

University of Nevada, Reno

**Response of Pinyon-juniper woodlands to fire, chaining, and hand thinning**

A thesis submitted in partial fulfillment of the requirements for the degree of Master of  
Science in Natural Resources and Environmental Science

By

Dominic M. Gentilcore

Sherman Swanson/Thesis Advisor

May 2015

UMI Number: 1591350

All rights reserved

INFORMATION TO ALL USERS

The quality of this reproduction is dependent upon the quality of the copy submitted.

In the unlikely event that the author did not send a complete manuscript and there are missing pages, these will be noted. Also, if material had to be removed, a note will indicate the deletion.



UMI 1591350

Published by ProQuest LLC (2015). Copyright in the Dissertation held by the Author.

Microform Edition © ProQuest LLC.

All rights reserved. This work is protected against unauthorized copying under Title 17, United States Code



ProQuest LLC.  
789 East Eisenhower Parkway  
P.O. Box 1346  
Ann Arbor, MI 48106 - 1346

Copyright Dominic M. Gentilcore 2015

All Rights Reserved



THE GRADUATE SCHOOL

We recommend that the thesis  
prepared under our supervision by

**DOMINIC M. GENTILCORE**

Entitled

**Response Of Pinyon-Juniper Woodlands To Fire, Chaining, And Hand Thinning**

be accepted in partial fulfillment of the  
requirements for the degree of

MASTER OF SCIENCE

Sherman Swanson, Ph.D., Advisor

Paul Verburg, Ph.D., Committee Member

Barry Perryman, Ph.D., Graduate School Representative

David W. Zeh, Ph.D., Dean, Graduate School

May, 2015

## Abstract

Pinyon-juniper (*Pinus monophylla* – *Juniperus osteosperma*) woodlands have expanded and infilled over the last 150 years to cover more than 40 million ha in the Great Basin. Many land managers seek to remove Pinyon-juniper trees using a variety of treatments. This thesis looks at six different Pinyon-juniper removal projects in Central and Eastern Nevada. We established a total of 73 vegetation and soil monitoring plots (38 treated, 35 adjacent untreated) across six Pinyon-juniper removal projects in Central and Eastern Nevada to look at the effects of fire, hand thinning, and chaining. The four burns examined together in Chapter 1 had similar elevation, precipitation, and pre-treatment vegetation communities in the untreated areas, but the treated areas had significantly different responses to treatment. With nonmetric multidimensional scaling (NMS), we found a useful 3-axis ordination of the plots (stress=7.1,  $R^2=.966$ ). Within ordination space, the treated plots were well grouped by parent material. These results informed a Poisson generalized linear model that found parent material factorized explained 86.5% of the deviance in cheatgrass (*Bromus tectorum*) cover at the treated plots. The projects on soils derived from welded tuff had very little cheatgrass while soils derived from limestone or mixed limestone/volcanics were dominated by cheatgrass. Parent material should be considered an important factor when planning Pinyon-juniper removal treatments. Chapter 2 examined the effects of a hand thinning. The hand thinning significantly reduced tree cover [ $F(1,10) = 7.43$ ,  $p = 0.027$ ] to less than 2%. Perennial grasses on the site are slightly higher in the treated area. There was a significant increase in perennial grass cover from 2013 to 2014 [ $F(1,10) = 14.5$ ,  $p = 0.003$ ]. The hand

thinning did not have significant effects on shrubs, annual grasses, annual forbs, perennial forbs, ground cover, stability, species richness, diversity, infiltration, or gap structure. Because hand thinning does not remove the shrubs or other perennials, site resistance can be maintained. With sufficient understory vegetation to maintain resistance post treatment (as in phase I or early phase II Pinyon-juniper woodlands), nonnative annual grasses are less likely to dominate after treatment. Chapter 3 examined the effects of a chaining. The effects of the 40-year old chaining are still significant even though Pinyon-juniper trees are reinvading and make up >5% of the cover in the treated area. The treated areas still have a much more productive understory than adjacent untreated areas. Perennial grass cover, frequency, and density was 2-5 times greater in the chained area. There were fewer large gaps (>100 cm) in the treated area. However, interspace infiltration times were slower in the treatment ( $t(4)=-2.14$ ,  $p=0.09$ ). Surface and subsurface soil aggregate stability remained significantly lower in the treatment for vegetation-protected and unprotected samples ( $t(4)=3.53$ ,  $p=0.024$ ;  $t(4)=3.10$ ,  $p=0.036$ ). Chainings have long-term benefits for vegetation, but also long term impacts on soils and hydrologic ecosystem processes. When planning Pinyon-juniper removal treatments, land managers should consider the plant community, temperature and precipitation regime, and soils at the potential treatment location to better achieve desired outcomes.

## Acknowledgements

I would like to thank all the members of my graduate committee whose advice and guidance was immensely helpful when completing this thesis. I am especially grateful to my advisor Sherman Swanson for regularly providing input on the project by lending advice and consul. I am indebted to all the great scientists who have contributed to a better understanding of Pinyon-juniper woodlands. I am thankful for the hard work of my field technicians Wailea Johnston and Sylvia Tran who assisted me with data collection. The project could not have been completed in a timely manner without their support and dedication. I am grateful to all members of the Swanson Lab especially Matt Church, Camie Dencker, Sabrina Murano, Sara McBee, John Swanson, and Carly Wagner for their friendship and daily advice in the office while I completed this project. I am indebted to the University of Nevada-Reno and all the staff and faculty who helped me throughout the course of my education. I am especially grateful for the taxpayers of the United States of America who provided the public funding for this project. I would also like to thank Carol Carlock, Cheri Howell, Kathleen Johnson, Jose Noriega, and Joshua Simpson at the United States Forest Service who directly funded this project, proposed the graduate position, helped with initial project selection, and provided past monitoring data.

## Table of Contents

Abstract.....	i
Acknowledgements.....	iii
Table of Contents.....	iv
List of Tables.....	vi
List of Figures.....	vii
Introduction.....	1
Thesis Overview.....	6
Research Methods.....	7
Study Area Description.....	7
Plot Selection.....	10
Project Descriptions.....	15
Cathedral.....	15
Currant Creek.....	16
Elkhorn 1 and Elkhorn 2.....	17
Holt Chaining.....	18
White Pine Hand Thinning.....	19
Experimental Design.....	20
Monitoring Protocol.....	20
Data Management.....	24
<b>Chapter 1: Influence of tuffaceous soils on cheatgrass susceptibility in burned Pinyon-juniper woodlands in Central Nevada.....</b>	<b>25</b>
Abstract.....	25
Introduction.....	26
Methods.....	28
Study Area.....	28
Sampling Design.....	30
Statistical Analysis.....	31
Results.....	32
Discussion.....	42
Management Implications.....	45
Literature Cited.....	47
<b>Chapter 2: The effects of hand thinning in Pinyon Juniper forests.....</b>	<b>52</b>
Abstract.....	52
Introduction.....	52
Methods.....	54
Results.....	58
Discussion.....	66
Management Implications.....	68
Literature Cited.....	70
<b>Chapter 3: Long-term effects of Pinyon-juniper chaining on soil stability.....</b>	<b>74</b>
Abstract.....	74
Introduction.....	74



Methods .....	77
Results .....	80
Discussion .....	86
Management Implications .....	88
Literature Cited .....	89
Conclusion .....	94
Management Implications Synthesis.....	95
When is Pinyon-Juniper Removal Appropriate?.....	98
Where Should Be Priority Treatment Areas?.....	99
What Treatment to Select? .....	101
Monitoring Successfulness.....	102
Literature Cited .....	103

## List of Tables

### Research Methods

<b>Table 1.</b> Macroplot location summary..	11
<b>Table 2.</b> Soils where plots occurred..	14
<b>Table 3.</b> Soil associations with soil series that are repeated in multiple associations where plots actually occurred.	15

### Experimental Design

<b>Table 1.</b> Soil Aggregate Stability Class Standard Characterization Criteria.....	22
--	----

### Chapter 1

<b>Table 1.</b> Summary of burned areas.....	29
<b>Table 2.</b> Resilience to disturbance and resistance to invasive annual grasses score for Cathedral, Currant, Elkhorn 1, and Elkhorn 2. ....	33
<b>Table 3.</b> ANOVA test results for differences in vegetation cover classes among projects. .....	34
<b>Table 4.</b> Proportion of variance explained by ordination axes.....	35
<b>Table 5.</b> Pearson's correlation with ordination axes for lifeform classes..	36
<b>Table 6.</b> Pearson's correlation with ordination axes for environmental and soil variables.. .....	36
<b>Table 7.</b> Best generalized linear model summary..	42

### Chapter 2

<b>Table 1.</b> Summary of Repeated Measures ANOVA results for vegetation cover classes.. .....	59
<b>Table 2.</b> White Pine vegetation summary. ....	59
<b>Table 3.</b> Summary of RM ANOVA results for ground cover..	62
<b>Table 4.</b> Summary of RM ANOVA results for species richness and diversity.....	63
<b>Table 5.</b> Summary of RM ANOVA results for gap. ....	66

### Chapter 3

<b>Table 1.</b> Holt vegetation summary. ....	81
<b>Table 2.</b> Paired t-test for vegetation cover classes between treated and untreated plots..	81
<b>Table 3.</b> Summary of ground cover at Holt.....	82
<b>Table 4.</b> Holt aggregate stability and gap summary.....	83
<b>Table 5.</b> Holt infiltration summary.....	86

# List of Figures

## Introduction

<b>Figure 1.</b> Project location overview map .....	4
--	---

## Research Methods

<b>Figure 1.</b> Annual precipitation from Eureka WRCC weather station 1965-2014.....	9
<b>Figure 2.</b> Annual precipitation from Ely WRCC weather station 1939-2014. ....	9
<b>Figure 3.</b> Macroplot elevation given in m.....	12
<b>Figure 4.</b> Macroplot hill slope given in % slope.....	12
<b>Figure 5.</b> Macroplot precipitation given in mm based on PRISM (2013). ....	13
<b>Figure 6.</b> Count of macroplots by aspect .....	14

## Experimental Design

<b>Figure 1.</b> Overview of a macroplot.....	21
<b>Figure 2.</b> Nested frequency frame design.....	21

## Chapter 1

<b>Figure 1.</b> 2014 Vegetation type cover in untreated areas.....	33
<b>Figure 2.</b> 2014 Vegetation type cover in treated areas.....	34
<b>Figure 3.</b> Ordination axis 1 versus axis 2 grouped by treatment type.....	37
<b>Figure 4.</b> Ordination axis 1 versus axis 2 grouped by project.....	38
<b>Figure 5.</b> Ordination axis 1 versus axis 2 grouped by parent material.....	40
<b>Figure 6.</b> Ordination axis 1 versus axis 3 grouped by project.....	41

## Chapter 2

<b>Figure 1.</b> Cover of vegetation grouped by functional group. Error bars show standard error.....	59
<b>Figure 2.</b> Boxplots for 2013 and 2014 ground cover at the White Pine hand thinning. ..	62
<b>Figure 3.</b> Box plots for 2013 and 2014 average species richness and average species diversity.....	63
<b>Figure 4.</b> Surface and subsurface aggregate stability values by cover class.....	64
<b>Figure 5.</b> Boxplots for 2013 and 2014 gap classes and canopy cover at the White Pine hand thinning. ....	66

## Chapter 3

<b>Figure 1.</b> Surface and subsurface aggregate stability values by cover class.....	83
--	----

## Introduction

Pinyon-juniper woodlands cover 40 million hectares across the Western United States (Romme et al. 2009). Over the last 200 years, Pinyon-juniper woodlands in the Western United States have expanded more than 10 fold from about 3 million ha to their current range (Miller and Tausch 2001). The majority of this expansion has occurred at the lower elevation ranges of the species (Weisberg et al. 2007), but now more infilling is in middle elevation areas (Floyd and Romme 2012). Multiple models have been proposed to explain the expansions of the woodlands focusing on potential causes such as climate change (Miller and Tausch 2001; Miller et al. 2008), differences in grazing practices (Blackburn and Tueller 1970), ecophysiological effects (Nowak et al. 1999), increased fire suppression (Burkhardt and Tisdale 1976), displacement of indigenous people who used to regularly burn some areas (Gruell and Swanson 2012), altered fire regime (Miller and Tausch 2001; Romme et al. 2002), and invasive species weakening the surrounding vegetation communities (Tausch and West 1988). None of these effects alone fully explains the expansion of Pinyon-juniper woodlands on their own (Bradley and Fleishman 2008). Sorting out the exact reason is difficult because several of these major changes started near the same time, shortly after European settlement.

Pinyon-juniper woodlands in the Great Basin continue to infill. In the last 150 years, the Great Basin has physically changed significantly, affecting land use. Changing climate, increased grazing, and increased fire suppression all push areas towards Pinyon-juniper tree dominance. As trees increase on a site and reduce the understory, site

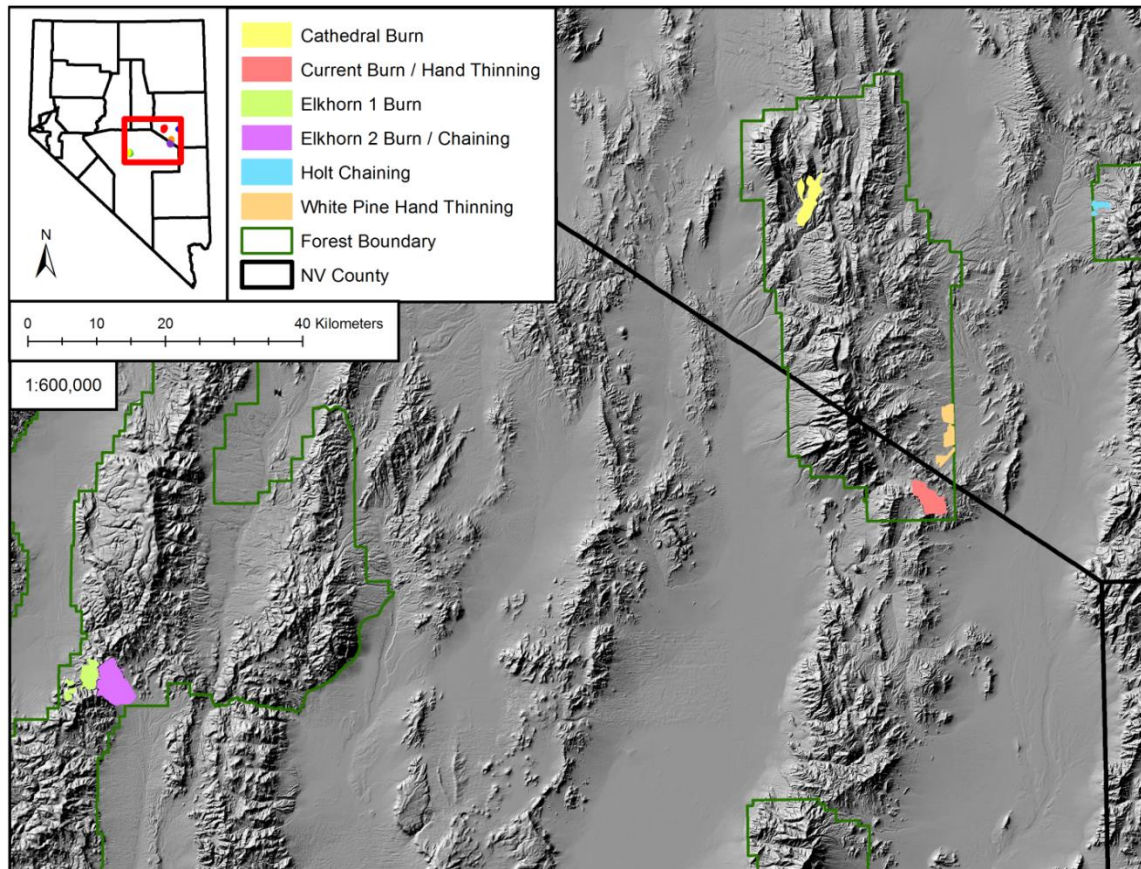
resilience is reduced (Roundy et al. 2014). Areas with low resilience are more likely to convert to undesirable vegetation such as invasive annual grasses after disturbance.

Land managers have attempted to control the expansion of Pinyon-juniper through a variety of methods including prescribed fires (Gruell and Swanson 2012), chainings (Bates et al. 2011), thinning (Floyd and Romme 2012), clear cutting (Pierson, Kormos and Williams 2008), mastication (Romme et al. 2009), and herbicide application (Tausch and Hood 2007). Management goals for the treatments are not always achieved. The same treatment applied to different areas can result in widely different vegetation responses. Some treated areas are recolonized by Pinyon-juniper seedlings almost immediately, and within 30-70 years, the treatment needs to be redone to retain benefits. Other areas successfully convert into a new non-reversible state and Pinyon-juniper do not reencroach, even many years later (Briske et al. 2005). While success is improving, it is still difficult to know whether a treated area will be prevented from crossing an ecological threshold into a woody-dominated state or to predict the vegetation phase that will dominate after treatment. State and transition models can identify degraded areas and help restore them by selecting treatments that may push an area into a new state or prevent an undesirable state transition (Briske et al. 2008).

In Central and Eastern Nevada, hundreds of thousands of acres of Pinyon-juniper treatments are planned for the next decade. These treatments include prescribed fire, hand cutting, mechanical sagebrush suppression, mechanical tree removal, herbicide treatment, mowing, and chaining. Better state and transition models will focus treatments in more appropriate areas.

At lower elevation Pinyon-juniper areas, it can be appropriate to focus on treatments that are less likely to affect resilience. Fire is more likely than mowing to lower resilience. Mowing is more likely than tebuthiuron to decrease resilience (Shaff et al. 2012). Resilience increases with elevation due to higher productivity, more favorable growing conditions, more rapid recovery after disturbance, and increased capacity to compete with invaders (Wisdom and Chambers 2009; Brooks and Chambers 2011). Aspect, slope, and soil characteristics also affect resilience (Chambers 2012). Cheatgrass is more likely to establish at middle elevation areas. It does not do as well in low elevation salt desert scrub or high elevation mountain sage. Cheatgrass is more of a concern at the lower elevation ranges of Pinyon-juniper woodland.

This thesis was funded by the Humboldt-Toiyabe National Forest who wished to know more about the long term results of past Pinyon-juniper removal treatments in the Ely and Austin-Tonopah districts in Central and Eastern Nevada. To evaluate this, we selected 6 Pinyon-juniper removal projects in the districts including four burns, one chaining, and one hand thinning (Figure 1). Pinyon-juniper woodlands can be separated into phases based on the dominance of the understory community. Phase I is herbaceous or shrub dominated as trees establish. In phase II, trees become dominant over the understory. Phase III is when trees dominate and shrubs are very limited or nonexistent (Tausch et al. 2009). The undisturbed woodlands around White Pine are late phase I or early phase II. Cathedral, Elkhorn 1, Elkhorn 2, Currant and Holt are phase III.



**Figure 1.** Project location overview map

Recent policy direction encourages the Bureau of Land Management (BLM), U.S. Forest Service (USFS), and Natural Resources Conservation Service (NRCS) to cooperatively identify and describe rangeland ecological sites for use in inventory, monitoring, evaluation and management of the Nation's rangeland (Caudle et al. 2013). State and transition models have been developed for the White Pine, Currant, Cathedral, and Holt projects. State and transition models have been developed for BLM adjacent to the Elkhorn 1 and 2 projects (Soil Survey Staff 2015). State and transition models seek to describe indicators that can be used to predict treatment outcomes and select among treatment options for an individual location of an ecological site (Stringham et al. 2003).

State and transition modeling predicts that an ecological site may exhibit vegetation characteristic of any one of many potential alternative states or phases of a state such as woodland, shrubland, or grassland (Westoby et al. 1989). Within each state, normal successional models operate as vegetation communities mature between disturbances. These are known as transitions. However, a combination of factors may lead to an area crossing an irreversible threshold and transitioning from one state to another (Briske et al. 2008). When this happens, the former vegetation will not recover after disturbance. Resilience of the former community has been lost and a new plant community will dominate the site. Efforts to return an area to an historic state are called restoration (Floyd and Romme 2012).

Rangeland degradation is characterized by changes in vegetation age structure, less seed production for heavily compromised species, and increased seed production for undesirable species. Degradation may cause a decrease in diversity, productivity, and a reduction in perennial plant cover. The final degraded state is when vegetation cover is lost, erosion increases, and soil salinity increases (Milton et al. 1994). Treatments should prevent or restore ecological functions after degradation.

This study also seeks to identify the effects of fire, chainings, and hand thinning treatments on soil and site stability, hydrologic function, and biotic integrity of Pinyon-juniper encroached areas across a range of elevations, slopes, aspects, and precipitation levels in three of the major mountain ranges within the Humboldt-Toiyabe National Forest in Central Nevada. Soil and site stability was assessed with soil stability tests, a soil pedon, and infiltration. Hydrologic function was assessed with infiltration. Biotic integrity was assessed with frequency, density, line point intercept, and canopy gap.



These measurements were recorded at 73 macroplots. This information was used to determine if the treatments triggered or prevented a state change. The information collected should help USFS managers plan treatments.

## **Thesis Overview**

Goal: Empower adaptive management of Pinyon-juniper in the Ely and Austin-Tonopah Districts and across the Western Great Basin ecosystem by determining how past Pinyon-juniper treatments have varied in their effect on sagebrush ecosystems, including the habitat for sage grouse and other sagebrush dependent wildlife species.

This thesis is a summary of the monitoring results from six Pinyon-juniper removal treatments. The three chapters present the result of the monitoring. The chapters were organized by treatment type and pre-treatment vegetation type. The first chapter deals with the four burns, the second chapter deals with the hand thinning, and the third chapter deals with the chaining. The studies will be directly relevant to land managers in the Humboldt-Toiyabe National Forest and other areas of the Great Basin where pre-treatment soils, vegetation, and precipitation patterns are similar. The conclusion presents a management implications synthesis that discusses the results of the previous chapters in relation to each other and discusses planning, implementing, and monitoring Pinyon-juniper removal treatments. Comparing different treatment types in Pinyon-juniper woodlands is useful for informing future land management decisions.

The first chapter looks at the four burns that occurred between 2005 and 2008 with similar untreated plant communities, but very different post-fire responses. Chapter two examines the White Pine Hand Thinning with less than half the untreated Pinyon-

juniper cover of the other projects. Chapter three looks at the late 1970s Holt Chaining, which had significantly more Pinyon-juniper cover in the adjacent untreated areas than the four burns or the White Pine Hand Thinning. The Holt Chaining was much older than any of the other treatments, done since 2005.

Chapter four presents a synthesis of management implications from the first three chapters and talks about how they relate to each other. The chapter goes on to make recommendations about selecting treatments for a Pinyon-juniper project to meet the goals of land managers.

## **Research Methods**

### **Study Area Description**

Projects are located within the Monitor, White Pine, and Egan mountain ranges, within Major Land Resource Area 28B - Central Nevada Basin and Range (Natural Resources Conservation Service 2006). Projects are all within the Great Basin ecoregion and range from 1910 m to 2847 m in elevation according to the National Elevation Dataset (NED) (Gesch et al. 2002; Gesch et al. 2007). Final Ecological Site Descriptions have been completed for Cathedral, Currant, Holt, and White Pine (Soil Survey Staff 2015). Ecological site descriptions exist for BLM land adjacent to the Elkhorn burns. The plant communities in the area include Pinyon-juniper woodlands, mahogany savannas, sagebrush shrublands, and grasslands; but all areas have at least some Pinyon-juniper trees currently present. Average annual precipitation ranges from 200 to 350 +cm per year based on the soil descriptions (Soil Survey Staff 2015). Average annual precipitation ranges from 246 to 387 mm according to PRISM (2013) extrapolation. The majority of

the precipitation falls in winter, but there are significant summer monsoons as well. The closest weather station to Elkhorn 1 and 2 is in Eureka, NV roughly 120-130 km NE of the projects. The closest weather station to Currant, White Pine, and Holt is in Ely, NV roughly 15-20 km NE of Holt and 45-65 km NE of Currant and White Pine. The weather stations are both 50-60 km from Cathedral with Ely to the E and Eureka to the NW. The weather station in Eureka is at 1996 m at the lower end of the range of plot elevations (1945 m – 2365 m). The weather station in Ely is slightly lower at 1905m. The Western Regional Climate Center (WRCC) precipitation data from the stations show that annual precipitation in the area is not consistent from year to year (Figure 1, Figure 2) (WRCC 2015). Large jumps in a year can be up to double or less than the half of the average precipitation. The years during monitoring – 2013 and 2014 – were both slightly below average at both weather stations.

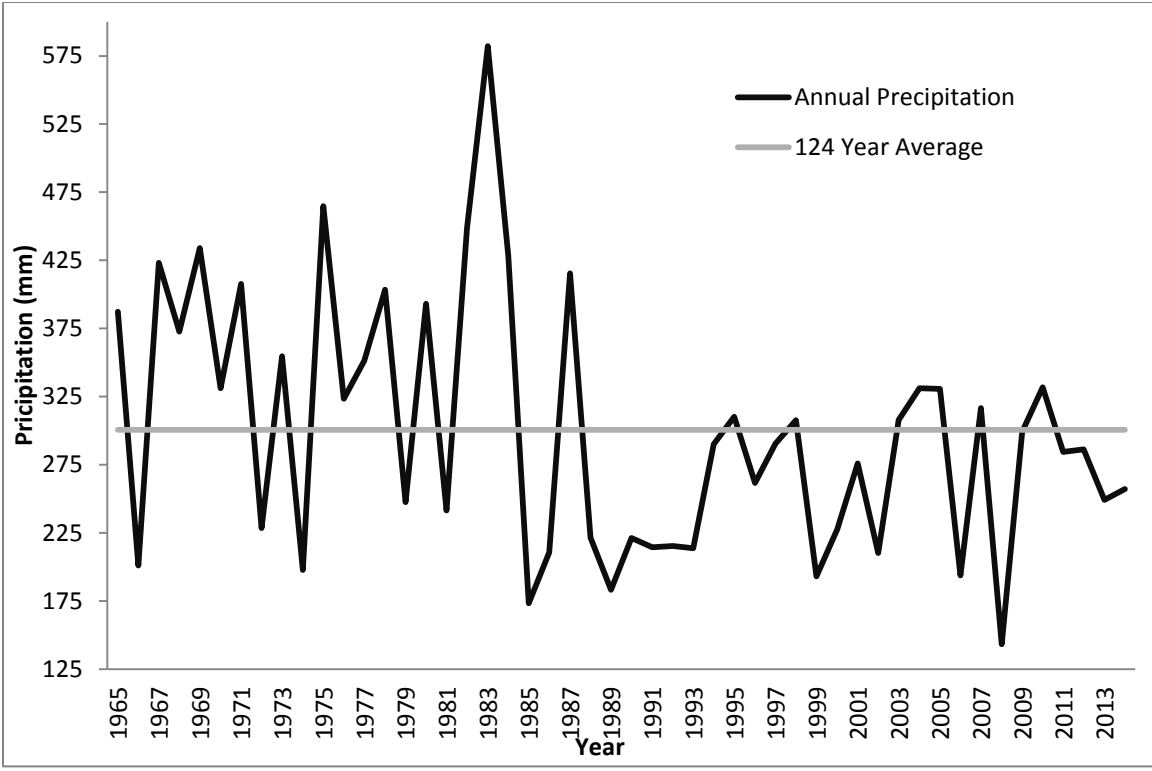


Figure 1. Annual precipitation from Eureka WRCC weather station 1965-2014.

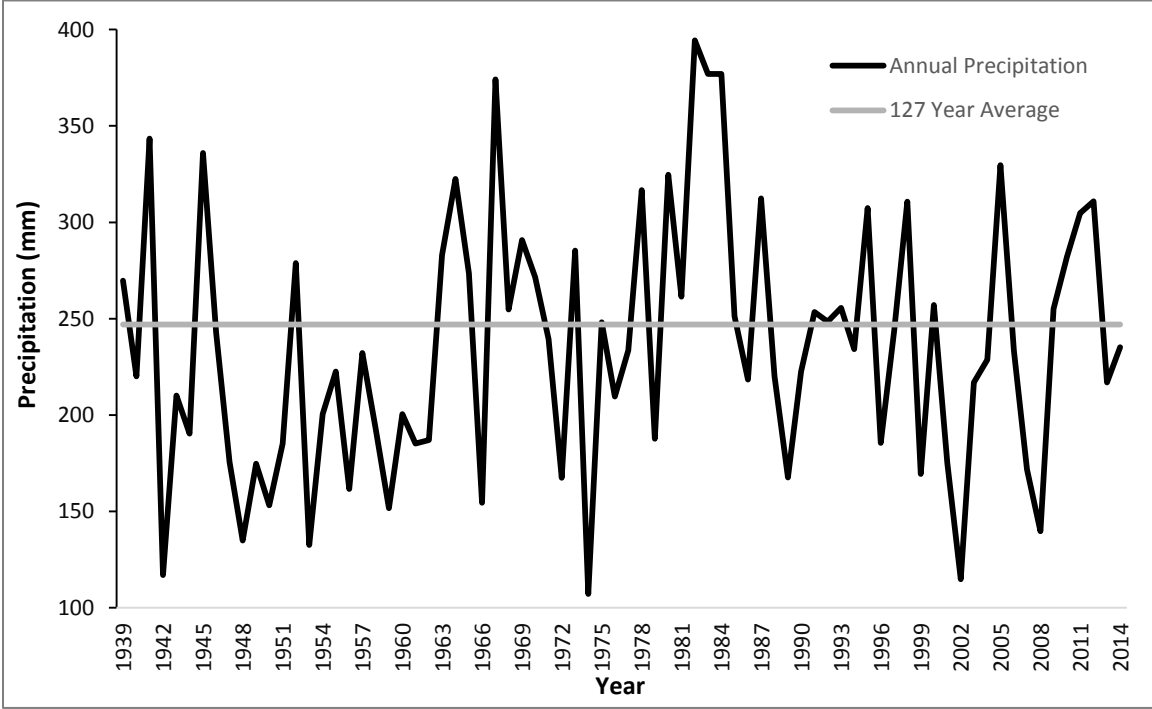


Figure 2. Annual precipitation from Ely WRCC weather station 1939-2014.

Project boundaries were established using pre-existing USFS project shape files and aerial imagery. The aerial imagery used was from the 2006 and 2010 National Agriculture Imagery Program (NAIP) (United States Geological Survey 2012), and 1994 and 1999 Digital Orthophoto Quarter Quads (DOQQs) (United States Geological Survey 2013). Stratified polygons were delineated to minimize variation between treatment and nearby untreated areas considering elevation, slope, aspect, precipitation and soil type. Within each polygon, three randomly located potential macroplot location points were selected. Only one point was monitored within each polygon, but multiple points were established in case the first or second point were not possible to monitor.

## **Plot Selection**

Untreated plot locations were paired to be on the same landform and soil association with no more than 50 m elevation difference based on the NED, 45 degrees of aspect difference based on the NED, 50 mm of precipitation difference based on PRISM (2013), and 10% slope difference based on the NED when compared with treated plots. NAIP imagery from 2006 and 2010 and DOQQs from 1994 and 1999 were used to examine the areas visually and attempt to pair areas that appeared to have similar landform. When pretreatment imagery was available (all projects except for Holt), attempts were made to select untreated areas that had a visually similar density of Pinyon-juniper trees. Plots were placed in the same grazing allotments and pastures.

After potential plot polygons were created, a 20 m buffer was added inside each polygon to ensure the plot could not run outside the chosen polygon. Three randomly selected points were generated within each potential plot polygon. If the first randomly

generated point was unsuitable for sampling, the plot moved the second, or if necessary the third randomly generated point. If the third point was unsuitable, the polygon was skipped. Points were deemed unsuitable if they were not in the expected treatment area or unsafe to monitor due to cliffs.

A total of 73 plots were monitored in 2013 and 2014. Plot elevations represented a 400 m elevation band at the lower range of elevation band where Pinyon-juniper occur. (Table 1, Figure 3). A wide range of slopes were samples from nearly flat to near the angle of repose for many of the soil types (Table 1, Figure 4). Annual precipitation at the plots represented a roughly 100 mm range (Table 1, Figure 5). All sites were semi-arid. Plots were sampled from all aspect classes (Figure 6). Aspect classes were evenly distributed for the potential plots site, but after sampling, the breakdown at the actual plots monitored was slightly biased with more plots on SE and E facing slopes than other slopes.

**Table 0-1.** Macroplot location summary. Project site count, area, elevation attributes, slope attributes, and precipitation attributes. Min = Minimum, Max = Maximum, SD = Standard Deviation.

		<u>Cathedral</u>	<u>Currant</u>	<u>Elkhorn 1</u>	<u>Elkhorn 2</u>	<u>Holt</u>	<u>White Pine</u>	<u>Total</u>
<b>Plot Count</b>		12	12	10	17	10	12	73
<b>Area (Ha)</b>		606	237	1103	1002	160	920	4028
<b>Elevation (m)</b>	<b>Min</b>	2095	2004	2281	2056	212	1945	1945
	<b>Mean</b>	2198	2086	2316	2109	215	1959	2130
	<b>Max</b>	2287	2158	2365	2250	221	1981	2365
	<b>SD</b>	56.38	58.98	30.15	63.12	29.5	9.17	114.42
<b>Hill Slope (%)</b>	<b>Min</b>	11	5	4	3	1	1	1
	<b>Mean</b>	26.5	22.83	10.4	13.88	9.5	3.08	14.58
	<b>Max</b>	48	46	26	44	23	7	48
	<b>SD</b>	11.58	14.50	6.77	15.33	6.75	2.02	13.42
<b>Precipitati</b>	<b>Min</b>	312	316.25	338.39	246.14	314.	279.37	246.14

on (mm)	<b>Mean</b>	339.04	326.58	349.82	259.46	330.	285.67	310.04
	<b>Max</b>	348.66	334.35	386.70	304.66	378.	289.77	386.70
	<b>SD</b>	12.68	7.37	14.32	18.55	20.5	3.82	36.86



Figure 3. Macroplot elevation given in m.

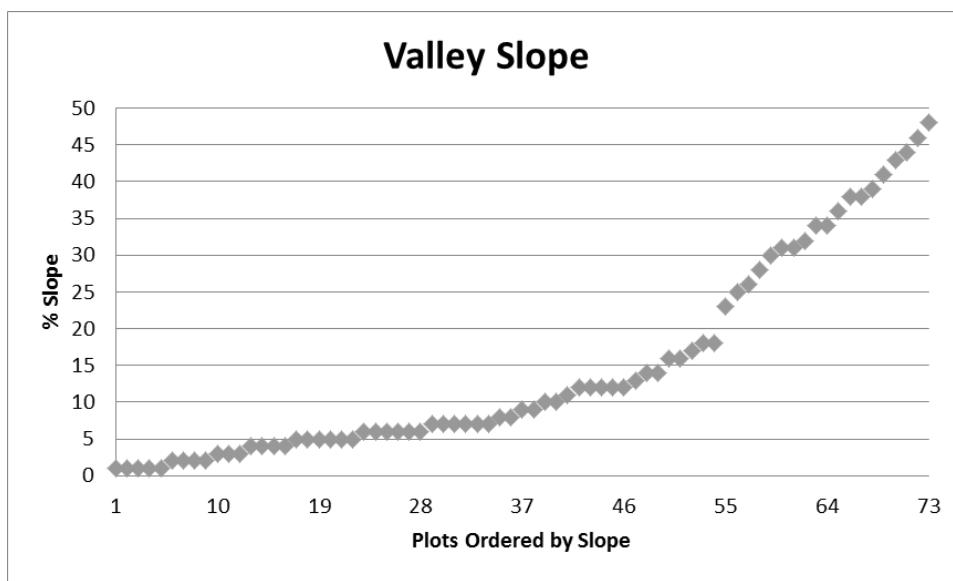
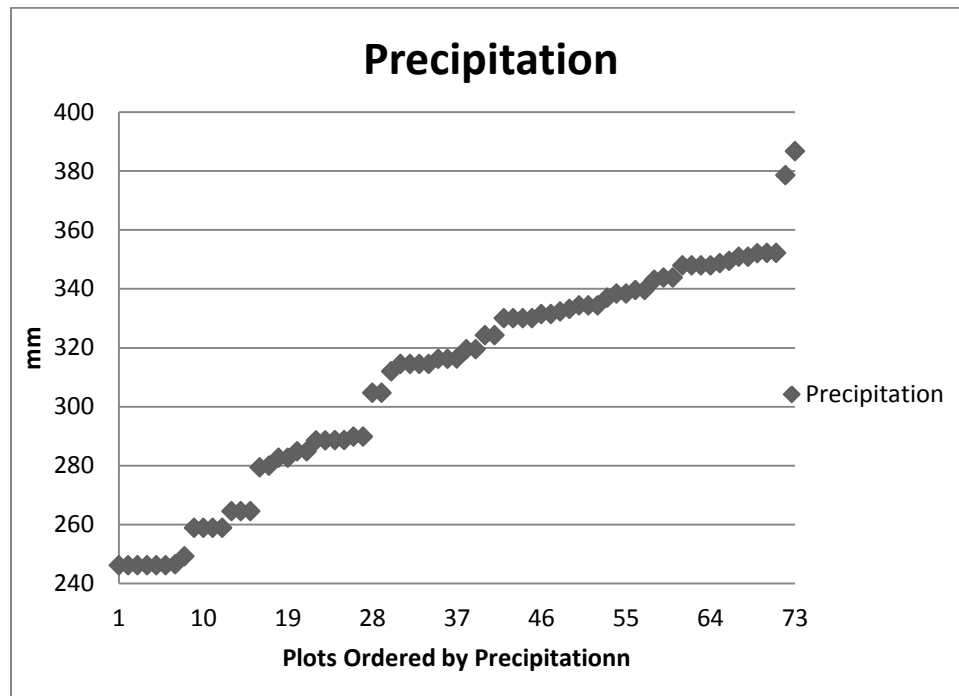
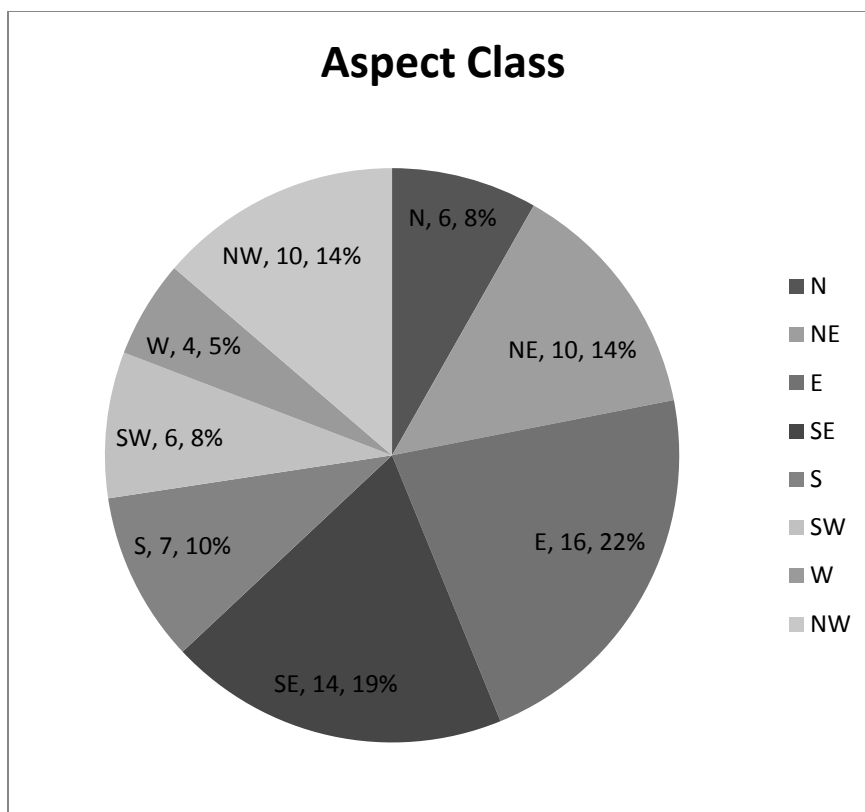


Figure 4. Macroplot hill slope given in % slope.



**Figure 5.** Macroplot precipitation given in mm based on PRISM (2013).





**Figure 6.** Count of macroplots by aspect.

Macroplots occurred in 21 different soils associations based on Soil Survey Staff (2015) (Table 2). Four of those associations had at least ten macroplots. Two soil associations occurred in multiple treatments. Several soil associations were similar to each other. Some individual soil series occurred in three or four soil associations (Table 3). These similar soils may have similar responses to treatment (Soil Survey Staff 2015).

**Table 2.** Soils where plots occurred. MUSYM is map unit symbol.

Association MUSYM	Association Name	Count of Plots in Association	Projects
150	<i>Gochea-Zadvar</i>	13	Elkhorn 2
166	<i>Cropper-Bellenmine-Rock outcrop</i>	10	Elkhorn 1
430	<i>Bellehelen-Rock outcrop</i>	4	Elkhorn 2
3860	<i>Hyzen-Zimbob-Rock outcrop</i>	4	Currant
4518	<i>Duffer-Pern-Belmill</i>	2	Cathedral

4526	Amelar-Birchcreek-Cavehill	2	Cathedral
6120	Tecomar-Pookaloo-Zimbob	14	Cathedral, Currant
6288	Palinor-Yody-Broland	14	Currant, White Pine
6296	Palinor-Urmafot-Palinor	2	Holt
6334	Parisa-Palinor-Shabliss	8	Holt

**Table 3.** Soil associations with soil series that are repeated in multiple associations where plots actually occurred. An “x” indicates a soil series that occurs within an association. The numbers in the left column refer to the MUSYM of a particular association. See Table 2 for association name.

	Mizpah	Palinor	Pyrat	Rock Outcrop	Sevenmile	Shabliss	Squawtip	Suak	Zimbob
150					x		x		
166				x				x	
430				x	x		x		
3860				x					x
4518				x					
4526				x				x	
6120			x	x					x
6288		x				x			
6296	x	x	x			x			
6334	x	x	x			x			

## Project Descriptions

### Cathedral

The Cathedral Fire is a wildfire in the White Pine Range. It burned 606 ha in July 2007. It ranges in elevation from 2129 to 2586 m. Slopes range from 0% to 94% with an average slope of 23%. There is no predominant direction of slope. The fire burned both sides of a SW to NE trending ridge as well as few adjacent slopes. Average annual precipitation for the site ranges from 312 mm to 349 mm based on PRISM (2013) extrapolation (Table 1). The soil associations indicate precipitation ranges from 200mm to 350+ mm. Soil data indicated the fire included portions of 11 soil associations. Thirty

paired polygons (15 treated, 15 untreated) were established in the area. Six pairs of plots were monitored from those 15 possible plot pairs. Two plots (one treated, one untreated) were monitored in 4518 – Duffer-Pern-Belmill soil association. Two plots (one treated, one untreated) were established in 4526 – Amelar-Birchcreek-Cavehill soil association. Eight polygons (four treated, four untreated) were established in 6120 – Tecomar-Pookaloo-Zimbob soil association (Soil Survey Staff 2015) (Table 2).

## **Currant Creek**

Currant Creek is a prescribed burn in the White Pine Range. It was completed in June 2010 covering 237 ha. After the burn, some of the standing tree stumps were removed for fuel wood by members of the public on a small portion of the fire. Slopes range from 0% to 71% with an average slope of 19%. There are aspects of all types and directions. The area ranges in elevation from 1910 m to 2356 m. Average annual precipitation for the site ranges from 316 mm to 334 mm based on PRISM (2013) extrapolation (Table 1). The soil associations indicate precipitation ranges from 200 to 350 mm. Soil data indicates the treatment contains portions of eight soil associations. Twenty-two paired potential polygons (11 treated, 11 untreated) were established in the area. Six pairs of plots were monitored from those 11 possible pairs. Two polygons (1 treated, 1 untreated) were monitored in soil 6288 – Palinor-Yody-Broland association. Six polygons (3 treated, 3 untreated) were monitored in 3860 – Hyzen-Zimbob-Rock outcrop soil association. Six plots (three treated, three untreated) were monitored in 6120 – Tecomar-Pookaloo-Zimbob soil association (Soil Survey Staff 2015) (Table 2).

## **Elkhorn 1 and Elkhorn 2**

The Elkhorns are a series of prescribed burns in the Monitor Range. They range in elevation from 2001 m to 2857 m. The Elkhorns burned adjacent sides of the Monitor Range. They come within 250 m of each other at the closest point near the top of the ridge, but their far points are more than 10 km away from each other. All of the Elkhorn 1 plots are at least 6 km away from the nearest Elkhorn 2 plots.

Elkhorn 1 was a prescribed burn from October 2005 and May 2006. It burned 1103 ha. Elevation ranges from 2291 m to 2816 m. Slopes range from 1% to 125% with an average slope of 26.7%. There is no predominant direction of slope. Average annual precipitation for the site ranges from 338 mm to 386 mm based on PRISM (2013) extrapolation (Table 1). The soil associations indicate precipitation ranges from 200 mm to 350+ mm. Preliminary soil data indicate the fires include portions of 6 soil associations. Four previously monitored and established macroplots exist within Elkhorn. An untreated plot was paired with each previously monitored point. An attempt was made to locate all previously established plots, but only one of the four was successfully located. An additional 10 paired polygons (5 treated, 5 untreated) were established in the area. A total of 12 plots were monitored. Based on preliminary soil data, 4 polygons (2 treated, 2 untreated) were established in soil 110 – Scuffe-Packer-Rock outcrop association. Sixteen polygons were established in soil 166 – Cropper-Bellenmine-Rock outcrop association (Soil Survey Staff 2015) (Table 2).

Elkhorn 2 was started as a prescribed burn, but exceeded expectations and was declared a wildfire. It burned 1002 ha in June 2008. After the burn, chaining was

implemented on 54 ha within the burned area. Elevation ranges from 2001 m to 2847 m. Slopes range from 0% to 130% with an average slope of 26%. Aspects are primarily N, NE, E and SE. Average annual precipitation for the site ranges from 246 mm to 305 mm based on PRISM (2013) extrapolation (Table 1). The soil associations indicate precipitation ranges from 200 mm to 350+ mm. Preliminary soil data indicates the fire includes portions of nine soil associations. At least three of these associations appear to extend onto adjacent mapped ecological sites. Five old monitoring plots exist within Elkhorn 2 and one old untreated plot exists just outside Elkhorn 2. Five of the six total old monitoring plots were resampled including two burned plots, two burned and chained plots, and one untreated plot. Two untreated plots were established and monitored that paired with existing old plots. Two sets of three polygons (burned, burned and chained, and untreated) were established and monitored in the area. An additional 12 paired polygons (six treated, six untreated) were established in the area. Four pairs of plots were monitored from those six possible pairs. In total, 17 plots were monitored on the Elkhorn 2 fire including six burned plots, four burned and chained plots, and seven untreated plots. Four plots (two treated, two untreated) were established in 430 – Bellehelen-Rock outcrop soil association. Thirteen plots (eight treated, five untreated) were monitored in 150 – Gochea-Zadvar soil association (Soil Survey Staff 2015) (Table 2).

## **Holt Chaining**

The Holt Chaining is a chaining treatment on Ward Mountain. It was done in the early 1970s. It treated 160 ha. It ranges in elevation from 2111 m to 2309 m. Slopes range from 1% to 20%. Aspects are predominantly N, NW, W, and SW facing and the overall

hillside slopes W. Average annual precipitation for the site ranges from 314 mm to 379 mm based on PRISM (2013) extrapolation (Table 1). The soil associations indicate precipitation ranges from 200 mm to 350 mm. Soil data indicates the fire includes portions of three soil associations. One previously monitored site exists within the Holt Chaining that was resampled. An untreated plot was paired with the previously monitored site. An additional 20 paired polygons (10 treated, 10 untreated) were established in the area. Five pairs of plots were monitored from those 10 possible pairs. Two plots (one treated, one untreated) were established in soil 6296 - Palinor-Urmafot-Palinor association. Eight plots (four treated, four untreated) were established in soil 6334 - Parisa-Palinor-Shabliss association (Soil Survey Staff 2015) (Table 2).

## **White Pine Hand Thinning**

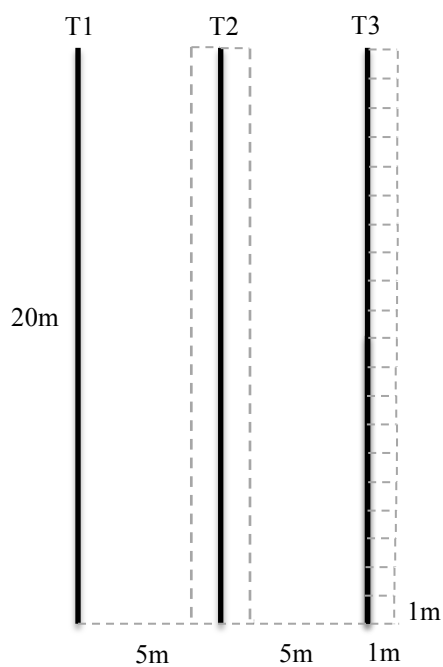
The White Pine is a hand thinning project around the White Pine Range. The plots were located in an area done between 2009 and 2012 on 920 ha. The White Pine hand thinning continued to expand while monitoring and by 2014, 1004 ha had been completed. It ranges in elevation from 1932 m to 2031 m. Slopes range from 0% to 18% with a mean of 3%. Aspects are predominantly NE, E, or SE and the overall hillside slopes E. Average annual precipitation for the site ranges from 279 mm to 290 mm based on PRISM (2013) extrapolation (Table 1). The soil associations indicate precipitation ranges from 200 mm to 300 mm. Soil data indicate the hand thinning includes portions of 3 soil associations. Twenty paired polygons (10 treated, 10 untreated) were established in the area. Six pairs of plots were monitored from those 10 possible pairs. Twelve plots (six

treated, six untreated) were established in soil 6288 – Palinor-Yody-Broland association (Soil Survey Staff 2015) (Table 2).

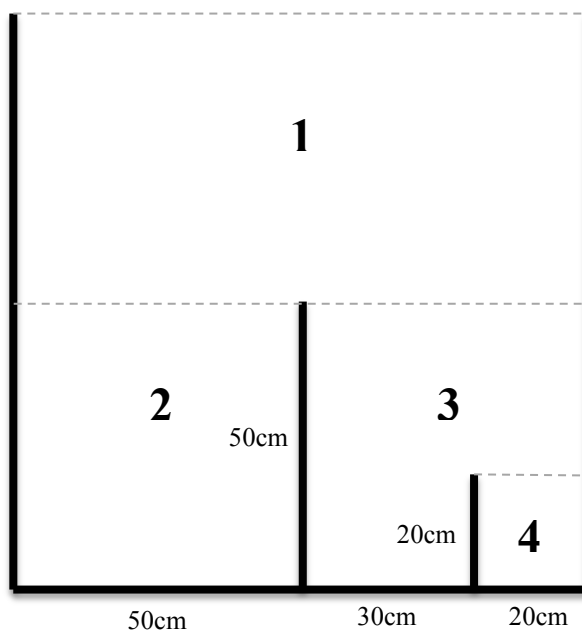
## **Experimental Design**

### **Monitoring Protocol**

Each macroplot consisted of three, 20 m long belt transects spaced 5 m apart (Figure 7). Along each belt transect, 20 frequency frames were placed at 1 m intervals along the uphill side of the belt transect. Nested frequency was recorded for each frame using the protocol from Coulloudon et al. (1999). The nested frequency frame is subdivided into four sections with areas of 1 m<sup>2</sup>, 0.5 m<sup>2</sup>, 0.25 m<sup>2</sup>, and 0.04 m<sup>2</sup> (Figure 8). Canopy gap intercept was recorded along each 20 m transect using the protocol from Herrick et al. (2009). Canopy line point intercept was recorded along each 20 m transect at 20 cm intervals to provide 100 canopy cover points per transect. Perennial density was recorded around the transects counting plants that occurred up to 1 m away on either side of the transect. Density along each transect covered a 2 x 20 m total area (40 m<sup>2</sup>). Each macroplot provided three samples of frequency, density, cover, and canopy gap.



**Figure 1.** Overview of a macroplot. The three transects, T1, T2, and T3 are each 20 m long. T2 shows the 40 m<sup>2</sup> area where density was recorded at each transect. T3 shows how the one m<sup>2</sup> nested frequency frames was placed at 1 m intervals along each transect. Canopy gap intercept and line point intercept were recorded along each transect.



**Figure 2.** Nested frequency frame design. A 1 m<sup>2</sup> frame with 4 sections. Section 1 is 1 m<sup>2</sup>. Section 2 is 0.5 m<sup>2</sup>. Section 3 is 0.25 m<sup>2</sup>. Section 4 is 0.04 m<sup>2</sup>.



The start and end of the first transect at each plot was monumented with lath installed in 2013 and relocated in 2014.

A soil stability test was completed at each macroplot during the second summer using the protocol described in Herrick et al. (2009). The test was done using a Jornada Experimental Range Soil Stability Test Kit (Seybold and Herrick 2001). The soil stability test included 9 surface and 9 subsurface samples taken from along the primary transect at each plot on the opposite side as frequency. Samples were taken at 2m intervals from 2m to 18m. Each sample was assigned to an aggregate stability class from 0 (low) to 6 (high) (Table 1) (Herrick et al. 2001). Soil stability was the first protocol performed after rolling out the transect tape so that effects of trampling were kept to a minimum. The surface sample was taken from the top of the soil surface (litter removed). The subsurface sample was taken from excavating down 2-5cm directly below the surface sample location.

**Table 1.** Soil Aggregate Stability Class Standard Characterization Criteria

Stability class	Criteria
0	Soil too unstable to sample
1	50% of structural integrity lost within 5 s of insertion in water
2	50% of structural integrity lost within 5 to 30 s of insertion in water
3	50% of structural integrity lost within 30 to 300 s of insertion in water OR < 10% of soil remains on sieve after five dipping cycles
4	10–25% of soil remains on sieve after five dipping cycles
5	25–75% of soil remains on sieve after five dipping cycles
6	> 75% of soil remains on sieve after five dipping cycles.

A soil pit was dug at each macroplot in 2014. The soil pits were located in open interspace areas no more than 15 m from the start transect one. The soil was classified into a soil series using the protocols described in Soil Survey Staff (1999).

A test for water infiltration was done at each macroplot during the second summer using the protocol described in Herrick et al. (2009). The test was done using a single-ring infiltrometer (Lowery et al. 1996) in interspace areas for every plot. The infiltration test was also done under the canopy of selected species that varied depending on the vegetation at the plot. Normally, 3-5 species were selected at each plot that represented the major lifeforms present including trees, shrubs, perennial forbs, perennial grasses, and annual grasses. The infiltration test timed how long 150ml of water took to infiltrate an area of 20.25 cm<sup>2</sup>.

Abiotic data were collected at each macroplot that included GPS location (with accuracy of <3.3m, 95% typical), elevation, transect azimuth, aspect, transect slope, and valley/hill slope using a clinometer. Field crews measured direction with a compass (declination 13 degrees east) and slopes with clinometer that can be read directly to 1%. Qualitative information about dominant vegetation and additional species in the general area of the plot not encountered during monitoring was recorded at each macroplot. The presence of wildlife and stock was also noted when grazing species or their evidence were encountered during monitoring. Notes were also made whenever signs of noxious weeds or OHV activity was encountered.

Photographs were taken from the start and end of the each transect. The primary photograph of each transect was taken from a point 1.2 m behind the start of the transect at a height of 1.5 m. The horizon was placed approximately level in the photograph one third down from the top of the frame. For transects that were previously photographed, an effort was made to recreate the photo as close as possible. Field crews took photographs with a Nikon D7000 or D600 camera, lens with 36 degree field of view, and attached

Aokatec AK-G7 GPS receiver. Photographs were georeferenced with location and bearing. GPS accuracy was  $\pm 5$ m. Compass accuracy was  $\pm 2$  degrees.

Field crews collected 121 herbarium specimens that represented 109 taxa from 77 genera and 30 families. The primary target of collections was the most common species encountered during monitoring as vouchers to confirm field identifications. Herbarium specimens were also included unknown or interesting plant species near plots. Most herbarium specimens were collected with three replicates. Herbarium specimen replicates were donated to the University of Nevada-Reno Herbarium in Reno, NV; Intermountain Herbarium at Utah State University in Logan, UT; and the Eastern Nevada Landscape Coalition herbarium in Ely, NV.

## **Data Management**

All field data were collected on paper data sheets. Data sheets for frequency and cover were taken from Coulloudon et al. (1999). Data sheets for soil stability were taken from Herrick et al. (2009). Data sheets for density, soils, and gap were created for this project. After returning from the field, data were stored in a fireproof safe. All paper datasheets were digitized and stored in a Microsoft Access 2010 database. Data summaries and statistical analyses were done using a combination of Microsoft Excel 2010, Microsoft Access 2010, R Statistical Software version 3.1.2, PC-ORD version 6, and ESRI ArcGIS 10.

# **Chapter 1: Influence of tuffaceous soils on cheatgrass susceptibility in burned Pinyon-juniper woodlands in Central Nevada.**

## **Abstract**

Despite a great deal of research being used to try to predict the outcomes of Pinyon-juniper removal treatments (*Pinus monophylla* – *Juniperus osteosperma*), outcomes vary among areas that appear to have similar resistance, resilience, and pre-treatment vegetation. We established 51 monitoring plots (27 treated, 24 adjacent untreated) across 4 projects in Central and Eastern Nevada to measure vegetation treatment effects on montane rangeland soils. The untreated area had similar elevation, precipitation, and pre-treatment vegetation communities, but the treated areas had significantly different responses to treatment. With nonmetric multidimensional scaling (NMS), we found a useful 3-axis ordination of the plots (stress=7.1,  $R^2=.966$ ). Within ordination space, the treated plots were well grouped by parent material. These results informed a Poisson generalized linear model that found factorized parent material explained 86.5% of the deviance in cheatgrass (*Bromus tectorum*) cover at the treated plots. The projects on soils derived from welded tuff had very little cheatgrass while soils derived from limestone or mixed limestone/other volcanics were dominated by cheatgrass. Tuffaceous soils near the top of cheatgrass' precipitation envelope were less likely to convert to cheatgrass following fire than other soils within the same precipitation zone. Parent material should be considered an important factor when planning Pinyon-juniper removal treatments and evaluating resistance to cheatgrass following fire.

## Introduction

The expansion and infilling of Pinyon-juniper (*Pinus monophylla* – *Juniperus osteosperma*) woodlands in the Western United States is a challenge for land managers concerned about production, resistance to invasives, resilience to disturbance, and diversity (Miller and Tausch 2001, Allen and Nowak 2008). Fire is one of the main tools used for removal on Pinyon-juniper and other vegetation both by land managers and as part of the natural succession process (Gruell 1999; Gruell and Swanson 2012). Prescribed fire and chaining are cheap compared to lop-pile-burn, lop and scatter, feller-buncher, and mastication (Cleaves et al. 1999; Provencher and Thompson 2014). In dense Pinyon-juniper woodlands, high intensity wildfire can nearly completely eliminate vegetation (Tausch 1999; West 1999).

Prescribed fire is not without disadvantages. It can be hard to control. Even “controlled burns” can escape containment and burn areas not intended to burn as did the Elkhorn 2 prescribed fire in this study. Possibly the most negative aspect of fires in the Great Basin is their tendency to convert to undesirable non-native annuals like cheatgrass (*Bromus tectorum*) after fire. Cheatgrass can establish on recently burned areas, increase the level of fine fuels, and lead to an increase in fire frequency known as a grass-fire cycle (D’Antonio and Vitousek 1992). Each burn through cheatgrass perpetuates cheatgrass dominance and further reduces perennials (Young and Evans 1973; Young and Clements 2009).

After dominating, cheatgrass ensures its continued expansion through shortening the fire cycle (D’Antonio and Vitousek 1992) and changes to the soil (Blank and Morgan

2013). When a fire burns through phase III Pinyon-juniper forests, the post-fire community is more susceptible to cheatgrass (Miller et al. 2014). Cheatgrass reduces organic material inputs compared with native vegetation, but increases surface microbial activity, porosity, and decomposition rates. This can lead to a long term depletion of soil organic material which can make a site difficult to restore to native vegetation (Norton et al. 2004). Once cheatgrass is established, it becomes difficult to eliminate (Knapp 1996). Targeted grazing can be used for cheatgrass control (Svejcar et al. 2014). Fires in phase III Pinyon-juniper burn hotter and more completely. Less vegetation survives the fires. Because the trees have outcompeted understory vegetation, there are not as many fire tolerant plants available to resprout after the burn. The hot fire can kill perennial grasses that would normally resprout under less intense burns. For these reasons, sites with high Pinyon-juniper cover and little understory are considered less resilient than sites with more diverse vegetation structure (Chambers et al. 2014). Cheatgrass density is reduced in areas where it competes with perennial grasses (Young et al. 1972; Beckstead and Augspurger 2004).

Pinyon-juniper dominance is widely distributed and occurs on a variety of soil types (Miller and Wigand 1994). Condon et al. (2011), in a landscape scale analysis of a 2,800 ha fire through phase III Pinyon-juniper, studied post fire cheatgrass invasion and sagebrush recovery. They found that the best predictors of cheatgrass cover post fire were incident solar radiation and perennial herbaceous species cover. The study looked at many variables including soil depth, but did not discuss differences in parent material. Additionally, cheatgrass was relatively infrequent at their plots having only around 8.81% total canopy cover. Because the cheatgrass cycle is established through changes to

the soil, we hypothesize that susceptibility to cheatgrass after fire will be influenced by the soil type.

## **Methods**

### **Study Area**

The study areas were located in the White Pine and Monitor Ranges in Nye and White Pine Counties, Nevada, U.S.A. (latitude 39°14'05" N to 38°31'13" N; longitude 116°44'02" W to 115°15'48" W). The treated area consists of four burns totaling 4,454 ha. The Cathedral burned area is in the White Pine Range about 50 km west of Ely, Nevada. The Carrant burned area is roughly 40 km south of Cathedral. The Elkhorn 1 and Elkhorn 2 burned areas are roughly 120 km west of Carrant. Elkhorn 1 and 2 burned opposite sides of the Monitor Range. The burned areas come within 250 m of each other at their closest point near the ridge top. However, the plots at Elkhorn 1 are all separated from plots on the prescribed fire and studied part of Elkhorn 2 by 6 km to 10 km because the lower elevations of the areas were targeted due to accessibility for monitoring and matching elevation with the plots at other projects. Cathedral was a wildfire that burned in July 2007. Carrant Creek was a prescribed burn in June 2010. Carrant had dead stumps hand thinned to reduce standing trees after the fire on a portion of the burn. Elkhorn 1 was a prescribed fire that burned in two parts in October 2005 and May 2006. Elkhorn 2 was a prescribed fire in July 2008 that exceeded expectations and was declared a wildfire. Elkhorn 2 was partially chained after fire. Elevation ranges in studied areas from 2004 m to 2365 m. Hill slope ranges from 3% to 48% with an average slope of 18.4%. Average annual precipitation is 246 to 387 mm (PRISM Climate Group 2013) (Table 1). The plots

cover a full range of aspects. The area surrounding the burns are dominated by phase III Pinyon-juniper trees according the classification system of Miller and Tausch (2001) with a minor understory containing 3-10% cover of shrubs including black sagebrush (*Artemisia nova*), low sagebrush (*Artemisia arbuscula*), and Wyoming sagebrush (*Artemisia tridentata* ssp. *wyomingensis*). The burned areas included portions of eight soil associations and 16 ecological sites in multiple disturbance response groups. The soils at Cathedral were derived from limestone. The soils at Currant were derived primarily from limestone with some volcanics such as rhyolite mixed in. The soils at Elkhorn 1 and Elkhorn 2 were primarily derived from welded tuff with some mixed granite and non-tuffaceous volcanics such as rhyolite.

**Table 0-1.** Summary of burned areas. Area given in ha. Elevation given in m. Hill slope given as a percentage. Precipitation given in mm.

		<u>Cathedral</u>	<u>Currant</u>	<u>Elkhorn 1</u>	<u>Elkhorn 2</u>	<u>Total</u>
<b>Treated Plots</b>		6	6	5	10	27
<b>Untreated Plots</b>		6	6	5	7	24
<b>Area</b>		606	237	1103	1002	2848
<b>Elevation</b>	<b>Minimum</b>	2095	2004	2281	2056	2004
	<b>Mean</b>	2199	2087	2316	2109	2177
	<b>Maximum</b>	2287	2158	2365	2250	2365
	<b>St. Dev.</b>	56.38	58.98	30.15	63.12	104.29
<b>Hill Slope</b>	<b>Minimum</b>	11	5	4	3	3
	<b>Mean</b>	26.50	22.83	10.40	13.88	18.40
	<b>Maximum</b>	48	46	26	44	48
	<b>St. Dev.</b>	11.58	14.50	6.77	15.33	7.52
<b>Precipitation</b>	<b>Minimum</b>	312.00	316.25	338.39	246.14	246.14
	<b>Mean</b>	339.04	326.58	349.82	259.46	318.73
	<b>Maximum</b>	348.66	334.35	386.70	304.66	386.70
	<b>St. Dev.</b>	12.68	7.37	14.32	18.55	36.86
<b>Plot Count By</b>	<b>Limestone</b>	9	10			19



<b>Parent Material</b>	<b>Mixed limestone</b>	3	2	5
	<b>Mixed granite and volcanics</b>		4	4
	<b>Non-tuffaceous volcanics</b>		4	4
	<b>Tuffaceous volcanics</b>		6	13
				19

We established 51 plots, 27 treated and 24 paired plots in adjacent untreated areas. Polygons were created so plots could be paired in areas that shared similar precipitation, pre-fire tree cover, soil association, slope, aspect, and elevation. Plots were then randomly located within the polygons.

### **Sampling Design**

To characterize vegetation, we collected data on canopy cover, ground cover, nested frequency, perennial density, and canopy gap in summers of 2013 and 2014. We recorded line point-intercept canopy cover and ground cover (Herrick et al. 2009) using a laser point projection device every 20 cm along three 20 m long parallel transects spaced 5 m apart with 100 points per transect and 300 points per plot. Frequency (Caulloudon 1999) was collected using a 1 m<sup>2</sup> nested frequency frame placed at 1 m intervals along the uphill side of each transect (20 frames per transect and 60 frames per plot). The nested frequency frame was subdivided into 4 sections with areas of 1 m<sup>2</sup>, 0.5 m<sup>2</sup>, 0.25 m<sup>2</sup> and 0.04 m<sup>2</sup>. Perennial density was recorded as a total count of all perennial plants rooted within 1 m on either side of the transect (40 m<sup>2</sup> per transect and 120 m<sup>2</sup> per plot). Canopy gap intercept (Herrick et al. 2009) was recorded along each transect with a minimum gap size of 20 cm. To characterize soils, we collected data on soil aggregate stability, infiltration, and taxonomy in 2014. An interspace soil pit was described within

15 m behind the start of each transect. The soil was classified to soil series (Soil Survey Staff 1999). Soil aggregate stability (Herrick et al. 2009) was tested along the first transect on the side opposite frequency and cover. We collected nine samples, each at 2 m intervals from 2 m to 18 m. At each sampling point, one sample was collected from the surface and another from 2-5 cm directly below. Each sample point was assigned to a canopy class based on vegetation type: tree, shrub, perennial grass, perennial forb, or no perennial canopy. Samples under the canopy of perennial vegetation were considered protected versus unprotected in interspaces or with only annual canopy cover.

Interspace soil infiltration was estimated using a single ring infiltrometer (Lowery et al. 1996) with 150ml of water (74mm depth) over an area of 20.25 cm<sup>2</sup>. Infiltrometer locations were not pre-wet. Duff and litter was not removed from the infiltration site. Infiltration times were recorded to the nearest second up to one hour, or as “more than one hour.”

### **Statistical Analysis**

The four burned areas were all scored using the *Score Sheet for Rating Resilience to Disturbance and Resistance to Invasive Annual Grasses in the Great Basin* (Miller et al. 2014). Because the projects crossed multiple ecological sites, values for overall projects were averages from values of different ecological sites within the projects. We compared treated versus untreated plots in plant functional groups (annual forb, annual grass, perennial grass, forb, seeded grass, shrubs, and trees), ground cover classes (bare, litter, rock, pavement, moss, scat), surface and subsurface soil aggregate stability, infiltration, gap, species richness, and species diversity. For each category, analysis of

variance (ANOVA) (Fisher 1918) was used in R Statistical Software version 3.1.2 (R Core Team 2014) to determine if statistically significant differences existed between projects.

Non-metric multidimensional scaling (NMS) (Kruskal 1964; Mather 1976) was used in PC-ORD version 6 (McCune and Mefford 2011) using the “slow-and-thorough” autopilot mode and a Sorensen distance measure (Sørensen 1948) to ordinate the plots based on similarities in vegetation. Sørensen distance is a proportional city-block distance measure of dissimilarity (McCune et al. 2002). A Monte Carlo test was used to compare the results of the ordination with randomized data (Metropolis and Ulam 1949). A biplot (Gabriel 1971) was used to check if the axes of the ordination were correlated with other environmental or soil variables by calculating Pearson’s correlation coefficient (Pearson 1931). The environmental and soils variables were easting, northing, precipitation, elevation, aspect, valley slope, side hill slope, subsurface soil stability, surface soil stability, average soil stability, and interspace infiltration.

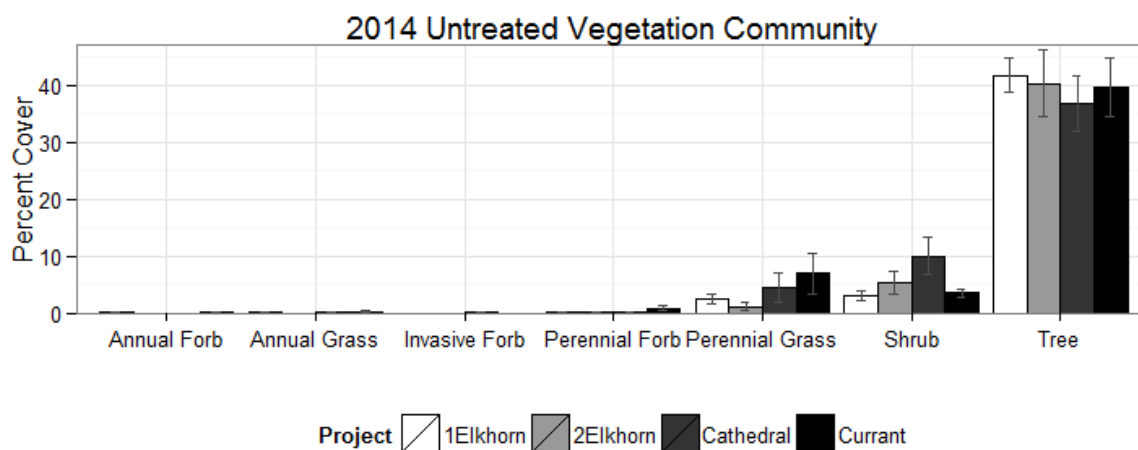
The results of NMS were used to inform a Poisson general linearized model (GLM) (McCullagh 1984) in R Statistical Software version 3.1.2 (R Core Team 2014) that predicted treated annual grass cover from parent material factorized, untreated annual herb cover, and untreated tree cover. A Poisson regression was selected for the dataset because there were many zeros in the data and the variables of interest followed a roughly Poisson distribution (Zuur et al. 2009). McFadden’s pseudo  $R^2$  was used to look at the proportion of variance explained by the GLM (McFadden 1973).

## **Results**

The Cathedral, Currant, Elkhorn 1, and Elkhorn 2 projects all received a resilience and resistance rating of “low” according to the *Score Sheet for Rating Resilience to Disturbance and Resistance to Invasive Annual Grasses in the Great Basin* (Miller et al. 2014) (Table 2). However, they had vastly different responses to fire. After burning, the vegetation communities at Cathedral and Currant became dominated by cheatgrass, whereas cheatgrass was only a minor component at Elkhorn 1 and Elkhorn 2. Plant communities in the adjacent untreated areas were very similar (Figure 1) with only insignificant differences in vegetation cover types among all projects (Table 3).

**Table 0-2.** Resilience to disturbance and resistance to invasive annual grasses score for Cathedral, Currant, Elkhorn 1, and Elkhorn 2.

	<b>Cathedral</b>	<b>Currant</b>	<b>Elkhorn 1</b>	<b>Elkhorn 2</b>
<b>Temperature score</b>	4.5	4	5	4.5
<b>Moisture score</b>	7	7.5	8	6
<b>Pre-treatment vegetation score</b>	1.2	1.2	1.2	0
<b>Total resilience and resistance rating</b>	12.7 (low)	12.7 (low)	14.2 (low)	10.5 (low)

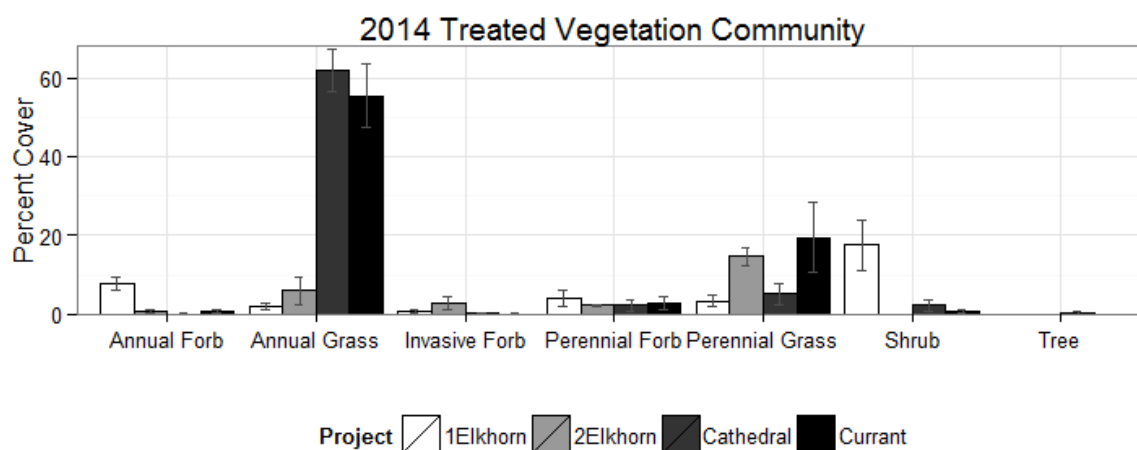


**Figure 1.** 2014 Vegetation type cover in untreated areas. Error bars show standard error.

**Table 0-3.** ANOVA test results for differences in vegetation cover classes among projects. DF = degrees of freedom (between groups, within groups), F = F test statistic, p = probability that test statistic could be obtained by chance. Bold values indicate significant differences of at the 10% confidence level.

	Untreated			Treated		
	DF	F	p	DF	F	p
Annual Forb	3, 20	1.791	0.181	3, 23	23.62	<b>&lt;0.001</b>
Annual Grass	3, 20	2.089	0.134	3, 23	37.85	<b>&lt;0.001</b>
Perennial Grass	3, 20	1.298	0.303	3, 23	2.571	<b>0.079</b>
Perennial Forb	3, 20	1.791	0.181	3, 23	0.343	0.795
Invasive Forb	3, 20	1.000	0.413	3, 23	1.462	0.251
Shrub	3, 20	2.361	0.102	3, 23	10.13	<b>&lt;0.001</b>
Tree	3, 20	0.156	0.925	3, 23	1.789	0.177

Despite similar vegetation in the untreated areas, the plant communities in the treated areas differed widely. Cathedral and Currant were strongly dominated by annual grasses which were only a minor component at Elkhorn 1 and 2 (Figure 2). There were significant differences among the cover of annual herbs, annual grasses, perennial grasses, and shrubs in the treated areas (Table 3). Currant and Elkhorn 2 had significantly more perennial grasses than Cathedral or Elkhorn 1. Elkhorn 1 had significantly more shrubs and annual forbs than any of the other project.



**Figure 2.** 2014 Vegetation type cover in treated areas. Error bars show standard error.

There were moderate correlations between some of the vegetation types at the untreated plots when compared with the treated plots. The cover of annual grasses and perennial grasses in the untreated areas were each correlated with their cover in the treated areas. The cover of perennial herbs in the treated area was moderately correlated with the cover of trees in the untreated area. However, the Pearson's correlations were misleading due to non-normal data and the presence of some values with large leverage. These large leverage values were not eliminated as outliers because they accurately represented the condition of vegetation on the ground and were within 3 standard deviation of the mean. To better understand the relationships within the data, a multivariate technique not sensitive to normality was used for evaluation.

The NMS provided a three-axis solution that was found to explain 96.6% of the variance (Table 4). The three axes were orthogonal with respect to each other. The best solution had a low stress of 7.1 which indicates a good ordination with little risk of drawing false inferences (Clarke 1993). Over 250 runs, the Monte Carlo test found a mean stress for randomized data was 12.7 and only 4.4% of the random runs has a stress equal to or lower than the best solution.

**Table 0-4.** Proportion of variance explained by ordination axes.

Axis	<u>R<sup>2</sup> value</u>	
	Increment	Cumulative
1	0.644	0.644
2	0.206	0.850
3	0.115	0.966

The primary axis was highly anti-correlated with tree cover and correlated with annual grass cover. The secondary axis was highly correlated with annual grass cover and

anti-correlated with invasive forbs. The tertiary axis was highly anti-correlated with shrubs and annual forbs (Table 5).

**Table 0-5.** Pearson's correlation with ordination axes for lifeform classes. Correlation with an  $R > 0.5$  ( $R^2 > 0.25$ ) shown in bold.

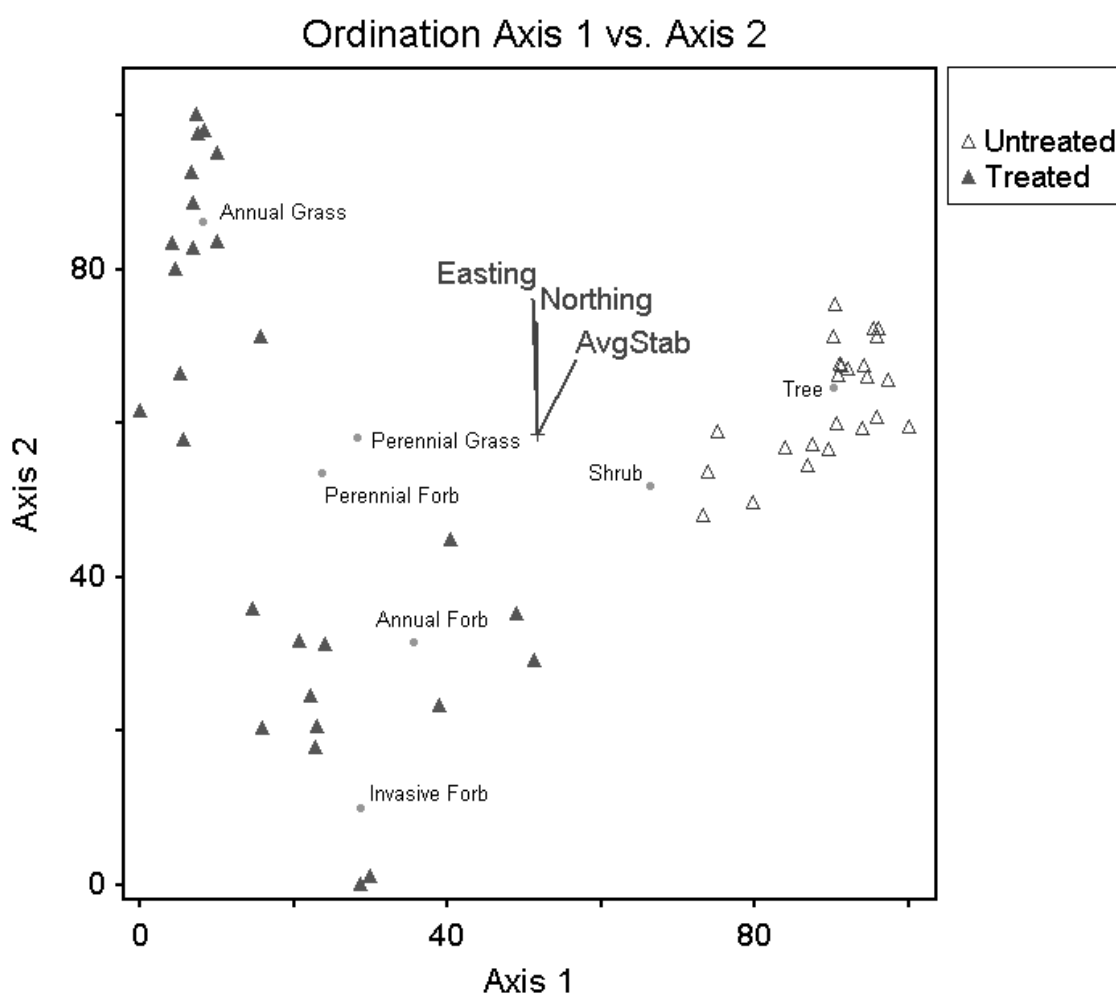
	Axis 1		Axis 2		Axis 3	
	R	R <sup>2</sup>	R	R <sup>2</sup>	R	R <sup>2</sup>
Annual Grass	<b>-0.682</b>	<b>0.465</b>	<b>0.674</b>	<b>0.454</b>	0.081	0.007
Annual Forb	-0.177	0.031	-0.462	0.214	<b>0.601</b>	<b>0.359</b>
Invasive Forb	-0.172	0.030	<b>-0.566</b>	<b>0.321</b>	0.058	0.003
Perennial Forb	-0.464	0.215	-0.129	0.017	0.112	0.012
Perennial Grass	-0.460	0.211	-0.010	0.000	-0.442	0.195
Shrub	0.251	0.063	-0.179	0.032	<b>0.682</b>	<b>0.465</b>
Tree	<b>0.901</b>	<b>0.812</b>	0.220	0.049	-0.205	0.043

For the environmental and soil variables, the primary axis was moderately correlated with average stability and subsurface stability. The secondary axis was highly correlated with easting and northing. The tertiary axis was highly anti-correlated with precipitation (Table 6).

**Table 0-6.** Pearson's correlation with ordination axes for environmental and soil variables. Correlation with an  $R > 0.5$  ( $R^2 > 0.25$ ) shown in bold.

	Axis 1		Axis 2		Axis 3	
	R	R <sup>2</sup>	R	R <sup>2</sup>	R	R <sup>2</sup>
Easting	-0.121	0.015	<b>0.594</b>	<b>0.353</b>	0.026	0.001
Northing	-0.079	0.006	<b>0.543</b>	<b>0.295</b>	0.122	0.015
Precipitation	0.056	0.003	0.445	0.198	<b>0.518</b>	<b>0.268</b>
Elevation	0.146	0.021	0.055	0.003	0.386	0.149
Aspect	0.223	0.050	-0.096	0.009	0.216	0.047
Valley Slope	0.025	0.001	0.425	0.180	-0.200	0.040
Side Hill Slope	-0.070	0.005	0.165	0.027	0.093	0.009
Subsurface Stability	0.312	0.097	0.320	0.103	-0.256	0.066
Surface Stability	0.212	0.045	0.405	0.164	0.184	0.034
Average Stability	0.319	0.102	0.441	0.194	-0.045	0.002
Interspace Infiltration	0.056	0.003	0.403	0.162	-0.225	0.051

After ordination we grouped plots using overlays based on treatment type, project, and parent material. There was a large difference in treated versus untreated plots. The ordination separated out these groups along the primary axis and there was no overlap of groups within ordination space (Figure 3). The untreated plots are all tightly clustered showing relatively little difference among the untreated plant communities. The treated plots are spread widely across ordination space due to a large degree of variability in plot responses to the treatment.



**Figure 3.** Ordination axis 1 versus axis 2 grouped by treatment type.



There is not a large difference among projects in ordination space (Figure 4). Control plots on the right overlap, showing that project was not a good predictor of untreated vegetation. Treated plots on the left show that Cathedral and Currant have large overlap and similar responses. Similarly, treated plots at Elkhorn 1 and Elkhorn 2 overlap showing similar responses. However the two groups, Cathedral and Currant versus Elkhorn 1 and 2 barely overlap with only two Elkhorn 2 plots farther down axis 1 and up axis 2.

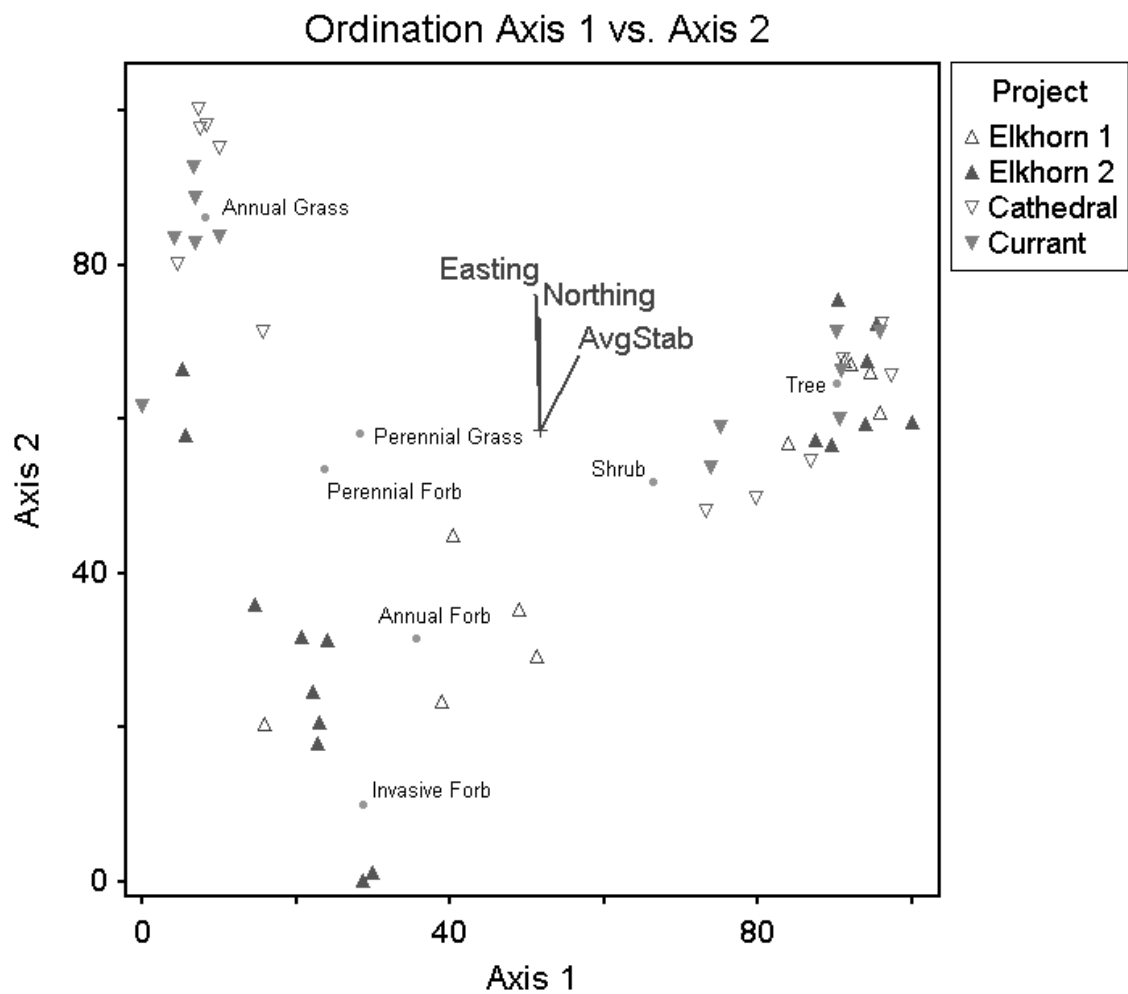
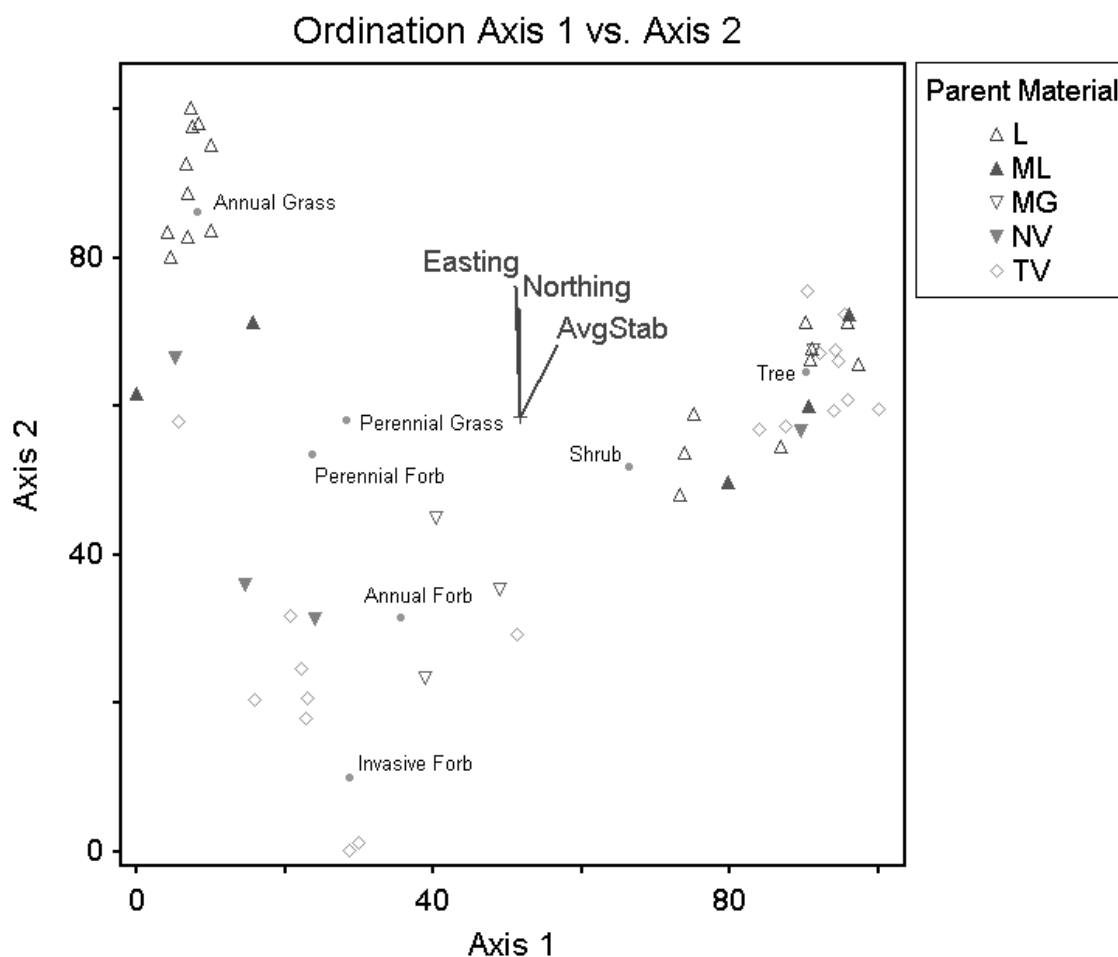


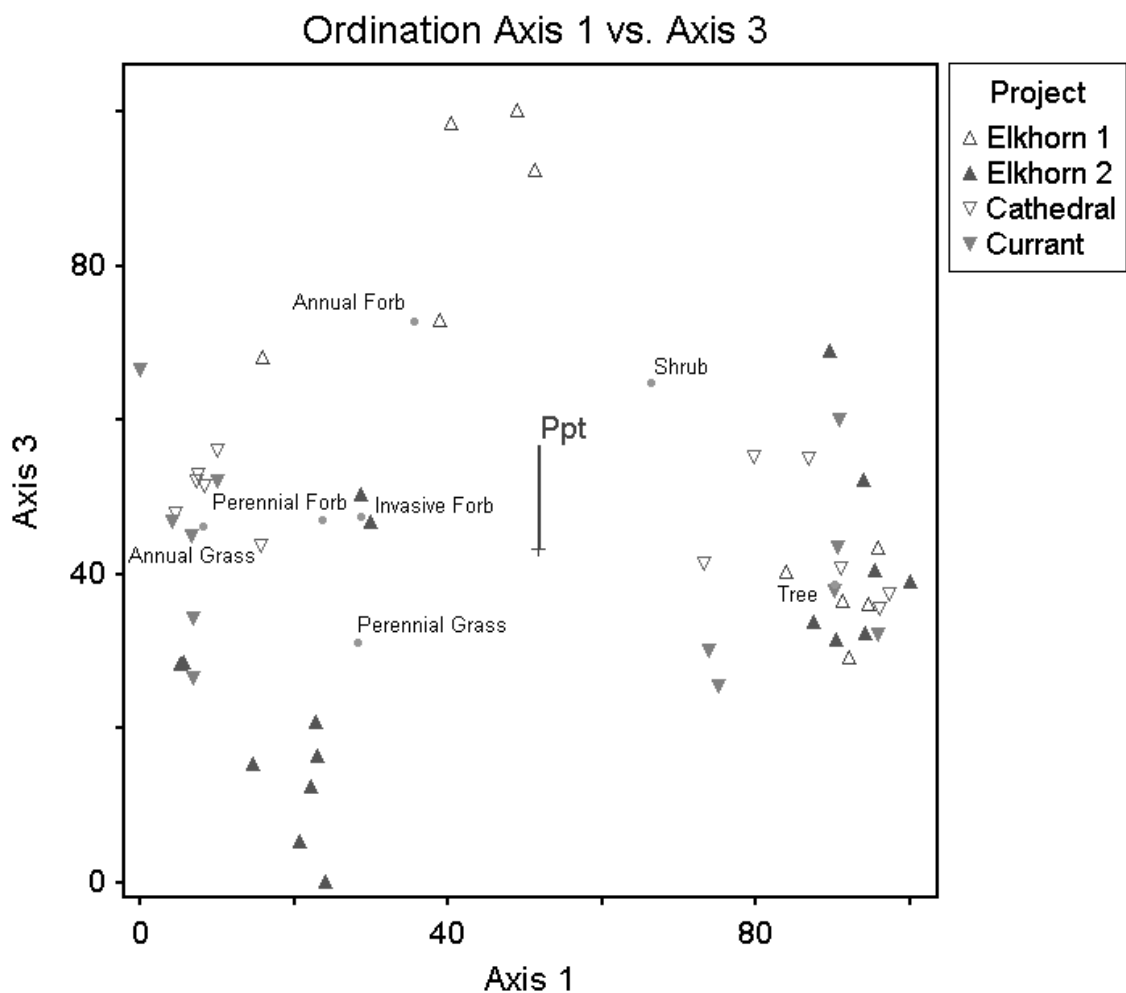
Figure 4. Ordination axis 1 versus axis 2 grouped by project.

When grouped by parent material instead of project, there is a much more obvious relationship between the treated plots and the responses of plant communities to treatment (Figure 5). The plots on soils derived from limestone are tightly grouped in one corner around high annual grass cover. The plots on soils derived from limestone mixed with volcanics have slightly less annual grass cover. The plots on soils derived from non-tuffaceous volcanics or mixed granite and volcanics have low annual grass cover. The plots on soils derived from tuffaceous volcanics have the lowest annual grass cover. Parent material is a better predictor of the post-fire response than project when the pre-fire plant community is similar.



**Figure 5.** Ordination axis 1 versus axis 2 grouped by parent material. L=limestone. ML=mixed limestone and volcanic, MG = mixed granite and volcanics, NV=non-tuffaceous volcanics, TV=tuffaceous volcanics.

Axis 3 of the ordination does a good job as separating treated plots at Elkhorn 1 from Elkhorn 2 though there is still some overlap (Figure 6). Elkhorn 1 was at a higher elevation, with more precipitation, and was more dominated by shrubs. Elkhorn 2 was more dominated by perennial grasses. Axis 3 was correlated most strongly with precipitation. The project with highest precipitation had more shrubs and the project with lowest precipitation had more grasses.



**Figure 6.** Ordination axis 1 versus axis 3 grouped by project.

The best GLM used parent material factorized as a predictor variable and treated annual grass cover as the response variable (Table 7). The McFadden's pseudo- $R^2$  value for the best more was 0.865 indicating that the model explained 86.5% of the deviance within the dataset. The beta coefficients show a positive relationship between plots derived from limestone as shown by the intercept term. The relationship was weakened in soils derived from mixed limestone and weakened further in soils derived from non-tuffaceous volcanics. Plots on soils derived from mixed other and non-tuffaceous

volcanics show the least increase in annual grass cover after treatment, but still have a positive relationship overall indicating that treated areas still have more annual grasses than untreated areas even when post-treatment cheatgrass is low. Attempts were made to construct a GLM that used other predictor variables normally recommend as important including untreated annual grass cover or untreated perennial grass cover, but these were not nearly as good predictors as parent material.

**Table 0-7.** Best generalized linear model summary. tAG = treated annual grass cover. PM = parent material, PM.ML = mixed limestone, PM.MG – mixed granite and volcanics, PM.NV = non-tuffaceous volcanics, PM.TV = tuffaceous volcanics.

Formula	<i>Round(tAG) ~ PM factorized</i>	
Beta Coefficients	(Intercept)	4.162
	PM.ML	-0.761
	PM.MG	-3.651
	PM.NV	-1.677
	PM.TV	-3.246
Null Deviance	843.3	
Residual Deviance	113.9	

## Discussion

The four burned areas had similar pre-fire vegetation, elevation, and precipitation; but two vastly different post-fire responses. Only Currant and Cathedral became dominated by cheatgrass. Cheatgrass was present and had the opportunity to establish at all projects. Differences in disturbance response can be best understood by grouping plots by parent material. The importance of ash content in soil has been know and observed by soil scientists (Soil Survey Staff 2015), but there has not been quantitate studies showing the differential response of cheatgrass to soils with and without ash. Plots with similar parent materials had similar disturbance responses.

The primary ordination axis is the treatment versus control axis. It separates the plots by their tree cover which creates a large separation between the untreated plots with high tree cover and the treated plots with very low tree cover. The axis also separates the treated plots based on tree cover remaining after the burn. The areas where tree cover was not completely removed may indicate areas of lower fire severity (Keeley 2009).

The secondary axis is the cheatgrass axis. It separates the plots by the amount of cheatgrass present post-treatment. This creates two clear response groups among the treated plots— the Elkhorn burns had very little cheatgrass and the Cathedral and Currant burns were dominated by cheatgrass. The untreated plots show relatively little variation here. All of the untreated plots had less than 1% cheatgrass. The secondary axis is correlated with easting and northing, but this is believed to be merely a relic of the fact that the White Pine Range formed mostly from limestone happened to be northeast of the Monitor Range where the soils were derived from welded tuff and granite. The tertiary axis is the shrub versus grass axis and was also correlated with precipitation. The areas with greater precipitation had more shrub cover in the post-fire community. The tertiary axis separates the native herbaceous dominated community, Elkhorn 2, from the native shrub dominated community, Elkhorn 1.

When grouped by soil parent material, a pattern emerges within ordination space. The plots on soils derived primarily from limestone were the soils that had the greatest amount of cheatgrass. The plots derived partially from limestone also had a large amount of cheatgrass. The plots on soils derived from volcanic and metamorphic rocks had lower cheatgrass. The plots on soils derived from tuffaceous volcanics had the lowest

cheatgrass. Grouping the plots by parent material created better response groups than grouping the plots by project location.

The GLM confirmed the results of the NMS and showed the importance of parent material for predicting to the cover of cheatgrass after fire. Using only the parent material as a predictor, 86.5% of the variance within the dataset was explained. The parent material is by far the most important factor that can be used to predict the differences in cheatgrass among the treated areas. Parent material was a better predictor than any other variable in the dataset.

Several possibilities could explain the mechanism for why cheatgrass behaves differently in soils derived from different parent material. Soils derived from tuff have a greater ash content than soils derived from other rocks (Soil Survey Staff 2015). The ash content of the soil increases the soil's moisture retention at field capacity due to differences in irregular surface area of ash particles (Kalra et al. 2000). This makes tuffaceous soils behave as if they are in a higher precipitation zone because they retain more moisture. This allows perennial grasses to access more soil water later in the growing season to better establish themselves following fire and resist cheatgrass invasion. Cheatgrass does not do well in areas that receive >400 mm of precipitation (Platt and Jackman 1946). This is mostly due to being outcompeted by native perennial grasses when more soil water is available throughout the year (Chambers et al. 2007). With tuffaceous soil types capturing precipitation and making more water available, perennials thrive and provide resistance to cheatgrass. Soils derived from tuff in the 250-350 mm precipitation range will not as readily convert to cheatgrass following fire as soils derived from limestone or non-tuffaceous volcanics. These effects may not occur in

regions with less precipitation where cheatgrass is better adapted to surviving. The soils also had different textures, physical, and chemical properties. Further investigation of soil samples could determine which physical or chemical soil properties are most important for cheatgrass resistance in tuffaceous soils. A difference in texture (Veihmeyer and Hendrickson 1931), pH (Reitemeier 1946), or phosphorous (He et al. 2002) could be related to the difference in moisture holding capacity.

## **Management Implications**

Aldo Leopold (1949) warned of simply accepting cheatgrass in “A Sand County Almanac” His observation is still relevant today. Cheatgrass eradication from the 4 million ha it dominates in the Great Basin (Bradley and Mustard 2008), is not the accepted goal. Cheatgrass is not on the Nevada noxious weed list (Nevada Department of Agriculture 2015) as it is too widespread to control. However, treatments should be done in a way to prevent, reduce, or control cheatgrass dominance when possible.

Cheatgrass only does well in semi-arid precipitation zones and does not do well where precipitation is less than 150 mm or more than 400 mm (Platt and Jackman 1946). The timing of the precipitation is important as well. Cheatgrass prefers spring precipitation (Bradley 2009). Cheatgrass is less likely to compete with native vegetation outside of its preferred precipitation regime (Loik 2007). Tuffaceous soils make the soils effectively act like they are deeper and wetter than they actually are. Also, despite more than a century since it became widespread in the Western United States (Novak and Mack 2001), cheatgrass has not totally replaced the native vegetation in the Great Basin. We still have tens of millions of acres of Pinyon-juniper and sagebrush that have not



converted to cheatgrass. Even fires that burn in areas with nearby cheatgrass seed sources and similar pre-fire vegetation do not always convert to cheatgrass as these results show. There are millions of acres in the Great Basin where cheatgrass has not and/or will not grow. It is important to identify these areas so that treatments can be targeted in areas where cheatgrass invasion is not likely to occur.

It is easy to focus solely on pre-treatment vegetation to describe a project because it is easily visible. However, the soils underneath the Pinyon-juniper have a wealth of diversity that impacts the disturbance response characteristics of the area. Understanding the soils is a critical point when planning restoration treatments. Treatments should ideally be planned on a per-ecological site basis. Yet, it is easy to accidentally cross into significantly different soil types without some pre-treatment soil identification.

Identifying down to soil series or ecological site often requires a field visit and inventory because a single soil mapping unit contains multiple soils series and ecological sites.

Parent material is a more broad classification that can often be identified with a good degree of certainty without a field visit by referencing currently existing Level 3 NRCS Soil Surveys.

When prioritizing treatments, land managers should consider the type of soils especially noting differences in parent material and precipitation regime. Areas that are less susceptible to cheatgrass invasion should be targeted when planning prescribed fires. Areas known to be susceptible to cheatgrass invasion should be actively protected from fire once they have crossed an ecological threshold. In those areas, managers should

implement practices that seek to promote shrubs dominance and fire resistant communities using tools other than fire (Chambers et al 2015).

The failure of the score sheet (Miller et al. 2014) to predict the response of these treatments shows that sometimes the best pre-treatment planning is not sufficient to ensure a desirable response. Monitoring is critical to understanding effectiveness of Pinyon-juniper removal treatments as the climate changes. Monitoring is the main source of information to inform present and future land managers about the long term effects of their actions and how to best achieve their management goals. While we already know a great deal about what impacts the outcome of land treatments, these results showed there are still lessons to be learned.

## Literature Cited

Allen, E. A., & R. S. Nowak. 2008. Effect of pinyon-juniper tree cover on the soil seed bank. *Rangeland Ecology & Management*, 61(1), 63-73.

Beckstead, J., & C. K. Augspurger. 2004. An experimental test of resistance to cheatgrass invasion: limiting resources at different life stages. *Biological Invasions*, 6(4), 417-432.

Blank, R. R., & T. Morgan. 2013. Soil Engineering Facilitates Downy Brome (*Bromus tectorum*) Growth-A Case Study. *Invasive Plant Science and Management*, 6(3), 391-400.

Bradley, B. A. 2009. Regional analysis of the impacts of climate change on cheatgrass invasion shows potential risk and opportunity. *Global Change Biology*, 15(1), 196-208.

Bradley, B. A., & J. F. Mustard. 2008. Comparison of phenology trends by land cover class: a case study in the Great Basin, USA. *Global Change Biology*, 14(2), 334-346.

Condon, L., P. J. Weisberg, & J. C. Chambers. 2011. Abiotic and biotic influences on *Bromus tectorum* invasion and *Artemisia tridentata* recovery after fire. *International Journal of Wildland Fire*, 20(4), 597-604.

- Coulloudon, B. 1999. *Sampling vegetation attributes: interagency technical reference*. Bureau of Land Management.
- Chambers, J. C., D. A. Pyke, J. D. Maestas, M. Pellant, C. S. Boyd, S. B. Campbell,... & A. Wuenschel. 2014. Using resistance and resilience concepts to reduce impacts of invasive annual grasses and altered fire regimes on the sagebrush ecosystem and greater sage-grouse: A strategic multi-scale approach.
- Chambers, J. C., B. A. Roundy, R. R. Blank, S. E. Meyer, & A. Whittaker. 2007. What makes Great Basin sagebrush ecosystems invasible by *Bromus tectorum*? *Ecological Monographs*, 77(1), 117-145.
- Clarke, K. R. 1993. Non-parametric multivariate analyses of changes in community structure. *Australian Journal of Ecology*, 18(1), 117-143.
- D'Antonio, C. M., & P. M. Vitousek. 1992. Biological invasions by exotic grasses, the grass/fire cycle, and global change. *Annual Review of Ecology and Systematics*, 63-87.
- Fisher, R. A. 1918. Studies in Crop Variation. I. An examination of the yield of dressed grain from Broadbalk. *Journal of Agricultural Science*, 11: 107-135.
- Gabriel, K. R. 1971. The biplot graphic display of matrices with application to principal component analysis. *Biometrika*, 58(3), 453-467.
- Gruell, G. E. 1999. Historical and modern roles of fire in pinyon-juniper. *Monsen, SB, Stevens, R.(Comps.), Proceedings: Ecology and Management of Pinyon–Juniper Communities in the Interior West. Proceedings RMRS-P-9. US Department of Agriculture Forest Service, Rocky Mountain Research Station, Ogden, UT*, 24-28.
- Gruell, G. E., & S. Swanson. 2012. *Nevada's changing wildlife habitat: an ecological history*. University of Nevada Press.
- He, Y. Q., Y. G. Zhu, S. E. Smith, & F. A. Smith. 2002. Interactions between soil moisture content and phosphorus supply in spring wheat plants grown in pot culture.
- Herrick, J. E., J. V. Zee, K. M. Havstad, L. M. Burkett, & W. G. Whitford. 2009. Monitoring manual for grassland, shrubland and savanna ecosystems. USDA-ARS Jornada Experimental Range. <http://jornada.nmsu.edu/monit-assess/manuals/monitoring> Accessed 22 March 2013.
- Kalra, N., R. C. Harit, & S. K Sharma. 2000. Effect of flyash incorporation on soil properties of texturally variant soils. *Bioresource Technology*, 75(1), 91-93.
- Keeley, J. E. 2009. Fire intensity, fire severity and burn severity: a brief review and suggested usage. *International Journal of Wildland Fire*, 18(1), 116-126.

- Kitchen, S. G. 2014. Learning to Live With Cheatgrass: Giving Up or a Necessary Paradigm Shift?. *Rangelands*, 36(2), 32-36.
- Knapp, P. A. 1996. Cheatgrass (*Bromus tectorum* L) dominance in the Great Basin Desert: history, persistence, and influences to human activities. *Global Environmental Change*, 6(1), 37-52.
- Kruskal, J. B. 1964. Nonmetric multidimensional scaling: a numerical method. *Psychometrika*, 29(2), 115-129.
- Loftin, S. R. 1999. Initial response of soil and understory vegetation to a simulated fuelwood cut of a pinyon-juniper woodland in the Santa Fe National Forest. *Proceedings: Ecology and Management of Pinyon-Juniper Communities within the Interior West. USDA Forest Service Proc. RMRS-P-9. Ogden, UT*, 311-314
- Loik, M. E. 2007. Sensitivity of water relations and photosynthesis to summer precipitation pulses for *Artemisia tridentata* and *Purshia tridentata*. *Plant Ecology*, 191(1), 95-108.
- Lowery, B., W. J. Hickey, M. A. Arshad, & R. Lal. 1996. Soil water parameters and soil quality. *Methods for assessing soil quality*, 143-155.
- Mather, P. M. 1976. *Computational methods of multivariate analysis in physical geography* (p. 532). London: Wiley.
- McCullagh, P. 1984. Generalized linear models. *European Journal of Operational Research*, 16(3), 2
- McCune, B. and M. J. Mefford. 2011. PC-ORD. Multivariate Analysis of Ecological Data. Version 6.0 MjM Software, Glenden Beach, Oregon, U.S.A.
- McCune, B., Grace, J. B., & Urban, D. L. 2002. *Analysis of ecological communities* (Vol. 28). Glenden Beach, Oregon: MjM software design.
- McFadden, D. 1973. Conditional logit analysis of qualitative choice behavior, in Zarembka, P. (ed.), *Frontiers in Econometrics*, pp.105-142, Academic Press, New York.
- Metropolis, N., & S. Ulam. 1949. The monte carlo method. *Journal of the American Statistical Association*, 44(247), 335-341.
- Miller, M. E., M. A. Bowker, R. L. Reynolds, & H. L. Goldstein. 2012. Post-fire land treatments and wind erosion—lessons from the Milford Flat Fire, UT, USA. *Aeolian Research*, 7, 29-44.

- Miller, R. F., J. C. Chambers, & M. Pellant. 2014. A field guide for selecting the most appropriate treatment in sagebrush and piñon-juniper ecosystems in the Great Basin: Evaluating resilience to disturbance and resistance to invasive annual grasses, and predicting vegetation response. Gen. Tech. Rep. RMRS-GTR-322-rev. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 66 p.
- Miller, R. F., & R. J. Tausch. 2001. The role of fire in pinyon and juniper woodlands: a descriptive analysis. In Proceedings of the invasive species workshop: the role of fire in the control and spread of invasive species. Tall Timbers Research Station Miscellaneous Publication (Vol. 11, wpp. 15-30).
- Miller, R. F., R. J. Tausch, E. D. McArthur, D. D. Johnson, & S. C. Sanderson. 2008. Age structure and expansion of piñon-juniper woodlands: a regional perspective in the Intermountain West.
- Nevada Department of Agriculture. 2015. Nevada Noxious Weed List. State of Nevada. [http://agri.nv.gov/Plant/Noxious\\_Weeds/Noxious\\_Weed\\_List/](http://agri.nv.gov/Plant/Noxious_Weeds/Noxious_Weed_List/) Accessed 20 April 2015
- Norton, J. B., T. A. Monaco, J. M. Norton, D. A. Johnson, & T. A. Jones. 2004. Soil morphology and organic matter dynamics under cheatgrass and sagebrush-steppe plant communities. *Journal of Arid Environments*, 57(4), 445-466.
- Novak, S. J., & R. N. Mack. 2001. Tracing Plant Introduction and Spread: Genetic Evidence from *Bromus tectorum* (Cheatgrass) Introductions of the invasive grass *Bromus tectorum* worldwide were broadly similar and closely tied to patterns of European human immigration. *Bioscience*, 51(2), 114-122.
- Pearson, E. S. 1931. The test of significance for the correlation coefficient. *Journal of the American Statistical Association*, 26(174), 128-134.
- Platt, K., & E. R. Jackman. 1946. *The cheatgrass problem in Oregon*. Corvallis, Or.: Federal Cooperative Extension Service, Oregon State College.
- PRISM Climate Group. 2013. Oregon State University, <http://prism.oregonstate.edu> Accessed 13 March 2013.
- Provencher, L., & J. Thompson. 2014. Vegetation Responses to Pinyon-Juniper Treatments in Eastern Nevada. *Rangeland Ecology and Management*, 67(2), 195-205.
- R Core Team. 2014. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. <http://www.R-project.org/>. Accessed 9 February 2015.

- Reitemeier, R. F. (1946). Effect of Moisture Content on the Dissolved and Exchangeable Ions of Soils of Arid Regions. *Soil Science*, 61(3), 195-214.
- Simpson, E. H. 1949. Measurement of diversity. *Nature*.
- Soil Survey Staff. 1999. *Soil taxonomy: A basic system of soil classification for making and interpreting soil surveys*. 2nd edition. Natural Resources Conservation Service. U.S. Department of Agriculture Handbook 436.
- Soil Survey Staff. 2015. Natural Resources Conservation Service, United States Department of Agriculture. Web Soil Survey. Available online at <http://websoilsurvey.nrcs.usda.gov/>. Accessed 15 March 2015.
- Sørensen, T. 1948. A method of establishing groups of equal amplitude in plant sociology based on similarity of species and its application to analyses of the vegetation on Danish commons. *Biol. Skr.*, 5, 1-34.
- Svejcar, T., C. Boyd, K. Davies, M. Madsen, J. Bates, R. Sheley,... & K. Havstad. 2014. Western land managers will need all available tools for adapting to climate change, including grazing: A critique of Beschta et al. *Environmental management*, 53(6), 1035-1038.
- Tausch, R. J. 1999. Historic pinyon and juniper woodland development. *Proceedings: ecology and management of pinyon-juniper communities within the Interior West. Ogden, UT, USA: US Department of Agriculture, Forest Service, Rocky Mountain Research Station, RMRS-P-9*, 12-19.
- Veihmeyer, F. J., & A. H. Hendrickson. 1931. The moisture equivalent as a measure of the field capacity of soils. *Soil Science*, 32(3), 181-194.
- West, N. E. 1999. 18 Juniper-Pinyon Savannas and Woodlands of Western North America.
- Young, J. A., & C. D. Clements. 2009. *Cheatgrass: fire and forage on the range*. University of Nevada Press.
- Young, J. A., & R. A. Evans. 1973. Downy brome: intruder in the plant succession of big sagebrush communities in the Great Basin. *Journal of Range Management*, 410-415.
- Young, J. A., R. A. Evans, & J. Major. 1972. Alien plants in the Great Basin. *Journal of Range Management*, 194-201.
- Zuur, A., E. N. Ieno, N. Walker, A. A. Saveliev, & G. M. Smith. 2009. *Mixed effects models and extensions in ecology with R*. Springer Science & Business Media.

## **Chapter 2: The effects of hand thinning in Pinyon Juniper forests.**

### **Abstract**

Land managers sometimes want to remove Pinyon-juniper on a site without significantly impacting the existing understory vegetation. Hand thinning is used as a treatment to selectively lop and scatter Pinyon-juniper trees. We established 12 plots (six treated, six untreated) to monitor the effects of a recent hand thinning in White Pine County, Nevada on the vegetation and soils at the site to determine the level of impacts caused by hand thinning. The hand thinning significantly reduced tree cover [ $F(1,10) = 7.43, p = 0.027$ ] to less than 2%. Perennial grasses on the site are slightly higher in the treated area. There was a significant increase in perennial grass cover from 2013 to 2014 [ $F(1,10) = 14.5, p = 0.003$ ]. The hand thinning did not have significant effects on shrubs, annual grasses, annual forbs, perennial forbs, ground cover, stability, species richness, diversity, infiltration, or gap structure. Because hand thinning does not remove the shrubs or other perennials, site resistance can be maintained. With sufficient understory vegetation to maintain resistance post treatment (as in phase I or early phase II Pinyon-juniper woodlands), nonnative annual grasses are less likely to dominate after treatment.

### **Introduction**

Pinyon-juniper trees have increased their area of domination over the last 130 years from covering 3 million ha to 30 million ha (Miller and Tausch 2001). As Pinyon-juniper cover increases, exposure of the soil surface increases because of reduced density of understory species and surface litter (Pierson et al. 2007). Hand thinning is one method

of mechanical Pinyon-juniper tree removal that can remove trees while leaving the majority of the understory community undisturbed. Over time, it increases total herbaceous vegetation (Bates, Miller, and Svejcar 2005). Removing Pinyon-juniper increases soil water availability (Roundy et al. 2014b). Manually harvesting trees is relatively labor intensive and expensive (Young et al. 1985), but remains a useful treatment in areas where other treatments are inappropriate or preserving the shrub community is a concern.

Hand thinning occurred long before people thought of it as a land management treatment. It is similar to cutting trees for wood harvest. Indigenous people used Pinyon-juniper wood products as the primary material for constructing shelters (Janetski 1999). After European settlement of the Great Basin, large areas around mining sites were deforested to supply wood for charcoal production, buildings, and fence posts (Young and Budy 1979). After the initial cut, others would return and dig out stumps when wood became scarce (Young and Budy 1979). As conifer encroachment became recognized as a threat to productivity (Tausch et al. 1981, Tueller et al. 1979), land managers began cutting trees for removal.

Hand thinning causes less soil disturbance than chaining (Loftin 1999) and is appropriate in sage grouse areas to remove Pinyon-juniper. Sage grouse avoid areas with more than 1% Pinyon-juniper cover and almost entirely stay out of areas with more than 4% Pinyon-juniper cover (Baruch-Mordo et al. 2013). Lop and scatter, with cut trees left on the site, can reduce pinyon-juniper cover to less than 1% (Provencher and Thompson



2014). Lop and scatter preserves biological crusts better than bulldozing, lop pile burn, feller-buncher, or mastication. (Provencher and Thompson 2014).

The moderate to long term effects of hand thinning include increases in total understory biomass, cover, seed production, and diversity (Bates et al. 2000, Bates 2005). After hand thinning, areas previously under Pinyon-juniper canopy respond differently than interspaces. Bates et al. (1998) found bluegrass (*Poa* spp.), bottlebrush-squirreltail (*Elymus elymoides*), and annual forbs were greatest in duff zones formerly under Pinyon-juniper canopy. Density of other perennial grasses and perennial forbs was greatest in the interspaces. Hand thinning reduces tree cover and bare soil (Loftin 1999) and increases total vegetation cover on sites not subjected to severe post-treatment livestock grazing (Everett and Sharrow 1985). By 10 years after treatment, cut sites have more perennial grasses, improved infiltration capacity, and less bare ground (Pierson et al. 2007). In comparison with other mechanical Pinyon-juniper removal treatments, cutting promotes sagebrush dominance more than burning or pile burning (O'Conner et al. 2013). Ross (2012) found that hand thinning significantly increased understory cover and was more effective than mastication treatments, but also causes an increase in cheatgrass (*Bromus tectorum*), but Roundy et al. (2014b) found hand thinning and mastication to produce similar increases in total perennial herbaceous and cheatgrass cover. Unlike chaining, aggregate stability remains high after clear cutting (Mohr et al. 2013).

## Methods

The study was located on the east side of the White Pine Range on (latitude 38°54'53" N to 38°49'52" N north to south; longitude 115°16'16" W to 115°14'41" W

west to east) on an alluvial fan slope about 50 km southwest of Ely, Nevada. Elevation ranges from 1945 to 1981 m. Average annual precipitation is 280 to 290 mm. Hill slope ranges from 1% to 7% with an average slope of 3%. The area surrounding the hand thinning is dominated by phase II Pinyon-juniper trees according the classification system of Miller and Tausch (2001). The cover of trees in the adjacent untreated areas is about the same as the cover of shrubs. Shrubs and trees are codominant, but the shrubs are much more dense and frequent. The area is mostly on the R028BY007NV — LOAMY 10-12 P.Z. ecological site with some plots also in the R028BY086NV — GRAVELLY CLAY 10-12 P.Z. ecological site. Both ecological sites are in disturbance response group 3B (Stringham et al. 2015). The adjacent untreated area is currently in the current potential phase 2.3 (at risk) with increasing Pinyon-juniper, according to the state and transition models for that disturbance response group. The treated area was dominated by black sagebrush (*Artemisia nova*) and native perennial grasses with some Wyoming big sagebrush (*Artemisia tridentata* spp. *wyomingensis*) also present. Other common shrubs in the area were green rabbitbrush (*Chrysothamnus viscidiflorus*), green ephedra (*Ephedra viridis*), and granite prickly phlox (*Linanthus pungens*). The most common perennial grasses were Indian ricegrass (*Achnatherum hymenoides*), bottlebrush squirreltail (*Elymus elymoides*), needle-and-thread grass (*Hesperostipa comata*), James' galleta (*Pleuraphis jamesii*), and Sandberg bluegrass (*Poa secunda*). Juniper trees made up 85-95% of the tree cover and tree frequency in the untreated areas. Several areas also had patches of the upland Douglas' sedge (*Carex douglasii*). The soils in the area are mesic, shallow, Xeric Haplodurids derived from limestone or limestone mixed with volcanics.

The treated 920 ha area was hand thinned using a lop and scatter method from 2009 to 2012. Lop and scatter consists of removing the major branches, felling trees, and leaving the biomass scattered on the site. We established 12 plots including six plots in the treated area and six paired plots in adjacent untreated areas. Polygons were created so plots could be paired in areas that shared the similar precipitation, soil association, slope, aspect, and elevation. Plots were then randomly located within the stratified polygons. The sampling plots consisted of three 20 m long parallel transects spaced 5 m apart.

To characterize vegetation, we collected data on canopy cover, ground cover, nested frequency, perennial density, and canopy gap in summers of 2013 and 2014. We recorded line point-intercept canopy cover and ground cover (Herrick et al. 2009) using a laser point projection device every 20 cm along three 20 m long parallel transects spaced 5 m apart with 100 points per transect and 300 points per plot. Frequency (Caulloudon 1999) was collected using a 1 m<sup>2</sup> nested frequency frame placed at 1 m intervals along the uphill side of each transect (20 frames/transect and 60 frames per plot). The nested frequency frame was subdivided into 4 sections with areas of 1 m<sup>2</sup>, 0.5 m<sup>2</sup>, 0.25 m<sup>2</sup> and 0.04 m<sup>2</sup>. Perennial density was recorded as a total count of all perennial plants rooted within 1 m on either side of the transect (40 m<sup>2</sup> per transect and 120m<sup>2</sup> per plot). Canopy gap intercept (Herrick et al. 2009) was recorded along each transect with a minimum gap size of 20 cm. To characterize soils, we collected data on soil aggregate stability, infiltration, and taxonomy in 2014. An interspace soil pit was described within 15 m behind the start of each transect. The soil was classified to soil series (Soil Survey Staff 1999). Soil aggregate stability (Herrick et al. 2009). was tested along the first transect on the side opposite frequency and cover We collected nine samples, each at 2 m intervals

from 2 m to 18 m. At each sampling point, one sample was collected from the surface and another from 2-5 cm directly below. Each sample point was assigned to a canopy class based on vegetation type: tree, shrub, perennial grass, perennial forb, or no perennial canopy. Samples under the canopy of perennial vegetation were considered protected versus unprotected in interspaces or with only annual canopy cover.

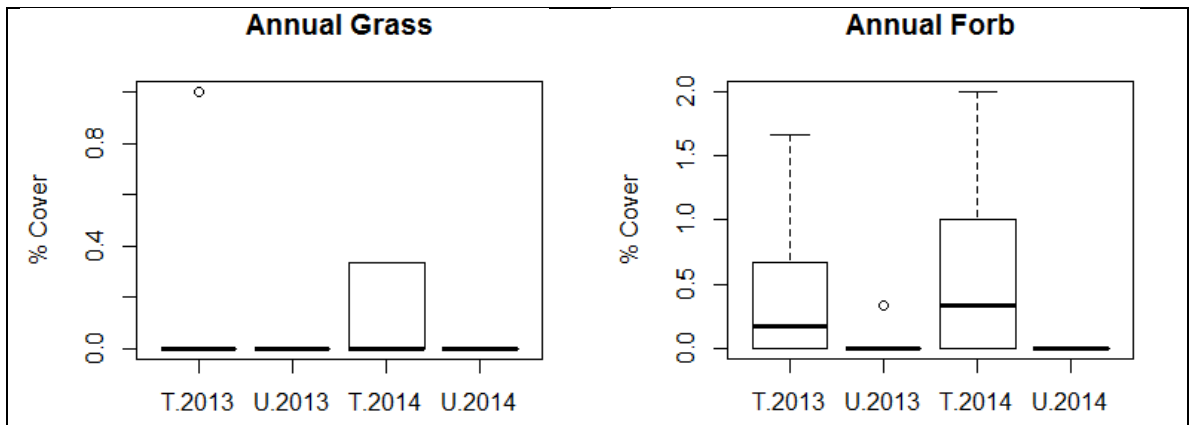
Interspace infiltration and shrub canopy infiltration were estimated using a single ring infiltrometer (Lowery et al. 1996) with 150ml of water (74mm depth) over an area of 20.25 cm<sup>2</sup>. Infiltrator locations were not pre-wet. Duff and litter was not removed from the infiltration site. Shrub infiltration time was collected under the canopy of black sagebrush and Wyoming big sagebrush. The same species was sampled at each plot pair. Infiltration times were recorded to the nearest second up to one hour, or as “more than one hour.”

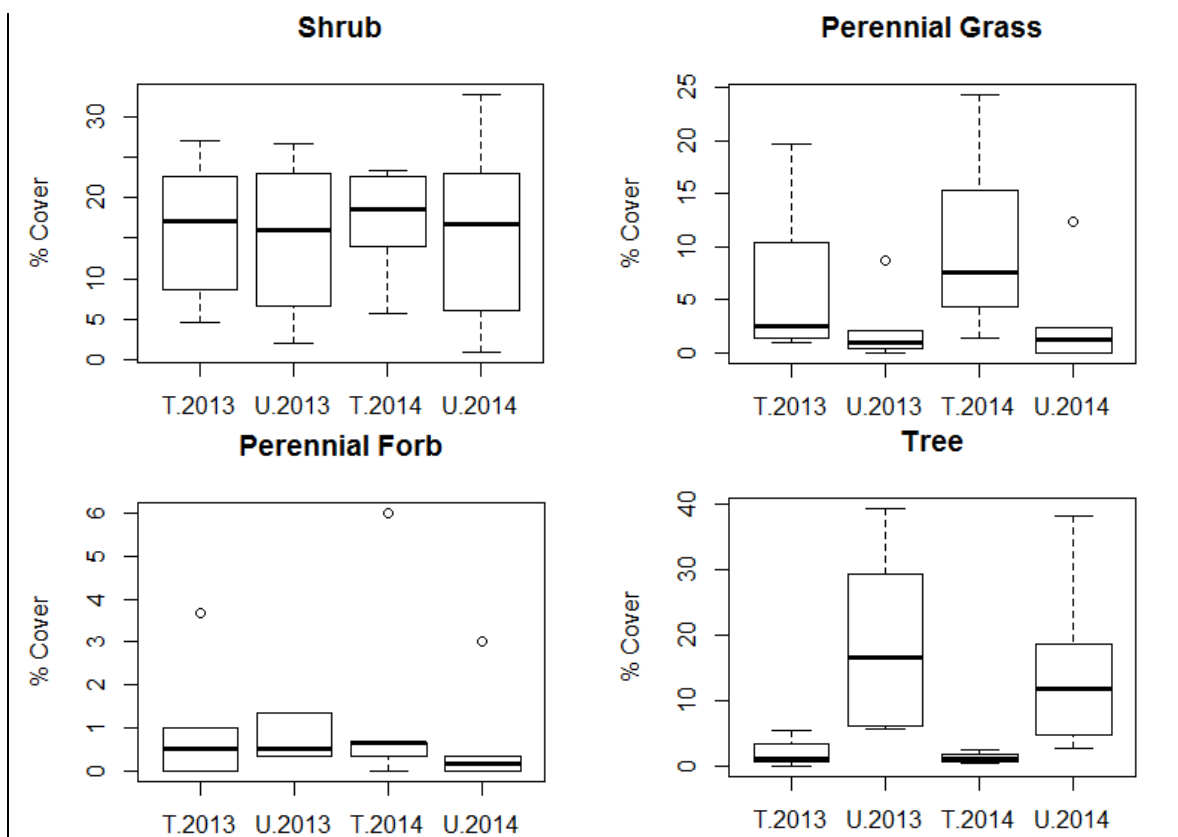
Species diversity was calculated as Simpson's diversity index for each plot (Simpson 1949). Species richness was calculated as a count of the total number of different taxa encountered at each plot. Gaps were grouped into classes as recommended by Herrick et al. (2001). The classes used were 25-50 cm, 51-100 cm, 101-200 cm, and >200 cm. We conducted two-way repeated measures analysis of variance (RM ANOVA) (Girden 1992) looking for statistically significant differences based on year and treatment among functional groups (annual forb, annual grass, perennial grass, forb, seeded grass, shrubs, and trees), ground cover classes (bare, litter, rock, pavement, moss, scat), gap classes, infiltration, species richness, and species diversity. Because stability was only collected in 2014, paired samples t-tests were used to determine significant difference

between treatment for surface soil aggregate stability and subsurface soil aggregate stability.

### Results

The White Pine hand thinning successfully removed trees from the site without damaging the understory community. There was a significant reduction in tree cover due to the treatment (Figure 1, Table 1). Some trees remained in the treated area (less than 2% of the vegetation cover). This is a 90% reduction in tree cover compared with the untreated areas where cover was around 14-18% (Table 2). Most of trees encountered were individuals that had survived the treatment. Some of the tree trunks were not fully severed by the chainsaw and a portion of the tree remained alive and appeared to be regrowing.





**Figure 1.** Cover of vegetation grouped by functional group. Error bars show standard error.

**Table 0-1.** Summary of Repeated Measures ANOVA results for vegetation cover classes. Degrees of freedom = 1,10 for all tests. F = test statistic, p = probability that the test statistic could be obtained by chance. Bold values indicate significance at the 95% confidence level.

	<u>Treatment</u>		<u>Year</u>		<u>Treatment*Year</u>	
	F	p	F	p	F	p
Annual Grass	1.623	0.231	0.172	0.687	0.172	0.687
Annual Forb	3.476	0.092	0.164	0.694	0.656	0.437
Shrub	0.043	0.840	0.564	0.470	0.000	1.000
Perennial Grass	2.449	0.149	<b>14.483</b>	<b>0.003</b>	<b>7.172</b>	<b>0.023</b>
Perennial Forb	0.385	0.549	0.354	0.565	0.984	0.345
Tree	<b>7.425</b>	<b>0.021</b>	<b>8.389</b>	<b>0.016</b>	3.981	0.074

**Table 0-2.** White Pine vegetation summary. There are six untreated and six treated plots. Relative frequency is shown as a percent of total frequency. Perennial density is shown as plants per square meter.

Relative Frequency

Percent Cover

Perennial Density

	<u>Untreated</u>		<u>Treated</u>		<u>Untreated</u>		<u>Treated</u>		<u>Untreated</u>		<u>Treated</u>	
	<u>2013</u>	<u>2014</u>	<u>2013</u>	<u>2014</u>	<u>2013</u>	<u>2014</u>	<u>2013</u>	<u>2014</u>	<u>2013</u>	<u>2014</u>	<u>2013</u>	<u>2014</u>
Annual Forb	5.6	5.2	12.2	10.6	0.1	0.0	0.4	0.6	0.0	0.0	0.0	0.0
Annual Grass	0.3	0.6	3.0	1.8	0.0	0.0	0.2	0.1	0.0	0.0	0.0	0.0
Perennial Grass	20.7	23.2	26.7	30.7	2.2	2.8	6.2	10.1	1.7	3.6	8.3	24.3
Perennial Forb	33.2	31.7	22.6	22.5	0.7	1.6	0.9	0.7	1.8	4.0	1.9	4.1
Shrub	36.3	36.1	33.6	33.2	15.1	16.0	16.2	17.1	3.1	3.2	3.5	4.0
Tree	3.9	3.3	1.0	1.2	18.9	14.7	1.9	1.2	0.3	0.2	0.1	0.1
Total	100	100	100	100	36.9	35.1	25.9	29.8	6.8	11.0	13.8	32.5

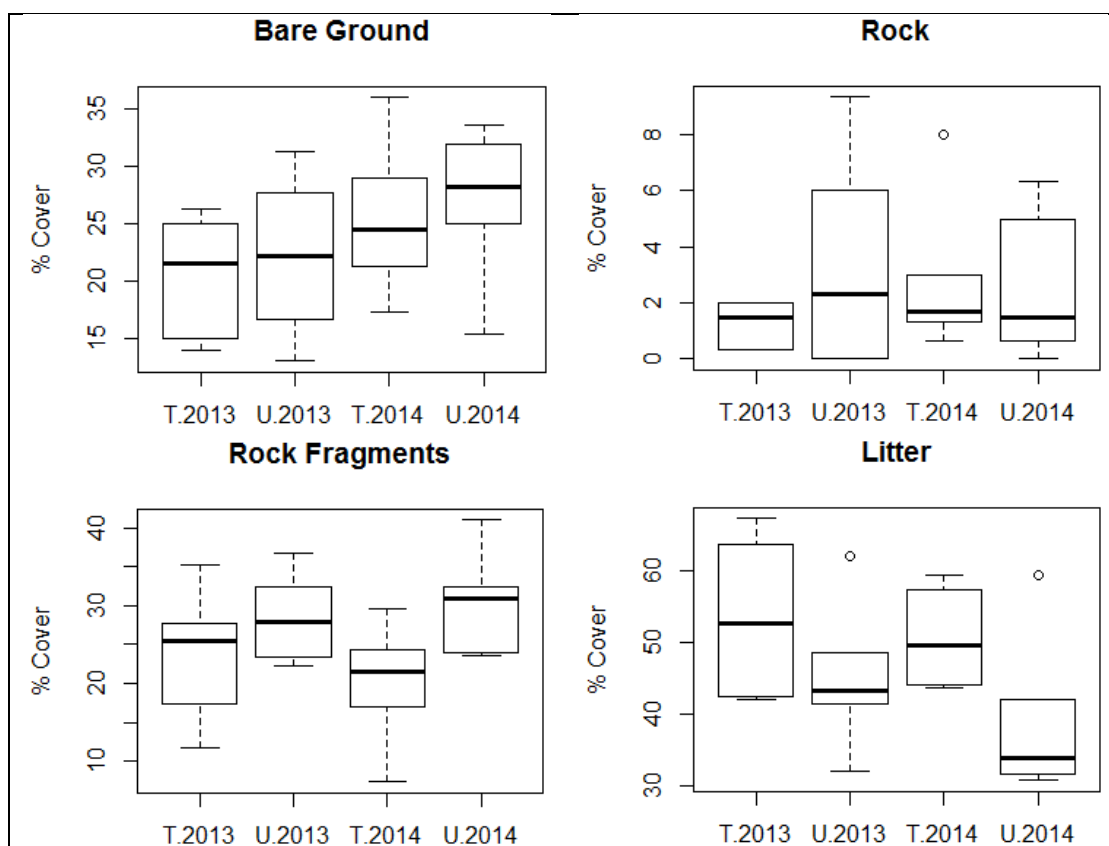
The sagebrush on the site mostly survived the hand thinning treatment. There were no significant differences in shrub cover between the treated and untreated areas (Figure 1, Table 1). The sagebrush on the treated site was still mostly large mature individuals and not recent recruits that established post-treatment. The frequency, cover, and density of shrubs showed very little difference between treated and untreated areas (Table 2). The sagebrush encountered during monitoring was almost entirely survivors that were alive before the treatment.

The native perennial grasses include both older bunchgrasses and seedlings or younger individuals that appeared to have established since the treatment time. Perennial grass cover was higher in the treated than the untreated areas, but the difference was not significant (Figure 1, Table 1). Perennial grasses also significantly increased from 2013 to 2014. There is a significant interaction between year and time as most of the gains in perennial grass cover were in the treated area in 2014. Perennial density and frequency were also higher in the treated areas (Table 2).

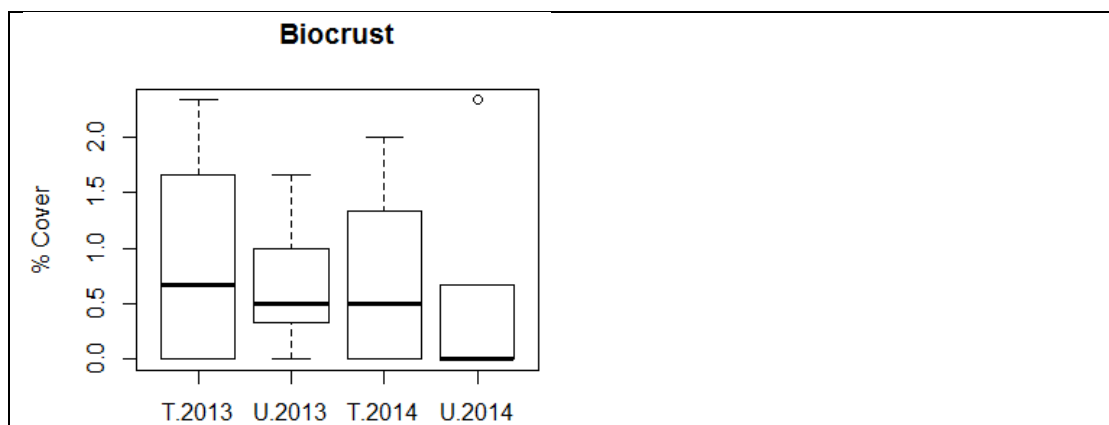
There were not any significant differences in the cover of annual forbs, annual grasses, or perennial forbs between treatment areas or years. There are few annual forbs

or annual grasses in the treated or untreated areas. Perennial forbs are a minor component of the treated and untreated areas making up around 1-2% cover.

There were not any significant differences in ground cover between the treated and untreated areas. There were some significant effects by year. From 2013 to 2014, there was a significant reduction in litter and a corresponding increase in bare ground cover at the treated and untreated sites. There was also a significant interaction between year and treatment for rock fragments. There were no significant differences in rock or biocrust cover by year or treatment (Figure 2, Table 3).





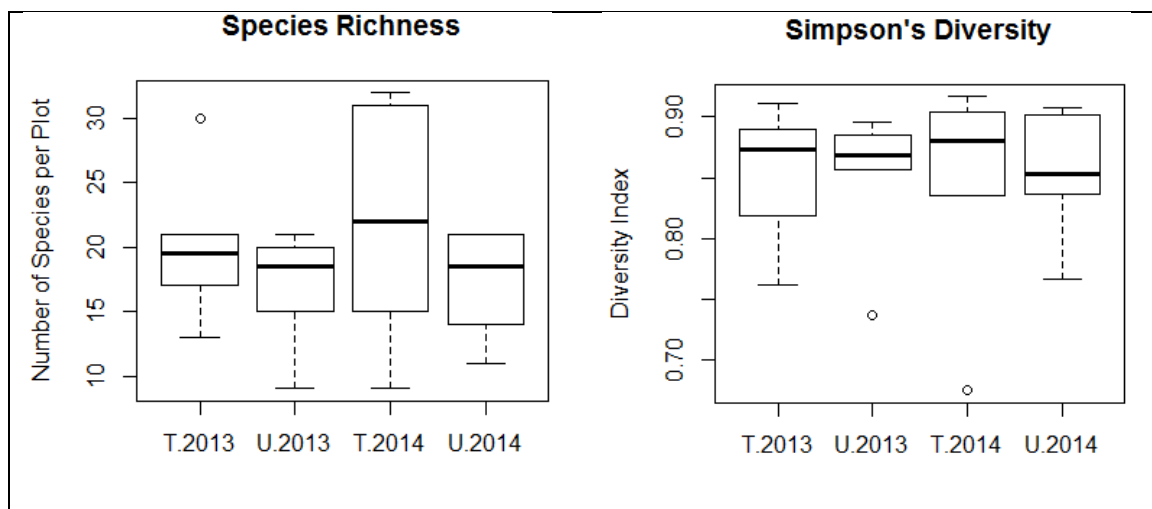


**Figure 2.** Boxplots for 2013 and 2014 ground cover at the White Pine hand thinning.

**Table 0-3.** Summary of RM ANOVA results for ground cover. Degrees of freedom = 1,10 for all tests. F = test statistic, p = probability that the test statistic could be obtained by chance. Bold values indicate significance at the 95% confidence level.

	<u>Treatment</u>		<u>Year</u>		<u>Treatment*Year</u>	
	F	p	F	p	F	p
Bare	0.201	0.664	<b>21.64</b>	<b>&lt;0.001</b>	0	1.000
Rock	0.399	0.542	0.268	0.616	3.717	0.083
Rock Fragments	3.761	0.092	1.028	0.335	<b>13.639</b>	<b>0.004</b>
Litter	3.229	0.103	<b>11.873</b>	<b>0.006</b>	1.832	0.206
Biocrust	0.220	0.649	1.034	0.333	0	1.000

Thinning did not have significant effects on species richness or species diversity (Figure 3, Table 4). Mean species richness was between 18-23 species per plot for all areas and years. Mean Simpson's diversity index was between 0.86 and 0.88 for all areas and years.



**Figure 3.** Box plots for 2013 and 2014 average species richness and average species diversity.

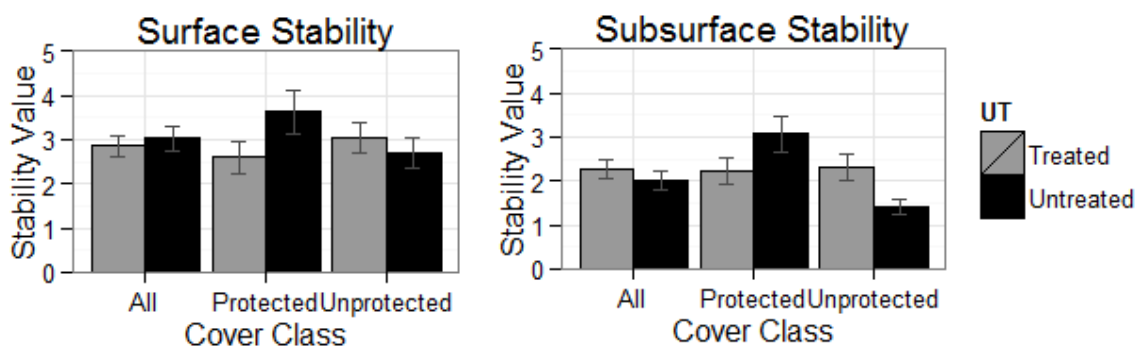
**Table 0-4.** Summary of RM ANOVA results for species richness and diversity. Degrees of freedom = 1,10 for all tests. F = test statistic, p = probability that the test statistic could be obtained by chance. Bold values indicate significance at the 95% confidence level.

	<u>Treatment</u>		<u>Year</u>		<u>Treatment*Year</u>	
	F	p	F	p	F	p
Species Richness	1.278	0.285	0.879	0.370	0.421	0.531
Simpson's Diversity	<b>0.001</b>	0.982	0.053	0.822	0.128	0.728

The hand thinning treatment did not have major effects on infiltration. The interspace infiltration time at the treated plots (M=2299, SD=723.3) was not significantly different than the interspace infiltration at the untreated plots (M=2283, SD=1553.6);  $t(5)=-0.025$ ,  $p=0.981$ ). The shrub infiltration time at the treated plots (M=639, SD=516.0) was not significantly different than the shrub infiltration at the untreated plots (M=555, SD=450.0);  $t(5)=-0.494$ ,  $p=0.642$ ).

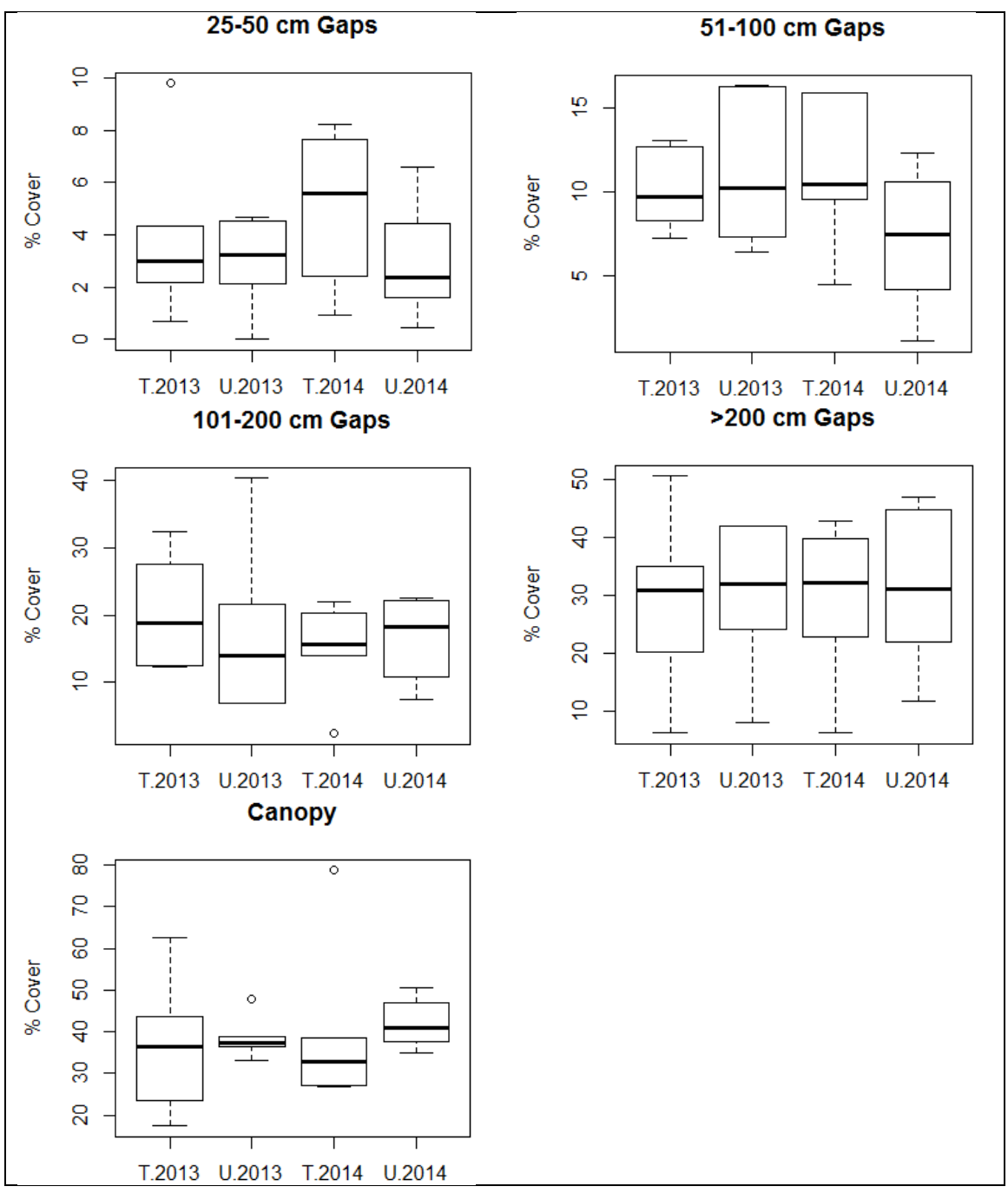
The hand thinning treatment did not have major effects on overall soil stability. The subsurface stability at the treated plots (M=2.27, SD=0.350) was not significantly

different than the subsurface stability at the untreated plots ( $M=2.02$ ,  $SD=0.679$ );  $t(5)=-0.905$ ,  $p=0.407$ . The surface stability at the treated plots ( $M=2.85$ ,  $SD=0.950$ ) was not significantly different than the subsurface stability at the untreated sites ( $M=3.13$ ,  $SD=1.121$ ),  $t(5)=0.381$ ,  $p=0.719$ ). The protected surface stability at the treated plots ( $M=2.61$ ,  $SD=1.210$ ) was not significantly different than the protected surface stability at the untreated plots ( $M=3.19$ ,  $SD=1.343$ );  $t(5)=0.565$ ,  $p=0.597$ . The unprotected surface stability at the treated plots ( $M=3.05$ ,  $SD=1.432$ ) was not significantly different than the unprotected surface stability at the untreated plots ( $M=2.82$ ,  $SD=1.162$ );  $t(5)=-0.321$ ,  $p=0.761$ . The protected subsurface stability at the treated plots ( $M=2.29$ ,  $SD=0.722$ ) was not significantly different than the protected subsurface stability at the untreated plots ( $M=2.83$ ,  $SD=1.019$ );  $t(5)=0.743$ ,  $p=0.491$ . There was a significant difference between the unprotected subsurface stability at the treated plots ( $M=2.21$ ,  $SD=0.649$ ) and the unprotected subsurface stability at the untreated plots ( $M=1.48$ ,  $SD=0.429$ );  $t(5)=-3.313$ ,  $p=0.021$ .



**Figure 4.** Surface and subsurface aggregate stability values by cover class. Protected indicates the sample was collected under a perennial forb, grass, shrub, or tree. Unprotected indicates the sample was not collected under a perennial forb, grass, shrub, or tree. Error bars show standard error.

There was no significant change in gap structure (total gap and canopy cover) between the treated and untreated areas. The areas also had similar amounts of gaps within each size class: 25-50cm, 51-100 cm, 101-200 cm, and >200 cm (Table 5, Table 5).



**Figure 5.** Boxplots for 2013 and 2014 gap classes and canopy cover at the White Pine hand thinning.

**Table 0-5.** Summary of RM ANOVA results for gap. Degrees of freedom = 1,10 for all tests. F = test statistic, p = probability that the test statistic could be obtained by chance. Bold values indicate significance at the 5% confidence level.

	<u>Treatment</u>		<u>Year</u>		<u>Treatment*Year</u>	
	F	p	F	p	F	p
<b>25-50 cm</b>	1.268	0.286	0.710	0.419	0.704	0.421
<b>51-100 cm</b>	0.611	0.452	1.405	0.263	3.979	0.074
<b>101-200 cm</b>	0.030	0.866	1.261	0.288	0.704	0.421
<b>&gt;200 cm</b>	0.042	0.842	0.118	0.739	0.033	0.859
<b>Canopy</b>	0.087	0.774	1.775	0.212	0.018	0.896

## Discussion

The White Pine hand thinning was successful in meeting its treatment goals of removing trees and allowing perennial grasses and forbs to increase. Just as important, the hand thinning did not significantly harm the shrub community, impact infiltration times, lower overall soil stability, decrease biological soil crust cover, or allow for much annual invasion. Cheatgrass or other nonnative species did not dominate the site after treatment.

As would be predicted for these ecological sites, the treatment moved the area through a phase pathway 2.3a from current potential phase 2.3 (at risk) with increasing Pinyon-juniper into current potential phase 2.1 with sagebrush and perennial grasses dominating, but with some annual non-native species present (Stringham 2015). The ecological site indicates that phase pathway 2.3a can be triggered by low severity fire, Aroga moth infestation, brush management with minimal soil disturbance, or late fall/winter grazing causing mechanical damage to sagebrush. The hand thinning can be considered brush management with little soil disturbance. In this case, the disturbance

response was well predicted by the state and transition models. The treatment did not cause a state transition. The treatment was done in a timely manner in phase II Pinyon-juniper before the area crossed a transition into a tree state.

A release of herbaceous plants is expected in hand thinned areas in this phase (Bates et al. 2000; Roundy et al. 2014b). That release is starting to be seen at in the increasing perennial grasses within the treated area. There was more litter in the treated areas, but the difference was not quite significant. Increased litter cover has several positive impacts including decreasing erosion potential (Davenport et al. 1998) and providing safe sites for seedling establishment (Fowler 1988). However, increased litter can also help cheatgrass establishment (Pierson and Mack 1990). The litter cover appears to be decreasing from 2013 to 2014. The decreasing litter corresponded with an increase in bare ground. After the litter is blown away, washed out, or decomposed, there was more bare soil left in its place at this site. There was not much of a change in the gap structure on the site.

The subsurface stability is higher in the protected sites at the untreated area because those protected sites are primarily under Pinyon-juniper trees which have high stability under their canopies. The unprotected sites are likely higher in the treated area because many of those unprotected sites were formerly under Pinyon-juniper canopy which was removed by the treatment, but the high stability has perpetuated despite the treatment. A similar pattern is seen with surface stability, but the slightly greater stability at the unprotected sites is not statistically significant. The difference in stability could be one of the reasons why Bates et al. (1998) found vegetative differences within hand

thinned sites between areas that used to be under Pinyon-juniper canopy and other areas that were not previously under Pinyon-juniper canopy. Overall, average stability was not significantly impacted by the treatment. Stability is a good indicator of overall range condition (Herrick et al. 2009). If Pinyon-juniper is removed using other mechanical methods that require heavy equipment, such as mastication or feller-buncher, there is a greater chance that soil stability would be negatively impacted (Ross et al. 2012).

The area had a low percentage of introduced species. Annual grasses were present at the treated sites, but not at the untreated sites. Still, annual grasses made up only a fraction of a percent of total cover and were relatively infrequent. Hand thinning can sometimes cause an increase in nonnative annuals (Roundy et al. 2014b; Ross 2012). But this increase is especially prominent at sites of hand thinning on infilled Pinyon-juniper (Roundy et al. 2014a). The White Pine hand thinning was not very densely infilled and had robust and diverse understory after treatment. The resistance of the intact understory communities was enough to keep cheatgrass from invading in this case. The White Pine hand thinning worked very well removing Pinyon Juniper canopy without damaging the shrub community or converting to strong annual domination post-treatment.

## **Management Implications**

Hand thinning is an appropriate way to reduce tree cover without impacting the shrub community. Because the shrub community is not removed, site resilience is not lost and the community is able to resist invasion from cheatgrass better than the site would if

a more intense treatment had been applied. Hand thinning is a very low intensity treatment and appropriate in areas of lower resilience.

Hand thinning is most appropriate for phase I or early phase II Pinyon-juniper encroachment before infilling occurs. Sites where hand thinnings are done on infilled Pinyon-juniper have a greater chance of post-treatment invasion by annuals (Roundy et al. 2014a). Ross (2012) found that hand thinning significantly increased understory cover and were more effective than mastication treatments, but also cause an increase in cheatgrass when done in phase III Pinyon-juniper woodland. The White Pine hand thinning resulted in a small but not significant increase in cheatgrass. Cheatgrass remained a very minor component of the post treatment community that made up less than 1% cover.

Hand thinning is not as damaging to understory shrubs or herbaceous vegetation as bulldozing (Young et al. 1985). Hand thinning maintains shrubs cover in phase I areas and increases perennial herbaceous cover in phase II and phase III areas (Roundy et al. 2014b). As long as sufficient understory remains after treatment to maintain site resistance, the area should respond well. That understory is not present in phase III Pinyon-juniper woodlands. Hand thinning is most appropriate for early phase Pinyon-juniper encroachment before infilling occurs.

Hand thinning is an especially important treatment in phase I or early phase II Pinyon-juniper woodlands that may be at risk of converting to cheatgrass or on steep slopes where erosion is a concern. Because hand thinning does not impact soil stability and leaves the understory vegetation on the site, it can be used on steep slopes without



fear of significant increases in erosion following treatment. Hand thinning can also be done on slopes that are too steep to drive heavy equipment safely.

Hand thinning is an appropriate treatment to be used in greater sage-grouse (*Centrocercus urophasianus*) habitat because Pinyon-juniper can be effectively removed without impacts on sage brush or perennial cover. Hand thinning is not as damaging to understory shrubs or herbaceous vegetation components as other mechanical treatments such as bulldozing (Young et al. 1985). Sage grouse avoid areas with more than 1% and almost entirely stay out of areas with more than 4-5% Pinyon-juniper cover (Baruch-Mordo et al. 2013, Knick et al. 2013). Lop and scatter can reduce pinyon-juniper cover to less than 1% (Provencher and Thompson 2014) which will satisfy the habitat requirements of the greater sage-grouse.

## Literature Cited

- Baruch-Mordo, S., J. S. Evans, J. P. Severson, D. E. Naugle, J. D. Maestas, J. M., Kiesecker,... & K. P. Reese. 2013. Saving sage-grouse from the trees: a proactive solution to reducing a key threat to a candidate species. *Biological Conservation*, 167, 233-241.
- Bates, J.D. 2005. Herbaceous Response to Cattle Grazing Following Juniper Cutting in Oregon. *Rangeland Ecology & Management*, May 2005, Vol. 58, No. 3, pp. 225-233.
- Bates, J. D., R. F. Miller, & T. Svejcar. 1998. Understory patterns in cut western juniper (*Juniperus occidentalis* spp. *occidentalis* Hook.) woodlands. *Western North American Naturalist*, 58(4), 363-374.
- Bates, J. D., R. F. Miller, & T. Svejcar. 2000. Understory dynamics in cut and uncut western juniper woodlands. *Journal of Range Management*, 119-126.
- Bates, J. D., R. F. Miller, & T. Svejcar. 2005. Long-term successional trends following western juniper cutting. *Rangeland Ecology & Management*, 58(5), 533-541.
- Bureau of Land Management. 1999. *Sampling vegetation attributes: interagency technical reference*.

- Davenport, D. W., D. D. Breshears, B. P. Wilcox, & C. D. Allen. 1998. Viewpoint: Sustainability of pinon-juniper ecosystems: A unifying perspective of soil erosion thresholds. *Journal of Range Management*, 231-240.
- Everett, R. L., & S. H. Sharrow. 1985. Understory response to tree harvesting of singleleaf pinyon and Utah juniper. *Western North American Naturalist*, 45(1), 105-112.
- Fowler, N. L. 1988. What is a safe site?: neighbor, litter, germination date, and patch effects. *Ecology*, 947-961.
- Girdenm, E.R. (Ed.). 1992. *ANOVA: Repeated measures* (No. 84). Sage.
- Goulden, C. H. 1939. *Methods of statistical analysis*. New York: Wiley.
- Herrick, J. E., J. V. Zee, K. M. Havstad, L. M. Burkett, & W. G. Whitford. 2009. Monitoring manual for grassland, shrubland and savanna ecosystems. USDA-ARS Jornada Experimental Range. <http://jornada.nmsu.edu/monit-assess/manuals/monitoring> Accessed 22 March 2013.
- Janetski, J. 1999. Role of pinyon-juniper woodlands in Aboriginal societies of the Desert West. In *USDA Forest Service Proceedings*.
- Knick, S. T., S. E. Hanser, & K. L. Preston. 2013. Modeling ecological minimum requirements for distribution of greater sage-grouse leks: implications for population connectivity across their western range, USA. *Ecology and Evolution*, 3(6), 1539-1551.
- Loftin, S. R. 1999. Initial response of soil and understory vegetation to a simulated fuelwood cut of a pinyon-juniper woodland in the Santa Fe National Forest. *Proceedings: Ecology and Management of Pinyon-Juniper Communities within the Interior West. USDA Forest Service Proc. RMRS-P-9. Ogden, UT*, 311-314
- Lowery, B., W. J. Hickey, M. A. Arshad, & R. Lal. 1996. Soil water parameters and soil quality. *Methods for Assessing Soil Quality*, 143-155.
- Miller, R. F., & R. J. Tausch. 2001. The role of fire in pinyon and juniper woodlands: a descriptive analysis. In *Proceedings of the invasive species workshop: the role of fire in the control and spread of invasive species*. Tall Timbers Research Station Miscellaneous Publication (Vol. 11, wpp. 15-30).
- Mohr, C. H., R. Coppus, A. Iroumé, A. Huber, & A. Bronstert. 2013. Runoff generation and soil erosion processes after clear cutting. *Journal of Geophysical Research: Earth Surface*, 118(2), 814-831.
- O'Connor, C., R. Miller, & J. D. Bates. 2013. Vegetation response to western juniper slash treatments. *Environmental Management*, 52(3), 553-566.

- Pierson, E. A., & R. N. Mack. 1990. The population biology of *Bromus tectorum* in forests: effect of disturbance, grazing, and litter on seedling establishment and reproduction. *Oecologia*, 84(4), 526-533.
- Pierson, F. B., J. D. Bates, T. J. Svejcar, & S. P. Hardegree. 2007. Runoff and erosion after cutting western juniper. *Rangeland Ecology & Management*, 60(3), 285-292.
- Provencher, L., & J. Thompson. 2014. Vegetation Responses to Pinyon-Juniper Treatments in Eastern Nevada. *Rangeland Ecology and Management*, 67(2), 195-205.
- Ross, M. R., S. C. Castle, & N. N. Barger. 2012. Effects of fuels reductions on plant communities and soils in a piñon-juniper woodland. *Journal of Arid Environments*, 79, 84-92.
- Roundy, B. A., R. F. Miller, R. J. Tausch, K. Young, A. Hulet, B. Rau,... & D. Eggett. 2014a. Understory cover responses to piñon-juniper treatments across tree dominance gradients in the Great Basin. *Rangeland Ecology and Management*, 67(5), 482-494.
- Roundy, B. A., K. Young, N. Cline, A. Hulet, R. F. Miller, R. J. Tausch,... & B. Rau. 2014b. Piñon-juniper reduction increases soil water availability of the resource growth pool. *Rangeland Ecology and Management*, 67(5), 495-505.
- Simpson, E. H. 1949. Measurement of diversity. *Nature*.
- Soil Survey Staff. 1999. Soil taxonomy: A basic system of soil classification for making and interpreting soil surveys. 2nd edition. Natural Resources Conservation Service. U.S. Department of Agriculture Handbook 436.
- Stringham, T.K., P. Novak-Echenique, P. Blackburn, C. Coombs, D. Snyder, and A. Wartgow. 2015. Final Report for USDA Ecological Site Description State-and-Transition Models, Major Land Resource Area 28A and 28B Nevada. University of Nevada Reno, Nevada Agricultural Experiment Station Research Report 2015-01. p. 1524.
- Tausch, R. J., N. E. West, & A. A. Nabi. 1981. Tree age and dominance patterns in Great Basin pinyon-juniper woodlands. *Journal of Range Management*, 259-264.
- Tueller, P. T., C. D. Beeson, R. J. Tausch, N. E. West, & K. H. Rea. 1979. Pinyon-juniper woodlands of the Great Basin: distribution, flora, vegetal cover. *USDA Forest Service research paper INT*.
- Young, J. A., & J. D. Budy. 1979. Historical use of Nevada's pinyon-juniper woodlands. *Forest & Conservation History*, 23(3), 112-121.

Young, J. A., R. A. Evans, & C. Rimbey. 1985. Weed control and revegetation following western juniper (*Juniperus occidentalis*) control. *Weed Science*, 513-517.

## Chapter 3: Long-term effects of Pinyon-juniper chaining on soil stability

### Abstract

Chaining was the most popular Pinyon-juniper removal treatment in the 1970s. Many of those old chainings are starting to have trees reencroach the area. The Holt chaining was one of those treatments completed in the early 1970s and has not been retreated in more than 40 years. We evaluated vegetation compositions and soil properties by comparing similar plots in chained and adjacent untreated areas. The effects of the chaining are still significant even with Pinyon-juniper (*Pinus monophylla*-*Juniperus osteosperma*) trees are reinvading and make up >5% of the cover on the treated area. The treated areas still have a much more productive understory than adjacent untreated areas. Perennial grass cover, frequency, and density was 2-5 times greater in the chained area. There were fewer large gaps (>100 cm) in the treated area. However, interspace infiltration times were slower in the treatment ( $t(4)=-2.14$ ,  $p=0.09$ ). Surface and subsurface soil aggregate stability remained significantly lower in the treatment for vegetation-protected and unprotected samples ( $t(4)=3.53$ ,  $p=0.024$ ;  $t(4)=3.10$ ,  $p=0.036$ ). Chainings have long-term benefits for vegetation, but also long term impacts on soils and hydrologic ecosystem processes.

### Introduction

Over the last 200 years, Pinyon-juniper woodlands in the Western United States have expanded more than 10 fold from about 3 million ha to their current range (Miller and Tausch 2001). Land managers in the Intermountain West are using active

management strategies to attempt to control the expansion of Pinyon-juniper (*Pinus monophylla-Juniperus osteosperma*) woodlands (Tausch and Hood 2001). The stated goals of Pinyon-juniper thinning treatments include improving livestock forage (Ansley et al. 2006), creating habitat for wildlife including sensitive and listed species (Fairchild, 1997; Monson et al. 2004; Skousen et al. 1989), maintaining species richness and diversity (Bunting et al. 1999; Fulbright 1996), and creating fuel breaks (Everett and Clary 1985).

One of the more common Pinyon-juniper control treatments has been chaining (Miller 2005). Chaining consists of dragging an anchor chain between two crawler tractors to fell and uproot trees. Single pass Ely chaining removes around 90% of the Pinyon-juniper on a site (Cain 1971). Double pass chaining has even higher removal rates, but flexible young trees may still survive (Monsen et al. 2004). Ely chains are boat anchor chains with I-beams or railroad rails welded crossways to some or all links of the chain. Ely chains are more effective than smooth chains at scarifying the soil and preparing the seedbed (Stevens 1997).

Chaining is an important Pinyon-juniper control tool for land managers in the Great Basin. These treatments reduce the cover of Pinyon-juniper trees and release more desirable perennial grasses and forbs from tree competition (Redmond et al. 2013). Chaining is one-fifth the cost or less than other mechanical treatments such as lop-pile-burn, lop-and-scatter, feller-buncher, or mastication, but still provides significant increases in understory canopy abundance (Provencher and Thompson 2014; Chadwick

et al. 1999). Chaining also increases the effectiveness of broadcast seeding treatments because it scarifies the soil surface and covers seeds (Stevens 1999).

Due to mechanical soil disturbance during the treatment, chaining has impacts on soil properties and erosion rates. Chaining increases water run-off and erosion rates (Wilcox 1994). Chaining slows infiltration rates (Roundy et al. 1978). Chaining increases sediment discharge (Roundy and Vernon 1999; Farmer et al. 1999). Williams et al. (1972) studied the presence of sand sized (0.2-2mm diameter) soil aggregates within chainings and found it was not a good indicator of infiltration rates, but said that larger aggregates than their method studied might be more important. Other mechanical mastication treatments have had at least short term negative impacts on soil aggregate stability (Ross et al. 2012). Pile-and-burn treatments have also been correlated with lower soil aggregate stability (Owen et al. 2009). Miller et al. (2012) found soil aggregate stability to be unaffected by fire, but lowered by aerial seeding than chaining or rangeland drill treatments. For their analysis, the two treatment types were lumped together and the relative effect of chaining versus drill treatments were not examined. Studies have not been published looking at the direct effects of chaining on aggregate stability using the Seybold and Herrick (2001) test kits. Soil aggregate stability is a good indicator of overall soil quality and rangeland health (Herrick et al. 2001). So, the effects of chaining on soil aggregate stability are an important measure of the impacts of chaining.

Chaining is usually thought of as a relatively temporary treatment. Afterward, treated areas tend to become reinvaded by Pinyon-juniper treatments over the course of

several decades. Tausch and Tueller (1977) found that Pinyon-juniper reencroachment in Eastern Nevada can happen in as little as 15 years and the understory will become as unproductive as it was before treatment. Schott and Pieper (1987) suggest that Pinyon-juniper redominate chained sites after 28 years. Redmond et al. (2013) found that while chaining was effective at reducing tree cover and increasing herbaceous cover, there were also long term (20-40 year) increases in bare soil cover and lower biological soil crust cover. This study looks at a 40 year old chaining treatment that has not been retreated to determine the long term effects of the chaining treatment on vegetation and soils.

## Methods

The study was located in the Egan Range on an upland slope on the west side of Ward Mountain about 15 km southwest of Ely, Nevada (latitude 39°10'35" N to 39°10'1" N north to south; longitude 115°0'10" W to 114°58'23" W west to east). The treated area consists of a 136 ha area that was chained over several years in the early 1970s. The area is part of the West Ward grazing allotment. Elevation ranges from 2129 to 2214 m. Average annual precipitation is 314 to 378 mm. Hill slope ranges from 1% to 23% with an average slope of 9.5%. We believe it was chained using a single pass Ely chain based on common management practices in the area during that time period. The site was seeded with crested wheatgrass (*Agropyron cristatum*), Russian wildrye (*Psathyrostachys juncea*), and intermediate wheatgrass (*Thinopyrum intermedium*) which are all still present in the area. The area surrounding the chaining is dominated by phase III Pinyon-juniper trees according the classification system of Miller and Tausch (2001) with a minor understory containing shrubs, grasses, and forbs. The earliest aerial imagery



available for the area taken in 1999 shows the Pinyon-juniper surrounding the treatment was already dense phase III Pinyon-juniper (United States Geological Survey 2001). It is not known how dense the Pinyon-juniper was during the time of treatment in the mid-1970s. The Sagebrush at the site was mostly black sagebrush (*Artemisia nova*) with some Wyoming sagebrush (*A. tridentata* ssp. *wyomingensis*) and mountain big sagebrush (*A. tridentata* ssp. *vaseyana*). Other common shrubs in the area were bitterbrush (*Purshia tridentata*), rabbitbrush (*Chrysothamnus viscidiflorus*), currant (*Ribes* sp.), and desert snowberry (*Symphoricarpos longiflorus*). The most common perennial grasses were Sandberg bluegrass (*Poa secunda*), crested wheatgrass, bluebunch wheatgrass (*Pseudoroegneria spicata*), thickspike wheatgrass (*Elymus lanceolatus*), muttongrass (*Poa fendleriana*), bottlebrush squirreltail (*Elymus elymoides*), intermediate wheatgrass, Indian ricegrass (*Achnatherum hymenoides*), western wheatgrass (*Pascopyrum smithii*), and smooth brome (*Bromus inermis*). Soils in the area are derived from mixed limestone and volcanic rocks with a component of loess.

We established ten plots including five plots in the treated area and five paired plots in adjacent untreated areas. Polygons were created so plots could be paired in areas that shared similar precipitation, soil association, slope, aspect, and elevation. Plots were then randomly located within the stratified polygons. The sampling plots consisted of three 20 m long parallel transects spaced 5 m apart.

To characterize vegetation, we collected data on canopy cover, ground cover, nested frequency, perennial density, and canopy gap in summers of 2013 and 2014. We recorded line point-intercept canopy cover and ground cover (Herrick et al. 2009) using a

laser point projection device every 20 cm along three 20 m long parallel transects spaced 5 m apart with 100 points per transect and 300 points per plot. Frequency (Caulloudon 1999) was collected using a 1 m<sup>2</sup> nested frequency frame placed at 1 m intervals along the uphill side of each transect (20 frames per transect and 60 frames per plot). The nested frequency frame was subdivided into 4 sections with areas of 1 m<sup>2</sup>, 0.5 m<sup>2</sup>, 0.25 m<sup>2</sup> and 0.04 m<sup>2</sup>. Perennial density was recorded as a total count of all perennial plants rooted within 1 m on either side of the transect (40 m<sup>2</sup> per transect and 120 m<sup>2</sup> per plot). Canopy gap intercept (Herrick et al. 2009) was recorded along each transect with a minimum gap size of 20 cm. To characterize soils, we collected data on soil aggregate stability, infiltration, and taxonomy in 2014. An interspace soil pit was described within 15 m behind the start of each transect. The soil was classified to soil series (Soil Survey Staff 1999). Soil aggregate stability (Herrick et al. 2009) was tested along the first transect on the side opposite frequency and cover. We collected nine samples, each at 2 m intervals from 2 m to 18 m. At each sampling point, one sample was collected from the surface and another from 2-5 cm directly below. Each sample point was assigned to a canopy class based on vegetation type: tree, shrub, perennial grass, perennial forb, or no perennial canopy. Samples under the canopy of perennial vegetation were considered protected versus unprotected in interspaces or with only annual canopy cover.

Interspace soil infiltration was estimated using a single ring infiltrometer (Lowery et al. 1996) with 150ml of water (74mm depth) over an area of 20.25 cm<sup>2</sup>. Infiltrometer locations were not pre-wet. Duff and litter was not removed from the infiltration site. Infiltration times were recorded to the nearest second up to one hour, or as “more than one hour.”

We looked for statistically significant differences (treated versus untreated) among functional groups (Annual forbs, annual grasses, perennial grasses, perennial forbs, shrubs, and trees), ground cover classes (bare, litter, rock, pavement, moss, scat by species), surface soil aggregate stability, subsurface soil aggregate stability, average gap, percent gap, species richness, and species diversity. For each category, paired sample *t*-test (Goulden 1939) was used in R Statistical Software version 3.1.2 (R Core Team 2014). We used analysis of variance (ANOVA) (Fisher 1918) to compare infiltration times for different species.

Species diversity was calculated as Simpson's diversity index for each plot (Simpson 1949). Species richness was calculated as a count of the total number of taxa encountered at each plot.

## **Results**

Vegetation responded well to chaining. The treatment remains effective at controlling Pinyon-juniper dominance and increasing perennial grasses even after nearly 40 years. Pinyon-juniper trees are starting to recolonize the treated area, but they are still a minor component totaling only one-tenth the cover of Pinyon-juniper trees in the untreated area (Table 1). There was a significant difference in tree cover between the treated and untreated areas (Table 2). Junipers made up slightly more cover in the untreated areas than Pinyon pines. The treated area can now be classified as an early phase I Pinyon-juniper woodland based on the classification system of Miller and Tausch (2001). The trees are between 1-2 m tall.

**Table 0-1.** Holt vegetation summary. There are five untreated and five treated plots. Relative frequency is a percent of total frequency. Perennial density is plants per square meter. The numbers are an average of 2013 and 2014 values. Bold indicates a significant difference.

	<u>Relative Frequency</u>		<u>Percent Cover</u>		<u>Perennial Density</u>	
	U	T	U	T	U	T
<b>Annual Forb</b>	<b>9.95</b>	<b>0.76</b>	<b>0.20</b>	<b>0.04</b>	n/a	n/a
<b>Annual Grass</b>	2.76	2.58	0.10	0.80	n/a	n/a
<b>Perennial Grass</b>	<b>38.14</b>	<b>68.81</b>	<b>6.74</b>	<b>32.91</b>	<b>10.45</b>	<b>40.01</b>
<b>Perennial Forb</b>	<b>25.02</b>	<b>13.45</b>	0.57	0.84	<b>4.80</b>	<b>3.50</b>
<b>Shrub</b>	15.75	13.46	<b>9.20</b>	<b>19.77</b>	2.26	3.88
<b>Tree</b>	<b>8.37</b>	<b>1.09</b>	<b>57.07</b>	<b>5.34</b>	<b>1.11</b>	<b>0.20</b>
<b>Total</b>	100.00	100.00	73.80	59.67	18.61	47.58

**Table 0-2.** Paired t-test for vegetation cover classes between treated and untreated plots. DF = degrees of freedom (between groups, within groups), t = test statistic, p = probability that test statistic could be obtained by chance. Significant p values shown in bold.

	<b>DF</b>	<u>Relative Frequency</u>		<u>Percent Cover</u>		<u>Perennial Density</u>	
		t	P	t	P	t	P
<b>Annual Forb</b>	4	5.979	<b>0.004</b>	2.236	<b>0.089</b>	n/a	n/a
<b>Annual Grass</b>	4	0.896	0.933	-1.177	0.305	n/a	n/a
<b>Perennial Grass</b>	4	-5.360	<b>0.006</b>	-9.325	<b>&lt;0.001</b>	-7.854	<b>0.001</b>
<b>Perennial Forb</b>	4	2.879	<b>0.045</b>	0.649	0.552	3.991	<b>0.016</b>
<b>Shrub</b>	4	0.440	0.683	-2.925	<b>0.043</b>	-0.786	0.476
<b>Tree</b>	4	3.868	<b>0.018</b>	8.467	<b>0.001</b>	3.961	<b>0.016</b>

Perennial grasses had significantly more cover in the treated than untreated areas (Table 2). A large number of seeded perennial grasses are a dominant component of the plant community in the treated area. Nonnative seeded grasses made up more than half of the total perennial grass cover. The cover of shrubs in the treated area was roughly twice untreated shrub cover. The total vegetation cover in the treated area was slightly less than in the untreated area (Table 1). The untreated area had more native annuals than nonnative invasive annuals. The treated area had more nonnative invasive annuals than native annuals. The most common nonnative invasive annual was cheatgrass (*Bromus tectorum*), but it occurred at low cover and frequency. Perennial forbs accounted for

around 0.5-1% total cover in the treated and untreated areas (Table 1). Ground cover at the treated and adjacent untreated sites was similar except the treated area had significantly more signs of both livestock and wildlife (cover of cow, elk, and deer scat) (Table 3).

**Table 0-3.** Summary of ground cover at Holt. Scat is cow, elk, and deer scat combined, SD = standard deviation, DF = degrees of freedom (between groups, within groups), F = test statistic, p = probability that test statistic could be obtained by chance. Significant p values <0.1 shown in bold.

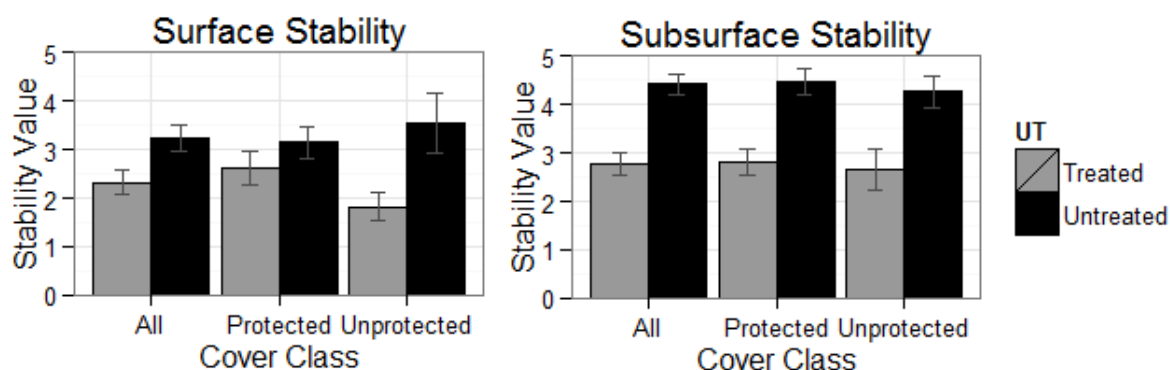
	Mean, SD		Paired t-test		
	Untreated	Treated	DF	t	p
<b>Bare soil</b>	14.40, 5.92	18.83, 4.80	4	-1.289	0.267
<b>Litter</b>	68.47, 10.35	65.83, 9.38	4	0.544	0.615
<b>Rock</b>	1.07, 1.49	0.77, 0.93	4	0.381	0.723
<b>Rock fragments</b>	15.73, 7.98	13.50, 6.54	4	1.276	0.271
<b>Scat</b>	0.23, 0.22	0.93, 0.68	4	-2.265	<b>0.086</b>
<b>Moss</b>	0.07, 0.09	0.10, 0.22	4	-0.408	0.704

There was a significant difference in surface aggregate stability at the untreated areas (M=3.24, SD=0.54) than at the treated areas (M=2.31, SD=0.40);  $t(4)=3.53$ ,  $p=0.024$ . There was a significant difference in subsurface aggregate stability at the untreated areas (M=4.42, SD=0.64) than at the treated areas (M=2.76, SD=0.87);  $t(4)=3.10$ ,  $p=0.036$ . Subsurface stability was consistently higher than surface stability. Average subsurface stability class in the treated area was below 3 for all samples indicating that less than 10% of the soil was in stable aggregates after five dipping cycles. There was little difference in subsurface stability between samples protected by perennial vegetation and unprotected samples. The average surface stability class in the treated area was even lower. The surface stability in the treated area was much lower for unprotected samples than protected samples. The subsurface soil stability for untreated area was

above 4 for all samples indicating that more than 25% of the soil was in stable aggregates. There was little difference in subsurface stability between samples protected and unprotected samples. The average surface stability class in the untreated area was above 3 for all samples indicating that between 10-25% of the soil was in stable aggregates (Table 4, Figure 1).

**Table 0-4.** Holt aggregate stability and gap summary. Gap size class shows the percentage of the plot taken up by gaps in that size class. Significant differences shown in bold.

		<u>Untreated</u>	<u>Treated</u>
<b>Simpson's Diversity Index</b>		<b>0.898</b>	<b>0.825</b>
<b>Species Richness</b>		24.7	20.4
<b>Total Canopy</b>		<b>71.73%</b>	<b>77.57%</b>
<b>Total Gap</b>		<b>28.27%</b>	<b>22.43%</b>
<b>Gap Size Class</b>	<b>25-50 cm</b>	<b>3.00%</b>	<b>9.18%</b>
	<b>51-100 cm</b>	6.64%	8.06%
	<b>101-200 cm</b>	<b>9.13%</b>	<b>4.44%</b>
	<b>&gt;200 cm</b>	<b>9.51%</b>	<b>0.75%</b>
<b>Aggregate Stability</b>	<b>Surface</b>	<b>3.24</b>	<b>2.31</b>
	<b>Subsurface</b>	<b>4.42</b>	<b>2.76</b>



**Figure 1.** Surface and subsurface aggregate stability values by cover class. Protected indicates the sample was collected under a perennial forb, grass, shrub, or tree. Unprotected indicates the sample was not collected under a perennial forb, grass, shrub, or tree. Error bars show standard error. n=180 total (90 surface, 90 subsurface).

There was a significant difference between treated areas and untreated areas in the amount and size of gaps. The total canopy gap at the treated area ( $M=0.224$ ,  $SD=0.054$ ) was significantly less than at the untreated area ( $M=0.283$ ,  $SD=0.080$ ),  $t(4)=2.492$ ,  $p=0.067$ . The same significant difference is shown by the total canopy. The percent of 25-50 cm gaps in the treated area ( $M=0.091$ ,  $SD=0.028$ ) was significantly more than in the untreated area ( $M=0.030$ ,  $SD=0.015$ ),  $t(4)=-3.412$ ,  $p=0.027$ . The percent of 51-100 cm gaps in the treated area ( $M=0.081$ ,  $SD=0.021$ ) was insignificantly more than in the untreated area ( $M=0.066$ ,  $SD=0.037$ ),  $t(4)=-0.655$ ,  $p=0.548$ . The percent of 101-200 cm gaps in the treated area ( $M=0.044$ ,  $SD=0.026$ ) was significantly less than in the untreated area ( $M=0.091$ ,  $SD=0.033$ ),  $t(4)=3.151$ ,  $p=0.034$ . The percent of >200 cm gaps in the treated area ( $M=0.007$ ,  $SD=0.010$ ) was significantly less than in the untreated area ( $M=0.095$ ,  $SD=0.072$ ),  $t(4)=2.841$ ,  $p=0.047$ . The treated area had more small gaps and fewer large gaps than the untreated area (Table 4).

Despite having more desirable species such as perennial grasses, the treated area still had a lower Simpson's diversity index and average species richness than the adjacent untreated area. The Simpson's diversity in the treated area ( $M=0.825$ ,  $SD=0.029$ ) was significantly lower than in the untreated area ( $M=0.898$ ,  $SD=0.009$ );  $t(4)=4.92$ ,  $p=0.008$ . The species richness in the treated area ( $M=20.4$ ,  $SD=4.1$ ) was lower than the species richness in the untreated area ( $M=24.7$ ,  $SD=5.2$ ), but the difference was not significant;  $t(4)=1.25$ ,  $p=0.280$ . We encountered 77 taxa while monitoring. We found 14 species found in untreated areas that were not found at treated areas including pussytoes (*Antennaria* sp.), thickstem wild cabbage (*Caulanthus crassicaulis*), curl-leaf mountain mahogany (*Cercocarpus ledifolius*), Douglas' dustymaiden (*Chaenactis douglasii*),

narrowstem cryptantha (*Cryptantha gracilis*), pinyon groundsmoke (*Gayophytum ramosissimum*), deervetch (*Lotus* sp.), three species of beardtongues (*Penstemon* spp.), Chambers' twinpod (*Physaria chambersii*), Douglas' knotweed (*Polygonum douglasii*), desert snowberry (*Symphoricarpos longiflorus*), and tufted Townsend daisy (*Townsendia scapigera*). We found 10 species found in treated areas that were not found at the untreated areas including pale agoseris (*Agoseris glauca*), smooth brome (*Bromus inermis*), rose heath (*Chaetopappa ericoides*), thickspike wheatgrass (*Elymus lanceolatus*), spotted fritillary (*Fritillaria atropurpurea*), flatspine stickseed (*Lappula occidentalis*), basin wildrye (*Leymus cinereus*), Lewis flax (*Linum lewisii*), thorn skeletonweed (*Pleiocanthus spinosus*), tall tumbled mustard (*Sisymbrium altissimum*), and stemless mock goldenweed (*Stenotus acaulis*).

The interspace infiltration time in the treated areas (M=1458, SD=915) was slower in the untreated areas (M=2093, SD=442) and the effect of treatment on interspace infiltration time was significant;  $t(4)=-2.14$ ,  $p=0.09$ . The effect of treatment on infiltration time under tree canopy was not significant [ $F(1,7) = 0.05$ ,  $p = 0.829$ ]. The effect of treatment on infiltration time under shrub canopy was not significant [ $F(1,5) = 0.446$ ,  $p = 0.534$ ]. There was a significant difference in the total infiltration time among interspace versus trees versus shrubs [ $F(2,23) = 16.24$ ,  $p = <0.001$ ]. Some hydrophobicity was encountered in both treated and untreated areas. The trees had the shortest infiltration time. The shrubs had longer infiltration times. The interspace had the longest infiltration time (Table 5).



**Table 0-5.** Holt infiltration summary. Mean infiltration time shown in seconds.

		<u>Untreated</u>	<u>Treated</u>	<u>Total</u>
<b>Infiltration Time</b>	<b>Interspace</b>	1457.80	2092.60	1775.20
	<b>Tree</b>	294.90	255.38	277.33
	<b>Shrub</b>	1018.25	707.67	885.14

## Discussion

The Holt chaining remains effective after more than 40 years, but Pinyon-juniper trees are established at the site and increasing. Christmas-tree sized Pinyon-juniper trees occur throughout the majority of the treatment, but perennial grasses dominate the treated area unlike the adjacent untreated Pinyon-juniper dominated areas. The perennial grass cover and density was 4-5 times greater in the treated area than the untreated area. In the treated area, perennial grasses covered more than one third of the area and made up more than half the total vegetation cover. In the untreated area, perennial grass are a minor component, cover less than seven percent of the area, and make up less than one tenth the total vegetation cover. There are large differences in the plant communities. Pinyon-juniper has yet to fully reencroach the treated area.

The initial chaining most likely initially left around 10% of the Pinyon-juniper trees on the site alive based on evidence from other chainings (Provencher and Thompson 2014). However, not all trees that appear to initially survive a chaining are able to survive long term and regrow. Stevens and Walker (1996) counted the number of live trees remaining on a chaining in 1964, marked them, then remonitored the area after 30 years and found 49% fewer remaining live trees. Some of the Pinyon-juniper trees that initially survive the chaining are too permanently damaged to survive long term. This implies that

chaining treatments are likely more effective at removing trees than indicated by studies that only monitor the first few years after treatment.

Even when tree recruitment is quick or trees survive the chaining, regrowth can be very slow. Redmond et al. (2013) found 40 year old trees had an average basal trunk diameter of less than 7 cm. This slow growth allows for perennial grasses and other understory components to thrive for decades after chainings before Pinyon-juniper trees again dominate. The 1-2 m tall trees in the treated area could possibly be as old as the treatment, as flexible seedlings are most likely to survive chaining treatments (Monsen et al. 2004).

The treated area had lower species richness and Simpson's diversity index than the untreated area. Dominance by nonnative grasses decrease species richness and diversity (Vernon et al. 2000). Crested wheatgrass, intermediate wheatgrass and Russian wildrye making up more than half of the perennial grass cover on the site probably contributed to these differences in richness and diversity. Average species diversity was unexpectedly higher in the untreated area than in the treated area. Despite the differences in species richness between the treated and untreated areas, the overall landscape is more diverse. There is greater species richness and diversity when the areas are considered together as there were some species in both areas that were not found in the other. Maintaining a matrix of clearings within Pinyon-juniper forests increases ecosystem production, diversity, and resilience (Fulbright 1996).

Soil aggregate stability links to infiltration rates (Bird et al. 2007), water runoff (Barthes and Roose 2002), erosion (Blackburn and Pierson 1994; Barthes and Roose

2002), sediment loads, percentage of bare ground, and biotic integrity (Herrick et al. 2009). The treated area did have significantly lower interspace infiltration times. The treated area also had a higher percent of bare ground, but the difference was not large enough to be significant. Erodibility depends on more static properties such as texture, mineral content, and slope length as well as dynamic properties such as aggregate stability, soil carbon, soil water content, biological soil crust development, and surface roughness (Blanco and Lal 2008). The lowered aggregate stability on the site may lead to increased erosion (Herrick et al. 2002). The lowered stability on the site could have been due to the chaining directly, differences in the post-treatment plant community (Meeuwig 1970), or different grazing intensities (Marble and Harper 1989).

Even after 40 years, the aggregate stability of the treated area is not as high as the untreated area and infiltration times are longer in the treated area. Additionally, despite a little more than 5% tree cover in the treated area, the area continues to produce more perennial grasses than the adjacent untreated Pinyon-juniper woodlands. Repeated chaining would be unwise here because it would likely not kill the small flexible trees. If retreated, hand thinning would be more appropriate and not costly as there are few small trees per acre.

## **Management Implications**

From the 1960s to the 1970s chaining was the most popular form of Pinyon-juniper removal treatment. It has since been overtaken by prescribed fire and other treatments (Miller 2005). Recently, some managers have begun recommending chaining again (Provencher and Thompson 2014) at least in part due to its cost effectiveness

relative to other treatments (Provencher and Thompson 2014; Evans and Workman 1994; Chadwick et al. 1999). Retreating chainings so early means that the “pre-treatment” vegetation condition for the next chaining would be very different than the last chaining. Hence, it is expected that there will be a different response to the treatment (Miller et al. 2014).

If a manager wishes to retreat the area, a second chaining would not be advised. The soil stability in the treated area is lower than the adjacent untreated area. The interspace infiltration times are higher in the treated area. These differences are most likely still left over from the chaining treatment. Chaining is known to cause an increase in water run-off, erosion, and sediment discharge rates (Wilcox 1994; Roundy and Vernon 1999; Farmer et al. 1999). These could be related to a loss in soil stability due to mechanical disturbance during the treatment. Additionally, the small flexible trees on the site may not be effectively removed by a chain. A treatment such as hand thinning would be preferable because it could remove the small trees on the site effectively and would be less intense causing less mechanical disturbance to the soil that could lead to a further reduction in stability.

Soil aggregate stability is a good indicator of biotic integrity, hydrologic function, and overall rangeland health (Herrick et al. 2009). Implementing soil aggregate stability measurements into a monitoring protocol is a cost effective way of gaining a great deal of information about ecosystem processes occurring in and around a treated area. Soil aggregate stability can indicate if an area has returned to pre-treatment soil conditions.

## **Literature Cited**

- Ansley, R. J., H. T. Wiedemann, M. J. Castellano, & J. E. Slosser. 2006. Herbaceous restoration of juniper dominated grasslands with chaining and fire. *Rangeland Ecology & Management*, 59(2), 171-178.
- Barthes, B., & E. Roose. 2002. Aggregate stability as an indicator of soil susceptibility to runoff and erosion; validation at several levels. *Catena*, 47(2), 133-149.
- Bird, S. B., J. E. Herrick, M. M. Wander, & L. Murray. 2007. Multi-scale variability in soil aggregate stability: implications for understanding and predicting semi-arid grassland degradation. *Geoderma*, 140(1), 106-118.
- Blackburn, W. H., & F. B. Pierson. 1994. Sources of variation in interrill erosion on rangelands. *Variability in rangeland water erosion processes*, 1-9.
- Bunting, S. C., J. L. Kingery, & E. Strand. 1999. Effects of succession on species richness of the western juniper woodland/sagebrush steppe mosaic. In *SB Monsen, S. Richards, RJ Tausch, RF Miller, and C. Goodrich (compilers). Proceedings Ecology and Management of piñon-juniper communities within the Interior West. USDA Forest Service, RMRS-P-9* (pp. 76-81).
- Cain, D. R. 1971. The eley chain. *USDI Bur. Land Manage. Ely, Nev*, 9.
- Chadwick, J. H., D. R. Nelson, C. R. Nunn, & D. A. Tatman. 1999. Thinning versus chaining: which costs more. In *Proceedings: Ecology and management of piñon-juniper communities within the Interior West. USDA Forest Service RMRS-P-9* (pp. 290-292).
- Chambers, J. C., B. A. Roundy, R. R. Blank, S. E. Meyer, & A. Whittaker. 2007. What makes Great Basin sagebrush ecosystems invisable by *Bromus tectorum*?. *Ecological Monographs*, 77(1), 117-145.
- Coulloudon, B. 1999. *Sampling vegetation attributes: interagency technical reference*. Bureau of Land Management.
- Evans, S. G., & J. P. Workman. 1994. Optimization of range improvements on sagebrush and pinyon-juniper sites. *Journal of Range Management*, 159-164.
- Everett, R. L., & W. Clary. 1985. Fire effects and revegetation on juniper-pinyon woodlands. In K. Sander, & J. Durham (Eds.), *Rangeland Fire Effects: A Symposium. USDI Bureau of Land Management, Idaho State Office, Boise, ID* (pp. 33-37).
- Fairchild, J. A. 1997, September. Pinyon-juniper chaining design guidelines for big game winter range enhancement projects. In *Monsen, SB; Stevens, R., comps. Proceedings: ecology and management of pinyon-juniper communities within the Interior West* (pp. 15-18).

- Farmer, M. E., K. T. Harper, & J. N. Davis. 1999. The influence of anchor-chaining on watershed health in a juniper-pinyon woodland in central Utah. *Proceedings, ecology and management of pinyon-juniper communities within the Interior West. Proceedings RMRS-P-9. Ogden, UT. USDA Forest Service. Rocky Mountain Research Station*, 299-301.
- Fisher, R. A. 1918. Studies in Crop Variation. I. An examination of the yield of dressed grain from Broadbalk. *Journal of Agricultural Science*, 11: 107-135.
- Fulbright, T. E. 1996. Viewpoint: a theoretical basis for planning woody plant control to maintain species diversity. *Journal of Range Management*, 554-559.
- Goulden, C. H. 1939. *Methods of statistical analysis*. New York: Wiley.
- Herrick, J. E., J. R. Brown, A. J. Tugel, P. L. Shaver, K. M. & Havstad. 2002. Application of soil quality to monitoring and management. *Agronomy Journal*, 94(1), 3-11.
- Herrick, J. E., W. G. Whitford, A. G. De Soyza, J. W. Van Zee, K. M. Havstad, C. A. Seybold, & M. Walton. 2001. Field soil aggregate stability kit for soil quality and rangeland health evaluations. *Catena*, 44(1), 27-35.
- Herrick, J. E., J. V. Zee, K. M. Havstad, L. M. Burkett, & W. G. Whitford. 2009. Monitoring manual for grassland, shrubland and savanna ecosystems. USDA-ARS Jornada Experimental Range. <http://jornada.nmsu.edu/monit-assess/manuals/monitoring> Accessed 22 March 2013.
- Marble, J. R., & Harper, K. T. (1989). Effect of Timing of Grazing on Soil-Surface Cryptogamic Communities in a Great Basin Low-Shrub Desert: A Preliminary Report. *Western North American Naturalist*, 49(1), 104-107.
- Meeuwig, R. O. (1970). Infiltration and soil erosion as influenced by vegetation and soil in northern Utah. *Journal of Range Management*, 185-188.
- Miller, R. F. 2005. *Biology, ecology, and management of western juniper (Juniperus occidentalis)*. Corvallis, Or.: Agricultural Experiment Station, Oregon State University.
- Miller, M. E., M. A. Bowker, R. L. Reynolds, & H. L. Goldstein. 2012. Post-fire land treatments and wind erosion—lessons from the Milford Flat Fire, UT, USA. *Aeolian Research*, 7, 29-44.
- Miller, R. F., J. C. Chambers, & M. Pellant. 2014. A field guide for selecting the most appropriate treatment in sagebrush and piñon-juniper ecosystems in the Great Basin: Evaluating resilience to disturbance and resistance to invasive annual grasses, and predicting vegetation response.

Miller, R. F., & R. J. Tausch. 2001. The role of fire in pinyon and juniper woodlands: a descriptive analysis. In Proceedings of the invasive species workshop: the role of fire in the control and spread of invasive species. Tall Timbers Research Station Miscellaneous Publication (Vol. 11, wpp. 15-30).

Monsen, S. B., R. Stevens, & N. L. Shaw. 2004. *Restoring western ranges and wildlands* (Vol. 2). United States Dept. of Agriculture, Forest Service, Rocky Mountain Research Station.

Owen, S. M., C. H. Sieg, C. A. Gehring, & M. A. Bowker. 2009. Above-and belowground responses to tree thinning depend on the treatment of tree debris. *Forest Ecology and Management*, 259(1), 71-80.

Paulson, D. D., & W. L. Baker. 2006. *The nature of southwestern Colorado: recognizing human legacies and restoring natural places*. University Press of Colorado.

Provencher, L., & J. Thompson. 2014. Vegetation Responses to Pinyon-Juniper Treatments in Eastern Nevada. *Rangeland Ecology and Management*, 67(2), 195-205.

R Core Team. 2014. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL <http://www.R-project.org/>. Accessed 9 February 2015.

Redmond, M. D., N. S. Cobb, M. E. Miller, & N. N. Barger. 2013. Long-term effects of chaining treatments on vegetation structure in piñon–juniper woodlands of the Colorado Plateau. *Forest Ecology and Management*, 305, 120-128.

Ross, M. R., S. C. Castle, & N. N. Barger. 2012. Effects of fuels reductions on plant communities and soils in a piñon-juniper woodland. *Journal of Arid Environments*, 79, 84-92.

Roundy, B. A., W. H. Blackburn, & R. E. Eckert Jr. 1978. Influence of prescribed burning on infiltration and sediment production in the pinyon-juniper woodland, Nevada. *Journal of Range Management*, 250-253.

Roundy, B. A., & J. L. Vernon. 1999. Watershed values and conditions associated with pinyon-juniper communities. *SB Monsen and R. Stevens, coordinators. Ecology and management of pinyon–juniper communities within the Interior West. USDA Forest Service, Rocky Mountain Research Station, Ogden, Utah, USA*, 172-187.

Tausch, R. J., & S. Hood. 2007. Pinyon/juniper woodlands. *Fire ecology and management of the major ecosystems of southern Utah. Fort Collins, CO, USA: US Department of Agriculture, Forest Service, RMRS-GTR-202*, 57-71.

- Tausch, R. J., & P. T. Tueller. 1977. Plant succession following chaining of pinyon-juniper woodlands in eastern Nevada. *Journal of Range Management*, 44-49.
- Simpson, E. H. 1949. Measurement of diversity. *Nature*.
- Schott, M. R., & R. D. Pieper. 1987. Succession of pinyon-juniper communities after mechanical disturbance in southcentral New Mexico. *Journal of Range Management*, 88-94.
- Skousen, J. G., J. N. Davis, & J. D. Brotherson. 1989. Pinyon-juniper chaining and seeding for big game in central Utah. *Journal of Range Management*, 98-104.
- Soil Survey Staff. 1999. *Soil taxonomy: A basic system of soil classification for making and interpreting soil surveys*. 2nd edition. Natural Resources Conservation Service. U.S. Department of Agriculture Handbook 436.
- Stevens, R., & S. C. Walker. 1996. Juniper-pinyon population dynamics over 30 years following anchor chaining. *United States Department of Agriculture Forest Service General Technical Report Int*, 125-128.
- Stevens, R. 1997, September. Mechanical chaining and seeding. In *SB Monsen and R. Stevens [COMPS.]. Proceedings—Ecology and Management of Pinyon-Juniper Communities Within the Interior West* (pp. 15-18).
- Stevens, R. 1999. Restoration of native communities by chaining and seeding. *Proceeding: Ecology and management of pinyon-juniper communities within the Interior West*, 285-289.
- United States Geological Survey. 2013. USGS GeoData Digital Orthophoto Quadrangles. Data available from the U.S. Geological Survey. <https://lta.cr.usgs.gov/DOQs>. Accessed 21 January 2013.
- Vernon, J. L., V. J. Anderson, & J. N. Davis. 2000. Response of residual native species after two-way chaining and seeding in Utah. *McArthur and DJ Fairbanks (COMPS.). Shrubland Ecosystem Genetics and Biodiversity Proceedings*, 13-15.
- Wilcox, B. P. 1994. Runoff and erosion in intercanopy zones of pinyon-juniper woodlands. *Journal of Range Management*, 285-295.
- Williams, G., G. F. Gifford, & G. B. Coltharp. 1972. Factors Influencing Infiltration and Erosion on Chained Pinyon-Juniper Sites in Utah. *Journal of Range Management*, 201-205.



## Conclusion

Land managers are tasked with maintaining diverse, productive rangelands. Their important decisions have broad effects on the environment and society. Management actions must be carefully considered and strategically implemented resource objectives (Swanson et al. 2006). The stakes are high and unintended consequences can have long lasting or even permanent effects on the landscape. It is important that managers carefully consider all likely potential outcomes when planning land management treatments such as Pinyon-juniper (*Pinus monophylla* – *Juniperus osteosperma*) removal.

Many tools are available to land managers who wish to remove Pinyon-juniper. Proven effective treatments for Pinyon-juniper removal include fire (Bruner and Klebenow 1979), chaining (Cain 1971), mastication (Johnson 1992), feller-buncher (Swan et al. 1997), hand thinning (Evans 1988), pile-and-burn (Gifford 1973), and herbicide (Tausch and Hood 2007). Different treatment types are known to have different advantages and disadvantages. Fire can remove nearly all vegetation, and easily cover large areas (Tausch and Hood 2007). However, fires can be unpredictable, hard to control, may burn more area than planned, and often result in conversions to invasive annuals after treatment (Miller and Tausch 2001). Hand thinning has the least impact on the soil, less danger of annual invasion, and works well on all slope types (Loftin 1999). Hand thinning does not remove the understory vegetation. Chaining kills large trees and some shrubs, but chaining is not as intense a disturbance as fire, and many more plants survive a chaining than a fire (Tausch and Tueller 1977). Chaining after a broadcast seeding helps cover seeds and incorporate them into the soil (Stevens 1999). However, it

also disturbs the soil surface, lowers soil stability, and increases erosion (Farmer et al. 1999; Redmond et al. 2013). Mastication compacts surface fuel loads, but does not remove fuel from the site (Hood and Wu 2006). Also, like chaining or a feller-buncher, mastication requires driving heavy equipment over a site which may lower soil stability or cause compaction. Pile and burn can remove fuel from a treatment area, but may negatively impact soil aggregate stability (Ross et al. 2012). Burned areas are also more susceptible to invasion by annuals. Selecting a treatment requires an understanding site conditions and effects of the various treatment types.

Land managers are faced with difficult decisions concerning Pinyon-juniper woodlands. They must consider when removal is appropriate, where, and what method should be used. Predicting results is difficult with so many interacting and unpredictable variables. For example, a seeding treatment will not take well in a drought (Chambers et al. 2007) or a boom in local rodents may eat the majority of the seeds (Nelson et al. 1970).

## **Management Implications Synthesis**

The six projects together represent only a small portion of the total number of Pinyon-juniper removal treatments on public lands. Chapter 1 examines four burned areas and provides information about how different parent materials respond to fire within the same elevation and precipitation. Chapter 2 and 3 both provide information about the effects of a single hand thinning and chaining respectively. However, there cannot be a quantitative conclusion drawn about how different treatments would impact the same site.

When land managers are selecting treatments they must consider a wide range of factors that are important in selecting treatments.

Very often, treatments are implemented on variable landscapes. In Nevada, those treatments are implemented at large scales. Often, there are many confounding environmental variables. When treatments do not respond as intended, it can be hard to identify which differences at the site are significant. In Chapter 1, the four burns were well paired with similar pre-treatment vegetation, but different soils. Finding sites such as these that are mostly similar and can be used as replicated is critical for obtaining the necessary data to try to explain why some treatments still do not always work as intended despite our best efforts.

There are still some qualitative conclusions that can be drawn from looking at how the projects relate to each other combined with knowledge from the literature. There were three projects within the White Pine Range – Cathedral, Curreant, and White Pine hand thinning. There are not any replicates of the hand thinning so that the treatment types could be effectively compared statistically, but it is interesting that the burns Cathedral and Curreant converted completely to cheatgrass while the White Pine hand thinning had very little cheatgrass in the post treatment community. It is possible this is due to the difference in treatment type, but the hand thinning also occurred in much different pre-treatment vegetation. Despite sharing many of the same soils between Curreant and White Pine, the White Pine hand thinning occurred in a phase I or early phase II Pinyon-juniper compared with phase III Pinyon-juniper at the other sites. Both of these factors were likely important, but without careful study and more replicates, it would be hard to

determine if the pre-treatment vegetation or treatment type difference was more important. It is likely that a prescribed burn on lower elevation limestone derived soils through phased III Pinyon-juniper in the White Pine range would convert to cheatgrass post-fire without intense post-fire restoration treatments. Prescribed burns would not be recommended in that area, but there could be a more desirable response to Pinyon juniper removal using prescribed fire in phase I Pinyon-juniper, at higher elevations with more precipitation, or using a different treatment type such as chaining or hand thinning.

The Holt chaining was just across the valley from the White Pine Range. It also had a positive response to treatment, but again there were too many things unique about the treatment to examine it with the others statistically. It was the only chaining and it occurred 30+ years earlier than the other treatments. The climate since then has changed and we cannot expect current treatments to react the same way under different climactic regimes. Still, it shows that chainings do have the potential to be successful treatments and have long term desirable impacts on vegetation and soils in an area. Chaining can still be an appropriate treatment for Pinyon-juniper removal.

The Elkhorn burns were on soils derived from different parent materials than the other treatment. Parent material is an important difference as shown in Chapter 1. The areas with soils derived from welded tuff should be managed differently than other areas in the district derived from limestone, granite, non-tuffaceous volcanic, or other parent materials. On soils similar to the Elkhorn burns, conversion to cheatgrass following fire is less likely. Prescribed burns are still relevant and can be useful in areas such as these that are resistant to cheatgrass.

For land managers, the successfulness of past treatments in similar locations can be a crucial guiding point. If planning a prescribed burn in one range, it is very useful to know the outcome of past prescribed burns in the same mountain range on similar soils. When available, the results of old treatments in similar areas should be referenced to better inform the treatment selection process.

### **When is Pinyon-Juniper Removal Appropriate?**

Pinyon-juniper removal is not always appropriate. The trees are an important part of the ecosystem in the Great Basin. Total elimination of Pinyon-juniper is not ever the landscape goal. Several types of areas should be totally avoided when planning treatments, especially old growth Pinyon-juniper woodlands and cultural areas used regularly for Pinyon nut collection.

Old growth Pinyon-juniper woodlands tend to be found in areas protected from large fires such as steep slopes with large amounts of rock outcroppings or other natural fire breaks (Miller et al. 1999). Old growth trees can be identified by their size and growth form without the need to core every or any tree. Additionally, coring can be incredible difficult in old desert trees (Many people who have attempted it have gotten their borer stuck. It is often necessary to cut down a tree to get an accurate ring count). Identifiable old growth characteristics for Utah juniper include fibrous bark with deep furrows, dead wood with lichen attached to the tree, a flattened or rounded crown, and dead terminal branch tips. Identifiable old growth characteristics for single-leaf Pinyon pine are similar, but the bark is thick and platy bark.

Pinyon pine nuts are a tasty and nutritious ancestral and modern food (Rosengarten 2004). They are harvested each year across the Great Basin by native people, commercial enterprises, and private individuals. Local tribes, commercial permit holders, and other members of the public should be asked where they regularly go to collect Pinyon nuts. Most users will have several areas they regularly use because Pinyon nut production varies from year to year based on precipitation (Breshears et al. 2005), so collectors need more than one site. Areas identified as culturally important Pinyon nut collection sites that regularly produce good crops of nuts should not be removed.

### **Where Should Be Priority Treatment Areas?**

Pinyon-juniper are broadly distributed and diverse. They dominate a quarter or more of the entire landmass of the Great Basin (Romme et al. 2009). They grow on a variety of soils types derived from many different parent materials. They grow in soils ranging from very shallow to deep. They occur on sites in association with many different sagebrush taxa. Managers should prioritize areas for treatment based on risks to resilience and resistance of the site. Managers should also prioritize areas of recent Pinyon-juniper expansion or infilling such as sagebrush shrubland being encroached by Pinyon-juniper.

There is a growing body of literature to help managers select the best treatments including agency programmatic treatment plans, scientific literature, and purpose made resources like the relatively new *A Field Guide for Selecting the Most Appropriate Treatment in Sagebrush and Pinyon-Juniper Ecosystems in the Great Basin* by Miller et al. (2014), the *Fire and Invasive Grass Assessment Tool (FIAT)* (Fire and Invasive Assessment Team 2014), or the *Fire and Fuels Management Contributions to Sage-*

Grouse Conservation (Havlina et al. 2014). The field guide includes a score sheet that can be used to rank the resilience and resistance of a site. Higher scores indicating greater resilience and resistance are given for cooler soil temperature, greater precipitation, and deeper soils. Mountain big sagebrush (*Artemisia tridentata* subsp. *vaseyana*) is more resistance and resilient than basin (*A. tridentata* subsp. *tridentata*) or Bonneville sagebrush (*A. tridentata* ssp. x *bonnevilleensis*) which are more resistant and resilient than Wyoming (*A. tridentata* subsp. *wyomingensis*), low (*A. arbuscula*), or black sagebrush (*A. nova*). Loamy soil is more resistant and resilient than silty, sandy, or clay loams which are more resistant and resilient than silt, sand, or clay. The highest resistance and resilience scores for vegetation are given to areas dominated by perennial grasses and forbs. Lower scores are given where perennial grasses and forbs are depleted to 5-15% foliar cover. Even lower scores are given where deep rooted perennial grasses have been depleted and replaced by Sandberg's bluegrass (*Poa secunda*). The lowest scores for vegetation are given for sites where perennial grasses are severely depleted and make up less than 5% of foliar cover. The total scores for temperature, moisture, and vegetation are then modified by the treatment type, low, moderate, and high severity and impacts on resilience and resistance.

After completing a rating form for an ecological site, land managers will be better informed about the potential impacts of their treatments. Sites with very low resilience and resistance should be managed to avoid disturbance. Sites with low resilience are probably not good candidates for high severity treatments, but can be prioritized for low severity treatments. Sites with moderate and high resilience are good candidates for

treatment types including high severity treatments. However their urgency is not yet great.

## **What Treatment to Select?**

After identifying areas where treatments would be appropriate, characteristics of the area can identify the most appropriate treatment types. Existing vegetation, soils, precipitation, slope, aspect, and ecological site properties should be examined and treatments selected that are appropriate for the area.

When burning, there is a large danger the post fire community will become dominated by invasive annuals that can change the fire cycle and physical soil properties to ensure they remain dominant (D'Antonio and Vitousek 1992). Once annual invasion has occurred, it is incredibly hard to reverse the process.

Fire can be used as a management tool in areas less likely to convert to annuals. Areas that have higher precipitation, tuffaceous soils, places with less pre-treatment tree cover, and greater pre-treatment grass cover are less likely to convert to annuals. In the Humboldt-Toiyabe districts studied, controlled burns would be more appropriate in the Monitor Range than the White Pine Range.

Fire is most appropriate in areas that are unlikely to convert to cheatgrass. This includes tuffaceous soils, higher elevation sites with greater precipitation (Young and Clements 2009), or areas far from existing cheatgrass seed sources. One of the best ways to resist cheatgrass invasion is with an effective seeding treatment that competes well (Chambers et al. 2007) In areas prone to cheatgrass, especially areas that get less than



250mm of precipitation per year, soils derived from limestone, or locations where old burns have converted to cheatgrass, fire should be avoided. Normally, a manager would not prescribe a fire if they were convinced that the area would convert to cheatgrass afterwards.

Hand thinning is appropriate when the shrub community is important in the short term such as for sage-grouse (*Centrocercus urophasianus*) habitat. Mastication can be used in areas where trees need to be removed, but there is less of a concern about damaging a percentage of the shrubs. A feller-buncher is effective at removing trees from a site. If a treatment is being done as a fire break, a feller buncher can remove a large amount of biomass from the site and reduce the available fuels in an area. A feller-buncher is one of the most expensive treatments for Pinyon-juniper removal (Provencher and Thompson 2014). Chaining is effective at removing trees and is good at incorporating seeds into the soil. However, it also disturbs the soil surface and may lower soil stability. Chaining is appropriate on resilient sites that are not susceptible to erosion.

## **Monitoring Successfulness**

All treatments should have stated quantitative objectives for success set before the treatment is implemented. After implementation, treatments should be monitored to determine if those objectives are reached. Ideally, monitoring would be ongoing, but limited budgets mean that treatments tend to be monitored for only short periods initially. For many land managers in the Great Basin, three years is the maximum time frame for which treatments are monitored. With irregular precipitation patterns across the Great Basin, three years is often not enough to determine if treatments have been successful.

Some seeds can remain dormant in the seedbank waiting for the next wet year to germinate (Young and Longland 1996). Long term studies have shown that how a treatment looks after three years is not very indicative of its long-term response or a good measure of overall success (Bates et al. 2005). Continued long-term monitoring is needed to ensure that areas declared successful remain that way and areas declared unsuccessful have not since begun to meet objectives.

Monitoring is especially useful when targeted to answer specific questions. Multiple projects should be examined that have similar pre-treatment conditions, so when there are varying responses to similar treatments, we can assume that there is an important variable that is not being monitored. At minimum, we already know that the pre-treatment vegetation community, climate, and soils are all critically important. Monitoring should record all of those conditions. Simply monitoring a plant community is not enough because widespread species occur on a variety of soil types which lead to differences in responses to treatment. Declaring a seeding treatment unsuccessful after three years is not productive when it takes six years for grasses to mature, especially if some of those years were in drought.

## Literature Cited

Bates, J. D., R. F. Miller, & T. Svejcar. 2005. Long-term successional trends following western juniper cutting. *Rangeland Ecology & Management*, 58(5), 533-541.

Bates, J. D., K. W. Davies, & R. N. Sharp. 2011. Shrub-steppe early succession following juniper cutting and prescribed fire. *Environmental management*, 47(3), 468-481.

Blackburn, W. H., & P. T. Tueller. 1970. Pinyon and juniper invasion in black sagebrush communities in east-central Nevada. *Ecology*, 51(5), 841-848.

- Bradley, B. A., & E. Fleishman. 2008. Relationships between expanding pinyon–juniper cover and topography in the central Great Basin, Nevada. *Journal of Biogeography*, 35(5), 951-964.
- Breshears, D. D., N. S. Cobb, P. M. Rich, K. P. Price, C. D. Allen, R. G. Balice,... & C. W. Meyer. 2005. Regional vegetation die-off in response to global-change-type drought. *Proceedings of the National Academy of Sciences of the United States of America*, 102(42), 15144-15148.
- Briske, D. D., B. T. Bestelmeyer, T. K. Stringham, & P. L. Shaver. 2008. Recommendations for development of resilience-based state-and-transition models. *Rangeland Ecology & Management*, 61(4), 359-367.
- Briske, D. D., S. D. Fuhlendorf, & F. E. Smeins. 2005. State-and-transition models, thresholds, and rangeland health: a synthesis of ecological concepts and perspectives. *Rangeland Ecology & Management*, 58(1), 1-10.
- Brooks, M. L., & J. C. Chambers. 2011. Resistance to invasion and resilience to fire in desert shrublands of North America. *Rangeland Ecology & Management*, 64(5), 431-438.
- Burkhardt, J. W., & E. W. Tisdale. 1976. Causes of juniper invasion in southwestern Idaho. *Ecology*, 472-484.
- Bruner, A. D., & D. A. Klebenow. 1979. Predicting success of prescribed fires in pinyon-juniper woodland in Nevada.
- Cain, D. R. 1971. The ely chain. *USDI Bur. Land Manage. Ely, Nev*, 9.
- Caudle, D., J. DiBenedetto, S. Karl, H. Sanchez, & C. Talbot. 2013. Interagency Ecological Site Handbook for Rangelands.
- Chambers, J.C. 2012. *Understanding the importance of resilience and resistance to restoration of sagebrush rangelands*. Society for Range Management 65th Annual Meeting. Spokane, Washington.
- Chambers, J. C., B. A. Roundy, R. R. Blank, S. E. Meyer, & A. Whittaker. 2007. What makes Great Basin sagebrush ecosystems invisable by *Bromus tectorum*?. *Ecological Monographs*, 77(1), 117-145.
- Coulloudon, B. 1999. Sampling vegetation attributes. BLM Technical Reference, 1734-4.
- D'Antonio, C. M., & P. M. Vitousek. 1992. Biological invasions by exotic grasses, the grass/fire cycle, and global change. *Annual review of ecology and systematics*, 63-87.

Evans, R. A. 1988. *Management of pinyon-juniper woodlands*. US Department of Agriculture, Forest Service, Intermountain Research Station.

Farmer, M. E., K. T. Harper, & J. N. Davis. 1999. The influence of anchor-chaining on watershed health in a juniper-pinyon woodland in central Utah. *Proceedings, ecology and management of pinyon-juniper communities within the Interior West. Proceedings RMRS-P-9. Ogden, UT. USDA Forest Service. Rocky Mountain Research Station, 299-301.*

Fire and Invasive Assessment Team. June 2014. Greater Sage-Grouse Wildfire, Invasive Annual Grasses & Conifer Expansion Assessment (Fire and Invasive Assessment Tool (FIAT)). Prepared by Fire and Invasive Assessment Team. (Appendix 5). 43 pages.

Floyd, M. L., & W. H. Romme. 2012. Ecological Restoration Priorities and Opportunities in Piñon-Juniper Woodlands. *Ecological Restoration, 30(1), 37-49.*

Gesch, D. B. 2007. The National Elevation Dataset, in Maune, D., ed., *Digital Elevation Model Technologies and Applications: The DEM Users Manual, 2nd Edition*: Bethesda, Maryland, American Society for Photogrammetry and Remote Sensing, p. 99-118.

Gesch, D., M. Oimoen, S. Greenlee, C. Nelson, M. Steuck, and D. Tyler. 2002 *The National Elevation Dataset: Photogrammetric Engineering and Remote Sensing*, v. 68, no. 1, p. 5-11.

Gifford, G. F. 1973. Runoff and sediment yields from runoff plots on chained pinyon-juniper sites in Utah. *Journal of Range Management, 440-443.*

Gruell, G. E., & S. Swanson. 2012. *Nevada's changing wildlife habitat: an ecological history*. University of Nevada Press.

Havlina, D.W., P. Anderson, L. Kurth, K.E. Mayer, J.C. Chambers,..., J. Vernon. 2014. *Fire and Fuels Management Contributions to Sage-Grouse Conservation*. Western Association of Fish and Wildlife Agencies. U.S. Fish and Wildlife Service, Denver, CO.

Herrick, J. E., J. V. Zee, K. M. Havstad, L. M. Burkett, & W. G. Whitford. 2009. Monitoring manual for grassland, shrubland and savanna ecosystems. USDA-ARS Jornada Experimental Range. <http://jornada.nmsu.edu/monit-assess/manuals/monitoring> Accessed 22 March 2013.

Herrick, J. E., W.G. Whitford, A. G. De Soyza, J. W. Van Zee, K. M. Havstad, C. A. Seybold, & M. Walton. 2001. Field soil aggregate stability kit for soil quality and rangeland health evaluations. *Catena, 44(1), 27-35.*

- Hood, S., & R. Wu. 2006, March. Estimating fuel bed loadings in masticated areas. In *Fuels Management—How to Measure Success: Conference Proceedings* (pp. 28-30).
- Johnson, R. E. 1992. Shred, don't burn--an alternative for treating slash on steep terrain. *Fire Management Notes*.
- Loftin, S. R. 1999. Initial response of soil and understory vegetation to a simulated fuelwood cut of a pinyon-juniper woodland in the Santa Fe National Forest. *Proceedings: Ecology and Management of Pinyon-Juniper Communities within the Interior West. USDA Forest Service Proc. RMRS-P-9. Ogden, UT, 311-314*
- Lowery, B., W. J. Hickey, M. A. Arshad, & R. Lal. 1996. Soil water parameters and soil quality. *Methods for assessing soil quality*, 143-155.
- Miller, R. F., J. C. Chambers, & M. Pellant. 2014. A field guide for selecting the most appropriate treatment in sagebrush and piñon-juniper ecosystems in the Great Basin: Evaluating resilience to disturbance and resistance to invasive annual grasses, and predicting vegetation response. Gen. Tech. Rep. RMRS-GTR-322-rev. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 66 p.
- Miller, R. F., & R. J. Tausch. 2001. The role of fire in pinyon and juniper woodlands: a descriptive analysis. In *Proceedings of the invasive species workshop: the role of fire in the control and spread of invasive species*. Tall Timbers Research Station Miscellaneous Publication (Vol. 11, wpp. 15-30).
- Miller, R., R. Tausch, & W. Waichler. 1999. Old-growth juniper and pinyon woodlands. In *USDA Forest Service Proceedings RMRS-P-9* (pp. 375-384).
- Milton, S. J., W. R. J. Dean, M. A. du Plessis, & W. R. Siegfried. 1994. A conceptual model of arid rangeland degradation. *BioScience*, 70-76.
- National Resources Conservation Service. 2006. Major Land Resource Areas of the United States, the Caribbean, and the Pacific Basin. *Agric. Handbk, 296*.
- Nelson, J. R., A. M. Wilson, & C. J. Goebel. 1970. Factors influencing broadcast seeding in bunchgrass range. *Journal of Range Management*, 163-170.
- Nowak, R. S., D. J. Moore, & R. J. Tausch. 1997, September. Ecophysiological patterns of pinyon and juniper. In *Proceedings of Ecology and Management of Pinyon-Juniper Communities within the Interior West* (pp. 15-18).
- Pierson, F. B., P. R. Kormos, & C. J. Williams. 2008, May. Hydrologic and Erosional Impacts of Pinyon and Juniper Encroachment Into Sagebrush Steppe Communities of the Great Basin, USA. In *Proceedings of the 15th International Congress of the International Soil and Water Conservation Organization*. Budapest, Hungary (pp. 1-4).

- PRISM Climate Group. 2013. Oregon State University. <http://prism.oregonstate.edu> Accessed 13 March 2013.
- Redmond, M. D., N. S. Cobb, M. E. Miller, & N. N. Barger. 2013. Long-term effects of chaining treatments on vegetation structure in piñon–juniper woodlands of the Colorado Plateau. *Forest Ecology and Management*, 305, 120-128.
- Romme, W. H., L. Floyd-Hanna, & D. D. Hanna. 2002. Ancient piñon-juniper forests of Mesa Verde and the West: A cautionary note for forest restoration programs. FIRE, FUEL TREATMENTS, AND ECOLOGICAL RESTORATION.[vp]. 16-18 Apr.
- Romme, W. H., C. D. Allen, J. D. Bailey, W. L. Baker, B. T. Bestelmeyer, P. M. Brown,... & P. J. Weisberg. 2009. Historical and modern disturbance regimes, stand structures, and landscape dynamics in pinon-juniper vegetation of the western United States. *Rangeland Ecology & Management*, 62(3), 203-222.
- Rosengarten Jr, F. 2004. *The book of edible nuts*. Courier Corporation.
- Ross, M. R., S. C. Castle, & N. N. Barger. 2012. Effects of fuels reductions on plant communities and soils in a piñon-juniper woodland. *Journal of Arid Environments*, 79, 84-92.
- Roundy, B. A., R. F. Miller, R. J. Tausch, K. Young, A. Hulet, B. Rau,... & D. Eggett. 2014. Understory cover responses to piñon-juniper treatments across tree dominance gradients in the Great Basin. *Rangeland Ecology and Management*, 67(5), 482-494.
- Provencher, L., & J. Thompson. 2014. Vegetation Responses to Pinyon-Juniper Treatments in Eastern Nevada. *Rangeland Ecology and Management*, 67(2), 195-205.
- Seybold, C. A., & J. E. Herrick. 2001. Aggregate stability kit for soil quality assessments. *Catena*, 44(1), 37-45.
- Shaff, S. E., D. A. Pyke, A. I. Lindgren, E. W. Schupp, J. Burnham, P. S. Doescher, & J. Chambers. 2012. Inter-perennial gaps may indicate invasibility of sagebrush systems. Proceedings from 97th ESA Annual Convention.
- Soil Survey Staff. 1999. *Soil taxonomy: A basic system of soil classification for making and interpreting soil surveys*. 2nd edition. Natural Resources Conservation Service. U.S. Department of Agriculture Handbook 436.
- Soil Survey Staff. 2015. Natural Resources Conservation Service. United States Department of Agriculture. Web Soil Survey. Available online at <http://websoilsurvey.nrcs.usda.gov/>. Accessed 15 March 2015.

- Stevens, R. 1999. Restoration of native communities by chaining and seeding. *Proceeding: Ecology and management of pinyon-juniper communities within the Interior West*, 285-289.
- Stringham, T. K., W. C. Krueger, & P. L. Shaver. 2003. State and transition modeling: an ecological process approach. *Journal of Range Management*, 106-113.
- Swan, L., J. McNeel, E. Nicita, B. M. Region, L. R. Ranch, & W. Tenino. 1997. Western juniper harvest systems comparisons project. *Final Report, Klamath County Economic Development Association, Klamath Falls, OR*.
- Swanson, S., B. Bruce, R. Cleary, B. Dragt, G. Brackley, G. Fults,... and D. Wilson. 2006. *Nevada rangeland monitoring handbook*. 2nd edition.
- Tausch, R. J., & S. Hood. 2007. Pinyon/juniper woodlands. *Fire ecology and management of the major ecosystems of southern Utah*. Fort Collins, CO, USA: US Department of Agriculture, Forest Service, RMRS-GTR-202, 57-71.
- Tausch, R. J., R. F. Miller, B. A. Roundy, & J. C. Chambers. 2009. Pinon and Juniper Field Guide: Asking the right questions to select appropriate management actions. Page 95 in U. D. o. t. Interior and U. G. Survey, editors. U.S. Geological Survey, Reston, Virginia.
- Tausch, R. J., & P. T. Tueller. 1977. Plant succession following chaining of pinyon-juniper woodlands in eastern Nevada. *Journal of Range Management*, 44-49.
- Tausch, R. J., & N. E. West. 1988. Differential establishment of pinyon and juniper following fire. *American Midland Naturalist*, 174-184.
- United States Geological Survey. 2012. USGS National Agriculture Imagery Program. Data available from the U.S. Geological Survey. <https://lta.cr.usgs.gov/NAIP>. Accessed 18 December 2012.
- United States Geological Survey. 2013. USGS GeoData Digital Orthophoto Quadrangles. Data available from the U.S. Geological Survey. <https://lta.cr.usgs.gov/DOQs>. Accessed 21 January 2013.
- Weisberg, P. J., E. Lingua, & R. B. Pillai. 2007. Spatial patterns of pinyon-juniper woodland expansion in central Nevada. *Rangeland Ecology & Management*, 60(2), 115-124.
- Western Regional Climate Center, Desert Research Institute. 2015. Available at <http://www.wrcc.dri.edu/summary/Climsmnv.html> Accessed 19 February 2015.
- Westoby, M., B. Walker, & I. Noy-Meir. 1989. Opportunistic management for rangelands not at equilibrium. *Journal of range management*, 266-274.

Wisdom, M. J., & J. C. Chambers. 2009. A landscape approach for ecologically based management of Great Basin shrublands. *Restoration Ecology*, 17(5), 740-749.

Young, J. A., & C. D. Clements. 2009. *Cheatgrass: fire and forage on the range*. University of Nevada Press.

Young, J. A., & W. S. Longland. 1996. Impact of alien plants on Grant Basin rangelands. *Weed Technology*, 384-391.