

AN ABSTRACT OF THE THESIS OF

Corinne N. Morozumi for the degree of Master of Science in Wildlife Science presented on August 19, 2015.

Title: Vegetation and Small Mammal Responses to Western Juniper (*Juniperus occidentalis*) Control in Eastern Oregon

Abstract approved:

Dana M. Sanchez

Shrub-steppe ecosystems of western North America provide habitat for many wildlife species, are important components of public and private rangelands, and offer recreational opportunities for millions of people. They are some of the most vulnerable ecosystems in the United States and have been altered by human activities such as livestock grazing, active fire suppression, conversion to agriculture, and urbanization of the west. Since the late 1800s woody encroachment of piñon-juniper species has also contributed to the loss of shrub-steppe habitat. Presently, land managers remove woody tree species in order to recover shrub-steppe although it is unclear how responses to these management activities differ due to site-specific conditions and existing woodland development. I studied post juniper-thinning responses in eastern Oregon at a wildlife area important as winter range for mule deer (*Odocoileus hemionus*). I investigated vegetative responses to western juniper (*Juniperus occidentalis*) thinning across a woodland development gradient and tested for interactive effects of juniper cutting and cattle exclusion. In addition, I explored plant and small mammal successional dynamics after juniper thinning and examined plant community responses within microhabitats created by the felled trees.

I compared vegetative responses to juniper thinning as well as cattle exclusion among sites where juniper were subordinate (Phase I), co-dominant (Phase II), and dominant to shrubs and grasses (Phase III, sensu Miller et al. 2005). At the Phase I site, thinning did not increase herbaceous biomass while thinned plots at the Phase II site had 2.71 times more median herbaceous biomass (99.17% CI: 1.37 to 5.37 times more biomass) than unthinned plots. Conversely, herbaceous biomass at the Phase III juniper woodland site was 0.36 times lower in thinned plots when compared to unthinned plots (99.17% CI: 0.17 to 0.78 times less biomass). Unfortunately, many of the responses were driven by exotic species release. Where juniper were subordinate (Phase I), mean percent cover by exotic grass increased by 24.58 percentage points in

thinned plots as compared to unthinned plots (98.30% CI: 0.27 to 48.90 percentage points higher). Thinning at the Phase II site increased mean exotic grass cover by an estimated 28.47 percentage points as compared to mean exotic cover in the same plots before treatment (98.30% CI: 4.15 to 52.79 percentage points higher cover). Median native bunchgrass cover at this site was 5.06 times greater after juniper treatment (99.7% CI: 1.78 to 14.35 times higher percent cover). I found few main or interactive effects of cattle exclosure after one year of treatment. Responses to grazing exclosure may take longer to develop. These results indicate that sites within the wildlife area respond differently to juniper management and that exotic grass control will be key to successful shrub-steppe recovery.

In addition, I used a time-since-juniper thinning chronoseries consisting of plots cut in 2008, 2009, 2010, 2012, and an adjacent uncut control to explore how shrub-steppe flora and fauna are responding to juniper treatment through time. Shrub cover and seedling density were low in each plot. I recorded the highest seedling abundance (mean of 0.25 seedlings) in the uncut control plot. Median grass cover in the uncut control was 9.50% while in the most recently treated plot (2012) it was 26.75%. Small mammal relative abundance and diversity was low across all time-since-treatment plots though highest in the plot with the greatest time-since-treatment. Deer mouse (*Peromyscus maniculatus*) was the most abundant species in all plots and accounted for 70-95% of all unique captures. Least chipmunk (*Tamias minimus*) were present in the plot with greatest time-since-treatment indicating the potential recovery of key native shrubs since these small mammals perform an important seed dispersal role.

I also investigated how potential microhabitats created by the felled juniper might support different plant communities. I assessed plant responses within zones created by 1) the felled tree (canopy zone), 2) the original duff zone, and 3) the between – tree interspace zone. The three zones had different plant community compositions as analyzed multivariately with non-metric multidimensional scaling (NMS). Canopy and duff zones were forb – dominated and had less exotic grass invasion while interspace zones were heavily invested with medusahead (*Taeniatherum caput-medusae*) and ventenata (*Ventenata dubia*). Exotic herbaceous productivity was lowest under the standing trees of the control plot and high in the interspace of all plots with juniper thinning. In the 2012 cut plot, grass cover was 27.90 percentage points less in the canopy zone compared to the interspace zone (98.75% CI: 17.64 to 38.16 percentage points less grass cover). In the uncut control, median native biomass was 2.08 times greater under the standing trees compared to the interspace (96.25% CI: 1.09 to 3.97 times more biomass). Results from the time-since-treatment chronoseries indicated that exotic grass dominance might be limiting shrub-steppe recovery. Active restoration of shrubs and native grasses may be necessary to address the

dominance of exotic grass after juniper thinning. Positive signs of habitat recovery included small mammal responses though I was unable to calculate population estimates due to plot size.

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Vegetation and Small Mammal Responses to Western Juniper (*Juniperus occidentalis*) Control in
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Major Professor, representing Wildlife Science

Head of the Department of Fisheries and Wildlife

Dean of the Graduate School

I understand that my thesis will become part of the permanent collection of Oregon State University libraries. My signature below authorizes release of my thesis to any reader upon request.

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CONTRIBUTIONS OF AUTHORS

My two advisors Dr. Dana Sanchez and Dr. Ricardo Mata-González assisted with study design, data collection, analysis, and editing of all chapters of this thesis. Dr. Sanchez also contributed in acquiring funding and research supplies.

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CHAPTER 1: INTRODUCTION TO THE THESIS

Rapid landscape level changes across the western United States have altered habitat quality in many ecosystems. One widespread change has been the expansion of piñon-juniper woodlands into neighboring ecosystems such as shrub-steppe. Conifer woodlands of this type are believed to have occupied only 3 million hectares before the 1800s, while they now occupy an estimated 30 million hectares across the western United States (Miller and Tausch 2000). Western juniper (*Juniperus occidentalis*) occupies 3.6 million hectares that previously had little to no conifer cover (Azuma et al. 2005; Miller et al. 2005). While many site-specific factors may be playing a role in the widespread community change, overgrazing as well as fire suppression are often cited as the main drivers of this woody encroachment. Furthermore, there are concerns that rapid climate change may further exacerbate the issue (Miller et al. 2005). Juniper establishment is likely regulated by fire, with seedlings being excluded from sites by high intensity fire and competition with other plants (Miller and Rose 1995; 1999; but see Romme et al. 2009). Fire return intervals have lengthened due to direct wildfire suppression (Burkhardt and Tisdale 1976). In addition, many sites now lack the fine fuels to sustain a juniper-killing fire through an area. These changes have led to increasingly dense stands of western juniper as well as a distributional expansion into, or invasion of many ecosystems. Trees that were present prior to this rapid expansion have been termed *pre-settlement* individuals while younger trees are called *post-settlement* individuals. Canopy closure of existing stands of pre-settlement (145-over 1000 years old) juniper is occurring as numerous post-settlement (less than 145 years old) trees, released from the population-controlling fires, have survived to reach mature height. In ecosystems where juniper were absent or rare, the changes in regulation have allowed the species to pioneer and prosper in spreading stands of post-settlement trees.

Most expansion of western juniper has been into sagebrush ecosystems though aspen forests have also been impacted (Miller et al. 2005; Miller and Rose 1995). Miller et al. (2005) separated juniper woodland succession into three phases. In Phase I juniper trees are subordinate to other vegetation and are thought to play a minor role in ecological processes on the site. As tree density increases, juniper woodlands transition into Phase II woodlands in which juniper, shrub, and herbaceous plants are co-dominant. Phase III woodlands are characterized by juniper dominance that drives most ecological processes on the site. Juniper encroachment leads to loss

of shrub cover, productivity, and plant density (Adams 1975; Bunting et al. 1999; Burkhardt and Tisdale 1969; Miller et al. 2000; Roberts and Jones 2000). Increasing dominance by juniper also reduces forb and grass cover in drier sites with shallow soils or those that are already degraded (Miller et al. 2005). Juniper dominance alters the spatial distribution of nutrients by the formation of resource islands under the canopy (Doescher et al. 1987). Dominance by western juniper is associated with increased rill erosion and sediment loss, which is thought to lead to a loss of nutrients from the system (Buckhouse and Mattison 1980; Pierson et al. 2013). Juniper removal increased available soil water at a mountain big sagebrush (*Artemisia tridentata subsp. vaseyana*) -Thurber's needle grass (*Achnatherum thurberianum*) association site with late juniper succession (Bates et al. 2000) though the impact to ecosystem level hydrological processes and water balance are not well understood (Miller et al. 2005).

In previously productive sage steppe habitats, increasingly dense juniper woodlands negatively affect wildlife species of concern such as the greater sage-grouse (*Centrocercus urophasianus*) (Baruch-Mordo et al. 2013), as well as economically important species such as big game animals, by reducing shrub cover. Relatively few studies though have directly examined the effect of juniper expansion in shrub-steppe on wildlife (Maser and Gashwiler 1978; Willis and Miller 1999). Compositional shifts in species that use these habitats have been found (reviewed in Miller et al. 2005; Reinkensmeyer et al. 2007). Early phase woodlands can be attractive to some wildlife species such as passerine birds due to the increased vertical diversity, nesting locations, and juniper berry production (Maser and Gashwiler 1978) but a net decrease in the faunal diversity of birds and mammals is expected as juniper woodland succession proceeds (Noson et al. 2006; Reinkensmeyer 2000).

Due to the various negative effects associated with late stage juniper woodland succession, land managers throughout the species' range have attempted to reduce density of post-settlement juniper. Restoration strategies including mechanical removal and prescribed fire have been implemented since the early 1960s (Miller et al. 2005). Currently, cutting individual trees by chainsaw is the most common method of juniper control though fire is still used on private lands and where understory fuel is still abundant enough to carry fire between trees (Miller et al. 2005).

While juniper is managed across its range on patches of both public and private lands, knowledge gaps still exist about ecosystem response to treatment and post-treatment community

dynamics. Past work compared spatially distinct sites within a common plant association type to evaluate the effect woodland successional phase had on response to juniper control (Bunting et al. 1999; Miller et al. 2000). However data are lacking on vegetative responses in cut and uncut plots across all woodland phases at one site.

The Phillip W. Schneider Wildlife Area (hereafter wildlife area) is an Oregon Department of Fish and Wildlife-owned site in Grant County that was purchased in the 1970s in order to conserve and manage critical winter range habitat for mule deer (*Odocoileus hemionus*). Bitterbrush (*Purshia tridentata*) stands provide high quality winter forage and cover for mule deer. These shrub stands are associated with lower predation by cougars and therefore may offer more protection than more open habitat dominated by smaller shrubs such as rabbitbrush (Pierce et al. 2004). Increased density and canopy cover of western juniper has coincided with invasion by several non-native species (i.e., medusahead (*Taeniatherum caput-medusae*) and ventenata (*Ventenata dubia*)). The resulting loss of habitat quality for many of the wildlife area's species and subsequent decrease in mule deer populations prompted active management of juniper via thinning.

Previous juniper removal studies have illustrated positive temporal responses of understory vegetation, especially perennial grasses, in treatment plots where juniper trees were cut compared to uncut control plots (Bates et al. 2000; 2005; Eddleman 2002). However these studies did not account for other management practices important to aboveground biomass production such as grazing, nor did they consider the effects of juniper treatment in the presence of invasive understory species. Rangelands of the intermountain west were historically subjected to intense grazing regimes, which led to degradation in many habitats since the introduction of domestic livestock in the 19th century (Sayre et al. 2012). Better grazing management techniques are now implemented, including limited stocking rates, active rotation and seasonal rest. In arid environments grazing may negatively affect plant diversity (Olf and Ritchie 1998) and plant primary productivity (Milchunas and Lauenroth 1993) potentially exacerbating understory response to juniper encroachment and making recovery more difficult. Whether grazing influences recovery after juniper management is not well understood. In one study, grazing after juniper removal decreased perennial grass productivity and seed production (Bates 2005). Therefore, I tested the independent and interactive effects of livestock grazing and juniper cutting at three sites within the wildlife area (Chapter 2). Oregon Department of Fish and Wildlife

managers had classified each site as a different juniper woodland phase (I-III). This enabled me to explore the responses of understory vegetation to the release from livestock grazing pressure as well as the release from overstory juniper canopy regulation across different levels of juniper woodland development. I investigated the ways in which vegetative responses to juniper treatment varied between grazed and ungrazed plots. My objectives for Chapter 2 were to understand the potentially main and interactive effects of grazing, juniper cutting, and juniper woodland phase on understory plant productivity (g/m^2) and foliar cover (%).

Successional dynamics has been a major focus of community ecology for the past century (Beisner et al. 2003; Clements 1916; Connell and Slatyer 1977; Gleason 1927; McIntosh and Odum 1969). While traditional succession processes such as transition to shrub-steppe stable state communities have been found following juniper cutting in some cases (Bates et al. 2005), many factors such as existing site conditions (Young et al. 1985) and alternative stable states affect the outcome after restoration (Briske et al. 2008). These dynamics are important as they will likely determine whether additional restoration efforts are needed once the juniper overstory has been removed. Ecological thresholds that prevent a site from returning to a state of high quality shrub-steppe habitat likely exist (Petersen et al. 2009) and should also influence management decisions by giving managers the predictive power to determine the likelihood of restoration success.

Knowledge gaps persist regarding the effect of time-since-treatment on vegetative understory and small mammal responses. Although data exist from long term monitoring of vegetative response to western juniper thinning (Bates and Svejcar 2009; Eddleman 2002), we understand less about the response of small mammals to such management action. Often overlooked, small mammals are potentially important to understanding shrub-steppe communities and their recovery potential. Chipmunks are major dispersers of bitterbrush shrub seeds (Vander Wall 1994), a key shrub component at the wildlife area. At the wildlife area managers have removed juniper from multi-hectare patches for several years. This offered an opportunity to understand the temporal dynamics of plants and animals after juniper cutting without having to implement a long-term study. I used a chronoserries with plots that differed in their time since juniper removal to estimate recovery of shrub-steppe habitat (Chapter 3). In addition, I examined how fine scale vegetative responses differed at microsites both within and across the chronoserries. My objectives for Chapter 3 were to 1) understand how time since juniper cutting influenced herbaceous productivity (g/m^2) and foliar cover (%), 2) determine if shrub seedling

recruitment was occurring within the plots, 3) compare responses between microsites both within and across chronoseried plots, and 4) compare relative abundance and diversity of small mammal across the chronoseried.

Results from this work will provide a foundation for understanding important aspects of our efforts to restore shrub steppe habitats along the juniper expansion front. My work addresses the influences of juniper woodland phase, livestock exclusion, and time-since-juniper removal on understory vegetation composition and productivity, along with relative abundance of small mammals. These findings will benefit managers tasked with restoring habitat quality for many shrub steppe species, including economically important game species such as mule deer.

CHAPTER 2: VEGETATIVE RESPONSES TO CATTLE EXCLUSION AND JUNIPER
THINNING WITHIN JUNIPER-INVADDED SHRUB-STEPPE IN EASTERN OREGON

Corinne N. Morozumi, Ricardo Mata-González, and Dana Sanchez

Fisheries and Wildlife Department
Oregon State University
Corvallis, OR

ABSTRACT

Expansion of western juniper (*Juniperus occidentalis* Hook.) into shrub-steppe ecosystems is responsible for declines of understory productivity, competitive elimination of shrubs throughout the northern Great Basin, and loss of habitat for fauna of the region. In an effort to improve shrub-steppe habitat quality and restore these imperiled communities, juniper thinning efforts have begun on public and private land across the region. While there is some consensus on the plant successional dynamics involved after thinning treatments it is unclear how low intensity grazing pressure by livestock might influence restoration success. In addition, few studies have evaluated the effects of these two factors across a gradient of juniper encroachment. In this study we evaluated herbaceous plant recovery within plots with four treatments across three areas where each area represented a different phase of juniper woodland succession. Responses of vegetative cover and biomass to juniper thinning were generally dependent on area while responses to grazing were similar across the three areas in the study. Herbaceous cover of native bunchgrasses increased in plots after thinning but this result was only significant at the area in Phase II condition with codominant juniper and shrub cover and relatively good initial site condition. Juniper thinning apparently released exotic grasses from overstory control and thinned plots had more exotic biomass and cover as compared to unthinned plots. These results highlight the role preexisting community structure and integrity has on restoration outcomes of juniper-invaded shrub-steppe.

INTRODUCTION

Piñon-juniper woodlands in the Intermountain West have rapidly expanded since the late 1800s (Miller et al. 2008; Tausch et al. 1981). Within the northern region of the Great Basin, western juniper (*Juniperus occidentalis* Hook.) has expanded into neighboring communities within its range since the second half of the 19th century and currently occupies 3.6 million hectares in eastern and central Oregon, northeastern California, as well as in northwestern Nevada and southwestern Idaho (Azuma et al. 2005; Miller et al. 2005). Before European settlement of the region, western juniper was generally relegated to rocky ridges and low sagebrush flats with shallow or unproductive pumice type soils. However, the species now occupies sites with deeper, more nutrient rich soils (Miller et al. 2005). Fire suppression likely contributed to releasing the species from the regulatory pressure of high intensity fire and plant competition (Miller and Rose

1995; Miller et al. 2008; but see Romme et al. 2009). In addition, historic overgrazing has contributed to the loss of fine fuel loads needed to carry a fire through an area. Mean fire return intervals for sagebrush grassland communities were historically 10 – 25 years but have increased to over 100 years across much of central and eastern Oregon (Miller and Tausch 2000). In general, western juniper is now found at higher densities as well as in plant communities previously dominated by other types of vegetation.

Transition to western juniper woodland can occur through rapid succession of plant community stages. Phase I woodlands are characterized by juniper being subordinate to shrubs and herbaceous plants. Once juniper is codominant with other vegetation and plays a role in shaping ecological processes such as hydrology and nutrient cycles, the woodland is classified as Phase II of juniper succession. Dominance by juniper, at which point the trees are the primary vegetative layer driving ecological processes, completes the succession into a Phase III woodland (Miller et al. 2005). Juniper trees established before the late 1800s are categorized as *presettlement trees* and were historically found in stands with less than 10 percent tree canopy cover. *Postsettlement trees* are those that established within the last 130 years and are found in stands with greater than 10 percent canopy cover (Miller et al. 2005).

The competitive advantage gained by western juniper in the last century has come at the detriment of plant communities such as sagebrush grasslands, shrub-steppe and aspen forests (Miller et al. 2005; Miller and Rose 1995). Dominance of a community by western juniper reduces shrub and herbaceous cover, production, and density (Adams 1975; Bunting et al. 1999; Burkhardt and Tisdale 1969; Roberts and Jones 2000; Schafer et al. 2003). There is also evidence that this expansion negatively impacts grasses and forbs, though this response might be dependent on the site characteristics such as shallow, root-restricting layers of soil and existing type of herbaceous cover or plant association type (Miller et al. 2005; Miller et al. 2000).

Due to the loss of understory productivity and cover induced by juniper woodland development, managers have begun thinning postsettlement juniper trees as a restoration treatment. Recovery after juniper thinning generally involves increased native plant cover, species diversity and primary productivity (Bates et al. 2000; Bates and Svejcar 2009; O'Connor et al. 2013; Rose and Eddleman 1994; Vaitkus and Eddleman 1986). After cutting, ecosystems often progress towards sagebrush grassland climax communities via traditional plant succession. However restoration outcomes can be complicated by factors such as invasion of non-native plant

species, concomitant interactions with fire and existing organisms, and the potential effects of cattle grazing. Where a site begins on a successional progression depends on the level of woodland closure found before treatment, existing understory vegetative cover, and presence of perennial species (Bates et al. 2000). Multiple stable states might exist that would make recovery to historic plant community type difficult or impossible (Miller et al. 2005; Petersen et al. 2009). Sites heavily invaded by non-native annual grasses might instead transition to an annual-dominated phase, for example. Studies are needed that evaluate the effects of juniper removal at sites with multiple drivers of plant community alteration.

One such change was the introduction of livestock to semi-arid ecosystems of western North America. Rangelands of the intermountain west were historically subjected to intense, poorly-timed grazing regimes, which led to degradation in many habitats after the introduction of domestic livestock in the 19th century (Sayre et al. 2012). Better grazing management techniques are now implemented, including limited stocking rates, active rotation and seasonal rest. In arid environments grazing may have little impact on or might negatively affect plant diversity (Olff and Ritchie 1998), though plant primary productivity is generally negatively affected by grazing (Milchunas and Lauenroth 1993). Impact to native vegetation may be mediated by interannual climatic variability producing stronger negative effects to native vegetative cover in dry years (Loeser et al. 2007). In western juniper woodlands of the northern Great Basin little is known about the effects of domestic grazing soon after juniper cutting. Only one study has assessed the effect of grazing in cut and uncut juniper woodlands. The experiment was conducted with short (4-5 days), intense grazing (0.78-0.96 cow-calf pairs per hectare) which was halted when 80% of perennial grass cover was removed or when grass development reached boot stage. Grazing had little effect on understory herbaceous cover though perennial grass productivity and seed production decreased (Bates 2005). Grazing regimes within managed juniper woodlands and shrub steppe ecosystems in Oregon tend to be of lower intensity for longer periods of time, with grazed periods ranging 1-5 months in duration (personal communication Molly Galbraith, June 2015). Current literature lacks data on how this type of grazing prescription affects recovery of native understory community composition, cover, and forage quantity and quality once juniper trees are felled and left in place.

We assessed the vegetative responses of herbaceous cover and productivity as well as shrub seedling recruitment to both juniper thinning and low intensity livestock grazing as well as

their potential treatment interactions. Our objective was to explore these responses among three sites that represented the three phases of juniper expansion within the same management area and grazing regime. We expected that thinning of juniper would tend to increase herbaceous productivity on all sites (Bates et al. 2000; 2005; Eddleman 2002) but would depend on grazing treatment and site. Percent cover of functional group (forbs, native bunchgrasses, shrubs) would respond differently to thinning (Bates and Svejcar 2009; O'Connor et al. 2013; Vaitkus and Eddleman 1986). Removal of juniper overstory would result in increased available resources such as light and nutrients. We expected exotic annual grasses would be able to take advantage of these changes quickly while slower growing shrubs would likely respond less vigorously. We expected functional plant groups susceptible to grazing such as native bunchgrasses, would respond to grazing negatively by decreasing biomass and cover. We expected browsing and grazing cattle to decrease cover of native shrubs and grass because palatability of invasive grass is low in the study area and because perennial bunch grasses often decrease with grazing (Milchunas and Lauenroth 1993). We expected grazing would also negatively affect primary productivity of native grasses and forbs.

MATERIALS AND METHODS

Site Description

We evaluated the effects of livestock grazing and mechanical juniper removal on vegetation understory within the Phillip W. Schneider Wildlife Area (lat 44°17'33"N, long-119°28'32"W). The wildlife area is located between the Ochoco and Malheur National Forests in the John Day Ecological Province approximately thirty miles west of John Day, OR. The John Day Ecological Province includes roughly 2.8 million hectares of central Oregon comprised of many finely dissected hills with vast plateaus and lone buttes. The hills and valleys are mainly composed of calcareous clayey soils originating from sedimentary and tuffaceous sources (Anderson et al. 1998).

The wildlife area receives an average of 34 centimeters of total annual precipitation (PRISM Climate Group 2004) with most of the precipitation falling in the winter months. From May to September precipitation per month averages 2.3 cm (ODFW weather station, Dayville OR). Annual snowfall is highly variable from 6 – 100 cm each winter on average. Annual

temperatures average 28°C in summer and -4°C in the winter. Elevation at the study site ranges from 1000 – 1500 m.

The entire wildlife area consists of 21,000 hectares of public land. The site is managed by Oregon Department of Fish and Wildlife (ODFW) and contains both ODFW deeded land as well as parcels owned by the Bureau of Land Management (BLM). Juniper woodlands of all three juniper densities (Phase I, II, and III) are represented in the wildlife area. The area is important winter range habitat for mule deer (*Odocoileus hemionus* Rafinesque) (Oregon Department of Fish and Wildlife 2011). Management action, including juniper thinning, is intended to increase usable habitat, habitat quality and browse availability for mule deer. Our three sites were within the ODFW-owned land: Murderer's Creek (MC), Flat Creek (FC), and Wiley Gulch (WG). Managers used site characteristics to designate juniper woodland status (sensu Miller et al. 2005): MC was estimated to be a Phase I site on which juniper cover averaged 1.8% \pm 2.5%. Due to the persistence of a shrub understory, FC was classified as Phase II of woodland development with an average of 19.2% \pm 12.3% canopy cover by juniper. Juniper canopy cover on the WG site was 11.4% \pm 16.1% but the virtual absence of almost all shrubs led to its designation as a Phase III juniper woodland (Ryan Torland, personal communication, October 2013). All three areas are within 15km of each other and are managed similarly. The historic native vegetation was a shrub and perennial bunchgrass steppe characterized by big sagebrush (*Artemisia tridentata* subsp. *tridentata* Beetle & Young, Welsh, *Artemisia tridentata* subsp. *wyomingensis* Beetle & Young, and *Artemisia tridentata* subsp. *vaseyana* Rydb. Beetle, low sagebrush (*Artemisia arbuscula* Nutt.) and antelope bitterbrush (*Purshia tridentata* Pursh). Since the 1980s, annual exotic grasses such as cheatgrass (*Bromus tectorum* L.), medusahead and most recently ventenata have invaded the wildlife area with the heaviest infestations near roads, disturbed areas, and in lowland flats.

Oregon Department of Fish and Wildlife implements a rest-rotation grazing program with two years of grazing in sequential months (May and June, typically) followed by a year of rest. The area includes 20 different pastures that are separated into primarily May and June use though several pastures are utilized until the beginning of September. Lower elevation sites are generally grazed earlier in the season as range readiness is determined by growing season (Daniel Marvin, personal communication, October 2013). Stocking rates are low (less than 0.1 cows/acre) and are determined using the animal unit per month (AUM) method.

EXPERIMENTAL DESIGN

We performed a blocked split-plot factorial experiment by manipulating juniper canopy cover and constructing grazing exclosures at the three sites, MC, FC, and WG ($n = 6$ blocks per site). In June 2012, ODFW placed 100 meter² (1 ha) blocks non-randomly to control for slope and aspect. We constrained block location to areas with constant northerly aspect with less than five degrees of slope. In fall 2012, we fenced 0.5 hectares (50x100m) of each block, using randomization to determine which 0.5 ha area was to be fenced. We then divided each 0.5 ha area into two plots, which we randomly assigned to either cut or uncut juniper treatments. Therefore each block consisted of four plots with treatment combinations: 1) uncut and grazed (UCG), 2) uncut and ungrazed (UCUG), 3) cut and grazed (CG), and 4) cut and ungrazed (CUG). We mechanically felled with chainsaw all juniper trees in the cut treatment plots within the same water year. We established treatments at Water Gulch in December 2012, at Flat Creek in March-April 2013, and at Wiley Gulch in October 2013. Because each juniper woodland phase was replicated only once assessing causation of effect of woodland phase on vegetative responses was not possible.

Vegetative sampling

In July 2012 we measured pretreatment understory vegetation in each plot along 30 m permanent transects. We established each transect using a randomly selected starting point and azimuth. We recorded cover (live vegetation by species, cut juniper, bare ground and duff) with the point intercept measurement method every 0.5 meters along the 30 meter transect in order to estimate percent cover. We collected standing live biomass of herbaceous species from a 0.75 x 0.75 m quadrat at each end of the transect clipped to 1 cm stubble. We dried biomass for 48 hours at 100°C before weighing dry mass to 0.01 grams. In 2014 we recorded posttreatment vegetative cover using point intercept and collected biomass from the same transects in each plot. We divided the 2014 biomass samples into native and exotic groups in the field. In 2014, we also recorded native shrub seedlings within a 1-m belt along each transect. We defined seedlings as individuals with stem diameter of less than 1 cm. We also counted juniper seedlings within the belt transects to understand establishment rates under different treatments.

Statistical Analysis

We used a general linear mixed model with split-plot blocked design to test for year, site, and treatment effects on our response variables: percent vegetative cover of native bunchgrass,

native forb, native shrub, and exotic grass, as well as total (and native and exotic) herbaceous biomass. Our model structure was:

$$Y_{ijkl} = \alpha_0 + \alpha_1 \text{Year}_{ijk} + \alpha_2 \text{Site}_{ijk} + \alpha_3 \text{Grazing}_{jk} + \alpha_4 \text{Thinning}_{ijk} + \alpha_5 (\text{Thinning} \times \text{Grazing} \times \text{Year} \times \text{Site}) + b_l + c_{ijk} + s_i + \varepsilon_{ijk}$$

Year referred to pretreatment (2012) and post-treatment (2014). There were three levels of site: MC, FC, and WG, as well as two levels for each treatment: grazed (G) and ungrazed (UG) and cut (C) and uncut (UC). Random error associated with model estimates (ε_{ijk}) and individual error for blocks (b_l), plots (c_{ijk}), and splitplots (s_i , repeated measures along the same transect) were also included.

We checked assumptions of constant variance and normality (symmetry) via modeled residuals before using the fitted models for parameter estimation or hypothesis testing. We log transformed biomass values as well as percent cover of bunchgrass and forbs to meet assumptions of normality necessary for use of linear mixed models (Gotelli and Ellison 2013). We report estimated means and in the case of log transformation, we report backtransformed medians. When comparing statistical differences of treatment group effects on response variables that have been log transformed we report ratios of medians.

Because we made multiple comparisons per response variable, we adjusted the family-wise error rate using false discovery rate (Benjamini and Hochberg 1995). We report the adjusted level of significance for each variable. This method controls the proportion of significant results which are false discoveries (type I error) while also reducing the chance of committing a type II error (Verhoeven et al. 2005) and is not as conservative as some family-wise comparison correction methods such as Bonferroni. We performed statistical analyses in R version 3.1.2 (R Development Core Team 2013) using the package *nlme* (Pinheiro et al. 2015) and the package *lsmeans* (Lenth and Hervé 2015).

RESULTS

Pretreatment overstory cover (%) of juniper differed among MC, FC, and WG. At MC, juniper overstory in plots randomly assigned to thinning or no thinning treatments was estimated to be $1.11\% \pm 2.05$ SD and $2.50\% \pm 2.71$ SD at thinned and unthinned plots, respectively. Similarly, juniper overstory cover was the same before thinning treatments at FC ($18.47\% \pm 10.38$ SD and $20.00\% \pm 14.44$ SD at thinned and unthinned plots, respectively). Percent cover of

juniper was lower at WG than expected at a Phase III site but comparable between thinned and unthinned plots ($11.67\% \pm 17.48$ SD and $11.11\% \pm 15.26$ SD at thinned and unthinned plots, respectively).

Productivity

When comparing pretreatment productivity to post-treatment productivity of total herbaceous biomass the effect of thinning treatment depended on site and year ($F_{2,60} = 3.379$, $P = 0.041$). In 2014, MC thinned plots had 1.29 times more median herbaceous biomass than unthinned plots but this was not statistically significant (99.17% CI: between 0.65 times less and 2.56 times more herbaceous biomass). Median herbaceous biomass significantly differed between thinned and unthinned plots at FC. Thinned plots at this site had 2.71 times more median herbaceous biomass and this was statistically significant (99.17% CI: 1.37 to 5.37 times more biomass). Biomass at WG was 0.36 times lower in thinned plots when compared to unthinned plots and this difference was statistically significant (99.17%: 0.17 to 0.78 times less biomass). Total herbaceous biomass differed among the three sites. Overall median herbaceous biomass was highest at MC and lowest at FC. Statistical tests of differences of means across these groups was not performed due to lack of replication at the site level.

Productivity of native and exotic herbaceous species in 2014 differed among the three sites (Figure 2.1). The ratio of native to exotic biomass depended on site ($F_{2,60} = 66.784$, $P < 0.001$). Mean native biomass at FC (Phase II) was $2.37 \text{ g} \cdot \text{m}^{-2}$ greater than exotic biomass (98.75% CI: between $11.18 \text{ g} \cdot \text{m}^{-2}$ less and $15.92 \text{ g} \cdot \text{m}^{-2}$ more native biomass) and not statistically significant. Both MC and WG were dominated by a non-native understory: MC (Phase I) mean native biomass was $57.96 \text{ g} \cdot \text{m}^{-2}$ less than exotic biomass (98.75% CI: $67.33 \text{ g} \cdot \text{m}^{-2}$ to $94.43 \text{ g} \cdot \text{m}^{-2}$ less native biomass) and mean native biomass at WG (Phase III) was $80.88 \text{ g} \cdot \text{m}^{-2}$ less than exotic biomass (98.75% CI: $44.41 \text{ g} \cdot \text{m}^{-2}$ to $71.51 \text{ g} \cdot \text{m}^{-2}$ less native biomass) at the site. We found no significant differences in herbaceous biomass one year post-grazing exclusion ($F_{1,15} = 0.227$, $P = 0.641$). Juniper removal significantly affected the ratio of native to exotic herbaceous biomass in 2014 ($F_{1,30} = 12.70$, $p = 0.001$) (Figure 2.1). We tested for differences in ratio of native to exotic mean herbaceous biomass between thinning treatments at each site individually although the interaction between site and thinning was not statistically significant ($F_{2,30} = 2.700$, $P = 0.083$). At MC, mean biomass of thinned plots was $19.21 \text{ g} \cdot \text{m}^{-2}$ greater than unthinned plots (98.75% CI: $-5.23 \text{ g} \cdot \text{m}^{-2}$ to $33.19 \text{ g} \cdot \text{m}^{-2}$ greater biomass). Mean

herbaceous biomass at FC was $11.33 \text{ g} \cdot \text{m}^{-2}$ greater in thinned plots than unthinned plots (98.75% CI: $-2.65 \text{ g} \cdot \text{m}^{-2}$ to $25.32 \text{ g} \cdot \text{m}^{-2}$ greater biomass). Mean biomass at WG increased $1.94 \text{ g} \cdot \text{m}^{-2}$ when comparing thinned versus unthinned plots but this was not statistically significant (98.75% CI: $-12.04 \text{ g} \cdot \text{m}^{-2}$ to $15.92 \text{ g} \cdot \text{m}^{-2}$ greater biomass).

Cover

Bunchgrass cover

We found evidence of interactive effects in our full model (thinning, grazing, site, and year) ($F_{2,60} = 3.800$, $P = 0.028$). This means that any difference in median bunchgrass cover between the thinning treatments depends on the grazing exclosure treatment, site, and whether the observation was taken pre or post treatment. We estimated ratios of median bunchgrass cover related to juniper thinning and grazing treatment comparisons for different sites and years separately. We found no differences between cut and uncut plots or grazed and ungrazed plots within a single year (Tables 2.1 and 2.2).

When comparing between years, post-thinning median native bunchgrass cover at FC was 5.06 times greater than pretreatment (99.7% CI: 1.78 to 14.35 times higher percent cover). There was no statistically significant difference in ratio of medians between any other groups. Median native bunchgrass cover in ungrazed plots was 2.28 times higher after cattle exclosure than the median bunchgrass cover in the same plot before treatment at FC (99.7% CI: 0.30 to 16.87 times higher percent cover). Thus, there was no statistical difference in median bunchgrass cover between these two groups. Similarly, median native bunchgrass cover in grazed plots was 1.01 times higher after treatment (99.7 CI: 0.46 to 2.23 times higher percent bunchgrass cover).

Compared to control plots (UCG) no statistical differences in ratio of median native bunchgrass cover were detected at any site and between any treatment combination compared to its respective control plot one year after treatment. Median native bunchgrass cover in CUG plots at FC was estimated to be 3.76 times higher than the median bunchgrass cover in UCG controls at FC (99.7% CI: 0.76 to 18.51 times higher percent cover), but this difference was not statistically significant.

Exotic annual grass cover

Estimated mean exotic cover was highest at MC and WG and lowest at FC (Appendix A). We found no effect of grazing exclusion ($F_{1,15} = 0.037$, $P = 0.850$) regardless of site, year, or treatment combination. There was strong evidence for an interaction between juniper thinning

and site ($F_{2,60}=17.007$, $P= 0.000$) so the effect of juniper thinning on exotic grass cover depended on site. We did not average between years when comparing pre and post exotic grass cover of thinned and unthinned plots.

At MC, there was no difference in mean percent cover of exotic grass between years. In unthinned plots mean exotic grass percent cover decreased by 15.97 percentage points from pretreatment to post-treatment but this was not statistically significant (98.3% CI: -40.29 to 8.35 percentage points higher cover; Table 2.3). On the other hand, in thinned MC plots mean exotic grass percent cover increased by 24.58 percentage points (98.3.0% CI: 0.27 to 48.90 percentage points higher). Thinning at FC increased mean exotic grass cover by an estimated 28.47 percentage points as compared to mean exotic cover in the same plots before treatment (98.3% CI: 4.15 to 52.79 percentage points higher cover).

Shrub cover

There was no evidence of an effect of thinning, cattle exclusion, or year (or any of the interactions among the fixed effects). There was a statistically significant effect of site ($F_{2,15}= 12.243$, $P< 0.001$). Mean shrub cover was estimated to be 7.19 percent at FC (98.3% CI: 3.52 to 10.86 percent cover) for all plots measured before or after thinning and exclusion of cattle (Appendix A).

Forb cover

We found no evidence of interactions among thinning, cattle exclusion, site, or time. There was no evidence of an effect of juniper treatment ($F_{1,30}=2.346$, $P= 0.136$) or cattle exclusion ($F_{1,15}= 1.483$, $P= 0.242$) on median forb cover. We did find evidence of forb cover differences across sites ($F_{2,15}= 17.974$, $P<0.001$). Median forb cover was statistically different across years as well ($F_{1,60}=89.28$, $P<0.001$). At all sites, between 2012 and 2014 in both the thinned plots and the unthinned plots median forb percent cover was statistically different (Table 2.4). At FC median forb cover was 4.11 times higher in 2014 than in 2012 within cut plots (95% CI: 2.58 to 6.54 times higher cover).

DISCUSSION

Woodland phase differences

In the past, studies have compared spatially distinct sites within a common plant association type to evaluate the effect woodland successional phase has on response to juniper

control (Bunting et al. 1999; Miller et al. 2000). To date, there are no data comparing cut and uncut plots at different successional phase woodlands at one site. Vegetative responses to juniper thinning and grazing treatments were dependent on site (Figure 2.1, Table 2.3) highlighting the importance of studying management responses across a gradient of juniper woodland stage. Our study supports previous findings that vegetative responses are dependent on pretreatment herbaceous community composition and structure (Miller et al. 2000).

Vegetative responses to juniper thinning

The response of herbaceous productivity to juniper thinning was dependent on site as we expected. All three areas had considerable exotic grass invasion that influenced responses to juniper thinning. Ratio of median native biomass to exotic biomass at MC and WG was unchanged in the first year post-thinning. While we observed an increase in the native: exotic biomass ratio at FC, this increase was not statistically significant. At this site exotics were far less dominant and the area retained a native bunchgrass and shrub community that had been replaced at the MC and WG sites. Overall, thinning resulted in an increase in exotic grass cover during the first year post-treatments. Exotic grass already established in the wildlife area appeared to be able to take advantage of disturbance and resources as a result of juniper thinning. This was likely due to the substantial seedbank and longevity of exotic grass seeds in the soil. We believe without active restoration these sites will remain in an alternative state of exotic dominance that will not reverse passively with the thinning of juniper alone.

Positive responses to thinning in bunchgrass cover responses were restricted to the FC area. Because bunchgrass recovery will likely be via primarily tillering and not recruitment of new genets (Bates 2005) we expected that areas within the site with existing bunchgrass communities would have the quickest responses. It is not yet known if recruitment by bunchgrass into areas that have lost all or most of the bunchgrass community will occur without active restorative efforts such as seeding.

Forb cover responded positively to juniper thinning at all sites but higher forb cover was found in untreated plots as well indicating interannual weather may have driven forb cover independent of juniper thinning (Table 2.4). Herbaceous cover by forbs responded strongest to thinning at the FC area. Because many forbs are annuals, the release of resources due to juniper thinning may have resulted in better germination rates or increased growth of the forb layer.

Quick response of forbs to juniper thinning has been observed in other studies (Bates et al. 2005; O'Connor et al. 2013; Vaitkus and Eddleman 1986).

Shrub cover did not respond to thinning after one year. However shrubs are slow growing plants thus responses to thinning may take a few years to become visible. The WG site no longer had any shrubs and we found no shrub seedlings. These results suggest this area and areas in similar condition in the site are unlikely to recover their shrub layer without active restoration such as removal of thatch layers and drill seeding. Previous research has shown significant shrub cover recovery in terms of recruitment (O'Connor et al. 2013) and biomass (Rose and Eddleman 1994) but cover was unchanged (Bates et al. 2005). All previous work has determined that pre-existing shrub condition is important in the recovery outcome.

Vegetative responses to grazing exclusion

We found very few first-year responses to livestock grazing exclusion. This is likely due to the short nature of this study as effects to grazing cessation can be slow to manifest (Anderson and Holte 1981) or not large in general (Curtin 2002; Stohlgren et al. 1999). In addition, the limited response to livestock exclusion can also be attributed to the low intensity grazing occurring at the site. At our site cattle grazing appeared to be playing a smaller role in driving vegetative cover and productivity than overstory juniper woodland development based on comparisons among treatment groups with grazing and those without in terms of productivity and cover. This study represents some of the first work assessing grazing and juniper thinning treatments concurrently but was over a short duration. Grazing has been associated with poor bitterbrush seed production in shrub-steppe communities (Clements and Young 2001) though we are unclear whether this is playing a role in the recruitment failures of shrub species at the wildlife area. More work that assesses the role of grazing to restoration outcomes over many years is still needed.

IMPLICATIONS

Our results highlight the importance of assessing preexisting conditions before undertaking juniper removal projects at the site. Areas retaining bunchgrass and shrub cover will likely respond well to juniper thinning without requiring additional active management to reestablish native understory. Areas with high exotic dominance and little native cover may take longer to recover or need more active restoration efforts to regain substantial native plant

communities. The effects of grazing by domestic livestock should be considered when managing juniper encroachment. Because blocks were sited only on relatively flat areas inference should only be made to other areas within the management area with similar topography. The effect of cattle grazing and thinning is expected to be quite different in steep areas or nearer to draws due to altered hydrology and use by cattle.

TABLES AND FIGURES

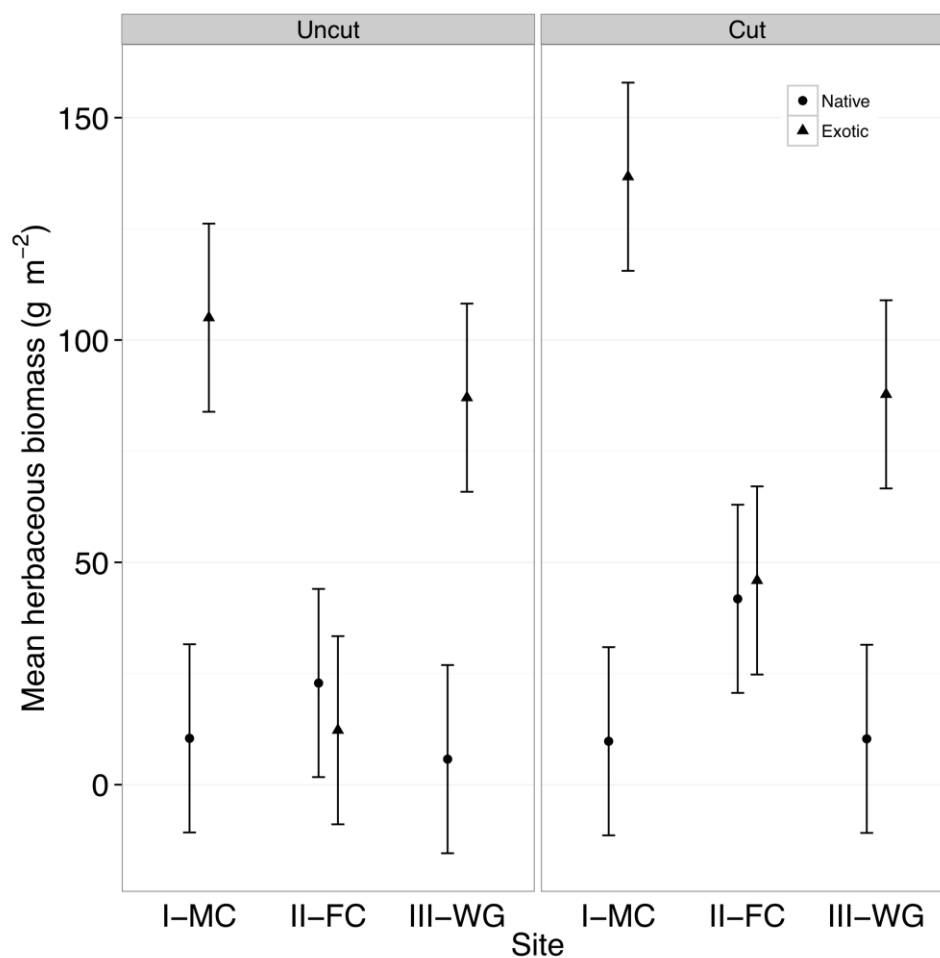


Figure 2.1. Mean native and exotic herbaceous productivity ($\text{g} \cdot \text{m}^{-2}$) for cut and uncut plots across three sites within the Phillip Schneider Wildlife Area, Grant County, Oregon, USA. Woodland successional phase (I-III) correspond to those defined in Miller et al. (2005) where Phase I sites have subordinate juniper cover with dominant understory vegetation, Phase II sites have co-dominant juniper cover but retain shrub understory layers, and Phase III sites where shrub and grass understory layers have been lost and juniper canopy is dominant. Sites MC, FC, and WG correspond to Murderer's Creek, Flat Creek, and Wiley Gulch, respectively. All herbaceous biomass (grams per square meter) for the 2013-2014 growing year was sorted into native and exotic categories in the field, dried, and then weighed.

Table 2.1. Estimated median bunchgrass cover ratio between juniper cut plots and uncut plots at three sites within the Phillip W. Schneider Wildlife Area, Grant County, Oregon. Data was collected in 2012 (pre) and 2014 (post) representing before and after treatment establishment. Treatments included juniper canopy manipulation (cut/uncut) and grazing exclosure (grazed/ungrazed). Sites MC, FC, and WG correspond to Murderer’s Creek, Flat Creek, and Wiley Gulch, respectively. Confidence intervals that did not contain a ratio of 1 (no difference) would be considered statistically significant.

Site	Contrast	Time	Grazing Treatment	Ratio of Medians ¹	Lower CI	Upper CI	SE	df
MC	Cut / Uncut	Pre	Grazed	0.874	0.275	2.771	0.343	30
MC	Cut / Uncut	Post	Grazed	1.322	0.417	4.193	0.343	30
FC	Cut / Uncut	Pre	Grazed	2.076	0.654	6.585	0.343	30
FC	Cut / Uncut	Post	Grazed	2.599	0.819	8.244	0.343	30
WG	Cut / Uncut	Pre	Grazed	1.122	0.354	3.561	0.343	30
WG	Cut / Uncut	Post	Grazed	1.587	0.500	5.036	0.343	30
MC	Cut / Uncut	Pre	Ungrazed	1.777	0.560	5.638	0.343	30
MC	Cut / Uncut	Post	Ungrazed	1.201	0.379	3.810	0.343	30
FC	Cut / Uncut	Pre	Ungrazed	0.382	0.120	1.210	0.343	30
FC	Cut / Uncut	Post	Ungrazed	1.899	0.599	6.024	0.343	30
WG	Cut / Uncut	Pre	Ungrazed	2.178	0.686	6.909	0.343	30
WG	Cut / Uncut	Post	Ungrazed	1.334	0.420	4.231	0.343	30

¹ Ratio of medians represent the number of times greater bunchgrass cover was between cut and uncut plots

* Denotes statistical significance at the 99.71% confidence level

Table 2.2. Estimated median bunchgrass cover ratio between grazed and ungrazed plots at sites within the Phillip W. Schneider Wildlife Area, Grant County, Oregon. Data was collected in 2012 (pre) and 2014 (post) representing before and after treatment establishment. Treatments included juniper canopy manipulation (cut/uncut) and grazing exclosure (grazed/ungrazed). Sites MC, FC, and WG correspond to Murderer’s Creek, Flat Creek, and Wiley Gulch, respectively. Only confidence intervals that did not contain a ratio of 1 (no difference) would be considered statistically significant.

Site	Contrast	Time	Juniper treatment	Ratio of Medians ¹	Lower CI	Upper CI	SE	df
MC	Grazed / Ungrazed	Pre	Cut	0.632	0.163	2.446	0.365	15
MC	Grazed / Ungrazed	Post	Cut	0.874	0.226	3.383	0.365	15
FC	Grazed / Ungrazed	Pre	Cut	1.849	0.478	7.162	0.365	15
FC	Grazed / Ungrazed	Post	Cut	0.862	0.223	3.339	0.365	15
WG	Grazed / Ungrazed	Pre	Cut	0.429	0.111	1.662	0.365	15
WG	Grazed / Ungrazed	Post	Cut	0.668	0.173	2.587	0.365	15
MC	Grazed / Ungrazed	Pre	Uncut	1.285	0.332	4.976	0.365	15
MC	Grazed / Ungrazed	Post	Uncut	0.794	0.205	3.074	0.365	15
FC	Grazed / Ungrazed	Pre	Uncut	0.340	0.088	1.316	0.365	15
FC	Grazed / Ungrazed	Post	Uncut	0.630	0.163	2.440	0.365	15
WG	Grazed / Ungrazed	Pre	Uncut	0.833	0.215	3.225	0.365	15
WG	Grazed / Ungrazed	Post	Uncut	0.561	0.145	2.173	0.365	15

¹ Ratio of medians represent the number of times greater bunchgrass cover was in grazed plots as compared to ungrazed plots

* Denotes statistical significance at the 99.71% confidence level

Table 2.3. Difference in mean exotic cover between 2014 and 2012 within juniper treated plots as well as the uncut plots. All comparisons are post-pre median exotic cover. All levels of grazing exclosure treatment have been pooled. Sites MC, FC, and WG correspond to Murderer's Creek, Flat Creek, and Wiley Gulch, respectively three sites within the Phillip W. Schneider Wildlife Area, Grant County, Oregon, USA. Confidence intervals that do not contain 0 (no difference) are considered statistically significant.

Site	Contrast	Juniper Treatment	Difference in Means ¹	Lower CI	Upper CI	SE	df
MC	Post - Pre	Cut	24.583 *	0.265	48.901	9.873	60
MC	Post - Pre	Uncut	-15.972	-40.290	8.346	9.873	60
FC	Post - Pre	Cut	28.472 *	4.154	52.790	9.873	60
FC	Post - Pre	Uncut	-16.403	-40.721	7.915	9.873	60
WG	Post - Pre	Cut	20.694	-3.623	45.012	9.873	60
WG	Post - Pre	Uncut	6.389	-17.929	30.707	9.873	60

¹ Difference in mean exotic cover between before and after treatments were established

* Denotes statistical significance at the 98.33% confidence level

Table 2.4. Ratio of median forb cover between 2014 and 2012 within juniper treated plots as well as the uncut plots. All comparisons are post/pre median forb cover. All levels of grazing enclosure treatment have been pooled. Sites MC, FC, and WG correspond to Murderer’s Creek, Flat Creek, and Wiley Gulch, respectively within the Phillip W. Schneider Wildlife Area, Grant County, Oregon, USA. Confidence intervals that do not contain a ratio of 1 (no difference) are considered statistically significant.

Site	Contrast	Juniper Treatment	Ratio of medians ¹	Lower CI	Upper CI	SE	df
MC	Post/ Pre	Cut	2.245 *	1.410	3.575	0.233	60
MC	Post/ Pre	Uncut	1.686 *	1.058	2.684	0.233	60
FC	Post/ Pre	Cut	4.107 *	2.579	6.541	0.233	60
FC	Post/ Pre	Uncut	1.761 *	1.106	2.804	0.233	60
WG	Post/ Pre	Cut	2.581 *	1.621	4.110	0.233	60
WG	Post/ Pre	Uncut	3.083 *	1.936	4.909	0.233	60

¹ Ratio of medians represent the number of times greater median forb cover was between the comparison in each plot

* Denotes statistical significance at the 95% confidence level

CHAPTER 3: RESPONSES OF FORMER SHRUB-STEPPE RANGE TO JUNIPER
THINNING ACROSS A TIME-SINCE-TREATMENT CHRONOSERIES

Corinne Morozumi, Dana Sanchez, Ricardo Mata-González

Fisheries and Wildlife Department
Oregon State University
Corvallis, OR

ABSTRACT

Shrub-steppe habitat has been degraded across the intermountain west of North America due to agriculture, land-use conversion, alteration of fire regimes, invasion by exotic species, and encroachment by coniferous trees. Due to its importance as wildlife habitat, efforts are underway to restore these ecosystems on public and private lands. Our study evaluated responses of a former sagebrush-bitterbrush shrub-steppe in eastern Oregon following the cutting of western juniper (*Juniperus occidentalis*). Through the use of a chronoserries of treatment plots, we evaluated plant and small mammal community responses to juniper removal between two and eight years after juniper overstory felling. In addition, we compared vegetative responses in the microhabitats created by the felled juniper. Vegetative cover and biomass was highest in plots with most recent juniper cutting and these responses were mainly driven by exotic annual grasses that were likely well-positioned to take advantage of the resource release. Microhabitats created by the felled juniper and the remnant duff area of the original standing trees harbored different plant communities. The duff zone in particular had high native biomass and species composition but this effect appeared to weaken as time since juniper cutting increased. Small mammal diversity was lowest in the plot that was cut most recently. Abundance of small mammal species captured was highest in the oldest treatment plot (i.e., longest time-since-treatment). Least chipmunk (*Tamias minimus*), a primary seed disperser of antelope bitterbrush (*Purshia tridentata*), was most numerous in the oldest treatment plot and in the uncut control plot but was limited in all other plots. Without bitterbrush seed dispersal into these areas, recovery of that key native shrub will be limited unless active recovery is undertaken.

INTRODUCTION

Shrub-steppe ecosystems occupy roughly 45 million hectares of semi-arid lands of the western US (West 1983) and are some of the most imperiled ecosystems in the country (Noss et al. 1995). These systems provide habitat for many wildlife species, recreational opportunities for millions of people, and make up large portions of public and private rangelands in the region. In eastern Oregon, historic shrub-steppe plant associations of big sagebrush (*Artemisia tridentata*)/bluebunch wheatgrass (*Pseudoroegneria spicata*), big sagebrush/Idaho fescue (*Festuca idahoensis*), and bitterbrush (*Purshia tridentata*)/bluebunch wheatgrass were once dominant. Shrub-steppe acts as obligate habitat for charismatic species of concern such as Greater sage-grouse (*Centrocercus urophasianus*) and as winter range for mule deer (*Odocoileus hemionus*). In particular, bitterbrush provides high quality winter forage and cover for mule deer. These shrub stands are associated with lower predation by cougars and therefore may offer more protection than smaller open-habitat shrubs such as rabbitbrush (Pierce et al. 2004). Because of their value for multiple native wildlife species, western shrub-steppe is a habitat of management concern.

Decades-long practices of active fire suppression, livestock overgrazing, and shrub clearing has led to large-scale modification to ecosystems of the intermountain west and ultimately loss of shrub-steppe communities. Across the intermountain western United States conifer encroachment by pine and juniper species has altered grassland, sagebrush, and shrub-steppe ecosystems (Miller et al. 2005). Increasing tree cover and density cause changes in plant community structure and function (Miller et al. 2000). As these woodland canopies close many sites lose much of their understory vegetation (Bunting et al. 1999; Knapp and Soulé 1998). In addition, annual exotic grasses have heavily invaded shrub-steppe. Abundance and palatability of mule deer forage declines with piñon pine-juniper stand closure (Oregon Department of Fish and Wildlife 2005; Short et al. 1977) while vegetation structure, forage species, and other resources critical to other native species likewise decline or disappear.

Managers in many western states have prioritized recovery and restoration of shrub-steppe ecosystems due to their importance as wildlife habitat. To increase habitat quality for wildlife species that rely on shrub-steppe, managers in western states have begun reducing the density of juniper trees. Control of western juniper (*Juniperus occidentalis*) can lead to increases in native plant cover, diversity and primary productivity (Bates et al. 2000; Bates and Svejcar 2009; O'Connor et al. 2013; Rose and Eddleman 1994; Vaitkus and Eddleman 1986). However,

restoration of these systems is bound to be complex. Existing site condition, such as persistence of perennial species and understory vegetative cover, as well as woodland stage development play a large role in determining recovery outcomes (Bates et al. 2000; Miller et al. 2000). Juniper woodland density can be classified into three phases: subordination (Phase I), co-dominance (Phase II), and dominance (Phase III) whereby the juniper play a more prominent role in driving ecosystem processes as they get more dense at a site (Miller et al. 2005).

Communities may pass through predictable phases towards sagebrush grassland climax communities although multiple stable states can complicate or preclude succession after juniper thinning (Briske et al. 2008; Petersen et al. 2009). Furthermore, juniper removal strategies vary. For example, cut juniper are left on some sites either because removal is infeasible or cost of piling and burning is prohibitive, whereas at other sites felled juniper are piled and burned (Bates and Svejcar 2009). Additionally, sites heavily invaded by non-native annual grasses might transition to an annual-dominated phase, and in some cases this tendency might be increased if fire is used (Bates et al. 2006). Studies are needed that evaluate the effectiveness of juniper management strategies across broad ranges of sites, conditions, and practices and as measured in both vegetative and wildlife responses.

Although felled junipers and reopened vistas are visible, immediate changes where juniper thinning has been implemented, we need to better understand the many ecologically important responses that occur at finer scales. In the short term after trees are felled, mule deer have been observed feeding on juniper needles (Ryan Torland, ODFW, personal communication) and birds use cavities in the felled trees as nest sites. However it is unknown whether there are significant differences in vegetation community composition within microsites between juniper trees (interspace), those beneath the felled tree (canopy zone), and those in the duff area (duff zone) of the original standing tree. More importantly, it is unknown whether these microsite communities might affect the recovery trajectory positively or negatively once the juniper trees are felled and whether these microsite communities vary in their responses as time-since-treatment increases. For example, areas beneath downed canopy likely provide shelter from grazers and browsers, might maintain soil moisture later in the season, and might harbor either native or exotic species. Some potential combinations of these factors would allow certain microsites to act as islands from which native recovery could spread, whereas other combinations would set microsite communities as bastions for further spread of invasives.

Finally, small mammals are another less visible, but potentially important group of actors in the potential restoration of juniper-invaded shrub-steppe. This group of animals can respond dynamically to vegetation changes and might present a useful indication of the impacts of vegetation management. Some members of the small mammal community perform important ecosystem functions such as seed dispersal and plant pollination, while others would act as seed predators challenging recovery of native understory. In the John Day ecological province researchers found increased abundance of yellow pine chipmunks (*Tamias amoenus*) in cut plots compared to uncut plots though this trend was not significant at another site in southeastern Oregon (Willis and Miller 1999). Because yellow pine chipmunks are thought to be the main seed dispersers of antelope bitterbrush in ponderosa pine-bitterbrush ecosystems (Vander Wall 1994) their increased abundance in cut plots might imply a potential for increased bitterbrush seedling establishment. Understanding the effectiveness of juniper removal as part of a restoration strategy must include investigation of which native small mammals persist in juniper-invaded areas, which are absent from the areas, and which species are likely to explore and reinhabit treated areas.

Management activities, such as juniper cutting across large patches of land over the course of many years, offer an opportunity to understand the temporal dynamics of shrub-steppe responses. We used a chronoserries of time-since-treatment plots to understand recovery of shrub-steppe habitat quality. We explored how plant and small mammal composition was influenced by time since juniper cutting. In addition, we examined how fine scale vegetative responses differed at microsites both within and across the chronoserries.

We expected that time since thinning of juniper canopy cover would positively increase primary productivity (Bates 2005; Rose and Eddleman 1994; Vaitkus and Eddleman 1986). By increasing access to resources and decreasing competition, we expected that juniper removal to stimulate growth of understory plants and that productivity and cover would tend to increase with time since thinning. We hypothesized that change in percent cover of shrubs would lag behind that of faster growing plants such as grasses and forbs and that any positive shrub response would only be observable in the oldest plots (greatest time since treatment). We predicted that as time since treatment increased small mammal abundance and richness would likewise increase (Willis and Miller 1999). At the microsite scale, we expected productivity to be highest in the interspace zone and that exotic grasses would dominate. We expected dominant cover to be different in each

zone. Forbs were expected to dominate the canopy zones while grasses and forbs would be present in the duff zones and that grasses would be dominant in the interspace. We expected these effects would be dependent on time since juniper treatment.

Study Site

The Phillip W. Schneider Wildlife Area (hereafter Wildlife Area) is located between the Ochoco and Malheur National Forests in the John Day Ecological Province about thirty miles west of John Day, Oregon. It encompasses 21,000 hectares of public land managed by Oregon Department of Fish and Wildlife (ODFW) and contains both ODFW deeded land as well as parcels owned by the Bureau of Land Management (BLM). Management action including juniper thinning aims to increase usable habitat and browse species for mule deer. The vegetation was once characterized by low sagebrush (*Artemisia arbuscula*), big sagebrush, and antelope bitterbrush with perennial bunchgrasses such as bluebunch wheatgrass, Idaho fescue and bottlebrush squirreltail (*Elymus elymoides*). Bitterbrush stands throughout the site have been in decline since the 1970s resulting in a loss of the shrub component in many parts of the Wildlife Area (Oregon Department of Fish and Wildlife 2011). In addition, western juniper is invading grassland and shrub steppe ecosystems in the Wildlife Area and becoming increasingly dense. Since the 1980s, annual exotic grasses such as cheatgrass (*Bromus tectorum*), medusahead (*Taeniatherum caput-medusae*) and recently North Africa grass (*Ventenata dubia*) began invading the wildlife area with the heaviest infestations near roads, disturbed areas and in lowland flats.

The Wildlife Area is important winter range habitat for mule deer (Oregon Department of Fish and Wildlife 2011). It is part of the Murderer's Creek Wildlife Management Unit (WMU), which contains an estimated 105,557 hectares of winter range habitat. Of that total winter habitat, 65,683 hectares are privately owned and 39,875 hectares are publicly owned. Mule deer populations within the Murderer's Creek basin have fluctuated greatly from an estimated (year) peak of 30,000 to a current (year) estimate of 5,000 animals. In 2010 Murderer's Creek WMU was included in the statewide Mule Deer Initiative managed by ODFW. This program addresses population declines of mule deer by increasing habitat quality, managing predators, and increasing compliance, among other objectives (Oregon Department of Fish and Wildlife 2011). Within the Wildlife Area habitat quality improvement has mainly focused on juniper thinning to date.

METHODS

We removed young juniper (estimated < 100 years old based on morphology, Miller et al. 2005) within contiguous plots along a tributary of Murderer's Creek (44.278°N, -119.478°W) in 2008, 2009, 2010, and 2012 (Figure 3.1). Plots were named for the year they were treated and were limited to single replicates. We mechanically cut with chainsaw and left in place all trees (except in 2010 when trees were piled and burned on slopes not exceeding 30%). In addition to the treated plots, we located a plot with similar juniper density and site conditions on adjacent Bureau of Land Management (BLM) owned property within the wildlife area to act as an untreated control. This plot is contiguous with the rest of the plots. It is within the same fence and under the same grazing regime as the thinned plots. We used these treated plots as well as the untreated control plot to assess temporal changes in plant community after juniper thinning. Plots used in the chronoserries were similar in initial shrub and herbaceous species composition except for the 2009 plot which had higher shrub cover (roughly 20% higher cover). Due to these inherent differences between the 2009 plot and the others in the project, we excluded the 2009 plot from any analyses with explicit comparisons of microsite vegetation.

Microsite vegetation sampling

In June 2014 we randomly selected 10 trees per plot, standardizing for slope and aspect across the plots when possible. We attempted to hold constant the total area from which trees were randomly selected because treatment plot size different among treatment years. At each tree, we measured biomass and cover within 0.75 x 0.75 m quadrats placed under the original canopy (duff zone), beneath the felled canopy (canopy zone), and in the area between trees (interspace zone) (quadrat location followed O'Connor et al. 2013). We randomly located two quadrats beneath each canopy (duff zone and canopy zone) for each tree. In the adjacent interspace we randomly placed two quadrats per tree ≥ 10 m from the edge of the duff zone. In the uncut control plot we selected 10 trees and placed two quadrats in the duff zone of the standing tree and two quadrats in the interspace. We recorded the number of shrub seedlings within each quadrat to calculate density. We assessed percent plant cover of the functional groups grass, shrub, forb, and total vegetation using the method determined by Bailey and Poulton (1968).

We calibrated and trained all observers by laying down cardboard shapes of known cover within a quadrat before recording plant cover estimation. We used a cover estimate reference sheet (Habitat Monitoring Committee 1996), which observers were able to reference throughout

the recording period. We clipped all aboveground herbaceous biomass to 1 cm within the quadrant. We dried quadrat samples for 48 hrs at 60°C before sorting the clipped biomass to native and exotic categorization and weighing.

Small mammals

In summer 2014 we trapped small mammals twice within chronoseries plots described above. In each plot, we deployed three 250-meter trapping transects with 50 trap stations per transect. In the 2008 plot we necessarily had to augment transect length and number due to the dimensions of the plot. We located transects along 350-degree bearings and separated the transects by 50 m when plot size and dimensions allowed. In order to standardize across the four treatment areas (and control), we oriented transects perpendicular to the slope, keeping elevation constant when possible. Each station contained two Sherman live traps (3x3.5x9 in) baited with rolled oats, peanut butter and mealworms. We provisioned each trap with a square of polyester bedding for insulation as well as a slice of potato for hydration. We covered traps with a sleeve for additional insulation, rain, and heat protection.

We pre-baited traps and locked them open for three nights prior to live trapping to acclimatize animals to the traps. We baited traps in the evening and checked for captures early the following morning. We reset triggered traps and re-provisioned them as necessary. We trapped for four consecutive nights because previous studies have found saturation of richness and abundance after four nights. Due to logistical constraints we were unable to trap within all areas at the same time. We trapped concurrently within the 2009, 2010, 2012 plots between 30 June 2014-3 July 2014 and between 15 July 2014-18 July 2014. We trapped concurrently within the 2008 plot and the control plot between 4 July 2014-7 July 2014 and between 20 July 2014-23 July 23 2014.

We transferred small mammals caught in live traps to a small plastic bag and identified each individual to species and determined sex. We recorded the mass of each individual with a calibrated Pesola scale. We marked individuals with non-toxic colored marker along the spine denoting day of capture and plot identification before releasing the animal at the point of capture. We calculated plot-specific abundance and Brillouin indexed diversity (Magurran 2004) in order to compare species richness and diversity in plots with different time-since-treatment.

We buried deceased individuals on-site according to protocols approved by OSU within the OSU Animal Care and Use Permit (ACUP # 4572) and the ODFW Scientific Taking Permit

(104-14). All animal handling was in accordance with guidelines provided by the American Society of Mammalogists (Gannon et al. 2007).

Shrub canopy cover estimation along trapping transects

In order to describe the understory canopy for each area in which we trapped, we recorded shrub and woody plant cover along each of the 250 m trapping transects. We used the line intercept method (Elzinga et al. 1998) to record all shrub or tree species along the transect, and considered gaps where breaks in canopy exceeded 10 cm. Due to high shrub cover in the 2009 plot, we collected shrub cover along a subsample of each 250 m transect in that plot. We divided the transects into 50m sections and randomly sampled 2 of the 5 possible sections on each transect. Shrub cover was then calculated along each transect by dividing the length of shrub cover for each species by the total length of sampled transect.

Data analysis

Differences in vegetative responses to juniper thinning across the chronoserries plots were graphed. We performed no statistical tests of vegetative or small mammal data among the time-since-treatment plots due to lack of replication of the chronoserries itself.

We built general linear mixed models with individual trees as a random effect to compare microsite vegetative responses at each zone. Because no (felled) canopy zone data were collected in the uncut control plot we removed the control plot when testing comparisons among the three zones. Likewise, to test for differences in means between the duff and interspace zones in the control plot we removed the canopy zone data. This was necessary because general linear mixed models require perfectly crossed factors. To control type I errors while making multiple comparisons, we used false discovery rate, which also increases power (Benjamini and Hochberg 1995). We report adjusted significance levels for each test. All univariate analyses were done using R version 3.1.2 (R Development Core Team 2013).

We performed multivariate analyses in PC-ORD version 6.0 (McCune and Mefford 2011). We used non-metric multidimensional scaling (NMS) using Jaccard's distance measure to ordinate sample units in species space as this method does not assume linearity between species abundances (McCune and Grace 2002). We removed rare species occurring in fewer than 5% of sample units before analysis to reduce noise in the data.

RESULTS

Microsite vegetation

Total vegetative cover, averaged across the three microsites, was comprised of shrubs, forbs, and grasses and was highly variable across the 2008, 2010, 2012, and control plots (Figure 3.2a). Control plots generally had less cover with a median total cover of 15.50%. All three of the other plots had a median cover of 26.75%. Grass cover also appeared to differ across the sites. Median grass cover in the uncut control was 9.50% while in the 2012 plot it was 26.75%. Variability across samples increased as time since cutting decreased (Figure 3.2b). Median forb cover was relatively low across the plots (2%-3.5% for all four plots, Figure 3.2c). Across the entire basin seedling establishment was nearly nonexistent. The highest native seedling mean abundance we recorded was 0.25 seedlings per zone in duff subplots located in the control plot.

Total cover was consistently lower in the canopy zone across the three cut areas (Figure 3.3a). In addition, total percent cover appears to be lower in the control plot compared to all three cut areas. Total percent cover in the duff zone seemed dependent on plot. Total percent cover in the interspace of cut plots did not vary significantly through the series of cuts but was consistently higher than total interspace cover in the control plot.

Grass cover was highest in interspace and duff zones of the most recently treated plots (Figure 3.3b, Appendix B.1). As time since juniper treatment increased grass cover in duff zones appeared to decrease. Median grass cover in the interspace zone of the control plot was 15.50% while median grass cover in the 2012 plot was 32.38%. Forb cover appeared to be higher in cut plots than in the control plot (Figure 3.3c, Appendix B.2). The effect of microsite on grass cover did not significantly differ by time since treatment ($F_{4,54} = 0.283$, $P = 0.838$). Mean grass cover was significantly different depending on time since treatment ($F_{3,36} = 14.124$, $P < 0.001$). Mean grass cover did vary by microsite zone ($F_{2,54} = 61.80$, $P < 0.001$) Difference in mean grass cover was largest between the canopy and interspace in the 2012 plot (Table 3.1).

Median forb cover across the different microsites was not dependent on cut year ($F_{3,36} = 0.620$, $P = 0.607$). Median forb cover differed among zones ($F_{1,36} = 16.786$, $P < 0.001$) but not among cut years ($F_{3,36} = 0.485$, $P = 0.695$). Median forb cover differed between the canopy and interspace zones in each juniper-cut plot (Table 3.2). The canopy zone of the 2010 plot had 6.12 times more forb cover than the interspace zone (96.67% CI: 2.63 to 14.27 times more forb cover).

Overall, median herbaceous biomass was highest in the interspace and duff zones and lowest in the canopy zone (Figure 3.4, Appendix B.3). Mean herbaceous biomass across the

different microsites was not dependent on cut year ($F_{4,54} = 0.342$, $P = 0.848$). We found mean biomass differences among zones ($F_{2,54} = 54.515$, $P < 0.001$) and also among cut years ($F_{2,27} = 5.187$, $P = 0.012$). In the 2010 plot the mean biomass (g/m^2) in the canopy was 110.61 g/m^2 less than the duff zones (97.98% CI: 62.31 to 158.91 g/m^2 less biomass) (Table 3.3).

Throughout the study area exotic herbaceous species were more productive than native herbaceous species (Figure 3.5, Appendices B.4 and B.5). The 2008 plot appeared to have less native productivity and this trend was especially pronounced in the interspace zone. Exotic herbaceous productivity was lowest in the duff zone of the control plot and high in the interspace of all plots with juniper thinning. Median native biomass across the different microsites was not dependent on cut year ($F_{3,36} = 1.616$, $P = 0.203$). We found median native biomass differences among zones ($F_{1,36} = 26.219$, $P < 0.001$) and among cut years ($F_{3,36} = 3.584$, $P = 0.023$). In the 2008 plot, median native biomass was 3.82 times higher in the duff zone as compared to the interspace (97.78% CI: 1.29 to 11.26 times more biomass). Median native biomass was not statistically different between the canopy and duff zones in the 2008 plot (0.31 times more biomass, 97.78% CI: 0.12 to 0.92 times more biomass). Median native biomass did not significantly differ between the canopy and interspace zones in 2008 (1.19 times more biomass, 97.78% CI: 0.40 to 3.50 times more biomass). In the 2010 plot, median native biomass did not statistically differ among any of the three zones. In the 2012 plot, median native biomass was 2.06 times higher in the duff zone compared to the interspace zone but this was not statistically significant (97.78% CI: 0.70 to 6.07 times more biomass). In the control plot, median native biomass was 2.08 times greater in the duff zone than the interspace (96.25% CI: 1.09 to 3.97 times more biomass). Median exotic biomass across the different microsites was not dependent on cut year ($F_{4,54} = 1.44$, $P = 0.235$). Median exotic biomass differed among zones ($F_{2,54} = 22.290$, $P < 0.001$) but not among cut years ($F_{2,27} = 1.060$, $P = 0.361$). We found no statistical differences in median exotic biomass among zones within each plot.

Composition of species overlapped greatly between all cut plots but differed in the space occupied by the control plot samples (Figure 3.6a). Ordination of sample units in species space resulted in a three dimensional solution. Axis 1 was driven mainly by differences among the microsite zones (Figure 3.6b). Species negatively correlated with Axis 1 included cleavers (*Gallium* sp.) ($r = -0.541$), prickly lettuce (*Lactuca serriola*) ($r = -0.632$), and *Amsinckia* sp. ($r = -0.349$) all found beneath the cut juniper in the canopy zone. North Africa grass ($r = 0.560$) and

Sandberg bluegrass (*Poa secunda*) ($r= 0.702$) were positively correlated with this axis. The second axis was driven by exotic grass dominance. Medusahead which forms dense stands, was positively correlated with axis 2 ($r=0.537$) as was North Africa grass ($r=0.313$), whereas perennial native species such as bluebunch wheatgrass ($r=-0.443$) and Idaho fescue ($r=-0.377$) were negatively correlated. Axis 3 expressed another aspect of grass cover, as squirreltail grass was strongly and positively correlated with the axis ($r= 0.647$) while cheatgrass was also a positive driver ($r=0.411$).

Small mammals

Small mammal relative abundance and diversity was low across all plots. The most prevalent species was deer mouse (*Peromyscus maniculatus*) that was found in all thinned plots as well as the control plot (Table 3.4). Brillouin indexed diversity was highest in the 2009 plot and lowest in the 2012 plot. We also calculated ratio of males and females within deer mouse and least chipmunk. Males made up a larger proportion of the total abundance in plots cut in 2012 and 2010 than plots cut in 2008, 2009, and the control plot. Mean shrub cover, assessed along mammal trapping transects, was less than 5% across all plots except 2009 (Table 3.5). Females of deer mouse and least chipmunk generally weighed more in each plot though median mass for both sexes of deer mouse was highest in the 2010 plot (Figure 3.7). Median mass of least chipmunks appears to be higher in the 2008 plot than the control plot (the only two plots where more than two chipmunks were trapped).

DISCUSSION

Understory responses to juniper thinning are mixed across the time-since-treatment chronoserries we sampled in the Phillip W. Schneider Wildlife Area. Initially, the release from juniper overstory seems to allow grasses to increase in cover (Figure 3.2) while shrub and forb cover were not significantly changed. In addition, herbaceous biomass increases shortly after juniper thinning. Unfortunately, this increase in biomass was mainly the result of increased growth of invasive annual grasses that dominate large areas in the Murderer's Creek basin. Species composition of herbaceous species within the cut plots is different from the control plot (Figure 3.6a). Shrub recruitment is not occurring in the treated plots. Together these results suggest that mule deer habitat is not improving due to juniper thinning in the near term. Positive shrub responses in the first few years after juniper treatment have been previously observed

(Bates and Svejcar 2009; O'Connor et al. 2013; O'Meara et al. 1981) but this response depends on the existing condition of the shrub layer when the juniper trees were removed (Miller et al. 2005). Because increasing habitat heterogeneity is important for many types of wildlife (reviewed in Tews et al. 2004) the lack of shrub response will likely limit the site as quality habitat for other non-game species as well.

Due to differences in site that predated juniper cutting we removed the 2009 plot from analyses with explicit comparisons. This plot has a median shrub cover of 8% shrub cover and nearly 22% cover by shrub in the interspace. In addition, most of the shrub cover in this plot was low sagebrush (*Artemisia arbuscula*) often found in areas with impermeable bedrock soil horizons closer to the soil surface than other shrub species (Fosberg and Hironaka 1964).

In many places managers cannot remove downed juniper trees due to financial or logistical reasons. Because of this our study focused on whether responses in the microhabitats created by these felled trees differed across time. Compared to the uncut control duff zones at felled trees in plots cut in 2008, 2010, and 2012 were more productive directly after juniper thinning (Figure 3.4) but this response was due to the release of exotic species (Figure 3.5). We found similar release of exotics in interspace plots and native productivity appeared to decrease as time since thinning increases.

Relative abundance and diversity of small mammals may be showing positive signs as time since juniper thinning increases (Table 3.5). This is largely driven by deer mouse abundance though we captured several chipmunks in the 2008 plot as well. Bitterbrush stands have persisted in low density in this plot, which may drive this trend. Chipmunk presence will likely assist bitterbrush recruitment in this plot. Anecdotally, we observed bitterbrush seeds within traps with chipmunk captures in the 2008 plot indicating that chipmunks are dispersing these seeds. Chipmunks were present in low abundance in the 2009 plot. This may have been due to the dominant cover of low sagebrush as abundance of least chipmunk was previously reported to be low in this shrub type (Feldhamer 1979). We found few chipmunks in the 2010 and 2012 plots as well. The low abundance of chipmunks in these plots will likely limit natural bitterbrush seedling establishment (Vander Wall 1994). However we caution interpretation of small mammal abundance as a parameter of habitat quality. This is because density and habitat quality are not always positively correlated (Van Horne 1983). Habitat of lower quality could be chosen by subordinate individuals that are in greater abundance than dominant individuals. We failed to

detect several small mammal species that we expected to use this site such as pinyon mouse (*Peromyscus truei*) Great basin pocket mouse (*Perognathus parvus*), Ord's kangaroo rat (*Dipodomys ordii*), grasshopper mouse (*Onychomys leucogaster*), and bushy-tailed woodrat (*Neotoma cinerea*). This could have been due to our summer trapping season but also may have been related to poor habitat. Exotic grass dominated sites have been found to have lower diversity of small mammals than intact sagebrush steppe (Ostoja and Schupp 2009).

Habitat quality can be assessed in several ways. Some studies have relied on capturing the demographic differences between animals using different possible habitats. Alternatively, physiological metrics such as presence of plasma metabolites and morphometric characteristics such as body condition are sometimes used to assess habitat quality. Other times the assessment of habitat quality has relied on measurements of environmental characteristics such as vegetative cover, forage quality, or potential nest locations. While estimating deer use of this site as winter range (assessing demographics, survival, as well as individual measures like body condition) will be important there may be a time lag where deer are distributed based on past habitat quality not present (found in birds Wiens et al. 1986). Therefore we have assessed mule deer winter range habitat based on several metrics such as small mammal abundance and plant cover and biomass that we think may react more quickly than the deer population itself. Responses to mule deer habitat improvement efforts are important to monitor as sites recover after juniper thinning. These monitoring efforts will be crucial for wise adaptive management decisions. In this study we used a space-for-time substitution within areas in a single site, which may be a useful strategy for managers to use when long term monitoring is impossible. These findings can be paired with population estimates obtained yearly at the site via helicopter survey as well as radio/GPS tracking in progress to holistically assess improvement to mule deer winter range habitat.

TABLES AND FIGURES

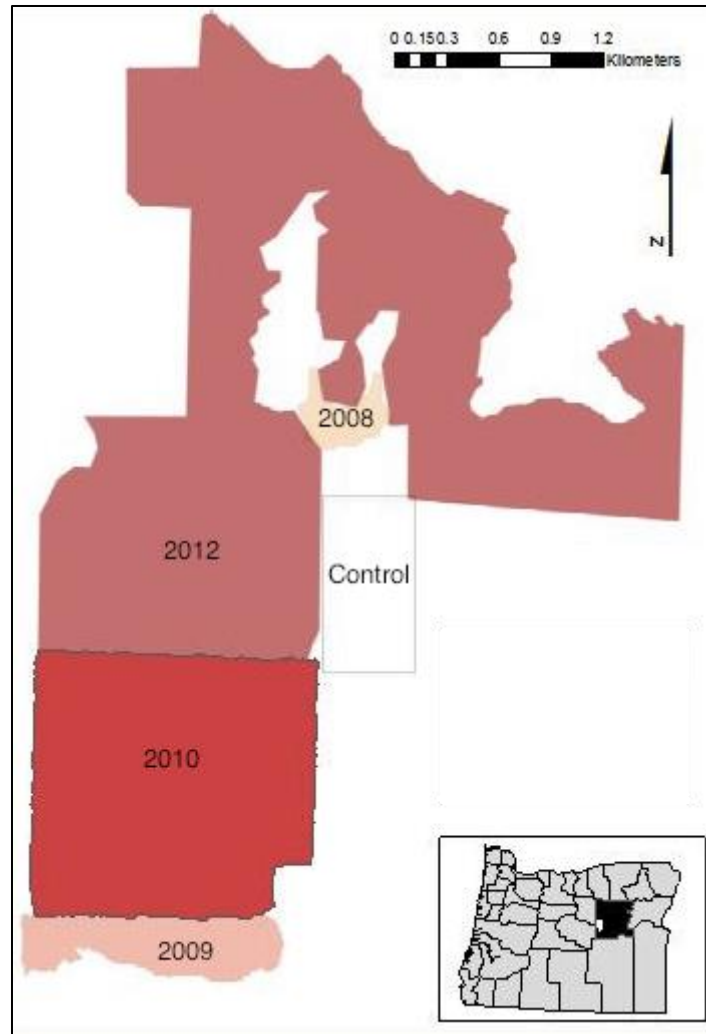


Figure 3.1. Study site map of the juniper thinning plots within Murderer's Creek basin. Overlays include the boundaries of the thinning treatments carried out in 2008, 2009, 2010, 2012 as well as the control area. In 2008 juniper were hand cut on 13.76 ha, in 2009 45.73 ha were cut. In 2010, 237.96 ha were cut and lastly in 2012 617.96 ha of post settlement juniper trees were cut. Trees within the 2010 area were piled and burned when felled on slopes not exceeding 30%. Inset shows location of the Phillip W. Schneider Wildlife Area operated by Oregon Department of Fish and Wildlife in Grant County, Oregon, USA.

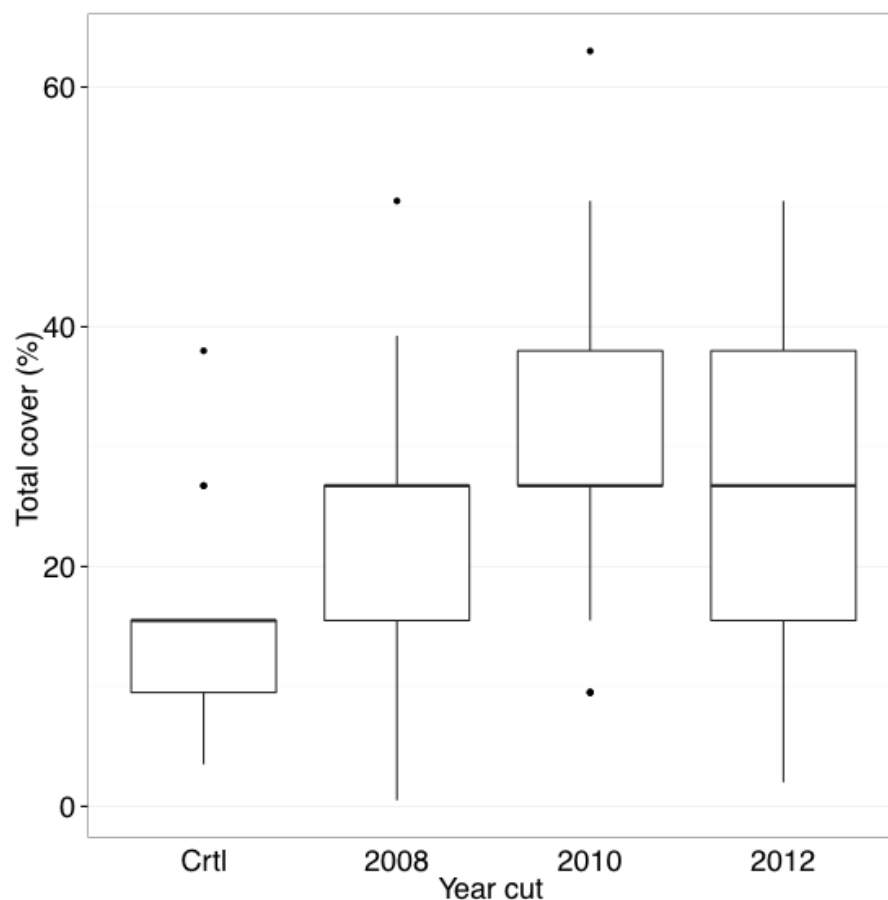


Figure 3.2. Total cover (A), grass cover (B), and forb cover (C) sampled in juniper-felled plots treated in 2008, 2010, 2012, as well as the uncut control area. Data were averaged from 10 cut trees per plot within microsites located under the felled tree (canopy), the original area under the standing tree canopy (duff) and 10 meters from the dripline of the tree (interspace). Samples were collected during July 2014 within the Phillip W. Schneider Wildlife Area. Boxes enclose the interquartiles, bolded line represents the median cover for each area. Whiskers extend to farthest point within 1.5 times that of the interquartiles. Points are outliers.

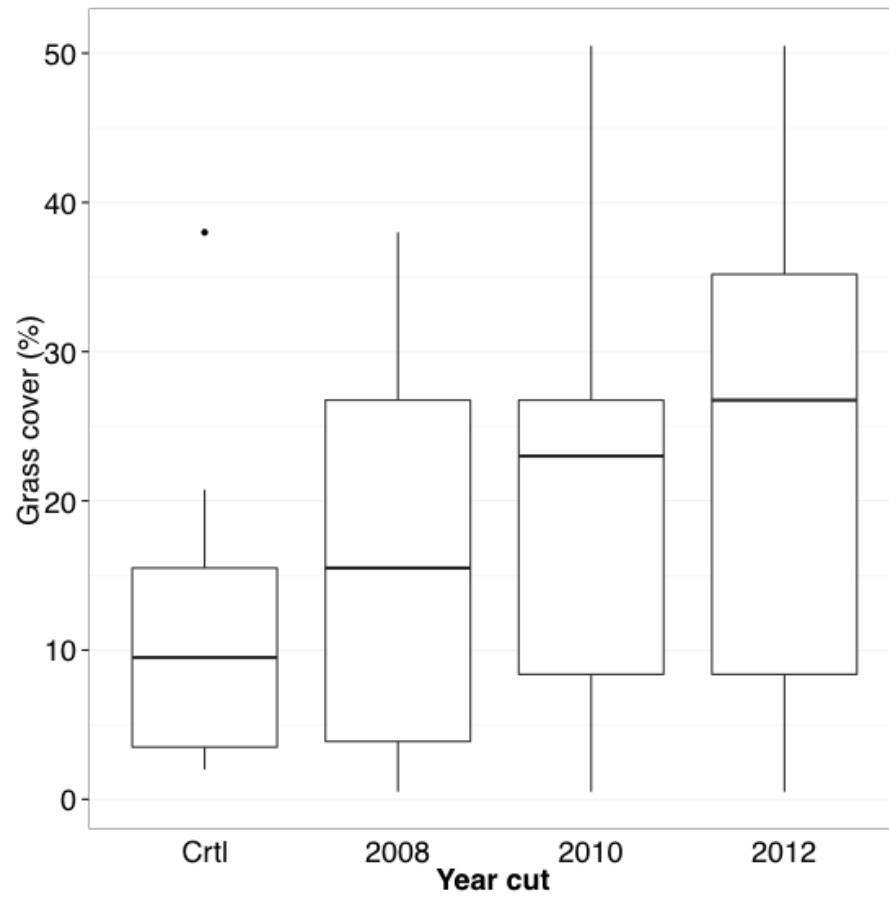


Figure 3.2. (B) (Continued)

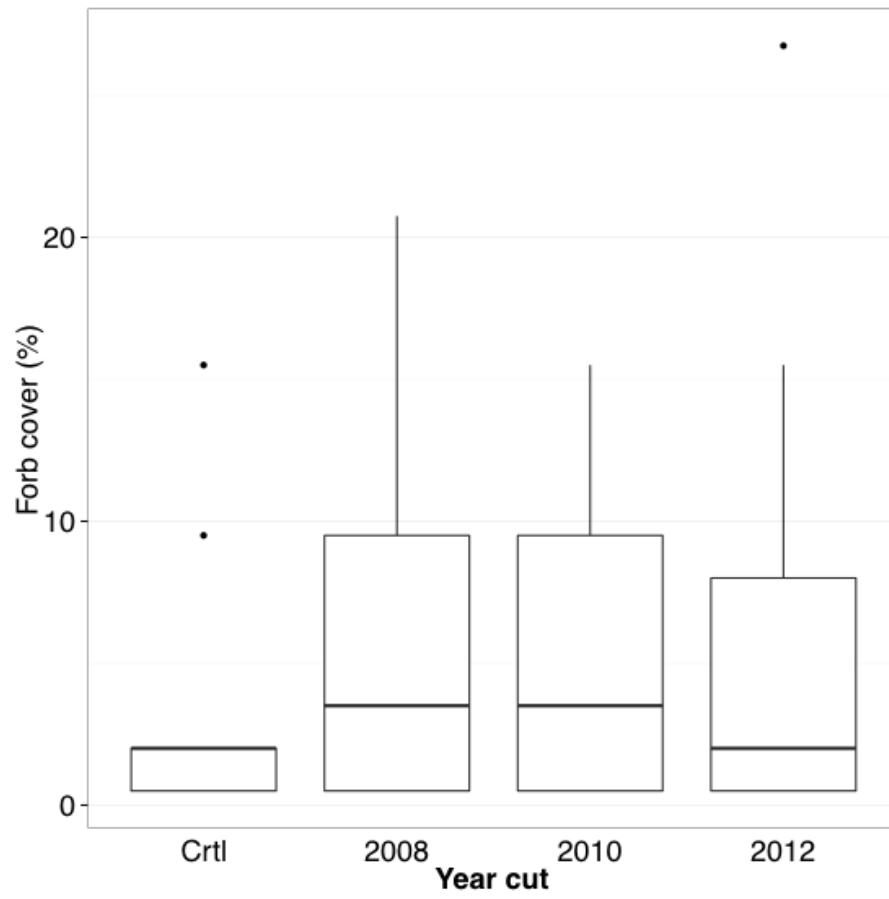


Figure 3.2. (C) (Continued)

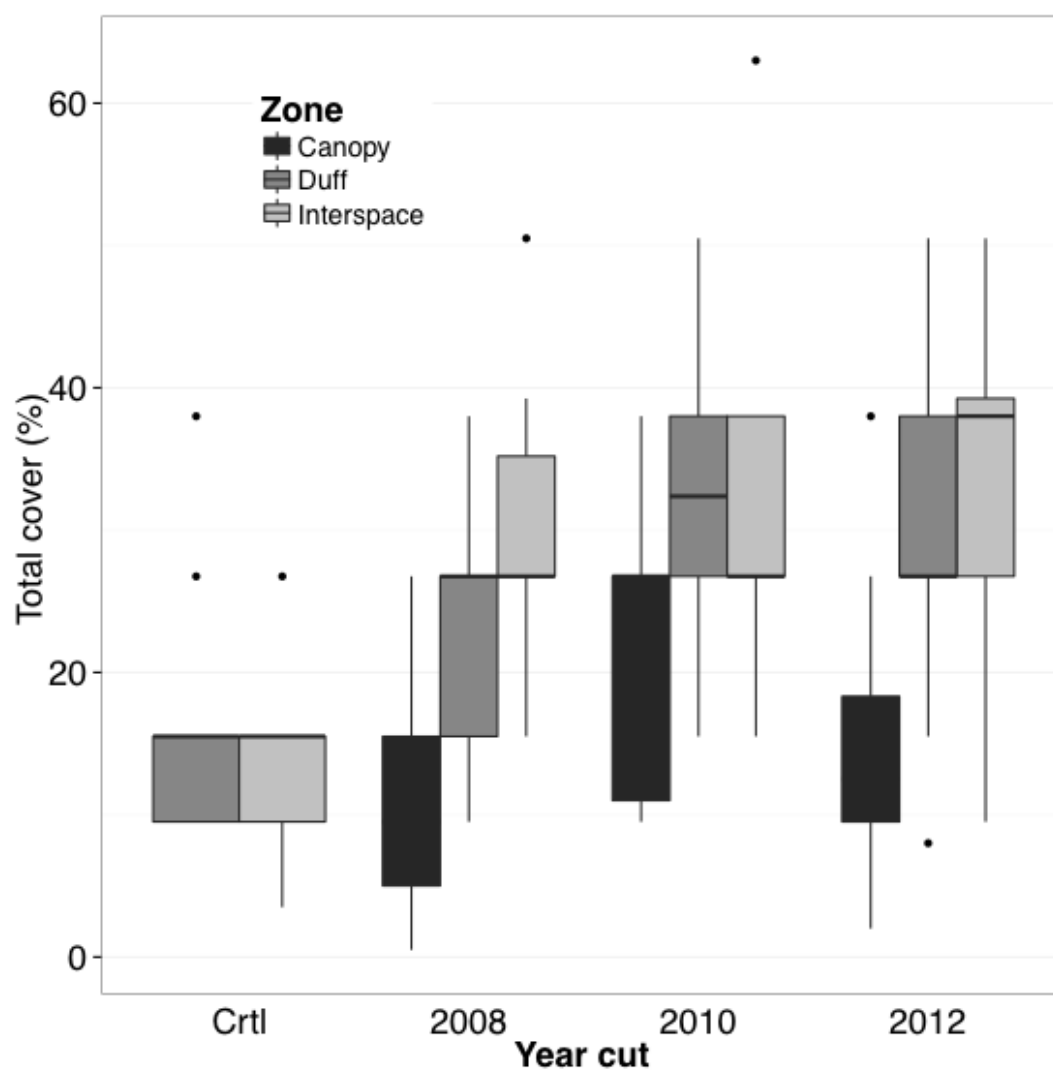


Figure 3.3. Total cover (A), grass cover (B), and forb cover (C) sampled in microsites located in the canopy, duff and interspace zones sampled within plots treated in 2008, 2010, 2012, and the uncut control area. Data were collected within two quadrats per zone at 10 cut trees per plot during July 2014 within the Phillip W. Schneider Wildlife Area. Boxes enclose the interquartiles, bolded line represents the median cover for each area. Whiskers extend to farthest point within 1.5 times that of the interquartiles. Points are outliers.

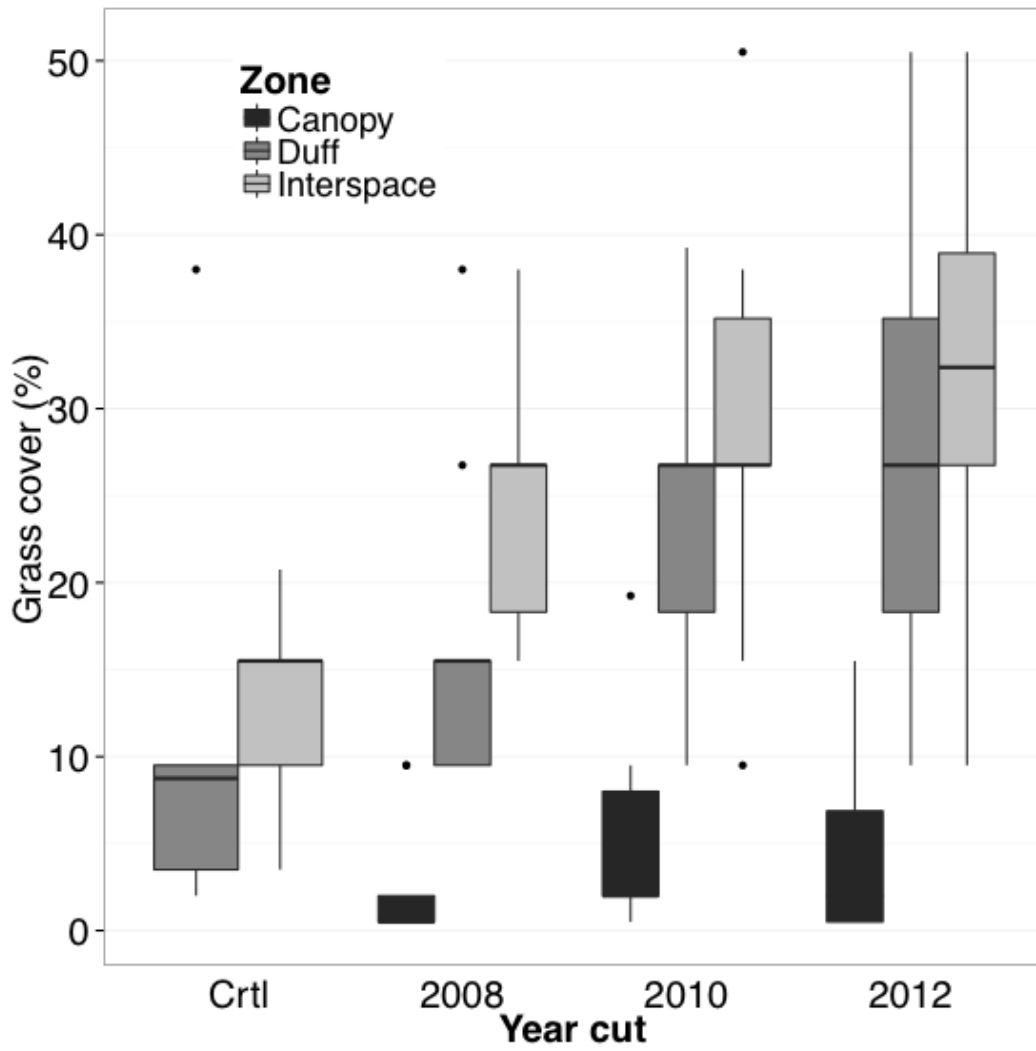


Figure 3.3 (B) (Continued)

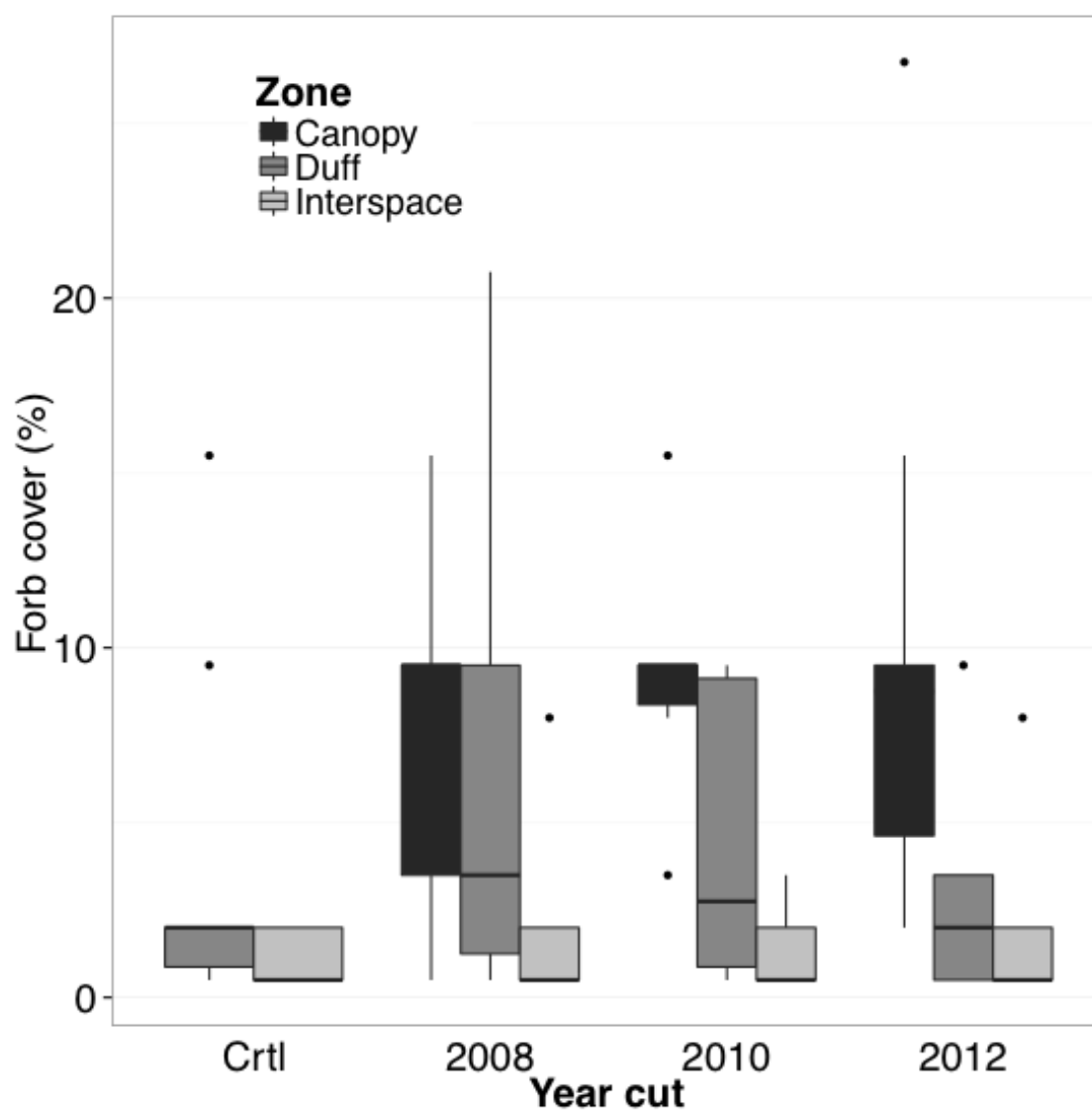


Figure 3.3 (C) (Continued)

Table 3.1. Difference in mean grass cover between microsite zones within juniper-felled plots treated in 2008, 2010, 2012, as well as the uncut control plot. Two quadrats (0.75 x 0.75m) were located under the felled tree (canopy), in the original area under the standing tree canopy (duff), and in the interspace 10 meters from the estimated dripline of the tree. Data were averaged across quadrats within the microsite zones. Data were collected at 10 cut trees per plot during July 2014 within the Phillip W. Schneider Wildlife Area.

Plot	Contrast	Difference in			SE	df
		means ¹	Lower CI	Upper CI		
2008	Canopy - Duff	-13.88 *	-24.14	-3.61	3.98	54
2008	Canopy - Interspace	-21.90 *	-32.16	-11.64	3.98	54
2008	Duff - Interspace	-8.03	-18.29	2.24	3.98	54
2010	Canopy - Duff	-19.78 *	-30.04	-9.51	3.98	54
2010	Canopy - Interspace	-23.15 *	-33.41	-12.89	3.98	54
2010	Duff - Interspace	-3.38	-13.64	6.89	3.98	54
2012	Canopy - Duff	-23.15 *	-33.41	-12.89	3.98	54
2012	Canopy - Interspace	-27.90 *	-38.16	-17.64	3.98	54
2012	Duff - Interspace	-4.75	-15.01	5.51	3.98	54
Control	Duff - Interspace	-2.78	-14.38	8.83	4.41	36

¹ Difference in mean estimated for the comparison

* Denotes statistical significance at the 98.75% confidence level

Table 3.2. Ratio of median forb cover between microsite zones within juniper-felled plots treated in 2008, 2010, 2012, as well as the uncut control plot. Subplots were located under the felled tree (canopy), in the original zone under the standing tree canopy (duff), and in the interspace 10 meters from the estimated dripline of the tree at 10 cut trees per treatment plot within the Phillip W. Schneider Wildlife Area, Grant County, Oregon, USA. Years refer to the year juniper thinning management was undertaken in the plot.

Plot	Contrast	Ratio of medians ¹	Lower CI	Upper CI	SE	df
2008	Canopy / Duff	1.54	0.66	3.59	0.33	54
2008	Canopy / Interspace	3.88 *	1.66	9.04	0.33	54
2008	Duff / Interspace	2.52 *	1.08	5.87	0.33	54
2010	Canopy / Duff	2.71 *	1.16	6.33	0.33	54
2010	Canopy / Interspace	6.12 *	2.63	14.27	0.33	54
2010	Duff / Interspace	2.26	0.97	5.26	0.33	54
2012	Canopy / Duff	3.38 *	1.45	7.89	0.33	54
2012	Canopy / Interspace	4.76 *	2.04	11.09	0.33	54
2012	Duff / Interspace	1.41	0.60	3.28	0.33	54
Control	Duff / Interspace	1.82	0.85	3.91	0.33	36

¹ Ratio of medians represent the number of times greater median forb cover was between the comparison in each plot

* Denotes statistical significance at the 96.67% confidence level for all 2008-2012 comparisons. Zone comparisons in the control plot were assessed at the 97.50% confidence level.

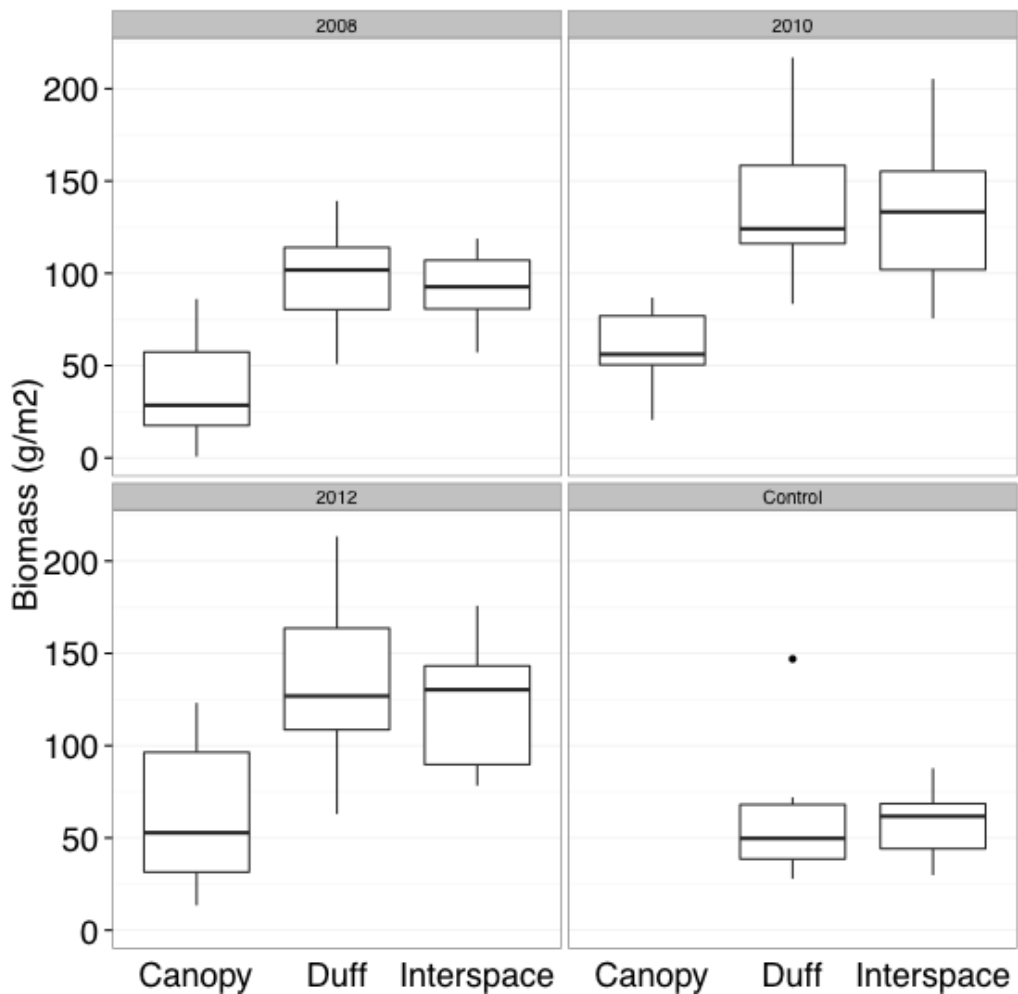


Figure 3.4. Herbaceous dry biomass (in grams per square meter) sampled in the canopy, duff, and interspace zones within juniper-cut plots treated in 2008, 2010, 2012, and the uncut control plot. Data were collected at 10 cut trees per plot within the Phillip W. Schneider Wildlife Area, Grant County, Oregon, USA. Subplots were located under the felled tree (canopy), in the original zone under the standing tree canopy (duff), and in the interspace 10 meters from the estimated dripline of the tree. Years refer to the year juniper thinning management was undertaken in the plot. Boxes enclose the interquartiles, bolded line represents the median cover for each area. Whiskers extend to farthest point within 1.5 times that of the interquartiles. Points are outliers.

Table 3.3. Difference in mean herbaceous biomass (g/m^2) between microsite zones within juniper-felled plots treated in 2008, 2010, 2012, as well as the uncut control plot. Subplots were located under the felled tree (canopy), in the original zone under the standing tree canopy (duff), and in the interspace 10 meters from the estimated dripline of the tree at 10 cut trees per treatment plot within the Phillip W. Schneider Wildlife Area, Grant County, Oregon, USA. Years refer to the year juniper thinning management was undertaken in the plot.

Plot	Contrast	Difference in means ¹	Lower CI	Upper CI	SE	df
2008	Canopy - Duff	-84.881 *	-133.18	-36.58	17.72	54
2008	Canopy - Interspace	-75.155 *	-123.46	-26.85	17.72	54
2008	Duff - Interspace	9.726	-38.58	58.03	17.72	54
2010	Canopy - Duff	-110.610 *	-158.91	-62.31	17.72	54
2010	Canopy - Interspace	-99.669 *	-147.97	-51.37	17.72	54
2010	Duff - Interspace	10.940	-37.36	59.24	17.72	54
2012	Canopy - Duff	-97.412 *	-145.72	-49.11	17.72	54
2012	Canopy - Interspace	-84.186 *	-132.49	-35.88	17.72	54
2012	Duff - Interspace	13.227	-35.08	61.53	17.72	54
Control	Duff - Interspace	0.691	-40.14	41.52	17.72	36

¹ Difference in mean estimated for the comparison

* Denotes statistical significance at the 97.98% confidence level

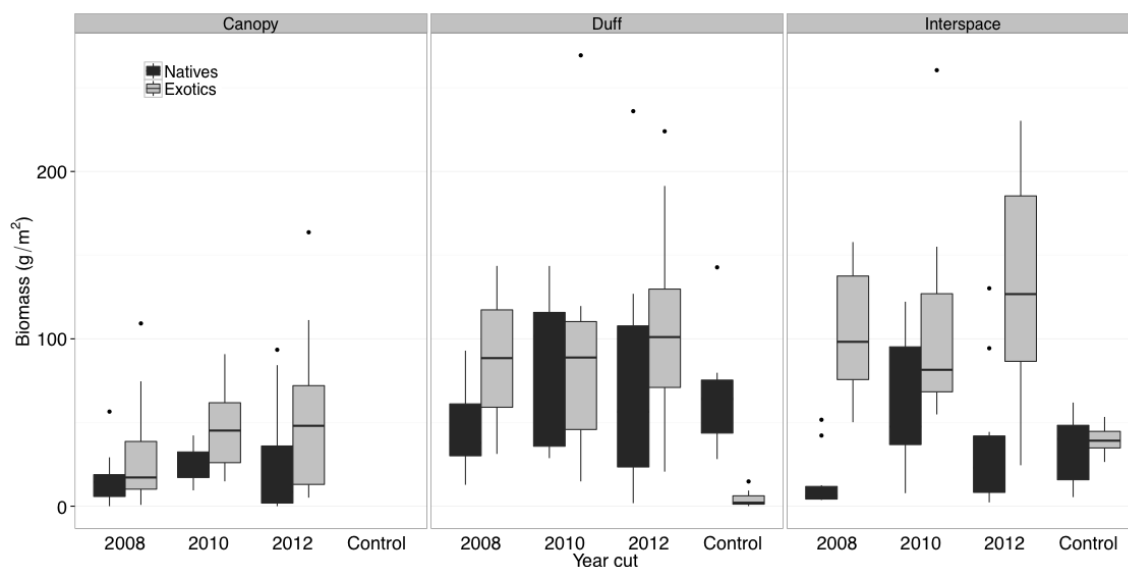


Figure 3.5. Native and exotic herbaceous biomass (in grams per square meter) in subplots located under the felled tree (canopy), in the original zone under the standing tree canopy (duff), and in the interspace 10 meters from the estimated dripline of the tree within 2008, 2010, 2012, and control plots (assessed in July 2014) at 10 cut trees per plot within the Phillip W. Schneider Wildlife Area, Grant County, Oregon, USA. Years refer to the year juniper thinning management was undertaken in the plot. Boxes enclose the interquartiles, bolded line represents the median cover for each area. Whiskers extend to farthest point within 1.5 times that of the interquartiles. Points are outliers.

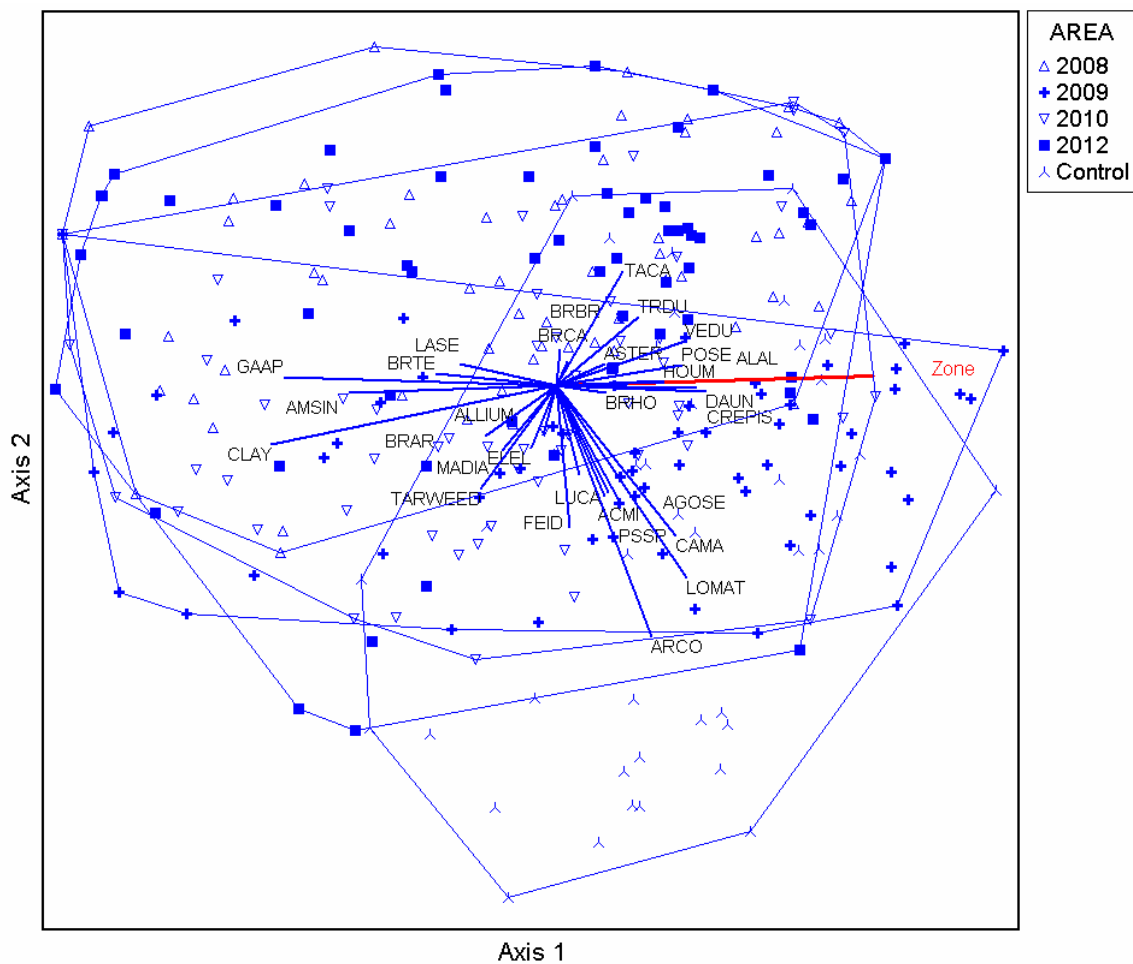


Figure 3.6. Non-metric multidimensional scaling (NMS) ordination plot of herbaceous species composition based on presence-absence data within samples. Rare species were removed prior to analysis. Points represent sample units in species space for all four juniper treatment plots and the uncut control. (A) Axes one and two are represented. Upward triangles are sample units from the 2008 plot, crosses are from the 2009 plot, downward triangles are from the 2010 plot, squares are from the 2012 plot and points symbolized by a λ are from the uncut control plot. Convex hulls have been drawn around plots. (B) Axes one and three are represented. Triangles are sample units from the canopy zone, squares are from the duff zone, and open circles are from the interspace zone. Convex hulls have been drawn around zones. In each representation, species have been overlaid as vectors that indicate direction and magnitude (length) of correlation between species and axes. Zone categorization has been overlaid as a joint plot overlay and is correlated with axis one. Distance measures were calculated by Jaccard dissimilarity and a three-dimensional solution was selected based on stress analysis. Species names are presented in Appendix C.

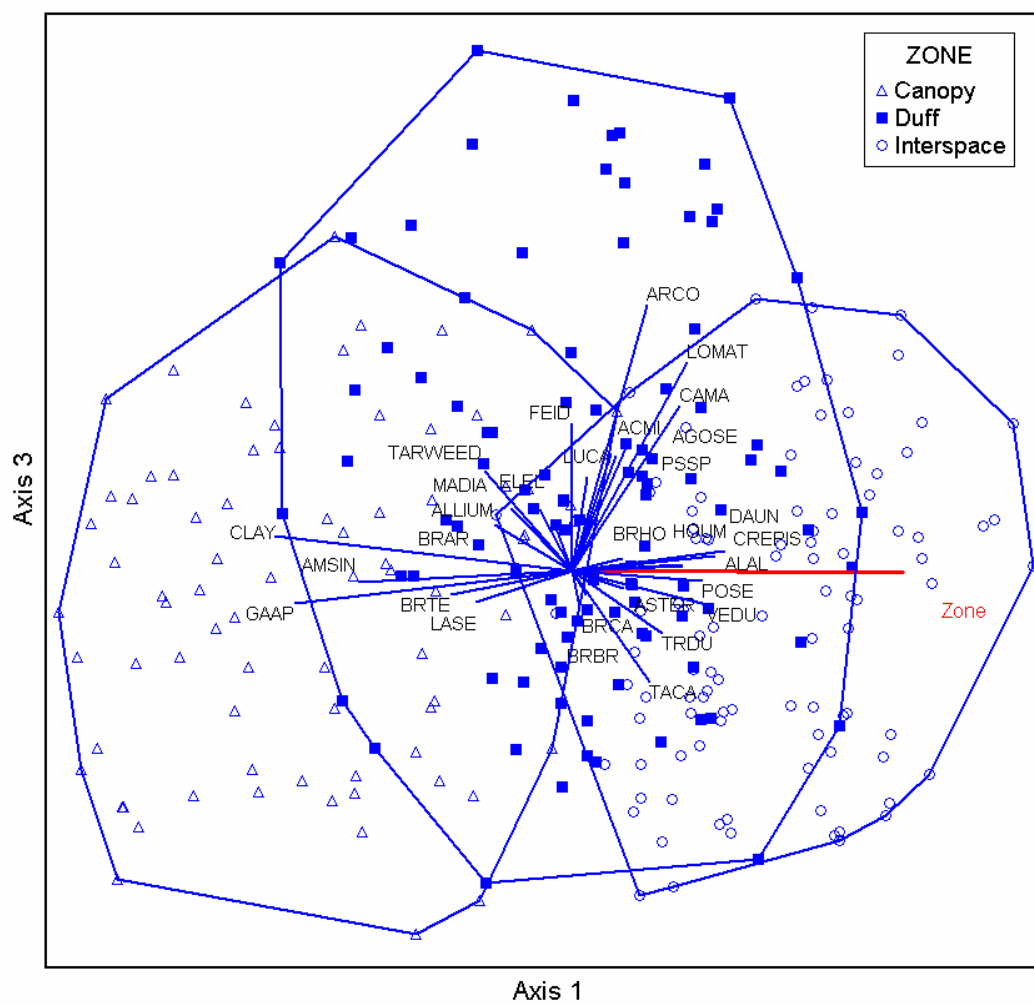


Figure 3.6. (B) (Continued)

Table 3.4. Species, abundance, sex ratio, and indexed diversity of small mammals trapped in June and July 2014. In the 2009, 2010, and 2012 plot we trapped 30 June 2014-3 July 2014 and 15 July 2014-18 July 2014. We trapped in the 2008 plot and the control plot between 4 July 2014-7 July 2014 and between 20 July 2014-23 July 23 2014. Abundance is pooled over the two 4-night trapping sessions conducted on each treatment plot within the Phillip W. Schneider Wildlife Area, Grant County, Oregon, USA, a juniper-invaded shrub-steppe. Junipers were felled in adjacent plots in 2008, 2009, 2010, and 2012 while trees were uncut in the neighboring control plot. Animals were live-trapped with Sherman 3x3.5x9 inch traps.

Plot	Species					Total Abundance	Brillouin indexed diversity
	PEMA ¹	TAMI ²	LECU ³	SYNU ⁴	Unknown ⁵		
2008	95 (43%)	13 (38%)	1	0	0	112	0.388
2009	26 (52%)	2 (50%)	0	6	0	37	0.590
2010	43 (56%)	1 (100%)	1	2	1	53	0.384
2012	80 (54%)	1 (100%)	0	1	0	85	0.107
Control	24 (39%)	6 (60%)	0	0	0	32	0.443

¹ PEMA is deer mouse (*Peromyscus maniculatus*)

² TAMI is least chipmunk (*Tamias minimus*)

³ LECU is sagebrush vole (*Lemmiscus curtatus*)

⁴ SYNU is mountain cottontail (*Sylvilagus nuttallii*)

⁵ One individual escaped before species identification

Table 3.5. Mean tree and shrub cover (%) along mammal trapping transects in juniper-felled plots treated in 2008, 2010, 2012, as well as the uncut control plot in the Phillip W. Schneider Wildlife Area, OR. Data were collected using the line intercept method in July 2014 and averaged from three 250 m transects per plot except in the 2008 plot, in which shorter transects were used due to the dimensions of the cut plot. In the 2009 plot we collected shrub cover data from a random subsample of the transect due to dense shrub cover. We divided the 250 m transect into 50 m sections and randomly sampled 2 of the 5 possible sections. To obtain a percentage of shrub cover point-intercept values for each species was divided by the total length of sampled transect and multiplied by 100.

Plot	JUOC	JUOC	PUTR ¹	ERNA ²	CHVI ³	ARAR ⁴	Rosa sp.	PERA ⁵	Total Shrub Cover
	seedling/sapling	Overstory							
2008	0.007	0	0.664	0.289	0	0	0	0	0.953
2009	0	0	1.677	0	0.001	22.210	0	0	23.887
2010	0.104	0	0.317	0	0	0	0.073	0.168	0.559
2012	0.069	0	0	1.613	1.613	0.759	0	0.261	4.247
Control	0	9.919	0.087	0	0	0	0	0	0.087
Average	0.022	2.019	0.549	0.381	0.323	4.594	0.015	0.086	5.947

¹ PUTR is antelope bitterbrush (*Purshia tridentata*)

² ERNA is rubber rabbitbrush (*Ericameria nauseosa*)

³ CHVI is green rabbitbrush (*Chrysothamnus viscidiflorus*)

⁴ ARAR is little sagebrush (*Artemisia arbuscula*)

⁵ PERA is wild crab apple (*Peraphyllum ramosissimum*)

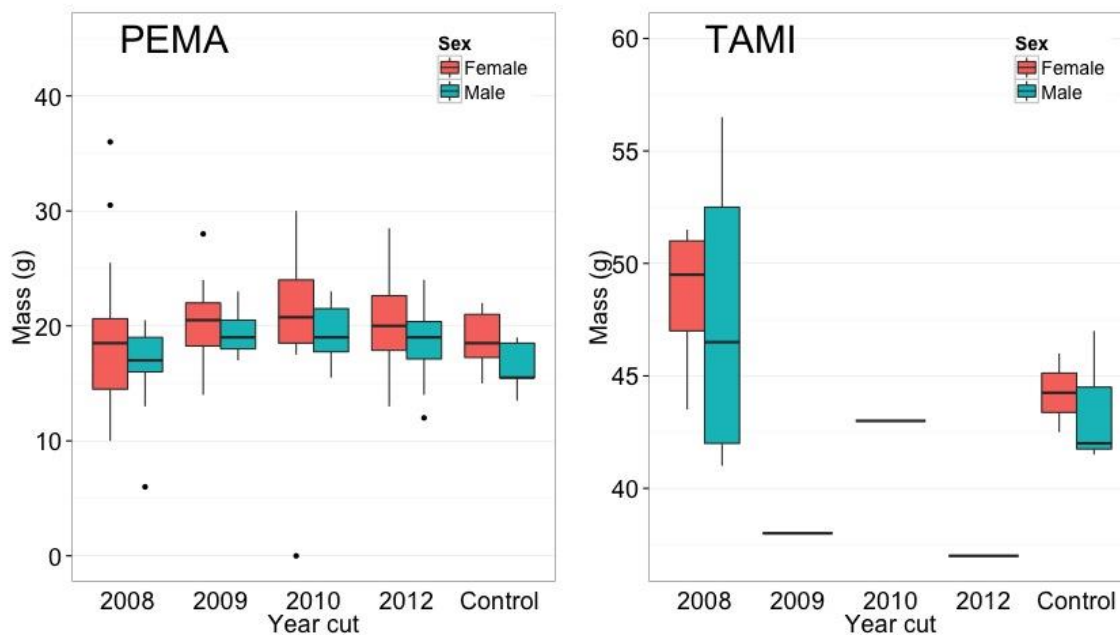


Figure 3.7. Body mass (grams) for males and female captured small mammals trapped in two 4-day trapping sessions for deer mouse (PEMA, *Peromyscus maniculatus*) (A) and least chipmunk (TAMI, *Tamias minimus*) (B) trapped from dusk to early morning within juniper thinned plots treated in 2008, 2009, 2010, and within an adjacent control plot. All mammals were trapped with Sherman 3x3.5x9 inch traps in the Phillip W. Schneider Wildlife Area, Grant County, Oregon, USA. Boxes enclose the interquartiles, bolded line represents the median cover for each area. Lines (whiskers) extend to 1.5 times that of the interquartiles and points are outliers that fall beyond the whiskers. Bolded lines with no box indicates only one individual of the species was trapped in each cut area.

CHAPTER 4: CONCLUSION TO THE THESIS

The objectives for my research were to 1) understand the independent and interactive effects of juniper cutting and livestock grazing on shrub-steppe recovery across different juniper woodland densities, and 2) assess the responses of small mammal communities and understory vegetation in microhabitats and along a time-since-treatment chronoserries. Both of these objectives address persistent knowledge gaps of shrub-steppe recovery. By understanding how other land use practices such as livestock grazing may influence post-juniper cutting, managers will be better equipped to make decisions concerning deferment of grazing after juniper management. Information about small mammal abundance and diversity across the chronoserries can be incorporated into site condition assessments. Lastly, knowledge of plant community dynamics associated with felled juniper left on the site will inform managers about possible juniper treatment options and post-juniper cutting management strategies.

To address the first set of objectives I compared treatment combinations of cattle exclusion and juniper cutting at three sites where each site was determined to be in a different phase of juniper woodland development (Chapter 2). I compared vegetative responses of cover and productivity before juniper cutting and cattle exclusion treatment to the same plots one year after treatments were established. I found few responses to juniper cutting and grazing shortly after treatment though magnitude and direction of responses depended on site. At the phase I site (MC), herbaceous productivity was unchanged in cut plots while cutting at the phase II site (FC) doubled herbaceous biomass. Unfortunately, most of this was due to the ability of exotics to capitalize on the release of resources. Cutting juniper at the phase III site (WG) decreased herbaceous productivity when compared to uncut plots. At MC and FC cut plots had statistically significant increases in exotic species cover when compared to plots before treatment establishment. Forbs responded positively to cutting at all three sites. However I documented a statistically significant increase in forb cover between 2012 and 2014 in plots that were untreated as well indicating that interannual variation in growing conditions favored forbs in 2014 irrespective to our treatment. Bunchgrass cover as well as the few shrubs remaining at the wildlife area did not respond to juniper cutting or grazing in the first year post-treatment. Because shrubs are slower growing I expected shrub responses to juniper cutting and cattle exclusion to be small in magnitude. However, I expected bunchgrasses to take advantage of the release from juniper

overstory within one year. Taken together these results indicate that recovery across the three sites is likely to be slow and site specific. Without active restoration to manage the post-cut increase in exotic grass, I expect MC and WG will tend to remain in an alternative stable state with little recovery of shrub-steppe components. I recommend monitoring of these sites to see if a lag in bunchgrass response is occurring.

In my second study, I compared responses of shrub-steppe vegetation and small mammal communities after juniper cutting across a chronoserries of plots cut in 2008, 2009, 2010, and 2012 (Chapter 3). Plots that were most recently cut had the highest exotic productivity and grass cover. Small mammals were found in low abundance and richness across the chronoserries though may be showing positive signs as time since juniper thinning increases. Key ecosystem drivers such as chipmunks were in low abundances in plots with little shrub cover. This will make recovery of the shrubs such as bitterbrush difficult without active restoration efforts. Within each plot I also assessed the responses of cover and productivity across microhabitats created by individual felled juniper trees. Grass cover was highest in the between-tree interspace and lowest under the canopy of the felled juniper. A concern for recovery efforts was the apparent loss of native productivity in the interspace as time since thinning increases. Due to lack of replication at the level of the chronoserries I was unable to statistically compare responses across time. Inference should only be made to this series of cuts though I believe other areas within the wildlife area will respond similarly. In addition, collecting a single year of post-treatment data limited my ability to understand the interannual variability inherent in the responses of vegetation and small mammals to juniper thinning.

Exotic grass dominance increased after juniper cutting at most sites I studied in the PWS Wildlife Area. Managers will need to actively reduce exotic cover and productivity to push these sites into a native-dominated stable state after juniper removal. I recommend further testing and refining of conceptual models proposed by state and transition models for juniper woodland and shrub-steppe ecosystems (Petersen et al. 2009). In addition, reapplication of juniper cutting will be necessary to combat the regeneration of these woodlands though we did not find appreciable seedling densities in cut areas. We did observe resprouting from remaining stems near the ground surface on cut trees and small sapling stage juniper that likely was too small to be detected during treatment.

Juniper management at the wildlife area was chiefly carried out to restore habitat for wintering mule deer. Survival of mule deer is driven by many factors including nutritional quality in both summer and winter ranges and predation. On winter ranges thermal protection (Parker and Gillingham 1990) and the forage quality (Garrott et al. 1987) affect mule deer occupancy and success. Mule deer diet is highly seasonal though shrubs and trees make up a large proportion of their diet throughout the year (Kufeld et al. 1973). In the winter, shrub and tree forage is especially important making up on average 74% of their diet across the species' range. At the wildlife area I did not detect recovery of key responses such as recruitment or change in cover of native shrub species. Shrub recovery was not found in either study so managers will need to find ways to actively restore this component of the shrub-steppe if this response continues to be minimal. I recommend more research at the PWS WA to determine the most successful strategies for shrub establishment.

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APPENDICES

APPENDIX A: MODEL ESTIMATES FOR CHAPTER 2

Table A.1. Estimated median biomass (g/m^2) within juniper treated plots as well as the uncut plots at sites within the Phillip W. Schneider Wildlife Area, Grant County, Oregon. Data were collected in 2012 (pre) and 2014 (post) representing before and after treatment establishment. Treatments included juniper canopy manipulation (cut/uncut) and grazing enclosure (grazed/ungrazed). All levels of grazing enclosure treatment have been pooled. Sites MC, FC, and WG correspond to Murderer's Creek, Flat Creek, and Wiley Gulch, respectively.

Site	Juniper	Year	Median biomass (g/m^2) ¹	Lower CI	Upper CI
MC	Cut	Pre	221.569	128.345	382.506
MC	Uncut	Pre	164.155	95.088	283.389
MC	Cut	Post	142.966	82.814	246.810
MC	Uncut	Post	110.478	63.995	190.723
FC	Cut	Pre	35.468	20.545	61.230
FC	Uncut	Pre	32.575	18.869	56.236
FC	Cut	Post	78.450	45.442	135.431
FC	Uncut	Post	28.899	16.740	49.890
WG	Cut	Pre	108.290	61.224	191.539
WG	Uncut	Pre	85.946	49.785	148.373
WG	Cut	Post	91.193	52.824	157.430
WG	Uncut	Post	85.951	49.788	148.381

¹ Median estimated by general linear mixed model

* 99.17% confidence intervals based on false discovery rate (FDR) correction

Table A.2. Estimated native and exotic biomass (g/m^2) within juniper treated plots as well as the uncut plots at sites within the Phillip W. Schneider Wildlife Area, Grant County, Oregon. Data were collected in 2014, one year after treatment. Treatments included juniper canopy manipulation (cut/uncut) and grazing exclosure (grazed/ungrazed). All levels of grazing exclosure treatment have been pooled. Sites MC, FC, and WG correspond to Murderer’s Creek, Flat Creek, and Wiley Gulch, respectively.

Site	Juniper	Native or Exotic	Mean biomass (g/m^2) ¹	Lower CI	Upper CI
MC	Cut	Native	9.770	-11.389	30.929
MC	Uncut	Native	10.424	-10.736	31.583
MC	Cut	Exotic	136.719	115.560	157.878
MC	Uncut	Exotic	105.019	83.860	126.178
FC	Cut	Native	41.803	20.644	62.962
FC	Uncut	Native	22.866	1.707	44.025
FC	Cut	Exotic	45.931	24.772	67.091
FC	Uncut	Exotic	12.247	-8.912	33.406
WG	Cut	Native	10.311	-10.848	31.470
WG	Uncut	Native	5.746	-15.413	26.905
WG	Cut	Exotic	87.780	66.621	108.939
WG	Uncut	Exotic	87.031	65.872	108.190

¹ Mean estimated by general linear mixed model

* 98.75% confidence intervals based on false discovery rate (FDR) correction

Table A.3. Estimated median bunchgrass cover (%) within juniper treated and grazing enclosure treatment plots at sites within the Phillip W. Schneider Wildlife Area, Grant County, Oregon. Data were collected in 2012 (pre) and 2014 (post) representing before and after treatment establishment. Treatments included juniper canopy manipulation (cut/uncut) and grazing enclosure (grazed/ungrazed). Sites MC, FC, and WG correspond to Murderer's Creek, Flat Creek, and Wiley Gulch, respectively.

Site	Juniper Treatment	Grazing Treatment	Year	Median cover (%) ¹	Lower CI	Upper CI
MC	Uncut	Grazed	Pre	2.404	0.938	6.163
MC	Uncut	Ungrazed	Pre	1.871	0.730	4.797
MC	Cut	Grazed	Pre	2.100	0.819	5.384
MC	Cut	Ungrazed	Pre	3.325	1.297	8.524
MC	Uncut	Grazed	Post	2.247	0.876	5.760
MC	Uncut	Ungrazed	Post	2.831	1.104	7.258
MC	Cut	Grazed	Post	2.970	1.158	7.614
MC	Cut	Ungrazed	Post	3.399	1.326	8.716
FC	Uncut	Grazed	Pre	2.404	0.957	6.039
FC	Uncut	Ungrazed	Pre	7.071	2.758	18.130
FC	Cut	Grazed	Pre	4.990	1.986	12.535
FC	Cut	Ungrazed	Pre	2.698	1.052	6.918
FC	Uncut	Grazed	Post	3.816	1.519	9.586
FC	Uncut	Ungrazed	Post	6.057	2.362	15.530
FC	Cut	Grazed	Post	9.916	3.947	24.911
FC	Cut	Ungrazed	Post	11.501	4.486	29.489
WG	Uncut	Grazed	Pre	1.871	0.730	4.797
WG	Uncut	Ungrazed	Pre	2.247	0.876	5.760
WG	Cut	Grazed	Pre	2.100	0.819	5.384
WG	Cut	Ungrazed	Pre	4.893	1.908	12.544
WG	Uncut	Grazed	Post	1.871	0.730	4.797
WG	Uncut	Ungrazed	Post	3.333	1.300	8.546
WG	Cut	Grazed	Post	2.970	1.158	7.614
WG	Cut	Ungrazed	Post	4.445	1.734	11.397

¹ Median estimated by general linear mixed model

* 99.7% confidence intervals based on false discovery rate (FDR) correction

Table A.4. Estimated mean exotic cover (%) within juniper treated plots as well as the uncut plots at sites within the Phillip W. Schneider Wildlife Area, Grant County, Oregon. Data were collected in 2012 (pre) and 2014 (post) representing before and after treatment establishment. Treatments included juniper canopy manipulation (cut/uncut) and grazing enclosure (grazed/ungrazed). All levels of grazing enclosure treatment have been pooled. Sites MC, FC, and WG correspond to Murderer’s Creek, Flat Creek, and Wiley Gulch, respectively.

Site	Juniper Treatment	Year	Mean Cover (%) ¹	Lower CI	Upper CI	SE	df
MC	Cut	Pre	94.028	62.239	125.817	9.674	15
MC	Cut	Post	118.611	86.822	150.400	9.674	15
MC	Uncut	Pre	121.389	89.600	153.178	9.674	15
MC	Uncut	Post	105.417	73.628	137.206	9.674	15
FC	Cut	Pre	30.417	-1.372	62.206	9.674	15
FC	Cut	Post	58.889	27.100	90.678	9.674	15
FC	Uncut	Pre	52.792	21.003	84.581	9.674	15
FC	Uncut	Post	36.389	4.600	68.178	9.674	15
WG	Cut	Pre	98.750	66.961	130.539	9.674	15
WG	Cut	Post	119.444	87.655	151.233	9.674	15
WG	Uncut	Pre	92.778	60.989	124.567	9.674	15
WG	Uncut	Post	99.167	67.378	130.956	9.674	15

¹ Mean estimated by general linear mixed model

* 98.33% confidence intervals based on false discovery rate (FDR) correction

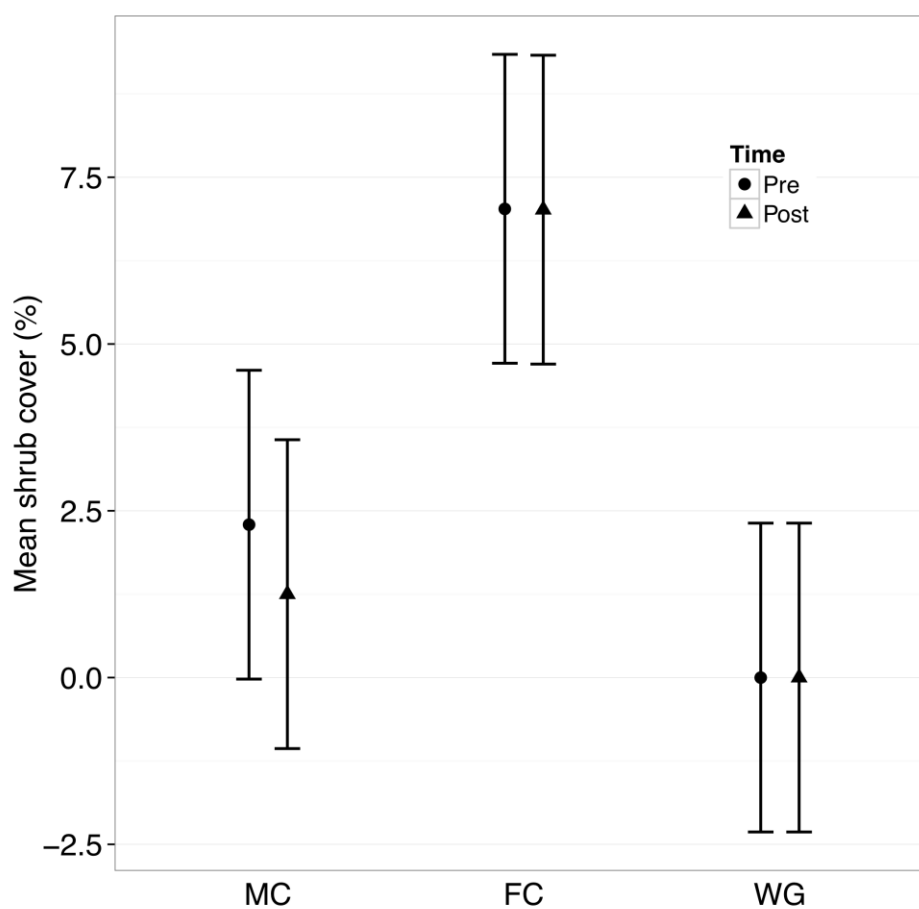


Figure A.1. Estimated mean shrub cover (%) of all plots at each three sites within the Phillip W. Schneider Wildlife Area, Grant County, Oregon, USA. Means were pooled across juniper and cattle exclusion treatments in 2012 (circles) and 2014 (triangles). Bars are 95% confidence intervals. Sites MC, FC, and WG correspond to Murderer's Creek, Flat Creek, and Wiley Gulch, respectively.

Table A.5. Estimated median forb cover (%) within juniper treated and grazing exclosure treatment plots at sites within the Phillip W. Schneider Wildlife Area, Grant County, Oregon. Data were pooled across 2012 and 2014 representing before and after treatment establishment. Treatments included juniper canopy manipulation (cut/uncut) and grazing exclosure (grazed/ungrazed). Sites MC, FC, and WG correspond to Murderer’s Creek, Flat Creek, and Wiley Gulch, respectively.

Site	Juniper Treatment	Grazing Treatment	Median forb cover (%) ¹	Lower CI	Upper CI
MC	Cut	Grazed	5.373	3.502	8.243
MC	Cut	Ungrazed	6.829	4.432	10.523
MC	Uncut	Grazed	7.508	4.894	11.519
MC	Uncut	Ungrazed	7.801	5.063	12.021
FC	Cut	Grazed	10.018	6.502	15.438
FC	Cut	Ungrazed	10.173	6.602	15.676
FC	Uncut	Grazed	7.809	5.068	12.034
FC	Uncut	Ungrazed	12.598	8.175	19.412
WG	Cut	Grazed	2.657	1.725	4.095
WG	Cut	Ungrazed	3.732	2.422	5.750
WG	Uncut	Grazed	4.898	3.178	7.547
WG	Uncut	Ungrazed	3.566	2.314	5.496

¹ Median estimated by general linear mixed model

* 95% confidence intervals based on false discovery rate (FDR) correction

APPENDIX B: MODEL ESTIMATES FOR CHAPTER 3

Table B.1. Estimated mean grass cover (%) within juniper-felled plots treated in 2008, 2009, 2010, 2012, as well as the uncut control plot at the Phillip W. Schneider Wildlife Area, Grant County, Oregon. Two quadrats (0.75 x 0.75m) were located under the felled tree (canopy), in the original area under the standing tree canopy (duff), and in the interspace 10 meters from the estimated dripline of the tree. Data were averaged across quadrats within the microsite zones. Data were collected at 10 cut trees per plot during July 2014.

Plot	Zone	Mean grass cover (%) ¹	Lower CI	Upper CI
2008	Canopy	2.600	-3.155	8.355
2008	Duff	16.475	10.720	22.230
2008	Interspace	24.500	18.745	30.255
2009	Canopy	5.375	-0.049	10.799
2009	Duff	18.050	12.151	23.949
2009	Interspace	10.700	4.801	16.599
2010	Canopy	5.375	-0.399	11.149
2010	Duff	25.150	19.376	30.924
2010	Interspace	28.525	22.751	34.299
2012	Canopy	4.250	-1.524	10.024
2012	Duff	27.400	21.626	33.174
2012	Interspace	32.150	26.376	37.924
Control	Duff	9.650	3.338	15.962
Control	Interspace	12.425	6.096	18.754

¹ Mean estimated by general linear mixed model

* 96.67% confidence intervals for all 2008-2012 plots. The control plot was assessed at the 98.75% confidence level.

Table B.2. Estimated median forb cover (%) within juniper-felled plots treated in 2008, 2009, 2010, 2012, as well as the uncut control plot at the Phillip W. Schneider Wildlife Area, Grant County, Oregon. Two quadrats (0.75 x 0.75m) were located under the felled tree (canopy), in the original area under the standing tree canopy (duff), and in the interspace 10 meters from the estimated dripline of the tree. Data were averaged across quadrats within the microsite zones. Data were collected at 10 cut trees per plot during July 2014.

Plot	Zone	Median forb cover (%) ¹	Lower CI	Upper CI
2008	Canopy	6.325	3.797	10.536
2008	Duff	4.105	2.464	6.839
2008	Interspace	1.631	0.979	2.716
2009	Canopy	12.425	7.497	20.593
2009	Duff	5.317	3.224	8.769
2009	Interspace	1.512	0.917	2.494
2010	Canopy	9.258	5.548	15.447
2010	Duff	3.410	2.044	5.690
2010	Interspace	1.512	0.906	2.523
2012	Canopy	7.756	4.648	12.941
2012	Duff	2.292	1.374	3.824
2012	Interspace	1.631	0.977	2.721
Control	Duff	2.627	1.558	4.428
Control	Interspace	1.443	0.855	2.436

¹ Median estimated by general linear mixed model

* 96.67% confidence intervals for all 2008-2012 plots. The control plot was assessed at the 97.50% confidence level.

Table B.3. Estimated mean herbaceous biomass (g/m^2) in microsite zones within juniper-felled plots treated in 2008, 2009, 2010, 2012, as well as the uncut control plot. Subplots were located under the felled tree (canopy), in the original zone under the standing tree canopy (duff), and in the interspace 10 meters from the estimated dripline of the tree at 10 cut trees per treatment plot within the Phillip W. Schneider Wildlife Area, Grant County, Oregon, USA. Years refer to the year juniper thinning management was undertaken in the plot.

Plot	Zone	Mean herbaceous productivity (g/m^2) ¹	Lower CI	Upper CI
2008	Canopy	51.811	20.979	82.643
2008	Duff	136.692	105.860	167.524
2008	Interspace	126.966	96.134	157.798
2009	Canopy	105.204	75.356	135.051
2009	Duff	167.925	138.909	196.940
2009	Interspace	76.928	47.912	105.943
2010	Canopy	81.245	50.313	112.176
2010	Duff	191.854	160.923	222.786
2010	Interspace	180.914	149.982	211.846
2012	Canopy	85.275	54.343	116.206
2012	Duff	182.687	151.756	213.619
2012	Interspace	169.460	138.529	200.392
Control	Duff	81.300	51.007	111.594
Control	Interspace	80.609	50.316	110.902

¹ Mean estimated by general linear mixed model

* 97.69% confidence intervals for all plots

Table B.4. Estimated median native herbaceous biomass (g/m^2) in microsite zones within juniper-felled plots treated in 2008, 2009, 2010, 2012, as well as the uncut control plot. Subplots were located under the felled tree (canopy), in the original zone under the standing tree canopy (duff), and in the interspace 10 meters from the estimated dripline of the tree at 10 cut trees per treatment plot within the Phillip W. Schneider Wildlife Area, Grant County, Oregon, USA. Years refer to the year juniper thinning management was undertaken in the plot.

Plot	Zone	Median Native Herbaceous Biomass (g/m^2) ¹	Lower CI	Upper CI
2008	Canopy	13.114	6.988	24.610
2008	Duff	42.152	22.462	79.105
2008	Interspace	11.046	5.886	20.729
2009	Canopy	51.111	28.932	90.293
2009	Duff	94.771	56.605	158.669
2009	Interspace	37.370	22.321	62.567
2010	Canopy	23.989	12.757	45.110
2010	Duff	73.913	39.306	138.991
2010	Interspace	47.749	25.392	89.790
2012	Canopy	11.996	6.379	22.558
2012	Duff	45.582	24.240	85.715
2012	Interspace	22.161	12.760	38.490
Control	Duff	61.501	35.410	106.816
Control	Interspace	29.534	17.005	51.295

¹ Median estimated by general linear mixed model

* 97.78% confidence intervals for all 2008-2012 plots. The control plot was assessed at the 96.25% confidence level.

Table B.5. Estimated median exotic herbaceous biomass (g/m^2) in microsite zones within juniper-felled plots treated in 2008, 2009, 2010, 2012, as well as the uncut control plot. Subplots were located under the felled tree (canopy), in the original zone under the standing tree canopy (duff), and in the interspace 10 meters from the estimated dripline of the tree at 10 cut trees per treatment plot within the Phillip W. Schneider Wildlife Area, Grant County, Oregon, USA. Years refer to the year juniper thinning management was undertaken in the plot.

Plot	Zone	Median exotic biomass (g/m^2) ¹	Lower CI	Upper CI
2008	Canopy	17.624	10.581	29.353
2008	Duff	80.859	48.548	134.674
2008	Interspace	99.469	59.721	165.671
2009	Canopy	29.987	18.110	49.653
2009	Duff	52.001	32.798	82.445
2009	Interspace	29.871	18.840	47.359
2010	Canopy	42.226	25.311	70.446
2010	Duff	73.723	44.190	122.992
2010	Interspace	97.072	58.186	161.945
2012	Canopy	34.835	20.881	58.116
2012	Duff	84.568	50.691	141.085
2012	Interspace	111.876	67.060	186.642
Control	Duff	2.211	1.366	3.576
Control	Interspace	38.850	24.014	62.852

¹ Median estimated by general linear mixed model

* 98.33% confidence intervals for all 2008-2012 plots. The control plot was assessed at the 98.75% confidence level.

APPENDIX C: SPECIES LIST

Table C.1. Common names, scientific names, and species codes for plants used as abbreviations in Figure 3.6.

Common name	Scientific name	Species Code
Common Yarrow	<i>Achillea millefolium</i>	ACMI
Agoseris	<i>Microseris sp.</i>	AGOSE
Pale alyssum, madwort	<i>Alyssum alyssoides</i>	ALAL
Onion	<i>Allium sp.</i>	ALLIUM
Yellow fiddleneck	<i>Amsinckia sp.</i>	AMSIN
Ballhead sandwort	<i>Arenaria congesta</i>	ARCO
Aster	<i>Asteraceae family</i>	ASTER
Rattlesnake grass	<i>Bromus briziformis</i>	BRBR
Mountain brome	<i>Bromus carinatus</i>	BRCA
Soft brome	<i>Bromus hordeaceus</i>	BRHO
Cheatgrass	<i>Bromus tectorum</i>	BRTE
Green-banded mariposa lily	<i>Calochortus macrocarpus</i>	CAMA
Miner's lettuce	<i>Claytonia sp.</i>	CLAY
Hawksbeard	<i>Crepis sp.</i>	CREPIS
Onespike danthonia	<i>Danthonia unispicata</i>	DAUN
Squirreltail	<i>Elymus elymoides</i>	ELEL
Bedstraw, Cleavers	<i>Galium aparine</i>	GAAP
Jagged chickweed	<i>Holosteum umbellatum</i>	HOUM
Prickly lettuce	<i>Lactuca serriola</i>	LASE
Biscuitroot	<i>Lomatium sp.</i>	LOMAT
Tailcup lupin	<i>Lupinus caudatus</i>	LUCA
Tarweed	<i>Madia sp.</i>	MADIA
Sandberg bluegrass	<i>Poa secunda</i>	POSE
Bluebunch wheatgrass	<i>Pseudoroegneria spicata</i>	PSSP
Medusahead	<i>Taeniatherum caput-medusae</i>	TACA
Tarweed II	<i>Madia sp.</i>	TARWEED
Salsify, Goat's beard	<i>Trogopogon dubius</i>	TRDU
Ventenata	<i>Ventenata dubia</i>	VEDU

