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Harvest Efficiency of Forage Grazed by Cattle and The Effect of Pinyon and Juniper Treatments on Vegetation Cover on the Grand Staircase Escalante National Monument

Ruger P. Carter
Utah State University

HARVEST EFFICIENCY OF FORAGE GRAZED BY CATTLE AND THE EFFECT

OF PINYON AND JUNIPER TREATMENTS ON VEGETATION COVER ON

THE GRAND STAIRCASE ESCALANTE NATIONAL MONUMENT

by

Ruger P. Carter

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Eric Thacker, Ph.D. Major Professor	Kevin Heaton, MS Committee Member
Thomas A. Monaco, Ph.D. Committee Member	Beth Burritt, MS Committee Member

Janis L. Boettinger, Ph.D.
Acting Vice Provost of Graduate Studies

Utah State University
Logan, Utah

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ABSTRACT

Harvest and Grazing Efficiency of Forage Grazed by Cattle and The Effect of Pinyon and Juniper Treatments on Vegetation Cover on the Grand Staircase Escalante National Monument

by

Ruger P. Carter, Master of Science

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Major Professor: Dr. Eric Thacker
Department: Wildland Resources

The Grand Staircase Escalante National Monument spans nearly 1.9 million acres, and is located in Kane and Garfield Counties in south-central Utah and is part of the West-Central portion of the Colorado Plateau. My research attempted to estimate the harvest and grazing efficiency coefficients of forage by cattle in the arid, bunchgrass dominated system found in the Grand Staircase Escalante National Monument. This research also determined the effect of pinyon and juniper reduction on two vegetation treatments.

Harvest and grazing efficiency coefficients measured in moderately used areas produced mixed results. This was likely to the uneven distribution of cattle across the landscape. It was found harvest and grazing efficiency may not be an appropriate tool to refine stocking rates on the Grand Staircase Escalante National Monument.

Meta-analysis was used to assess the effects of pinyon and juniper reductions and reseeding. I looked at the changes in cover of grasses, forbs, shrubs, bare ground, and litter on two different sites on the Grand Staircase Escalante National Monument. Observations were recorded one year before treatment and one to five years after treatment. I found that pinyon and juniper removal and reseeding had a positive effect on perennial grass, perennial forb, and litter cover. Bare ground and native annual grass (*F. octoflora*) cover was negatively affected by treatment. Annual forb, invasive annual grass, and shrub were not found to be significantly affected by treatment of pinyon and juniper.

(86 pages)

PUBLIC ABSTRACT

Harvest Efficiency of Forage Grazed by Cattle and The Effect of Pinyon and Juniper Treatments on Vegetation Cover on the Grand Staircase Escalante National Monument

Ruger P. Carter

The Grand Staircase Escalante National Monument (GSENM), located in central south Utah, currently has 76,957 active grazing animal unit months on the monument. Recently, there has been questions whether the harvest and grazing efficiency coefficients developed in the Midwest are applicable to the arid, bunch grass dominated systems of the GSENM. Harvest and grazing efficiency defines the percentage of allocated forage that is being ingested by the animal, and the percentage that is being wasted. Harvest and grazing efficiency coefficients were calculated on the Lower Cattle allotment on the GSENM by taking total forage production and dividing that by expected cattle intake. Expected cattle intake was estimated by calculating stocking rates in study areas using a resource selection function that predicted cattle distribution. Total forage production was calculated using the paired plot method.

The GSENM has also needed data analyzed from pinyon and juniper removal projects. The effect size of pinyon and juniper removal treatments on the GSENM were also analyzed to find changes in grasses, forbs, shrubs, bare ground, and litter. Land managers need this analysis to better inform their decisions and determine the success of their treatments.

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CONTENTS

LIST OF TABLES

LIST OF FIGURES

CHAPTER 1

INTRODUCTION AND LITERATURE REVIEW

Harvest and Grazing Efficiency of Forage by Cattle

Rangelands

Rangelands are an important resource that provides energy, water, recreation, timber, minerals, and food to the human population (Holechek et al. 2011). The Society for Range Management (1998) defines Rangelands as “Land on which the indigenous vegetation (climax or natural potential) is predominantly grasses, grass-like plants, forbs, or shrubs and is managed as a natural ecosystem. If plants are introduced, they are managed similarly. Rangeland includes natural grasslands, savannas, shrub-lands, many deserts, tundra, alpine communities, marshes and meadows.” Recent estimates suggest that rangelands cover 18-80% of the earth’s surface. These variations in these estimates are due to differences definitions used to define rangelands (Lund 2007.) However, it is agreed upon that rangelands are an important part of the world’s natural resources.

Rangeland Management

The Society for Range Management (1998) defines rangeland management as “A distinct discipline founded on ecological principles and dealing with the use of rangelands and range resources for a variety of purposes. These purposes include use as watersheds, wildlife habitat, grazing by livestock, recreation, and aesthetics, as well as other associated uses.” Due to the importance of rangelands, it is important to properly manage them to sustain and increase the resources found thereon.

However, rangelands have not always been properly managed throughout history. Wildfire, livestock grazing, drought, wildlife, and humans have contributed to the degradation of rangelands throughout North America (Milton et al. 1994, Fleischner 1994, Scasta et al. 2016, Cingolani et al. 2005, Jones 2000). Livestock grazing has the most widespread influence on rangelands in Western North America, and over-utilization of vegetation can lead to degradation of rangelands. (Crumpacker 1984, Fleischner 1994, Jones 2000).

History of Livestock Grazing

During the homesteading era, the rangelands of Western North America were often over-utilized leading to degradation of the Range. In response to the over-utilization of forage, the Taylor Grazing Act (U.S Dept. of the Interior 1934) was passed to regulate grazing on public lands. The Taylor Grazing Act created grazing districts and a fee for animal unit months (AUM). An AUM is the amount of forage that a cow/calf pair will eat in one month.

A large portion of rangelands that are being grazed in the Western United States are owned by the Bureau of Land Management and the Forest Service. Currently, there are 19,689,128 AUMs grazing under 27,696 permits on public lands (LaFrance and Watts 1995).

Because of the negative impact of historical grazing regimes, livestock grazing on public lands has become one of the most controversial natural resource topics in the Western United States (Holechek 1991).

Stocking Rate

Selecting the correct stocking rate is one of the most important range management decision a land manager can make (Holechek et al. 1999). To avoid over-utilization that could lead to degradation of rangelands, many land managers use the take half, leave half method where 50% of the forage is allocated to livestock, and 50% is left for range and watershed health (Green and Brazee 2012). However, not all the forage allocated to livestock is consumed by the animal. During grazing, part of the forage used by livestock is 'ingested', and part is 'wasted' through trampling or spoilage via manure, urine, and bedding (Green and Brazee 2012, Galt et al. 2000). Other factors not attributed to livestock that can utilize forage include wildlife, insects, and weathering (Quinn and Hervey 1970). Calculating a stocking rate based on estimates of how much the animals consumes, and not considering waste could lead to over-utilizing rangeland.

Harvest Efficiency

To select a proper stocking rate that reduces the risk of over-utilization, harvest efficiency needs to be considered. The National Range and Pasture Handbook (Butler et al. 2003) defines harvest efficiency as "The percentage of forage actually ingested by the animals from the total forage produced." This harvest efficiency percentage shows how much forage is being consumed by the target animals.

Equation for Harvest Efficiency:

*Intake / Total Forage Production * 100 = Harvest Efficiency Percentage (Figure 1.1 Green and Brazee, 2012)*

Grazing Efficiency

Grazing efficiency is also a helpful figure when considering efficiency. The NRPH defines grazing efficiency as, “Of all forage utilized (this includes what is wasted), that portion actually ingested by the animal is grazing efficiency.” (Green and Brazee 2012). Grazing efficiency is closely related to harvest efficiency and gives an estimate of how much of the allocated forage is being consumed, and how much is being wasted.

Equation for grazing efficiency:

*Intake/ Total Forage Production - Residual) * 100 = Grazing Efficiency (Figure 1.2 Green and Brazee, 2012)*

Implications

Harvest and grazing efficiency are intended to help producers better manage grazing. Ultimately, it encourages grazing managers to shorten the time animals spend in pastures. To increase harvest efficiency managers, increase the stocking density and shorten the time spent in pastures in order to waste less forage. This happens because livestock will consume forage before it can be wasted.

Calculating a Stocking Rate Using Harvest Efficiency

To select a stocking rate using harvest efficiency, first calculate total forage production then multiply total forage production by the harvest efficiency for the specific rangeland. This will provide total forage available for consumption. Then divide that by the expected monthly intake per animal, which will provide the number of AUMs.

Calculating Animal Unit Months Using Harvest Efficiency:

*Pounds of Forage per Acre * Number of Acres = Total Pounds of Production*

*Total Pounds of Production * Harvest Efficiency = Forage Available for Consumption*

Forage Available for Consumption / Expected Monthly Intake Per Animal = Number of Animal Unit Months

Example:

*1000 Pounds of Forage per Acre * 100 Acres = 100,000 Pounds of Total Forage Production*

*100,000 lbs. Total Forage Production * 25% Harvest Efficiency = 25,000 lbs. Available Forage.*

25,000 lbs. of available forage / 913 lbs. Expected Monthly Intake = 27.38 Animal Unit Months

Calculating a Stocking Rate Using Grazing Efficiency

To select a stocking rate using grazing efficiency, first calculate total forage production, and then multiply that by desired utilization percentage. This will provide the total forage available for utilization. Multiply total forage available for utilization by the grazing efficiency coefficient to calculate the amount of forage to be consumed. Divide the amount of forage to be consumed by the estimated monthly intake per animal, which will provide the number of AUMs. Note that grazing efficiency and harvest efficiency are both related and using each method to calculate a stocking rate will produce the similar result.

Calculating Animal Unit Months Using Grazing Efficiency:

*Pounds of Forage per Acre * Number of Acres = Total Pounds of Production*

*Total Pounds of Production * Desired Utilization = Available Forage*

*Available Forage * Grazing Efficiency = Forage to be Consumed*

Forage to be Consumed / Expected Monthly Intake = Number of Animal Unit Months

Example:

*1000 Pounds of Forage per Acre * 100 Acres = 100,000 Pounds of Total Forage Production*

*100,000 lbs. Total Forage Production * 50% Utilization = 50,000 lbs. Available Forage.*

*50,000 lbs. Available Forage * 50% Grazing Efficiency = 25,000 lbs. Forage to be Consumed*

25,000 lbs. Consumed Forage / 913 lbs. Expected Monthly Intake = 27.38 Animal Unit Months

Harvest and Grazing Efficiency Research

There is limited research on harvest and grazing efficiency. Current research on harvest and grazing efficiency has been conducted on the Great Plains in mixed-grass prairie. Smart et al. (2010) found that moderate stocking rates with 50% utilization have a harvest efficiency of 25%, meaning approximately 25% of the forage is wasted and/or spoiled, while 25% of the forage is ingested by the animal. Galt et al. (2000) made detailed evaluations of actual forage use on several New Mexico rangelands. They consistently found that actual use was 10-15% higher than intended. They attributed this to wildlife and natural disappearance. Paulsen and Ares (1962) recommended a 35% utilization rate, however, they found that the harvest efficiency coefficient should be set at 30% to obtain a 35% utilization. They attributed this to trampling, ingestion by wildlife, and weathering loss.

However, little is known about how harvest efficiency differs in other rangeland types, especially more arid range types dominated by bunch grasses and shrubs. Balph and Malecheck (1985) found that cattle avoid stepping on elevated bunch grasses, which would decrease the amount of waste by trampling in bunch-grass dominated systems. This would increase harvest efficiency due to less forage being trampled and wasted by livestock, leaving more forage for consumption. Large interspaces between plants would also decrease waste of forage by defecation, urination, and bedding. Therefore, harvest efficiency coefficients from the Great Plains may not be applicable to grazing in more arid bunch-grass dominated rangelands like what is found on the Grand Staircase Escalante National Monument.

There has been limited research on harvest and grazing efficiency on arid, bunch-grass dominated rangelands. Galt et al. (2000) recommended a harvest efficiency coefficient of 25% to reach utilization levels of 30-35% for most western rangelands. This would allow livestock to consume 25% of the forage, while 10-15% of forage is utilized through trampling, wildlife, and weathering. Table 1.1 shows the recommended harvest efficiency coefficients and utilization percentages for the Great Plains, Chihuahua Desert, and the mixed grass-shrub ranges of Arizona and New Mexico.

Research Questions

Land managers looked at using harvest efficiency coefficients in order to refine stocking rates on the Grand Staircase Escalante National Monument. Using harvest efficiency coefficients could potentially change the amount of animal unit months (AUMs) allowed on each allotment. Harvest efficiency is affected by forage type, forage maturity, forage distribution, topography, livestock distribution, and stocking density. (Meehan et al. 2018). Due to the differences in these factors, harvest and grazing efficiency coefficients developed in the North American Great Plains may not be applicable to the Grand Staircase Escalante National Monument. My research attempted to quantify harvest and grazing efficiency of forage by grazing cattle in the arid, bunch grass dominated system found on the GSENM.

1. What is the harvest efficiency of forage by cattle on the GSENM?
2. What is the grazing efficiency of forage by cattle on the GSENM?

The Effect of Pinyon and Juniper Treatments on Vegetation Cover

Pinyon and Juniper

Pinyon (*Pinus spp.)* and juniper (*Juniperus spp.*) woodlands cover over 44 million acres in the Intermountain West (Miller et al. 2008) and cover 100 million acres in the Western United States (Romme et al. 2009). The juniper species found in the Western United States are western juniper (*J. occidentalis*), Utah juniper (*J. osteosperma*), one-seed juniper (*J. monosperma*), Rocky Mountain juniper (*J. scopulorum*), and alligator juniper (*J. deppeana*). The pinyon pine species found in the Western United States are single-leaf pinyon (*P. monophylla*), two-needle pinyon (*P. edulis*), and Mexican Pine (*P. cembroides*).

Pinyon and juniper woodlands are primarily used for livestock grazing. It is estimated that 80% of pinyon and juniper woodlands in the United States are used for livestock grazing (Evans 1998). Pinyon and juniper woodlands are also used for fuelwood, pinyon pine nut harvesting, recreation, and lumber. They also provide habitat for wildlife, and a watershed and hydrologic value to rangelands (Evans 1988, Paulin et al. 1999).

Pinyon and Juniper Encroachment

Since European settlement, pinyon (*Pinus* spp.) and juniper (*Juniperus* spp.) have expanded in range and density. There have been several studies that have documented the expansion of pinyon and juniper woodlands into shrub-steppe and grassland communities (Van Auken 2000, Bradley and Fleishman 2008, Blackburn and Tueller, 1970, Miller and Wigand, 1994). Pinyon and juniper have expanded in range and density due to fire

suppression, livestock grazing, natural range expansion, altered climate patterns, and elevated carbon dioxide levels (Romme et al. 2009, Evans 1988, Miller et al. 2019).

Miller et al. (2008) showed pinyon and juniper have increased between 125-625%, and they found that since 1860, the area occupied by pinyon and juniper has increased between 125-625% in Idaho, Oregon, Nevada, and Utah. In addition, they found the majority of the woodlands were in early stages of development. With the absence of disturbance, woodlands will continue to expand, mature, and close. This can lead to negative impacts on forage availability, wildlife habitat, and watersheds. When pinyon and juniper invade grasslands and shrub steppes understory cover declines, which leads to a loss of forage and habitat for livestock and wildlife (Miller et al. 2005). Pinyon and juniper invasion changes soil fertility, alters the plant community (Miller and Tausch, 2000), and increases soil erosion (Wilcox and Breshears, 1994). Pinyon and juniper expansion can alter wildlife distribution and survival as well. Pinyon and juniper encroachment into sagebrush has been documented to negative impact the distribution and survival of the greater sage-grouse (Coates et al. 2017).

Healthy Forest Restoration Act

To address the problem of expanded woodlands, the Healthy Forest Restoration Act (HFRA 2003) was introduced to conduct hazardous fuel reduction treatment on federal lands. The purpose of this act is to reduce high severity fires, restore forest ecosystems, and protect habitat for threatened and endangered species. In response to the HFRA, there have been many programs and projects implemented to reduce hazardous fuels. The Utah Watershed Restoration Initiative (UWRI) is one example of these

programs. UWRI restoration projects restores and prevents the destruction of watersheds by promoting positive changes to reduce future problems, primarily by reducing pinyon and juniper that have encroached into shrubland (UWRI 2019).

Benefits of Pinyon and Juniper Removal

There are many benefits to removing pinyon and juniper from the landscape. Some benefits include a decrease in fuel load (Redmond et al. 2013), an increase in herbaceous plant cover and diversity (Brockway et al. 2002), decreased soil erosion (Hastings et al. 2003), and increased soil moisture (Roundy et al. 2014). Redmond et al. (2013) studied the long term (20-40 year) effects of chaining treatments on vegetation structure in pinyon and juniper woodlands. They found treated areas had a significant decrease in pinyon and juniper when compared to untreated sites (Figure 1.4).

Bates et al. (2000) found pinyon and juniper removal increased soil water availability and enhanced understory vegetation cover. Roundy et al. (2017) suggested tree removal by chaining combined with seeding, increased vegetation cover and reduced runoff and erosion. Williams et al. (2019) found that pinyon and juniper removal treatments can initially improve infiltration and limit hillslope runoff and erosion if tree debris is sufficiently distributed into bare patches and in contact with the soil surface.

Pinyon and juniper removal can also positively impact animal populations. Peterson et al. (2017) found that pinyon and juniper removal has a positive impact on small mammal populations, due to microhabitats created when pinyon and juniper are removed. Bergman et al. (2014) found that pinyon and juniper removal increased desirable browse species, and had a positive impact on mule deer fawn survival.

Commons et al. (1988) saw an increased population of male sage grouse in areas where pinyon and juniper were reduced. Frey et al. (2013) found that sage grouse use increased in areas where pinyon and juniper were removed, and decreased in areas still dominated by pinyon and juniper.

Disadvantages of Pinyon and Juniper Removal

Pinyon and juniper treatments can be controversial. (Jones 2019, Review of the Literature). There are also risks involved in pinyon and juniper removal, including undesirable impacts on plant community composition and an increase in invasive species (Bates et al. 2000). Baughman et al. (2010) reported an increase in downy brome (*Bromus tectorum*) when pinyon and juniper were removed. Bybee et al. (2016) reported low resistance to invasive annuals where few pre-treatment shrubs, grasses, and forbs remained.

Certain wildlife species also are negatively impacted by pinyon and juniper removal (Bombaci and Pejchar 2016). Pavlacky and Anderson (2001) found that pinyon and juniper obligate birds favored areas with greater pinyon pine cover and high canopy height. They recommend maintaining pinyon pine is critical to providing quality habitat for these species. Francis et al. (2011) found that 86% of nests in live trees that belonged to open cup and cavity nesting birds occurred in juniper trees, and recommended that the selective removal of juniper be avoided when thinning juniper woodlands.

Pinyon and Juniper Removal Methods

When pinyon and juniper encroach into shrublands and grasslands, the general methods used to remove the trees include chaining, mastication, hand thinning, burning,

and herbicide (Clary 1974, Miller et al. 2005). For the purpose of this literature review, mechanical treatments and hand thinning will be reviewed.

Chaining is used to treat large areas, and it unselective of which trees are removed. Tausch and Tueller (1977) found a significant increase in understory vegetation in the years immediately following chaining. They found that maximum forb cover was achieved 2 years post treatment, and maximum perennial bunch grass cover was achieved 3-4 years post treatment.

Mastication of pinyon and juniper is used as a selective way to remove trees. Ross et al. (2012) found that total understory cover in sites that have been masticated in the previous two growing seasons was 5–16 fold higher than controls. Johnston (2014) compared three different thinning methods (Chaining, Roller, Mastication). They found that two years post-treatment, the responses of desirable perennials was similar among mechanical treatment types, with all treatments producing 10-15 times higher grass biomass, 2-3 times higher grass cover, and higher shrub biomass (non-significant trend) than control plots.

Hand thinning using chainsaws is used in smaller areas, and can be used to selectively harvest trees. There are two different hand-thinning treatments: pile burn; where trees are cut with chainsaws, and debris is placed in piles that are burned, and lop and scatter; where trees are cut with chainsaws, and the debris is scattered across the site (Ross et al. 2012). Ross et al. (2012) found that following hand thinning, understory plant cover was 4–5.5 fold higher in the pile burn and lop & scatter respectively relative to the untreated control. Loftin (1998) found a significant increase in grass and forb cover following hand thinning of pinyon and juniper in the Santa Fe National Forest.

Forage Response to the Removal of Pinyon and Juniper on the Colorado Plateau

Many studies have shown a significant effect on forage production and cover following pinyon and juniper removal on the Colorado Plateau. In Arizona, Clary and Jameson (1981) found that average production following pinyon and juniper removal increased from pretreatment values by the following proportions: grasses, 10.5 times; forbs, 6 times; shrubby plants, 1.67 times; and total herbage 6.67 times.

Stephens et al. (2016) found that mechanical removal of pinyon and juniper in Northwestern Colorado can result in increased understory vegetation in relation to untreated areas 2 years post treatment. Specifically, they found that grass biomass increased 10-15 times post treatment.

Bybee et al (2016) found shredding trees maintained shrub cover and increased perennial herbaceous on sites throughout the Colorado Plateau and the Great Basin. After shredding or shredding and seeding, perennial herbaceous understory cover increased (generally to >20%) to equal or exceed that at early phases of infilling (<10% tree cover), at mid (15-35%) to high (90%) ranges of pretreatment tree cover.

Forage Response to the Removal of Pinyon and Juniper on the Grand Staircase Escalante National Monument

Redmond et al. (2013) studied the long term (20-40 years) effects of chaining treatments on vegetation structure in pinyon and juniper woodlands on the Grand Staircase Escalante National Monument (GSENM). They found that past chaining treatment methods were effective at increasing understory cover, even 40 years post-treatment (Figure 1.3). Total herbaceous cover was over four times as high (8.1% as

opposed to 1.7%) on sites that had been treated as compared to untreated sites. However, Redmond et al. (2013) did not see a long term effect on understory plant diversity like what has been found in other research (O'Meara et al. 1981).

Evangelista et al. (2004) studied the vegetation response to fire and postburn seeding treatments in juniper woodlands on the GSENM. They found that native species richness, percent cover of native species, and total biological soil crust cover were higher on unburned plots. They attributed this to the site characteristics, and a high domination of cheatgrass (*B. tectorum*). It was suspected that burned areas provided ideal conditions for cheatgrass. When compared to Redmond et al. (2013) research, mechanical removal of pinyon and juniper may be a more appropriate method at increases forage cover.

Research Questions

The GSENM is interested in analyzing existing data from previously conducted pinyon and juniper reduction treatments on the monument. The objective of this study is to determine the effect on forage cover when pinyon and juniper are removed from the landscape on the GSENM. My main hypothesis is that when pinyon and juniper are removed, forb and perennial grass cover will increase, and bare ground will decrease. My rationale for this hypothesis comes from previous research that has found similar findings (Clary and Jameson 1981, Roundy et al. 2017, Roundy et al. 2014, Redmond et al. 2013).

1. What is the effect of pinyon and juniper removal on herbaceous vegetation, shrub, and bare ground cover?
2. What is the mean change in herbaceous vegetation, shrub, and bare ground cover when pinyon and juniper are removed?

Treatment Objectives:

1. Remove 100% of encroaching pinyon and juniper.
2. Re-introduce perennial grasses, forbs and shrubs on 3,293 acres that are being displaced by encroaching pinyon and juniper.
3. Re-establish perennial grasses, forbs and shrubs beneficial to sage grouse and other sagebrush species.
4. Restore percent canopy cover of shrubs to 30%, forbs to 5% and grasses to 30%.

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Tables and Figures

Table 1.1 - Recommended harvest efficiency coefficients and utilization percentages.

Author	Location of Study	Recommended Harvest Efficiency Coefficients	Utilization
Smart et al. 2010	Great Plains	25%	50%
Galt et al, 2000	Chihuahua Desert of New Mexico	25%	30-35%
Paulsen and Ares 1962	Mixed Grass-Shrub Ranges of Arizona and New Mexico	30%	35%

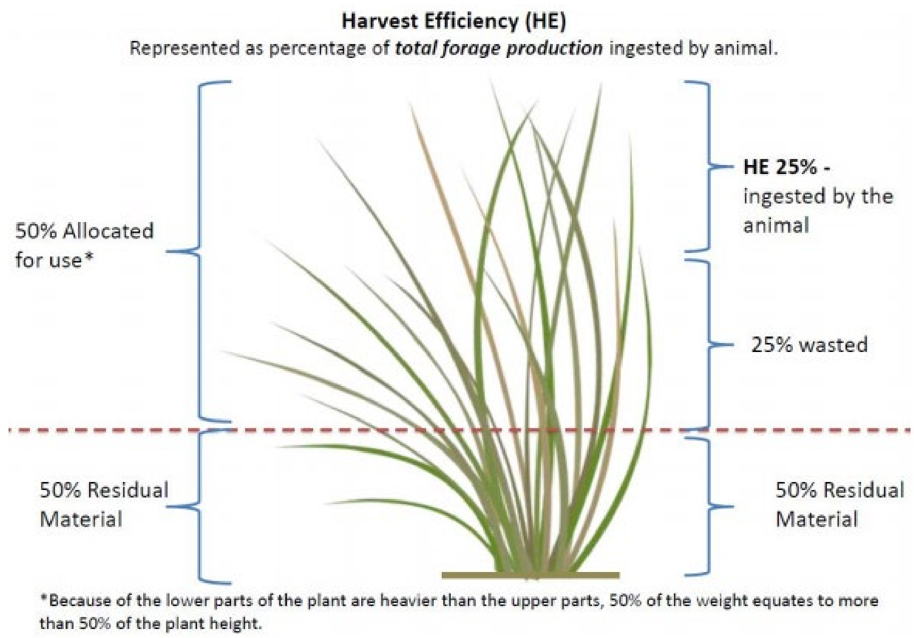

Figure 1.1 - Representation of harvest efficiency, adapted from Green and Brazee 2012.

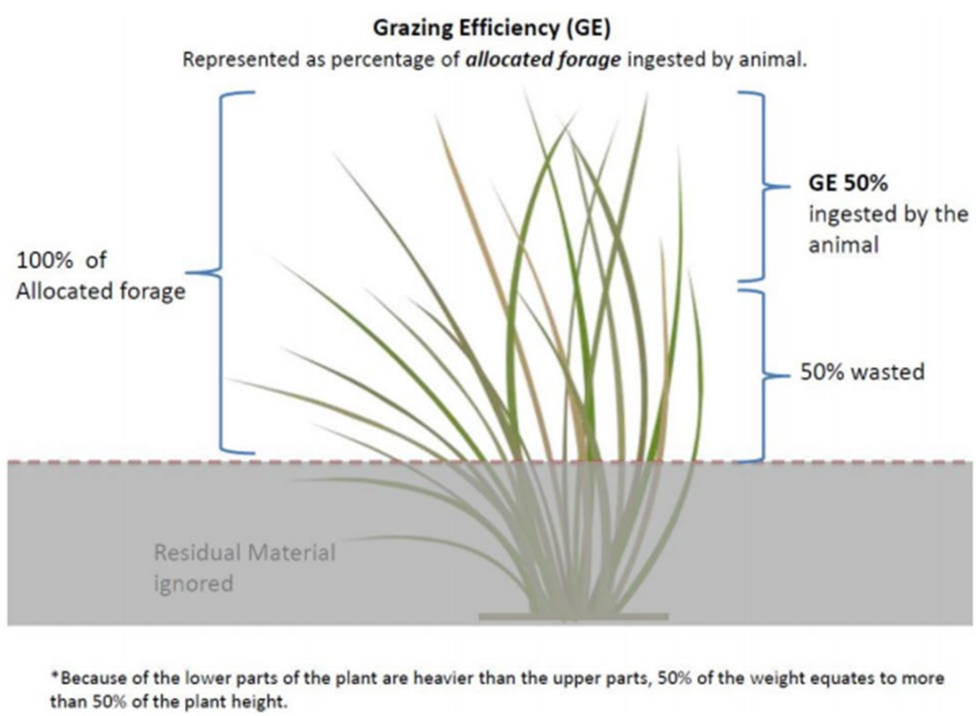

Figure 1.2 - Representation of grazing efficiency, adapted from Green and Brazee 2012.

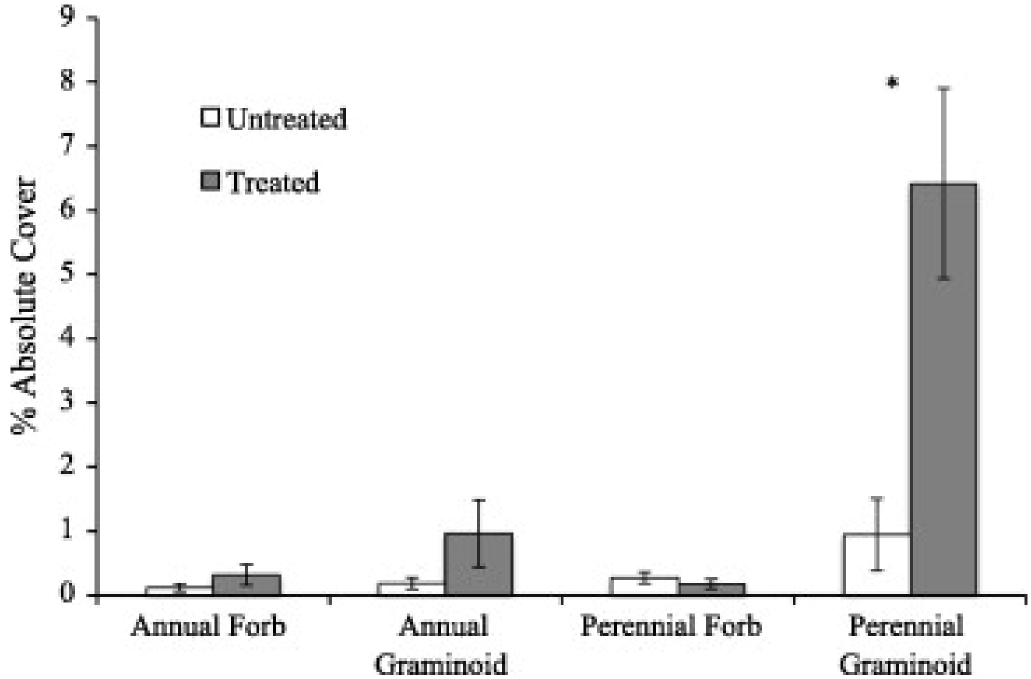

Figure 1.3 - Differences between treated (chained and seeded) and untreated sites in J. osteosperma and P. edulis seedling (BTD <2.5 cm) and sapling (BTD <5 cm and ⩾2.5 cm) densities at Grand Staircase-Escalante National Monument, Utah. Data are means ± 1 SE and an asterisk denotes significant differences between treated and untreated sites, with α = 0.05 (Redmond et al. 2013).

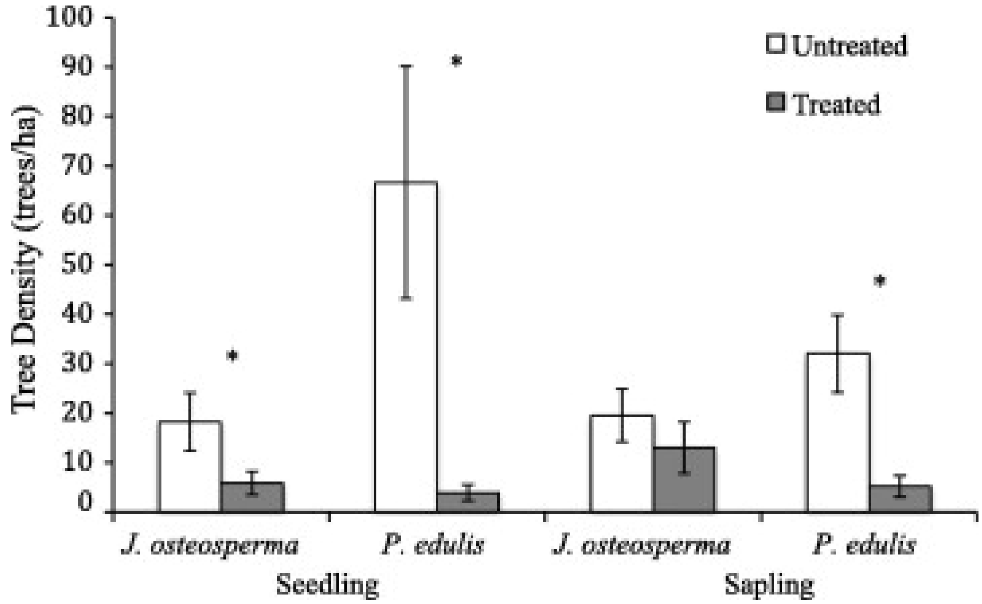

Figure 1.4 - Percent absolute cover of annual forbs, annual graminoids, perennial forbs, and perennial graminoids in treated (chained and seeded) and untreated sites at Grand Staircase-Escalante National Monument, Utah. Data are means ± 1 SE and an asterisk denotes significant differences between treated and untreated sites, with $\alpha = 0.05$ (Redmond et al. 2013).

CHAPTER 2

HARVEST AND GRAZING EFFICIENCY OF FORAGE GRAZED BY CATTLE ON THE GRAND STAIRCASE ESCALANTE NATIONAL MONUMENT

Abstract

Harvest and grazing efficiency has been used to refine stocking rates on rangelands throughout the North American Great Plains. The harvest efficiency coefficients developed on these rangelands show that a 25% harvest efficiency coefficient is needed to obtain 50% utilization under a moderate stocking rate. However, there is little information on harvest efficiency estimates on other rangeland types, especially arid landscapes that are dominated by bunch-grasses and shrubs. The purpose of this research was to quantify harvest and grazing efficiency coefficients for the Grand Staircase Escalante National Monument. In order to calculate harvest and grazing efficiency, utilization, and forage intake by cattle were calculated. I found that harvest and grazing efficiency differs across the landscape due to the heterogeneous nature of the allotment. Because of this, it may not be appropriate to refine stocking rates using harvest and grazing efficiency in the way it is has been used in other rangeland and pasture settings that are more homogenous in nature.

Introduction

Selecting the correct stocking rate is one of the most important range management decision a land manager can make (Holechek et al. 1999). To avoid overutilization of forage that could lead to degradation of rangelands, many land managers use the take half, leave half method where 50% of the forage is allocated to livestock, and 50% is left

for range and watershed health. However, not all the forage allocated to livestock is consumed by the animal. During grazing, part of the forage used by livestock is 'ingested', and part is 'wasted' through trampling or spoilage via manure, urine, and bedding (Green and Brazee 2012, Galt et al. 2000). Other factors not attributed to livestock that can utilize forage include wildlife, insects, and weathering (Quinn and Hervey 1970). Calculating a stocking rate based on estimates of how much the animals consumes, and not considering waste could lead to over-utilizing forage.

To select a proper stocking rate that reduces the risk of overutilization, harvest efficiency needs to be considered. The National Range and Pasture Handbook (Butler et al. 2003) defines harvest efficiency as "The percentage of forage actually ingested by the animals from the total forage produced." The harvest efficiency percentage shows how much forage is being consumed by the target animals.

Grazing efficiency is also a helpful figure when considering efficiency. The NRPH defines grazing efficiency as, "Of all forage utilized (this includes what is wasted), that portion actually ingested by the animal is grazing efficiency." (Green and Brazee 2012).

In some rangelands across the North American Great Plains, harvest and grazing efficiency has been used as a tool to refine stocking rates (Smart et al. 2010). The harvest efficiency coefficients developed on these rangelands show that a 25% harvest efficiency coefficient is needed to obtain 50% utilization under a moderate stocking rate. Similarly, a grazing efficiency coefficient of 50% is needed to achieve 50% utilization under a moderate stocking rate. However, there is little information on harvest and grazing efficiency estimates on other rangeland types, especially arid landscapes that are

dominated by bunch-grasses and shrubs. The purpose of my research was to quantify harvest and grazing efficiency of forage by cattle on the Grand Staircase Escalante National Monument. The scientific research I conducted was inspired by questions raised by land managers and ranchers about the validity of using harvest and grazing efficiency as a tool to refine stocking rates on the Grand Staircase Escalante National Monument.

Study Area

My research was conducted on the Lower Cattle Allotment on the Grand Staircase Escalante National Monument (GSENM). The GSENM spans nearly 1.9 million acres and is located within Kane and Garfield Counties in Southern Utah and the West-Central portion of the Colorado Plateau. The monument is bordered by the Dixie National Forest to the north, Capitol Reef National Park to the east, Glen Canyon National Recreation Area to the southeast, Bureau of Land Management to the south and west, and Bryce Canyon National Park to the northwest.

The area studied within the GSENM is found on the Lower Cattle grazing allotment 20 miles southeast of Escalante, Utah, and is on the Kaiparowits Plateau region of the monument. The allotment is 32,921.17 acres and is bounded by the 50 Mile Mountain on the west and the Glen Canyon National Recreation Area to the east (BLM 2000).

The Lower Cattle allotment ranges in elevation from approximately 1500 to 1700 m, with an average precipitation of 25.5 cm. The major soil type found in this allotment are find sands and are mainly dominated by perennial warm and cool season grasses and shrubs. The graminoids commonly found in this site include needle-and-thread

(*Hesperostipa comata*), Indian ricegrass (*Achnatherum hymenoides*), sand dropseed (*Sporobolus cryptandrus*), blue grama (*Bouteloua gracilis*), and James galleta (*Pleuraphis jamesii*). Common shrubs include fourwing saltbush (*Atriplex canescens*), sand sagebrush (*Artemisia filifolia*), and blackbrush (*Coleogyne ramosissima*).

The Lower Cattle allotment is a winter allotment, and cattle graze the allotment from October 1st to April 15th. There are four different permittees grazing in the Lower Cattle allotment. There are currently 7,488 active animal unit months (AUMs) on this allotment, with an average of 4,481 AUMs used between the years 1996 and 2013 (BLM 2015). Water and extreme slope are generally the limiting resource on this allotment, with some areas receiving limited use on years with a lack of precipitation. In the 2018-19 grazing year when this research was conducted, 60% of the active AUMs were used on this allotment due to a prolonged drought that has caused a decrease in forage production.

Methods

To estimate harvest and grazing efficiency of forage by cattle on the Lower Cattle allotment, total forage production, forage utilization, and cattle intake were calculated. The equations found below were used to calculate harvest and grazing efficiency.

Equation for Harvest Efficiency:

*Intake / Total Forage Production * 100 = Harvest Efficiency Percentage (Figure 1.1 Green and Brazee, 2012)*

Equation for grazing efficiency:

*Intake/ Total Forage Production - Residual) * 100 = Grazing Efficiency (Figure 1.2 Green and Brazee, 2012)*

Total Forage Production and Utilization

To calculate total forage production, utilization, and residual forage, 4 study areas (transects) were randomly placed in areas of moderate use described by land managers. 36 – 0.91m x 0.61 m cages were systematically placed in the 4 study areas (Figure 2.1). These cages excluded all ungulates from grazing (Figure 2.2). The BLM paired plot method was used at the end of the grazing season to estimate total forage production and utilization (Coulloudon et al. 1999). At the time when forage production and utilization were calculated, there were 3,297 animal unit equivalents (AUEs) that had grazed the Lower Cattle allotment.

Estimating Cattle Distribution and Intake
Using a Resource Selection Function

Due to the size of the allotment, and the unequal distribution of cattle, I created a resource selection function (RSF) to estimate the number of AUEs that grazed in the areas the cages were placed. To create the resource selection function, adult female cattle (n = 8), between 4 and 8 years old, were fitted with Advanced Telemetry Systems G5-2D Iridium/GPS Collars. The collars recorded locations every hour, and GPS points were collected from Oct 01, 2018, to April 15th, 2019. Abiotic variables (slope, aspect, elevation) and biotic variables (existing vegetation type) were obtained from USGS LANDFIRE Data (2018a, 2018b). A distance to water layer was created using ARC Map 10.7.1 (ESRI 2019). Existing vegetation type was split into five classifications. These classifications were shrub, grass, tree, none, and other. Red rock with no vegetation was classified as other in this RSF. These layers used in the RSF are found in appendix A.

The resource selection function was created using a generalized linear model (GLM) (used/available design; Boyce et al. 2002). To create the available points, I sampled every pixel in the allotment, which provided 359,955 available points. The cattle GPS points provided 29,762 used points. I then checked for linearity between the different variables but found none exceeding 0.6. To allow for comparison of variables, I standardized variables by subtracting the mean and divided by the standard deviation. To compare models, I used the Akaike Information Criterion (AIC; Boyce et al. 2002) to select the model with the lowest AIC value. I used R (R Core Team 2019) and RStudio (RStudio Team 2019) to perform all analyzes, along with the packages lme4 (Bates et al. 2015) and AICcmodavg (Mazerolle 2017). The top model used for this RSF was (used/available ~ slope, elevation, aspect, distance to water, vegetation type). The RSF was used to create a probability of use map (Figure 2.3) to provide an estimate of cattle distribution on the Lower Cattle allotment.

Allocation of AUMs and Predicting Cattle Intake

To allocated AUMs across the allotment and to calculate a stocking rate at study sites, the Lower Cattle allotment acreages were binned in 10% probability of use increments, and a reverse weight was placed on each probability of use bin. AUMs were then allocated based upon the acre*weight correction (Table 2.1).

A stocking rate was calculated for each transect location (Table 2.2) by using the probability of use from the RSF with the AUM allocation from table 2.1. Daily forage intake was estimated at 2.5% of the animal’s body weight. The average weight of the

cattle grazing this allotment was 545 kg. This equates to 415.56 kg of forage intake per AUM.

Results

Due to the heterogeneity of the landscape and the severe drought, forage production differed between the 4 transect locations. Transect 1 had 241.19 kg of forage/hectare. Transect 2 had 191.52 kg of forage/hectare. Transect 3 had 98.80 kg of forage/hectare. Transect 4 had 58.39 kg/hectare (Table 2.2).

Stocking rates were calculated for each study site based on the AUM allocation (Table 2.2). The allocation of AUMs using the RSF predicted the stocking rate to be 4.78 hectares/ AUM for Transects 1, 2, and 4. Transect 3 had a probability of use of 61%, so the stocking rate for this location was set at 3.41 hectares/AUM.

Grazing and harvest efficiency varied across the 4 treatment areas. Grazing efficiency was calculated at 48.58%, 71.13%, and 100% on Transects 1-3 respectfully. Harvest efficiency was calculated at 36.05%, 45.39%, and 100% on transects 1-3 (Table 2.3). Transect 4 did not receive any forage utilization by cattle, so grazing and harvest efficiency could not be calculated at this study site.

Discussion

Harvest and grazing efficiency of forage by cattle varied between the study sites. Harvest and grazing efficiency is influenced by forage type, forage maturity, forage distribution, topography, livestock distribution, and stocking density. Stocking density is influenced by cattle selection of resources on the landscape. Areas that contain desirable and abundant forage, have gentle slopes, and are near water are going to see higher

stocking densities. When calculating probability of use by cattle on the Lower Cattle allotment, it was found that 59% of the total 32,921 hectares falls below 20% probability of use. These areas are found far from water sources, on steep slopes, and have little to no forage. These areas are going to see a decreased stocking density, and therefore see a decreased harvest and grazing efficiency. Due to the heterogeneity of the rangelands found on the GSENM, harvest and grazing efficiency most likely changes with the landscape characteristics. As stocking density increases, so does harvest and grazing efficiency (Smart et al. 2010). As shown in the cattle probability of use heat map (Figure 2.3), there are a wide range of stocking densities on the Lower Cattle allotment. Therefore, it can be assumed that harvest and grazing efficiency will differ across the landscape.

Due to the size of the allotment, differences in forage resources, and the severe drought leading up to the grazing season, there was large differences in forage production across the transects. Cattle most likely were selecting areas with greater forage production. Utilization rates calculated at these areas supports this suggestion (Figure 2.4). Forage production was not captured with the resource selection function, which is why the resource selection function predicted probability of use to be similar across transects 1, 2, and 4, even though utilization differed between transects. Transect 1 had the highest production, and the highest utilization, followed closely by transect 2. Transect 3 saw little utilization, but had a low quantity of forage. Transect 4 had the lowest amount of forage production, and it did not see any utilization. Residual forage on transect 1, 2, and 3 never fell below the total forage production in transect 4 (Figure 2.5).

This can be attributed to the way cattle utilize their tongues to graze, and couldn't graze the remaining forage that was close to the ground.

Because cattle distribution differed significantly across the allotment and cattle utilization was not equal across the study sites, the grazing and harvest efficiency coefficients calculated are likely higher on transects 1 and 2 where there was higher utilization. Similarly, harvest and grazing coefficients that I calculated on transects 3 and 4 are probably lower than the 100% reported.

Smart et al. (2010) conducted their research on harvest and grazing efficiency in pastures ranging from 12.83 to 128 hectares, compared to my study that covered 32,921 hectares. The differences in acreage, forage type, forage distribution, topography, and livestock distribution between the two studies may be the reason why we did not see similar harvest and grazing efficiency coefficients.

Implications

Due to the heterogeneous nature of this allotment, and unequal distribution of cattle, there is a wide range of harvest and grazing coefficients found on the allotment. It may be impractical to refine stocking rates using harvest and grazing efficiency in the way it is used in other rangeland and pasture settings that are more homogenous in nature. It is recommended that this research is repeated in other areas to better understand harvest and grazing efficiency in bunch-grass dominated systems. Furthermore, research is needed to estimate how a resource selection function can be used to predict cattle utilization across the landscape, and how predicting livestock distribution and utilization using a resource selection function can help refine stocking rates.

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Tables and Figures

Table 2.1 - Lower Cattle allotment acres that fall within each probability of use bin. A reversed weighting system was used to account for higher utilization in areas with a greater probability of use. AUMs were then split among the weighted acreages, which allows for a stocking rate to be created at each study area location.

Probability of Use	Hectares	Weight	Acre*Weight Correction	Percent Area	Allocated AUMS	Stocking Rate HA/AUM
0-9%	9718.45	1	9718.45325	0.12344	404.05	23.88
10-19%	9751.99	2	19503.98515	0.24773	814.17	11.94
20-29%	6684.93	3	20054.77945	0.25473	836.32	7.96
30-39%	4697.88	4	18791.52583	0.23868	787.39	5.97
40-49%	1789.01	5	8945.03827	0.11362	380.45	4.78
50-59%	248.48	6	1490.90064	0.01894	64.86	3.98
60-69%	20.84	7	145.85494	0.00185	6.47	3.41
70-79%	7.04	8	56.29489	0.00072	2.32	2.98
80-89%	2.28	9	20.56226	0.00026	0.86	2.65
90-100%	0.27	10	2.74163	0.00003	0.11	2.39
Total	32921.18		78730.1363	3297	3,297	

Table 2.2 - Calculations of stocking rates at each study site using the probability of use calculated from the Resource Selection Function.

Study Area	Forage Production	Utilization	Utilization %	Probability of Use	HA/AUM
1	241.19 kg/ha	178.96 kg	74.20%	45.55%	4.78
2	191.52 kg/ha	122.21 kg	63.81%	40.41%	4.78
3	98.80 kg/ha	12.86 kg	13.05%	60.97%	3.41
4	58.39 kg/ha	0 kg	0%	43.77%	4.78

Table 2.3 - Calculations of harvest and grazing efficiency at each cage location using estimated stocking rates.

Transect Location	HA/ AUM	Available Forage	Estimated Intake	Actual Utilization	Harvest Efficiency	Grazing Efficiency
1	4.78	1,152.89 kg	36.05%	74.20%	36.05%	48.58%
2	4.78	915.47 kg	45.39%	63.81%	45.39%	71.13%
3	3.41	336.91 kg	100%	13.05%	100%	100%
4	4.78	279.10 kg	100%	0%	NA	NA

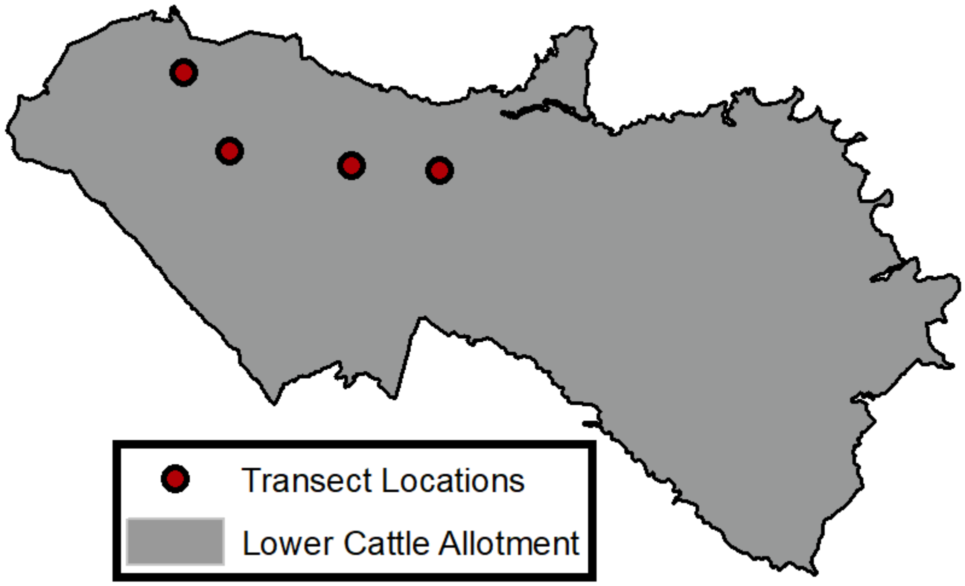

Figure 2.1 – Lower Cattle allotment and study areas.

Figure 2.2 – Photo of the 0.61m x 0.91m cages used to calculate total forage production.

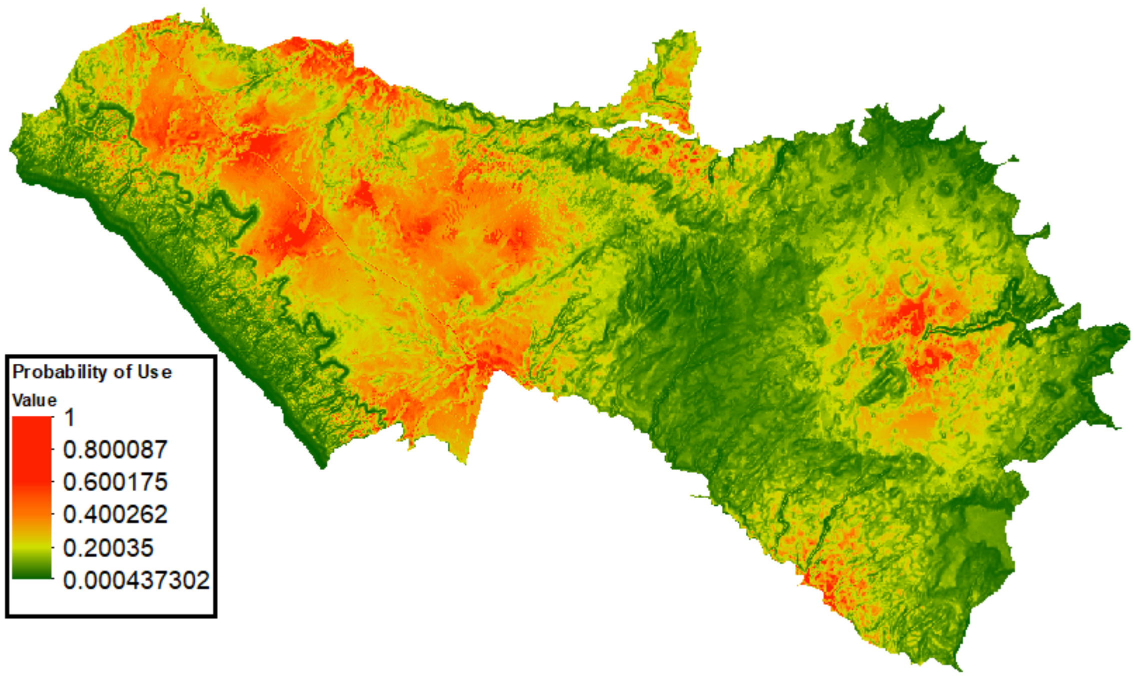

Figure 2.3 – Probability of use heat map created from the Resource Selection Function. The function used for this resource selection function is: (used ~ slope, elevation, aspect, distance to water, vegetation type)

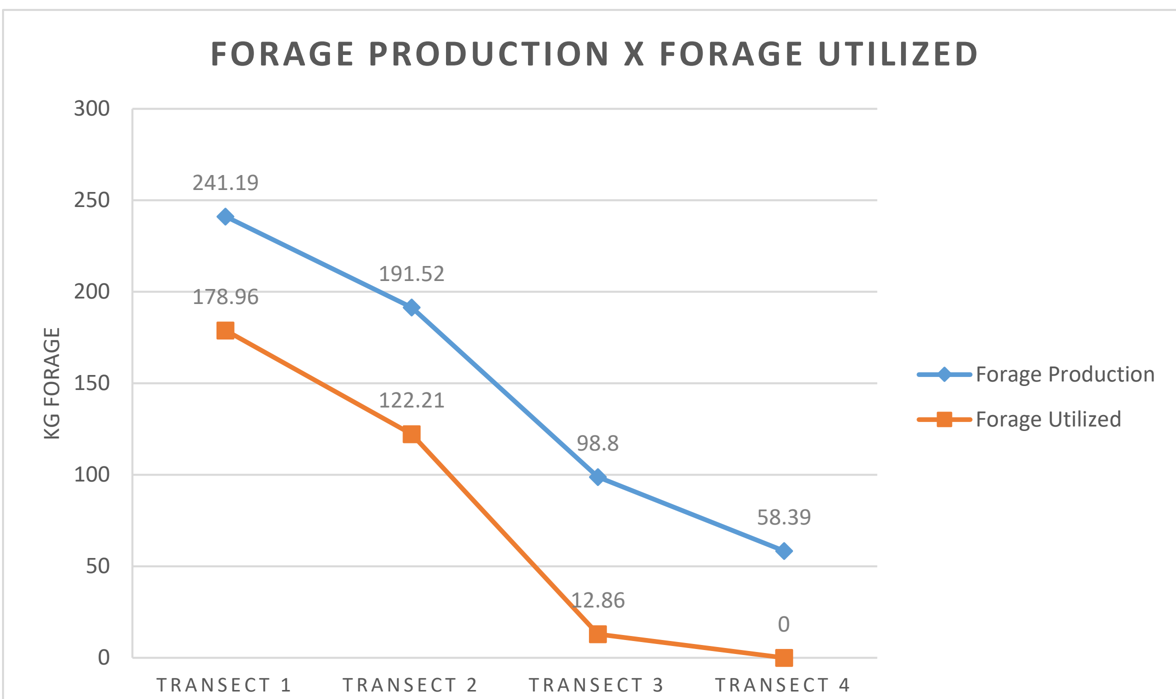

Figure 2.4 – Total forage production (kg/ha) plotted against forage utilized (kg/ha) on the 4 different study areas within the Lower Cattle Allotment.

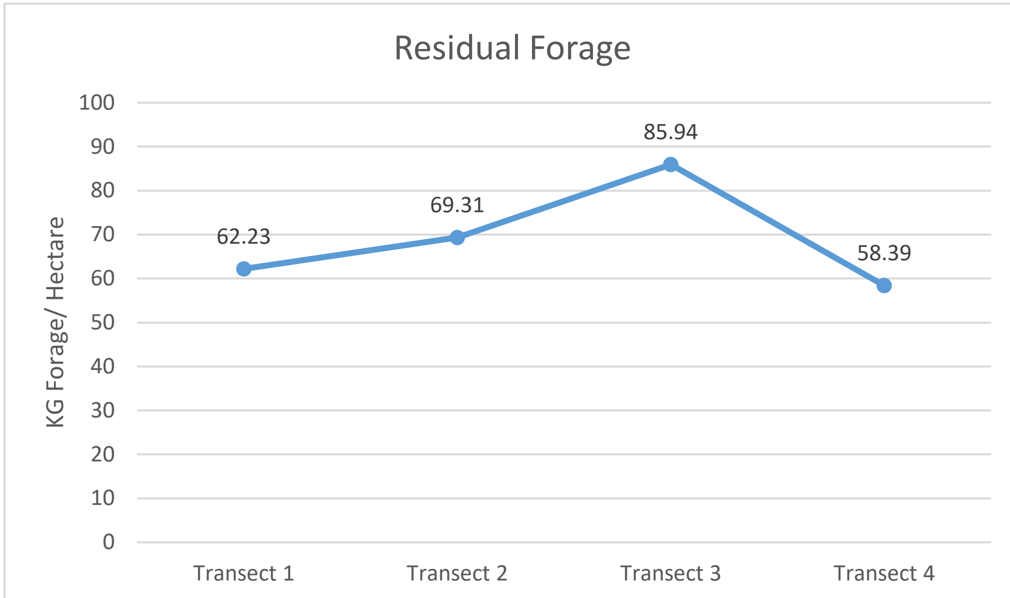

Figure 2.5 – Residual forage for each of the 4 study areas (transects) within the Lower Cattle allotment.

CHAPTER 3

THE EFFECT OF PINYON AND JUNIPER TREATMENTS ON VEGETATION COVER ON THE GRAND STAIRCASE ESCALANTE NATIONAL MONUMENT

Abstract

Encroaching pinyon and juniper negatively affects herbaceous cover, which leads to a decrease in forage production for livestock and wildlife. I used meta-analysis techniques to assess the effects of pinyon and juniper removal and reseeding on cover of grasses, forbs, shrubs, bare ground, and litter on two different treatment sites on the Grand Staircase Escalante National Monument. Observations were recorded one-year pre-treatment and four to five years post years post-treatment. It was found that treatment had significant positive effects on perennial grass, perennial forb, and litter cover. It was also found that treatment had a significant negative effect on bare ground and native annual grass cover. Annual forb, invasive annual grass, and shrub did not change significantly following treatment of pinyon and juniper. Further research is needed to understand the long term affects (6+ years post-treatment) of pinyon and juniper removal, and to confirm success criteria is being met on treatment sites.

Introduction

Since European settlement, pinyon (*Pinus spp.*) and juniper (*Juniperus spp.*) have significantly expanded in range and density. It is estimated that pinyon and juniper woodlands cover over 44 million acres in the Intermountain West (Miller et al. 2008) and cover 100 million acres in the Western United States (Romme et al. 2009). Many studies

have documented the expansion of pinyon and juniper woodlands into shrub-steppe and grassland communities (Van Auken 2000, Bradley and Fleishman 2008, Blackburn and Tueller 1970, Miller and Wigand 1994). Miller et al. (2008) found that since 1860, the area occupied by pinyon and juniper has increased between 125-625% percent in Idaho, Oregon, Nevada, and Utah.

In the study Miller et al. (2008) conducted where they found pinyon and juniper had increased between 125-625%, and it was found that most of the woodlands were in early stages of development. With the absence of disturbance, woodlands will continue to expand, mature, and close. This can lead to negative effects on forage production, wildlife, and watersheds. When pinyon and juniper invade grasslands and shrub steppes, understory cover declines, which leads to a loss of forage and habitat for livestock and wildlife (Miller et al. 2005). Pinyon and juniper invasion changes soil fertility, alters the plant community, and increases soil erosion (Wilcox and Breshears 1994, Miller and Tausch 2000, Miller et al. 2005). Pinyon and juniper expansion can alter wildlife distribution and survival. Pinyon and juniper encroachment into sagebrush negatively impacts the distribution and survival of the greater sage-grouse (Coates et al. 2017).

There are many benefits to removing pinyon and juniper from the landscape. Some benefits include an increase in herbaceous plant cover and diversity (Brockway et al. 2002, Stephens et al. 2016, Bybee et al. 2016, Redmond et al. 2013), decreased soil erosion (Hastings et al. 2003), and increased soil moisture (Roundy et al. 2014). Pinyon and juniper reduction can also benefit wildlife (Coates et al. 2017, Bergman et al. 2014, Peterson et al. 2017).

Though there are many benefits of removing encroaching pinyon and juniper, there are also risks involved in pinyon and juniper removal, including change in plant community composition, and potential increases in invasive species (Baughman et al. 2010).

The purpose of my research is to determine the effect of pinyon and juniper reduction on forage cover. Specifically, I sought to calculate the effect of pinyon and juniper reduction on perennial grass, native annual grass, invasive annual grass, perennial forb, annual forb, shrub, bare ground, and litter.

Study Site

The Grand Staircase Escalante National Monument (GSENM) spans nearly 1.9 million acres and is located within Kane and Garfield Counties in Southern Utah and the West-Central portion of the Colorado Plateau. The monument is bordered by the Dixie National Forest to the north, Capitol Reef National Park to the east, Glen Canyon National Recreation Area to the southeast, Bureau of Land Management to the south and west, and Bryce Canyon National Park to the northwest. Treated areas in this study include the Pine Point hand thin treatment, and the Ford Pasture hand thin and mastication treatment.

The Pine Point and Ford Pasture treatment areas are located 40 kilometers northeast of Kanab, Utah on the Skutumpah Terrace (UTM - 37.67 E, 41.23 N). The Pine Point pasture was hand thinned and seeded (Figure 3.1). The Pine Point Pasture had 1024 hectares treated. The Ford pasture was hand thinned and masticated (Figure 3.2). The

hand thinned area of the project consisted of 151 hectares. The mastication area consisted of 158 hectares.

The project areas once consisted of grass species, sagebrush, and other browse species. Before the treatment, pinyon and juniper were expanding into these areas. This increase in pinyon and juniper had negatively affected plant community composition and diversity, increased erosion, and altered fire regimes and wildlife habitat. These projects were proposed to maintain areas, open travel corridors, and provide benefits to the southernmost population of greater sage grouse in the United States. Additionally, the project area is highly important mule deer grounds for the Paunsaugunt and Kaiparowits deer herds and important habitat for elk and many shrub steppe birds. The watersheds also have a high potential for erosion due to the soil types and lack of herbaceous understory in the project area (McQuivey 2013).

The Pine Point hand thinning was completed in the fall of 2012. The Ford Pasture hand thinning and mastication treatment was conducted in the fall and winter of 2005/2006. Both areas treated were rested from grazing for a minimum of two complete growing seasons.

Treatment Objectives (McQuivey 2013):

1. Remove 100% of encroaching pinyon and juniper.
2. Re-introduce perennial grasses, forbs and shrubs on 1,333 hectares that are being displaced by encroaching pinyon and juniper.
3. Re-establish perennial grasses, forbs and shrubs beneficial to sage grouse and other sagebrush species.
4. Restore percent canopy cover of shrubs to 30%, forbs to 5% and grasses to 30%.

Methods

To assess the change in vegetation and ground cover following pinyon and juniper reductions, I used data collected by the Bureau of Land Management (BLM) on BLM & Utah Watershed Restoration Imitative (UWRI) projects. UWRI is a partnership based program in Utah to improve high priority watersheds throughout the state. Specifically, they look at improving watersheds through reducing invasive and over abundant plant species, limiting destructive wildfire, restoring degraded streams and riparian areas, and reversing aspen forest decline (UWRI 2019).

Treatment types and a seeded species list were compiled from project site records maintained by the BLM. The two treatment types that were included in this analysis were hand thinning and mastication of pinyon and juniper. The Pine Point pasture treatment was hand thinned and aerial seeded (Figure 3.1), with 1024 hecatres treated. The Ford pasture was hand thinned and masticated (Figure 3.2), and seed was broadcasted. The hand thinned area of the project consisted of 151 hectares. The mastication area consisted of 158 hectares. Both treatment areas were seeded with a customized mix of species suitable to the area.

Canopy cover of grasses, forbs, shrubs, bare ground, and litter were monitored using the standard protocol used by the BLM. Each site was sampled randomly inside the treatment area by establishing one 50 meter transect. Vegetation was monitored along the transect using 20 nested frequency quadrats spaced 2.5 meters apart. Canopy cover was estimated inside the quadrat using the ocular method (Elzinga et al. 1998). For grasses, forbs, shrubs, bare ground, and litter, I calculated average cover and a standard deviation over the 20 quadrats. The Ford pasture treatment cover readings consisted of 1 pre-

treatment reading and 5 post-treatment readings on 5 transect locations. The Pine Point treatment cover readings consisted of 1 pre-treatment reading, and 4 post treatment readings on 4 transect locations.

Data Analysis

Given the limitations of our study design, (i.e. variable seeding and treatment years, monitoring years, different seeding mixes, lack of repetition), I chose to calculate a standardized metric of effect size to quantify changes in species cover and analyze this data using meta-analysis. I chose meta-analysis, as it is appropriate for situations when results across multisite, longer-term experiments are used to assess and synthesize outcomes of different management strategies (Koricheva and Gurevitch 2014). To do this, I calculated mean and standard deviation from each transect and time frame for perennial grass, native annual grass, invasive annual grass, perennial forb, annual forb, shrub, bare ground, and litter. These were used to calculate an effect size as the natural log of the ratio between post- and pre-treatment (ln[post/pre] = lnRR) that were weighted by the inverse of study site variance (Hedges and Vevea 1998, Gurevitch and Hedges 1999). Despite the differences in sample sizes and the limitations outlined above, meta-analysis allowed comparisons between the two treatments.

Effect size estimates were computed and analyzed with the metafor package (Viechtbauer 2010) for R (www.r-project.org) using the RStudio console (www.rstudio.com) (RStudio Team 2019). I used a multi-level model with random effects for transect site and year. Effect size estimates were graphed with 95% confidence intervals (CI) to visually compare effects, which were considered significantly different

from zero if 95% Cis did not overlap zero (Nakagawa and Cuthill 2007). Actual pre- and post-treatment cover data (i.e. mean ± SE) were summarized by vegetation class for each timeframe.

Results

Pine Point Treatment – Actual Vegetation Cover Values

On the pine point pasture treatment area, perennial grass cover held constant in the first two years following treatment, but gradually increased in post-treatment years three and four. Native annual grass (*Festuca octoflora*) varied year to year. Invasive annual grass (*Bromus tectorum*) was recorded in low amounts in the pre-treatment reading, and the following three years post-treatment. However, four years post-treatment, there was an increase in invasive annual grass. Perennial forb increased one-year post treatment, then decreased in subsequent years, but was always above the pre-treatment observations. Annual forb cover did not change one-year post treatment, but steadily increased in years two to four years post-treatment. Shrub cover did not significantly change between pre- and post-treatment readings. Bare ground cover decreased following treatment, and litter cover increased following treatment. (Figure 3.3)

Ford Pasture Treatment – Actual Vegetation Cover Values

In the Ford pasture, perennial grass cover remained constant in the first two years following treatment, afterward it increased significantly in the third year post treatment. Native annual grass (*Festuca octoflora*) was found in low quantities in the pre-treatment

readings but was absent in the five years post-treatment observations. Invasive annual grass slightly increased two years following treatment, and significantly increased three to five years post-treatment. Perennial forb decreased the first year post treatment, but increased significantly in years two to five post-treatment. Similarly, annual forb decreased the first year post-treatment, but increased in years two to five years post treatment. Shrub cover declined one-year post treatment, but increased to pre-treatment readings in the subsequent years. Bare ground decreased in years following treatment, and litter increased. (Figure 3.4)

Treatment Effects

When calculating effect size on vegetation cover one to two years post treatment, effect sizes were found to be insignificant for most variables, except for native annual grass, bare ground and litter (Figure 3.5, Appendix B1). However, when effect sizes were calculated three to five years post treatment, effect sizes were found to be significant for more variables (Figure 3.5).

When calculating effect sizes between three to five years post treatment (pooled between treatment), it was found that the treatments had a positive effect on perennial grass ($P = <0.0001$), perennial forb ($P = 0.0313$) and litter ($P = <0.0001$). Pinyon and juniper treatments also had a negative effect on native annual grass ($P = 0.0281$), and bare ground ($P = 0.0015$). It was found that invasive annual grass, annual forb, and shrub were not affected three to five years post-treatment (Figure 3.5, Appendix B2).

When calculating effect sizes on the Ford Pasture restoration site three to five years post treatment, I found the treatment had a positive effect on perennial grass (P =

0.0004), and litter (P = .0002). Treatment reduced bare ground (P = 0.0154) as well. Native annual grass, invasive annual grass, perennial forb, annual forb, and shrub found to not be significantly affected by the treatment (Figure 3.6, Appendix B3).

When calculating effect sizes for the Pine Point restoration site three to five years post treatment, it was found the treatment had a positive effect on perennial grass (P = 0.0002), perennial forb (P = 0.0342), annual forb (P = 0.0059), and litter cover (P = 0.0088). Treatment had a negative effect on bare ground (P = 0.0137) as well. Native annual grass, invasive annual grass, and shrub cover were found to not to be significantly affected by the treatment (Figure 3.6, Appendix B4).

Discussion

By calculating effect size with data from three years post treatment, the effect size for perennial grass was significant, but not when calculating effect size one to two years post treatment. This is likely due to a delay in response of seeded species. Tausch and Tueller (1977) found that maximum perennial bunch grass cover was achieved three to four years post treatment. Cover values and the significant effect size for perennial grass three to five years post treatment aligns with this theory. Actual cover was highest in year three on the Ford Pasture treatment, and in year four of the Pine Point treatment. The increasing trend of perennial grass following pinyon and juniper reduction may indicate that the effect size could still be increasing long term (five to 10 years post treatment), similar to what has been found in previous research (Redmond et al. 2013, Dulfon 2016).

The effect size calculated for perennial forbs was significant in the Pine Point treatment, but not in the Ford Pasture treatment. However, when pooling the two

treatments together, perennial forb cover was significantly affected three to five years post-treatment, but it was not significant one to two years post treatment. This differs from previous research, where Tausch and Tueller (1977) found that maximum forb cover was achieved two years post treatment. The delay of forb cover in this study may also be a result of a delayed response of seedling germination and growth. There seems to be a decreasing trend in forb cover, which may be concerning, because the effect on perennial forbs cover may not be as significant in the long term (five to ten years post treatment).

When calculating effect sizes for each treatment, native annual grass (*F. octoflora*) was found to not be significantly affected by the treatment. However, it was found that native annual grass was negatively affected by the treatment when pooling the Ford and Pine Point pastures. This was most likely due to a lack of data at each transect site. Native annual grass may have decreased due to the increase in competition by other vegetation types, or an increase in litter cover, which may decrease the area it used to occupy before the treatment.

Treatment had a significant effect on annual forb cover on the Pine Point treatment, but not on the Ford Pasture treatment. When pooling treatment together, the effect of treatment was not significant three to five years post treatment for annual for cover. Actual annual forb cover showed an increasing trend, so it may be possible that treatment did have a positive effect on annual forb, but due to the limited data collected from these studies, the effect size was not significant when pooled between the two treatments.

Invasive annual grass and shrub did not result in a significant effect size when using meta-analysis. Cover of invasive annual grass (*B. tectorum*) seems to have increased when looking at the change in cover over time, however, the effect size of the treatment was not significant. Other studies have shown a positive effect on invasive annual grasses from pinyon and juniper treatment (Baughman et al. 2010, Ross et al. 2012). It is possible that treatment did have a positive effect on invasive annual grass, but due to the lack of data and the wide fluctuations found across sites, I did not see a significant effect size. Shrub cover was not found to be affected by the treatment, however, shrub cover response in the long term is unknown. Tausch and Tueller (1977) found that large shrubs may not reach peak production until 5 or more years post treatment. Therefore, it is possible that treatment would have a significant effect on shrubs $\geq$ 6 years following treatment.

It was found that bare ground was negatively affected and litter was positively affected following treatment. The majority of the litter increase was most likely due to the dead pinyon and juniper that were left on the treatment areas following hand thinning and mastication. The decrease in bare ground can be primarily attributed to the increase in litter, as well as an increase in perennial forb and perennial grass cover.

Due to the limitations of this study (lack of replication in data collection), not all of the cover variables that maybe significant were found to be significant when computing effect size. It may be possible that non-significant variables would become significant if there was more repetition in this study. Specifically, we do not know the long-term effects (five to ten years post-treatment) in this study, which would change the results.

Implications

The goals of these projects were to remove 100% of the encroaching pinyon and juniper, re-introduce perennial grasses, forbs, and shrubs, and increase the percent canopy cover of shrubs to 30%, forbs to 5%, and grasses to 30%. In the first year following treatment, there were no pinyon and juniper recorded on the treatment areas. However, juniper returned on two transects 2 years post treatment. With the goal of removing 100% of the pinyon and juniper, these areas may need to be treated again to prevent pinyon and juniper encroachment.

The goal of re-introducing perennial grasses, forbs, and shrubs was most likely met in the case of perennial grasses and forbs. Perennial grasses and forbs significantly increased post-treatment, however, shrub cover was not found to be significantly affected by the treatments.

The goal of increasing the percent canopy cover of shrubs to 30%, forbs to 5%, and grasses to 30% was met in the case of forbs on the Ford pasture, but not the Pine Point pasture. Shrub cover on both sites never exceeded 12%, and perennial grass cover never exceeded 20%. Therefore, the goals of this project were not met, at least not in the short term. It is not known with this limited data if the treatment areas would continue to trend up, leading to the project goals stated above.

It may also be possible that the areas being treated are not capable of meeting the criteria set. The ecological site description of the treatment area states that shrub cover should fall between 2-15%, grass cover 20-30%, and forb cover to be 3-8% (R035XY307UT). From this criteria, the treatment was successful for shrub cover on both treatment areas, successful for grass cover on the Ford Pasture treatment area and is

trending towards being successful for grass cover on the Pine Point treatment area. Other portions of the treatment area fall under ecological site descriptions classified as pinyon and juniper woodlands (R035XY314UT)), and these areas may also not be capable of reaching the successful criteria set. In the future, it may be important to continue to monitor treatment areas to see if project goals are being met. It is possible that the effect size on herbaceous plant cover may be different in the long term (≥ 6 years post treatment).

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Tables and Figures

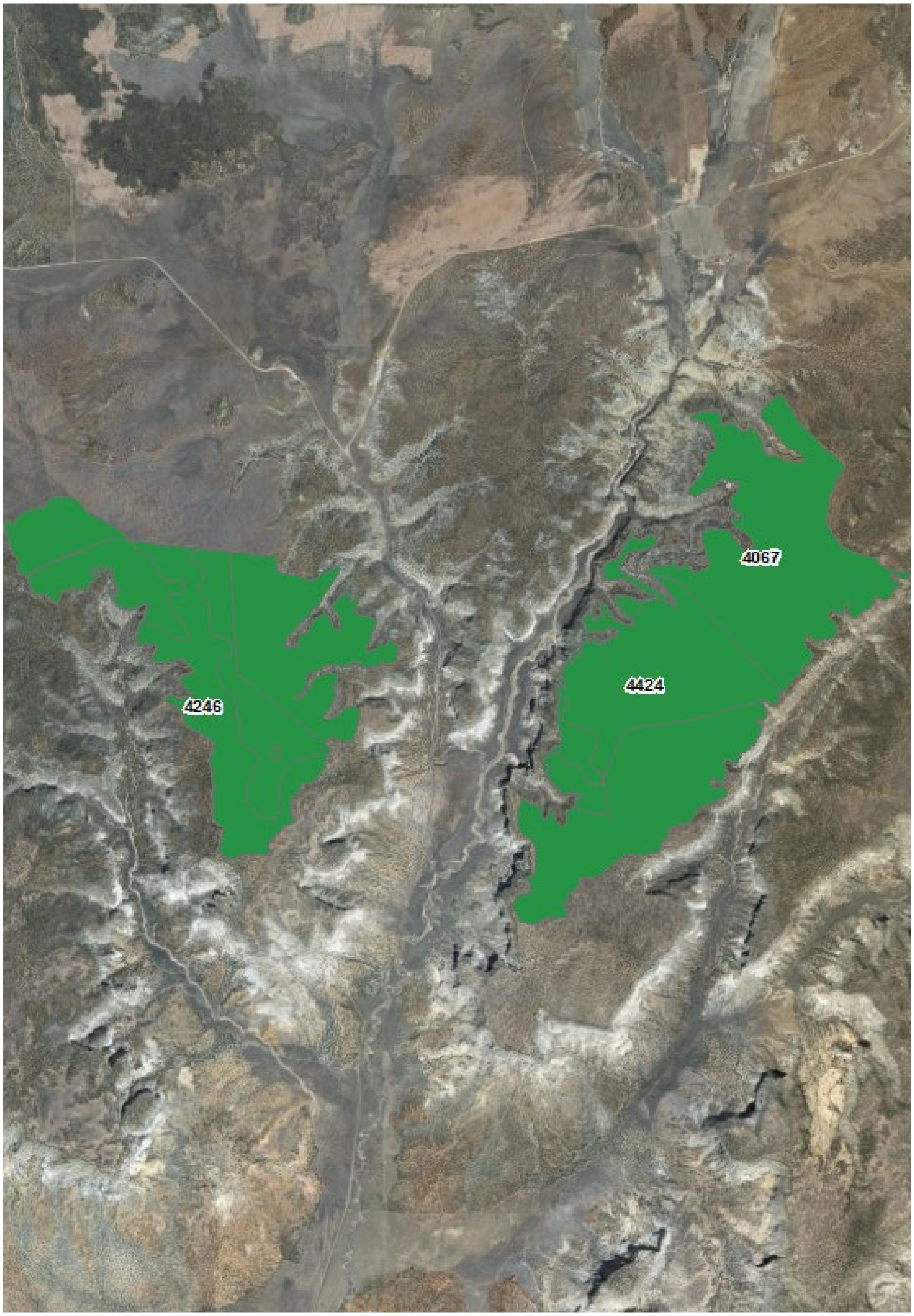


Figure 3.1 – Pine Point treatment area map.

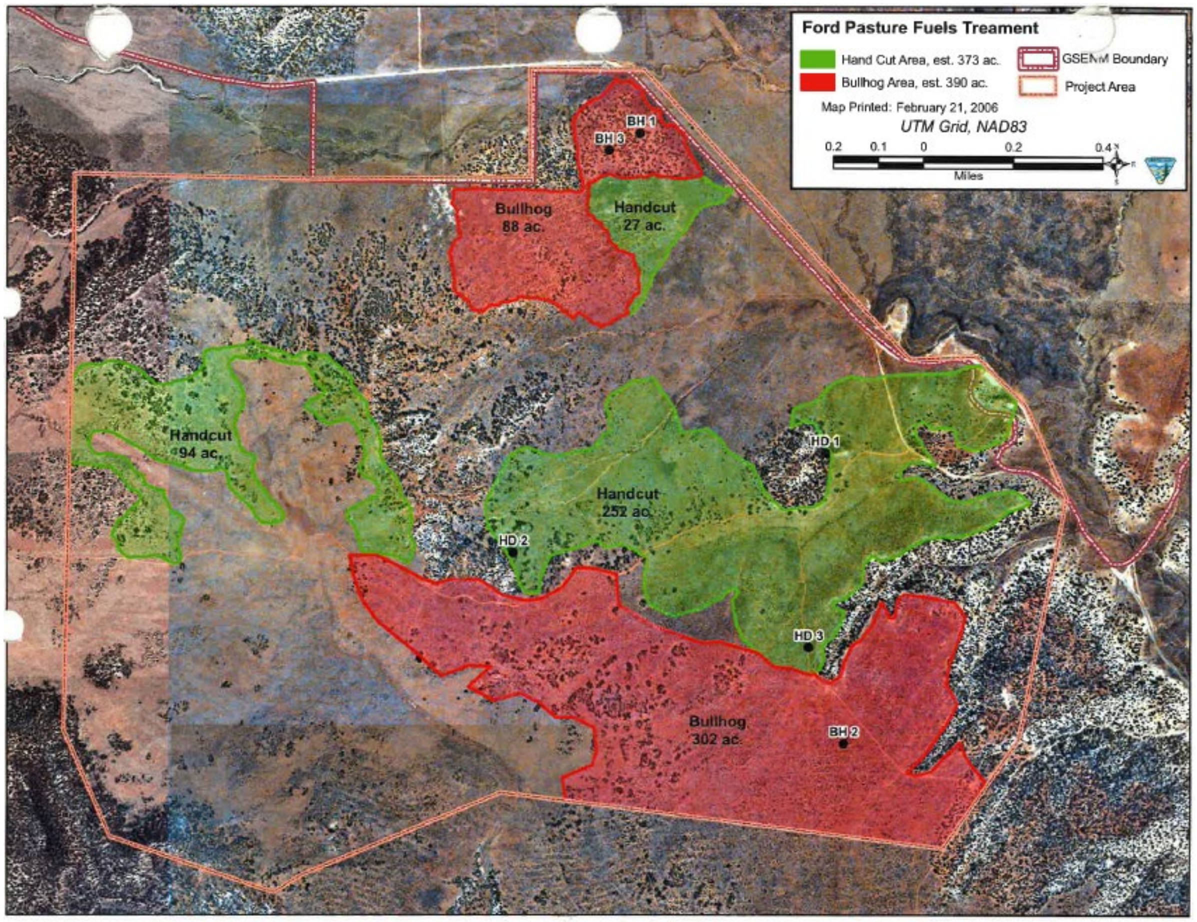


Figure 3.2 – Ford Pasture Treatment areas with locations of transects.

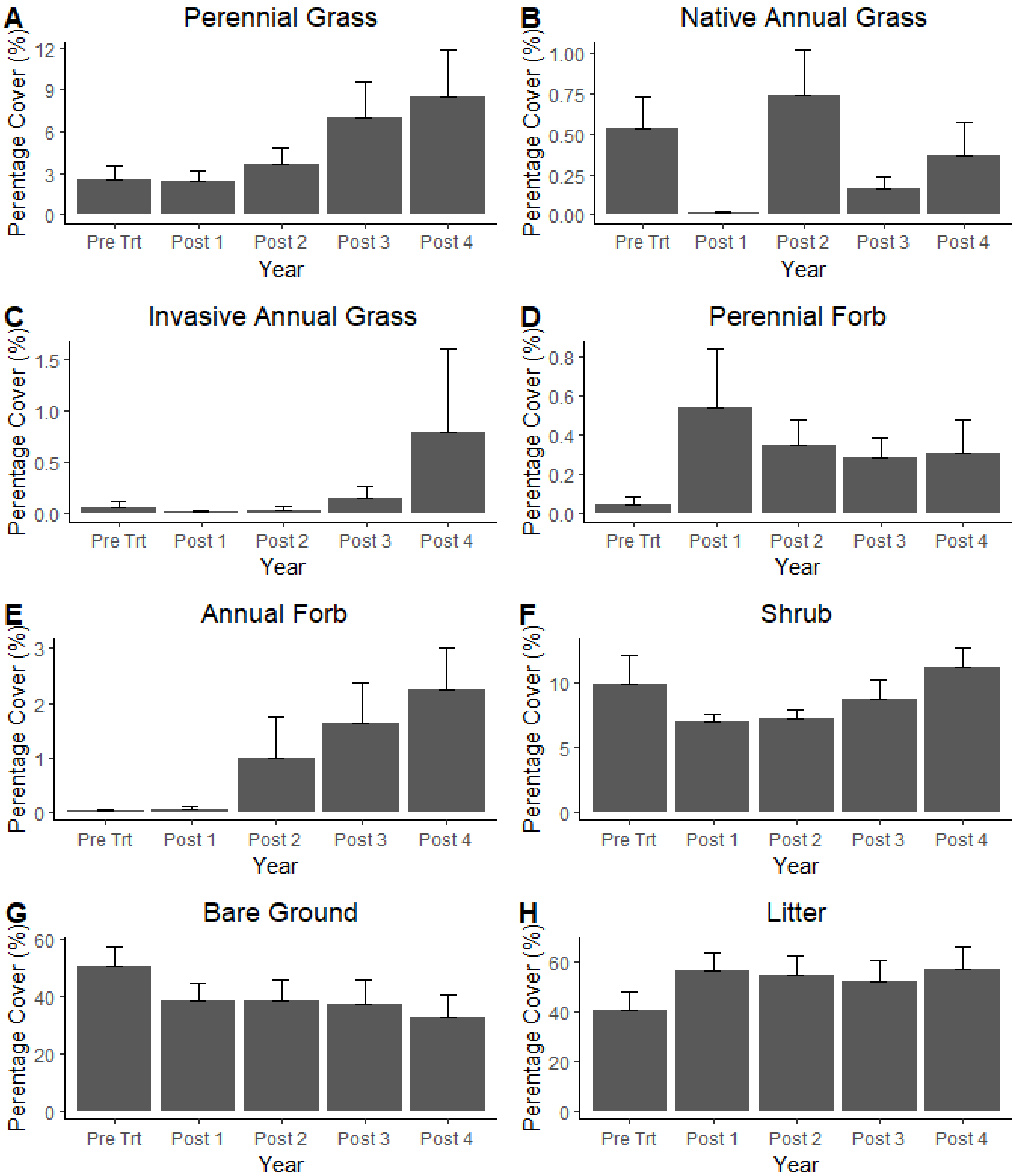


Figure 3.3 – Mean (+SE) cover values of vegetation classes measured 1-year pre-treatment and 4-years post-treatment following seeding and pinyon and juniper removal by hand thinning on the Pine Point restoration site. (A: Perennial Grass Cover, B: Native Annual Grass, C: Invasive Annual Grass, D: Perennial Forb, E: Annual Forb, F: Shrub, G: Bare Ground, H: Litter.)

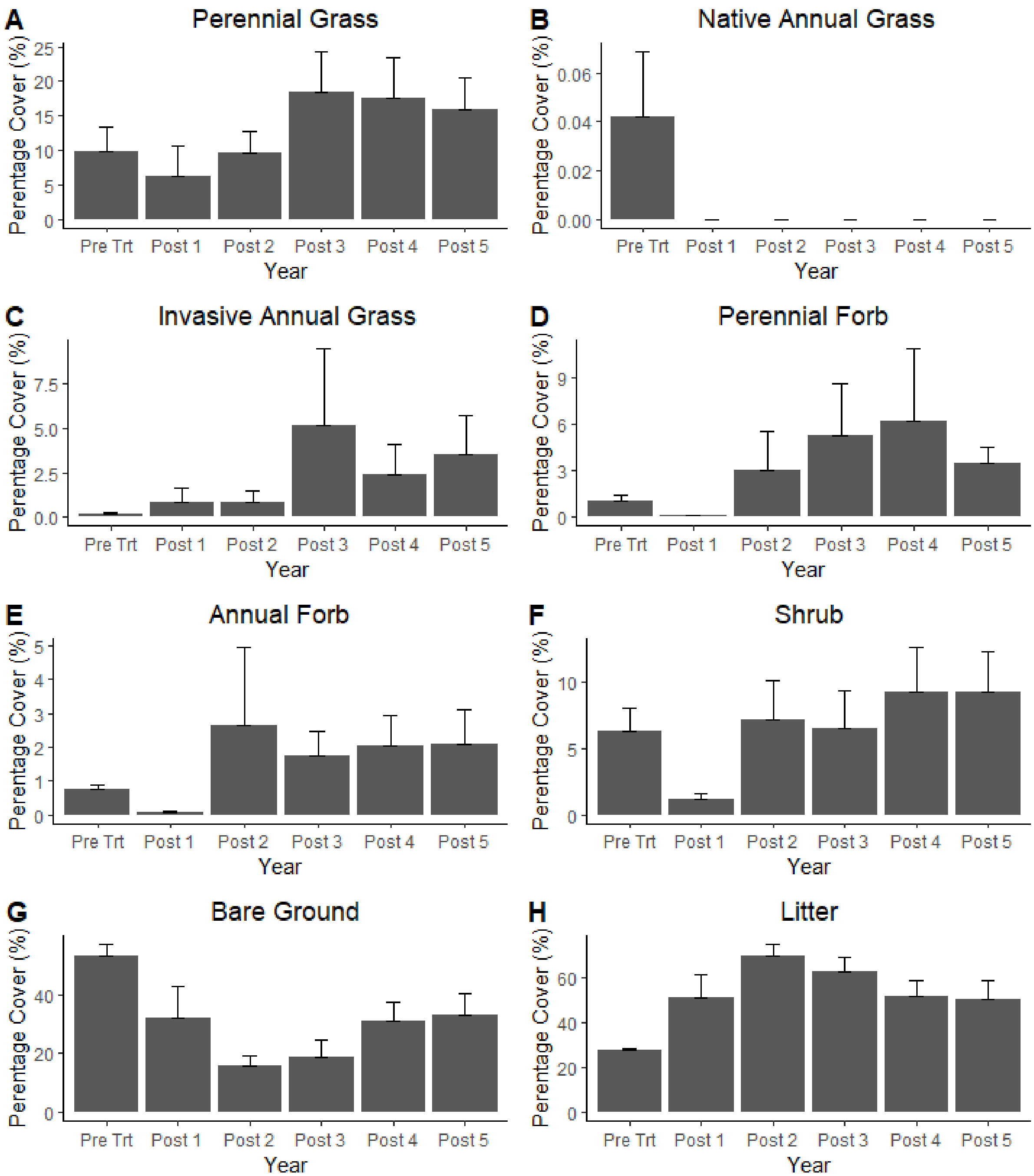


Figure 3.4 – Mean (+SE) cover values of vegetation classes measured 1-year pre-treatment and 5 years post-treatment following seeding and pinyon and juniper removal by hand thinning and mastication on the Ford Pasture restoration site. (A: Perennial Grass Cover, B: Native Annual Grass, C: Invasive Annual Grass, D: Perennial Forb, E: Annual Forb, F: Shrub, G: Bare Ground, H: Litter.)

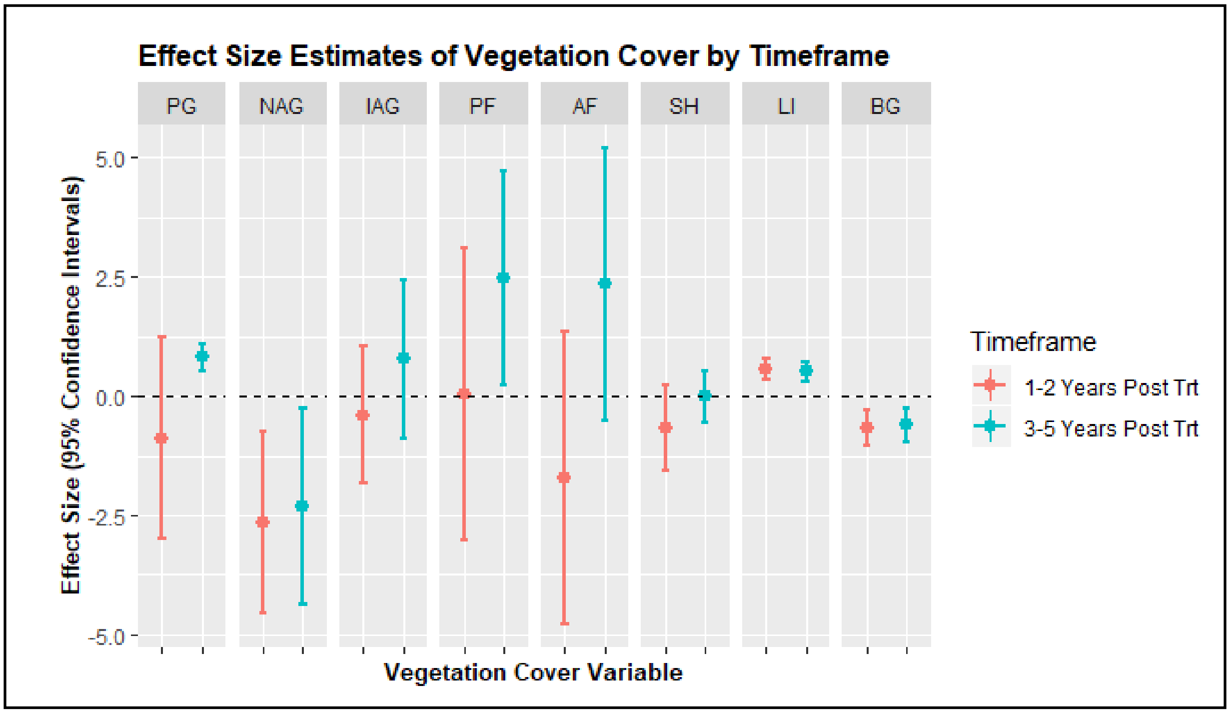


Figure 3.5 – Mean (± 95% CI) effect size estimates for perennial grass (PG), native annual grass (NAG), invasive annual grass (IAG), perennial forb (PF), annual forb (AF), shrub (SH), litter (LI), and bare ground (BG) 1-2 years post treatment and 3-5 years post treatment (pooled between the Ford Pasture treatment and Pine Point treatment).

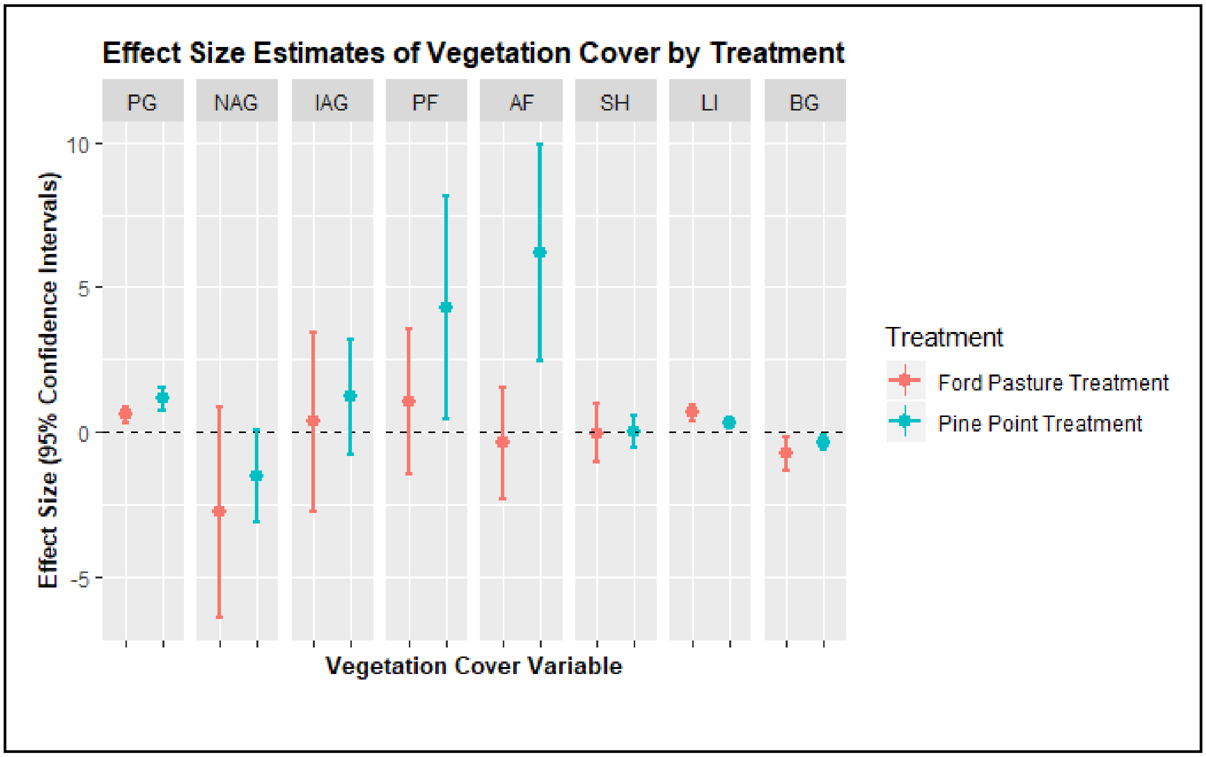


Figure 3.6 – Mean ($\pm$ 95% CI) effect size estimates for perennial grass (PG), native annual grass (NAG), invasive annual grass (IAG), perennial forb (PF), annual forb (AF), shrub (SH), litter (LI), and bare ground (BG) split by treatment, (pooled 3-5 years post treatment).

CHAPTER 4

SUMMARY AND CONCLSUIONS

On the Grand Staircase Escalante National Monument (GSENM), and the rangelands throughout Southern Utah, livestock grazing and vegetation treatments have been used as part of an adaptive resource management approach to sustain and increase the forage resources. Forage resources, rangeland and watershed health, livestock, wildlife, and humans benefit from a holistic approach that monitors and evaluates the effectiveness of vegetation treatments and livestock grazing.

I evaluated 2 vegetation treatments, and the harvest and grazing efficiency of forage by livestock on the GSENM.

The results from the harvest and grazing efficiency study were mixed. This was due to the fact that harvest and grazing efficiency is influenced by forage type, forage maturity, forage distribution, topography, livestock distribution, and stocking densities. Because of the heterogeneous nature of the GSENM, harvest and grazing efficiency differs across the landscape. Therefore, using harvest and grazing efficiency as a tool to refine stocking rates may not be appropriate. However, it was found that cattle distribution is not equal across the landscape, and by modeling cattle distribution using a resource selection function, land managers can better understand the efficiency of cattle grazing an allotment. Utilizing resource selection functions or other tools to estimate distribution may be a more appropriate tool for refining stocking rates. More research is needed to understand how stocking rates can be refined by predicting cattle distribution.

Based on my results from the vegetation treatments, short term vegetation response (1-2 years post-treatment) to pinyon and juniper removal and reseeding was rarely significant. However, by 4 and 5 years post-treatment, vegetation responses and effect sizes increased. Pinyon and juniper removal can positively affect vegetation cover, especially 4-5 years post treatment. Bare ground decreases when pinyon and juniper were removed, which would result in less erosion.

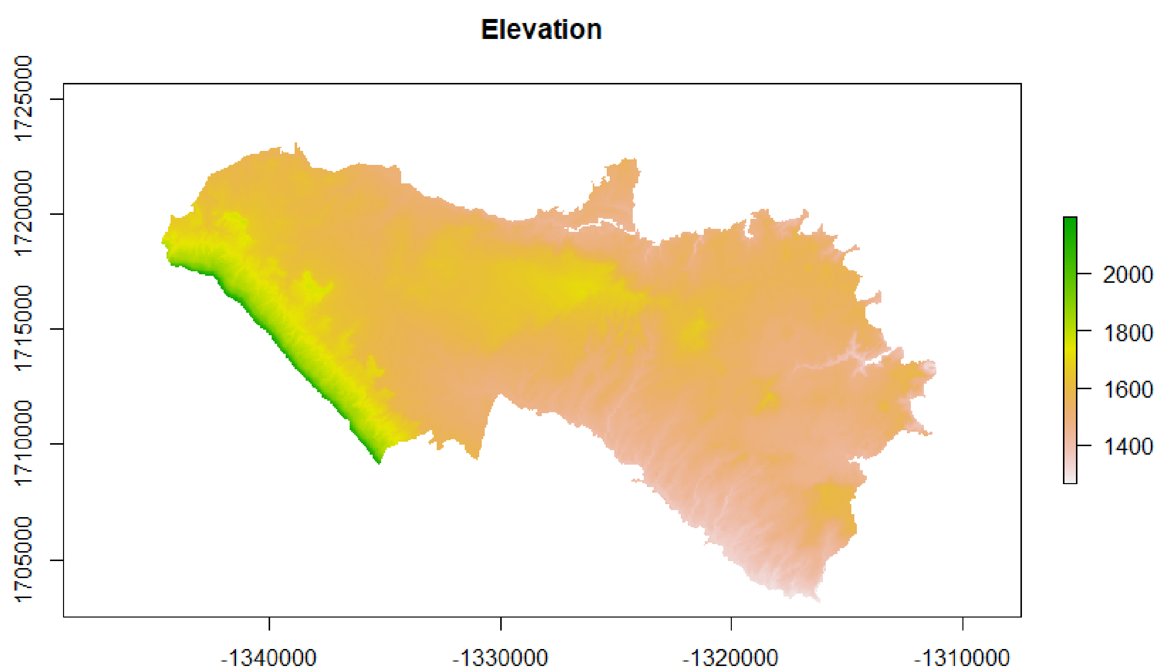
By understanding the effect size of pinyon and juniper treatments and stocking rates, land managers and livestock producers can make informed rangeland management decisions that increase and sustain the rangeland resources.

APPENDICES

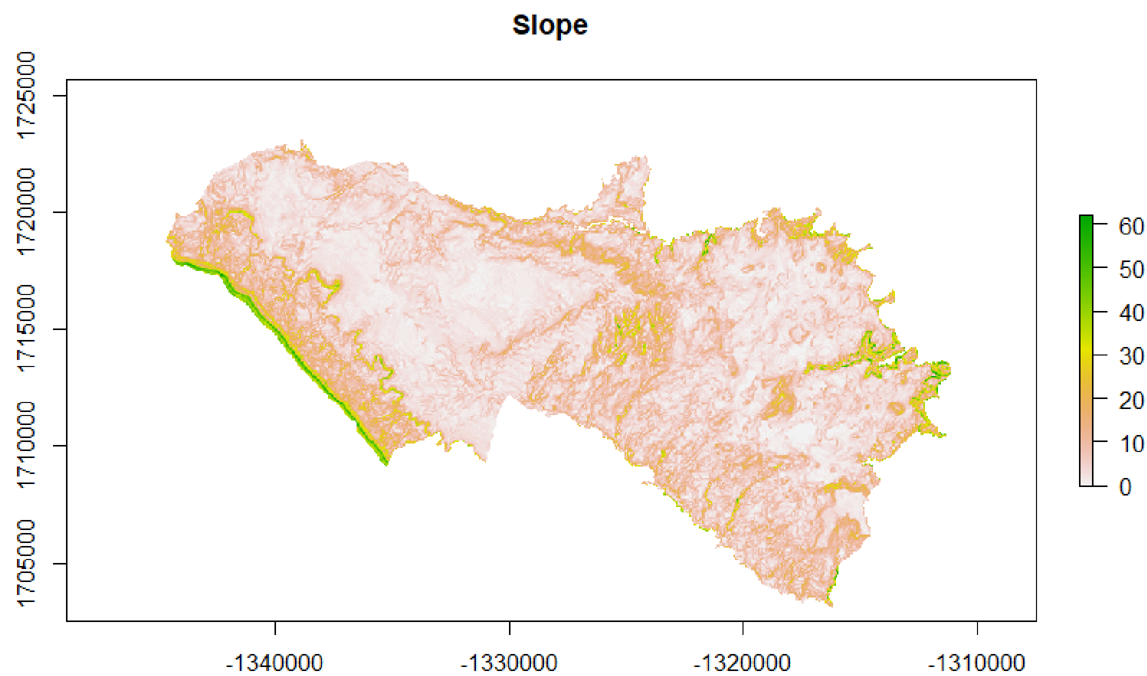
APPENDIX A

Appendix A9 – Estimates, standard errors, z values, and probability of the model (Used ~ Slope, Elevation, Aspect, Distance to Water, Vegetation type). Shrub was used as the intercept for vegetation type.

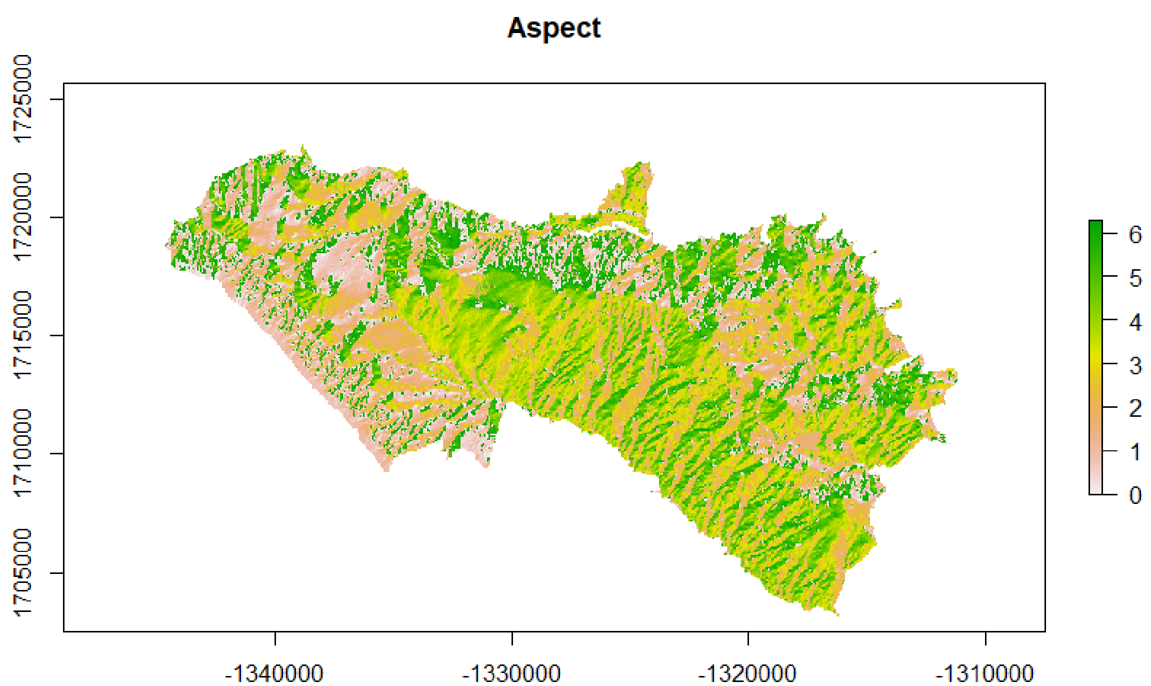
	Estimate	Std. Error	Z Value	Pr (> |Z|)
Intercept	-2.812099	0.011524	-244.029	< 2e-16
Slope	-0.664268	0.014943	-44.453	< 2e-16
Distance to Water	-0.503967	0.007996	-63.030	< 2e-16
Elevation	-0.105505	0.009522	-11.080	< 2e-16
Aspect	-0.140047	0.006530	-21.446	< 2e-16
Veg Grass	0.487686	0.041865	11.649	< 2e-16
Veg None	-0.269774	0.090294	-2.988	0.00281
Veg Other	-0.175598	0.019335	-9.082	< 2e-16
Veg Tree	-0.372725	0.033211	-11.223	< 2e-16



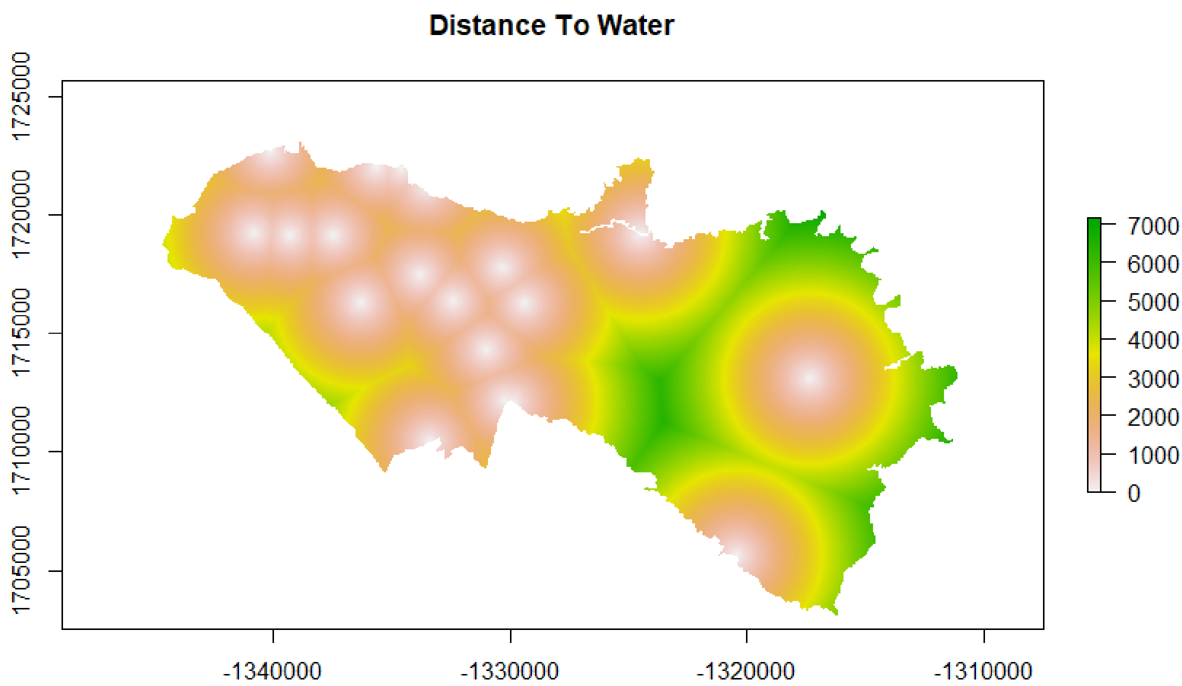
Appendix A1 – Lower Cattle allotment elevation layer used to create the Resource Selection Function.



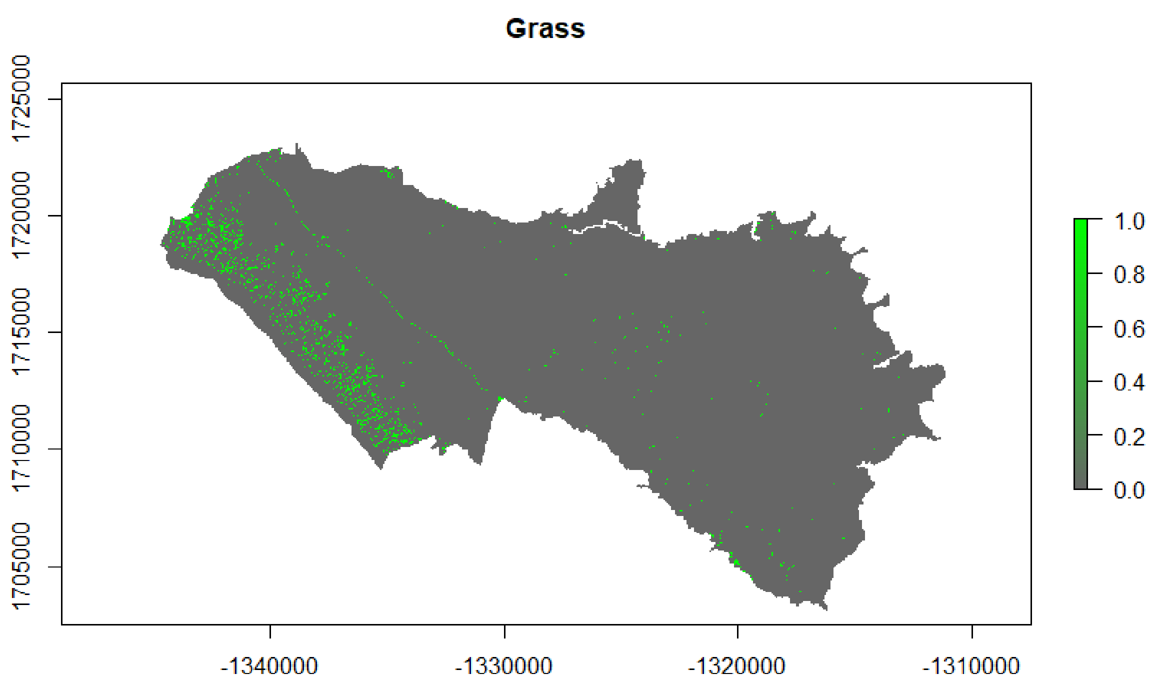
Appendix A2 – Lower Cattle allotment slope layer used to create the Resource Selection Function.



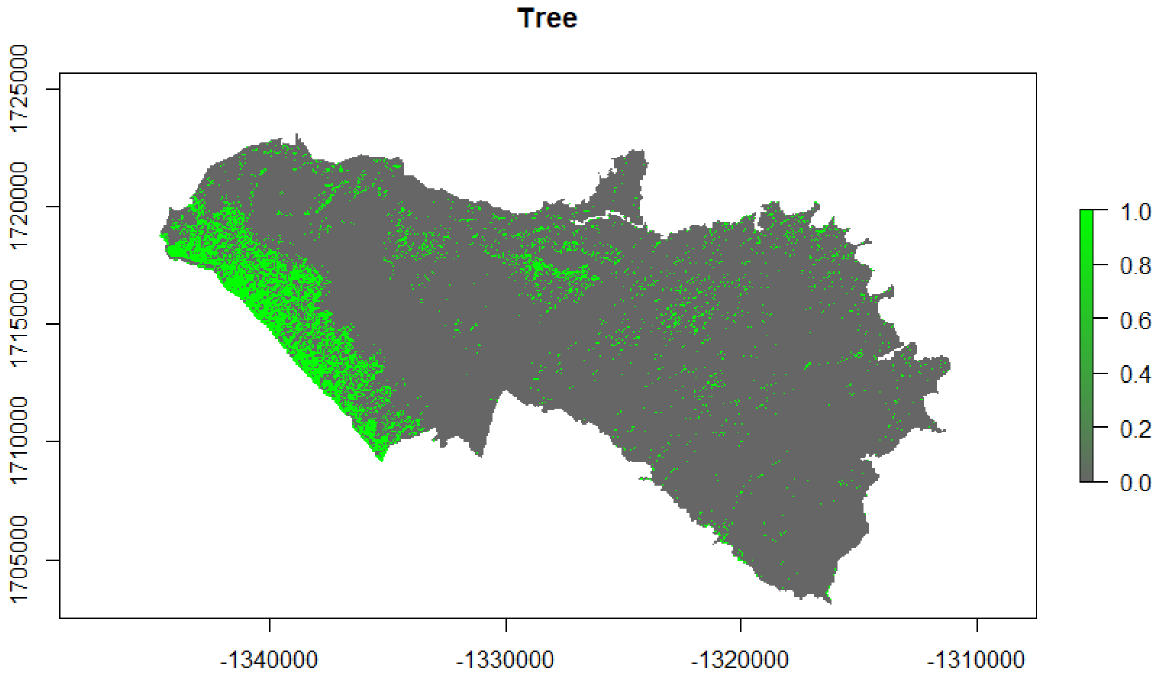
Appendix A3 – Lower Cattle allotment aspect layer used to create the Resource Selection Function.



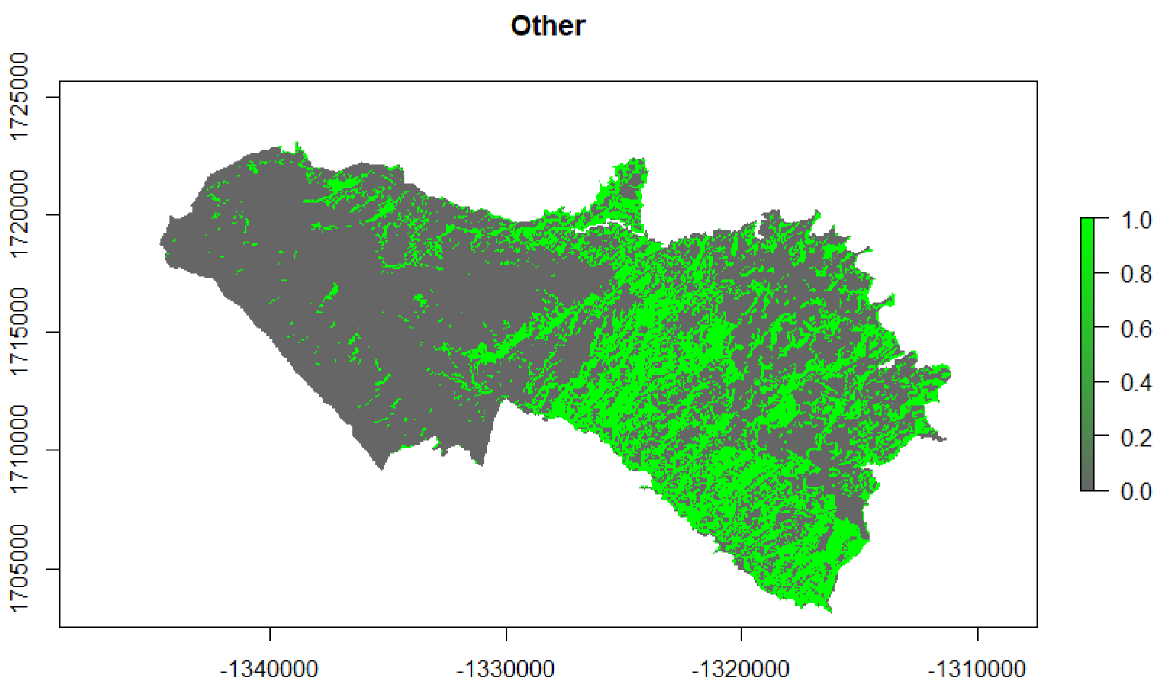
Appendix A4 – Lower Cattle allotment distance to water layer used to create the Resource Selection Function.



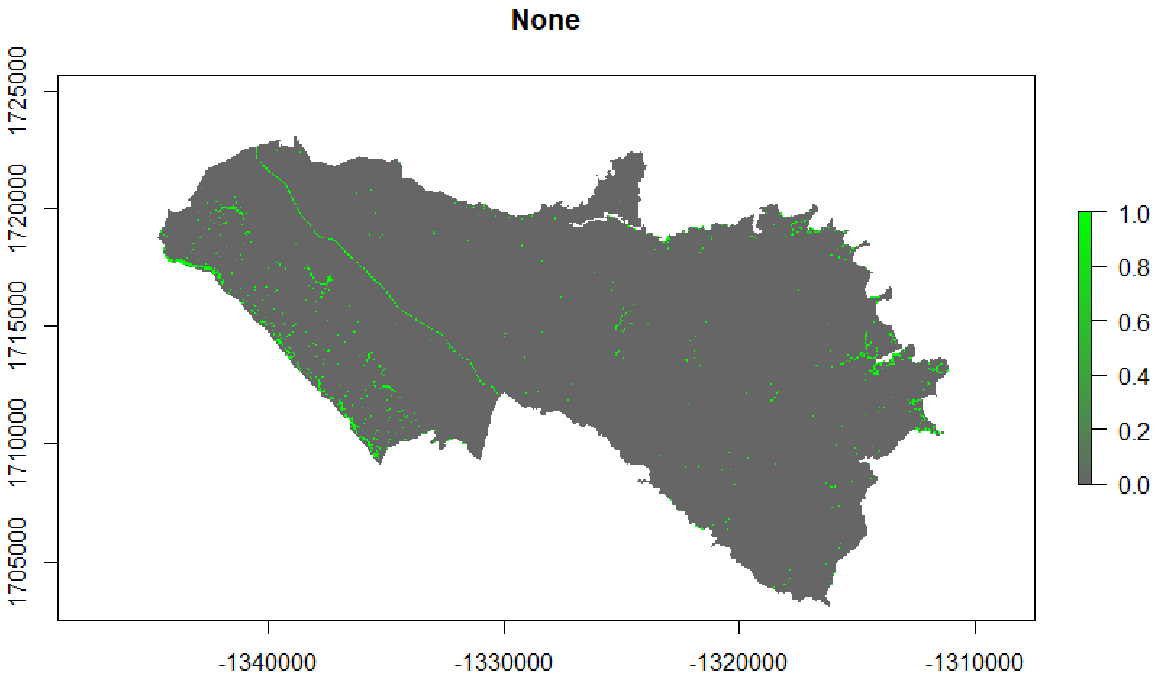
Appendix A5 – Lower Cattle allotment grass layer used to create the Resource Selection Function.



Appendix A6 – Lower Cattle allotment tree layer used to create the Resource Selection Function..



Appendix A7 – Lower Cattle allotment other vegetation layer used to create the Resource Selection Function. Red rock with no vegetation is classified as other in this study.



Appendix A8 – Lower Cattle allotment other no vegetation layer used to create the Resource Selection Function.

APPENDIX B

Appendix B1 – Meta-analysis test of moderators (Q_m) for perennial grass, native annual grass, invasive annual grass, perennial forb, annual forb, shrub, litter, and bare ground 1-2 years post treatment (pooled between the Ford Pasture treatment and Pine Point treatment). Significant effect sizes labeled with **.

	Estimate	SE	p-value	CI LB	CI UB
Perennial Grass	-0.8772	0.9970	0.3920	-2.9907	1.2364
Native Annual Grass**	-2.6682	0.8993	0.0091	-4.5745	-0.7619
Invasive Annual Grass	-0.3987	0.6774	0.5644	-1.8347	1.0373
Perennial Forb	0.0502	1.4490	0.9728	3.0214	3.1219
Annual Forb	-1.7128	1.4396	0.2515	-4.7646	1.3390
Shrub	-0.6741	0.4261	0.1332	-1.5774	0.2292
Litter**	0.5732	0.1063	<0.0001	0.3479	0.7985
Bare Ground**	-0.6612	0.1736	0.0015	-1.0291	-0.2933

Appendix B2 – Meta-analysis test of moderators (Q_m) for perennial grass, native annual grass, invasive annual grass, perennial forb, annual forb, shrub, litter, and bare ground 3-5 years post treatment (pooled between the Ford Pasture treatment and Pine Point treatment). Significant effect sizes labeled with **.

	Estimate	SE	p-value	CI LB	CI UB
Perennial Grass **	0.810	0.1349	<0.0001	0.5303	1.0897
Native Annual Grass **	-2.3122	0.9832	0.0281	-4.3512	-0.2731
Invasive Annual Grass	0.7732	0.7966	0.3423	-0.8789	2.4253
Perennial Forb**	2.4795	1.0782	0.0313	0.2435	4.7154
Annual Forb	2.3465	1.3733	0.1016	-0.5016	5.1946
Shrub	-0.0104	0.2643	0.9690	-0.5585	0.5377
Litter**	0.5096	0.1017	<0.0001	0.2987	0.7206
Bare Ground**	-0.6068	0.1679	0.0015	-0.9551	-0.2586

Appendix B3 – Meta-analysis test of moderators (Q_m) for perennial grass, native annual grass, invasive annual grass, perennial forb, annual forb, shrub, litter, and bare ground 3-5 years post treatment on the Ford Pasture treatment area. Significant effect sizes labeled with **.

	Estimate	SE	p-value	CI LB	CI UB
Perennial Grass**	0.5891	0.1266	0.0004	0.3175	0.8607
Native Annual Grass	-2.7652	1.6975	0.1256	-6.4060	0.8757
Invasive Annual Grass	0.3384	1.4547	0.8194	-2.7817	3.4585
Perennial Forb	1.0554	1.1756	0.3845	-1.4661	3.5769
Annual Forb	-0.3877	0.8967	0.6720	-2.3109	1.5355
Shrub	-0.0413	0.4790	0.9325	-1.0687	0.9861
Litter**	0.6543	0.1343	0.0002	0.3663	0.9422
Bare Ground**	-0.7615	0.2759	0.0154	-1.3533	-0.1696

Appendix B4 – Meta-analysis test of moderators (Q_m) for perennial grass, native annual grass, invasive annual grass, perennial forb, annual forb, shrub, litter, and bare ground 3-5 years post treatment on the Pine Point treatment area. Significant effect sizes labeled with **.

	Estimate	SE	p-value	CI LB	CI UB
Perennial Grass **	1.1410	0.1656	0.0002	0.7494	1.5327
Native Annual Grass	-1.5296	0.6625	0.0543	-3.0961	0.0368
Invasive Annual Grass	1.2015	0.8391	0.1953	-0.7827	3.1856
Perennial Forb**	4.2986	1.6379	0.0342	0.4255	8.1718
Annual Forb**	6.1701	1.5839	0.0059	2.4247	9.9155
Shrub	-0.0002	0.2431	0.9993	-0.5750	0.5745
Litter**	0.2958	0.0823	0.0088	0.1013	0.4904
Bare Ground**	-0.3609	0.1104	0.0137	-0.6220	-0.0999