with the mechanical removal of 70% of cover two-needle pinon pine, and one-seed juniper (*Juniperus monosperma*), will result in a pollinator abundance increase over time. These results indicate a mechanical treatment that reduces the amount of pinyon and juniper may increase pollinator abundance. There was no difference in pollinator abundance in either the cut and leave or burn treatments. However, Rhode et al. (2010) found that fewer ants, butterflies, and moths were found within a burn site vs their control sites. Pollinator richness did not show any strong relationship to the three treatment types within my study. Day et al. (2019) found that burned treatments did not have any impacts on insect abundance, but that they did have significant impacts on ant species richness.

IMPLICATIONS

The Intermountain West plays a crucial role for ecosystem services, it provides habitat for wildlife and pollinator species, maintains vegetation biodiversity, and contributes to carbon sequestration. Over the years this region has experienced various land management strategies. This has created a challenge for land managers to determine the best practices for improving and sustaining these ecosystems. With pollinator species declining globally, and the loss and fragmentation of sagebrush ecosystems it is very important to continue restoring and preserving these ecosystems.

My study aimed to assess the effects of common land management practices, such as prescribed burning, tree and brush removal, and seeding, on soil carbon and nitrogen, plant biodiversity, and pollinator abundance and diversity in the Intermountain West. My research addressed key questions about the overall impact of these practices on ecosystem services in this region through a meta-analysis of existing peer-reviewed studies. Ultimately, this research aims to inform land managers and policymakers about the most effective strategies for improving and sustaining the health, biodiversity, and functionality of these ecosystems.

The methodology employed in my study involved a comprehensive review of literature spanning over a century and focused on studies that examine the effects of land management practices on the targeted ecosystem services. My study found that there was a positive association with pollinator abundance and mechanical treatments, which indicates that these treatments had a beneficial impact on pollinator species. However, the cut and leave and burn treatments did not significantly affect pollinator abundance. Furthermore, my study revealed that none of the treatments had an impact on pollinator richness. Total herbaceous abundance was also not affected by any treatment. Overall, the results suggest that the selected treatments did not have substantial effects on most of the measured variables. These results highlight the complex dynamics involved in ecosystem service response to common management practices. Further research is needed to gain a better understanding of the underlying mechanisms and potential long-term impacts of these treatments on ecological communities.

The findings of this study will contribute to the broader knowledge of sustainable land management practices and guide future efforts in preserving and enhancing the valuable ecosystem services provided by sagebrush rangelands within the Intermountain West. It is important to continue researching the impacts of these common land management practices so that we can continue to provide resources for all stakeholders involved.

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TABLES

Table 3.1. The mean, standard error (SE), z-value, and p-value for abundance of all treatment types and plant functional groups. These statistics are from a meta-analysis of vegetation treatment effects on rangeland vegetation communities in the Intermountain West.

Treatment Type	Functional Group	mean	SE	z-value	p-value
Disturbance	Annual Exotic Abundance	0.48	0.16	2.91	0.004
Disturbance	Bare Ground Abundance	-0.14	0.13	-1.08	0.280
Disturbance	Perennial Grass Abundance	-0.36	0.47	-0.78	0.438
Disturbance	Shrub Abundance	-0.68	0.25	-2.71	0.001
Disturbance	Perennial Forb Abundance	-0.22	0.11	-1.95	0.051
Disturbance	Total Herbaceous Abundance	0.41	0.22	1.87	0.061
Disturbance and Seeding	Annual Exotic Abundance	0.04	0.16	0.23	0.822
Disturbance and Seeding	Bare Ground Abundance	0.37	0.58	0.65	0.517
Disturbance and Seeding	Perennial Grass Abundance	-0.29	0.21	-1.38	0.167
Disturbance and Seeding	Shrub Abundance	-0.75	0.30	-2.51	0.012
Disturbance and Seeding	Perennial Forb Abundance	-0.59	0.41	-1.46	0.144
Disturbance and Seeding	Total Herbaceous Abundance	0.07	0.11	0.63	0.529
Seeding	Annual Exotic Abundance	0.06	0.26	0.25	0.805

Seeding	Bare Ground Abundance	0.02	0.39	0.04	0.965
Seeding	Perennial Grass Abundance	0.05	0.24	0.20	0.844
Seeding	Shrub Abundance	0.34	0.38	0.91	0.364
Seeding	Perennial Forb Abundance	0.37	0.19	1.93	0.053
Seeding	Total Herbaceous Abundance	-0.15	0.28	-0.53	0.596

Table 3.2. The mean, standard error (SE), z-value, and p-value for all treatment types and plant functional groups related to species richness. These statistics are from a meta-analysis of vegetation treatment effects on rangeland vegetation communities in the Intermountain West.

Treatment Type	Functional Group	mean	SE	z-value	p-value
Disturbance	Annual Exotic Richness	0.34	0.22	1.54	0.124
Disturbance	Perennial Grass Richness	0.13	0.19	-1.43	0.152
Disturbance	Perennial Forb Richness	-0.27	0.22	0.57	0.569
Disturbance	Total Herbaceous Richness	0.03	0.15	0.22	0.825
Disturbance and Seeding	Annual Exotic Richness	0.20	0.13	1.46	0.144
Disturbance and Seeding	Perennial Grass Richness	0.12	0.15	0.84	0.403
Disturbance and Seeding	Perennial Forb Richness	0.13	0.23	0.51	0.612
Disturbance and Seeding	Total Herbaceous Richness	0.03	0.11	0.24	0.809
Seeding	Annual Exotic Richness	0.23	0.24	0.96	0.336
Seeding	Perennial Grass Richness	0.16	0.18	-2.08	0.037
Seeding	Perennial Forb Richness	-0.37	0.23	0.67	0.503
Seeding	Total Herbaceous Richness	-0.01	0.10	-0.05	0.957

Table 3.3. The mean, standard error (SE), z-value, and p-value for all treatment types and functional groups related to carbon and nitrogen. These statistics are from a meta-analysis of vegetation treatment effects on rangeland vegetation communities in the Intermountain West.

Treatment Type	Functional Group	mean	SE	z-vale	p-value
Disturbance	Soil Nitrogen	0.40	0.05	0.24	0.810
Disturbance	Soil Carbon	0.01	0.30	1.34	0.180
Disturbance and Seeding	Soil Carbon	NA	NA	NA	NA
Disturbance and Seeding	Soil Nitrogen	NA	NA	NA	NA
Seeding	Soil Carbon	NA	NA	NA	NA
Seeding	Soil Nitrogen	NA	NA	NA	NA

FIGURES

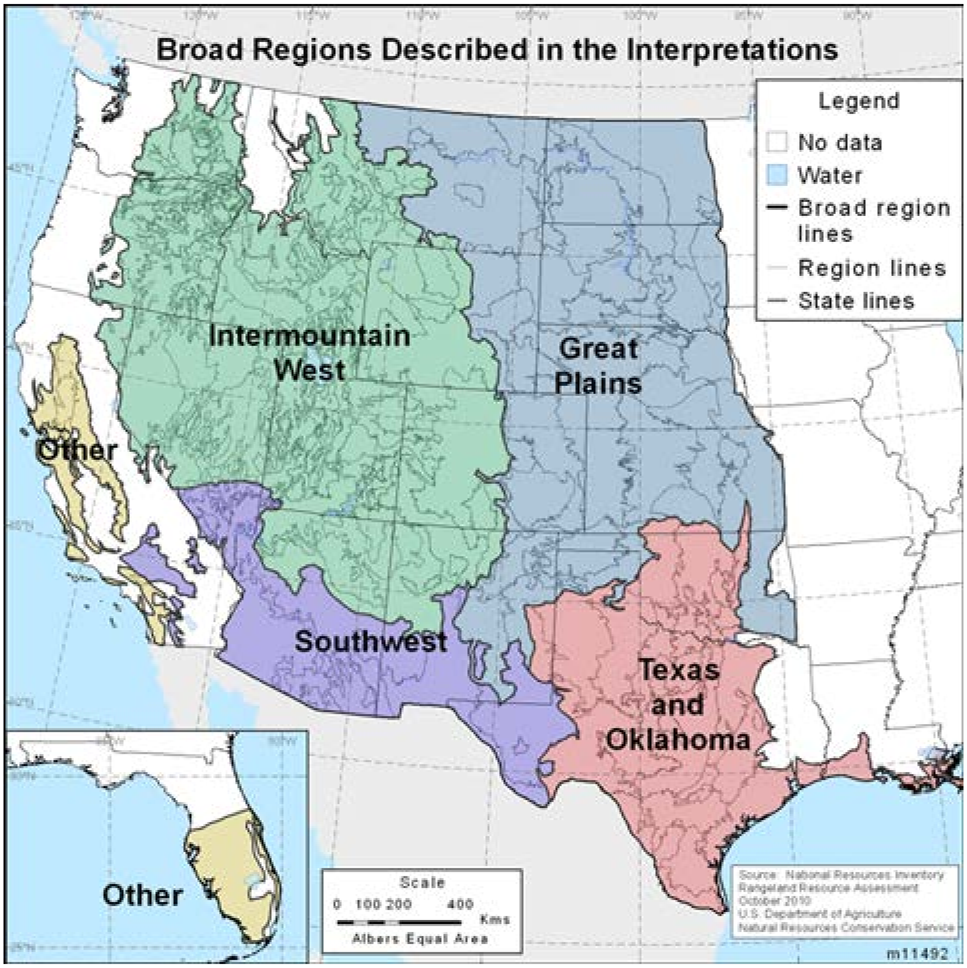

Figure 3.1. Broad regions described in the interpretations from USDA-NRCS (2018).

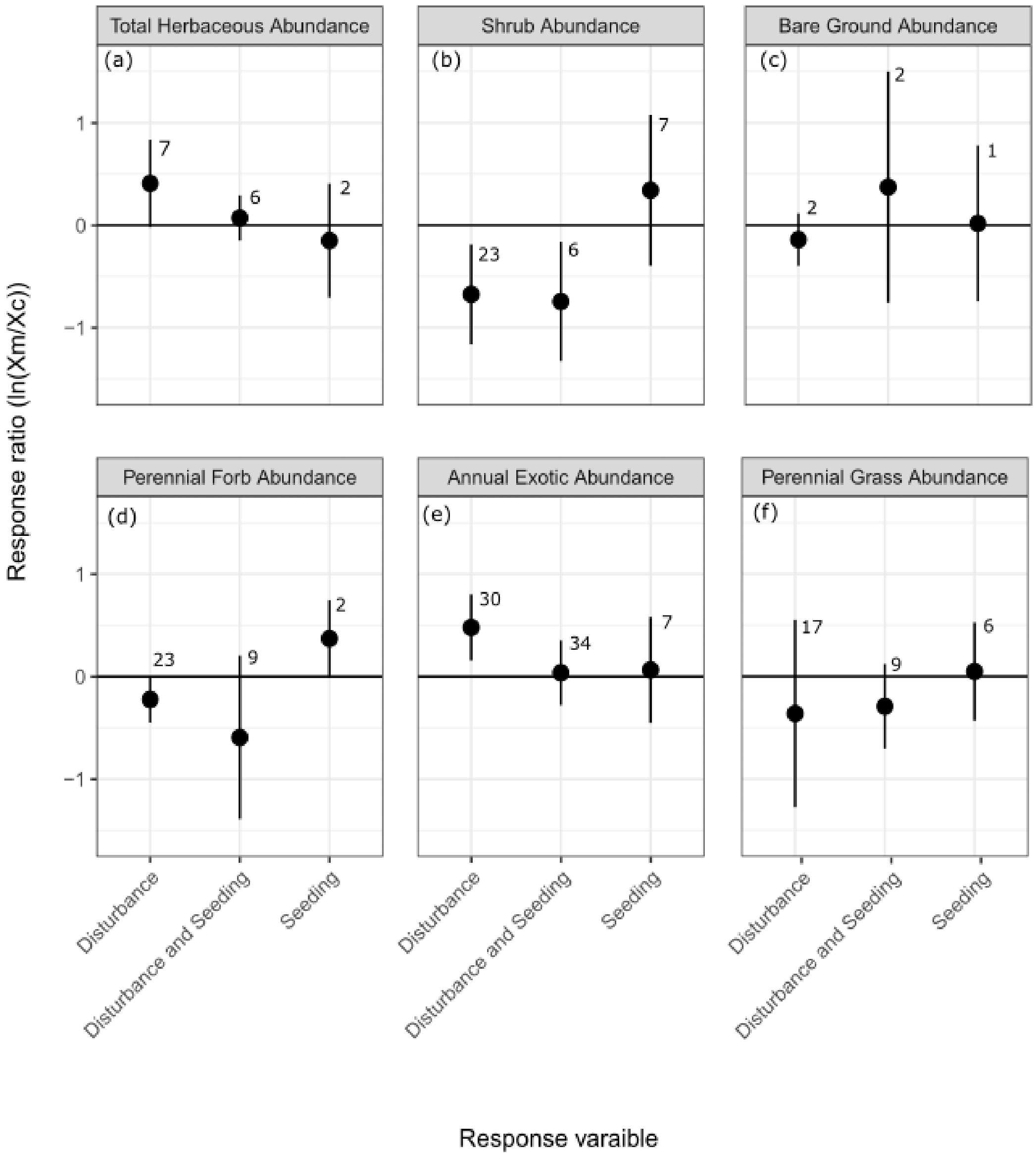

Figure 3.2. The response ratio shown for the vegetation abundance in relation to treatment type. Mean and 95% confidence intervals are shown for treatment types: disturbance, disturbance and seeding, and seeding. The number of case studies are presented next to the results. The line of no difference is indicated by zero. The response ratios that are below the 0 line are significantly lower and the ones above the 0 line are significantly higher.

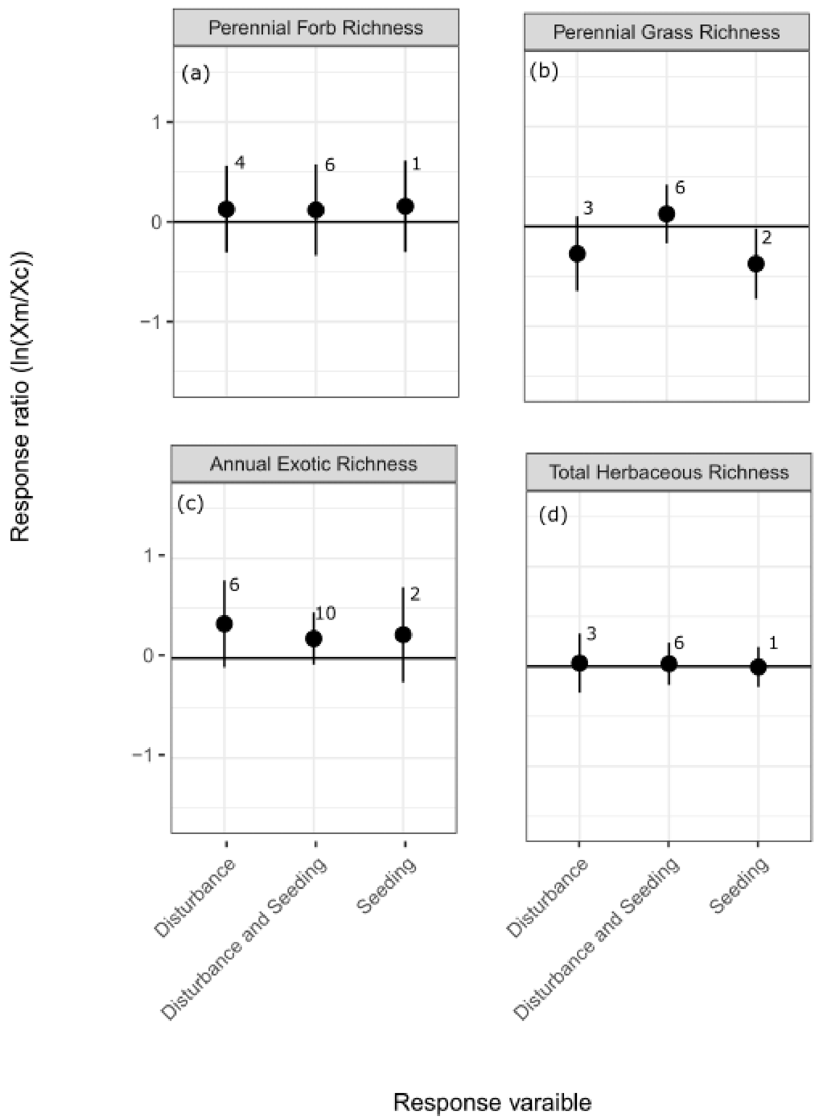

Figure 3.3. The response ratio shown for the vegetation richness in relation to treatment type. Mean and 95% confidence intervals are shown for treatment types: disturbance, disturbance and seeding, and seeding. The number of case studies are presented next to the results. The line of no difference is indicated by zero. The response ratios that are below the 0 line are significantly lower and the ones above the 0 line are significantly higher.

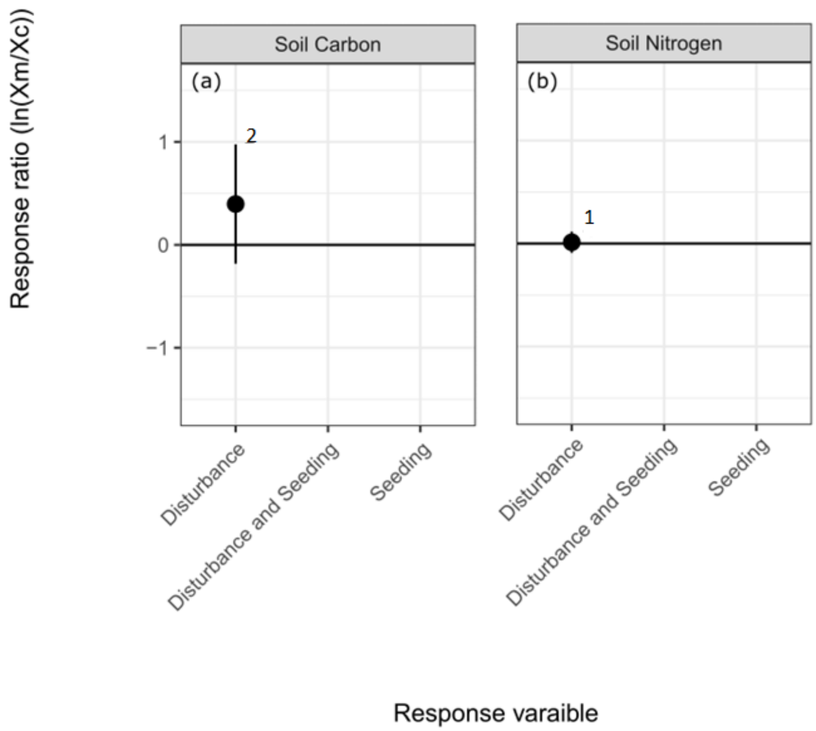

Figure 3.4. The response ratio shown for the soil carbon and soil nitrogen in relation to treatment type. Mean and 95% confidence intervals are shown for treatment types: disturbance, disturbance and seeding, and seeding. The number of case studies are presented next to the results. The line of no difference is indicated by zero. The response ratios that are below the 0 line are significantly lower and the ones above the 0 line are significantly higher.

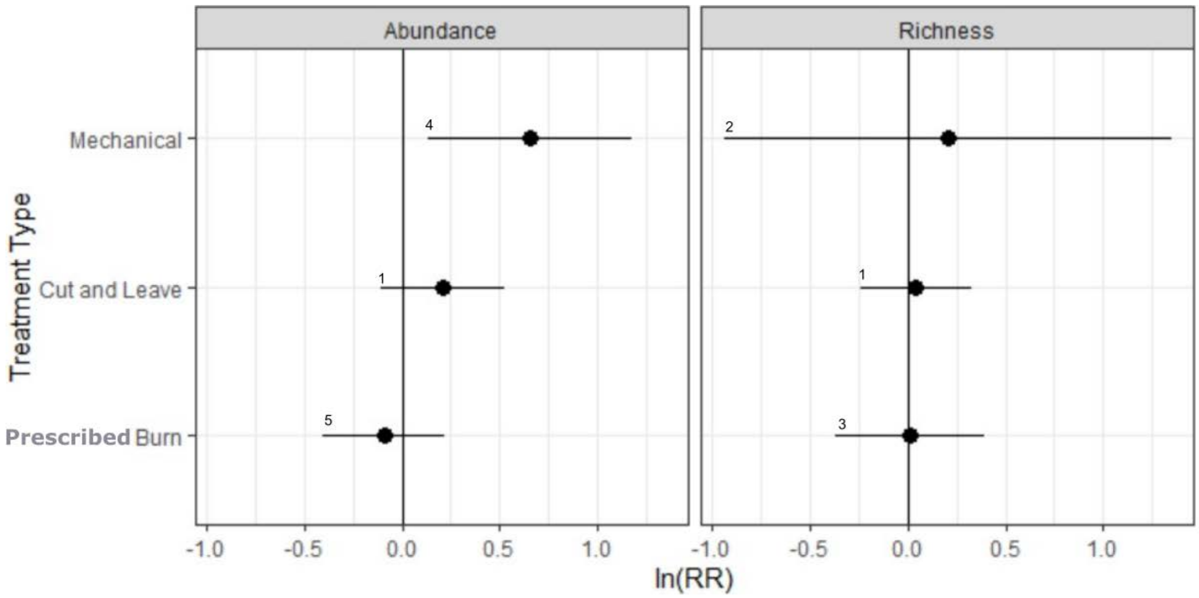

Figure 3.5. The log response ratio shown for pollinator species in relation to treatment type. Mean and 95% confidence intervals are shown for treatment types: mechanical, cut and leave, and burn. The number of case studies are presented next to the results. The line of no difference is indicated by zero. The response ratios that are to the left of the 0 line are significantly lower and the ones to the right of the 0 line are significantly higher.