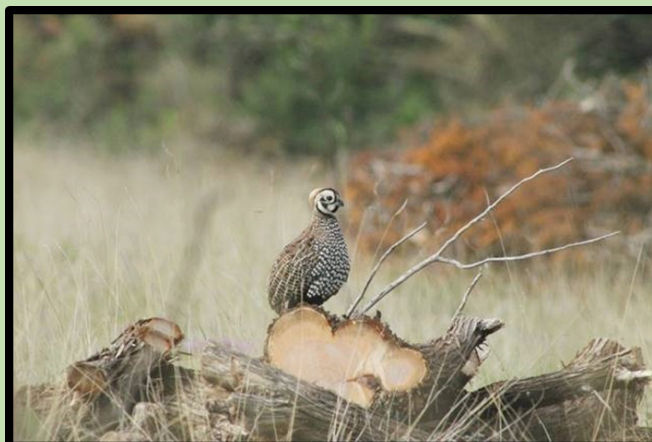


# Symposium Proceedings on Piñon-Juniper Habitats: Status and Management for Wildlife–2016

October 12–14, 2016  
Albuquerque, New Mexico



## Abstract

Piñon-juniper vegetation types, including juniper woodland and savannah, piñon-juniper, and piñon woodland, cover approximately 40 million ha in the western United States, where they provide ecosystem services, wildlife habitat, and cultural and aesthetic value (Romme et al. 2009). These ecosystems are also the sites of oil and gas activities, grazing, and urban development and are impacted by changing climate and wildfire. The realization that piñon-juniper ecosystems are being lost and degraded by human activities and changing climate (Cole et al. 2008, Williams et al. 2010, Clifford et al. 2011, McDowell et al. 2016) has stimulated interest in management of these habitats for wildlife. The goal of the 2016 symposium, Piñon-juniper Habitats: Status and Management for Wildlife, was to bring together information on the management of piñon-juniper ecosystems for the wildlife that depend on them.

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**Keywords:** adaptation, climate change, drought, fire, forage, invasive species, precipitation, restoration, thinning, vulnerability

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## Acknowledgments

The idea for this symposium was suggested in 2015 by Leland Pierce of the New Mexico Department of Game and Fish (NMDGF), in a conversation with Peggy Darr, NMDGF ornithologist, and Kristine Johnson, zoology coordinator at Natural Heritage New Mexico, University of New Mexico (NHNM). The symposium was organized by the steering committee, Bryan Dykstra and Karl Malcolm (U.S. Forest Service), Casey Hendricks (NM State Land Office), Kristine Johnson (NHNM), David Kreuper and Kammie Kruse (U.S. Fish and Wildlife Service), and Marikay Ramsey and Zoe Davidson (NM BLM). Marikay Ramsey provided funding from NM BLM. Kristine Johnson organized the scientific program. Jacqueline Smith (NHNM) organized additional logistics, including communications, check-in, and the field trip. Rebecca Keeshen (NHNM) took care of accounting and payment. Brian Dykstra, David Kreuper, Karl Malcolm, Esther Nelson (U.S. Forest Service), and Marikay Ramsey led discussion groups. Craig Allen (U.S. Geological Survey), Esteban Muldavin (NHNM), and Bill Romme (Colorado State University) led the field trip. Corrie Reasner (NHNM) and Zoe Davidson managed the slide presentations. NHNM made the symposium poster and posted it on their web site.

Karl Malcolm served as technical editor-in-chief for this volume. Other technical editors were: Brian Dykstra, Kristine Johnson, David Lightfoot (Museum of Southwestern Biology, University of New Mexico), Esteban Muldavin, and Marikay Ramsey. Deborah Finch and Patricia Cohn (U.S. Forest Service) edited the volume.

We especially thank the speakers for presenting their work and manuscripts. The organizing committee apologizes if we have failed to acknowledge anyone who contributed to this effort. We appreciate all who attended and participated in discussion groups.

## Introduction

This volume includes both full papers based on the symposium talks and abstracts of talks, in the case of papers submitted or in press elsewhere. The symposium program included several talks not represented here; only talks by those authors who provided full manuscripts or abstracts are included in this volume. The first group of papers focuses on piñon-juniper vegetation. William Romme gave two talks, one addressing the different types of piñon-juniper vegetation and their historical disturbance regimes, a second on tailoring management to the variability in these ecosystems. For purposes of these proceedings, these two talks were combined into a single contribution. Robert Parmenter and co-authors talked about the environmental drivers of mast production in piñon-juniper-oak ecosystems in the Southwest. With their talk on root-associated microbes and implications for management, Lela Andrews and Catherine Gehring added an often unappreciated perspective on the underground ecology of piñon trees and its importance for management. Sam Flake and Peter Weisberg reported on the influence of stand structure and aridity on piñon

mortality and implications for climate resilience. Jack Triepke presented models of piñon-juniper vulnerability to climate change in the Southwest. Ellis Margolis discussed the role of low-severity fire in a piñon-juniper savanna. Together, these chapters provide valuable background information on piñon-juniper natural history, ecosystem dynamics, disturbance regimes, and current threats.

The second group of talks focused on piñon-juniper habitat use and wildlife management. Three chapters address habitat needs and management for individual piñon-juniper wildlife species. Kristine Johnson and co-authors presented models of Pinyon Jay habitat use at home range, nesting colony, and nest scales at three study sites in New Mexico and discussed management implications for this declining bird. Lynn Wickersham and co-authors presented multi-scale models of Gray Vireo habitat use at four study sites in New Mexico and presented management recommendations. Lou Bender discussed habitat needs of deer and elk and assessed management practices in piñon-juniper woodlands.

The final group of talks discussed management actions and their effects on wildlife. Ryan Luna and co-authors reported on changes in habitat use of Montezuma Quail in response to tree canopy reduction in the Capitan Mountains, New Mexico. Pat Magee discussed the responses of avian communities to piñon-juniper woodland thinning in the Arkansas River Valley, Colorado. In a related talk from the same study, Jonathan Coop presented results of an assessment of the impacts of fuels treatments on woodland vegetation, fuels, birds, and modeled fire behavior. He suggested ways that fuels treatments could be optimized to maintain valued ecosystem components and still reduce fire hazards. David Lightfoot and co-authors discussed a study of the effects of thinning treatments on rodents, birds, and large mammals in piñon-juniper and ponderosa pine habitats in the Manzano Mountains, New Mexico. Alex Laurence-Traynor and co-authors presented the Bureau of Land Management's Assessment, Inventory, and Monitoring strategy (AIM, used in the Taos, NM BLM Field Office) as a method of monitoring treatment effects on vegetation.

These talks and the associated papers and abstracts in this volume provide a picture of piñon-juniper woodland condition and threats, wildlife use, and potential effects of woodland management practices on wildlife. Presenters emphasized the need for ongoing monitoring of wildlife populations and continued research on the effects of woodland treatments on wildlife habitats. The organizers hope that this volume will stimulate land and wildlife managers to carefully consider the habitat needs of piñon-juniper wildlife when planning thinning, burning, and other piñon-juniper management and to incorporate research and monitoring as a regular component of all management actions.

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# Piñon-Juniper Is Not Just One Vegetation Type: Key Variation in Composition, Structure, and Disturbance Dynamics, With Implications for Wildlife and Ecosystem Management

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**KEYWORDS**—piñon-juniper savannas, persistent piñon-juniper woodlands, wooded shrublands, wildlife, fire, restoration, ecosystem management

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Piñon-juniper vegetation covers 100 million acres across the western United States, in areas of diverse topography, climate, and soils. Consequently, piñon-juniper vegetation varies greatly in structure and disturbance processes at both the stand and landscape scales. Until recently, however, this variation was not widely recognized, and piñon-juniper vegetation was interpreted and managed in much the same way throughout its range (Jacobs 2008a). Early studies were conducted primarily in the Great Basin and Mexican borderlands, where juniper expansion into grasslands and shrublands was a major ecological process and management concern. Results of these studies were then applied uncritically to piñon-juniper vegetation elsewhere, notably on the Colorado Plateau, where more recent research has revealed very different stand and landscape dynamics.

This later research resulted in a workshop in August, 2006, that brought together 15 scientists with piñon-juniper expertise from across the range of piñon and juniper in the western USA. These researchers summarized available data and understanding of these varied ecosystems and published their synthesis in three venues: an academically oriented, invited review paper

in the journal *Rangeland Ecology and Management* (Romme et al. 2009), and two shorter articles geared toward managers and policy-makers (Romme et al. 2007, 2008). Key findings from these three articles and recent related literature are summarized below.

## Variation in Tree Species Composition

Two piñon species and five juniper species are common components of piñon-juniper woodlands across the western USA (names from USDA Plants database, <https://plants.usda.gov/java/>). Local composition varies with geographic location, elevation, and synoptic climate (Jacobs 2008a). Utah juniper (*Juniperus osteosperma*) and two-needle pinyon, also called Colorado piñon (*Pinus edulis*), dominate woodlands in regions where precipitation is winter-dominated or bimodal, e.g., on the northern Colorado Plateau. Oneseed juniper (*J. monosperma*), alligator juniper (*J. deppeana*), and two-needle piñon dominate summer monsoonal regions, e.g., in central and southern Arizona and New Mexico. Rocky Mountain juniper (*J. scopulorum*) is common at higher elevations throughout most of the Rocky Mountain cordillera. Singleleaf pinyon (*P. monophylla*) and western juniper (*J. occidentalis*)

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In: Malcolm, Karl; Dykstra, Brian; Johnson, Kristine; Lightfoot, David; Muldavin, Esteban; Ramsey, Marikay. 2020. Piñon-juniper habitats: Status and management for wildlife. Proceedings RMRS-P-77. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 128 p.

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dominate the summer-dry country of the Great Basin and eastern Sierra Nevada. Two additional piñons and four additional junipers are found along the southern and eastern boundaries of the American Southwest (Jacobs 2008a).

### **Variation in Stand Structure and Dynamics**

The scientists at the 2006 workshop identified three fundamentally different kinds of piñon-juniper vegetation. This classification is based not on species composition, but on functional characteristics including typical stand structures, historical disturbance regimes, and temporal dynamics.

Piñon-juniper savannas are composed of trees within a grassland matrix, and are found in areas of summer-dominated precipitation, e.g., in southern and eastern Arizona and New Mexico, and southeastern Colorado. Tree densities probably were limited historically by periodic fires, but in the last century junipers have expanded extensively into former grasslands, especially on concave or depositional landforms, converting those grasslands to young savannas and woodlands in many places (e.g., Fuchs 2002, Jacobs 2008b, Poulos et al. 2009, Parker 2009, Margolis 2014).

Wooded shrublands are composed of trees within a shrub matrix, e.g., sagebrush (*Artemisia* spp.), Gambel or wavyleaf oak (*Quercus gambelii* or *Q. undulata*), mountain-mahogany (*Cercocarpus montanus*), and Utah serviceberry (*Amelanchier utahensis*). Tree density naturally waxes and wanes in response to decadal-scale variations in climate, fire, and insect activity. These woodlands are probably most prevalent in areas of winter-dominated or bimodal precipitation, e.g., in eastern Utah, western Colorado, and northwestern New Mexico.

Persistent piñon-juniper woodlands are found throughout the distribution of piñon-juniper woodlands in places that burn very infrequently, such as dry rocky upland sites, but especially on the northern Colorado Plateau where precipitation is winter-dominated or bimodal. Notably, these are not young woodlands created by recently expanding junipers and piñons, but are often ancient stands with centuries-old trees, snags, and fallen boles, and little or no evidence of fire. The paucity of fire in persistent piñon-juniper woodlands

apparently is due to a generally discontinuous fuel structure (Romme et al. 2009). Heavy grazing has reduced fine fuels in many piñon-juniper woodlands, but fuels tend to be discontinuous and fires infrequent in persistent woodlands even where grazing has been absent or minimal for nearly a century, e.g., Mesa Verde National Park (Floyd 2003).

This classification, recognizing three broad kinds of piñon-juniper woodlands, is obviously a huge simplification of an extensive and varied vegetation type. Local floristic composition can vary greatly within each of these three functional piñon-juniper types. For example, Romme and Jacobs (2015; table 4) identified three floristic plant associations within the persistent piñon-juniper type and one within the piñon-juniper savanna type at El Malpais National Monument, New Mexico. Nevertheless, a functional classification of this kind highlights some of the important differences in composition, structure, and ecological dynamics to be found within piñon-juniper woodlands across the West and provides a first step in characterizing any specific woodland stand or landscape.

### **Some Applications to Wildlife and Ecosystem Management**

Management opportunities and constraints are greatly influenced by the kind of piñon-juniper (PJ) ecosystem being considered. For example, the greatest departure from historical conditions is likely to be seen in PJ savannas, especially where oneseed juniper has converted former grasslands into new savannas and woodlands during the past century. Persistent PJ woodlands, on the other hand, generally have changed the least from historical conditions. Persistent PJ woodlands are also most likely to support a unique suite of wildlife species that are closely linked with and dependent upon this particular kind of habitat, e.g., Pinyon Jay (*Gymnorhinus cyanocephalus*) and Juniper Titmouse (*Parus inornatus*). Thinning or burning these woodlands is likely to reduce habitat quality for these PJ specialists.

Reducing tree density to improve forage production for livestock, or habitat for grassland-associated wildlife species, will often be most successful in PJ savannas, where soils and climate are inherently favorable for grasses, and least successful in persistent PJ woodlands where rocky soils favor trees and shrubs over grasses.

Prescribed fire can be applied effectively in many PJ savannas for a number of objectives, but prescribed fires in persistent PJ woodlands notoriously fail to spread through surface fuels, or spread uncontrollably into the canopy. In wooded shrublands dominated by shrub species that resprout after fire, e.g., Gambel oak (*Quercus gambelii*) or mountain-mahogany (*Cercocarpus montanus*), prescribed fire can be an effective way to improve forage for deer (*Odocoileus hemionus*) and elk (*Cervus elaphus*). However, if the dominant shrubs tend to be killed by fire without resprouting, e.g., big sagebrush (*Artemisia tridentata*), then prescribed burning may produce little improvement in forage but promote an unwanted increase in nonnative plants such as cheatgrass (*Bromus tectorum*).

Piñon-juniper vegetation is managed for a wide range of objectives, and in some cases the same kind of treatment can achieve several different objectives. But this is not always true. For example, the term *restoration* is sometimes used loosely to bundle multiple objectives, but restoration has a precise definition as “... an intentional activity that initiates or accelerates the recovery of an ecosystem with respect to its health, integrity and sustainability ... attempts to return an ecosystem to its historic trajectory,” (Society for Ecological Restoration, <http://www.ser.org/>). Genuine restoration often does not achieve some of the other objectives we have for PJ vegetation, and accomplishing some of those other objectives may actually move an ecosystem farther from its historical condition.

Floyd and Romme (2012) explore some of the conceptual issues in restoration of PJ ecosystems. Restoration may focus on structural parameters, e.g., canopy density and understory composition, or on *functional aspects*, e.g., fire frequency and effects. A structural restoration can return a site to its condition at a particular time in the past, but that may be only one point within the natural range of variability on that site, and it may not persist without repeated treatment. Therefore, a functional or process-oriented restoration may be more sustainable, by allowing natural dynamics to continually reshape the ecosystem within historical bounds. Restoration may be passive or active. *Passive* restoration merely removes any chronic stressors from an ecosystem that is generally in good condition, e.g., minimizing impacts of grazing or recreation,

without directly altering structure or process by actions like thinning or planting. *Active* restoration directly alters the ecosystem to restore parameters that have been lost and cannot be recovered without intervention.

The specific type of restoration needed depends on the current condition and history of a PJ ecosystem. Floyd and Romme (2012) suggest some very general guidelines and review specific examples of restoration experience in different kinds of PJ. If a PJ ecosystem remains within or very close to its historical range of variability (Weins et al. 2012), then passive restoration may be all that is needed to maintain that condition; active treatments like canopy thinning could actually degrade stands by disturbing the soil and encouraging non-native plant establishment. An example requiring only passive restoration is the old-growth PJ woodlands on Mesa Verde (Floyd 2003). In other cases the overstory may be in good shape—within the historical range of tree density, composition, and cover—but the understory has been degraded by excessive grazing or recreational impacts.

Piñon-juniper woodlands on the Kaiparowitts Plateau fit this description: native graminoids and herbs were greatly reduced by more than a century of summer grazing, and nonnative cheatgrass has proliferated in place of the natives, even though the overstory remains intact (Floyd et al. 2008). Restoration of this ecosystem requires both passive and active approaches: passive to reduce grazing intensity and active to reduce the cheatgrass and possibly to also plant seed of species native to the site, but the overstory needs no treatment.

In other places both the overstory and understory have been degraded by past land use, and active treatment is needed in both components of the ecosystem. Such a stand in Bandelier National Monument was restored by mechanically thinning the overly dense overstory and scattering the slash over the bare ground between retained tree canopies. Within 3-5 years, understory plant cover increased several-fold and run-off and sediment production were reduced by an order of magnitude (Jacobs 2015). The treated stand also was more resilient in the face of a severe drought, a wildfire, and a bark beetle outbreak that occurred a few years after the treatment.

A final class of PJ ecosystems that may be in special need of restoration is found in areas that have recently burned in high-severity wildfires. A large portion of the old-growth woodland on Mesa Verde burned between 1989 and 2008 (Floyd et al. 2004). Severe fires in this kind of ecosystem are natural, although infrequent, but postfire vegetation dynamics have been anything but natural. Nonnative annual and biennial plant species such as cheatgrass, muskthistle (*Carduus nutans*), and Canada thistle (*Cirsium arvense*) have expanded rapidly into the burned areas and are the dominant species in many places. Notably, the stands having understories of resprouting shrubs, e.g., Gambel oak or Utah serviceberry, have been more resistant to invasion by nonnative plants than have the stands with few sprouting shrubs. This may be because the sprouting shrubs compete strongly with the nonnatives. Active restoration by means of planting native grass seeds in the burned areas has substantially reduced the abundance of the nonnatives, although it has not eliminated them altogether (Floyd et al. 2006).

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# Environmental Drivers of Tree Mast Production in Piñon-Juniper-Oak Woodlands of Central New Mexico

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**ABSTRACT**—The mast dynamics of two-needle piñon pine (Pinaceae: *Pinus edulis*), oneseed juniper (Cupressaceae: *Juniperus monosperma*), and Sonoran scrub oak (Fagaceae: *Quercus turbinella*), were analyzed in relation to weather variables (precipitation, air/soil temperatures, relative humidity, vapor pressure deficit, wind) over different time frames (current year, 1- and 2-yr lags) at 6 sites over 15-19 years (1997–2015) within the Sevilleta National Wildlife Refuge, New Mexico, USA. Juniper mast production was positively correlated to current year winter-spring precipitation, combined with a negative correlation with current-year summer temperatures. Oak mast production was positively correlated only with current year winter/spring precipitation and was not affected by subsequent summer temperatures. Piñon pine mast production was positively correlated with 1-year-lagged total annual precipitation, and negatively correlated with both summer temperatures and 2-year lagged autumn temperatures. Increased precipitation led to a greater proportion of trees producing mast, as well as greater mast production per tree. A series of hypotheses concerning mast dynamics—resource matching, resource switching, cycling and delta-temperature—were evaluated. The observed relationships between weather variables and mast production indicate that future climate warming will likely reduce mast production in extant Southwestern woodland ecosystems and will contribute to geographic shifts in future woodland distributions.

**KEYWORDS**—climate change, mast, precipitation, temperature, weather

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In: Malcolm, Karl; Dykstra, Brian; Johnson, Kristine; Lightfoot, David; Muldavin, Esteban; Ramsey, Marikay. 2020. Piñon-juniper habitats: Status and management for wildlife. Proceedings RMRS-P-77. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 128 p.

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# Root-associated Microbes in Pinyon Pine: Implications for Management

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**ABSTRACT**—Colorado pinyon pine (*Pinus edulis*) forms mutualistic associations with root-associated ectomycorrhizal (EM) fungi which confer improved fitness under stressful conditions and thus may be critical to the success of this species in response to global climate change. We review studies of the EM fungal associations of *P. edulis* with an emphasis on the literature focused on drought and associated *P. edulis* mortality. We also describe the importance of other soil microbes including nonmycorrhizal fungi and bacteria, and present data describing dominant soil microbial taxa associated with *P. edulis*. Finally, we discuss the relevance of the patterns observed in these studies to the management of pinyon-juniper (*Pinus* spp.-*Juniperus* spp.) ecosystems. We conclude that *P. edulis* reestablishment or restoration following drought-induced mortality may be limited in some areas because of reduced availability of EM fungal inoculum. Genetically based differences in *P. edulis* drought tolerance are also associated with differences in EM fungal species composition. Given that the diversity of mycorrhizal fungi can be directly tied to host plant performance, loss of EM fungal diversity after differential *P. edulis* mortality may negatively affect recruitment in natural populations or in restoration efforts. Yet drought-tolerant *P. edulis* genotypes and their associated EM fungi may be critical to survival of both host and fungus during ongoing drought.

**KEYWORDS**—climate change, drought, ectomycorrhizal fungi, *Pinus edulis*, pinyon pine, rhizosphere

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## INTRODUCTION

Plants and fungi frequently form intimate associations with one another that can be mutually beneficial, neutral, or parasitic. Among the most widespread of these associations are the mycorrhizas, mutualistic associations between soil fungi and the fine roots of plants in which plants exchange carbon fixed through photosynthesis for soil resources (Smith and Read 2008). In addition to promoting plant survival and growth, mycorrhizal fungi can influence larger scale processes such as plant species richness (Teste et al. 2017; van der Heijden et al. 1998), the abundance and species composition of soil organisms (Smith and Read 2008), nutrient cycling (Phillips et al. 2013), and soil stability

(Wilson et al. 2009). There are several types of mycorrhizal associations (Johnson and Gehring 2007). The most widespread are the arbuscular mycorrhizas (AM), in which the fungi involved are limited in diversity (~300 species in a single phylum), and the plants are highly diverse (~85 percent of land plants) and widespread (Wang and Qiu 2006). Arbuscular mycorrhizas generally produce hyphae without septa. They also produce characteristic exchange structures (arbuscules) and storage structures (vesicles) that occupy the cortex of host roots and must be stained to be observed.

Ectomycorrhizal (EM) fungi form a diagnostic morphological structure within roots, the Hartig net, but

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also envelope root tips within a sheath or mantle of fungal hyphae. Ectomycorrhizas are also widespread because they dominate on plants that occur in many temperate and boreal forest ecosystems. In this case, fewer plant taxa are involved with thousands of species of fungi from two major phyla, the Basidiomycota and the Ascomycota (Tedersoo et al. 2008, 2011). Prominent EM host trees include pine (*Pinus* spp.) and spruce (*Picea* spp.) (Wang and Qiu 2006), which have broad distributions across the forest ecosystems of the northern hemisphere and are functionally obligately dependent on EM fungal associations (Molina et al. 1992). Importantly, members of only a few plant families are able to form associations with both AM and EM fungi; this specificity can reduce mycorrhiza formation on plants in some settings (Chen et al. 2000).

Numerous studies demonstrate the importance of mycorrhizal fungi to their host plants, but there also is significant evidence of functional variation among fungal species. For example, the species composition of EM fungi associated with plants varies with environmental variables such as precipitation (Karst et al. 2014), soil nitrogen (N) availability (Lilleskov et al. 2011), and disturbance such as fire (Hewitt et al. 2016; Kipfer et al. 2011). Similarly, species of EM fungi vary in their ability to access different forms of N (Smith and Read 2008) and to promote drought tolerance in their host plants (Peay et al. 2007). The AM fungi also vary in distribution and function (Kivlin et al. 2011); recent studies suggest that many AM fungal species are distributed across six continents (Davison et al. 2015). However, for both AM and EM fungi, it is important to understand not only whether inoculum is available at a site, but also whether that inoculum consists of suitable fungal species for a given environment. Associated fungal species composition can vary among both species (Molina et al. 1992) and genotypes of plant hosts (Lamit et al. 2016), and some fungal taxa demonstrate considerable host specificity (Molina et al. 1992), such as between members of the EM fungal family Suillaceae and members of the Pinaceae (Bruns et al. 2002).

Although many forested ecosystems contain plant species that form either AM or EM associations, pinyon-juniper (*Pinus* spp.-*Juniperus* spp.) woodlands are unusual in that the codominant trees that characterize the ecosystem form different types of mycorrhizal

associations, AM for juniper and EM for pinyon. In addition, the grasses and shrubs that form most of the remainder of the vegetation in pinyon-juniper woodlands form AM associations (Haskins and Gehring 2005). This distribution means that pinyons are often the only host for EM fungi across a substantial portion of their geographic range (Gehring et al. 2016). Colorado pinyon pine (*Pinus edulis*) has experienced significant mortality across the southwestern United States, with models projecting further contraction of its range (Cole et al. 2008; Rehfeldt et al. 2006). In this paper, we discuss what is known about the mycorrhizal associations of *P. edulis*, with particular emphasis on applications for management given its sensitivity to drought and its recent drought-induced mortality across much of the Southwest (Breshears et al. 2005; Garrity et al. 2013; Mueller et al. 2005a). We also describe research on other microbes associated with *P. edulis* and their potential importance to pinyon ecology.

## PINYONS, EM FUNGI, AND PLANT NEIGHBORS

Given that across much of the range of pinyon-juniper woodlands, pinyons are often the sole host for EM fungi, available EM inoculum and successful pinyon reestablishment could be limited following drought-induced mortality. Gehring et al. (2016) estimated that *P. edulis* is the only host for EM fungi across nearly 60 percent of its geographic range. At sites where junipers dominate, pinyons show reduced overall abundance of EM fungi, and seedlings are less likely to have any EM fungal associations when grown in soil from juniper-dominated areas (Haskins and Gehring 2005). Pinyons experimentally isolated from neighboring junipers respond with a rapid doubling of fine-root biomass; EM fungi respond similarly, also doubling in abundance (Haskins and Gehring 2004). Allen et al. (2010) observed that the EM associations of *P. edulis* were more sensitive to N deposition than the AM associations of juniper. These results indicate that junipers and their associated soil microbial communities (including AM fungi) may influence the environment through competitive or antagonistic effects (or both) that render *P. edulis* less able to establish and survive.

A similar competitive release was seen when AM-associated shrubs, principally *Fallugia paradoxa*, were

removed from beneath mature pinyons in the field, resulting in increases in root biomass, needle length, and EM abundance (McHugh and Gehring 2006). Associations with this species of shrub help juvenile pinyons to survive and grow in stressful environments (Sthultz et al. 2007). At low-stress sites, this relationship becomes more competitive, with both aboveground and belowground factors contributing to the variation, though additional research is necessary to clarify mechanisms (Sthultz et al. 2007). The dynamics of these relationships are likely to change as climate continues to warm and dry, with increased likelihood that relationships between shrubs and pinyons will be beneficial when trees are young and detrimental as they age.

Research from other forested ecosystems also demonstrates the importance of EM plants to one another. A large-scale study of northern temperate forest trees showed that species that form EM associations are generally positively influenced by soils in which conspecifics had grown previously and negatively influenced by soil in which heterospecifics had grown previously. The formation of mycorrhizas in EM-dependent trees was more efficient when seedlings were grown in soil collected beneath a conspecific tree rather than beneath a heterospecific tree, and this difference was further reflected in the overall performance of the seedling as a function of biomass (Bennett et al. 2017). This result could be due to decreased or less-compatible EM fungal inoculum from heterospecific soil, or an increase in pathogen load, or both. Combined with previous studies of *P. edulis*, these results suggest that EM seedlings establishing in areas lacking conspecifics will perform poorly, potentially due to lack of EM inoculum or other complex differences in local soil communities. Inoculation with live soil or other sources of EM inoculum may remedy inoculum limitation.

An experiment conducted in *Pinus patula* demonstrated that soil collected from a *P. patula* plantation offered an effective source of EM fungal inoculum, as did mixed cultures of certain EM fungal species (Restrepo-Llano et al. 2014). Similarly, de Oliveira et al. (2006) isolated and cultured EM fungi from a loblolly pine (*P. taeda*) plantation which was then successfully used to inoculate greenhouse seedlings. It therefore may be possible to supplement soil with

ectomycorrhizal inoculum where pine restoration is desired following high-mortality events. Because culturing techniques for most EM fungal taxa associated with *P. edulis* have yet to be developed, inoculation with soil from nearby areas with living pinyons is more likely to provide inoculum-limited pinyons with beneficial EM fungal taxa than inoculation with soil fungal cultures or commercially produced generalist EM fungi that may not be native to the area (Schwartz et al. 2006).

In cases where *P. edulis* is extirpated from an area due to drought or other disturbance, reestablishment may occur with EM fungal inoculum that persists in the soil. However, reestablishment requires that EM fungal communities persist in the absence of a host plant. Little is known about how long existing EM fungal inoculum remains viable or which taxa persist after disturbance events. Long-term experimental data on the ability of dormant EM fungal spores to colonize bishop pine (*P. muricata*) yield mixed results. Several *Rhizopogon* species (Basidiomycota) show an improved potential to colonize roots over time (Bruns et al. 2009), whereas other EM fungal taxa were undetectable after several years (Nguyen et al. 2012).

Though these data indicate that at least some EM fungal propagules will persist and facilitate growth of pines over time, they also imply an eventual loss of EM fungal diversity in the absence of the EM host. This may help to explain the result of Haskins and Gehring (2005), who observed decreased colonization by EM fungi among pinyon seedlings grown in soil collected from juniper-dominated stands. This observation implies a density-dependent relationship as stands previously shared by pinyons and junipers shift toward a juniper-only woodland following drought-induced pinyon mortality events. Further, the effect of persistent juniper dominance may result in an altered soil microbial community that specifically favors the growth of junipers over pinyons as soil communities become locally adapted to the juniper-dominated regime (Revillini et al. 2016).

Besides examining roots directly to assess mycorrhizal colonization and EM fungal species composition, another approach to describing EM fungal communities (or other microbes) found in pinyon-juniper woodlands is the use of next-generation DNA sequencing

technologies. These techniques can provide data on the relative abundance of all fungi or bacteria present in a soil sample (e.g., Caporaso et al. [2012]). In a preliminary survey, we used this technology at Sunset Crater National Monument (SCNM) in northern Arizona, which revealed differences in EM and AM fungal composition between root-associated soil collected beneath live, adult *P. edulis* ( $n = 31$ ), and interspace zones devoid of vegetation for a diameter of 6 m ( $n = 26$ , collected in four transects within the population of adult trees). Taxa were identified from polymerase chain reaction amplicon sequencing of the ITS2 rDNA gene for each sample using fungal-specific primers (Taylor et al. 2016) on an Illumina® MiSeq Desktop Sequencer (Illumina Inc., San Diego, California). Paired t-tests in which the data were matched by sample were used for comparing relative abundances of AM and EM taxa within each group (interspace or adult trees). These tests revealed no

difference between AM and EM abundances in the interspace soil ( $t = 1.4488$ ,  $p = 0.1536$ ), and a strong difference between AM and EM abundances in soil from adult trees ( $t = 14.4659$ ,  $p < 0.0001$ ).

Unpaired t-tests were used to compare AM or EM taxa abundances separately between each group, revealing that differences exist for both AM ( $t = 4.3331$ ,  $p < 0.0001$ ) and EM ( $t = 3.8931$ ,  $P = 0.0003$ ) fungal relative abundances. Pinyon-associated soil was rich with EM fungal symbionts at this site (12.1 percent relative abundance), consisting mostly of the genera *Geopora* and *Rhizopogon*, while there was a marked absence of AM fungi (0.8 percent relative abundance) (fig. 1). These EM taxa have been observed consistently at this site in conjunction with pinyon roots (Gehring et al. 1998, 2014b; Gordon and Gehring 2011; Sthultz et al. 2009b). EM taxa were also abundant in the interspace soils (8.1 percent relative

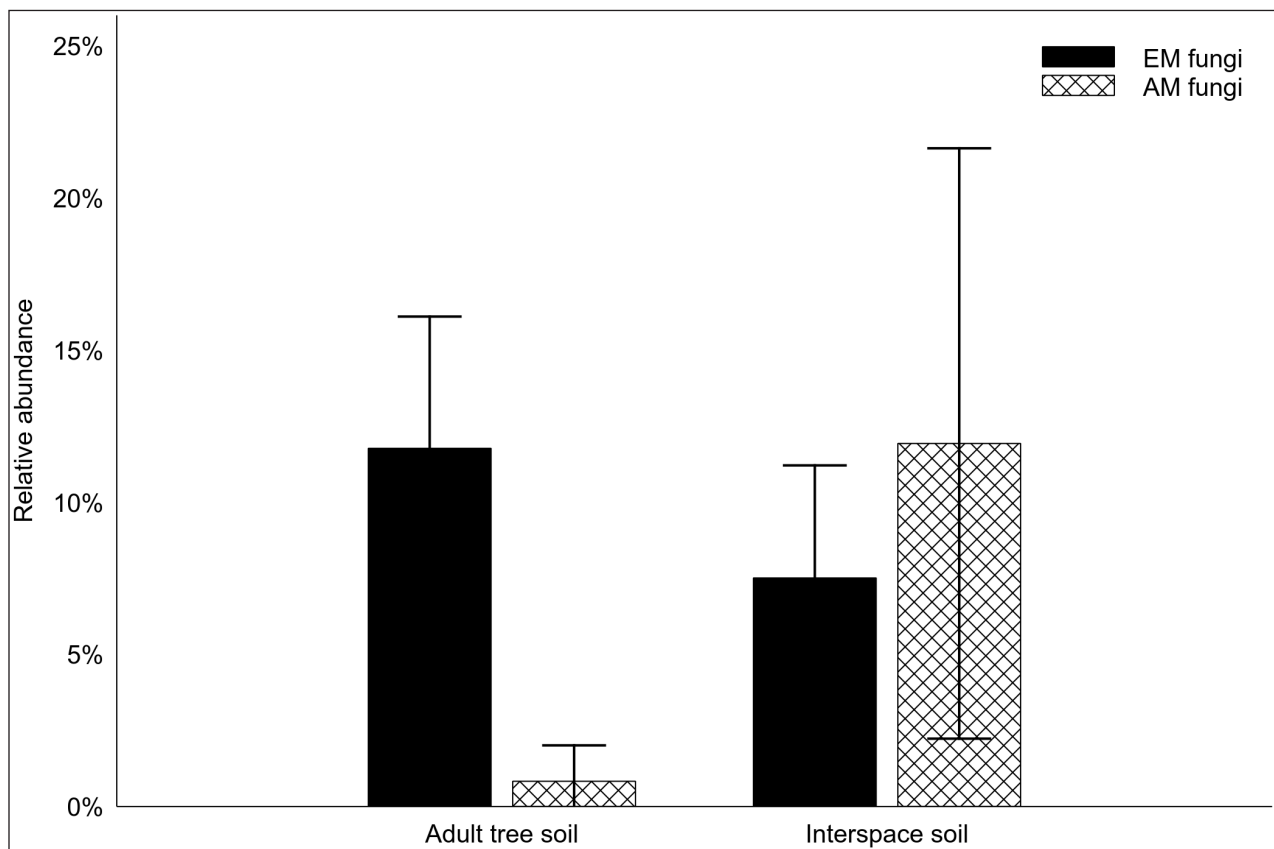


**Figure 1**—Average relative abundances of total ectomycorrhizal (EM) or arbuscular mycorrhizal (AM) fungal taxa observed by DNA sequencing of root-associated soil associated with adult *Pinus edulis* ( $n = 31$ ) or from vegetation-free interspaces ( $n = 26$ ) at Sunset Crater National Monument, Arizona. Error bars represent  $\pm$  one standard deviation. *Pinus edulis* hosted a significantly larger fraction of EM than AM fungi ( $p < 0.0001$ ), in contrast with interspaces ( $p = 0.1536$ ). *Pinus edulis* hosted a larger fraction of EM fungi than observed in interspaces ( $p = 0.0003$ ) while the opposite was true for AM fungi ( $p < 0.0001$ ).

abundance), locations where it would not have been possible to observe them in the absence of pinyon roots without the use of this DNA sequencing technique. We note the presence of both AM (12.4 percent relative abundance) and EM fungal taxa in the inter-space soils, which implies that mycorrhizal symbionts are readily available for establishing seedlings of AM- and EM-associated plant species in this ecosystem. Though these data represent just a single time point for a single site, they serve as a useful illustration of the challenges that EM hosts may face during seedling recruitment in heterospecific soils.

Across approximately 40 percent of its geographic range, *P. edulis* occurs with other hosts for EM fungi, such as ponderosa pine (*P. ponderosa*) and a variety of species of oaks (*Quercus* spp.) (Gehring et al. 2016). *Pinus edulis* and *P. ponderosa* habitats overlap near the upper elevational limit of *P. edulis*. When the rhizospheres of *P. edulis* and *P. ponderosa* overlapped, they shared 88 percent of the fungal taxa observed (Hubert

and Gehring 2008) (fig. 2). The EM taxa shared between *P. edulis* and *P. ponderosa* were dominated by what were then defined as unknown Pezizalean (Ascomycete) fungi (Hubert and Gehring 2008), and are probably what we now recognize as a group of EM fungi in the genus *Geopora* of particular significance to arid *P. edulis* ecosystems (Flores-Renteria et al. 2014; Gordon and Gehring 2011). Such zones of host-species overlap may provide important refugia for EM fungal taxa important to *P. edulis*.

Less is known about overlap with other EM hosts such as oaks. To our knowledge, the EM community composition of *P. edulis* and *Quercus* species has not been examined in areas where their distributions overlap. However, data from other systems demonstrate the importance of host-overlap zones for maintenance of EM taxonomic diversity. For example, Horton et al. (1999) observed that in mixed stands of Douglas-fir (*Pseudotsuga menziesii*) and manzanita (*Arctostaphylos* spp.), most trees were colonized by

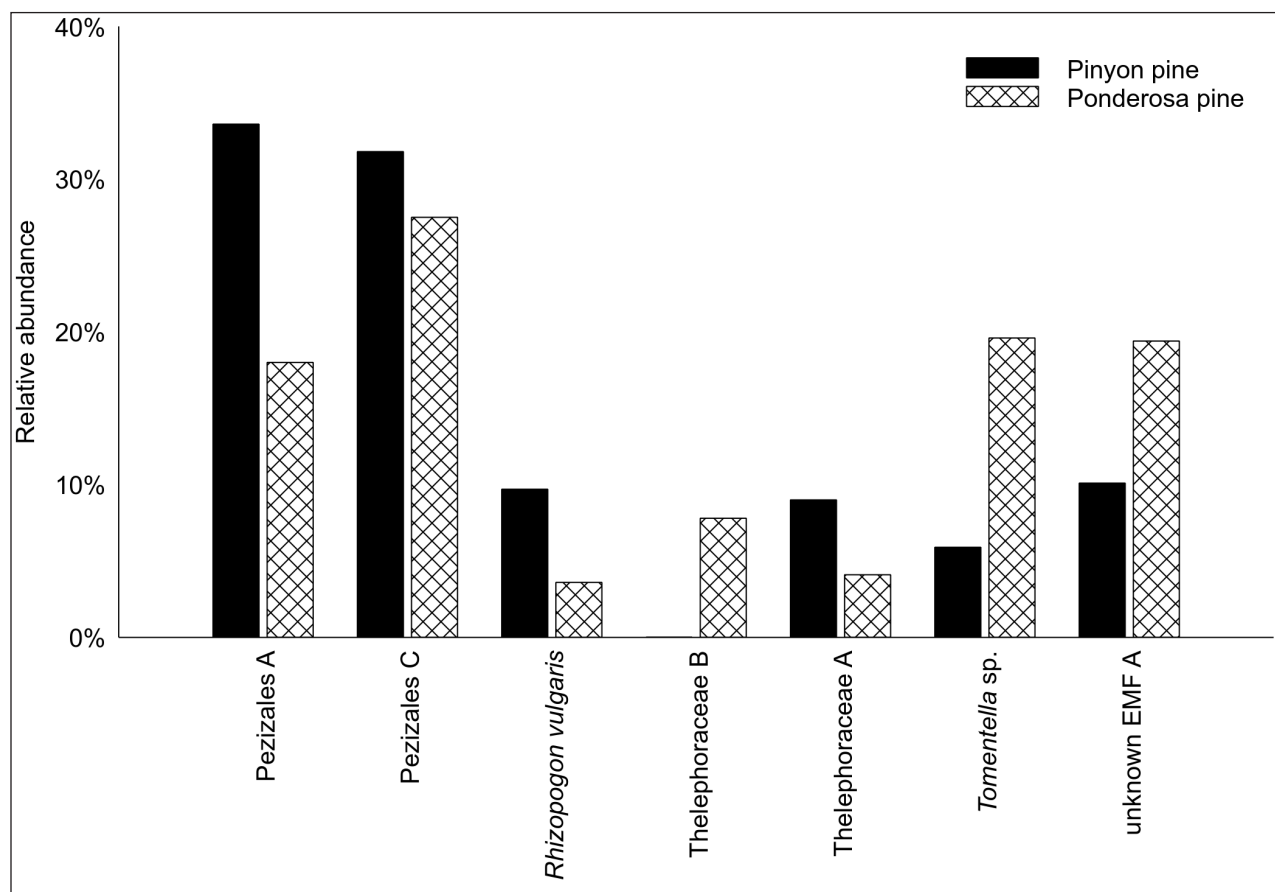


**Figure 2**—Average relative abundance of ectomycorrhizal fungal taxa observed on roots of *P. edulis* and *P. ponderosa* when the rooting zones of the two species overlapped. The two pine species shared six of seven EM fungal taxa with similar relative abundance (redrawn from Hubert and Gehring [2008]).

EM fungal taxa which regularly colonize the roots of either host, illustrating an ecological linkage between gymnosperm and angiosperm populations. Likewise, Cullings et al. (2000) examined a mixed forest of Engelmann spruce (*Picea engelmannii*) and lodgepole pine (*P. contorta*), and discovered that 95 percent of the observed EM fungi colonized both hosts. These results demonstrate little evidence for host specificity in their respective systems and imply that the presence of one host tree species may have benefits for maintaining EM fungal diversity that could benefit other tree species dependent on EM fungi.

### HOST GENETICS, DROUGHT TOLERANCE, AND EM FUNGI

*Pinus edulis* is especially vulnerable to drought stress relative to other regional forest tree species, and experiences high rates of mortality under prolonged periods of drought (Gitlin et al. 2006). This vulnerability can be magnified by the effects of climate change, which are predicted to include a continued increase in mean average temperature for the present range of *P. edulis* (Adams et al. 2009) and a greater likelihood of more frequent and severe drought events across the southwestern United States (Seager et al. 2007). Indeed, projections of future plant distribution based on climate modeling data suggest the current range of *P. edulis* may no longer be suitable habitat for these trees by the end of this century (Rehfeldt et al. 2006). An extreme drought event from 2002 to 2003 resulted in high rates of pinyon mortality across the range of the species (Breshears et al. 2005). Later observations revealed that survival or death of *P. edulis* during this event was correlated with lower precipitation and vapor pressure levels (Clifford et al. 2013).

*Pinus edulis* mortality can be highly variable among stands; dead and surviving trees may occur side by side, suggesting individual variation in drought tolerance (Mueller et al. 2005a; Ogle et al. 2000). Sthultz et al. (2009a) showed that at several sites near SCNM, genetic differences across the population correlated with survival or demise during the 2002–2003 drought, indicative of a genetic basis for drought tolerance or intolerance. At these sites, drought-intolerant trees, initially present as a high proportion of the population, were reduced to a similar proportion to that of

drought-tolerant trees, even as the overall population density contracted (Sthultz et al. 2009a).

The genetic basis of drought tolerance and intolerance in these trees was distinguishable because of variation in susceptibility to a stem-boring moth whose chronic herbivory on mature trees alters their architecture. Susceptible trees chronically attacked by the moth have a prostrate, shrub-like architecture in contrast to the normal upright architecture of moth-resistant trees (Whitham and Mopper 1985). Subsequent studies demonstrated a genetic basis to moth resistance or susceptibility (Mopper et al. 1991). Sthultz et al. (2009a) observed that moth-resistant trees were approximately three times more likely than moth-susceptible trees to perish under drought stress, indicating that this phenotypic differentiation is indicative of drought tolerance in the moth-susceptible group. Seedlings of moth-susceptible trees also survived longer under drought conditions experimentally imposed in a greenhouse, demonstrating the genetic basis of drought tolerance (Sthultz et al. 2009a). Such phenotypically dimorphic populations present important opportunities for study as not every population will provide such a simple diagnostic tool for examining genetic effects on ecological interactions, and resources for studying the genetic composition of pinyon pine populations remain limited (Krohn et al. 2013; Lesser et al. 2012).

Sthultz et al. (2009b) observed that the differential phenotypes of the pinyon population previously described associate with specific communities of EM fungi. Specifically, the drought-tolerant group hosted more Pezizalean EM fungi, including the genus *Geopora* (Ascomycota), and this pattern held whether data were from a relatively wet or dry year. Gehring et al. (2014a) explored the EM fungal communities of drought-tolerant and -intolerant trees over time and observed that drought-tolerant trees hosted a consistent community as the regional climate became more arid, whereas the drought-intolerant trees exhibited a significant shift in EM community during the same period. In removing shrubs from beneath mature *P. edulis*, a competitive release was observed in which the focal pinyon exhibited increased growth (Gehring et al. 2014a). This competitive release was more significant in the drought-tolerant trees, which maintained a consistent EM fungal community, than in the drought-intolerant trees, whose EM fungal community

also shifted following shrub removal. The improved performance of the drought-tolerant trees after shrub removal is most likely due to added benefits provided from the associated EM community, implying that the EM community commonly observed among drought-tolerant trees is composed of mutualists superior to those observed among drought-intolerant trees (Gehring et al. 2014b, 2017). These superior mutualists may confer upon drought-tolerant trees the opportunity to perform more consistently under variable climatic conditions, though members of the genus *Geopora* have increased in abundance with drought both within and among sites (Gehring et al. 2014b; Gordon and Gehring 2011), possibly benefiting drought-intolerant trees as well.

Given the evolutionary patterns of *P. edulis* mortality following recent drought, in which climatic pressure selected against drought-intolerant genotypes (Sthultz et al. 2009a), it is possible that drought-intolerant genotypes will be extirpated during subsequent drought events. A potential consequence of this differential mortality is the loss of EM fungal diversity across local or regional scales due to the differences in EM fungal communities among the different genotypes. It is unclear how the loss of *P. edulis* genetic diversity and EM fungal biodiversity may affect recruitment of seedlings in natural settings or in restoration efforts.

The idea that it may be important to maintain intra-specific diversity of either the host plant or the fungal symbiont is not a new one (Johnson et al. 2012), and may be particularly relevant to the *P. edulis* system. While exploring the effect of AM fungal diversity on plant communities, van der Heijden et al. (1998) observed that increased AM fungal species richness yielded greater hyphal mass in the soil, increased plant biomass, and increased plant species richness. These effects corresponded with a depletion of available phosphorus in the soil, implying that the increase in AM fungal hyphae provided an improved mechanism for foraging nutrients from the environment (van der Heijden et al. 1998). More recently, Teste et al. (2017) demonstrated the importance of diverse soil biota to maintaining taxonomic and functional diversity in Australian shrublands. While experimental data showed that different resource acquisition strategies by plants (e.g., EM-dependent or AM-dependent) resulted in variability in plant response to soil microbial content, simulations indicated that the presence of these

different strategies combined with their associated soil microbial communities contributed strongly to maintenance of plant species and functional diversity across the environment (Teste et al. 2017). Although comparable data for pinyon-juniper woodlands are not available, these studies speak to the importance of maintaining soil microbial diversity in order to benefit plant communities. Because *P. edulis* is considered a foundation species, negative environmental effects on this species may exert a disproportionate effect on many dependent organisms (Whitham et al. 2003).

## OTHER ROOT AND SOIL MICROBES

Many organisms besides mycorrhizal fungi inhabit the soil among plant roots, but our overall understanding of their species composition and interactions remains limited (Ingleby and Molina 1991). In addition to pine-associated EM fungal communities, non-EM fungi and bacteria are present and participate in ecological processes in the soil. Bacteria are most abundant by cell count, regardless of host type (Neal et al. 1964; Rouatt and Katznelson 1961), and represent the bulk of soil genetic diversity, which may contain thousands of genomes per gram of soil (Torsvik and Øvreås 2002). The contribution of soil bacteria to host tree performance was first observed by comparing the growth of *P. radiata* in fumigated and nonfumigated soils, though it was unclear at the time which microbial group was mediating this effect (Ridge and Theodorou 1972). Later, it was recognized that various bacteria may either benefit the host plant directly as plant growth-promoting rhizobacteria (PGPR) (Kloepper and Schroth 1978) or indirectly by promoting the interaction between the host plant and potential mycorrhizal associates as mycorrhizal helper bacteria (MHB) (Garbaye 1994). Archaea are less well studied than bacteria due to their relative scarcity and a lack of available tools for examining these organisms. However, recent work indicates that although they represent just a small percentage of soil microbial communities, they may exert a powerful influence on nutrient-cycling processes such as ammonia oxidation (Leininger et al. 2006).

Nonmycorrhizal fungi represent a class of potent decomposers which release the bulk of present but unavailable nutrients into the ecosystem (Schlesinger and Bernhardt 2013). Much of the matter consumed

during decomposition consists of various plant tissues, so these taxa rely on the plant community even as they make nutrients available to that community. The apparent relationship between plants and microbial taxa gets even more complicated when we consider that, despite the well-known symbiotic mutualisms such as mycorrhizal associations in such communities, microbes and plants effectively compete for available nutrients (Harte and Kinzig 1993), implying a relationship that is simultaneously competitive and mutualistic. It is likely that the relationships among plant and microbial community members, combined with the abiotic characteristics of the surrounding soil, are the main influences that define the community structure in plant ecosystems.

Besides EM fungi, heterotrophic fungi and bacteria have been examined in pinyon-associated soils (Kuske et al. 2003). Patterns were observed with differences in host genetics, tree age, and soil type for drought-tolerant and -intolerant *P. edulis*. Tree age consistently

affected bacterial content with older (ca. 150 years old) trees hosting about three times that of younger (ca. 60 years old) trees, whereas fungi do not correlate with age (Kuske et al. 2003). These results demonstrate that bacterial communities may become denser as the trees age, which could improve survival beyond a certain age if those communities possess adequate levels of PGPR or MHB taxa. Conversely, because some soil bacteria may be pathogens (Revillini et al. 2016), the host may instead require more consistent associations with EM fungi to better survive increased antagonistic pressure through the protective effects of EM (Laliberté et al. 2015).

DNA sequencing data that we have collected for the microbial content of pinyon-associated soils describe the nonmycorrhizal fungi (based on ITS2 rDNA gene sequencing) (fig. 3) and bacteria (based on 16S v4 rDNA gene sequencing) (fig. 4) in finer detail than previously possible. Importantly, the largest observed

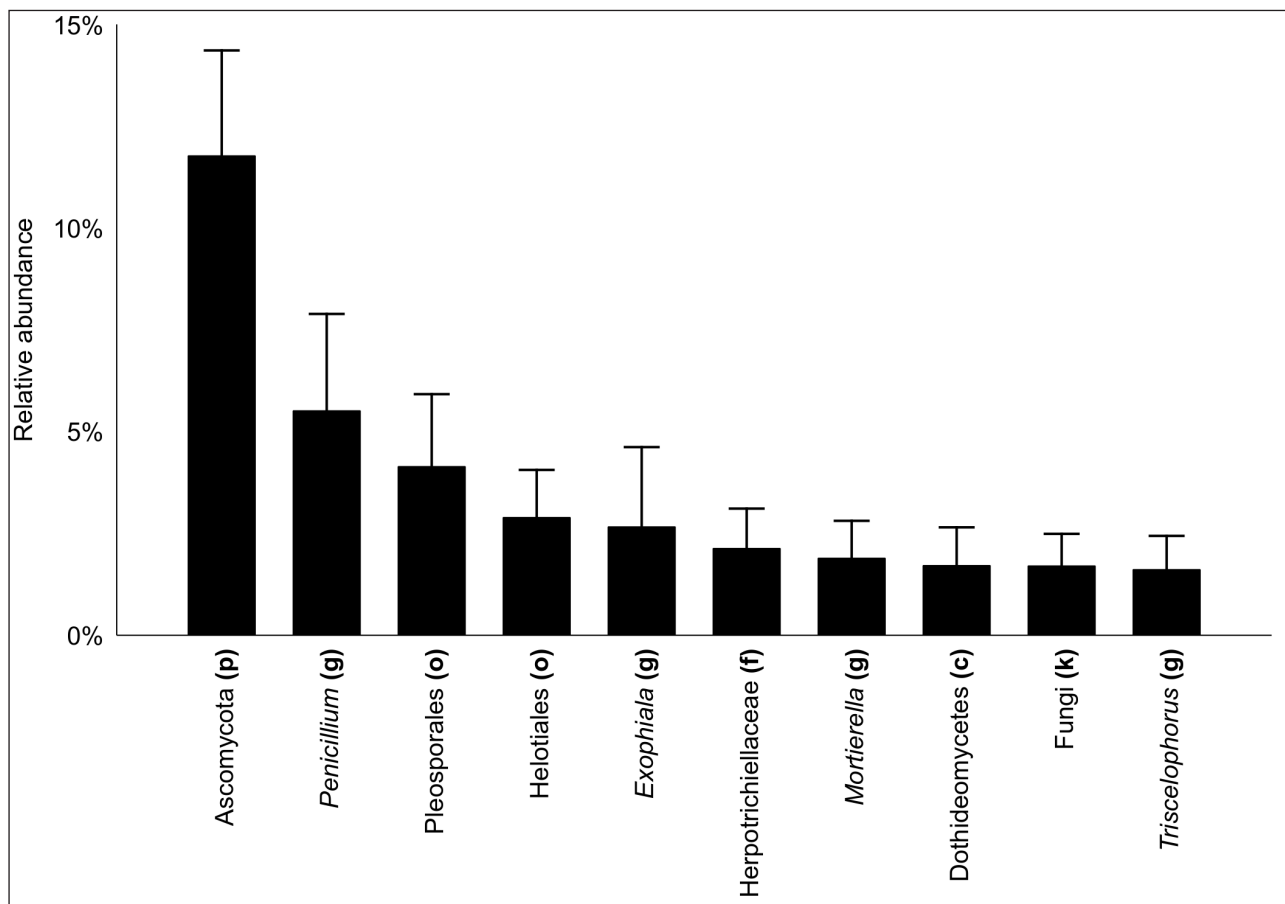


**Figure 3**—Average relative abundance of nonmycorrhizal fungal taxa observed by DNA sequencing of soil from roots of *P. edulis*. Data are summarized to the deepest confident taxonomic designation for each observation. Hierarchical taxonomic depth for each classification is indicated in bold parentheses (k = kingdom, p = phylum, c = class, o = order, f = family, g = genus). Error bars represent +/- one standard deviation.

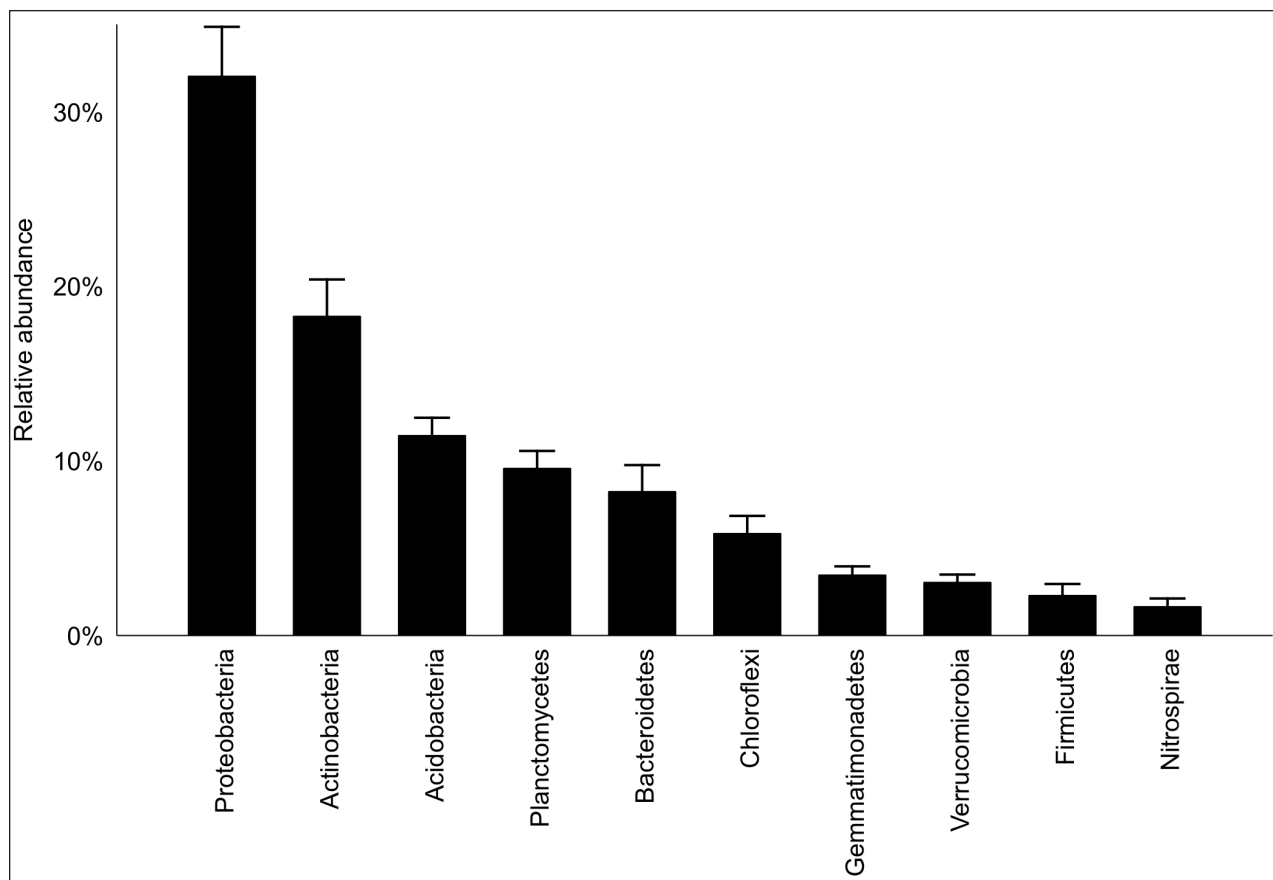


**Figure 4**—Average relative abundance of bacterial taxa observed by DNA sequencing of soil from roots of *P. edulis*. Data are summarized to the phylum level for all observations. Error bars represent +/- one standard deviation.

component of the nonmycorrhizal fungal community includes undescribed taxa in the phylum Ascomycota, highlighting the need for continued basic research describing the biodiversity of fungi in the environment. High proportions of fungi from the genus *Penicillium* indicate ample presence of saprotrophs (Kirk et al. 2008), as well as potential pathogen pressure as various species of *Penicillium* may also cause disease in plants (Ballester et al. 2015; Louw and Korsten 2013). Intriguing is the considerable proportion of fungi from the order Pleosporales, taxa that generally belong to the less well understood class of plant-symbiotic fungi known as dark septate endophytes (DSE) (Jumpponen and Trappe 1998). These fungi may turn out to be important players, as a meta-analysis revealed that associations by a variety of plant species with DSE generally confer an increased growth response in a host plant (Kivlin et al. 2013). Identification of soil bacteria using short DNA sequences generally fails to provide taxonomic information deeper than family, and we present only phylum-level classifications here (fig. 4).

In terms of the dominant phyla, these data are generally consistent with previous sequencing-based analyses from other forest soil bacteria (Roesch et al. 2007).

Numerous studies have implicated specific nonmycorrhizal taxa as either direct or indirect facilitators, including in various *Pinus* systems (e.g., Kataoka et al. [2009]). While studying the grass *Brassica rapa*, Lau and Lennon (2011) observed that more complex soil microbial communities facilitated an increase in plant biomass, leaf number, and flower count, due in part to the bacterial community composition. They later found that soil microbial communities adapted to environmental conditions, offering an alternative adaptive strategy to the host plant, which can utilize the benefits of a well-adapted soil microbial community as a proxy for evolutionary adaptation in order to better survive under stressful conditions (Lau and Lennon 2012). Such effects emphasize that beneficial soil microbes include those besides mycorrhizal fungi, and that these microbes can also have a profound impact on their plant hosts.

## MANAGEMENT IMPLICATIONS

At sites where *P. edulis* has been lost due to drought and EM fungal inoculum may be limited, provision of EM fungal inoculum during restoration may be necessary. Although there are few data on the direct effect of fire on pinyon EM fungi, a recent review showed that fire and drought have similar consequences for EM fungal populations and communities, particularly when they represent a loss of pines from the landscape (Karst et al. 2014). Because pines may exhibit specificity for certain EM taxa and vice versa (Bruns et al. 2002), commercially available EM fungal inoculum may be ineffective. A better option may be to grow plants in soil inoculated with live soil from surviving pinyons. Data on variation in drought tolerance among *P. edulis* genotypes (Sthultz et al. 2009a) indicate that it may also be important to manage pinyon ecosystems to maximize diversity of individual trees, which would be expected to translate into improved genetic diversity. In turn, greater genetic diversity of the *P. edulis* host is likely to contribute to maintenance of a more complete resource of belowground EM fungal propagules which will facilitate recruitment of new seedlings in the future as well as improve survival during restoration or conservation projects.

A feasible strategy for restoration of *P. edulis* in drought-stressed areas could make use of drought-tolerant genotypes, at least initially. These trees would better survive additional drought stress (Sthultz et al. 2009a), and may help to establish an adequate reservoir of EM fungal inoculum to eventually expand the genetic diversity of the host in order to further improve the resident EM inoculum potential. However, this strategy may have additional challenges. For instance, the drought-tolerant genotypes at SCNM often produce significantly fewer female cones (Mueller et al. 2005b), implying that collection of seeds from the desired genotypes could be more difficult. However, annual application of the systemic insecticide Cygon (Drexel Chemical Co., Memphis, Tennessee) is sufficient to control insect herbivory such that insect damage on drought-tolerant trees becomes equivalent to that observed on drought-intolerant trees, and cone production is similarly restored after just 4 years of treatment (Mueller et al. 2005b). Not all drought-tolerant genotypes will share the same ecological pressures as those observed at SCNM that result in easily identifiable drought-tolerant genotypes. Adequate study of individual populations at different sites will be

required to identify trees that possess drought-tolerant traits, and additional tools will need to be developed to better understand the genetics behind such traits.

When managing land for wildlife, land management agencies are often concerned with maintaining healthy plant populations that serve as food and shelter for game and nongame animals. The studies and data presented here suggest that soil micro-organisms contribute in important ways to plant and soil health. Of particular importance are the mycorrhizal fungal communities on which plant communities and their dependent organisms rely for efficient nutrient extraction from the soil (Smith and Read 2008) and for efficient deposition and storage of belowground carbon (Treseder and Holden 2013). For a recent, comprehensive review of how management practices relate to the management of mycorrhizal fungi, see Tomao et al. (2017). It is important to recognize that this review, which focuses on silvicultural practice, relates more directly to forest stands of limited composition and relatively uniform age. The authors specifically highlight the need for more research into the growth of mushrooms (which generally constitute available EM fungi in forest ecosystems) under forest conditions more typical of pinyon-juniper woodlands.

Another review by Karst et al. (2014) describes findings directly relevant to EM fungi and pines in general with respect to fire, drought, and herbivory, but again concludes by highlighting the many gaps in our understanding of the dynamics of EM fungal communities and their hosts. Based on the data available, we suggest that land managers regard effects of stand-reducing events (e.g., fire, drought, insect infestation) in pinyon pine and EM fungi similarly. Such reductions in host populations alter EM abundance, species richness, or community composition, thus decreasing the resiliency of the landscape to recover, at least temporarily. However, though fire will have an indiscriminate effect across a population, drought or insect infestation is likely to target certain host plant genotypes with greater consequences than others, reducing available genetic diversity of forest trees, and in turn, altering the composition of available EM fungi across an affected area. Including soil micro-organisms, such as mycorrhizal fungi that link aboveground and belowground portions of ecosystems, in future research and management of pinyon-juniper woodlands could improve outcomes for host plants, soils, and associated wildlife. Given these

challenges, we implore forest management agencies to direct available funding specifically to exploration of EM fungal communities in both healthy and affected *P. edulis* ecosystems.

## CONCLUSIONS

Associations between plants and microbes are increasingly recognized as key to plant performance and ecosystem function. The studies we have described here demonstrate the importance and complexity of microbial associations in pinyon-juniper woodlands, with an emphasis on mycorrhizal fungi. The mycorrhizal associations of pinyon-juniper woodlands are unique in ways that may affect their management, including the potential for EM fungal inoculum limitation following *P. edulis* mortality and the tight association between pinyon genotype and a genus of poorly described EM fungi. As studies of microbial communities remain limited spatially and temporally, much remains to be learned, though several recent studies have begun to explore the geographic distribution of mycorrhizal fungi on broad scales (Davison et al. 2015; Kivlin et al. 2011; Opik et al. 2010; Swaty et al. 2016; Tedersoo et al. 2011). Newer, culture-independent DNA sequencing methods are making characterization of microbial communities more rigorous and less costly (e.g., Caporaso et al. [2012]). A host of emerging methods are also elucidating the complexities of microbial function. These techniques may improve our understanding of the drivers of pinyon-juniper ecosystem function and better inform management of these ecosystems in an era of rapid environmental change.

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# Stand Structure and Drought Alter Tree Mortality Risk in Nevada's Pinyon-Juniper Woodlands

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**KEYWORDS**—piñon, stand density, Great Basin, forest mortality, forest inventory, semi-arid woodlands, aridity, competition

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Severe droughts in the southwestern USA in 2002–2003 revealed that pinyon pine and juniper trees are susceptible to widespread mortality caused by severe water deficits and drought-facilitated insect outbreaks (Breshears et al., 2005; Clifford et al., 2008). Despite increased interest in forest susceptibility to drought, the influence of stand structure on the drought resilience of pinyon-juniper woodlands remains uncertain (Meddens et al., 2015). Stand structure is of particular interest because it is readily manipulated by silvicultural practices such as thinning. Page (2008) suggests, for example, thinning pinyon-juniper woodlands to between 5 and 25 percent of maximum stand density index (SDI) in order to reduce fuel loads and increase understory vegetation. If trees are more susceptible to drought in denser stands, thinning may also be a viable strategy to increase resilience to drought.

We measured 98 plots in central Nevada pinyon-juniper woodlands before and after a severe multi-year drought in order to assess the effect of stand density on drought-related tree mortality and canopy dieback (Flake, 2016). The stands are dominated by singleleaf pinyon pine (*Pinus monophylla*) and Utah juniper (*Juniperus osteosperma*) with a small component of curl-leaf mountain mahogany (*Cercocarpus ledifolius*). Each plot measured 0.1 ha in area, and the plot network spanned a wide range of elevations (1820 m.a.s.l. to 2360 m.a.s.l), mean annual precipitation (230 mm to 414 mm), and soil properties (soil depth 7.4 to 43

cm). Plots were inventoried in 2005 (Greenwood and Weisberg, 2008, 2009) and resampled in 2015. For each tree, we estimated the amount of its canopy that was dead (dead branches, brown needles, or defoliated) in both 2005 and 2015, allowing us to calculate canopy loss over the study period.

We modeled tree canopy loss and the probability of stem mortality using generalized linear mixed-effects models. We first estimated the effects of environmental variables such as temperature, precipitation, vapor pressure deficit, climatic water deficit (Stephenson, 1998), Forest Drought Stress Index (Williams et al., 2013), soil depth, and soil available water capacity. We then added variables reflecting stand structure and tree attributes, including tree diameter and height; the basal area of neighboring trees within 4-m of the focal tree (*4-m BA*), the distance which was most strongly correlated with canopy dieback; the distance to the nearest neighboring tree (*ENN Dist*); and the ratio of the basal area of the nearest neighbor to that of the focal tree (*Neighbor Ratio*). We fit separate models for pinyon canopy dieback, juniper canopy dieback, and pinyon stem mortality.

On average, pinyons lost  $23.7 \pm 0.58$  percent of their canopy and junipers only  $10.8 \pm 0.46$  percent. In our study area, 10.9 percent of the pinyon pines but only 0.6 percent of junipers had died over the past decade. This mortality was driven largely by

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soil characteristics, with greater mortality on deeper soils, and by site aridity, with the greatest mortality at sites with the greatest climatic water deficit and vapor pressure deficits, but was mediated by stand structure. We found that the basal area of neighboring trees had a large effect on tree dieback and mortality after accounting for climate and soil effects. In pinyon pine, the effect of stand structure depended upon the annual water deficit of the site. At the wettest sites (10th percentile of climatic water deficit (CWD)), an increase in neighborhood basal area was associated with decreased canopy loss, while the opposite pattern was seen at the driest sites (90th percentile of CWD), where trees were predicted to lose about 10 percent of their canopy as neighborhood basal area increased from its minimum (0 m<sup>2</sup>) to maximum (1.4 m<sup>2</sup>). Increased neighborhood basal area was also associated with greater pinyon mortality, with an increase of mortality risk of 3.8 times from minimum to maximum basal area. For juniper, the effect was weaker and not statistically significant. Pinyon canopy health was negatively associated with proximity to adjacent trees. For each 1-m reduction in distance between a tree with its nearest neighbor, canopy health was predicted to decline by 1.6 percent. Proximity to neighboring trees

was not important for juniper dieback or for pinyon stem mortality, which were driven more by abiotic factors such as soil and climate.

These results indicate that on drier sites, competition among trees reduces the resilience of pinyon-juniper woodlands during drought. This finding implies that risk of drought-induced mortality may change during stand development. Drought may precipitate self-thinning in denser, intermediate-aged stands, while sparse young stands or old-growth may be less susceptible to drought. Severe drought is not uncommon in the Great Basin (fig. 1), and it may play an important role in the long-term dynamics of pinyon-juniper woodlands by introducing structural and habitat heterogeneity.

Silvicultural treatments in pinyon-juniper woodlands may be a useful strategy in the management of denser stands. Moreover, because the effect of competition depends upon site conditions, with more arid sites more prone to competition-induced drought sensitivity, thinning may be more useful in sites with higher stocking relative to the maximum stand density index. We encourage the collection of baseline data before and after thinning treatments, as well as in comparable

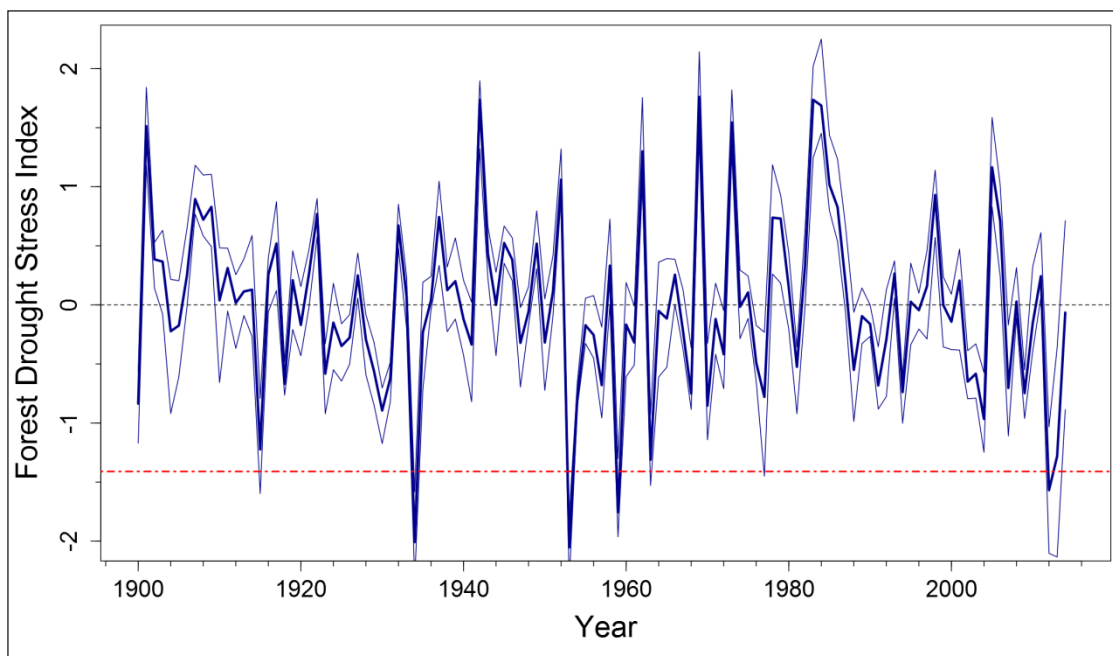


**Figure 1**—Forest drought stress index (FDSI; Williams et al., 2013) over the entire PRISM climate time series (PRISM Climate Group) for our central Nevada plot network. Lower values indicate greater drought stress, and values of FDSI below the horizontal line at FDSI = -1.41 are linked historically with decreased tree vigor.

control sites, in order to create datasets required to more precisely quantify stand density influences on drought resilience. Further study of stand-level processes in pinyon-juniper woodlands (e.g. self-thinning) is also needed to develop basic silvicultural guidelines (e.g. density management) appropriate for highly heterogeneous, topographically and edaphically complex woodland systems. It is probable that the thinning of dense patches of trees, particularly on dry sites likely to have low maximum pinyon-juniper stocking, will enhance drought resilience. However, additional controlled experiments manipulating densities and stand structure are needed to provide a critical test of the role of stand density in pinyon-juniper resilience to drought, and for development of silvicultural prescriptions aimed at maintaining healthy woodlands given drought conditions and climate change processes.

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# Vulnerability of Piñon-Juniper Habitats to Climate Change in the Southwestern USA

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**KEYWORDS**—adaptation, climate envelope, climate vulnerability, fire severity, Global Circulation Models, piñon-juniper, realignment

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Wildlife managers are concerned about the impacts of climate change on piñon-juniper habitats and on the ecosystem services that they provide (Friggens et al. 2013). Contrary to common perception, this extensive ecosystem type is greatly varied in structure, composition, ecology, and habitat value (Comer et al. 2003; Moir and Carlton 1986; Romme et al. 2009), along with the expected manifestations of climate change vulnerability (Comer et al. 2012; Rehfeldt et al. 2006). This work resulted in a geospatial climate vulnerability surface that included five types of piñon-juniper across Arizona and New Mexico.

To predict vulnerability, the landscape was spatially stratified into recognizable ecosystem types that repeat across the landscape. Then, base-level polygons were generated for the analysis area with each segment representing similar site potential at the scale of individual plant communities. Segments were attributed with biophysical, contemporary climate, and projected climate for multiple Global Circulation Models (GCMs). Climate envelopes were developed for each type of piñon-juniper based on pre-1990 climate data and the most discriminating climate variables. The discriminant analysis of 21 climate variables resulted in the selection of five key variables for building climate envelopes for each piñon-juniper type. Finally, each segment was assigned a vulnerability score based on the projected departure in future climate from the characteristic climate—an inference of probable change in

site potential climate envelopes. In the context of this analysis, vulnerability ratings are a consequence of the breadth of the envelope for a given type of piñon-juniper, the current climate status of a given location relative to its envelope, and the magnitude of projected change in climate at that location.

Of the five major types of piñon-juniper in the southwestern USA, the Piñon-Juniper Sagebrush system appears to be the most vulnerable with nearly 90 percent of its area occurring as high vulnerability (table 1). The percentage of high vulnerability in the remaining types varies considerably, between 22-64 percent.

In all cases, by the late 21st-century, the majority of area currently classified as piñon-juniper will have at least exceeded historic envelopes, suggesting important changes ahead for dependent wildlife. The efficacy of the vulnerability surface was tested by assessing contemporary fire severity patterns (Eidenshink et al. 2007) against the predicted vulnerability for all piñon-juniper types collectively, and found that low vulnerability sites were over 38 percent more likely to experience stand replacement fire than the background levels suggested in the vulnerability surface. Conversely, high vulnerability areas were nearly 33 percent less likely to experience stand replacement fire, with vulnerability playing a hypothetical role on plant productivity (Parks et al. 2016; Rocca et al. 2014). A follow-up study on the Lincoln

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**Table 1**—Vulnerability of the five major types of piñon-juniper to climate change in the southwestern USA.

| <b>Piñon-Juniper ERU</b>                           | <b>Vulnerability category</b> | <b>Percent</b> |
|----------------------------------------------------|-------------------------------|----------------|
| PJ Evergreen Shrub<br>15,908 km <sup>2</sup>       | Low                           | 28             |
|                                                    | Moderate                      | 50             |
|                                                    | High+                         | 22             |
| PJ Woodland (persistent)<br>22,199 km <sup>2</sup> | Low                           | 8              |
|                                                    | Moderate                      | 42             |
|                                                    | High+                         | 50             |
| PJ Grass<br>24,607 km <sup>2</sup>                 | Low                           | 7              |
|                                                    | Moderate                      | 30             |
|                                                    | High+                         | 64             |
| Juniper Grass<br>37,488 km <sup>2</sup>            | Low                           | 3              |
|                                                    | Moderate                      | 43             |
|                                                    | High+                         | 55             |
| PJ Sagebrush<br>9,173 km <sup>2</sup>              | Low                           | 0              |
|                                                    | Moderate                      | 12             |
|                                                    | High+                         | 88             |
| All Piñon-Juniper<br>109,375 km <sup>2</sup>       | Low                           | 8              |
|                                                    | Moderate                      | 38             |
|                                                    | High+                         | 54             |

National Forest in south-central New Mexico is being conducted to evaluate the relationship of vulnerability and plant productivity, while also considering the respective influences of vulnerability and uncharacteristic fire on tree regeneration patterns. Finally, the vulnerability surface was focused on vegetation and not wildlife habitat specifically; but, because vegetation serves as the principle structural component of habitat, the surface can offer an inference of habitat vulnerability and a means for underpinning analyses for specific species of wildlife that utilize piñon-juniper.

In applying the vulnerability surface for a given type of piñon-juniper, biologists could focus adaptation management for habitat in those landscapes with the highest vulnerability. An adaptive capacity process of analysis, planning, and management can link outputs from the vulnerability surface to practitioners and decisionmakers by integrating vulnerability predictions with knowledge of resilience-resistance characteristics and plant functional traits. For example, prescribed burning and tree thinning, particularly in the fire-adapted types (Juniper Grass and Piñon-Juniper Grass) that have been substantially altered from reference conditions, may improve adaptive capacity and facilitate realignment, while reducing the risk of high-severity disturbance to piñon-juniper (Bradford

and Bell 2017). For all vulnerable piñon-juniper types, managers can plan and apply treatments adaptively with particular emphasis on ecosystem function as a useful operational framework for responding to climate change for piñon-juniper habitat in the southwestern United States.

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# Fire Exclusion Linked to Increased Forest Density in a New Mexico Piñon-Juniper Savanna Landscape

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**KEYWORDS**—fire scar, tree ring, *Pinus edulis*, *Juniperus scopulorum*, Piñon-Juniper savanna

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Piñon-juniper (PJ) ecosystems cover approximately 30 million ha in the western United States (West 1999). High variability in species assemblages and structure exists across this large geographic range, driven in part by varying climate. Among the three functional PJ types (persistent PJ woodlands, PJ savannas, and PJ shrublands), the least is known about ecosystem processes and vegetation dynamics in PJ ecosystems containing a substantial grass component - PJ savannas (Romme et al. 2009). The best supported model for fire regimes in PJ is one of infrequent, high-severity fire (e.g., Floyd et al. 2004). However, it is logical that: (1) the grass component of PJ savannas could support frequent, low-severity fire, and (2) exclusion of frequent fire is a plausible mechanism for the observed increases in PJ tree density in PJ savannas. To assess these hypotheses I used dendroecological methods to reconstruct the history of fire, forest age structure, and composition on a ~ 30,000 ha PJ-dominated mesa landscape (Rowe Mesa, New Mexico), located at the PJ (two-needle pinyon (*Pinus edulis*)-Rocky Mountain juniper (*Juniperus scopulorum*), grassland, ponderosa pine (*Pinus ponderosa*) (PIPO) ecotone (Margolis 2014). At seven sites located on a 7-km grid I collected fire scars, aged trees, and measured forest composition and density. I cross-dated 112 fire-scarred trees (8% juniper, 17% pinyon, and 74% ponderosa) containing 630 fire scars that burned during 87 unique fire years (1547-1899). Fires recorded synchronously at multiple sites (spreading fires) occurred every 7.8 years on

average. Fires recorded at over half of the sites (wide-spread fires) occurred every 23.7 years on average. Evidence of high-severity fire (e.g., even-aged stands or groups of fire-killed snags or logs) was not observed. Instead, multi-decadal variability in tree establishment was negatively correlated with fire frequency (i.e., frequent fire reduced tree establishment). Peak PJ establishment was synchronous with the collapse of the frequent fire regime and dry conditions in the late 19th century. I conclude that frequent, low-severity fires burning grass fuels historically maintained low tree densities across what was a PJ-PIPO savanna landscape. Further, late 19th century fire exclusion associated with the removal of fine fuels from intensive livestock grazing was the primary driver of current high tree densities on Rowe Mesa dominated by PJ species (mean density = 881 live trees ha<sup>-1</sup>, 94 percent of which were < 130 years old and likely would not have been present when the frequent fire regime was active). Wildlife species that favor a more open, grassy environment would not be favored by the documented changes, whereas other species that prefer a dense woodland environment could benefit from the changes.

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# Pinyon Jay Habitat Use and Management Recommendations in New Mexico Piñon-Juniper Woodlands

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**KEYWORDS**—*Gymnorhinus cyanocephalus*, habitat, management, piñon-juniper, Pinyon Jay, spatial scale

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Pinyon Jays and piñon pines have a mutualism whereby the trees provide mast crops of highly nutritional seeds that enhance the jays' population viability (Marzluff and Balda 1992), and the jays cache the seeds, serving as the tree's main long-distance seed disperser (Ligon 1978). Breeding Bird Survey data indicate that Pinyon Jay populations have been declining for 50 years (Sauer et al. 2014), and they are currently the fastest-declining piñon-juniper bird species (Boone et al. in press). Ongoing decline of the piñon tree's main seed disperser will limit the potential of the trees to reestablish in areas of high mortality, colonize higher elevations, or shift distributions northward in response to climate change.

We modeled Pinyon Jay habitat use to inform management and benefit both the bird and tree species. To understand Pinyon Jay habitat needs, we modeled habitat use at the home range, nesting colony, and nest scales at three study sites across New Mexico, USA. At the landscape scale (Ferrari and Ferrarini

2008), the home range of a Pinyon Jay flock can range from 3000–5000 ha. At the ecosystem scale (Ferrari and Ferrarini 2008), a nesting colony can range from 1–50 ha. The nest scale includes the nest tree and its immediate surroundings, an area of about 1 ha. Study areas were the Oscura Mountains, White Sands Missile Range, NM; the Manzanita Mountains, Kirtland Air Force Base, NM; and the Bureau of Land Management Farmington, NM Resource Area.

At the landscape scale, we created habitat maps of the three study areas. For the Oscura Mountains, we used an existing vegetation map (Muldavin et al. 2000) in combination with 1-m natural color aerial photography taken in 2009. For the Manzanita Mountains, we created a vegetation classification based on 6-inch color digital ortho-imagery taken in 2008 (Johnson et al. 2014, 2016). For the Farmington Resource Area, we used National Agriculture Imagery Program 1-m visible and near-infrared digital aerial photography from 2014 and Landsat 8 satellite imagery to create a land

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cover classification (Johnson et al. 2017). Map units were defined based on Pinyon Jay habitat use. Year-round home ranges of two Pinyon Jay flocks in central NM averaged 4,779 ha (Johnson et al. 2014). The summer range of one flock in northwestern NM was 4,034 ha (Johnson et al. 2015). Jays used mainly Juniper Woodland and Savanna, Piñon-Juniper Woodland, and Piñon Woodland habitats (Johnson et al. 2014, 2017). Higher elevation Piñon-Juniper Woodland and Piñon Woodland were used for nesting, while lower elevation Juniper Woodland and Savanna was used more often in the nonbreeding season (Johnson et al. 2016).

We created a predictive habitat model of colony-scale habitat using the landscape-scale habitat model (above) land cover data combined with geospatial data such as slope, aspect, elevation, solar radiation, and NDVI (greenness). We first classified colony sites, then created a supervised classification of the entire study area based on the geospatial measures. For the models, we retained areas conforming to the geospatial and vegetation profiles of known nesting colonies. A comparison of the predictive model to validation colonies at the Oscura Mountains and Manzanita Mountains study sites showed 100 percent and 76 percent overlap of the validation colonies with the predictive model, indicating good predictability of the colony-scale model (Johnson et al. 2016).

Using conditional logistic regression, we modeled nest-scale habitat use at nine nesting colonies, three at each of the three study sites. We collected data on 5-m and 11.3-m radius BBird (Martin et al. 1997) plots at each nest and a paired, random plot 100 m away from the nest plot. Data collected included diameter and height of nest tree and trees on plot, nest height, nest aspect, canopy cover, ground cover, number of trees and shrubs, and so on. Tree density at nests averaged 965 trees/ha at seven colony sites in central NM and 436 trees/ha at two colonies in northwestern NM. Pinyon Jays nested on plots with higher canopy cover, larger trees, and higher litter cover in central NM (Johnson et al. 2014). In northwestern NM, they nested in larger-diameter trees (measured as root crown diameter, RCD) and taller trees, compared to random plots, while avoiding the tallest, most emergent trees (Johnson et al. 2015). Combining nest-scale data from all sites for a case-controlled conditional logistic

regression, we found that nest plots were located on cooler, northeast facing slopes and had higher total canopy cover than random plots. Nest trees were larger diameter, taller, more asymmetrical, and were surrounded by smaller trees, than corresponding trees on random plots.

In New Mexico piñon-juniper woodlands, management for Pinyon Jays should include adequate area for a flock's home range, at least 5,000 ha. Multiple land cover types such as Juniper Woodland and Savanna, Piñon-Juniper Woodland, and Piñon Woodland should be included to account for variation in use across seasons and life history stages. Sagebrush shrubland or grassland components may also be included, especially for wintering habitat. Home ranges should include multiple areas containing 1) many large (>15 cm RCD), mast-producing trees, 2) multiple colony-sized patches (50 ha) of nesting habitat, and 3) water sources, especially near colonies.

Management for Pinyon Jays at the nesting colony-scale should include Piñon-Juniper Woodland and/or Piñon Woodland and have relatively high canopy cover. Colony sites should 1) be at least 50 ha in area, 2) contain large nest trees, 3) include > 20 dense clumps containing potential nest trees, 4) be within 1 km of a water source, 5) have minimal fragmentation by roads, well pads, and so on, and 6) have minimal noise and foot traffic from March—June.

Management for Pinyon Jays at the nest-scale should include: 1) tall piñon or Utah juniper trees, 2) large diameter piñon or Utah juniper trees (mean for this study 35.4 cm, range 7 – 100 cm), 3) high tree density (400–1200+ trees /ha), 4) high canopy cover (in this study, 40% measured from the ground; mean aerial 30%), and 5) healthy trees with dense foliage. We believe these general recommendations are applicable to areas other than these three sites (e.g., nesting in larger trees, higher canopy cover), but specific numerical recommendations (e.g., number of trees/ha, mean tree diameter) will vary depending on the characteristics of each site. In general, retaining large piñon trees and high canopy cover would also benefit mule deer, Juniper Titmouse, Gray Flycatcher, Bewick's Wren, and Black-throated Gray Warbler (Pavlacky and Anderson 2001).

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# Habitat Use at Multiple Scales by Nesting Gray Vireos in New Mexico

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**KEYWORDS**—Gray Vireo, piñon-juniper, habitat, New Mexico, *Vireo vicinior*

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Piñon-juniper (*Pinus edulis*–*Juniperus* spp.) woodlands cover approximately 40 million hectares of the western United States (Romme et al. 2009) and provide habitat for a diverse suite of avian species. Several U.S. Fish and Wildlife Service (FWS) Birds of Conservation Concern (BCC) depend on piñon-juniper habitats for breeding, including the Gray Vireo (*Vireo vicinior*). Gray Vireos breed within a limited range only in the hot, arid southwestern USA (Barlow et al. 1999). In New Mexico, they occupy the piñon-juniper habitats of foothills, canyons, mesas, and rolling hills. Their populations are disjunct, however, with most occupied sites containing few, often less than 10, territories (DeLong and Williams 2006). The New Mexico Department of Game and Fish (NMDGF) currently lists the Gray Vireo as a threatened species (NMDGF 2016).

We evaluated habitat use by Gray Vireos at landscape, territory, and nest scales on U.S. Department of Defense (DOD) and Bureau of Land Management (BLM) lands in New Mexico. We collected data

between 2009 and 2012 on three DOD sites: White Sands Missile Range (WSMR; Sierra County), Kirtland Air Force Base (KAFB; Bernalillo County), and Camel Tracks Training Area (CTTA; training grounds on BLM land in Santa Fe County). We expanded the study to BLM lands in the Farmington Field Office from 2013 to 2016, including Crow Mesa (Sandoval County), Pump Canyon and Pump Mesa (San Juan County), and several canyons and rolling hills north and west of the City of Aztec (San Juan County). To date, we have completed landscape- and territory-scale analyses only at DOD sites and nest-scale analyses at both DOD and BLM sites.

At the landscape scale, the vast majority ( $\geq 75$  percent) of Gray Vireo detections and nests at DOD sites occurred in juniper-dominated woodlands or savanna habitat types containing less than 25 percent piñon trees. At KAFB and WSMR, 10 percent or fewer detections/nests also occurred in piñon-juniper woodlands (25–50 percent piñons). Additionally at WSMR, 10 percent or fewer detections/nests occurred

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in shrubland and  $\leq 4$  percent in arroyo riparian habitat types dominated by shrub species such as wavyleaf oak (*Quercus x pauciloba*), mountain mahogany (*Cercocarpus montanus*), acacia (*Acacia* spp.), catclaw mimosa (*Mimosa aculeaticarpa* var. *biuncifera*), featherplume (*Dalea Formosa*), common sotol (*Dasyllirion wheeleri*), Apache plume (*Fallugia paradoxa*), resinbush (*Viguiera stenoloba*), and desert willow (*Chilopsis linearis*; see Johnson et al. 2014).

Mean territory size at DOD sites was 0.9 ha at WSMR ( $n=44$ ), 2.8 ha at CTTA ( $n=24$ ), and 3.1 ha at KAFB ( $n=49$ ); overall mean territory size was 2.3 ha. Important predictor variables for the territory-scale analysis included slope, aspect, elevation, curvature, solar radiation, and greenness; however, the importance and/or relationship of these predictors varied across sites. At CTTA, territories occurred on terrain with more north-facing aspects, higher elevations, intermediate slopes, and lower overall solar radiation compared with non-territories. At KAFB, territories occurred at more intermediate aspects (e.g., east- or west-facing), lower elevations, and in areas with more intermediate evergreen greenness than non-territories. Finally, at WSMR, territories were situated in areas with more southerly aspects, negative (bowl-shaped) curvature, intermediate elevations, lower slopes, and lower solar radiation than non-territories. Differences in the relationship of some territory predictor variables occupancy—specifically slope, aspect, and elevation—probably reflect inherent differences in topography and elevation across the three study sites rather than specific preferences by vireos (see Johnson et al. 2014).

We found 89 Gray Vireo nests across DOD sites, all in juniper trees. Of 65 nests at BLM sites, 82 percent were in junipers, 15 percent in piñon trees, and 3 percent in big sagebrush (*Artemisia tridentata*). All but one of the piñon tree nests were from a single site, Crow Mesa, where the majority (75 percent) of vireo nests were in piñon trees. At that site, the ratio of piñon to juniper trees in the area immediate surrounding vireo nests was considerably higher (0.7) compared with the other BLM and DOD sites ( $\leq 0.2$ ).

Nest-scale habitat use models at both DOD and BLM sites indicated that Gray Vireos selected nest sites in areas with more and taller trees relative to random points within their territories; however, actual tree density

and height varied across sites. Mean tree density on BLM sites was almost three times greater (316.1 trees/ha) than on DOD sites (113.0 trees/ha). Conversely, mean tree height surrounding nests on DOD lands averaged higher (3.3–4.0 m across sites) than BLM lands (2.5–3.4 m). At DOD sites, Gray Vireos also selected nest locations with more south-facing aspects and negative (bowl-shaped) curvature compared with randomly-selected locations within their territories. Our aspect data contradict an earlier study by DeLong and Cox (2005) in Socorro and Santa Fe, NM, who reported that Gray Vireos tend to nest on west-facing aspects. At BLM sites, vireos also exhibited a weak preference for nest trees with smaller, on average, tree foliage width than random trees within their territories.

Our data indicate that Gray Vireos mainly occupy juniper-dominated woodlands and savannas but can be flexible in choosing nest substrates. Likely, preferences for territory characteristics differ across landscapes based on varying topographic features and variation in woodland species composition. While vireos may occupy woodlands of varying density, height, and composition, they prefer nesting around taller and denser trees within their territories. Thus, piñon-juniper management activities should strive to retain older, more mature stands with similar tree densities as documented in our study.

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# Elk, Deer, and Piñon-Juniper in New Mexico: Needs, What Works, and What Doesn't

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**ABSTRACT**—Elk (*Cervus elaphus*) and mule deer (*Odocoileus hemionus*) differ in nutritional and structural habitat needs; consequently, management of pinyon-juniper (*Pinus* spp.-*Juniperus* spp.) woodlands to provide optimal habitat differs for these herbivores. Diet quality requirements are higher for deer, and are highest for both species during late gestation, lactation, and antler growth. Properly timed treatments, especially prescribed burning, can enhance forage quality and alter forage species composition, providing dietary benefits during these times. Late-winter burns provide a nutrient flush during these critical periods, and burn intervals can be manipulated to selectively benefit deer or elk by favoring browse or herbaceous species in the understory. Burning at other times can increase forage biomass, but has fewer beneficial effects on forage quality. Overstory thinning can also increase forage biomass given proper slash treatment, but excessive thinning can degrade cover attributes. For example, deer prefer higher levels of security cover, and most movements are near denser woodland patches, which ideally compose approximately 25 percent of each home range on the landscape mosaic. Elk seek thermal cover when average daily temperatures exceed their thermal neutral zone, but this can be provided by single trees or clumps. Other treatments, such as chipping or shredding, may decrease forage and reduce habitat quality if residual biomass is not treated properly.

**KEYWORDS**—elk, fire, forage, mule deer, New Mexico, nutrition, pinyon-juniper, thinning

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## INTRODUCTION

Habitat quality is the main determinant of individual and population performance of mule deer (*Odocoileus hemionus*) and elk (*Cervus elaphus*), primarily through the influence of nutrition on individual body condition and consequent effects on demographic vigor (Cook 2002; Hanks 1981; Wakeling and Bender 2003). Because availability and nutrient content of forage largely determine the condition of individuals (Cook 2002; Verme and Ullrey 1984; Wakeling and Bender 2003), a direct link exists between habitat and population performance (Bender 2012; Cook et al. 2004; Gaillard et al. 2000; Hanks 1981; Hoenes and Bender 2012). Thus, attributes of habitat, including plant community composition, phenological development, and availability of cover, influence the condition of individuals and hence population performance (Hoenes

and Bender 2012; Wakeling and Bender 2003), although the mechanisms that drive these habitat-performance relationships are often poorly appreciated by managers.

Expansion and increasing density of pinyon-juniper (*Pinus* spp.-*Juniperus* spp.) woodlands, particularly juniper, is a contentious issue for management of mule deer and elk habitat in the southwestern United States including New Mexico (Lutz et al. 2003; Short and McCullough 1977; Short et al. 1977). Pinyon-juniper (PJ) woodlands are one of the most extensive vegetation types in North America (Romme et al. 2009), and include nearly 3 million ha in New Mexico (Van Hooser et al. 1993), almost 10 percent of the total area of the state. Most of this woodland is actual or potential habitat for mule deer and elk, making PJ woodlands the most abundant habitat type for mule deer and

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elk in New Mexico. However, a number of attributes associated with these woodlands can limit their value to deer and elk, including:

1. High concentrations of secondary plant compounds, especially terpenes, in foliage. The antimicrobial properties of these limit fermentation and thus rumen function (Schwartz et al. 1980a,b). Consequently, forage value of juniper is limited despite high digestible energy.
2. Presence of other secondary plant compounds such as phenolics, which may have community-level effects on the herbaceous and woody shrub understory, such as inhibiting growth due to leachate (Ramsey 1989).
3. Limited understory development due to competition for light, water, and other nutrients (Hoenes and Bender 2012; Lutz et al. 2003; Pieper 1990; Short and McCullough 1977; Short et al. 1977). This limits the availability of ungulate forage in denser PJ stands.

The lack of understory development in particular has driven most management efforts intended to improve PJ woodlands as deer and elk habitat. Most treatments attempt to increase understory production through partial or complete removal of the PJ canopy, because an inverse relationship between overstory PJ cover and understory production is frequently seen (e.g., Brockway et al. [2002]; Halbritter and Bender [2011b]; Hoenes and Bender [2012]; Howard et al. [1987]; Pieper [1990]; Short et al. [1977]). Understory species include potentially important forages such as forbs and woody browse (Hoenes and Bender 2012).

An implicit assumption behind overstory reduction treatments is that understory biomass equals food, which ignores differing foraging strategies and forage quality requirements of large herbivores (Hofmann 1985) as well as differing nutritional quality and palatability of resultant understory communities. Moreover, the magnitude and type of understory response can be influenced by a variety of factors, including the type, timing, and size of treatments, and the composition of extant understory. (Bender 2011, 2012; Bergman et al. 2014). Consequently, the habitat-performance relationship is far more complex than the simple generalization that less PJ equals more food. Because

of this, the anticipated benefits of treatments are often not realized. In a recent review of published material that assessed population responses of wildlife to PJ treatments, Bombaci and Pejchar (2016) found that less than 10 percent of mechanical removal treatments (i.e., chaining, bulldozing) and less than 20 percent of thinning treatments resulted in any positive response by ungulates.

The lack of positive responses raises the issue of what role PJ actually plays in deer and elk habitat-performance relationships; in other words, does the presence of untreated PJ negatively affect performance of elk and deer or not? Further, inconclusive findings may not solely result from a simplistic “less PJ equals more food” assumption. Habitat-herbivore interactions and resultant individual and population dynamics are highly complex, and require understanding of how habitat attributes fundamentally affect individuals in a population, and how individual responses may (or may not) be reflected in population responses. Understanding these fundamental relationships is necessary for developing informed management treatments to benefit deer and elk.

Moreover, mule deer and elk are very different herbivores, with very different forage quality and cover requirements (Cook 2002; Hofmann 1985; Verme and Ullrey 1984; Wakeling and Bender 2003). Hence, there is no single prescription for optimal management of PJ woodlands or other habitats for elk and deer or large herbivores, as often presented in management guidelines that tend to emphasize thermal cover (e.g., Miller et al. [2005]; Thomas et al. [1979]; Wittmer et al. [1985]). Other issues may also cloud the results of herbivore-PJ studies, including the use of evaluation metrics (e.g., population size, survival, pellet group, or telemetry location density) that are either problematic indices of habitat quality or far removed from fundamental responses to habitat; a poor understanding of treatment effects on elk and deer habitat needs; and use of management treatments ostensibly intended for deer or elk but actually intended to achieve other, often unstated, management goals. A common example is the continued emphasis on winter thermal cover (Thomas et al. 1979) in Federal lands management for elk and deer, despite the demonstration that winter thermal cover provides no energetic benefit (Cook et

al. 1998; Freddy et al. 1986) and that security cover can be provided by multiple seral states other than large sawtimber or old growth.

These issues contribute to the challenge of identifying successful treatments of PJ for elk and deer. To illustrate, Bombaci and Pechjar (2016) recently reviewed more than 40 primary papers regarding PJ treatments and elk or deer. A January 2017 search of Web of Knowledge (Biological Abstracts) revealed at least 67 papers on mule deer and PJ, and at least 26 on elk and PJ, and this search did not include literature from sources such as USDA Forest Service publications. Given all of these publications and knowledge, it would be a reasonable expectation that any number of management treatments that consistently improve PJ woodlands as elk and deer habitat would have been identified. Instead, the actual efficacy of most treatments is uncertain (Bombaci and Pechjar 2016). Reasons for this uncertainty include the limitations of many of these studies, particularly the lack of relating habitat management practices to increases in individual performance, which must be present to realize population-level responses.

Consequently, understanding fundamental relationships to habitat (i.e., habitat-individual performance), how these differ between mule deer and elk, and how these are (or are not) met by PJ habitats, and why, is necessary to develop informed actions to benefit these large herbivores. Herein, I attempt to clarify these issues in order to guide effective habitat management. My goals included:

1. Clarifying the fundamental role of habitat for elk and deer
2. Illustrating how this fundamental relationship affects individual and population performance
3. Evaluating the influence of PJ on performance of deer and elk, and
4. Evaluating preliminary results from selected ongoing long-term studies of management treatments in terms of providing for elk and deer needs for forage and cover.

To do this I review the role of habitat in individual and population productivity (i.e., the

habitat-condition-productivity link), and existing literature on fundamental relationships between mule deer, elk, and PJ in New Mexico. I further use existing and new datasets from multiple populations and study areas in New Mexico to define needs of mule deer and elk relative to PJ woodlands, and to identify treatments that provide for these needs in the hotter, more arid woodlands of New Mexico.

## What is Elk and Deer Habitat?

Understanding how habitat fundamentally benefits individuals, and how or why populations then may (or may not) respond to treatments, is key to designing effective habitat management strategies for deer and elk. The fundamental role of habitat is to maximize the energy balance of individuals, because every life process of homeothermic organisms is fundamentally influenced by nutrition and individual condition (e.g., Cook 2002; Cook et al. 2004; Hanks 1981; Mautz 1978; Verme and Ullrey 1984; Wakeling and Bender 2003). Habitat affects individual energetics of wildlife both directly and indirectly. The primary direct influence is providing nutrients, that is, foraging habitat, especially forage quality. The indirect influences include:

1. Minimizing disturbance, that is, hiding or security cover
2. Acquiring food, that is, ambush or stalking cover
3. Life security, that is, hiding or security cover, and
4. Moderating environmental effects, that is, thermal cover.

Each of these ultimately focuses on optimizing individual body condition through acquiring energy, minimizing loss of energy, or preventing the individual from being converted to a different individual's energy.

To maximize individual energy balance, individuals must balance food vs. cover needs. With respect to PJ woodlands, the relevant questions thus are: How does PJ affect individual energetics, why, and how can management treatments affect these fundamental relationships?

Evaluating the influences of PJ on deer and elk, and the efficacy of management treatments to enhance

individual and population performance, requires understanding of how individuals and populations respond to resource availability. Individuals (and populations) respond to resource stress in a hierarchical and predictable manner (Gaillard et al. 2000; Hanks 1981), including:

1. Declines in individual body condition
2. Declines in juvenile conception and fecundity
3. Declines in juvenile survival
4. Declines in adult fecundity, and
5. Declines in adult survival.

Thus, resource stress is first evident in declining nutritional condition of adults, which results in delayed sexual maturation of yearlings and decreased litter size of both primiparous and multiparous females (Bender and Hoenes 2017a; Bishop et al. 2009; Cook et al. 2004; Gaillard et al. 2000; Johnstone-Yellin et al. 2009). Decreased condition further results in later birth dates and lighter birth weights of neonates, which decreases preweaning survival (Bender and Hoenes 2017a; Cook et al. 2004; Lomas and Bender 2007). Only under extreme stress may adult survival be affected (Gaillard et al. 2000). Thus, individual and population productivity are most sensitive to stress, adult survival least.

Performance metrics that assess individual condition and juvenile growth and fecundity are much closer to indexing the fundamental role of habitat on deer and elk, and thus capturing real relationships between habitat attributes and deer and elk performance (Bergman et al. 2014; Hanks 1981). Conversely, metrics such as adult survival or population rate-of-increase may not be affected until extreme levels of resource stress are reached because of compensatory responses in survival dynamics of juveniles and adults, both within and among age classes (Bender 2008). Additionally, survival and population size, being further removed from the fundamental influence of habitat, can be affected by many extraneous variables, including harvest, accidents, severe density-independent events (e.g., weather, some diseases), and population age-structure, particularly in relatively small geographic areas such as most treatment test sites or study areas. Hence, the

distance of these metrics from the fundamental effect can lead to incorrect or inconclusive evaluations of habitat treatments. They are poor evaluation metrics, and their use should be avoided (Van Horne 1983).

Nutritional requirements of individuals vary throughout the annual cycle of deer and elk (fig. 1), and these differences must also be understood if management treatments are to benefit individual energetics. For both deer and elk, the greatest nutritional requirement occurs during the last trimester of gestation and during primary lactation (Cook 2002; Wakeling and Bender 2003), equivalent to early spring through mid-summer. The magnitude of this effect is apparent by the overwhelming negative effect of lactation on individual condition under all circumstances below optimal nutrition (Bender and Hoenes 2017b; Cook et al. 2004; Piasecke and Bender 2011). Because free-ranging populations rarely if ever attain optimal nutrition, lactation status has a greater effect on individual condition than most habitat or environmental influences in free-ranging mule deer and elk populations in New Mexico (Bender and Hoenes 2017b; Piasecke and Bender 2011). This impact, however, can be reduced as habitat quality increases (Piasecke and Bender 2011). The importance of the individual energetic cycle with regard to habitat management of PJ is that management treatments must provide increased nutrition or decreased disturbance when most needed. This means, for deer and elk, during late gestation and early lactation. Consequently, treatments should be designed to provide high quality forage or a nutrient flush during spring and early summer (Bender 2011, 2012).

Cover requirements also vary. Security cover is important throughout the annual cycle of deer and elk, both to minimize disturbance and to keep the individual alive. Mule deer are more vulnerable than elk to predation, and thus their preferences for security cover are much stronger (e.g., Bender et al. [2007b]; Heffelfinger et al. [2006]; Hoenes [2008]; Lomas and Bender [2007]; Lutz et al. [2003]; Short et al. [1977]). Minimizing disturbance is more critical during periods of high energetic needs or low resource availability; for example, Shively et al. (2005) documented decreased productivity of elk in response to increased disturbance during the calving season.

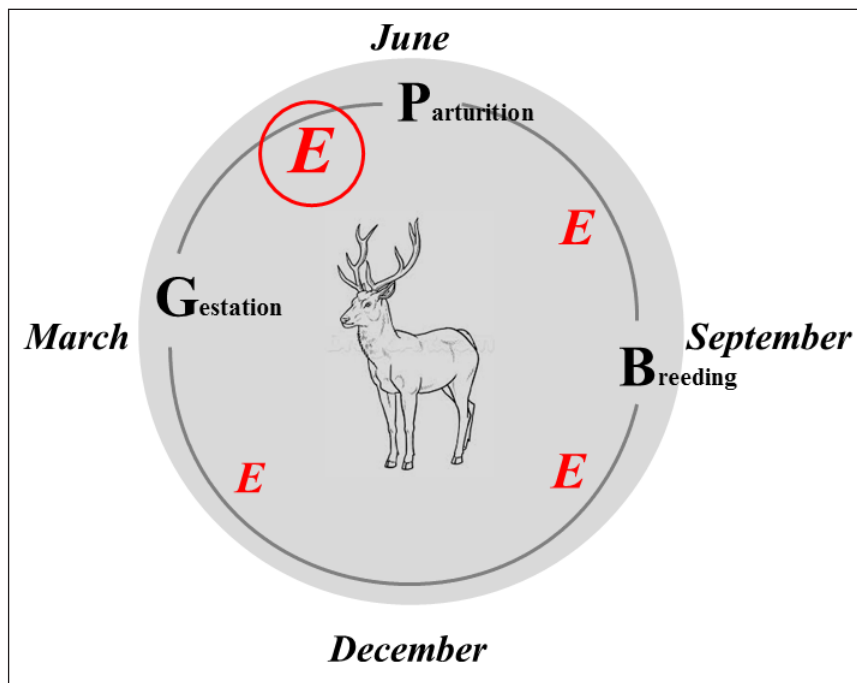


**Figure 1**—Energetic requirements (E) of elk relative to key periods in their life history; the larger the E, the greater the requirements. Energetic requirements are greatest during late gestation and early lactation, corresponding to late April through June in most of New Mexico. Mule deer show the same general pattern, but later parturition periods “twist” the figure clockwise approximately 0.5 to 1 month depending on latitude.

The need of elk and mule deer for winter thermal cover (Cook et al. 1998; Freddy et al. 1986) has been discounted, although the thermal tolerance of both species may be challenged in hotter climates (Parker and Gillingham 1990; Parker and Robbins 1984). In particular, relatively recent colonization (or recolonization) of hotter, more arid PJ woodlands by elk in New Mexico (e.g., Chaco Canyon area, Oscura and San Andres Mountains) makes summer thermal tolerance of elk a relevant issue in PJ management. The upper thermal neutral zone of elk is approximately 26.5 °C (Parker and Robbins 1984); above this, they need to actively cool themselves. Consequently, elk shift activity patterns when mean monthly high temperature exceeds their thermal tolerance, including avoiding activity during midday hours and seeking bedding sites with high overhead cover, particularly overstory trees (Bender et al. 2012a). Conversely, summer thermal cover needs of mule deer can be satisfied with under-story shrubs (Tull et al. 2001).

Last, mule deer and elk are not the same animal, despite being frequently treated as such, and both differ from cattle in their dietary needs. Forage quality requirements of elk and mule deer are much higher than

for cattle; for example, good diet quality for elk is 60 to 64 percent digestibility (or about 2.75–2.90 kcal/g of forage) (Cook et al. 2004). Needs of mule deer are even higher (Hanley 1997; Wakeling and Bender 2003), and thus mule deer are more sensitive to the specific characteristics and species composition of foraging environments. Consequently, cattle—not deer or elk—are often the primary beneficiary of management treatments that are intended to benefit wild ungulates but are poorly planned with respect to timing and type of forage responses.

These diet quality differences are reflected in the food habits of mule deer and elk. Woody browse and forbs are the main components of mule deer diets (Boeker et al. 1972; Krausman et al. 1997; Short et al. 1977; Wakeling and Bender 2003), whereas elk include much higher proportions of grass in their diets (Cook 2002; Short et al. 1977). Mule deer require higher quality forages (i.e., higher cell soluble:cell wall ratio) because their smaller rumeno-reticulum limits their ability to digest forages higher in cellulose (such as grasses) as compared to elk (Hanley 1997; Hofmann 1985; Putnam 1988). Thus, mule deer are far more selective of higher quality forage, spend more time

foraging, and are more sensitive to species composition and structure of plant communities (Hoenes and Bender 2012; Wakeling and Bender 2003). Because of this environmental sensitivity, woody browse is generally the predominant year-long component of deer diets (Boeker et al. 1972; Krausman et al. 1997), as its availability compared to forbs is less influenced by environmental factors such as precipitation (Hoenes and Bender 2012). But many forbs are better sources of nitrogen (N), digestible energy (DE), and other nutrients (Boeker et al. 1972; Urness et al. 1971; Wallmo et al. 1977). Optimal forage for mule deer is thus perfectly suitable for elk. However, elk can be highly productive on ranges dominated by lower quality foods such as grasses in which mule deer could not be productive.

Understanding the fundamental role of habitat helps clarify the current dynamics of cervids in New Mexico, and the importance of habitat quality is clearly seen in population demographics. Survival and performance of multiple elk and especially mule deer populations throughout New Mexico were limited by poor habitat quality through effects on individual body condition (Bender et al. 2007a, 2011, 2012b; Bender and Hoenes 2017b; Bender and Piasecke 2010; Halbritter and Bender 2011a; Hoenes 2008; Lomas and Bender 2007; Piasecke and Bender 2011), which strongly affected adult survival (Bender et al. 2007a, 2011; Halbritter and Bender 2011a), production and survival of juveniles (Bender and Piasecke 2010; Hoenes 2008; Lomas and Bender 2007), and predisposition to mortality (Bender et al. 2007a, 2011, 2012b; Bender and Rosas-Rosas 2016; Hoenes 2008; Lomas and Bender 2007). Consequently, the leading cause of mortality among mule deer populations was malnutrition (Bender et al. 2007a, 2011, 2012b; Hoenes 2008; Lomas and Bender 2007).

In response, mule deer populations have significantly declined throughout much of New Mexico (Bender et al. 2007a, 2011, 2012b), mostly because of declines in the quantity and quality of food and also because of seasonal drought (particularly from conception through parturition, approximately January–June) (Bender et al. 2007a, 2011, 2012b; Hoenes 2008). Conversely, elk are increasingly colonizing (or recolonizing) new habitats, including hotter, arid habitats such as PJ, which is typically associated with environmental conditions that challenge both thermoregulatory and water needs

of elk (Bender et al. 2010, 2012a). However, population productivity of elk is similarly strongly related to individual condition and hence forage quality in New Mexico (Bender and Piasecke 2010; Halbritter and Bender 2011a, 2015). Thus, while both cervids share the same fundamental relationships with habitat, trends of each species in response to recent past and current changes in habitats, including PJ expansion and related natural and anthropogenic changes, are dissimilar, highlighting the fundamental differences between these large herbivores.

### **How Does PJ Function as Elk and Deer Habitat?**

In this section, I review studies from New Mexico that assessed the fundamental quality of PJ as mule deer and elk habitat, in other words, the influence of PJ on accrual of condition (e.g., body fat). I also review the influence of PJ on home range size, an index of habitat quality (Bender 2012) that is likely to reflect cover requirements more so than nutritional needs. Last, I review the role of PJ as cover for mule deer and elk. I include only data for females, since their dynamics drive population rate-of-increase (White and Bartmann 1998).

#### ***Mule deer***

On the Corona Range and Livestock Research Center in east-central New Mexico, Bender et al. (2013) found that body fat of mule deer was not significantly related to either the proportion of PJ in spring-summer-autumn (SSA) home ranges or to the proportion of deer locations in PJ during the SSA period. However, body fat was positively related to the proportion of locations in mechanically thinned PJ during the SSA. Size of SSA home ranges was also not related to the proportion of PJ in SSA home ranges, but was negatively related to the number of locations of deer in PJ during the SSA period, as well as the number of locations in mechanically thinned PJ; the latter two indicate that habitat quality for deer apparently increased as more PJ was included in their home ranges. Additionally, more than 88 percent of locations of mule deer were less than 200 m from unmanaged PJ woodlands or savannas (Bender et al. 2013), indicating that deer required PJ woodlands for security cover in this site (Bender et al. 2013).

Similarly, body fat was unaffected by proportion of PJ in SSA home ranges or use of PJ during SSA in the San Andres Mountains in south-central New Mexico (Hoenes 2008; L. Bender, unpublished data). Size of SSA home ranges was negatively related to the proportion and use of PJ during the SSA. Deer were also located in PJ in SSA from 4 to 16 times more often than the availability of PJ in SSA ranges would predict, and 1.2 to 24 times more often than PJ availability in annual home ranges would predict.

In contrast, accrual of body fat was negatively impacted by increasing proportion of PJ in SSA home ranges and increasing use of PJ during the SSA period in north-central New Mexico (Bender et al. 2007b). Size of SSA ranges was not related to any PJ measure. Piñon-juniper was overrepresented by 12.4 percent in SSA ranges and 13.0 percent in annual home ranges as compared to its availability on the landscape, and deer were located in PJ at least 1.1 times more frequently than predicted by landscape availability (Bender et al. 2007b).

Results from these individual sites suggest a mixed response to PJ by mule deer, similar to the inconclusive results reviewed by Bombaci and Pejchar (2016). Condition appears to be unaffected or negatively affected by untreated PJ, yet many metrics of distribution suggest positive effects, likely driven by cover needs. Consequently, gains in mule deer habitat can be attained by management of PJ woodlands to increase forage quantity and quality (e.g., Bergman et al. [2014]), but care must be taken to preserve adequate structure for cover needs of mule deer (Bender 2012).

## Elk

Results from elk studies were also variable. In the Chaco Canyon area of northwestern New Mexico, Bender et al. (2012a) found accrual of body fat to be unrelated to the proportion of PJ in SSA home ranges, while size of both annual and SSA home ranges was negatively related to the proportion of PJ in home ranges. Elk home ranges included PJ at 1.8 to 2.4 times its availability on the landscape, and elk were found in PJ 1.4 to 1.5 times more frequently than availability on the landscape would predict. Additionally, PJ woodlands were an important micro-site element of elk habitat. Twenty-seven percent of

elk bedding sites were located under pinyon or juniper trees (second only to rock ledges and caves, at 35 percent), and bedding sites averaged 46 to 56 percent overhead cover and 55 to 63 percent horizontal cover (Bender et al. 2012a).

In the Lincoln National Forest of south-central New Mexico, Halbritter and Bender (2011b) found body fat to be negatively related to the proportion of PJ in SSA home ranges. In contrast to the results from the Chaco Canyon area, elk included PJ in SSA home ranges at only 0.05 to 0.16 times its availability on the landscape. The Lincoln study area was characterized by predominant availability of more mesic forest types such as aspen (*Populus tremuloides*) and mixed conifer (Halbritter and Bender 2011b), which were not present on the Chaco site. Availability of these more mesic elk habitats very likely decreased the use of PJ woodlands for cover.

Similar to mule deer, results from these individual sites suggest a mixed response to PJ by elk. Unmanaged PJ woodlands appear to have no positive impact on body condition, but metrics of distribution suggest positive effects, at least in hotter, more arid sites such as Chaco Canyon. Also similar to mule deer, gains in elk habitat potentially could be attained by management of the PJ woodlands to increase forage quantity and quality. In hotter, more arid sites, adequate amounts of thermal cover need to be maintained for elk (Bender et al. 2012a). This management practice may be particularly relevant given that elk are increasingly using these hotter, lower elevation PJ woodlands (e.g., Chaco Canyon, the Oscura and San Andres Mountains), and the thermal benefits of PJ may be a key element to maintain elk in such areas given their thermal tolerance limitations (Bender et al. 2012a).

## METHODS

### Study Areas

#### *Deer and Elk Condition and Distribution*

My study areas for mule deer condition and distribution included past projects on the Corona Range and Livestock Research Center (CRLRC), the San Andres Mountains (SAM), and Colfax County (Northcentral). Elk study areas included past projects on Chaco Culture National Historic Park (Chaco) and Lincoln

National Forest (LNF), and ongoing work in the Oscura Mountains and Chupadera Mesa (Oscura). Herein, I restricted individuals from past and current studies to those with home ranges that included only untreated PJ woodlands, and eliminated individuals with home ranges that included areas of treated PJ regardless of treatment type. Brief descriptions of these sites follow; see respective citations for additional details.

The CRLRC is a 113-km<sup>2</sup> working research ranch located in east-central New Mexico. Average high temperature in July was 29 °C and average low temperature in January was −6 °C. Mean annual precipitation was 40 cm, most of which occurs in July and August as high intensity, short duration convectional thunderstorms. During this study, annual precipitation averaged 0.93 times the normal (range = 0.72–1.22 times). Pinyon-juniper (*J. monosperma*) woodlands made up approximately 20 percent of the study site (Bender et al. 2011, 2013).

The SAM study area encompassed approximately 11,000 km<sup>2</sup> primarily within White Sands Missile Range (WSMR), south-central New Mexico. Average high temperature in July was 35 °C and average low temperature in January was −3 °C at WSMR. Precipitation averaged 20 to 35 cm annually, with the bulk of moisture occurring as short, intense rainstorms from July through September. During this study, annual precipitation averaged 1.05 times the normal (range = 0.50–1.46 times). Pinyon-juniper (primarily *J. monosperma* with occasional *J. deppeana*) woodlands made up about 7 percent of the study site (Bender et al. 2012b; Hoenes and Bender 2012).

The Northcentral study area encompassed approximately 4,860 km<sup>2</sup> in Colfax County, north-central New Mexico. Average high temperature in July was 28 °C and average low temperature in January was −8 °C at Raton, New Mexico. Mean annual precipitation averaged 44 cm, with 62 percent falling from May through August. During this study, annual precipitation averaged 1.03 times the normal (range = 0.72–1.56 times). Pinyon-juniper (primarily *J. monosperma*) woodlands made up approximately 16 percent of the study site (Bender et al. 2007a,b).

The Chaco study area covered approximately 308 km<sup>2</sup> on and adjacent to Chaco Culture National Historical Park in northwestern New Mexico. Average high temperature in July was 32 °C and average low temperature in January was −11 °C. Average annual precipitation was 23 cm, with 52 percent falling from July through October. During this study, annual precipitation averaged 0.92 times normal (range = 0.84–1.02 times). Pinyon-juniper (primarily *J. monosperma*) woodlands made up approximately 15 percent of the study site (Bender and Piasecke 2010; Bender et al. 2012a).

The LNF study area encompassed much of the 1,823-km<sup>2</sup> Sacramento Ranger District in the southern Sacramento Mountains of south-central New Mexico. Average high temperature in July was 22 °C and average low January temperature was −7 °C in Cloudcroft, New Mexico. Annual precipitation averaged 67 cm with more than 50 percent falling from June through September. During this study, annual precipitation averaged 0.82 times the normal (range = 0.66–0.96 times). Pinyon-juniper (primarily *J. monosperma* with occasional *J. deppeana*) woodlands made up approximately 10 percent of the study site (Halbritter and Bender 2011a,b).

The Oscura study area encompassed approximately 870 km<sup>2</sup> of the Oscura Mountains and adjacent Chupadera Mesa in south-central New Mexico. Average high temperature in July was 33 °C and average low January temperature was −6 °C in Bingham, New Mexico. Annual precipitation averaged 27 cm with 66 percent falling from July through October, although this average underestimates actual precipitation received in the higher elevation PJ zone. During this study, annual precipitation averaged 0.97 times the normal (range = 0.74–1.07 times). Pinyon-juniper (*J. monosperma* and scattered *J. deppeana*) woodlands made up approximately 48 percent of the study site.

### *Pinyon-juniper treatment sites*

I also include preliminary results from a subset of long-term studies assessing responses of preferred mule deer and elk forages to differing treatments of PJ woodlands on private ranches across the Southwest. Because these

studies are ongoing and several landowners preferred not to have their property identified, I refer to these ranches by general geographic location only.

Pinyon-juniper treatment areas were located in south-central (SCNM1 and SCNM2), north-central (NCNM), and central (CNM) New Mexico. Climatic and PJ relationships for the SCNM sites were similar to those listed for the Oscura site. Research on SCNM1 evaluated the influence of light long-term (i.e., decades) seasonal suitability and best pasture grazing on understory composition; this was the only site on which livestock grazing occurred. Research on the SCNM2 site has evaluated the effects of broadcast burning (i.e., understory burning of the entire ranch) on understory species composition since 2011. During this study, annual precipitation measured at the site averaged 0.86 times (range = 0.50–1.27 times) the approximate norm of 32 cm recorded at the nearest Western Regional Climate Center ([wrcc.dri.edu](http://wrcc.dri.edu)) station. This site was previously thinned approximately 10 years before implementation of the burning trials.

Conditions for the NCNM site were similar to those listed for the Northcentral site, except that PJ woodland was more varied, ranging from savanna with less than 10 stems/ha to denser woodland with more than 200 stems/ha. During this study, annual precipitation at the site averaged 0.94 times (range = 0.67–1.17 times) the approximate norm of 41 cm recorded at the nearest Western Regional Climate Center station. Described treatments were limited to the PJ savanna on this site, where the effect of broadcast burning on understory composition has been evaluated since 2008.

Conditions for the CNM site were similar to those for the CRLRC. During this study, annual precipitation at the site averaged 1.00 times (range = 0.89–1.18 times) the approximate norm of 40 cm recorded at the nearest Western Regional Climate Center station. This site has been used to evaluate the effects of overstory thinning and chipping or mastication since 2013.

### *Deer and Elk Condition and Distribution*

I captured adult female elk and mule deer by darting or net-gunning from a Bell 206B Jet Ranger/OH58 helicopter using carfentanil citrate and xylazine hydrochloride in late fall at the seasonal peak of condition

on each site (Bender et al. 2007a, 2011, 2012b; Bender and Piasecke 2010; Halbritter and Bender 2011a; L. Bender, unpublished data). I blindfolded individuals to minimize stress during handling, and administered penicillin, vitamin B, vitamin E and selenium (MU-SE), and an eight-way *Clostridium* bacterin to alleviate capture stress. I radio-collared each individual with a very high frequency collar (Advanced Telemetry Systems, Asanti, Minnesota) and attached unique numbered ear tags (relative to site).

Individuals were located from the ground at least once per week, and locations were recorded with a handheld GPS unit and plotted using the geographic information system ArcGIS 10.0 (ESRI, Redlands, California). I calculated 95-percent minimum convex polygons (MCPs) (Mohr 1947) for each individual annually and for SSA (April–November) locations only, and a composite 100-percent MCP with a 2.5-km buffer from all locations pooled to define the overall herbivore use area. I used annual and SSA home ranges of each individual as a clip coverage in ArcGIS 10.0 to clip 30-m raster vegetation covertype coverages to determine the vegetation composition of annual and SSA home ranges. I delineated vegetation covertypes from the U.S. Geological Survey's Southwest ReGAP land cover classification ([https://www.usgs.gov/core-science-systems/science-analytics-and-synthesis/gap/science/land-cover-data-overview?qt-science\\_center\\_objects=0#qt-science\\_center\\_objects](https://www.usgs.gov/core-science-systems/science-analytics-and-synthesis/gap/science/land-cover-data-overview?qt-science_center_objects=0#qt-science_center_objects)).

I used a SonoVet 2000 ultrasound (Medison, Seoul, South Korea) with a 5 mHz probe to measure subcutaneous fat thickness along a straight line midway between the spine, at its closest point to the coxal tuber (hip bone), and the ischial tuber (pin bone) (MAXFAT) (Cook et al. 2001; Stephenson et al. 2002). I also determined a rump body condition score (rBCS) by palpation of the soft tissue of the rump along the sacrosciatic (ischial) ligament and scored results of rBCS from standards which ranged from 1 (emaciated) to 5 (obese) in intervals of 0.25 (Bender et al. 2007a; Cook et al. 2001). I estimated body fat of elk using rBCS and MAXFAT following Cook et al. (2001), and of mule deer following Bender and Hoenes (2016).

As noted earlier, I limited this analysis to individuals that had only untreated PJ woodlands in their annual and SSA home ranges in order to assess the influence

of PJ woodlands on condition in the absence of woodland treatments. I used a linear mixed model (PROC GLIMMIX; SAS Institute Inc., Cary, North Carolina) to model the effect of proportion of PJ in annual and SSA home ranges of mule deer and elk on accrual of body fat and size of annual and SSA home ranges. I normalized data within populations to account for differences in magnitude of home ranges and fat levels among populations. Population was treated as a random effect in all models, and I used a hierarchical model for body fat to account for the effect of lactation status on condition of mule deer and elk (Bender and Hoenes 2017b; Piasecke and Bender 2011). If PJ had a significant effect on body fat, I further modeled the lactation  $\times$  pinyon-juniper interaction to see if PJ cover had differing effects on condition of lactating and dry females.

## Forage Responses

I used line-point intercept transects to determine plant cover and species composition during mid- to late August or September annually depending on location, near the end of the growing season. I randomly placed a randomly oriented 100-m transect in each of 15 to 20 patch or stand replicates for landscape treatments (or 10–15 transects within each stand for stand-level treatments), and recorded ground cover as bare, rock, litter, or vegetation at every 1-m interval along transects, with vegetation being further identified to species. I segregated plants into highly palatable, high quality food for deer or elk, or both (hereafter, preferred), or food for other species. I used diet studies from New Mexico (e.g., Boeker et al. 1972] Hoenes 2008; Short et al. 1977; L. Bender, unpublished data) and diets and associated diet quality data collected from many of the previously mentioned study areas used for elk and deer condition as well as ongoing studies to identify preferred forages at each site (L. Bender, unpublished data).

I estimated woody species densities using the point-center quarter method (Wyoming Game and Fish 1982). Center points were established every 10 m along transects, with the four quadrants defined by the transect line and a line perpendicular to it. The nearest shrub ( $<5.1$  cm diameter at breast height [d.b.h.]) and tree ( $>5.1$  cm d.b.h.) located within each quadrant was identified to species and the distance from the center point to each tree or shrub measured. Canopy coverage was also estimated at each 10-m interval along

transects by using a spherical densiometer (Lemmon 1956), taking four separate estimates (one in each cardinal direction) and using the average of all readings as the interval estimate. I also determined the volume ( $m^3$ ) of preferred browse species by measuring canopy height (at the center) and diameter (mean of two perpendicular measurements) and using the formula for the shape of a solid that best approximated growth form of the shrub following Ludwig et al. (1975). Not all vegetation measures were collected at all sites.

Last, I used historical USDA Soil Conservation Service and Natural Resources Conservation Service Range Site Descriptions (RSDs) and Ecological Site Descriptions (ESDs) (<https://www.nrcs.usda.gov/wps/portal/nrcs/main/national/technical/ecoscience/desc/>) to infer historical species cover for ecological sites that occurred on the study areas.

## Pinyon-Juniper Treatments

Burn treatments were conducted in late winter or early spring (March–early April) depending on site following Bender (2011, 2012). Burn areas ranged from approximately 2 to 10 sections in the SCNM2 and NCNM sites. I assessed fire intensity using ground char and flame length classes following Ryan and Noste (1985).

Thinning and chipping or masticating treatments were performed throughout the summer on six 0.1-acre (0.04 ha) plots in the CNM site. An adjacent untreated plot was blocked with one thinned and one chipped plot into three groups. Thinned plots had slash lopped and scattered during treatment. Chip treatment (i.e., chip, and leave or remove) plots were untreated for the following two springs, then chips were manually raked and removed in the fall after the second spring because of a lack of understory growth.

I used a linear mixed model (PROC GLIMMIX; PROC GLM; SAS Institute Inc.) to analyze landscape-level data: time since burn and precipitation effects on plant cover on the NCNM site. I used patch or stand as the random effect. I used MANOVA (PROC GLM; SAS Institute Inc.) to analyze the following within-stand data: cover of forage classes preburn, 1-year postburn, and 4-years postburn in the SCNM2 site and among untreated, thinned, and chip treatments in the CNM site. I also compared preburn and 1-year

postburn cover at the NCNM site using randomization tests (Efron and Tibshirani 1993). Where appropriate, I contrasted candidate models using Akaike's information criteria (Burnham and Anderson 1998) and assessed contributions of covariates using 86-percent confidence intervals (CIs) (Arnold 2010).

## RESULTS

### Deer and Elk condition and Distribution

Overall, proportion of PJ in both annual ( $F_{1,86} = 4.8$ ;  $P = 0.025$ ) and SSA home ranges ( $F_{1,86} = 5.2$ ;  $P = 0.031$ ) was negatively related to the amount of body fat that mule deer were able to accrue (table 1), although lactating and dry females were differentially affected. Pinyon-juniper cover had a negative effect on body fat accrual of lactating females (annual:  $P = 0.008$ ; SSA:  $P = 0.011$ ), but no effect on dry females (annual:  $P = 0.209$ ; SSA:  $P = 0.194$ ). Proportion of PJ in annual ( $F_{1,69} = 2.5$ ;  $P = 0.122$ ) or SSA ( $F_{1,69} = 2.0$ ;  $P = 0.166$ ) home ranges did not affect accrual of body fat in elk (table 1).

Size of annual home ranges of mule deer was negatively related to proportion of PJ in annual ( $F_{1,120} = 13.8$ ;  $P < 0.001$ ) and SSA ( $F_{1,201} = 12.7$ ;  $P < 0.001$ ) home ranges. Similarly, size of SSA home ranges of mule deer was negatively related to proportion of PJ in annual ( $F_{1,120} = 17.9$ ;  $P < 0.001$ ) and SSA ( $F_{1,201} = 20.9$ ;  $P < 0.001$ ) home ranges. Proportion of PJ in either annual or SSA home ranges of elk was not related to size of either annual (annual:  $F_{1,90} = 0.3$ ;  $P = 0.610$ ; SSA:  $F_{1,172} = 0.5$ ;  $P = 0.475$ ) or SSA (annual:  $F_{1,90} = 0.01$ ;  $P = 0.936$ ; SSA:  $F_{1,172} = 0.9$ ;  $P = 0.343$ ) home ranges.

### Vegetation Composition

Understory vegetation cover of mule deer and elk study sites reflected the lack of understory development in denser untreated PJ woodlands (table 2). The same pattern was present on treatment areas (table 3). Preferred forage was only a small proportion of the extant understory on all sites (tables 2 and 3). Among untreated sites, preferred grasses accounted for 65 percent (90-percent  $CI_{BOOTSTRAP} = 51$ –79 percent) of total grass cover, preferred forbs 30 percent (90-percent  $CI_{BOOTSTRAP} = 21$ –39 percent) of total forb cover, and

preferred shrubs 37 percent (90-percent  $CI_{BOOTSTRAP} = 21$ –53 percent) of total shrub cover (tables 2 and 3). Availability of nutritious herbaceous forage during late gestation and early lactation was even rarer; cool-season grasses composed only 12 percent (90-percent  $CI_{BOOTSTRAP} = 8$ –20 percent) of total grass cover.

Among the three treatment sites, treatments generally increased cover of preferred grasses (+6 percent [90-percent  $CI_{BOOTSTRAP} = 3$ –9 percent]), forbs (+11 percent [90-percent  $CI_{BOOTSTRAP} = 7$ –15 percent]), and shrubs (+5 percent [90-percent  $CI_{BOOTSTRAP} = 0$ –10 percent]), but not cool-season grasses (+2 percent [90-percent  $CI_{BOOTSTRAP} = -1$ –4 percent]). The grazing treatment (SCNM) was not included as it had only 1 year of data without a pretreatment or other control.

Preferred forage was generally comparable between present and historical plant communities (table 4), with the exception of cool-season grasses, which were less abundant in current communities in six of nine comparisons. I am uncertain how relevant these data are, however. Development of RSDs began in the 1930s and 1940s and ESDs were developed mostly starting in the late 1980s, after decades of generally static management of dynamic rangelands. Because herbivory impacts associated with overuse by large herbivores can occur rapidly (Irwin et al. 1994; Noy-Meir 1981) and eliminate preferred forages (Irwin et al. 1994; Klein 1981; Severson and Urness 1994), past herbivory may have eliminated preferred forages or altered understory compositions prior to development of ESDs.

### Prescribed Burning

Fire intensity on the SCNM2 site was light (mean char class = 1.3 [standard error (SE) = 0.08]; mean flame length class = 1.6 [SE = 0.09]), corresponding to light ground char and flame lengths of less than 1 m (Ryan and Noste 1983). Burning did not significantly alter PJ density ( $F_{2,42} = 0.2$ ;  $P = 0.861$ ). Preburn, 1-year postburn, and 4-year postburn vegetation cover differed ( $F_{4,40} = 42.8$ ;  $P < 0.001$ ) (table 5). Grass ( $F_{2,42} = 5.5$ ;  $P = 0.008$ ) and forb ( $F_{2,42} = 3.3$ ;  $P = 0.048$ ) cover increased significantly after burning, then declined (table 5). Shrub cover ( $F_{2,42} = 3.3$ ;  $P = 0.047$ ) declined initially, then recovered by 4 years postburn. Burning decreased successional status of preferred browse species (mountain mahogany [*Cercocarpus montanus*]:

**Table 1**—Relationships between the proportion of pinyon-juniper woodlands in annual (annual) and spring–autumn (SSA) home ranges of mule deer and elk, and the size of annual home range (annual HR), spring–autumn home range (SSA HR), and percent body fat among study areas in New Mexico.

|           |                  | Proportion of pinyon-juniper woodlands in home range |           |                 |          |                       |                              |           |                 |          |                       |
|-----------|------------------|------------------------------------------------------|-----------|-----------------|----------|-----------------------|------------------------------|-----------|-----------------|----------|-----------------------|
|           |                  | Annual home range                                    |           |                 |          |                       | SSA home range               |           |                 |          |                       |
| Species   | Variable         | <i>P</i> (rand) <sup>a</sup>                         | $\beta^b$ | SE <sup>c</sup> | <i>P</i> | <i>N</i> <sup>d</sup> | <i>P</i> (rand) <sup>a</sup> | $\beta^b$ | SE <sup>c</sup> | <i>P</i> | <i>N</i> <sup>d</sup> |
| Mule deer | Size (annual HR) | 0.922                                                | −0.32     | 0.09            | <0.001   | 122                   | 0.939                        | −0.30     | 0.08            | <0.001   | 205                   |
|           | Size (SSA HR)    | 0.900                                                | −0.39     | 0.09            | <0.001   | 122                   | 0.977                        | −0.31     | 0.07            | <0.001   | 205                   |
|           | Body fat         | 0.414                                                | −0.25     | 0.11            | 0.025    | 91                    | 0.311                        | −0.22     | 0.10            | 0.031    | 91                    |
| Elk       | Size (annual HR) | 0.554                                                | 0.46      | 0.89            | 0.610    | 93                    | 0.675                        | 0.47      | 0.65            | 0.475    | 176                   |
|           | Size (SSA HR)    | 0.556                                                | −0.05     | 0.58            | 0.936    | 93                    | 0.471                        | 0.29      | 0.31            | 0.343    | 176                   |
|           | Body fat         | 0.319                                                | −0.19     | 0.12            | 0.122    | 74                    | 0.116                        | −0.15     | 0.10            | 0.166    | 74                    |

<sup>a</sup>Probability of random effect in model.

<sup>b</sup>Coefficient.

<sup>c</sup>Standard error.

<sup>d</sup>Number of individuals sampled.

**Table 2**—Proportion of mule deer and elk study areas in pinyon-juniper (PJ) cover, percent understory grass, cool-season grass (CSG), forb, and woody shrub cover; percent preferred (P) grass, forb, and shrub cover; shrub and tree density; and canopy cover (CC) of untreated pinyon-juniper woodlands in mule deer and elk study areas<sup>a</sup>, New Mexico. Values in parentheses are standard errors.

|               | Mule deer |          |              | Elk     |      |           |
|---------------|-----------|----------|--------------|---------|------|-----------|
|               | CRLRC     | SAM      | Northcentral | Chaco   | LNF  | Oscura    |
| Proportion PJ | 0.20      | 0.07     | 0.16         | 0.15    | 0.10 | 0.48      |
| Grass         | 15 (7)    | 12 (2)   | 26 (13)      | 23 (8)  | ---  | 27 (5)    |
| Grass (P)     | 9 (6)     | 7 (2)    | 12 (6)       | 12 (5)  | ---  | 17 (6)    |
| Grass - CSG   | 2 (2)     | 2 (1)    | 3 (2)        | 11 (5)  | ---  | 1 (<1)    |
| Forb          | 2 (2)     | 3 (1)    | 0.3 (0.9)    | 5 (3)   | ---  | ---       |
| Forb (P)      | <1 (<1)   | 1 (1)    | 0 (0)        | 2 (2)   | ---  | ---       |
| Shrub         | 4 (4)     | 9 (1)    | 6 (7)        | 27 (9)  | ---  | ---       |
| Shrub (P)     | 2 (1)     | 3 (3)    | 6 (7)        | 10 (6)  | ---  | ---       |
| Shrub/ha      | ---       | 541 (40) | ---          | ---     | ---  | 642 (179) |
| Shrub/ha (P)  | ---       | 274 (66) | ---          | ---     | ---  | 257 (77)  |
| Tree/ha       | ---       | 223 (18) | 188 (41)     | ---     | ---  | 106 (30)  |
| Tree (% CC)   | 62 (3)    | 25 (3)   | ---          | 33 (12) | ---  | ---       |

<sup>a</sup>CRLRC: Corona Range and Livestock Research Center; SAM: San Andres Mountains; Northcentral: Colfax County; Chaco: Chaco Culture National Historic Park; LNF: Lincoln National Forest; Oscura: Oscura Mountains and Chupadera Mesa.

**Table 3**—Percent understory grass, cool-season grass (CSG), forb, and woody shrub cover; percent preferred (P) grass, forb, and shrub cover; shrub and tree density; and canopy cover (CC) of pinyon-juniper treatment areas<sup>a</sup>, New Mexico. Values in parentheses are standard errors.

|             | SCNM1   | SCNM2    | NCNM    | CNM     |
|-------------|---------|----------|---------|---------|
| Grass       | 16 (3)  | 15 (2)   | 19 (2)  | 2 (2)   |
| Grass (P)   | 12 (3)  | 12 (2)   | 16 (2)  | <1 (<1) |
| CSG         | <1 (1)  | 1 (<1)   | 1 (<1)  | 0 (---) |
| Forb        | <1 (<1) | 4 (1)    | 1 (<1)  | 6 (2)   |
| Forb (P)    | <1 (<1) | 1 (1)    | 0 (---) | 1 (1)   |
| Shrub       | <1 (<1) | 22 (3)   | <1 (<1) | 4 (1)   |
| Shrub (P)   | 0 (---) | 6 (4)    | 0 (---) | 2 (1)   |
| Tree/ha     | 29 (9)  | 131 (37) | 5 (3)   | ---     |
| Tree (% CC) | ---     | ---      | ---     | 76 (6)  |

<sup>a</sup>SCNM1 and SCNM2: south-central New Mexico; NCNM: north-central New Mexico; CNM: central New Mexico.

**Table 4**—Percent cover of all grasses, cool-season grasses (CSG), forbs, and woody browse species (shrub), and preferred forage (P) within these classes, in current (Ext) understory and in historical (Hist) plant communities in mule deer and elk study areas, New Mexico<sup>a</sup>. Extant cover that is lower than historical is in **bold**.

|           | No specific treatment |      |           |      |              |      |           |      |          |      |              |      | Treated      |      |              |      |              |      |
|-----------|-----------------------|------|-----------|------|--------------|------|-----------|------|----------|------|--------------|------|--------------|------|--------------|------|--------------|------|
|           | CRLRC                 |      | SAM       |      | Northcentral |      | Chaco     |      | Oscura   |      | SCNM1        |      | SCNM2        |      | NCNM         |      | CNM          |      |
|           | Ext                   | Hist | Ext       | Hist | Ext          | Hist | Ext       | Hist | Ext      | Hist | Ext          | Hist | Ext          | Hist | Ext          | Hist | Ext          | Hist |
| Grass     | <b>15</b>             | 20   | <b>12</b> | 13   | 26           | 21   | <b>23</b> | 24   | 27       | 20   | 16           | 12   | 15           | 13   | 19           | 16   | <b>2</b>     | 20   |
| G (P)     | <b>9</b>              | 12   | <b>7</b>  | 9    | <b>12</b>    | 14   | 12        | 12   | 17       | 15   | 12           | 9    | 12           | 9    | 16           | 14   | <b>&lt;1</b> | 12   |
| CSG       | <b>2</b>              | 3    | 2         | 1    | 3            | 3    | 11        | 11   | <b>1</b> | 2    | <b>&lt;1</b> | 2    | <b>&lt;1</b> | 1    | <b>1</b>     | 4    | <b>0</b>     | 3    |
| Forb      | 2                     | 2    | 3         | 1    | <b>&lt;1</b> | 3    | 5         | 2    | ---      | 2    | <1           | 1    | 4            | 1    | 1            | 1    | <b>1</b>     | 2    |
| Forb (P)  | <b>&lt;1</b>          | 1    | 1         | <1   | <b>0</b>     | 1    | 2         | 1    | ---      | 1    | <1           | <1   | 1            | <1   | ---          | 1    | 1            | 1    |
| Shrub     | 4                     | 3    | 9         | 8    | <b>6</b>     | 13   | 27        | 10   | ---      | 12   | <b>&lt;1</b> | 3    | 22           | 8    | <b>&lt;1</b> | 2    | 4            | 3    |
| Shrub (P) | 2                     | 1    | 3         | 3    | <b>6</b>     | 11   | 10        | 6    | ---      | 4    | <b>0</b>     | 2    | 6            | 3    | <b>0</b>     | 1    | 2            | 1    |

<sup>a</sup>CRLRC: Corona Range and Livestock Research Center; SAM: San Andres Mountains; Northcentral: Colfax County; Chaco: Chaco Culture National Historic Park; Oscura: Oscura Mountains and Chupadera Mesa; SCNM1 and SCNM2: south—central New Mexico; NCNM: north-central New Mexico; CNM: central New Mexico.

**Table 5**—Percent cover of forage classes and preferred forage (P), and tree density, prior to burning, 1 year after burning, and 4 years after burning, south-central New Mexico. Values in parentheses are standard errors. Values followed by different letters differ at the  $P < 0.001$  significance level.

|                   | Preburn  | Postburn (1 year) | Postburn (4 years) |
|-------------------|----------|-------------------|--------------------|
| Grass             | 15 (2)A  | 31 (4)B           | 23 (5)AB           |
| Grass (P)         | 12       | 25                | 24                 |
| Cool-season grass | <1       | 1                 | 1                  |
| Forb              | 4 (1)A   | 11 (3)B           | 5 (2)A             |
| Forb (P)          | 1        | 4                 | 2                  |
| Shrub             | 22 (3)A  | 14 (2)B           | 23 (4)A            |
| Shrub (P)         | 6        | 6                 | 5                  |
| Tree (trees/ha)   | 131 (37) | 105 (30)          | 115 (33)           |

preburn volume = 0.63 [SE = 0.11]); postburn volume = 0.30 [SE = 0.06];  $P_{\text{RAND}} < 0.001$ ). This decrease in plant volume was reflected in the drop in shrub cover 1 year postburn.

Fire intensity was similarly light on the NCM site (mean char class = 1.6 [SE = 0.07]; mean flame length class = 1.9 [SE = 0.09]), corresponding to light-moderate ground char and flame lengths of less than 1 m (Ryan and Noste 1983). Burning increased cover of all grasses (+21 percent;  $P_{\text{RAND}} = 0.007$ ) and warm-season grasses (+19 percent;  $P_{\text{RAND}} = 0.014$ ) by 1 year postburn, but not cool-season grasses, forbs, or shrubs

( $P_{\text{RAND}} < 0.166$ ). Among candidate models including time since burn, annual precipitation, growing season precipitation, precipitation during the previous winter, and their additive and interaction effects, only time since burn ( $F_{1,34} = 23.0$ ;  $P < 0.001$ ) and time since burn ( $F_{1,33} = 25.4$ ;  $P < 0.001$ ) + winter precipitation ( $F_{1,33} = 4.7$ ;  $P = 0.037$ ) were potentially informative (i.e.,  $\Delta\text{Akaike's information criterion [AICc]} < 3$ ). Time since burn (summed AIC  $\omega = 1.00$ ) was the predominant variable influencing trends in grass cover following burning for the first 7 years postburn (table 6); grass cover declined approximately 3 percent annually each year.

**Table 6**—Candidate models of grass cover as influenced by time since burning (Time), precipitation (Precip), and time since burning (Time) in north-central New Mexico. Values of variables with 86-percent confidence intervals that do not include 0 are in **bold**. P(w) = winter precipitation.

| Model      | $\Delta\text{AICc}$ | AIC $\omega$ | Time          | SE           | Precip       | SE           |
|------------|---------------------|--------------|---------------|--------------|--------------|--------------|
| Time       | 0                   | 0.81         | <b>-0.027</b> | <b>0.006</b> | ---          | ---          |
| Time, P(w) | 2.9                 | 0.19         | <b>-0.027</b> | <b>0.005</b> | <b>0.023</b> | <b>0.011</b> |

Precipitation received during the previous winter (summed AIC  $\omega = 0.19$ ) was the only other variable that had 86-percent CIs that excluded 0 in some models (table 6). Warm-season grass and all grass cover showed nearly identical relationships, whereas cover of cool-season grasses, forbs, and shrubs was not related to time since burning ( $P > 0.185$ ) or winter or growing season precipitation ( $P > 0.693$ ), very likely because of scarcity at this site (fig. 2).

Similar results were seen following burning in 2015. Burning increased cover of all grasses (+15 percent;  $P_{\text{RAND}} = 0.013$ ) and warm-season grasses (+13 percent;  $P_{\text{RAND}} = 0.025$ ) by 1 year postburn, but not cool-season grasses, forbs, or shrubs ( $P_{\text{RAND}} < 0.209$ ).

## Thinning and Chipping

Overstory canopy cover was reduced from more than 70 percent to less than 20 percent with each treatment (table 7). Vegetation responses differed among treatment combinations ( $F_{3,5} = 36.2$ ;  $P < 0.001$ ). Three years following treatment, the chip treatment did not differ ( $P \geq 0.266$ ) from the control in any forage class (table 7). The thinning treatment had greater cover of all forage classes ( $P \leq 0.032$ ) as compared to the chip treatment and control, with the exception of forb cover, which was similar between the thinning treatment and control ( $P = 0.166$ ).

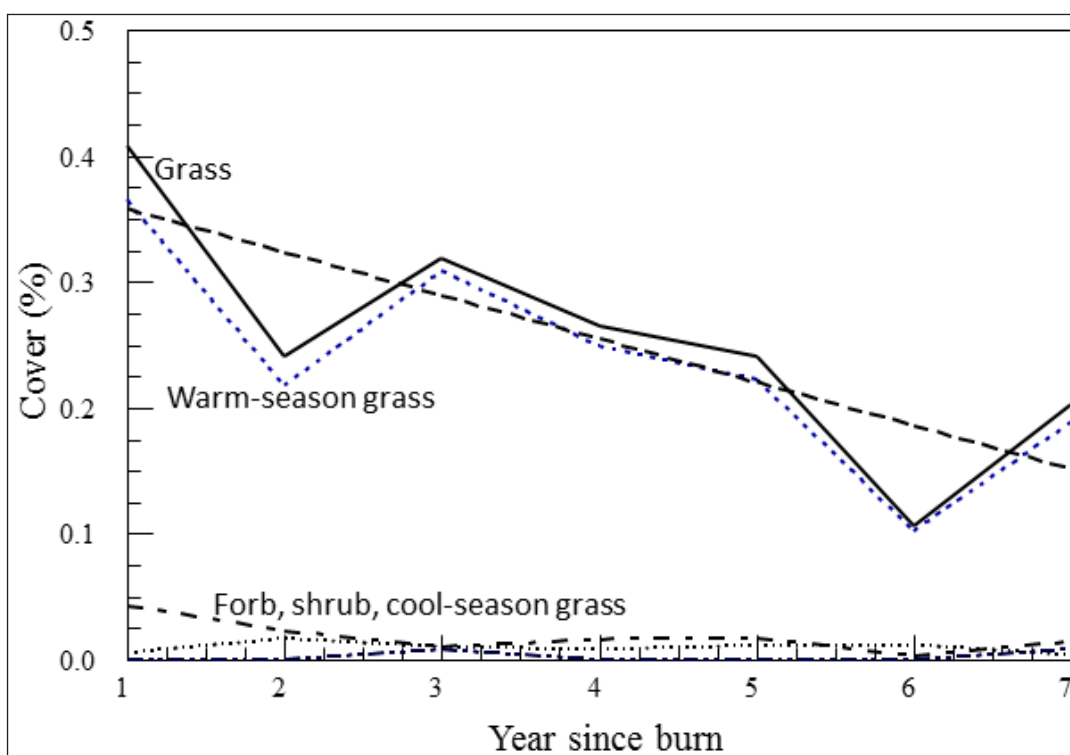


**Figure 2**—Trends in percent cover of all grasses, warm-season grasses, cool-season grasses, forbs, and shrubs following prescribed burning in late winter in the north-central New Mexico (NCNM) site. Also shown is the regression line for total grass cover, indicating an approximate 3-percent decline each year postburn.

**Table 7**—Percent cover (SE) of forage classes and preferred forage (P), and tree canopy cover, after 3 growing seasons on pinyon-juniper control sites, chip treatment sites, and thinned sites, central New Mexico. Values in parentheses are standard errors. Values followed by different letters differ at the  $P < 0.001$  significance level.

|                   | Control | Chip and leave or remove | Thin    |
|-------------------|---------|--------------------------|---------|
| Grass             | 2 (2)A  | 6 (3)A                   | 27 (7)B |
| Grass (P)         | <1      | <1                       | 8       |
| Cool-season grass | 0       | 0                        | 1       |
| Forb              | 6 (2)AB | 4 (1)A                   | 9 (2)B  |
| Forb (P)          | 1       | 1                        | 2       |
| Shrub             | 4 (1)A  | 4 (1)A                   | 12 (3)B |
| Shrub (P)         | 2       | 2                        | 6       |
| Tree (% CC)       | 76 (6)  | 19 (3)                   | 17 (2)  |

## Grazing

The SCNM1 site was included as an example of a site representative of light multipasture seasonal suitability and best pasture grazing. Differences between current and historical plant composition on this site showed patterns similar to those on the other, ungrazed sites (table 4). Cool-season grasses and preferred shrubs were particularly lacking relative to historical plant community composition.

## DISCUSSION

The lack of understory associated with untreated PJ woodlands was reflected in the nutritional condition of deer and elk using these sites, as inclusion of untreated PJ woodlands in home ranges negatively affected nutritional condition of mule deer, and had no positive effect on elk (table 1). These results were similar to individual site results, which showed variable responses within species, although condition was never positively associated with untreated PJ in New Mexico (Bender et al. 2007b, 2012a, 2013; Halbritter and Bender 2011b; Hoenes 2008; see Bergman et al. 2014 for similar results in Colorado). Particularly rare in untreated PJ stands were cool-season grasses (12 percent of grass cover) and preferred browse (37 percent of shrub cover), forages that would be palatable and nutritious during the critical period of late gestation through early lactation (Short et al. 1977). Cool-season grasses were also often underrepresented in current understory communities compared to their

historical presence (table 4). Because cool-season grass was the only forage class that did not increase among treatments, preferred cool-season herbaceous native forages (or substitutes) may have to be reestablished in many PJ woodlands if forage attributes are to be enhanced significantly. This approach would particularly aid mule deer given their higher (relative to elk) diet quality requirements during the pre-monsoon late-gestation period. The generally positive responses of preferred warm-season grasses to a variety of treatments indicate that forage preferred by elk (Cook 2002; Short et al. 1977) can be significantly enhanced by many treatment options (tables 5 and 7).

Because of the current composition of most understory communities, enhancement of woody browse has the highest potential to increase the quality of PJ woodlands, especially for mule deer. Woody browse is the predominant item in mule deer diets (Boeker et al. 1972; Krausman et al. 1997; Short et al. 1977; Wakeling and Bender 2003) because it is available year-round with at least moderate dietary quality. Woody browse is also less susceptible to drought than herbaceous species, especially forbs, which is an important consideration for forage management in arid Southwestern woodlands (Bender 2012; Hoenes and Bender 2011). Although seasonal drought will always decrease productivity of cervids (Bender et al. 2007a, 2011; Hoenes 2008), the presence of sufficient relatively drought-tolerant browse can maintain survival of adults (Bender et al. 2012b; Hoenes and Bender 2011).

Similar to herbaceous forages (excluding preferred warm-season grasses, which are primarily an important forage component for elk) (Cook 2002; Short et al. 1977), preferred browse was a limited component of the overall shrub understory (table 4) and may not necessarily increase in cover in the short term with treatment of PJ woodlands (among treatments, effect = +5 percent [90-percent CI<sub>BOOTSTRAP</sub> = 0–10 percent]).

Many treatments, particularly mechanical and herbicidal, have been used to increase browse in PJ woodlands (e.g., Bender et al. [2013]; Howard et al. [1987]; Miller et al. [2005]; Short et al. [1977]). However, Bender (2012) showed that mechanical removal of overstory PJ (to 11 percent CC) did not increase the cover of preferred browse as compared to unthinned PJ (62 percent CC) or PJ savanna (17 percent CC) on the CRLRC site. Despite this, body fat of mule deer was positively affected by increasing use of mechanically treated PJ sites during SSA (Bender et al. 2013). Although the mechanism behind this response was not identified, it is likely to have included both increased nutritional quality of individual plants (because of increased solar and other resource capture) and decreased successional status of individual plants (because of damage, i.e., crushing) associated with mechanical treatments (Cook and Harris 1950; Heffelfinger et al. 2006). On other sites, thinning increased protein levels of preferred shrubs by 1.2 to 1.5 times that of unthinned sites (L. Bender, unpublished data). Similarly, decreasing individual plant height and volume can increase browse availability and nutritional quality because of increased younger, less lignified, growth (Cook and Harris 1950; Heffelfinger et al. 2006; Hoenes and Bender 2011; Severson and Urness 1994).

Despite the positive relationship between mechanically treated PJ and body fat of mule deer, body fat levels of deer on the CRLRC were low ( $\leq 5.7$  percent annually for lactating females,  $< 1/3$  of what mule deer can accrue under optimal nutrition) (Bender et al. 2013). Thus, while mechanical clearing or thinning treatments may increase forage quality, increases in quantity of preferred forages are also needed to significantly enhance the quality of foraging environments, at least for mule deer. An increase in preferred browse was seen in other thinned sites such as the CNM site (table 7). But the magnitude of response (to 6 percent cover from 2 percent) was relatively low, and the total cover lower

than that seen on the CRLRC site ( $\leq 6$  percent vs. 11 percent), although the species composition differed. It is likely that cover of preferred browse will increase with additional time since treatment, however, and further monitoring is needed to see the level ultimately attained.

Overstory reduction of PJ by herbiciding is not recommended as a management strategy for elk or deer. Broadcast application of tebuthiuron decreased live PJ canopy cover from more than 60 percent to less than 30 percent on CRLRC, but also resulted in a decrease in cover of preferred mule deer browse from 11 to 18 percent down to less than 3 percent (Bender 2012; Bender et al. 2013). Individual tree treatments also decreased preferred browse to less than 10 percent cover, because browse often is established adjacent to or under the canopy of PJ trees (Bender 2012; Bender et al. 2013).

Overstory thinning in combination with broadcast burning, or broadcast burning under previously thinned stands, has shown the most potential to enhance forage in PJ, more than doubling grass and forb cover in the first year following burning in the SCNM2 (table 5) and NCNM sites as well as several other sites in the Southwest (L. Bender, unpublished data). Although shrub cover was initially decreased, it typically recovered by the third year postburn. Further, some of the decrease in shrub cover was desirable, because it reflected loss of older, decadent canopies, as shrub volume decreased with burning. The younger growth from resprouting increased the forage quality of these shrubs significantly despite the lessened cover (see later discussion). Preferred browse followed these trends, but remained a small proportion of overall woody browse cover (table 5).

Vegetative protein levels were not assessed on the SCNM2 and NCNM sites, but burning increased protein levels of preferred grasses more than 1.6 times and shrubs more than 2 times that of unburned sites in early spring and summer following late-winter burns, although this effect was lessened by year 2 and absent by year 3 (L. Bender, unpublished data). Similar responses have been seen elsewhere (e.g., DeWitt and Derby 1955; Dills 1970; Einarsen 1946; Grelen and Epps 1967; Hobbs and Spowart 1984; Miller et al. 2005), and result from the freeing of N (Covington et al. 1991; Rau et al. 2009). Burned

sites also greened-up on average 14 days earlier than unburned across several southwestern sites (L. Bender, unpublished data). Fire speeds spring green-up, at least in the first year, because ash warms in the sun, creating a warmer microclimate for plants to germinate or initiate growth (Hobbs and Spowart 1984). Both of these responses increase diet quality of cervids during the critical late-gestation and early-lactation periods.

Burning also has nutritional benefits for elk and deer beyond those seen with mechanical treatments (which can also increase protein levels, as discussed earlier). Properly timed burns increase the quality of diets by removing dead plant material, allowing access to newly emergent forages in early phenological states (Hobbs and Spowart 1984; Severson and Urness 1994). Frequent fire can cause species shifts away from perennial warm-season grasses to forbs (Ford and McPherson 1996), which can also be preferentially increased in abundance by spring fires (Brewer and Platt 1994). Fire also tends to decrease crude fiber of plants, increasing digestibility. Fire therefore results in increased energy available, both within plants (because of increased digestibility) and overall (because of increased plant biomass) (Edwards et al. 2004). Many of these additional effects are more long-lived than the 1- to 2-year protein flush (Bender 2011), and benefit both deer and elk.

Similar to protein responses, live understory cover may be lost with increasing time since burning (fig. 2). Herbaceous responses were lost each year following burning, and lasted approximately 4 to 5 years in arid New Mexico woodlands (fig. 2, table 5) regardless of precipitation received during the winter or growing season. Thus, not only should burns be conducted at appropriate times to provide needed nutritional benefits (i.e., nutrient flush, earlier green-up, increased preferred forage cover) (Bender 2011) to cervids, but fire intervals should be timed to maintain the desired understory response(s) as well. Bender (2011, 2012) recommended that sites with moderate soil productivity be burned every 5 to 7 years, whereas sites should be burned on 10- to 15-year intervals if low productivity soils dominate or if increased woody browse is a management goal. Burning must be done with care, however, as data from several areas indicate that even light- to moderate-intensity (i.e., flame lengths  $\leq 1$  m) burns in thinned PJ can result in substantial mortality

of remaining trees, decreasing residual cover below desired levels. Even when the goal is increased woody browse, periodic burning is necessary to alter successional status of shrubs, increasing palatability and availability.

Additional treatments beyond properly timed prescribed burning may be needed to achieve high coverage of preferred herbaceous forage in understories. In such cases, forage seedings are frequently used to establish palatable forages (Bender 2012). For example, dryland alfalfa (*Medicago sativa* var. Ladak) was seeded in a thinned (to about 30 percent canopy cover) PJ stand at 5 to 10 lb/acre (6 to 12 kg/ha) in mid-summer following a late-winter burn near the SCNM2 site. By fall, cover of alfalfa was 12 percent (SE = 4) on the site, and increased to 20 percent (SE = 6) the next spring. Dryland alfalfa is a highly palatable legume of great nutritional quality, and hence is highly preferred by mule deer and elk. Because it is a cool-season species, it provides needed nutrition during late gestation and throughout lactation, as it maintains quality throughout the warm season. Dryland alfalfa plantings frequently fail because seeds are planted too deep (Bender 2012), and relatively shallow-rooted seedlings are susceptible to freeze-out during their first winter (L. Bender, unpublished data). However, once it or similar seedings are established, they can maintain high cover of extremely high quality forage that responds well to appropriate prescribed burning timing and intervals (Bender 2012).

In areas including livestock grazing, warm-season grazing is a tool that may favor increased cover of cool-season grasses and forbs (Chris Allison, New Mexico State University, Las Cruces, New Mexico, personal communication). Warm-season grazing may also increase the foraging efficiency of mule deer and elk due to the removal of coarse herbaceous cover (Severson and Urness 1994). Understory composition data from the SCNM1 site (table 4), however, suggested that even lightly grazed systems that include cool-season use may not maintain cover of cool-season grasses. More frequent burning of understories may decrease cover of warm-season grasses in favor of greater coverage by forbs, including cool-season species (Bender 2011; Ford and McPherson 1996; Launchbaugh 1964; Wright and Bailey 1980). Very short fire intervals (<3 years), however, can eliminate

most preferred woody shrubs (Bender 2011; Ford and McPherson 1996). Thus, this strategy should be limited to areas such as perennial grasslands that are proximate to PJ woodlands, and not the woodlands themselves, if mule deer habitat is a management goal. Burns should also be in late winter, as late-spring burns may decrease cover of cool-season grasses (Hart 1990).

Other treatments, such as chipping and other mastication treatments, may show little understory response if biomass is left on the ground, particularly in hotter and more arid PJ woodlands (table 7). Juniper wood (240:1) and leaves (55:1) have high carbon (C):N ratios (Bates 1996; Miller et al. 2005), and Bates (1996) found that C:N ratios in soil increased by 32 to 43 percent where western juniper (*J. occidentalis*) slash was deposited. High C:N ratio litter can cause soil microbes to immobilize available N (Schimel and Firestone 1989), slowing decomposition (Miller et al. 1979). Consequently, chip treatments may have lower total nitrogen, phosphorus, organic phosphorus, calcium, and potassium concentrations than untreated, lop-and-scatter, or broadcast burn treatments (Waldrop et al. 2003).

Chipping can also result in lower forest floor enzyme activity, which may slow decomposition rates up to 60 percent (Waldrop et al. 2003). Other work suggests that though chipping and mastication treatments may not be detrimental to N mineralization in the short term, slow decomposition of masticated material and widening C:N ratios may be a concern in the long term (Gottfried et al. 2009). Because decomposition can be influenced by temperature and moisture (Klopatek et al. 1995), these dynamics may be a particular challenge in the hotter, more arid PJ woodlands of New Mexico, west Texas, and northern Mexico. Studies showing positive understory responses to chipping or mastication were in cooler, moister sites, or regions with different precipitation patterns than the CNM site, which has a high (>50 percent) monsoonal index (e.g., Klopatek et al. 1995; Ross et al. 2012).

Leaching of secondary plant compounds from residual chip biomass may also inhibit understory responses. Ramsey (1989) found that phenolic leachates did not affect emergence, but generally inhibited growth, such as root and shoot biomass and tiller numbers, of several grass species (Ramsey 1989). In contrast, leaving

biomass may promote plant growth by increasing soil moisture because of the mulch effect (Benson 2006; Bergman et al. 2014; Brockway et al. 2002). However, this also makes follow-up burning more difficult, particularly in late winter or early spring, because lower chip layers and adjacent soil remain moist. For example, I was unable to burn chipped areas in the CNM site 1 year and 2 years post-chipping during late winter because of saturation of chips in contact with the ground.

A further concern with this treatment is that chipping or mastication treatments may decrease soil temperature in spring (Brockway et al. 2002; Waldrop et al. 2003). This effect is particularly relevant to the annual energetic cycle of mule deer and elk, because it would result in later green-up, delaying availability of early phenology forage during late gestation when nutritional needs increase dramatically (Cook et al. 2004; Verme and Ullrey 1984; Wakeling and Bender 2003).

Most documented responses of preferred forages (other than warm-season grasses) to treatments have been relatively small (tables 5, 7). However, even a small change in plant community composition may result in a significant change in elk or deer habitat quality. For example, it takes only an approximately 10-percent change in diet quality to significantly increase elk productivity (Cook et al. 2004). Diet and diet quality data from pronghorn (*Antilocapra americana*) suggest that even a relatively small increase—5 to 7 percent—in preferred forages may be able to produce such a change in diet quality (L. Bender, unpublished data). Similar changes were seen with cattle diet quality (Holechek and Vavra 1983; McGinty et al. 1983). However, further work is needed to identify what level of change in preferred forage can significantly increase diet quality.

For treatments that remove woodland overstory and horizontal cover, care must be taken to maintain sufficient canopy cover or stem density to meet security and thermal requirements. Inclusion of untreated PJ woodlands was negatively related to home range size of mule deer, indicating that presence of denser PJ increased habitat quality. This beneficial effect was probably because of cover attributes. Previously, Bender et al. (2013) found that the majority of mule deer activity was near unthinned PJ woodlands on the CRLRC. Elk decreased diurnal activities in the Chaco area when

mean high temperatures exceeded their thermal tolerance, with juniper trees being an important diurnal bedding site (Bender et al. 2012a). For example, radio-collared elk in the Oscura site significantly decreased use of portions of home ranges on Chupadera Mesa following prescribed burning that resulted in canopy loss of 70 to more than 90 percent and near elimination of shrubs (at least in the near term). Observations of mule deer in these areas also decreased by more than 90 percent after these burns (L. Bender, personal observation). Additionally, some cool-season grasses are positively correlated with overstory PJ canopy cover (Schott and Pieper 1985), so too much canopy reduction may negatively impact some key foods as well.

In PJ-dominated sites with few other cover options, Bender (2012) recommended a “Rule of 4s” to maintain adequate cover for mule deer when managers are treating PJ stands. This strategy recommends that for each section of PJ (which approximates the average home range size of female mule deer in New Mexico) (Bender et al. 2007a, 2011; Hoenes 2008), one-fourth should remain in untreated (i.e., PJ canopy cover > 60 percent; if existing canopy cover is <60 percent, these areas should be allowed to develop to >60 percent canopy cover) patches of 40 acres (16 ha) or more, one-fourth should be thinned to no less than 30 percent PJ canopy cover, and the remaining two-fourths should be thinned to no less than 10 to 15 percent canopy cover. This would create a mosaic in which one-fourth of each home range is in untreated PJ ideal for security cover, one-fourth is in a structural state that provides both minimal cover requirements and increased forage (30 percent canopy cover), and one-half is in a structural state that provides scattered thermal cover and optimal foraging attributes (10–15 percent canopy cover). At a relative fine scale (i.e., ≤40-acre treatments), these treatment allocations would result in 100 percent of the landscape being suitable for mule deer (Bender 2012).

While designed for the security preferences and needs of mule deer, this strategy also provides abundant summer thermal cover for elk, and probably does not decrease the nutritional carrying capacity of elk range

significantly. In areas of high disturbance, the extra security cover beyond the likely requirements of elk may be useful to avoid unnecessary movements and thus help maintain energy balance and productivity (Shively et al. 2005). Further, distribution of patches or stands should be allowed to shift across the landscape over time as succession, disturbance, or management treatments change the overstory PJ cover, thus maintaining a dynamic landscape.

The potential for enhancing extant PJ woodlands for mule deer and elk is a viable and relevant opportunity for managers of public or private land. These woodlands include almost 10 percent of the total area of New Mexico, making PJ woodlands the most abundant habitat type capable of meeting all forage and cover requirements for mule deer and elk in the State. Though current habitat quality of many woodlands may be relatively low, this may not have been true historically. In several of my study areas, agency and landowner survey data indicate much higher mule deer densities in PJ woodlands historically. For example, mule deer densities in the San Andres and Oscura Mountains were approximately 5 to 20 times greater during the 1970s and 1980s as compared to the early 2000s (Bender et al. 2012b, 2017; White Sands Missile Range: Bender, unpublished data). Similar trends have occurred in the Corona and NCNM sites (Bender et al. 2007).

Density can be a misleading index of habitat quality, as mentioned earlier (Van Horne 1983), but differences of these magnitudes probably at least partially reflect significant changes in habitat quality. While many site-specific factors can affect local deer and elk populations (e.g., disease, drought, overharvest) (Bender et al. 2007), one constant among these sites has been a lack of disturbance appropriate for rejuvenating woody forage, especially fire. Currently, deer and elk are productive in PJ habitats in New Mexico, as reviewed earlier, but not as productive as they can be. Public and private land managers who desire to increase the quality of mule deer and elk habitat in PJ woodlands in New Mexico thus have historical context indicating that habitat quality can very likely be improved substantially.

## MANAGEMENT IMPLICATIONS

1. Needs and opportunity exist to improve the quality of PJ woodlands as foraging habitat for elk and mule deer.
  - Increases in understory, however, do not necessarily equal increases in food.
  - It is likely that relatively small increases in preferred forages can have a significant nutritional benefit.
2. Increased nutrients (protein, digestible energy) need to be provided during late gestation, or beginning in approximately mid-April.
  - Broadcast burning during late winter (March) appears to be the single most effective tool to achieve this goal. Burns can be done following thinning with lop-and-scatter of slash, in previously thinned stands, or in savannas.
  - Agency emphasis on “natural” lightning-driven fire regimes cannot provide a nutrient flush at appropriate times.
  - Piling and burning of slash should be avoided unless piling sites are to be seeded to a highly preferred cool-season forage (i.e., a food plot).
3. Cool-season forages should be encouraged because they provide critical early-phenology forage during late gestation through early lactation.
  - Late-winter burning and warm-season livestock grazing may help increase this component, but more information is needed.
  - Plantings are a further alternative.
4. “Xeriscaping” foraging habitats by providing an abundant and diverse browse component can mitigate the effects of drought in many cases.
  - This is particularly important for habitat for mule deer because their greater diet selectivity emphasizes forbs from herbaceous communities, which are more susceptible to drought.
5. Methods to assess responses of mule deer and elk to management treatments must be closely related to the fundamental influence of habitat.
  - Individual condition, juvenile growth, and juvenile fecundity are appropriate direct metrics.
  - Measures of forage quality of preferred forages and diet quality of mule deer and elk are appropriate indirect methods.
  - Removed metrics (i.e., adult survival, population rate of increase) that can be confounded by numerous environmental and human-caused influences should not be used.
  - Remotely sensed metrics often used to infer forage quality such as the normalized difference vegetation index are not strongly related to nutritional condition of deer or elk and should be avoided unless better indices are developed (Caltrider 2012).
6. Mule deer and elk are different herbivores with differing nutritional and cover requirements.
  - High-quality mule deer habitat will benefit elk.
  - High-quality elk habitat may not benefit mule deer.
7. Management treatments should retain adequate cover for mule deer and elk.
  - Mule deer are likely to require significant areas of high density, high canopy cover for security.
  - Elk require much less cover, primarily for summer thermal constraints.
  - Landscape prescriptions that balance forage and cover such as the “Rule of 4’s” can result in up to 100 percent of the landscape being suitable for mule deer and elk.
  - Cover and forage patches should be allowed to shift across the landscape over time in response to planned or unplanned perturbations, thereby maintaining a dynamic landscape. This subjects all forage to periodic quality-enhancing disturbance over time.

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## Changes in Habitat use of Montezuma Quail in Response to Pinyon-Juniper Canopy Reduction in the Capitan Mountains of New Mexico

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**ABSTRACT**—Montezuma quail (*Cyrtonyx montezumae*) are unique among quail with respect to clutch size, diet, covey dynamics, and habitat use. With the exception of a few notable early studies, there is relatively little information on the ecology of Montezuma quail. Previous research has indicated that one of the primary habitats utilized by Montezuma quail is pinyon-juniper woodlands. Throughout many areas of the southwestern United States, pinyon-juniper woodlands are often targeted for thinning projects. Many studies have been conducted on the amount of tree canopy cover needed by other quail species, however, no data is currently published on Montezuma quail tree canopy cover needs and their response to thinning projects. The goal of this research was to evaluate Montezuma quail responses to common silviculture practices, specifically pinon juniper thinning in the Capitan Mountains of New Mexico. Results of our research indicated that Montezuma quail selected sites thinned to reduce tree canopy cover to 30-40 percent. Selection for this habitat was much higher than selection for the surrounding area which consisted of  $\geq 70$  percent tree canopy cover (Manly-Chesson Selectivity Index=1.68). Overall, this study provides important information for managers planning pinyon-juniper thinning projects in Montezuma quail habitat.

**KEYWORDS**—*Cyrtonyx montezumae*, habitat management, Montezuma quail, population characteristics, tree canopy cover, thinning

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### INTRODUCTION

Montezuma quail (*Cyrtonyx montezumae*), also known as Mearns quail, fools quail, or crazy quail, are a cryptic species and little is known about their ecology, roosting behaviors, or behavioral changes to adjust to habitat modifications. As a result, this quail

species is one of the least understood upland game birds in North America. A possible factor contributing to the paucity of literature pertaining to this species can be attributed to difficulties in locating coveys and abnormal methods required for trapping. Montezuma quail are secretive in nature (crouch and freeze instead of flushing) which can make detection of coveys

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difficult (Hernandez et al. 2006; Oberholser 1974; O'Connor 1936). Additionally, Montezuma quail differ from other quail species in that they primarily dig for subterranean plants such as soft bulbs, sedge tubers, and seeds (Bishop and Hungerford 1965, Harveson et al. 2007, Stromberg 2000). As a result of their different foraging behavior, methods of trapping that work well for other quail species (e.g., funnel traps) typically don't have a high success rate with Montezuma quail (Hernandez et al. 2006, Stromberg 1990). Due to difficulties trapping and locating this species, little is known about their preferred habitat and how they respond to various management practices such as thinning juniper (*Juniperus* spp.).

One way to assess Montezuma quail response to thinning juniper is through radio-telemetry. Aside from Stromberg (1990), Hernandez et al. (2006), and Greene (2013) little data have been collected on Montezuma quail with the aid of radio-telemetry. Montezuma quail are most frequently associated with habitat consisting of pine-juniper woodlands (*Pinus* spp. and *Juniperus* spp.), montane meadows, and desert grasslands (Brown 1982; Hernandez 2004; Holdermann 1992; Holdermann and Holdermann 1998).

It is known that even though Montezuma quail are found in a variety of habitats, there is one factor that all Montezuma quail habitats have in common. Montezuma quail habitats require adequate grass cover (e.g. bunchgrasses) and forbs and grasses that support subterranean foods. It is known that Montezuma quail need bunchgrasses, but little is known about how tree canopy cover influences their habitat selection. Many studies have addressed the amount of tree or herbaceous canopy cover needed by other quail species (Bidwell et al. 1991; Guthery et al. 2000; Johnson and Guthery 1988; Kopp et al. 1998; Rice et al. 1993; Schroeder 1985). However, there is currently no data on Montezuma quail tree canopy cover preferences or how they respond to tree thinning projects. Therefore, studies are warranted to fill in this missing data, which will help to increase our knowledge on the habitat requirements of Montezuma quail and allow us to make informed decisions on thinning projects in areas occupied by Montezuma quail.

The goal of this research is to assess Montezuma quail responses to common silviculture and rangeland

practices in the Capitan Mountains of New Mexico. The primary objective of this research was to assess habitat use of Montezuma quail before and after thinning a pinyon-juniper woodland resulting in a tree canopy cover reduction from 70-80 percent down to 30-40 percent.

## METHODS

### Study Area

The Capitan Mountains represent the northeastern range of Montezuma quail in the greater Sacramento Mountain complex in Lincoln and Otero counties of south-central New Mexico. The 30-year average temperature in the Capitan Mountains range from -5 to 29 °C and annual average precipitation is 6.69 cm (NOAA, 2017). Elevations of the Capitan Mountains range from 1,524m to over 3,353m. The study site is located in the Madrean Encinal and Madrean Pine-Oak Forest and Woodland area of the Apache Highlands and AZ-NM Mountains Ecoregion (TNC Ecoregions). The study site consists of approximately 500 ha within the Capitan Mountains 2 km west of Fort Stanton, New Mexico. Dominant trees on the study site were oneseed juniper (*Juniperus monosperma*), pinyon pine (*Pinus edulis*), alligator juniper (*J. deppeana*) in the higher elevations, and gray oak (*Quercus grisea*) along the rocky hillsides and canyons (USDA Plants). Woody plants in the pinyon-juniper zone include fragrant mimosa (*Mimosa borealis*), skunkbush sumac (*Rhus trilobata*), and at least two species of rabbit brush (*Chrysothamnus* spp.) are noted along draws and dry hillsides. Grass species include sandhill muhly (*Muhlenbergia pungens*), bush muhly (*Muhlenbergia porteri*), mountain muhly (*Muhlenbergia montana*), slender muhly (*Muhlenbergia monticola*), scratchgrass (*Sporobolus asperifolius*), sacaton (*Sporobolus wrightii*), alkali sacaton (*Sporobolus airoides*), Fendler threeawn (*Aristida longiseta*), Arizona three-awn (*Aristida arizonica*), common wolfstail (*Lycurus phleoides*), James' galleta (*Hilaria jamesii*), and tobosagrass (*Hilaria mutica*) (USDA Plants). Also common are tumblegrass (*Schedonnardus paniculatus*), sideoats grama (*Bouteloua curtipendula*), sixweeks grama (*Bouteloua barbata*), black grama (*Bouteloua eriopoda*), Mexican lovegrass (*Eragrostis mexicana*), western wheatgrass (*Agropyron smithii*), foxtail barley (*Hordeum jubatum*), Canada wild rye (*Elymus canadensis*), and blue grama (*Bouteloua*

*gracilis*) (USDA Plants). Common forbs in the study area include Carruth's sagewort (*Artemisia carruthii*), ragweed (*Ambrosia* spp.), Rocky Mountain zinnia (*Zinnia grandiflora*), common mullein (*Verbascum thapsus*), fleabane (*Eriogeron* spp.), scarlet globemallow (*Sphaeralcea coccinea*), flaxflowered ipomopsis (*Ipomopsis longiflora*), scarlet gilia (*Ipomopsis aggregata*), yellow nutsedge (*Cyperus esculentus*), and silverleaf nightshade (*Solanum elaeagnifolium*).

### Tree Canopy Cover Reduction

Within the 500 ha study area, an area consisting of 151 ha was selected for pinyon juniper thinning (fig. 1). The area selected for thinning treatment had a tree canopy cover of approximately 70-80 percent. The rest of the study site had tree canopy cover that ranged from 20-80 percent. Tree canopy cover was assessed using NDVI (Normalized Difference Vegetation Index, Earth Explorer, June and September 2015-2016). Some of the study site had steep slopes and canyons. These areas were not included to undergo the tree canopy reduction due to them having too great a slope to be prime Montezuma quail habitat. Within the area selected for thinning, trees were chosen at random to be harvested by chainsaw. A 5-acre plot was thinned (August 2015) as an example of what was to be completed across the entire area designated for thinning. A commercial tree crew was hired to complete the thinning treatment. During the thinning process, slash was collected and piled, each pile with an estimated size of around 1x2x1m in width, length, height respectively. The slash was burned after it was piled. The thinning was completed in December 2015 and resulted in a tree canopy cover that varied from 30-40 percent.

### Trapping and Handling

To assess how Montezuma quail respond to pinyon-juniper thinning projects, we collected pre-thinning (2014, 2015) and post-thinning data (2015, 2016) from November through May. To adequately survey the 500 ha study site, we used ArcGIS to generate a stratified random sampling pattern consisting of 10 transects. Each of these transects were 500m in length. We traversed each transect with trained pointing dogs every other week in November and again in February (2014 and 2015) to facilitate detection of Montezuma quail. We employed 2 methods of capture for Montezuma

quail. The first method was a modified version of the capture technique described by Brown (1978) using trained pointing dogs and large O-shaped dip nets. The majority of transects were searched within 4 hours of sunset. Once coveys were located during the day, a global positioning system (GPS) was used to mark the covey location. When a dog went on point, a researcher equipped with a large O-shaped dip nets net would try to locate the covey and place the net over them prior to covey flush.

The second and main trapping technique that we used for capturing Montezuma quail involved trained dogs, large hoop nets, and captures at night similar to methods described in Hernandez et al. 2006. Transect searches were conducted in the evenings (2 hours prior to sunset) using the dogs. Once a dog located a covey, the covey was marked on a GPS, and the research team backed out of the area until nightfall. Capture crews revisited the GPS covey locations  $\geq 1$  hour after sunset accompanied by a bird dog. A lighted collar and tracking device (Garmin Alpha TT15) was used for monitoring the dog at night. Search efforts at night were focused in the general area of the original detection site but expanded to approximately a 200-m radius. If the dog was able to relocate the covey and went on point, we used headlamps to locate the exact location of the roost. The researcher in closest proximity to the covey would then attempt to maneuver a hoop net on top of a covey prior to covey flush.

Captured birds were carefully removed from the net and placed individually in 18x28cm soil sampling bags. Each individual bird was weighed, aged and sexed. Captured birds were then fitted with either a VHF (2014, 2015) or GPS (2015) backpack style radio transmitter (4-6 g, GPS: Telemetry Solutions Quail\_5219, Lotek Pinpoint 120, VHF: Advanced Telemetry Systems quail, Frequency range 148-151Mhz), and banded with an individually numbered aluminum leg band. After all data were obtained from the captured birds, we released them at the site of their capture. During 2014, VHF backpacks were attached to record Montezuma quail movements. In 2015, technology advanced so that store on board GPS units were available for quail. Therefore, in 2015, we primarily deployed GPS backpacks. These GPS backpacks were programed to obtain a location every 4 hours between the hours of 06:00 and 22:00. The

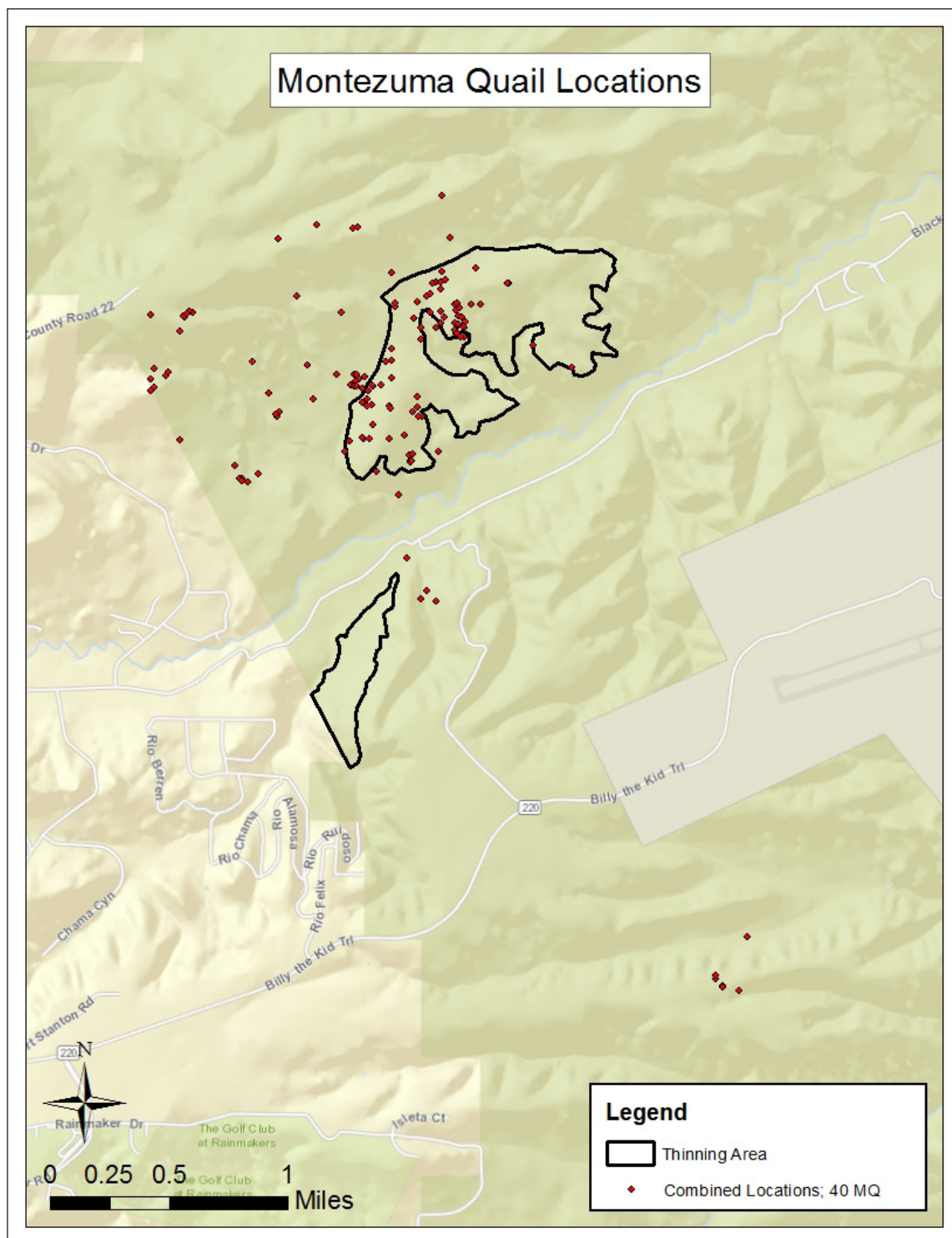


**Figure 1**—Montezuma quail habitat use on the study area prior to commencing the pinyon-juniper thinning project (November 2014 through May 2015).

goal was to have a minimum of 2 birds from each covey fitted with the GPS backpacks, while remaining birds were fitted with VHF backpacks. We determined VHF locations 1 time per day, 5 days a week, every other week. The GPS backpacks were used to assess finer scale daily movements. Once a covey had  $\geq 1$  transmitted bird, the covey was relocated at night using a night-netting technique initially described by Labisky (1959, 1968). Researchers located the radio-marked birds at night with the use of a radio receiver and a yagi antenna (ATS, R4000). When a covey was located, an O-shaped dip net was placed over as many birds in the covey as possible. Previously captured birds were examined and weighed, while newly captured individuals were aged, sexed, and had backpacks and leg bands attached. All trapping activities were conducted in accordance with state (New Mexico Department of Game and Fish Scientific Authorization Permit #3595) and university policy (SRSU IACUC #SPR-0592-525).

### **Spatial Patterns and Habitat Use**

Monitoring of radio-marked birds was carried out with the use of a receiver (ATS R4000, Isanti, MN) and a yagi antenna. Locations of each individual were determined 2-5 times every other week from November 2014 through May 2015. Locations were imported into ArcGIS (ESRI, Redlands, California, USA) where 95 percent and 50 percent MCPs (Minimum convex polygons) were determined for each individual and covey. MCPs were determined to ensure that covey breakup had not been initiated. If the covey was still intact, MCPs were used to assess overall covey habitat use instead of individual habitat use.

We developed habitat use maps for the study site and evaluated habitat selection pre and post thinning, with specific interest in post thinning data (use of unthinned versus areas where thinning projects reduced tree canopy cover to 30-40 percent). A Manly-Chesson Selectivity Index was used to assess selection of the thinned area versus those areas with greater tree canopy cover.

## **RESULTS**

From the pre-thinning sample period, a total of 292 Montezuma quail locations were obtained between November 2014 and May 2015. The locations were acquired from 40 Montezuma quail that we captured from 5 different coveys (fig. 2). Of the 292 locations, 60 percent were within the area designated to undergo the pinyon-juniper canopy reduction. Assessing the covey movements, the 95 percent MCP was  $50.65\text{ha} \pm 14.2\text{ha}$ , and ranged from 2 to 103 ha over the 7 month study period. The 50 percent core use area for the coveys was  $8.65\text{ha} \pm 4.68\text{ha}$ . After the tree canopy reduction had taken place, we were able to capture 53 individuals and equip them with radio-telemetry (GPS) backpacks. The 95 percent MCP was slightly larger than the previous year/pre-thin period. The post-thin 95 percent MCP was  $70.67\text{ha} \pm 36.80\text{ha}$  with a range of 3.2ha to 94.6ha. However, the 50 percent MCP core use area was less than the pretreatment. The post-thinning 50 percent MCP was  $5.36\text{ha} \pm 3.38\text{ha}$ . Overall, the area that had undergone the pinyon-juniper canopy reduction accounted for 85 percent of the total locations obtained from Montezuma quail. Given the assumption of equal use between thinned and non-thinned areas, our Manly-Chesson Selectivity Index was 1.68. This selectivity index centered at 1, which would indicate use at the level of availability. Any number below 1 would indicate less use or avoidance, and any number above 1 would indicate that it is selected for, or used more than available. Our selectivity index indicates Montezuma quail on our study site were selecting for the 30-40 percent tree canopy cover over the  $\geq 70$  percent tree canopy cover surrounding the thinning project (fig. 1). An unintentional finding of this study was that Montezuma quail also utilized the slash piles created on the thinning site. We recorded coveys as well as individuals using these slash piles for cover during loafing and roosting periods. However, use of slash piles could not be fully determined due to the piles being burned as part of the thinning contract.

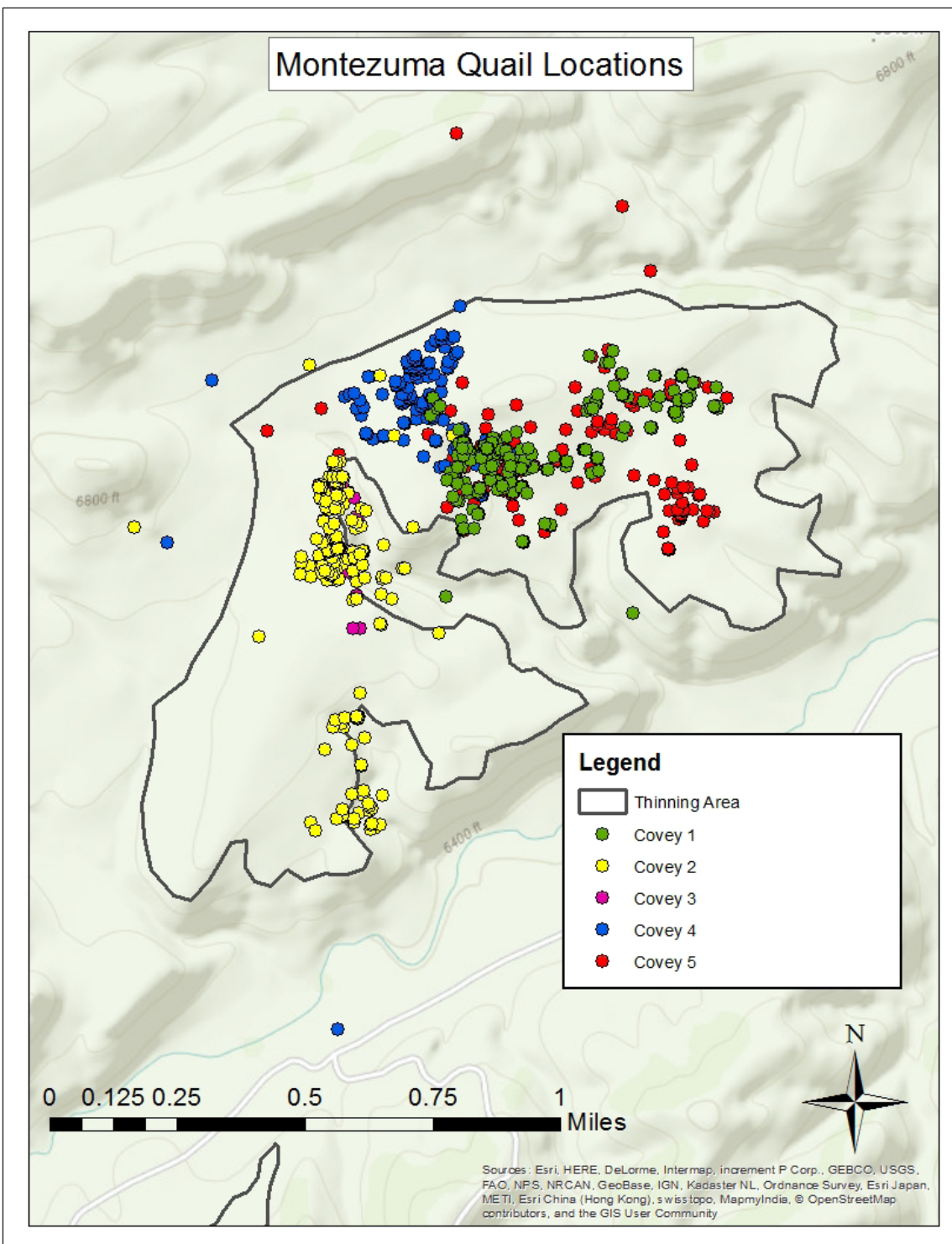


**Figure 2**—Montezuma quail locations from November 2015 through May 2016. Locations indicate habitat use with respect to utilization of a thinning project that reduced the tree canopy cover from  $\geq 70\%$  to 30–40% in the Capitan Mountains near Ft. Stanton, New Mexico.

Each of our coveys monitored during the post-thinning treatment had what appeared to have been exploratory movements of up to 2.4 km, thereby inflating overall home range estimates. However, each covey returned to their core range after these movements. One of the coveys had a very small home range (3.21 ha, SD = 0.01). If this covey is excluded from the overall average, the average home range size for coveys on the study site would increase to 70.68 ha (SD = 24.33).

## DISCUSSION

Our study indicated that by reducing the tree canopy cover of pinyon-juniper woodlands from 70 percent down to around 30-40 percent, use of the area by Montezuma quail increased by 25 percent. Additionally, our selectivity index indicated that areas that underwent thinning were selected for compared to other available habitat. Future studies should incorporate factors to assess Montezuma quail use of slash piles post-thinning.

During the pre-thinning data collection period, the Montezuma quail coveys that we were able to locate on the study site seemed to be relatively dispersed throughout the 500 ha study area. After the thinning was completed, transects that were close to or in thinned areas of the study site were much more productive than those that were in the non-treated portions of the study area. This observation was supported by the radio-telemetry data we received from the coveys. Overall, the majority of the movements were associated with the area that had undergone tree canopy reduction. Each covey did have what appeared to be exploratory movements. During these movements, the covey ventured off of the area where tree canopy reduction had taken place and made a large loop, only to return back to the area that had undergone tree canopy reduction. These exploratory movements increased the 95 percent MCP and resulted in a post-thinning 95 percent MCP (70.67ha) that was greater than the pre-thinning 95 percent MCP (50.65ha). However, the 50 percent core range MCP was smaller post-thinning (5.36ha versus 8.65ha), which could indicate that habitat requirements were met on a smaller scale in the treatment area. Further experiments are warranted to determine if this is actually the case. Overall, our home range estimates during the study period were

similar to those reported in other Montezuma quail literature (Chavarria et al. 2017; Stromberg 1990).

Stromberg (1990) was the first to describe home range and movements of Montezuma quail. However, his sample size (15 radio-marked birds) and the number of relocations (<25) could be a limiting factor to determine true Montezuma quail home range size. Because of our greater sample size, the results of this study should be more indicative of true habitat use during the study period. Leopold and McCabe (1957) suggested 4–10 ha for general home range estimates, but these estimates were based on observations of non-marked coveys in Mexico. One key component that previous studies did not account for was the difference between total home range size and that of core areas. Other studies, namely Greene et al. (2013), recorded movements up to 12.7 km in the Davis Mountains of Texas. The much smaller movements observed during our study compared to those reported in Greene et al. (2013) are likely the result of our study being conducted prior to the breeding season while Greene et al. (2013) incorporated breeding season movements. Another factor that could be influencing movements is the differences in habitat between study sites in the literature. Differences in the Montezuma quail habitats in the Davis Mountains in Texas or study sites in Arizona could influence how much individuals are able to travel in attempts to locate mates. Home range estimates recorded during this study were much closer to those reported by Stromberg (1990) which indicated home ranges of 50.65 ha and core use areas of 1.69 ha. Home range estimates for Montezuma quail are thought to be generally less than scaled quail, but similar to Gambel's quail. Zornes and Bishop (2009) estimated home range sizes for Gambel's quail that were similar to those of Montezuma quail (8–38 ha), while scaled quail home ranges estimates were generally much larger and ranged from 10-882ha.

The small core areas recorded in our study indicate that these coveys are likely showing feeding site fidelity. Leopold and McCabe (1957) suggested Montezuma quail showed feeding site fidelity which could result in limited daily movements. Brown (1978) observed that coveys normally have home ranges <6 ha. Before pairing season, coveys generally do not make as large of movements as they do in the breeding

season; however, movements observed in our study were still larger than the 6 ha suggested by Brown (1978). During our post-thinning study period, home ranges spanned from 3.21 ha to 94.63 ha. Although this is a wide margin between individual covey home range sizes, there appeared to not be major differences attributed to the type of habitat selected between each covey since all were primarily utilizing the area that underwent thinning. Overall, our study indicated a positive response with habitat selection and use associated with reducing tree canopy cover from  $\geq 70$  percent down to 30-40 percent.

## MANAGEMENT IMPLICATIONS

Montezuma quail habitat use can be influenced by tree canopy cover. This could be associated with escape cover. Thick tree canopies likely restrict flight movements, and might influence a Montezuma quail's ability to evade predators. Therefore in some areas, tree canopy reductions may be warranted. Our research provides a baseline for future thinning projects in pinyon-juniper woodlands that contain Montezuma quail. This research indicated that quail spent approximately 85 percent of their time in areas that have tree canopy cover in the 30-40 percent range. Within our study area, habitat that contained tree canopy cover that exceeded 40 percent received relatively little use. The 50 percent MCP core use area resided within the thinning project boundary for all 5 coveys that we tracked during this study. This study yields vital information for managers considering implementing thinning projects in pinyon-juniper woodlands that contain Montezuma quail. By reducing tree canopy cover to 30-40 percent, one can greatly increase habitat that is selected for by this enigmatic quail species.

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# Effects of Piñon-Juniper Woodland Thinning on Avian Communities in the Arkansas River Valley, Colorado

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**KEYWORDS**—piñon pine, juniper, piñon-juniper woodlands, avian occupancy, mastication, woodland thinning

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Piñon-juniper woodlands, the third largest vegetation type in the United States, encompass 40 million ha of western North America, and occur in 10 western U.S. States (Tausch and Hood 2007, Romme et al. 2009, West 1984, Laylock 1999). Piñon-juniper thinning and mastication treatments have been used by public lands managers and private landowners throughout the western United States to mitigate fire risk, reduce encroachment, enhance wildlife and livestock forage production, and improve wildlife habitat. The Bureau of Land Management along with private and public partners have treated approximately 10,000 ha of piñon-juniper woodlands in the Royal Gorge Field Office area of Colorado primarily to reduce fire risk. The influence of tree removal on piñon-juniper bird communities is not well documented, however, and leads to varied, species-specific responses (Bombaci and Pejchar 2016). Piñon-juniper woodlands provide breeding habitat for more than 70 bird species, the highest of any terrestrial ecosystem in the western USA (Balda and Masters 1980), including several at-risk species experiencing long-term population declines (Sauer et al. 2014). The piñon-juniper bird community is unique compared to other ecosystems and includes many obligate species that differ substantially from those of other ecosystems (Paulin et al. 1999, Francis et al. 2011). The purpose of our study was to

determine the effects of thinning efforts on bird populations and communities in piñon-juniper woodlands in southcentral Colorado. We measured multiscale avian occupancy on 29 paired (control/treatment) sites in piñon-juniper woodlands in the Arkansas River valley of central Colorado.

The treatments consisted of 24 masticated (with hydro-ax) and 5 hand-thinned (using chainsaws) sites. Our study allowed for a broad evaluation of bird response to thinning across a variable piñon-juniper landscape. Treatments varied in size and shape and significantly reduced canopy cover from over 30 percent in control sites to less than 15 percent in treatment sites. Most treatment configurations included patchy cuts with clustered trees remaining within the treatment boundary. Some treatments occurred in dense piñon-juniper woodlands, whereas others were in more open sites. Some were located at higher elevations mixing with ponderosa pine and other treatments were prescribed at lower elevation grassland/woodland ecotones. Controls were located adjacent to treatments to reduce site variation between controls and treatments.

We documented avian occupancy (presence or absence) using 10-minute surveys at each of 232 point count stations three times each season in 2014 and

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2015. We used hierarchical multiscale modeling (Program Mark; Pavlacky et al. 2012) to obtain unbiased estimates of landscape ( $\Psi$ ) and local occupancy ( $\theta$ ; i.e., probability of use) in treated and untreated sites for 31 bird species identified in the study. We also measured seven landscape scale covariates to gain insight into the mechanism of treatment effects on avian occupancy. These included mean annual temperature (MAT), mean annual precipitation (MAP), Heat Load Index (HLI; incident radiation integrated across slope, aspect and latitude), elevation, forest cover at 10-ha and 100-ha scales, and time elapsed (years) since treatment was initiated (time elapsed = 0 for control sites; oldest treatments occurred 11 years prior to sampling). At the local scale we measured 11 covariates including percent cover of bare ground, herbs, and shrubs; density and basal area of all live trees, live piñon pines only, and live junipers only; mean tree height; and the standard deviation of tree height (a measure of woodland structural complexity).

We used a three-step modeling procedure to analyze effects of thinning and mastication treatments on local and landscape scale avian occupancy. For each species we first identified a best-fitting structure for modeling detection probability to provide unbiased occupancy estimates. Secondly, we used the best-fitting detection model as we modeled treatment effects on landscape scale ( $\Psi$ ) or local scale ( $\theta$ ) occupancy, or both. Finally, we fixed both detection and treatment structures and

modeled effects of habitat-specific covariates representing mechanisms that drove treatment effects on occupancy.

We identified 77 bird species in the study area during surveys. The spotted towhee was the most numerous and ubiquitous species present in the study area. We also documented the presence of five piñon-juniper obligates including the black-throated gray warbler, Virginia's warbler, gray flycatcher, pinyon jay, and juniper titmouse.

Thirteen species had lower occupancy on treated sites, whereas six species showed increased occupancy on treated sites (table 1). At the landscape scale six species had strong negative treatment effects. These included two piñon-juniper obligates, Virginia's warbler and gray flycatcher. The mature conifer specialist, mountain chickadee, along with a forest generalist, white-breasted nuthatch, and two open-conifer associated birds, Steller's jay and Clark's nutcracker showed strong negative treatment effects. In addition, three more species (Townsend's solitaire of open conifer, western tanager of open conifer, and northern flicker, a forest generalist) showed moderate negative treatment effects at the landscape scale. In contrast, two species showed positive treatment effects at the landscape scale including the mountain bluebird (edge species) and the pinyon jay (piñon-juniper obligate). At the local spatial scale, the black-headed grosbeak,

occupancy) and  $\theta$  (local scale occupancy). Where the 95percent confidence interval overlapped 0 by less than 10 % the effect was deemed strong, whereas moderate treatment effects included betas with confidence intervals overlapping zero by 10-20 percent.

| Occupancy parameter                | Negative treatment effect | Positive treatment effect |
|------------------------------------|---------------------------|---------------------------|
| Landscape Occupancy ( $\Psi$ )     |                           |                           |
| Strong effect                      | Virginia's warbler        | Mountain bluebird         |
|                                    | Gray flycatcher           |                           |
|                                    | Mountain chickadee        |                           |
|                                    | Steller's jay             |                           |
|                                    | Clark's nutcracker        |                           |
|                                    | White-breasted nuthatch   |                           |
| Moderate effect                    | Townsend's solitaire      | Pinyon jay                |
|                                    | Western tanager           |                           |
|                                    | Northern flicker          |                           |
| Local Scale Occupancy ( $\theta$ ) |                           |                           |
| Strong effect                      | Black-headed grosbeak     | Lark sparrow              |
|                                    | Pinyon jay                | American robin            |
| Moderate effect                    | Broad-tailed hummingbird  | Western bluebird          |
|                                    | Ash-throated flycatcher   | Blue-gray gnatcatcher     |

broad-tailed hummingbird, ash-throated flycatcher, and pinyon jay showed significant negative treatment effects. The lark sparrow, American robin, western bluebird, and blue-gray gnatcatcher, in contrast, had elevated local occupancy in treatments.

In a covariate analysis, we found that bird occupancy not only varied by treatment, it was also influenced by landscape and local scale habitat, topographic, climatic, and temporal factors. Each species had a unique set of covariates that were associated with occupancy, but generally, reduced occupancy of forest species was associated with declines in tree density, basal area, and forest cover, whereas occupancy of species associated with more open habitats increased with less forest cover.

Piñon-juniper woodlands are rich in bird species and provide habitat for a suite of high profile conservation species, many of which are experiencing long-term population declines. Our data suggest that mastication and hand-thinning treatments may reduce local and landscape occupancy of several of these species while simultaneously increasing occupancy for species associated with more open habitats and characterized by less urgent conservation needs. Woodland and piñon-juniper obligate species showed the strongest negative responses to treatments. According to Bombaci and Pejchar (2016), woodland birds are “losers” when considering wildlife functional group responses to tree removal; however, this negative response is buffered following less extensive tree thinning. As a group, birds in this study did not respond uniformly to mastication and hand-thinning treatments. Similarly, avian taxa and functional group, as well as treatment type led to divergent responses among birds across numerous studies (Bombaci and Pejchar 2016). Further, our results highlighted the differential treatment effects on occupancy across spatial scales and among and even within species. Pinyon jays, for example, had lower occupancy on treated sites at the local scale; however, occupancy was greater in treatments at the landscape scale. pinyon jays may find treated landscapes suitable for occupancy as long as they contain fairly dense

patches, but within treated forest stands, fragmentation may exceed a threshold that pinyon jays can tolerate for nest and roost habitat selection.

Mastication and hand-thinning treatments may not be needed for fire mitigation in the future as climate driven piñon-juniper tree mortality has increased in the 21st century (Breshears et al. 2005). In the wildland urban-interface, protection of property and life are priorities, and fire mitigation treatments will continue in these areas. Managers may be able to alter treatment prescriptions that favor forest bird habitat such as leaving more live trees and further clustering treatments rather than clearcutting or leaving evenly dispersed savannas (Bombaci et al. 2016, Francis et al. 2011, Pavlacky and Anderson 2001). In our study, canopy cover was reduced beyond what was needed to moderate fire risk. Species such as the pinyon jay and black-headed grosbeak were positively associated with tree density and would benefit if fewer trees were removed in mastication treatments.

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# Identifying and Mitigating Social-Ecological Tradeoffs: Vegetation, Birds, Fuels, and Modeled Fire Behavior in Piñon-Juniper Fuel Reduction Treatments

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**KEYWORDS**—crown fire, invasive species, hydro-ax, *Juniperus monosperma*, *Juniperus scopulorum*, *Pinus edulis*

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Mastication and hand-thinning treatments are increasingly used by land managers as a means of reducing tree cover for fire hazard mitigation and other habitat objectives in piñon-juniper (P-J) woodlands. (Battaglia et al. 2010; Redmond et al. 2013). However, the effects of these treatments on ecological processes including fire, and on a wide range of species, particularly vulnerable P-J obligate birds, are incompletely understood. To address these knowledge gaps, we measured vegetation and fuels, and conducted bird point counts at 232 sites in 29 pairs of 1 to 11-year-old treatments and untreated adjacent controls in P-J woodlands of the Arkansas River Valley, Colorado. We used a suite of statistical approaches including paired t-tests, mixed-effects models, and a principal component analysis to assess treatment impacts on vegetation composition and structure, bird abundance, fuels. We also used BehavePlus (Andrews et al. 2005) to develop fire behavior models to examine expected treatment impacts on fire behavior along gradients of fuel loads and under varying fuel moisture scenarios.

Treatments drove major, persistent ecological shifts relative to controls. Tree cover and canopy fuels were reduced; concomitantly, down woody surface fuels, forbs, and grass cover increased (Coop et al. 2017).

Treatments exhibited rapid, large, and persistent increases in the frequency, richness, and cover of 20 non-native plant species including cheatgrass (*Bromus tectorum*). Exotic plant expansion was associated with reductions in tree canopy, alterations to ground cover (distribution of mastication debris and bare soil), and other treatment activities (Coop et al. 2017). Effective mitigation of nonnative species may necessitate additional pre- and posttreatment control measures. Decreased cover by piñon and juniper in treatments was also closely associated with reduced frequency of observations of a suite of P-J obligate bird species.

Reduced canopy fuels and increased herbaceous surface fuels, including exotic annuals, are expected to alter potential fire behavior through reduced active crown fire probability, but also increased surface fire intensity, flame length, and rate of spread. Modeled fire behavior suggests that under most conditions, treatments will be highly effective at reducing active crown fire risk, but also that treatments generally removed more trees than necessary to mitigate this risk. For example, average canopy cover was 6 percent in treatments but only 29 percent in untreated control units. However, even under fairly extreme 97th percentile conditions, models indicated that sustained

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active crown fire required greater than 30 to 50 percent canopy cover. Retention of more trees within treated sites will likely benefit P-J obligate birds. Fire behavior models also indicated that residual trees within treatment sites, particularly in masticated sites, were still highly susceptible to passive crown fire (torching), due to abundant surface fuels, increased flame lengths, and low tree crown base heights. Models, including simulated treatment modifications, suggest that this remaining crown fire risk could mostly be eliminated through follow-up surface fuel reductions (e.g., prescribed fire) and/or crown base height increases (e.g., pruning). We propose that managers consider additional posttreatment fuels interventions to increase the fire resistance of residual trees, which is anticipated to yield both social and ecological benefit. We encourage managers carrying out P-J fuels reductions projects to explicitly consider potential trade-offs between desired treatment outcomes and unintended ecological impacts, and how these might be mitigated. In areas distant from the wildland-urban interface, and in persistent P-J woodlands without imminent restoration needs, we also question whether or not tree removal treatments may be useful given anticipated climate change impacts to these woodlands (e.g., McDowell et al. 2016).

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# Wildlife Responses to Small-Scale Tree Thinning in Central New Mexico Pinyon-Juniper and Ponderosa Pine Woodlands

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**ABSTRACT**—Small-scale (<5 ha) tree thinning treatments were implemented at two pinyon-juniper (*Pinus edulis-Juniperus monosperma*) and two ponderosa pine (*Pinus ponderosa*) wildland-urban interface woodland sites in central New Mexico. We used paired watershed, Before/After/Control/Impact sampling designs with 3 years of prethinning measurements followed by 5 years of posttreatment measurements. Characteristics of soils, vegetation, and wildlife were measured. Manual thinning treatments reduced small trees, removed firewood, and chipped slash. The treatments reduced wildfire fuel loads, changed the woodlands to more open stands of proportionately larger trees, increased herbaceous understory vegetation canopy cover, and caused some shifts in bird, rodent, and larger wildlife species assemblages. Animal species that prefer tree cover either declined slightly or showed no response to thinning, while species that prefer open habitats increased slightly. Birds showed primarily neutral responses, but several avian species showed positive responses. Rodents showed primarily neutral or negative responses, but dominant pinyon mice (*Peromyscus truei*) declined. Larger native mammals and wild turkeys (*Meleagris gallopavo*) showed negative responses while domestic livestock increased on thinned locations.

**KEYWORDS**—Southwest, pinyon, birds, rodents, mammals, ecology, restoration, climate change

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## INTRODUCTION

Pinyon-juniper (*Pinus* spp.-*Juniperus* spp.) woodlands are one of the most widespread woodland vegetation types across western North America (Romme et al. 2009), and they support a great diversity of wildlife species (Bombaci and Pejchar 2016), including approximately 70 species of breeding birds (Balda and Masters 1980; Paulin et al. 1999), about 60 species of mammals (Chung-MacCoubrey 2005; Finch and Ruggiero 1993), and hundreds of insects and other arthropod species (Frischknecht 1975; Furniss and Carolin 1977; Higgins et al. 2014; Lightfoot et al. 2008; Nyoka 2010). Amphibian and reptile associations are poorly documented (Bombaci and Pejchar 2016), but are probably represented by about 20 species in New Mexico (Degenhardt et al. 2005). Pinyon and juniper provide not only physical habitats for animals, but also food, especially when these plants produce cones and berries. The physical structure of pinyon-juniper woodlands supports a mix of animal species that prefer patchy to open woodlands, and species that require some dense tree structure (Bombaci and Pejchar 2016; Gottfried et al. 1995).

Pinyon-juniper woodlands are widespread at mid-elevations across the West and are composed of three primary ecological stand types: 1) relatively long-term stable and persistent pinyon or pinyon-juniper woodlands on steeper, rocky environments; 2) dynamic pinyon-juniper savannas on well-developed loamy soils; and 3) dynamic wooded shrublands in environments that favor shrubs, but also support pinyon trees (Romme et al. 2009). Pinyon pine and juniper are represented by several species in North America (Romme et al. 2009). The species in central New Mexico are twoneedle pinyon pine (also called Colorado pinyon or in Spanish, piñon; *Pinus edulis*) and oneseed juniper (*Juniperus monosperma*) (U.S. Department of Agriculture 2016).

The temporally dynamic pinyon-juniper savannas of central New Mexico are transitional between grasslands and conifer forests, and consist largely of pinyon-juniper savannas mixed with persistent pinyon stands. The pinyon-juniper savannas have expanded into the lower elevation grasslands over the past 150 years, primarily due to livestock overgrazing of perennial grasses, wildfire suppression, and climate change (Dick-Peddie

1993; Miller and Wigand 1994; Romme et al. 2009). Despite the great extent of pinyon-juniper woodlands across the Southwest, the management of both vegetation and wildlife is still poorly developed due to the regional or landscape-scale variability of pinyon-juniper stand types, the many wildlife species that utilize pinyon-juniper habitats, and the different management approaches used to reduce pinyon-juniper stand densities (Bombaci and Pejchar 2016; Romme et al. 2009).

The historical within-stand overgrowth and landscape expansion of pinyon-juniper, followed by the recent declines in pinyon-juniper stands in response to climate change as atmospheric temperature warming intensifies drought (Gutzler 2013; Gutzler and Robbins 2011; Llewellyn and Vaddey 2013; Petrie et al. 2014; Seager et al. 2008), have created a natural resource management challenge; the overgrown stands are now at risk of drought-induced tree mortality (Breshears et al. 2005, 2015; Clifford et al. 2008; Ganey and Vojta 2011; Gaylord 2014; Gaylord et al. 2013, 2015; Hicke and Zeppel 2013; McDowell and Allen 2015; McDowell et al. 2016; Williams et al. 2013) and increasing high-severity wildfire (Allen et al. 2010; Savage et al. 2013). These conditions all interact to create a critical need for natural resource managers to employ the most effective methods for managing the natural resources of pinyon-juniper and ponderosa pine (*Pinus ponderosa*) woodlands across the West, including their associated wildlife species.

Tree thinning management practices (Agee and Skinner 2005) and associated research about the effects of tree thinning on forest and watershed health and function across the Southwest have focused on commercially valuable ponderosa pine and mixed-conifer stands (e.g., Abella 2009; Abella and Covington 2004; Allen et al. 2002; Friederici 2003; Fulé et al. 2001, 2007; Gaylord 2014; Griffis et al. 2001; Moore et al. 2006; Noss et al. 2006; Overby et al. 2015; Robles et al. 2014; Stoddard et al. 2011; Thomas and Waring 2015). In contrast, relatively little research and information are available on the effects of thinning in pinyon-juniper woodlands (e.g., Baker and Shinnerman 2004; Gottfried et al. 2005; Ross et al. 2012). Tree thinning in ponderosa pine stands has been demonstrated to increase the health and productivity of remaining trees (Keane et al. 2002; Kolb et al.

1998; Ronco et al. 1985; Skov et al. 2005; Wallin et al. 2004; Zausen et al. 2005) and to increase the cover and production of understory vegetation (Abella 2009; Abella and Covington 2004; Matchett et al. 2010; Moore et al. 2006; Ramirez et al. 2008; Stoddard et al. 2011; Thomas and Waring 2015), but thinning effects on plant species composition and species diversity are less clear and generally minor (Abella 2009). Tree thinning also has been found to reduce soil erosion, while often increasing surface water yields from watersheds (Baker 1986, 2003; Keane et al. 2002; Moghaddas 2013; Robles et al. 2014; Stednick 1996; Troendle et al. 2010). How soils and hydrology are affected depends on the method of thinning and the weather conditions during and after thinning (Elliott et al. 1996; Moghaddas 2013; Wyatt 2013).

Tree thinning affects wildlife by changing the physical structure, microclimates and patch pattern of pinyon-juniper and ponderosa pine forest/woodland habitats, and understory vegetation composition and productivity (Allen et al. 2002; Bombaci and Pejchar 2016; Kalies et al. 2009; Noss et al. 2006; Pilliod et al. 2006; Stephens et al. 2012; Thomas and Waring 2015). Reducing tree densities creates open stands of habitat for animal species that prefer less tree canopy cover and more productive understory vegetation (Verschuyl et al. 2011), such as mule deer (*Odocoileus hemionus*), North American elk (*Cervus canadensis*), and others (Thomas and Waring 2015).

Birds tend to be diverse in Southwestern forests and woodlands, and because of their species richness, species-specific habitat preferences, and relatively high abundances, birds are good indicators of the effects of tree thinning treatments on native animal communities (Hutto 1998; Hutto et al. 2014). Tree thinning treatments in Southwestern conifer woodlands have been found to have neutral to positive effects on bird assemblages (Bombaci and Pejchar 2016; Gaines et al. 2010; Hurteau et al. 2008; Hutto et al. 2014; Kalies et al. 2009; Pilliod et al. 2006; Stephens et al. 2012; Verschuyl et al. 2011; White et al. 2013). However, bird responses to thinning treatments tend to be specific to the species or foraging guild, and variable over time relative to weather conditions and vegetation changes (Hutto et al. 2014; Kalies et al. 2009; Pilliod et al. 2006; Stephens et al. 2012; Yegorova 2013).

Small mammal or rodent assemblages also have been found to have positive or neutral responses to tree thinning treatments (Bombaci and Pejchar 2016; McIver et al. 2013; Pilliod et al. 2006; Stephens et al. 2012). Ground-dwelling rodent species tend to respond positively to small-diameter tree removal (Kalies 2010; Zwolak 2009). The presence of downed woody slash following treatments has been found to increase habitat and abundance of several ground-dwelling rodent species (Bagne and Finch 2010; Converse et al. 2006a,b,c; Severson 1986; Stephens et al. 2012). However, the effects of tree thinning projects on wildlife are variable and depend on characteristics of the local environment, the type of tree reduction method used (e.g., mechanical, manual, variable prescriptions), the ecology of each animal species, and time since thinning treatments were imposed (Hutto et al. 2014; Kalies et al. 2009; Pilliod et al. 2006; Stephens et al. 2012).

Tree thinning treatment effects on wildlife across the Southwest have largely focused on ponderosa pine forests (Converse et al. 2006a,b,c; Kalies et al. 2009; Kennedy and Fontaine 2009; Pilliod et al. 2006; Stephens et al. 2012), and less is known about tree thinning effects on wildlife in pinyon-juniper woodlands (O'Meara et al. 1981; Severson 1986). Bombaci and Pejchar (2016) provide a comprehensive literature review of studies addressing the effects of pinyon-juniper woodland tree reduction on wildlife and have found that 69 percent of studies showed no change in overall wildlife abundance. However, they found both positive and negative responses to be more frequent when single species or guilds were evaluated. As tree thinning becomes a common management tool in pinyon-juniper woodlands, our need to understand the environmental effects of thinning on wildlife becomes more urgent.

The research reported in this article is part of a larger research project on effectiveness of tree thinning in the Estancia Basin, New Mexico. The Estancia Basin Watershed Health, Restoration and Monitoring Program (Watershed Restoration Program) is a cooperative consortium of local government agencies that acquired funding to conduct forest thinning projects on private lands in the Manzano Mountains of central New Mexico. The goals of the Watershed Restoration Program are to reduce the threat of catastrophic

high-severity wildfires and to improve forest and watershed environmental health and function. The tree thinning treatments follow prescriptions developed by New Mexico State Forestry that emphasize hand thinning practices with low environmental impact. Large-diameter wood is removed and used for firewood, and small-diameter wood or slash is chipped and spread over the soil surface to reduce soil erosion potential. The individual project areas tend to be relatively small in size, ranging from around 2 to 12 ha, typical for wildland-urban interface (WUI) tree thinning treatments (Martinuzzi et al. 2015; Matchett et al. 2010).

Under this program, we developed and implemented an experimental monitoring study to evaluate the effects of small-scale (<5 ha) tree thinning, chipping, and wood chip distribution, in pinyon-juniper and ponderosa pine woodlands in the Manzano Mountains. Effectiveness monitoring for the experimental tree thinning provides data appropriate to evaluate trends in the management goals relative to forest thinning treatments. The objectives of the restoration effectiveness monitoring were (1) to evaluate the success of tree thinning treatments relative to decreasing excess small trees and fuels; (2) to reduce high severity wildfire and enhance woodland watershed function, health and growth of remaining trees, and understory herbaceous vegetation productivity; and (3) to improve wildlife habitat and animal diversity and abundance. Here, we report on wildlife-specific research questions developed for the project: (1) How do small-scale tree thinning treatments change wildlife habitat (overall physical structure and vegetation species composition), and (2) how does small-scale tree thinning affect wildlife species composition and abundance? Wildlife addressed in this study includes diurnal birds, nocturnal rodents, and medium-sized to large mammals.

## METHODS

### Study Sites

The study area was on the eastern slopes of the Manzano Mountains 10 to 40 km northwest of Mountainair, New Mexico (fig. 1). We used a Before/After/Control/Impact, or BACI, experimental monitoring design, where thinning treatment impact was evaluated by comparing results for paired control and treated catchments or watersheds (Green 1979;

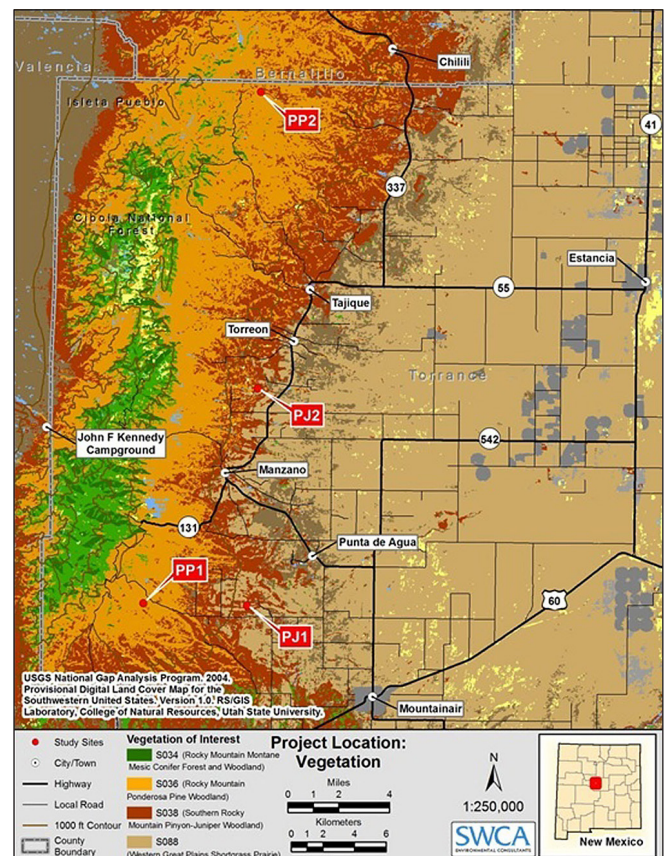


**Figure 1**—Southwest Regional Gap Analysis Project land cover map of the study region and location of study sites on the eastern slopes of the Manzano Mountains, Torrance County, New Mexico.

Neary 2016; Quinn and Keough 2007). We installed two paired watershed sites in ponderosa pine forest and two paired watershed sites in pinyon-juniper woodland in the study area in fall 2007. This study was conducted over an 8-year period, with 2.5 years of pretreatment measurements from 2008 through 2010, and 5 years of posttreatment measurements from 2011 through 2015. Two ponderosa pine study sites, PP1 and PP2, were located in the Arizona/New Mexico Mountains, Montane Conifer Forests ecoregion (U.S. Environmental Protection Agency Ecoregions) (Griffith et al. 2006) and within the Southern Rocky Mountain Ponderosa Pine Woodland (Southwest Regional Gap Analysis Project Land Cover) (Lowry et al. 2007), or Lower Montane, Ponderosa Pine-Gambel Oak Series vegetation type (Dick-Peddie 1993: 66), at about 2,300 m in elevation. Two pinyon-juniper sites, PJ1 and PJ2, were located in the Southwestern Tablelands, Central New Mexico Plains ecoregion (Griffith et al. 2006) and within the Southern Rocky Mountain Pinyon-Juniper Woodland (Lowry et al.

2007) or Pinyon-Juniper Woodland—*Pinus edulis-Juniperus monosperma*/S/*Bouteloua gracilis*—vegetation type (Dick-Peddie 1993:89), and both sites were at about 2,100 m in elevation. Pinyon-juniper woodlands of the region and both pinyon-juniper study sites were within the pinyon-juniper savanna type described by Romme et al. (2009). Refer to Dick-Peddie (1993) for details on floristic composition, climates, and ecological succession within those vegetation types.

Domestic livestock, primarily cattle but also some horses, were present at three of the study sites, but not at one ponderosa pine site (PP1). All sites except PP1 were lightly to moderately grazed by livestock (*sensu* Holechek et al. 1999); the two pinyon-juniper sites had similar levels of moderate grazing (~40% utilization), and the PP2 ponderosa pine site had light grazing (<30% utilization). All control and treatment watersheds per site were exposed to the same amount of grazing by the same animals.

The selection of paired watershed study sites was subjectively based on (1) landowner permission from a set of available landowners involved in the private lands tree thinning program, and (2) by selecting adjacent paired watershed locations representing similar topography, soils, and vegetation (using topography and soils maps and aerial images), ranging in size from 2 to 3 ha each. A mini-meteorological (mini-met) station, a Parshall flume, a soils and vegetation measurement plot, and a wildlife and vegetation measurement plot were located near the center of each of the eight paired watersheds, and are described later. One set of ponderosa pine and pinyon-juniper study sites (PP1 and PJ1) was located within the same larger scale watershed, and the other set of sites (PP2 and PJ2) was located within another larger scale watershed. Each site, PP1, PP2, PJ1, and PJ2, consisted of two adjacent paired watersheds, one randomly selected for the thinning treatment and one left alone as a control, for a total of eight study watersheds.

The original PP1 site that was established in 2007 burned in the high-severity Trigo wildfire that burned the eastern slopes of the Manzano Mountains in May 2008, and data from that site are not addressed here. The new PP1 site was established in fall 2008, farther to the north in a different watershed. Below we refer

to the four primary study locations with paired watersheds as sites, and we refer to each of the eight small paired study watersheds as watersheds (entire paired hydrologic catchment drainages), treatments (control or tree reduction treatments applied to each paired watershed). With each treatment there are smaller study plots that are specific measurement plot locations where various data for plants and animals were collected.

## Regional Climate

The project area is located within the Arizona/New Mexico Mountains Ecoregion (Griffith et al. 2006). The climate across the project area is semiarid temperate and characterized by winter and summer rain patterns, a wide range of diurnal and annual temperatures, and low relative humidity. Most precipitation tends to occur in July and August from summer convective storms. The mean annual precipitation (including water from snow) for Mountainair (1902–2015) is 36.6 cm, average total snowfall is 69.6 cm, average maximum annual temperature is 19.7 °C, and average minimum air temperature is 2.0 °C (Western Regional Climate Center 2016). Mountainair is at an elevation similar to the pinyon-juniper sites and there were no long-term meteorological stations at elevations similar to the ponderosa pine sites.

## Measurement Plots

Three different environmental measurement study plot designs were installed on each of the eight watersheds. The study plots were subjectively located within the center of each watershed, such that measurement plots were at least 100 m apart, and located on similar topography, soils, and vegetation structure and composition. Tree measurements were made following the Forest Service, U.S. Department of Agriculture (Forest Service) Forest Inventory and Analysis (FIA) plots and protocols (Forest Service 2005). Rangeland monitoring plots and protocols (Herrick et al. 2005) were installed and used for measuring understory vegetation. The rangeland monitoring plots were approximately the same overall size and dimensions as the FIA plots, and both plot types were overlaid on each other to provide combined tree and understory vegetation measurement plots. Soil variables also were measured from sampling locations based on the overlaid plot design,

and those overlaid plots are hereafter referred to as “vegetation/soils plots.” Tree data presented here were measured on the vegetation/soils plots.

A separate wildlife and vegetation (wildlife habitat) measurement plot (hereafter, “wildlife plots”) was also installed within 20 m of each of the eight vegetation/soils plots. The wildlife plots had the same general topography, soils, and vegetation as the adjacent vegetation/soils plots. Each wildlife plot was 50 m × 50 m and composed of a 6-station × 6-station grid of 36 permanently marked rodent trap stations and thirty-six 1-m<sup>2</sup> vegetation measurement quadrats at 10-m intervals (fig. 2).

## Variables Measured

### Weather

An automated mini-met station (WatchDog model 2425, Spectrum Technologies, Aurora, Illinois) was installed in an open location adjacent to the vegetation/soils plot at each of the eight watersheds (i.e., four BACI sites) and programmed to record data at 1-hour intervals continuously year-round. Each of the eight mini-met stations was enclosed in a barbed-wire fence to prevent livestock from damaging the equipment. Solar-powered electric wires were attached to the fences at the ponderosa pine sites to prevent American black bears (*Ursus americanus*) from damaging the equipment (damage occurred without protection). Each mini-met station was equipped with a tipping bucket rain gauge, an ambient temperature recorder, and a soil moisture probe and temperature probe at 10 cm below the soil surface. Graduated cylinder rain gauges also were installed at each mini-met station to provide backup rainfall data. Graduated cylinder rain gauges contained small amounts of mineral oil to prevent evaporation of precipitation. Cumulative precipitation amounts in the rain gauges were recorded every month, and all mini-met station data were offloaded monthly year-round.

### Soils, Vegetation, and Hydrology

Detailed measurements on soils (texture, chemistry, water content, surface stability, and erosion), vegetation (plant species composition, canopy cover, and standard stand metrics), and hydrology (surface

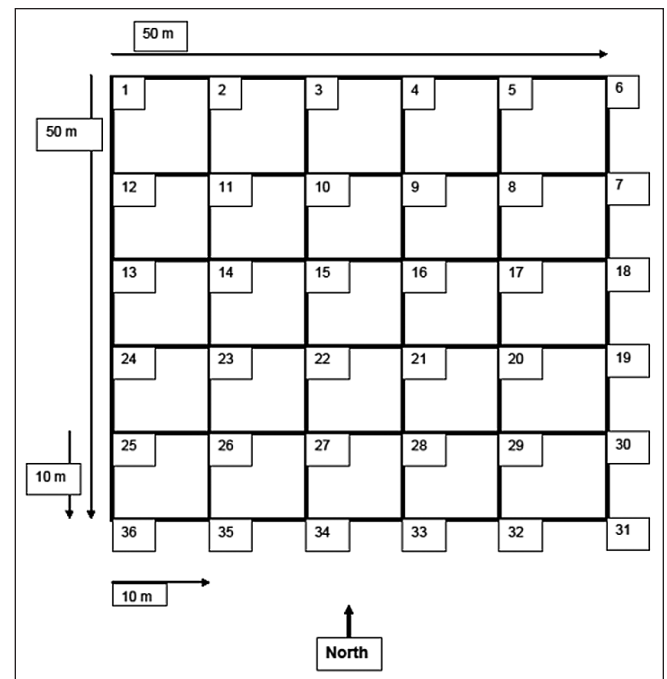


**Figure 2**—Diagram of a wildlife and vegetation measurement plot. Numbers 1 through 36 are 1-m<sup>2</sup> vegetation quadrat numbers. Components of the diagram are not to scale for ease of viewing.

runoff) were taken, but soils and hydrology data are not reported in this article. Some summary data on vegetation are presented here, because vegetation is a primary component of wildlife habitat.

### Vegetation—Trees

All trees on both the pinyon-juniper and ponderosa pine plots were tallied on each of three 14.6-m-diameter tree subplots of each individual study plot. All trees were identified to species and tagged for future reference. All tree measurement protocols followed the standard methods in the FIA guidelines (2005). Tree diameter was measured by using a diameter tape at diameter at breast height (d.b.h.) for ponderosa pine trees or diameter at root collar (d.r.c.) for pinyon and juniper. Along with diameter, tree stand structure, species composition, and tree health were measured on all four FIA tree plots, on each of the eight watersheds. Tree basal area was measured by point sampling techniques (Avery and Burkhart 2002) with a basal area factor of 20. These measurements were taken on all plots in 2008 prior to thinning treatments, and again in 2011 and 2015 following treatments. Measures of tree health, including crown dieback (mortality percentage

of the canopy of each tree), and tree mortality were recorded for all trees across all watersheds once each year in September from 2008 to 2015.

### ***Vegetation Vertical Canopy Cover***

The amount of vertical vegetation (herbaceous and tree) above the ground surface to a height of 2 m was measured with a canopy structure pole as described by Herrick et al. (2005) and made from 2.5-cm-diameter white polyvinylchloride (PVC) pipe. The 2-m pole was partitioned into four consecutive 0.5-m-tall subsections, and each of the four subsections was further divided into five 10-dm subsegments alternating black (black vinyl tape) and white, for a total of twenty 1-dm subsegments in vertical sequence above the ground surface. One biologist held the pole in a vertical position at 10 permanently established measurement points that were located at 10-m intervals on each wildlife plot. The measurement points were at the rodent trap and vegetation quadrat locations 10, 15, 22, 27, 34, 14, 15, 16, 17, and 18 on each wildlife plot (see fig. 2). Measurements were taken once each year from 2010 through 2015 at the end of the summer growing season. The hollow PVC pipe pole was always placed over the permanent rebar marker stake at each point. A second observer stood 10 m away, at points 3, 10, 15, 22, and 27 (north to south view), and at points 13, 14, 15, 16, and 17 (west to east view) (see fig. 2) and recorded whether each of the 1-dm subsegments was obscured 50 percent or greater by vegetation or tree foliage, stems, and branches. An overall visual obstruction average score was then calculated for each 0.5-m segment of the pole over all 20 sampling points per plot.

### ***Tree Crown Canopy Cover***

Tree crown canopy densities were measured with a spherical densiometer (Cook et al. 1995) at the same 11 measurement/observation points where vertical canopy cover was measured. Given that densiometer readings often differ among observers (Cook et al. 1995), the same observer took all densiometer measurements from 2010 through 2015 to reduce observer error. The spherical densiometer also is known to overestimate canopy density due to the shape of the mirror (Cook et al. 1995), but the primary objective of measuring the tree canopy densities in this study was to directly compare the canopy densities of paired

plots over time, so that error was consistent. Reducing observer bias and taking 11 measurements from each plot provided us with a useful tool to compare changes in tree canopy densities over the study plots and over time, despite known measurement problems with the spherical densiometer.

### ***Pinyon Cone and Seed Production***

Pinyon is a mastling plant species where individual trees collectively produce large seed crops infrequently (every several years) on an irregular basis (Janetski 1999). Given the uncertainty of cone production, we did not initially design sampling protocols to measure cone crop production. From 2008 to 2015, pinyon trees produced cone crops at our study sites in 2009 and again in 2015. In 2015 we attempted to measure cone production by visually estimating the percentage of each tree's canopy containing cones from all trees in each of twenty-five 10-m × 10-m plot cells of each wildlife plot (see fig. 2). The same observer recorded cone production from all eight study plots in September 2015.

### ***Understory Vegetation and Ground Surface Cover***

Vegetation growing below tree canopies and cover of soil surface features (e.g., litter and duff, rocks, biotic crusts) were measured in late September each year from 2010 through 2015 from the permanent 1-m<sup>2</sup> quadrats located at each of the 36 permanently marked rodent trapping stations on each wildlife plot (see fig. 2). All herbaceous plant species, cacti, and woody shrubs were measured on each quadrat. The total canopy cover and maximum height (cm) of each species was measured per quadrat. Vegetation quadrat data were also categorized by growth form (i.e., grass [graminoid], forb, succulent, and woody shrubs and subtrees such as oaks [*Quercus* spp.]) and life history (annual or perennial). We followed the taxonomic classification and common names of plants provided by the U.S. Department of Agriculture (2016) PLANTS Database. Voucher plant specimens were photographed, collected, and identified in the laboratory as needed to provide accurate species-level identifications.

In addition to vegetation, soil surface cover categories were measured on the quadrats, including the percent cover of bare soil, leaf litter (and dead and downed

woody material <2 cm diameter), dead and downed wood (>2 cm diameter), rock, and biotic soil surface crusts (blue-green algae, algae, lichens, and moss growing on the soil surface). Measures of wood chip ground coverage resulting from tree thinning and chipping of small branches and foliage were added in 2011 following the tree thinning treatments.

## Wildlife

Birds, rodents, and medium-sized to large mammals were measured on the study sites to evaluate the effects of tree thinning on the species composition and relative abundances of all species over time. Birds and rodents were sampled from each site twice each year, once in the spring (May) and once in the late summer (September) from 2008 through 2015. Automated wildlife cameras or camera traps were installed on the wildlife study plots and run continuously year-round, from 2012 through 2015.

Birds were sampled by fixed-center point counts (see Ralph et al. 1995) from one sample point located at the center of each wildlife plot (see fig. 2). The woodland trees blocked the view of observers, so no distance limits (e.g., fixed-distance circular plots) were imposed on observations, and many observations were by call rather than sight, without a center point-to-bird distance measured or estimated. An experienced bird observer recorded all birds visually sighted or heard calling (or both) during a 20-minute period during the early morning hours for three consecutive mornings at each site, once in May and once in September of each year. Efforts were made to record each bird only once during each 20-minute observation period, but because individuals were not marked, a single individual may have been recounted on each of the 3 days.

Bird counts by species over each 3-day seasonal sampling period were averaged to provide a count per day per species for each period to account for potential multiple daily recounts of individuals. Some bird species were routinely detected calling from great distances or flying over at high altitudes and were not utilizing the local study site habitats. Those species are reported in the Results and were excluded from data analysis because they were not present on study plots. The May observations were intended to capture territorial breeding and nesting birds, whereas the September

observations were meant to capture migrating and local resident birds. Bird species common names, Latin names, and taxonomic classification followed the American Ornithological Society's seventh-edition checklist of North American birds (American Ornithological Society 2016).

Rodents were sampled using 23-cm aluminum live-traps (H.B. Sherman Traps, Inc., Tallahassee, Florida). Each wildlife sampling plot consisted of 36 permanent rodent trap stations at 10-m intervals along the grid pattern (see Measurements and fig. 2). Each rodent trap was fitted with a 30.5-cm section of white PVC roofing gutter, placed upside down over the top of each trap to provide protection from rainfall and the sun. Each trap was baited with oatmeal and also contained cotton to provide captured animals with nesting material. Rodent trapping was conducted twice each year in May and in September at the same time that bird point counts were conducted. Rodent traps were placed on each wildlife plot for 3 consecutive nights (days) and checked early each morning for the presence of animals.

All animals captured were identified to species, gender, and age (juvenile or adult) and weighed to the nearest gram with a spring scale (Pesola, Schindellegi, Switzerland). Each captured animal was temporarily marked with a Sharpie® permanent marker on the upper chest area; blue was used on control plots and red on treatment plots. The color markings wore off over the 3- to 6-month intervals between trapping sessions. All recaptured animals were recorded as recaptures so they could be differentiated from new animals captured on subsequent nights. Persons handling the traps and rodents took safety precautions to avoid exposure to the Sin Nombre virus (Centers for Disease Control and Prevention 2016) and hanta virus from deer mice (*Peromyscus maniculatus*).

Some mice (*Peromyscus* spp.) were difficult to identify. In such cases, key body measurements (ear length, hind foot length, body length, and tail length) were measured and recorded to the nearest millimeter, and photographs were taken to provide dorsal and lateral views for fur colors and patterns. Those data were later evaluated to verify questionable field identifications. Traps and covers were removed from study plots for the remainder of each year. Since animals were

marked, only data for newly captured individuals over each 3-night trapping period were used for analysis. Recaptured animal counts were omitted to provide nonredundant count data for each individual per each 3-night seasonal sampling period. Rodent names and taxonomic classification followed Findley et al. (1975).

Measurements of other larger mammals were not originally included in the study design, but were added in 2012, 1 year following thinning treatments. Medium-sized to large animals (mostly mammals) were sampled from each wildlife plot by permanently mounted, automated wildlife cameras or camera traps (Truth® Cam 35, model 63010, Primos Hunting, Flora, Mississippi). One camera was mounted on a 2-m-tall steel T fence post, approximately 1.5 m above the ground surface, facing a relatively open area on each wildlife plot. Cameras were left on year-round, with settings for low sensitivity and a 1-minute pause between photographs (to avoid many photographs of wind-blown vegetation). Images stored on secure digital camera cards were offloaded once each month from each camera. Qualified wildlife biologists reviewed images and recorded all animals photographed from each plot for each month. Each discernible individual was recorded only once if multiple photographs of the same individual were taken during a given time period. All animals, including domestic livestock, were recorded. Data were summed for each individual of each species from each camera over each calendar year. Mammal names and taxonomic classification followed Findley et al. (1975).

### ***Tree Thinning Treatments***

One watershed of each pair was randomly selected to be treated by thinning trees and the other watershed of the pair was left intact as a control for comparison. New Mexico State Forestry provided the standard tree thinning prescriptions for the region and forest stand types. A summary of the thinning prescription (in English units) for the pinyon-juniper watersheds was a reduction in conifer tree basal area to an average of 40 to 60 ft<sup>2</sup>/acre (9.2–13.7 m<sup>2</sup>/ha), leaving a variety of tree size classes, groups of trees, and random open areas. The thinning prescription for the ponderosa pine watersheds was a reduction in tree basal area to an average of 60 ft<sup>2</sup>/acre (5.6 m<sup>2</sup>/ha) and removal of all small trees under the driplines of remaining large trees,

leaving a variety of size classes in groups of two to seven trees, and random openings. Both prescriptions additionally called for selectively removing diseased and insect-damaged trees, but leaving one to five dead standing trees and one to five logs at least 12 feet (3.6 m) long, per acre for wildlife habitat, and remaining larger diameter wood to be removed for firewood. Small-diameter wood or slash was chipped or masticated and the chips spread to an average depth of 2 inches (5 cm), but no greater than 6 inches (15 cm), with no chips under the driplines of remaining trees.

The tree thinning crew was composed of workers who also performed the same thinning treatments at other locations in the area as part of the Watershed Restoration Program. The same crew thinned all four treatment watersheds for this study, and all four thinning treatments were inspected and approved by New Mexico State Forestry for compliance with prescriptions. Thinning treatments for all four watersheds were completed between November 2010 and March 2011.

### **Data Analysis**

Measurement data from environmental variables were used to compare variable values from each of the four paired control and treatment watersheds on a per site basis (i.e., each watershed pair per site was analyzed and compared separately from the other sites). There was not enough site replication to use sites as statistical replicates. Data were analyzed in different ways depending on the variables measured, sampling designs, and purposes for analysis. Statistical testing for differences in variables between paired control and treatment watersheds was performed with both parametric two-tailed paired t-tests for mean values of variables measured with suitable within-plot sample replication (e.g., understory vegetation canopy cover and ground surface cover variables), and nonparametric Chi-Square goodness of fit tests were used for nonnormally distributed animal count data. The similarity of plant and animal species compositions among all of the study plots since 2010 was evaluated with nonparametric multivariate hierarchical group-average cluster analysis (McCune et al. 2002), where the relationships among the eight watersheds were based on the similarity of species counts (animals) or canopy cover (plants). SAS 9.4 statistical software (SAS Institute Inc., Cary, North Carolina) was used for those analyses.

All statistical testing for differences in mean values of variables between control and treatment watersheds was set at a standard p-value of 0.05 or less ( $\leq 5$ -percent probability of data values supporting the null hypothesis of no difference between treatments). We consider a test criterion of  $p \leq 0.05$  to be excessively precise for field-measured data and therefore highly conservative. Given such conservative testing, we did not apply further conservative Bonferroni adjustments for cases of multiple testing (e.g., multiple species tested separately for the same control versus treatment differences). Only data pertinent to the effects of tree thinning on wildlife are addressed here, and in some cases only data for key periods of time relative to the before- and after-treatment dates, such as the years 2010 (1 year pretreatment), 2011 (1 year posttreatment), and 2015 (5 years posttreatment), are presented here.

## RESULTS

### Weather Conditions

An extreme drought event occurred across the region and most of the Southwest during the 8-year period during which this study was conducted (U.S. Drought Monitor 2016). The study area was not in drought in 2008, 2009, and 2010, but then went into extreme drought in 2011, severe drought in 2012 and 2013, abnormally dry in 2014, and no drought in 2015. Annual precipitation amounts for 2009 through 2015 averaged over each year from hourly data from each of the four mini-met stations at the two ponderosa pine sites and from each of the four mini-met stations at the two pinyon-juniper sites are presented in fig. 3. American black bears damaged the weather stations several times at both ponderosa pine sites in 2008, creating gaps in weather data for that year, so we chose not to report annual precipitation and ambient temperature averages for 2008. The solar-powered electric fences placed around the stations in 2009 reduced bear damage in later years.

Annual precipitation averaged from hourly recordings for each year, over both mini-met stations (control and treatment watersheds) at each of the four study sites, was slightly above the long-term average for the region in 2009 through 2010; then drought conditions occurred from 2011 and 2012, with near-average

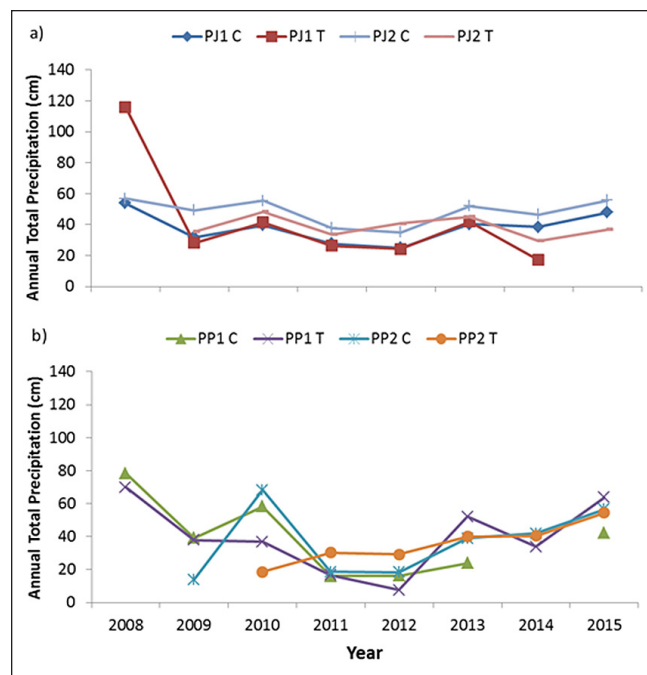


**Figure 3**—Total annual precipitation values for (a) the two pinyon-juniper (PJ1 and PJ2) and (b) the two ponderosa pine (PP1 and PP2) sites, 2008–2015. Data gaps were due to rain gauge outages, primarily from American black bear damage.

precipitation in 2013 through 2015. Annual average ambient temperatures for 2009 through 2015 averaged over each year from hourly data from all four weather stations at the two ponderosa pine sites and from all four weather stations at the two pinyon-juniper sites are presented in fig. 4. Ambient temperatures generally increased steadily from 2009 to 2015, with a slight decrease in 2013.

### Trees

#### Basal Area

In 2010 prior to the thinning treatments, tree basal area averaged around 200 ft<sup>2</sup>/acre (46 m<sup>2</sup>/ha) at the ponderosa pine sites and around 140 ft<sup>2</sup>/acre (32 m<sup>2</sup>/ha) at the pinyon-juniper sites with no differences between control and treatment watersheds. Most of the trees on all watersheds were smaller diameter trees located close together. In 2011, tree basal area was reduced to 85 ft<sup>2</sup>/acre (19.5 m<sup>2</sup>/ha) on the treated ponderosa pine watersheds, while at the pinyon-juniper sites basal area was reduced to around 40 ft<sup>2</sup>/acre.

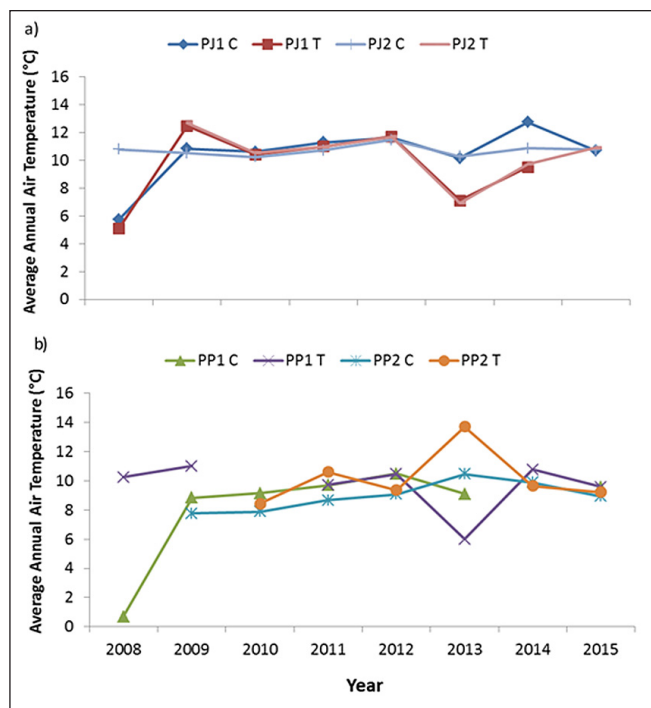


**Figure 4**—Average annual temperatures for (a) the two pinyon-juniper (PJ1 and PJ2) and (b) the two ponderosa pine (PP1 and PP2) sites, 2008–2015. Data gaps were due to temperature gauge outages, primarily from American black bear damage.

## Stand Structure

Measurements of tree diameters (DBH and DRC) were taken in 2010 prior to thinning treatments and again in 2012 following treatments to assess the impacts of the treatments on stand structure. Most trees across both the pinyon-juniper and the ponderosa pine watersheds were within the smaller size classes (>5 inches [12.7 cm], 5–9 inches [12.7–22.9 cm]) with the majority of the trees smaller than 5 inches in diameter, before the thinning treatments. The thinning treatments then created more of a uniform distribution of trees across all size classes with the number of trees less than 8 inches (20.3 cm) in diameter greatly reduced.

Historical forestry and land management practices on both the control and treated plots had resulted in very few if any large-diameter old-growth trees (unpublished data). The remaining trees on the treated pinyon-juniper plots were more evenly distributed in the 5- to 9-inch and 9- to 12-inch (22.9–30.5 cm) diameter classes with only a few juniper trees in the 12- to 18-inch (30.5–45.7 cm) and 18- to 24-inch (45.7–61.0 cm) size classes. There were no trees in the <5-inch

class left on any of the pinyon-juniper measurement plots. The remaining trees on the ponderosa pine plots were more evenly distributed in the 5- to 9-inch and 9- to 12-inch diameter classes with a small proportion of trees less than 5 inches and greater than 12 inches.

## Tree Mortality

Tree mortality measurements taken across both the ponderosa and pinyon-juniper plots showed less than 5 percent mortality, with one exception, from 2008 through 2015 with no significant differences between the control and treatment plots. During the height of the regional drought in 2012, however, there was an extensive pinyon ips (*Ips confusus*) bark beetle outbreak across the region that killed large stands of pinyon trees. Pinyon trees on both the pinyon-juniper treatment and control plots were impacted by this outbreak but not at a significant level. One of the pinyon-juniper plots had more than 5 percent mortality in 2012; beetles killed a cluster of five pinyon trees on a treatment plot. Even though the area had been thinned to reduce competition between trees, the number of beetles inhabiting the surrounding area overwhelmed this cluster of trees. Other pinyon trees within the treatment and control plots were impacted by the beetles as well, but they were resilient enough to survive the attacks. In contrast to the pinyon-juniper sites, the ponderosa pine sites on average had less than 2 percent individual tree mortality over the study period, with no significant differences between control and treatment plots.

## Lower Tree Vertical Canopy Cover

Lower tree vertical canopy cover from ground level to a height of 2 m showed a general reduction in canopy structure on the plots that were thinned in 2011 as compared to control plots (fig. 5). In 2010, before tree thinning, pretreatment plots at each of the two ponderosa pine sites had significantly lower (paired t-test,  $n = 12$  for all tests) canopy structure; PP1 had significantly greater cover structure on the control plot ( $p = 0.02$ ), while PP2 had significantly greater cover structure on the pretreatment plot ( $p = 0.006$ ). Pretreatment and control plots at the two pinyon-juniper sites were not different. In 2011 after tree thinning, lower tree canopy cover structure was significantly lower on treated plots at both ponderosa pine sites (PP1 site,  $p = 0.001$ ;

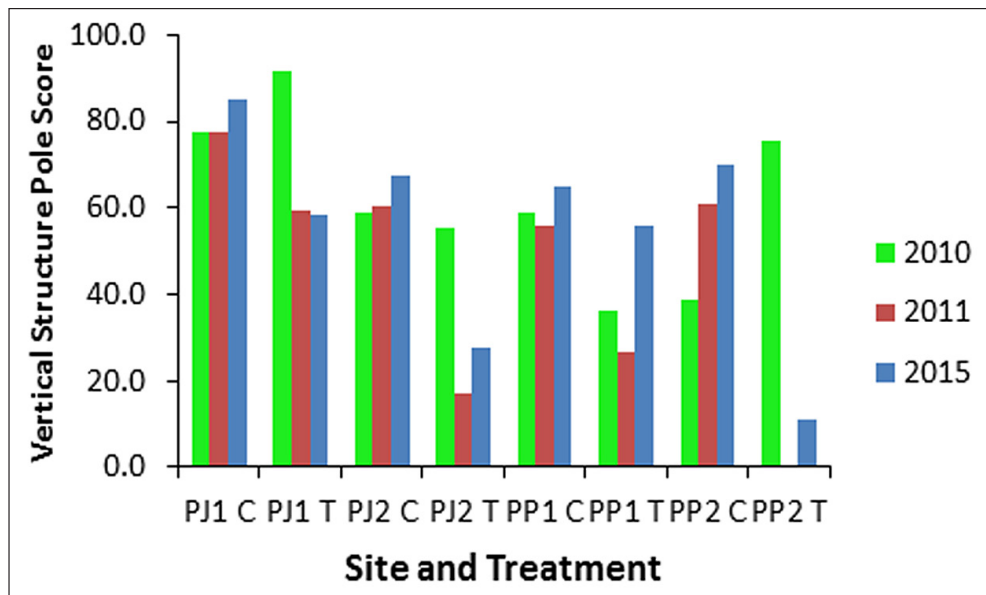


**Figure 5**—Average (averaged over 11 points per plot) vertical vegetation canopy cover structure 0 to 3 m above the ground surface for the two pinyon-juniper (PJ1 and PJ2) and the two ponderosa pine (PP1 and PP2) sites. C = control; T = treatment.

PP2 site,  $p < 0.0001$ ) and at one of the pinyon-juniper sites (PJ2,  $p < 0.0001$ ), but not significantly different at the PJ1 site (fig. 5). In 2015, vertical canopy cover structure was significantly lower on treatment plots than on control plots at the PP2 site ( $p < 0.0001$ ) and at both pinyon-juniper sites (PJ1 site,  $p = 0.026$ ; PJ2 site,  $p < 0.0001$ ), but not at the PP1 site (fig. 5).

#### *Tree Crown Canopy Cover*

Tree crown horizontal canopy cover as measured by a spherical densiometer showed a significant reduction in tree upper canopy cover on all of the treatment plots in 2011 compared to the control plots following tree thinning in late 2010 (fig. 6). Before tree thinning in

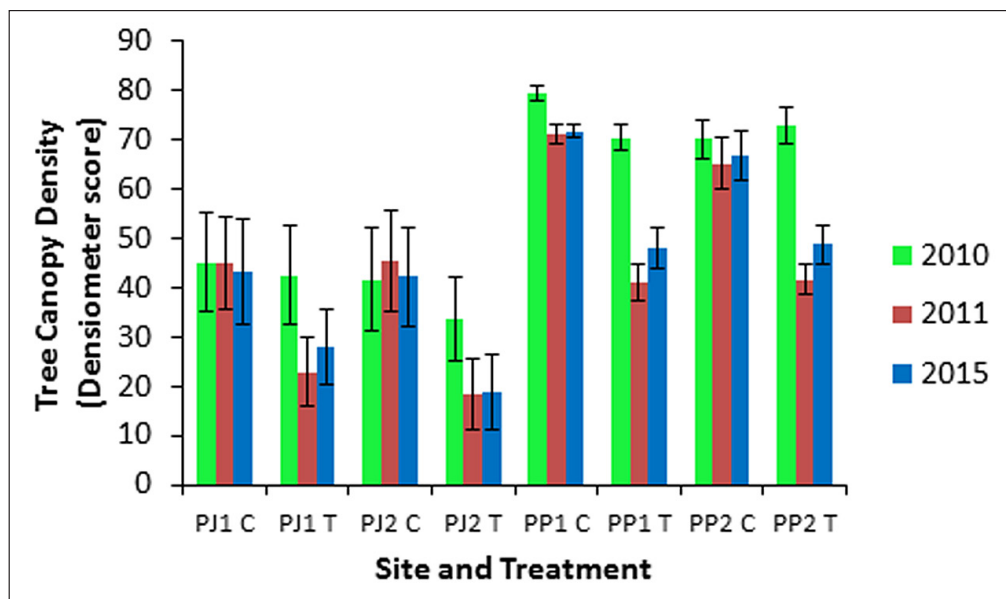


**Figure 6**—Average densiometer score (averaged over 11 points per plot) for horizontal tree crown canopy. Error bars represent  $\pm$  one standard error of the mean.

2010, there were no significant differences (paired t-test,  $n = 11$  for all tests) in the tree canopy cover between the control and pretreatment plots at both pinyon-juniper sites and at one ponderosa pine site (PP2). But at the PP1 site, the control plot had greater prethinning cover than the pretreatment plot. In 2015, tree canopy cover within plots was not significantly different from 2011. In 2015, control plots remained significantly different from the treated plots at the PP1 ( $p < 0.0001$ ) and PP2 ( $p = 0.01$ ) sites, but not at either of the pinyon-juniper sites.

### Pinyon Cone and Seed Production

Results of testing for differences between the percentages of tree canopies with cones between control and treatment plots using the Wilcoxon test revealed that trees at the PJ2 site on the treatment plot had a significantly greater percentage of tree canopies bearing cones (57 percent vs. 85 percent) ( $p = 0.02$ ,  $n = 25$ ). At the PJ1 site, the difference between treatment and control was not significant.

### Understory Vegetation Foliage Canopy Cover

Results for herbaceous understory vegetation canopy cover measured from the thirty-six 1-m<sup>2</sup> quadrats from 2010 through 2015 are presented in fig. 7. Within-year paired t-test results for differences between control and treatment watersheds are presented in table 1. Total herbaceous canopy cover was not significantly different between control and treatment plots at any site in 2010 before tree thinning treatments, but total herbaceous cover was significantly greater on treated plots at all sites except PP2 in 2011 through 2015 following thinning treatments. The amount of herbaceous vegetation cover on treatment plots tended to increase even more with each successive year relative to control plots at the three sites other than PP2 through 2015, except for a leveling off and decline in total canopy cover at the PJ1 site (fig. 7, table 1). The PJ2 site demonstrated the greatest increase in herbaceous vegetation cover on the treatment plot compared to the control plot over time. Total herbaceous vegetation cover was consistently about twice as high at the two pinyon-juniper sites compared to the two ponderosa pine sites over time.

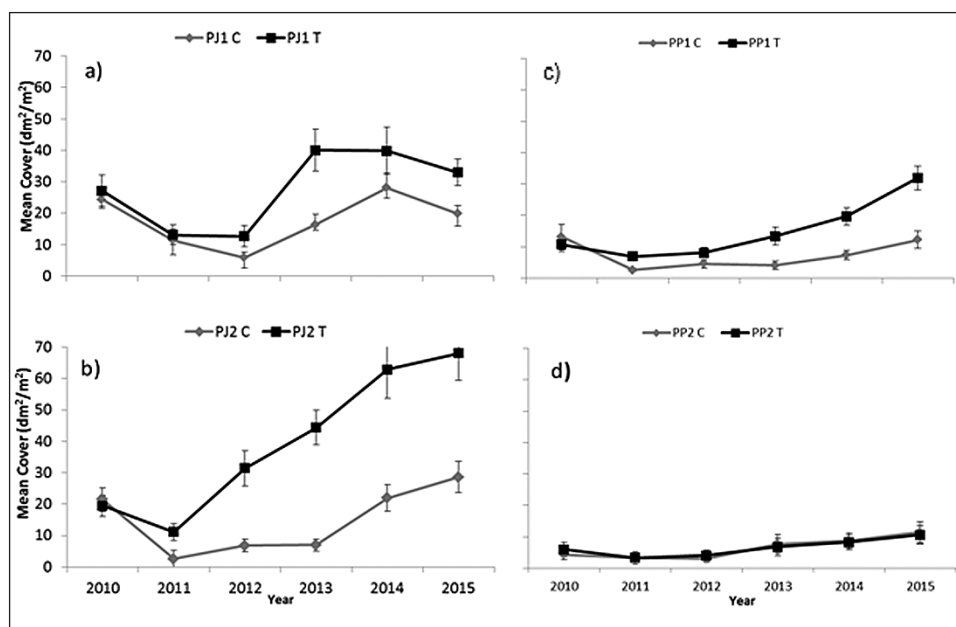


**Figure 7**—Mean herbaceous plant canopy cover values across all vegetation quadrats on (a,b) the two pinyon-juniper (PJ1 and PJ2) and (c,d) the two ponderosa pine (PP1 and PP2) sites, 2010–2015. C = control; T = treatment. Thinning treatments occurred on the treatment plots between 2010 and 2011. Error bars represent  $\pm$  one standard error of the mean.

**Table 1**—Test results for paired t-tests of no difference between mean values of vegetation and ground cover types measured from vegetation quadrats on each study plot pair at the two pinyon-juniper (PJ1 and PJ 2) and the two ponderosa pine (PP1 and PP2) study sites prior to thinning treatments in 2010, and in 2011 and 2015 following thinning treatments.

| Year | Site | Ground cover      | Control mean | Treatment mean | P-value (significance) |
|------|------|-------------------|--------------|----------------|------------------------|
| 2010 | PJ1  | All herbs         | 24.3         | 27.2           | 0.6573                 |
|      |      | Wood chips        | 0.0          | 0.0            | na                     |
|      | PJ2  | All herbs         | 21.5         | 19.5           | 0.6952                 |
|      |      | Wood chips        | 0.0          | 0              | na                     |
|      | PP1  | All herbs         | 13.1         | 10.7           | 0.5888                 |
|      |      | Wood chips        | 0.0          | 0.0            | na                     |
|      | PP2  | All herbs         | 4.4          | 5.9            | 0.6092                 |
|      |      | Wood chips        | 0.0          | 0.0            | na                     |
| 2011 | PJ1  | All herbs         | 11.4         | 13.1           | 0.7227                 |
|      |      | <b>Wood chips</b> | <b>34.11</b> | <b>0.0</b>     | <b>&lt;0.0001</b>      |
|      | PJ2  | All herbs         | 6.4          | 11.1           | 0.2121                 |
|      |      | Wood chips        | 41.6         | 0.0            | <0.0001                |
|      | PP1  | <b>All herbs</b>  | <b>2.1</b>   | <b>7.0</b>     | <b>0.0016</b>          |
|      |      | <b>Wood chips</b> | <b>33.9</b>  | <b>0.0</b>     | <b>&lt;0.0001</b>      |
|      | PP2  | All herbs         | 3.3          | 3.3            | 0.9980                 |
|      |      | <b>Wood chips</b> | <b>40.2</b>  | <b>0.0</b>     | <b>&lt;0.0001</b>      |
| 2015 | PJ1  | <b>All herbs</b>  | <b>19.8</b>  | <b>33.0</b>    | <b>0.0103</b>          |
|      |      | <b>Wood chips</b> | <b>0.0</b>   | <b>22.8</b>    | <b>&lt;0.0001</b>      |
|      | PJ2  | <b>All herbs</b>  | <b>28.6</b>  | <b>68.0</b>    | <b>0.0002</b>          |
|      |      | <b>Wood chips</b> | <b>0.0</b>   | <b>20.1</b>    | <b>&lt;0.0001</b>      |
|      | PP1  | <b>All herbs</b>  | <b>12.2</b>  | <b>32.0</b>    | <b>&lt;0.0001</b>      |
|      |      | <b>Wood chips</b> | <b>0.0</b>   | <b>18.0</b>    | <b>&lt;0.0001</b>      |
|      | PP2  | All herbs         | 3.4          | 2.9            | 0.8953                 |
|      |      | <b>Wood chips</b> | <b>0.0</b>   | <b>15.6</b>    | <b>&lt;0.0001</b>      |

Note: Results in rows that are presented in **bold** were significantly different ( $P < 0.05$ ) between control and treatment plots. All tests were with sample sizes of 36; P-values of less than 0.05 represent significant differences.

The majority of forbs were native species such as *Chenopodium* species that grew on disturbed soils and wood chips. The exotic invasive weed species redstem stork's bill (*Erodium cicutarium*), prickly Russian thistle (*Salsola tragus*), puncture vine (*Tribulus terrestris*), and common mullein (*Verbascum thapsus*) were found only at one of the four sites (PJ2, treated) and in very low numbers in 2014 and in 2015. Dominant grasses at the two pinyon-juniper sites that responded positively to tree thinning were perennial species such as blue grama (*Bouteloua gracilis*) and James' galleta (*Pleuraphis jamesii*). Those perennial grasses grew through the wood chips from existing individual plants that were in place prior to thinning treatments, unlike annual forbs that colonized the disturbed soils and wood chips from seeds. One species of invasive exotic grass, cheatgrass (*Bromus tectorum*), was found only at one of the four sites (PJ2, treated) and in very low numbers in 2015.

### Understory Plant Species Composition

Cluster analysis dendrograms (trees) for all paired plots within sites in 2010, 2011, and 2015 are presented in fig. 8. Before the tree thinning treatments in 2010 and 2011, the paired plots at both ponderosa pine sites and at both pinyon-juniper sites shared more similar understory plant species compositions within each site location than between site locations. There were no groupings of treatment plots or groupings of control plots; all groupings or similarities in plant species compositions were based on location (fig. 8a).

Within 1 year following the tree thinning treatments, the location-based site and plot cluster groupings observed in 2010 were less pronounced, and the treated plots at the two ponderosa pine sites grouped together, not by location as before thinning (fig. 8b). In 2011, the PJ2 control plot grouped with the ponderosa pine sites, while the PJ2 treatment plot grouped with both PJ1 plots. The ponderosa pine plots were most similar in understory plant species compositions by location in 2010 before treatment, then were most similar by treatment in 2011; but they became more similar by location again by 2015 (fig. 8). The pinyon-juniper control plots from the two different pinyon-juniper sites grouped together based on treatment, while the PJ2 treatment plot had plant species compositions distinctly different from any of the other pinyon-juniper

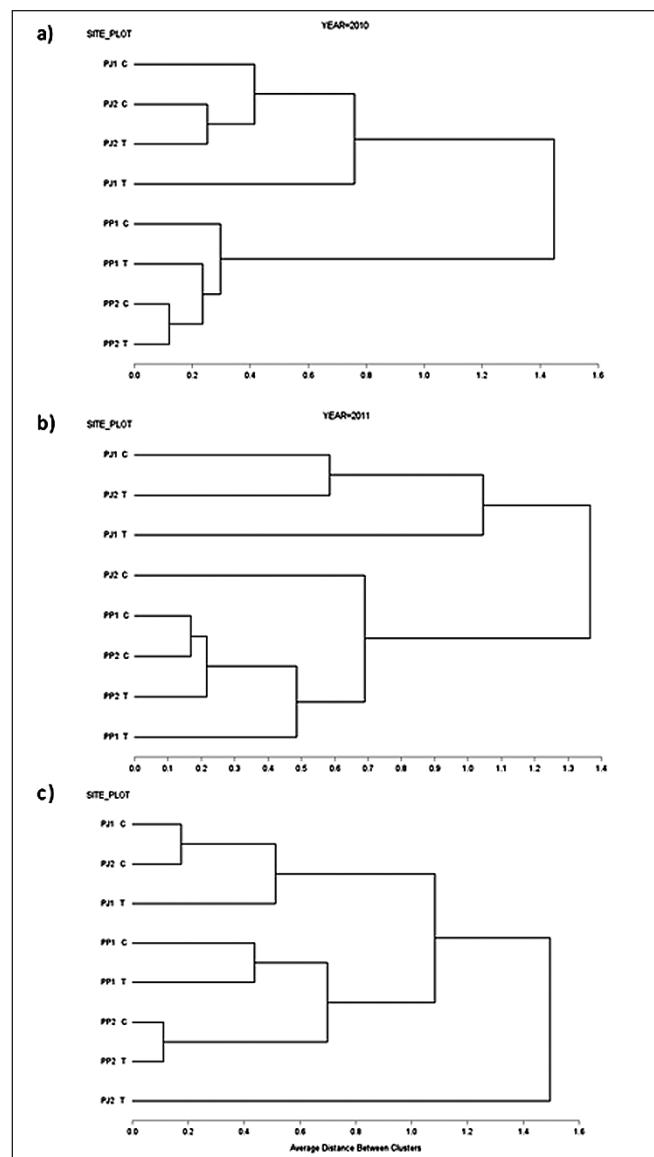


**Figure 8**—Cluster analysis results showing the similarity of sites and paired plots based on similarity of the herbaceous plant community species compositions in (a) 2010, (b) 2011, and (c) 2015.

or ponderosa pine sites or plots (fig. 8c). Overall, cluster analysis results showed that the tree thinning treatments resulted in large changes in the plant species compositions of the treated plots at both pinyon-juniper sites, but tree thinning did not change the understory vegetation species compositions of the ponderosa pine sites (fig. 8).

### Wood Chips

Wood chips were not present before the thinning treatments in late 2010, and they were applied only to the treated watersheds and plots in late 2010. Wood

chip cover on the treated plots declined by about 20 percent at all sites between 2011 and 2012, indicating some decomposition or redistribution, or increased herbaceous plant canopy cover over wood chips (or a combination thereof) (fig. 9). From 2012 through 2015, wood chip cover remained fairly constant at the two pinyon-juniper sites. In 2015, wood chip cover increased to about 20 percent cover again at the two ponderosa pine sites.

## Wildlife

### Birds

Lists of bird species encountered across all four study sites, along with their daily average counts averaged over 3 years before tree thinning and averaged over 5 years after tree thinning, are presented in table 2 for the pinyon-juniper sites and table 3 for the ponderosa pine sites. In total, 1,359 birds representing 66 species were recorded from the pinyon-juniper sites, and 1,064 birds representing 61 species were recorded from the

ponderosa pine sites from 2008 through 2015. Birds that could not be identified to species were recorded to the most precise taxonomic level possible (e.g., flycatcher, sparrow, passerine, hummingbird), and those records accounted for a small percentage of the total records (tables 2 and 3). Bird species that were recorded from point counts on the study plots, but were typically observed or heard some distance away from the sites and were not actually utilizing the local study plot habitats, included American crow (see tables 2 and 3 for scientific names of all bird species mentioned in this section), common raven, great blue heron, northern harrier, red-tailed hawk, Swainson's hawk, and turkey vulture. Those species data were excluded from the analysis.

The numbers of birds recorded across the two pinyon-juniper study sites during the spring breeding season from 2009 (no spring counts in 2008) through 2015 and during the fall migratory season from fall 2008 through fall 2015 were generally more abundant on the treatment plots, even before the tree thinning

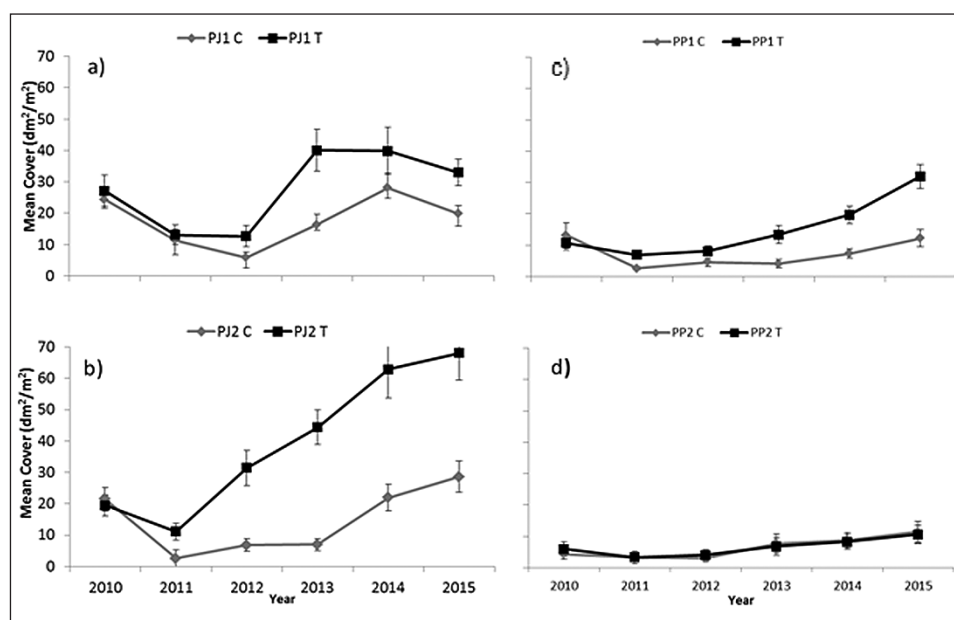


**Figure 9**—Mean wood chip cover values across all vegetation quadrats on (a,b) the two pinyon-juniper (PJ1 and PJ2) and (c,d) the two ponderosa pine (PP1 and PP2) sites, 2010–2015. Thinning treatments occurred on the treatment (T) plots between 2010 and 2011. Error bars represent  $\pm$  one standard error of the mean.

**Table 2**—Pinyon-juniper bird species and mean counts of individuals per species averaged over counts per 3-day sample period, averaged over the seasons and years prior to tree thinning from fall 2008 through fall 2010, and averaged over the seasons and years after tree thinning from spring 2011 through fall 2015.

| Bird species on pinyon-juniper (PJ) sites |                                    |                       | Before thinning, 2008–2010 |         |         |         |         |         | After thinning, 2011–2015 |         |             |         |  |  |
|-------------------------------------------|------------------------------------|-----------------------|----------------------------|---------|---------|---------|---------|---------|---------------------------|---------|-------------|---------|--|--|
| Common name                               | Scientific name                    | AOU <sup>1</sup> code | PJ1                        |         | PJ2     |         | PJ1     |         | PJ2                       |         | Row average |         |  |  |
|                                           |                                    |                       | Control                    | Treated | Control | Treated | Control | Treated | Control                   | Treated | Control     | Treated |  |  |
| American crow                             | <i>Corvus brachyrhynchos</i>       | AMCR                  | 0.5                        | 0.7     | 0.9     | 0.7     | 0.9     | 0.7     | 0.5                       | 0.6     | 0.7         |         |  |  |
| American goldfinch                        | <i>Spinus tristis</i>              | AMGO                  | 0.0                        | 0.3     | 0.0     | 0.0     | 0.0     | 0.0     | 0.0                       | 2.0     | 0.3         |         |  |  |
| American kestrel                          | <i>Falco sparverius</i>            | AMKE                  | 0.0                        | 0.0     | 0.3     | 0.0     | 0.0     | 0.3     | 0.0                       | 0.3     | 0.1         |         |  |  |
| American robin                            | <i>Turdus migratorius</i>          | AMRO                  | 0.3                        | 0.3     | 0.5     | 0.5     | 0.7     | 0.5     | 0.8                       | 0.4     | 0.5         |         |  |  |
| Ash-throated flycatcher                   | <i>Myiarchus cinerascens</i>       | ATFL                  | 0.3                        | 0.3     | 0.5     | 0.3     | 0.4     | 0.3     | 0.5                       | 0.3     | 0.4         |         |  |  |
| Audubon's warbler                         | <i>Setophaga coronata auduboni</i> | AUWA                  | 0.3                        | 0.3     | 0.3     | 0.8     | 0.3     | 0.3     | 0.3                       | 0.4     | 0.4         |         |  |  |
| Barn swallow                              | <i>Hirundo rustica</i>             | BARS                  | 0.3                        | 0.0     | 0.0     | 0.0     | 0.0     | 0.3     | 0.0                       | 0.0     | 0.1         |         |  |  |
| Black-capped chickadee                    | <i>Poecile atricapillus</i>        | BCCH                  | 0.3                        | 0.3     | 0.3     | 0.3     | 0.0     | 1.0     | 0.0                       | 0.0     | 0.3         |         |  |  |
| Black-chinned hummingbird                 | <i>Archilochus alexandri</i>       | BCHU                  | 0.0                        | 0.0     | 0.0     | 0.0     | 0.0     | 0.3     | 0.5                       | 0.5     | 0.2         |         |  |  |
| Bell's vireo                              | <i>Vireo bellii</i>                | BEVI                  | 0.0                        | 0.0     | 0.0     | 0.0     | 0.3     | 0.0     | 0.3                       | 0.0     | 0.1         |         |  |  |
| Bewick's wren                             | <i>Thryomanes bewickii</i>         | BEWR                  | 0.3                        | 0.4     | 0.7     | 0.6     | 0.3     | 0.7     | 0.3                       | 0.3     | 0.4         |         |  |  |
| Blue-gray gnatcatcher                     | <i>Poliophtila caerulea</i>        | BGGN                  | 0.0                        | 0.0     | 0.0     | 0.0     | 0.0     | 0.8     | 0.0                       | 0.0     | 0.1         |         |  |  |
| Broad-tailed hummingbird                  | <i>Selasphorus platycercus</i>     | BTAH                  | 0.3                        | 0.0     | 0.0     | 0.3     | 0.3     | 0.3     | 0.0                       | 0.4     | 0.2         |         |  |  |
| Black-throated gray warbler               | <i>Setophaga nigrescens</i>        | BTYW                  | 0.0                        | 0.0     | 0.3     | 0.0     | 0.0     | 0.0     | 0.0                       | 0.0     | 0.0         |         |  |  |
| Bush tit                                  | <i>Psaltiriparus minimus</i>       | BUSH                  | 0.0                        | 0.3     | 0.0     | 0.0     | 0.6     | 0.5     | 0.0                       | 0.0     | 0.2         |         |  |  |
| Canyon towhee                             | <i>Melospiza fusca</i>             | CANT                  | 0.0                        | 0.0     | 0.0     | 0.0     | 0.0     | 0.5     | 0.0                       | 0.0     | 0.1         |         |  |  |
| Chipping sparrow                          | <i>Spizella passerina</i>          | CHSP                  | 0.3                        | 0.4     | 0.7     | 0.8     | 1.3     | 0.5     | 0.7                       | 0.6     | 0.7         |         |  |  |
| Cordilleran flycatcher                    | <i>Empidonax occidentalis</i>      | COFL                  | 0.0                        | 0.0     | 0.0     | 0.0     | 0.0     | 0.0     | 0.3                       | 0.3     | 0.1         |         |  |  |
| Cooper's hawk                             | <i>Accipiter cooperii</i>          | COHA                  | 0.0                        | 0.0     | 0.0     | 0.0     | 0.0     | 0.3     | 0.0                       | 0.7     | 0.1         |         |  |  |
| Common raven                              | <i>Corvus corax</i>                | CORA                  | 0.3                        | 0.4     | 0.8     | 0.6     | 0.7     | 0.8     | 0.6                       | 1.0     | 0.6         |         |  |  |
| Dark-eyed junco                           | <i>Junco hyemalis</i>              | DEJU                  | 0.0                        | 1.0     | 1.0     | 0.3     | 0.0     | 0.0     | 0.0                       | 0.0     | 0.3         |         |  |  |
| Great blue heron                          | <i>Ardea herodias</i>              | GBHE                  | 0.0                        | 0.0     | 0.0     | 0.3     | 0.0     | 0.0     | 0.0                       | 0.0     | 0.0         |         |  |  |
| Gray flycatcher                           | <i>Empidonax wrightii</i>          | GRFL                  | 0.0                        | 0.0     | 0.0     | 0.3     | 0.0     | 0.3     | 0.3                       | 0.0     | 0.1         |         |  |  |
| Gray vireo                                | <i>Vireo vicinior</i>              | GRVI                  | 0.0                        | 0.0     | 0.0     | 0.0     | 0.0     | 0.3     | 0.3                       | 0.5     | 0.1         |         |  |  |

(CONT.)

| Bird species on pinyon-juniper (P-J) sites |                                   |           | Before thinning, 2008–2010 |         |         |         | After thinning, 2011–2015 |         |         |         | Row average |     |
|--------------------------------------------|-----------------------------------|-----------|----------------------------|---------|---------|---------|---------------------------|---------|---------|---------|-------------|-----|
| Common name                                | Scientific name                   | AOU¹ code | PJ1                        |         | PJ2     |         | PJ1                       |         | PJ2     |         |             |     |
|                                            |                                   |           | Control                    | Treated | Control | Treated | Control                   | Treated | Control | Treated |             |     |
| Hairy woodpecker                           | <i>Picoides villosus</i>          | HAWO      | 0.0                        | 0.3     | 0.0     | 0.3     | 0.0                       | 0.3     | 0.0     | 0.0     | 0.3         | 0.1 |
| Hermit thrush                              | <i>Catharus guttatus</i>          | HETH      | 0.0                        | 0.0     | 0.0     | 0.0     | 0.3                       | 0.0     | 0.0     | 0.0     | 0.0         | 0.0 |
| House finch                                | <i>Haemorhous mexicanus</i>       | HOFI      | 0.0                        | 0.0     | 0.0     | 0.0     | 0.0                       | 0.0     | 1.0     | 0.0     | 0.0         | 0.1 |
| Juniper titmouse                           | <i>Baeolophus ridgwayi</i>        | JUTI      | 0.3                        | 0.6     | 0.7     | 0.5     | 0.4                       | 0.6     | 0.6     | 0.6     | 0.6         | 0.5 |
| Lark sparrow                               | <i>Chondestes grammacus</i>       | LASP      | 0.0                        | 0.0     | 0.0     | 0.0     | 0.0                       | 0.7     | 0.0     | 0.9     | 0.2         | 0.2 |
| Ladder-backed woodpecker                   | <i>Picoides scalaris</i>          | LBWO      | 0.0                        | 0.0     | 0.0     | 0.3     | 0.3                       | 0.3     | 0.0     | 0.0     | 0.0         | 0.1 |
| Lesser goldfinch                           | <i>Spinus psaltria</i>            | LEGO      | 0.0                        | 0.0     | 0.0     | 0.0     | 0.3                       | 0.0     | 0.3     | 0.5     | 0.1         | 0.1 |
| Mountain bluebird                          | <i>Sialia currucoides</i>         | MOBL      | 0.0                        | 0.0     | 1.7     | 5.0     | 0.0                       | 0.4     | 1.0     | 0.5     | 1.1         | 1.1 |
| Mountain chickadee                         | <i>Poecile gambeli</i>            | MOCH      | 0.3                        | 0.3     | 0.5     | 0.3     | 0.5                       | 0.4     | 0.0     | 0.4     | 0.4         | 0.4 |
| Mourning dove                              | <i>Zenaida macroura</i>           | MODO      | 0.7                        | 0.9     | 0.5     | 0.5     | 0.3                       | 0.8     | 0.4     | 0.9     | 0.6         | 0.6 |
| Northern flicker                           | <i>Colaptes auratus</i>           | NOFL      | 0.4                        | 0.4     | 0.5     | 0.6     | 0.4                       | 0.4     | 0.5     | 0.5     | 0.5         | 0.5 |
| Northern harrier                           | <i>Circus cyaneus</i>             | NOHA      | 0.3                        | 0.3     | 0.0     | 0.0     | 0.0                       | 0.0     | 0.0     | 0.0     | 0.1         | 0.1 |
| Northern mockingbird                       | <i>Mimus polyglottos</i>          | NOMO      | 0.0                        | 0.0     | 0.3     | 0.0     | 0.0                       | 0.0     | 0.4     | 0.5     | 0.1         | 0.1 |
| Northern rough-winged swallow              | <i>Stelgidopteryx serripennis</i> | NRWS      | 0.0                        | 0.0     | 0.0     | 0.0     | 1.7                       | 0.5     | 0.5     | 0.5     | 0.4         | 0.4 |
| Orange-crowned warbler                     | <i>Oreothlypis celata</i>         | OCWA      | 0.7                        | 0.0     | 0.0     | 0.0     | 0.0                       | 0.0     | 0.3     | 0.0     | 0.1         | 0.1 |
| Pinyon jay                                 | <i>Gymnorhinus cyanocephalus</i>  | PIJA      | 0.3                        | 0.3     | 0.5     | 0.0     | 1.7                       | 2.3     | 2.8     | 2.3     | 1.3         | 1.3 |
| Plumbeous vireo                            | <i>Vireo plumbeus</i>             | PLVI      | 0.0                        | 0.0     | 0.0     | 0.3     | 0.3                       | 0.0     | 0.3     | 0.4     | 0.2         | 0.2 |
| Red-breasted nuthatch                      | <i>Sitta canadensis</i>           | RBNU      | 0.0                        | 0.0     | 0.0     | 0.3     | 0.3                       | 0.0     | 0.0     | 0.3     | 0.1         | 0.1 |
| Ruby-crowned kinglet                       | <i>Regulus calendula</i>          | RCKI      | 0.0                        | 0.0     | 0.0     | 0.0     | 0.0                       | 0.3     | 0.0     | 0.0     | 0.0         | 0.0 |
| Red crossbill                              | <i>Loxia curvirostra</i>          | RECR      | 0.0                        | 0.0     | 0.7     | 0.0     | 0.0                       | 0.0     | 0.0     | 0.0     | 0.1         | 0.1 |
| Red-tailed hawk                            | <i>Buteo jamaicensis</i>          | RTHA      | 0.0                        | 0.0     | 0.0     | 0.0     | 0.3                       | 0.3     | 0.0     | 0.9     | 0.2         | 0.2 |
| Rufous hummingbird                         | <i>Selasphorus rufus</i>          | RUHU      | 0.3                        | 0.0     | 0.0     | 0.0     | 0.0                       | 0.0     | 0.0     | 0.0     | 0.0         | 0.0 |
| Say's phoebe                               | <i>Sayornis saya</i>              | SAPH      | 0.0                        | 0.0     | 0.0     | 0.0     | 0.0                       | 0.3     | 0.3     | 0.4     | 0.1         | 0.1 |
| Savannah sparrow                           | <i>Passerculus sandwichensis</i>  | SAVS      | 0.0                        | 0.0     | 0.0     | 0.0     | 0.0                       | 1.0     | 0.0     | 0.0     | 0.1         | 0.1 |
| Spotted towhee                             | <i>Pipilo maculatus</i>           | SPTO      | 0.3                        | 0.3     | 0.6     | 0.6     | 0.3                       | 0.3     | 0.3     | 0.4     | 0.4         | 0.4 |
| Sharp-shinned hawk                         | <i>Accipiter striatus</i>         | SSHA      | 0.0                        | 0.0     | 0.0     | 0.3     | 0.0                       | 0.0     | 0.0     | 0.0     | 0.0         | 0.0 |

(CONT.)

| Bird species on pinyon-juniper (P-J) sites |                               |                       | Before thinning, 2008–2010 |         |         |         |         |         | After thinning, 2011–2015 |         |         |         |     |     |             |
|--------------------------------------------|-------------------------------|-----------------------|----------------------------|---------|---------|---------|---------|---------|---------------------------|---------|---------|---------|-----|-----|-------------|
| Common name                                | Scientific name               | AOU <sup>1</sup> code | PJ1                        |         | PJ2     |         | PJ1     |         | PJ2                       |         | PJ1     |         | PJ2 |     | Row average |
|                                            |                               |                       | Control                    | Treated | Control | Treated | Control | Treated | Control                   | Treated | Control | Treated |     |     |             |
| Steller's jay                              | <i>Cyanocitta stelleri</i>    | STJA                  | 0.0                        | 1.0     |         | 0.0     | 0.0     | 0.0     | 0.0                       |         | 0.0     | 0.0     | 0.5 | 0.5 | 0.3         |
| Swainson's hawk                            | <i>Buteo swainsoni</i>        | SWHA                  | 0.0                        | 0.0     |         | 0.0     | 0.0     | 0.0     | 0.0                       |         | 0.0     | 0.0     | 0.3 | 0.0 | 0.1         |
| Swainson's thrush                          | <i>Catharus ustulatus</i>     | SWTH                  | 0.0                        | 0.0     |         | 0.0     | 0.0     | 0.3     | 0.0                       |         | 0.0     | 0.0     | 0.0 | 0.0 | 0.0         |
| Townsend's solitaire                       | <i>Myadestes townsendi</i>    | TOSO                  | 0.3                        | 0.8     |         | 0.3     | 0.3     | 0.3     | 1.0                       |         | 0.7     | 0.3     | 0.3 | 0.4 | 0.5         |
| Tree swallow                               | <i>Tachycineta bicolor</i>    | TRES                  | 0.0                        | 0.0     |         | 0.3     | 0.0     | 0.0     | 0.0                       |         | 0.0     | 0.0     | 0.0 | 0.0 | 0.0         |
| Turkey vulture                             | <i>Cathartes aura</i>         | TUVU                  | 2.2                        | 0.3     |         | 0.7     | 0.5     | 0.7     | 0.7                       |         | 0.5     | 0.4     | 0.7 | 0.7 | 0.7         |
| Vesper sparrow                             | <i>Poocetes gramineus</i>     | VESP                  | 0.0                        | 0.0     |         | 0.0     | 0.0     | 0.0     | 0.0                       |         | 0.0     | 0.0     | 0.0 | 0.3 | 0.0         |
| Violet-green swallow                       | <i>Tachycineta thalassina</i> | VGSW                  | 0.0                        | 0.0     |         | 0.0     | 0.0     | 0.0     | 1.0                       |         | 0.0     | 0.0     | 0.0 | 0.0 | 0.1         |
| White-breasted nuthatch                    | <i>Sitta carolinensis</i>     | WBNU                  | 0.3                        | 0.3     |         | 0.5     | 0.0     | 0.3     | 0.3                       |         | 0.3     | 0.3     | 0.3 | 0.5 | 0.3         |
| Western bluebird                           | <i>Sialia mexicana</i>        | WEBL                  | 0.0                        | 0.0     |         | 0.0     | 1.0     | 1.6     | 0.8                       |         | 1.2     | 1.0     | 0.7 | 0.7 | 0.7         |
| Western kingbird                           | <i>Tyrannus verticalis</i>    | WEKI                  | 0.0                        | 0.0     |         | 0.0     | 0.0     | 0.0     | 0.3                       |         | 0.3     | 0.5     | 0.5 | 0.1 | 0.1         |
| Western meadowlark                         | <i>Sturnella neglecta</i>     | WEME                  | 0.3                        | 0.3     |         | 0.0     | 0.0     | 0.4     | 0.3                       |         | 0.3     | 0.3     | 0.8 | 0.3 | 0.3         |
| Western tanager                            | <i>Piranga ludoviciana</i>    | WETA                  | 0.0                        | 0.0     |         | 0.0     | 0.0     | 0.0     | 0.3                       |         | 0.0     | 0.0     | 0.0 | 0.0 | 0.0         |
| Williamson's sapsucker                     | <i>Sphyrapicus thyroideus</i> | WISA                  | 0.3                        | 0.0     |         | 0.0     | 0.3     | 0.0     | 0.0                       |         | 0.0     | 0.0     | 0.0 | 0.0 | 0.1         |
| Wild turkey                                | <i>Meleagris gallopavo</i>    | WITU                  | 0.0                        | 0.0     |         | 0.0     | 0.0     | 0.0     | 0.0                       |         | 0.0     | 0.6     | 0.0 | 0.0 | 0.1         |
| Woodhouse's scrub-jay                      | <i>Aphelocoma woodhouseii</i> | WOSJ                  | 0.7                        | 0.9     |         | 1.0     | 1.3     | 0.4     | 0.4                       |         | 0.6     | 0.5     | 0.5 | 0.7 | 0.7         |
| White-winged Dove                          | <i>Zenaida asiatica</i>       | WWDO                  | 0.0                        | 0.0     |         | 0.0     | 0.0     | 0.0     | 2.7                       |         | 0.0     | 0.0     | 0.0 | 0.3 | 0.3         |
| UNKNOWN BIRD                               | none                          | UNKNOWN               | 0.7                        | 0.7     |         | 3.7     | 0.3     | 2.0     | 0.0                       |         | 0.7     | 0.0     | 0.0 | 1.0 | 1.0         |
| UNKNOWN HUMMINGBIRD                        | none                          | UKHUMM                | 0.0                        | 0.0     |         | 0.0     | 0.0     | 0.0     | 0.3                       |         | 0.0     | 0.0     | 0.0 | 0.0 | 0.0         |
| UNKNOWN PASSERINE                          | none                          | UKPASS                | 0.0                        | 0.0     |         | 0.0     | 0.0     | 0.4     | 0.6                       |         | 0.6     | 0.3     | 0.3 | 0.2 | 0.2         |
| UNKNOWN SPARROW                            | none                          | UKSPAR                | 0.0                        | 0.0     |         | 0.0     | 0.0     | 0.0     | 0.3                       |         | 0.0     | 1.7     | 0.3 | 0.3 | 0.3         |
| UNKNOWN VIREO                              | none                          | UKVIRE                | 0.0                        | 0.0     |         | 0.0     | 0.0     | 0.0     | 0.0                       |         | 0.0     | 0.0     | 0.3 | 0.0 | 0.0         |
| UNKNOWN WOODPECKER                         | none                          | UKWOOD                | 0.0                        | 0.0     |         | 0.0     | 0.0     | 0.0     | 0.3                       |         | 0.3     | 0.0     | 0.0 | 0.1 | 0.1         |
|                                            |                               | Column average        | 0.2                        | 0.2     |         | 0.3     | 0.3     | 0.3     | 0.4                       |         | 0.3     | 0.4     | 0.3 | 0.4 | 0.3         |

<sup>1</sup>American Ornithological Union.

**Table 3–Ponderosa pine bird species and mean counts of individuals per species averaged over counts per 3-day sample period, averaged over the seasons and years prior to tree thinning from fall 2008 through fall 2010, and averaged over the seasons and years after tree thinning from spring 2011 through fall 2015.**

| Birds on ponderosa pine (PP) sites |                                    |                       | Before thinning, 2008–2010 |         |         |         |         |         | After thinning, 2011–2015 |         |             |         |     |  |
|------------------------------------|------------------------------------|-----------------------|----------------------------|---------|---------|---------|---------|---------|---------------------------|---------|-------------|---------|-----|--|
| Common name                        | Scientific name                    | AOU <sup>1</sup> code | PP1                        |         | PP2     |         | PP1     |         | PP2                       |         | Row average |         |     |  |
|                                    |                                    |                       | Control                    | Treated | Control | Treated | Control | Treated | Control                   | Treated | Control     | Treated |     |  |
| American crow                      | <i>Corvus brachyrhynchos</i>       | AMCR                  | 0.3                        | 0.3     | 0.0     | 0.0     | 0.0     | 0.0     | 0.7                       | 0.3     | 0.0         | 0.0     | 0.2 |  |
| American robin                     | <i>Turdus migratorius</i>          | AMRO                  | 0.4                        | 0.4     | 0.5     | 0.3     | 0.3     | 0.4     | 0.3                       | 0.4     | 0.5         | 0.6     | 0.4 |  |
| Ash-throated flycatcher            | <i>Myiarchus cinerascens</i>       | ATFL                  | 0.0                        | 0.0     | 0.4     | 0.3     | 0.0     | 0.0     | 0.0                       | 0.0     | 0.5         | 0.7     | 0.2 |  |
| Audubon's warbler                  | <i>Setophaga coronata auduboni</i> | AUWA                  | 0.0                        | 0.4     | 0.0     | 0.0     | 0.5     | 0.9     | 0.4                       | 0.6     |             |         | 0.4 |  |
| Black-capped chickadee             | <i>Poecile atricapillus</i>        | BCCH                  | 0.3                        | 0.9     | 0.6     | 0.6     | 0.3     | 0.0     | 0.0                       | 0.0     | 0.0         | 0.0     | 0.3 |  |
| Black-chinned hummingbird          | <i>Archilochus alexandri</i>       | BCHU                  | 0.0                        | 0.0     | 0.0     | 0.0     | 0.0     | 0.0     | 0.3                       | 0.3     |             |         | 0.1 |  |
| Blue-gray gnatcatcher              | <i>Polioptila caerulea</i>         | BGGN                  | 0.0                        | 0.0     | 0.0     | 0.0     | 0.0     | 0.0     | 0.0                       | 0.0     | 0.0         | 0.3     | 0.0 |  |
| Black-headed grosbeak              | <i>Pheucticus melanocephalus</i>   | BHGR                  | 0.0                        | 0.0     | 0.0     | 0.0     | 0.0     | 0.3     | 0.0                       | 0.0     | 0.0         | 0.0     | 0.0 |  |
| Brown creeper                      | <i>Certhia americana</i>           | BRCR                  | 0.0                        | 0.7     | 0.0     | 0.0     | 0.3     | 0.3     | 0.3                       | 0.3     | 0.0         | 0.0     | 0.2 |  |
| Broad-tailed hummingbird           | <i>Selasphorus platycercus</i>     | BTAH                  | 0.0                        | 0.0     | 0.3     | 0.4     | 0.4     | 0.3     | 0.0                       | 0.3     | 0.0         | 0.3     | 0.2 |  |
| Bushtit                            | <i>Psaltirparus minimus</i>        | BUSH                  | 0.0                        | 0.0     | 0.3     | 1.3     | 0.0     | 2.3     | 2.0                       | 1.7     |             |         | 1.0 |  |
| Chipping sparrow                   | <i>Spizella passerina</i>          | CHSP                  | 0.0                        | 0.0     | 0.0     | 1.7     | 0.9     | 0.6     | 0.3                       | 0.5     |             |         | 0.5 |  |
| Cordilleran flycatcher             | <i>Empidonax occidentalis</i>      | COFL                  | 0.0                        | 0.0     | 0.0     | 0.0     | 0.3     | 0.3     | 0.0                       | 0.0     | 0.0         | 0.0     | 0.1 |  |
| Cooper's hawk                      | <i>Accipiter cooperii</i>          | COHA                  | 0.0                        | 0.0     | 0.3     | 0.0     | 0.0     | 0.0     | 0.3                       | 0.0     | 0.0         | 0.0     | 0.1 |  |
| Common raven                       | <i>Corvus corax</i>                | CORA                  | 0.5                        | 0.5     | 0.5     | 0.5     | 0.4     | 0.3     | 0.6                       | 0.5     |             |         | 0.5 |  |
| Dark-eyed junco                    | <i>Junco hyemalis</i>              | DEJU                  | 0.3                        | 0.3     | 1.1     | 0.4     | 0.7     | 0.9     | 1.0                       | 0.4     |             |         | 0.6 |  |
| Downy woodpecker                   | <i>Picoides pubescens</i>          | DOWO                  | 0.0                        | 0.0     | 0.0     | 0.0     | 0.3     | 0.0     | 0.3                       | 0.5     |             |         | 0.1 |  |
| Evening grosbeak                   | <i>Coccothraustes vespertinus</i>  | EVGR                  | 0.0                        | 0.7     | 0.0     | 0.0     | 0.0     | 0.0     | 0.0                       | 0.0     | 0.0         | 0.0     | 0.1 |  |
| Great horned owl                   | <i>Bubo virginianus</i>            | GHOW                  | 0.0                        | 0.0     | 0.0     | 0.0     | 0.3     | 0.0     | 0.0                       | 0.0     | 0.0         | 0.0     | 0.0 |  |
| Gray flycatcher                    | <i>Empidonax wrightii</i>          | GRFL                  | 0.5                        | 0.0     | 0.0     | 0.0     | 0.0     | 0.0     | 0.0                       | 0.3     |             |         | 0.1 |  |
| Grace's warbler                    | <i>Setophaga graciae</i>           | GRWA                  | 0.5                        | 0.5     | 0.5     | 0.6     | 0.5     | 0.5     | 0.5                       | 0.5     |             |         | 0.5 |  |
| Hairy woodpecker                   | <i>Picoides villosus</i>           | HAWO                  | 0.4                        | 0.4     | 0.5     | 0.3     | 0.5     | 0.4     | 0.3                       | 0.0     |             |         | 0.3 |  |
| Hermit thrush                      | <i>Catharus guttatus</i>           | HETH                  | 0.0                        | 0.0     | 0.3     | 0.4     | 0.3     | 0.7     | 0.0                       | 0.0     | 0.0         | 0.0     | 0.2 |  |
| Juniper titmouse                   | <i>Baeolophus ridgwayi</i>         | JUTI                  | 0.0                        | 0.8     | 0.0     | 0.0     | 0.0     | 1.4     | 1.0                       | 0.5     |             |         | 0.5 |  |
| Ladder-backed woodpecker           | <i>Picoides scalaris</i>           | LBWO                  | 0.0                        | 0.0     | 0.0     | 0.0     | 0.0     | 0.3     | 0.0                       | 0.0     | 0.0         | 0.0     | 0.0 |  |
| Lesser goldfinch                   | <i>Spinus psaltria</i>             | LEGO                  | 0.0                        | 0.0     | 0.0     | 0.0     | 0.0     | 0.7     | 0.0                       | 0.0     | 0.0         | 0.0     | 0.1 |  |

(CONT.)

| Birds on ponderosa pine (PP) sites |                                  |                       | Before thinning, 2008–2010 |         |         |         |         |         | After thinning, 2011–2015 |         |             |         |     |  |
|------------------------------------|----------------------------------|-----------------------|----------------------------|---------|---------|---------|---------|---------|---------------------------|---------|-------------|---------|-----|--|
| Common name                        | Scientific name                  | AOU <sup>1</sup> code | PP1                        |         | PP2     |         | PP1     |         | PP2                       |         | Row average |         |     |  |
|                                    |                                  |                       | Control                    | Treated | Control | Treated | Control | Treated | Control                   | Treated | Control     | Treated |     |  |
| Lucy's warbler                     | <i>Oreothlypis luciae</i>        | LUWA                  | 0.0                        | 0.0     | 0.0     | 0.0     | 0.0     | 0.0     | 0.0                       | 0.0     | 1.0         | 0.0     | 0.1 |  |
| Mountain bluebird                  | <i>Sialia currucoides</i>        | MOBL                  | 0.0                        | 0.0     | 0.0     | 0.0     | 0.0     | 0.0     | 0.0                       | 0.0     | 0.0         | 1.7     | 0.2 |  |
| Mountain chickadee                 | <i>Poecile gambeli</i>           | MOCH                  | 0.4                        | 0.3     | 0.6     | 0.7     | 1.1     | 0.7     | 0.8                       | 0.8     | 0.8         | 0.8     | 0.7 |  |
| Mourning dove                      | <i>Zenaida macroura</i>          | MODO                  | 0.5                        | 0.3     | 0.3     | 0.3     | 0.0     | 0.0     | 0.0                       | 0.7     | 0.7         | 0.3     | 0.3 |  |
| Northern flicker                   | <i>Colaptes auratus</i>          | NOFL                  | 0.3                        | 0.5     | 0.4     | 0.5     | 0.4     | 0.4     | 0.3                       | 0.4     | 0.4         | 0.4     | 0.4 |  |
| Northern mockingbird               | <i>Mimus polyglottos</i>         | NOMO                  | 0.0                        | 0.0     | 0.0     | 0.0     | 0.0     | 0.3     | 0.0                       | 0.0     | 0.0         | 0.0     | 0.0 |  |
| Orange-crowned warbler             | <i>Oreothlypis celata</i>        | OCWA                  | 0.0                        | 0.0     | 0.3     | 0.0     | 0.0     | 0.0     | 0.0                       | 0.0     | 0.0         | 0.0     | 0.0 |  |
| Pinyon jay                         | <i>Gymnorhinus cyanocephalus</i> | PIJA                  | 0.0                        | 0.0     | 0.0     | 0.0     | 0.0     | 0.0     | 0.3                       | 0.3     | 0.3         | 0.1     | 0.1 |  |
| Pine siskin                        | <i>Spinus pinus</i>              | PISI                  | 0.5                        | 0.5     | 0.0     | 0.0     | 0.0     | 0.7     | 0.9                       | 0.7     | 0.7         | 0.4     | 0.4 |  |
| Plumbeous vireo                    | <i>Vireo plumbeus</i>            | PLVI                  | 0.4                        | 0.5     | 0.4     | 0.5     | 0.4     | 0.5     | 0.5                       | 0.4     | 0.4         | 0.4     | 0.4 |  |
| Pygmy nuthatch                     | <i>Sitta pygmaea</i>             | PYNU                  | 0.0                        | 0.0     | 0.0     | 0.0     | 0.7     | 0.0     | 0.0                       | 0.0     | 0.0         | 0.0     | 0.1 |  |
| Red-breasted nuthatch              | <i>Sitta canadensis</i>          | RBNU                  | 0.3                        | 0.7     | 0.0     | 0.3     | 0.3     | 0.3     | 0.3                       | 0.5     | 0.5         | 0.3     | 0.3 |  |
| Ruby-crowned kinglet               | <i>Regulus calendula</i>         | RCKI                  | 0.3                        | 0.4     | 0.0     | 0.7     | 0.7     | 0.7     | 0.0                       | 0.3     | 0.3         | 0.4     | 0.4 |  |
| Red crossbill                      | <i>Loxia curvirostra</i>         | RECR                  | 0.8                        | 1.2     | 1.4     | 1.2     | 0.7     | 0.0     | 0.0                       | 5.0     | 5.0         | 1.3     | 1.3 |  |
| Red-tailed hawk                    | <i>Buteo jamaicensis</i>         | RTHA                  | 0.0                        | 0.0     | 0.0     | 0.0     | 0.0     | 0.3     | 0.0                       | 0.0     | 0.0         | 0.0     | 0.0 |  |
| Rufous hummingbird                 | <i>Selasphorus rufus</i>         | RUHU                  | 0.0                        | 0.0     | 0.3     | 0.3     | 0.0     | 0.0     | 0.0                       | 0.0     | 0.0         | 0.0     | 0.1 |  |
| Spotted towhee                     | <i>Pipilo maculatus</i>          | SPTO                  | 0.0                        | 0.3     | 0.3     | 0.3     | 0.0     | 0.3     | 0.0                       | 0.0     | 0.0         | 0.2     | 0.2 |  |
| Sharp-shinned hawk                 | <i>Accipiter striatus</i>        | SSHA                  | 0.0                        | 0.0     | 0.0     | 0.0     | 0.3     | 0.3     | 0.0                       | 0.3     | 0.3         | 0.1     | 0.1 |  |
| Steller's jay                      | <i>Cyanocitta stelleri</i>       | STJA                  | 0.4                        | 0.3     | 0.4     | 0.4     | 0.5     | 0.7     | 0.5                       | 0.5     | 0.5         | 0.4     | 0.4 |  |
| Swainson's hawk                    | <i>Buteo swainsoni</i>           | SWHA                  | 0.0                        | 0.0     | 0.0     | 0.0     | 0.0     | 0.0     | 0.3                       | 0.0     | 0.0         | 0.0     | 0.0 |  |
| Townsend's solitaire               | <i>Myadestes townsendi</i>       | TOSO                  | 0.5                        | 0.0     | 0.0     | 0.4     | 0.0     | 0.3     | 0.0                       | 0.3     | 0.3         | 0.2     | 0.2 |  |
| Tree swallow                       | <i>Tachycineta bicolor</i>       | TRES                  | 0.0                        | 0.0     | 0.0     | 1.0     | 0.0     | 0.0     | 0.0                       | 0.0     | 0.0         | 0.1     | 0.1 |  |
| Turkey vulture                     | <i>Cathartes aura</i>            | TUVU                  | 0.3                        | 0.7     | 0.3     | 0.0     | 1.7     | 0.3     | 0.3                       | 0.0     | 0.3         | 0.5     | 0.5 |  |
| Violet-green swallow               | <i>Tachycineta thalassina</i>    | VGSW                  | 0.0                        | 0.0     | 0.0     | 0.0     | 0.0     | 0.7     | 0.0                       | 0.6     | 0.6         | 0.2     | 0.2 |  |
| White-breasted nuthatch            | <i>Sitta carolinensis</i>        | WBNU                  | 0.4                        | 0.6     | 0.5     | 0.4     | 0.4     | 0.4     | 0.5                       | 0.4     | 0.4         | 0.4     | 0.4 |  |
| White-crowned sparrow              | <i>Zonotrichia leucophrys</i>    | WCSP                  | 0.0                        | 0.0     | 0.3     | 0.3     | 0.0     | 0.0     | 0.0                       | 0.0     | 0.0         | 0.1     | 0.1 |  |
| Western bluebird                   | <i>Sialia mexicana</i>           | WEBL                  | 0.0                        | 0.0     | 0.0     | 0.3     | 0.8     | 0.8     | 1.4                       | 0.7     | 0.7         | 0.5     | 0.5 |  |
| Western kingbird                   | <i>Tyrannus verticalis</i>       | WEKI                  | 0.0                        | 0.0     | 0.0     | 0.0     | 0.0     | 0.0     | 0.0                       | 0.3     | 0.3         | 0.0     | 0.0 |  |
| Western tanager                    | <i>Piranga ludoviciana</i>       | WETA                  | 0.0                        | 0.0     | 0.0     | 0.0     | 0.0     | 0.8     | 0.3                       | 0.0     | 0.0         | 0.1     | 0.1 |  |
| Western wood-pewee                 | <i>Contopus sordidulus</i>       | WEWP                  | 0.0                        | 0.5     | 0.0     | 0.3     | 0.0     | 0.0     | 0.0                       | 0.0     | 0.0         | 0.1     | 0.1 |  |

(CONT.)

| Birds on ponderosa pine (PP) sites |                               |                       | Before thinning, 2008–2010 |         |         |         |         |         | After thinning, 2011–2015 |         |             |     |  |  |
|------------------------------------|-------------------------------|-----------------------|----------------------------|---------|---------|---------|---------|---------|---------------------------|---------|-------------|-----|--|--|
| Common name                        | Scientific name               | AOU <sup>1</sup> code | PP1                        |         | PP2     |         | PP1     |         | PP2                       |         | Row average |     |  |  |
|                                    |                               |                       | Control                    | Treated | Control | Treated | Control | Treated | Control                   | Treated |             |     |  |  |
| Williamson's sapsucker             | <i>Sphyrapicus thyroideus</i> | WISA                  | 0.0                        | 0.0     | 0.5     | 0.0     | 0.3     | 0.0     | 0.0                       | 0.3     | 0.3         | 0.1 |  |  |
| Wild turkey                        | <i>Meleagris gallopavo</i>    | WITU                  | 0.0                        | 0.0     | 0.0     | 0.0     | 0.0     | 0.3     | 0.3                       | 0.0     | 0.0         | 0.1 |  |  |
| Wilson's warbler                   | <i>Cardellina pusilla</i>     | WIWA                  | 0.0                        | 0.0     | 0.3     | 0.0     | 0.0     | 0.0     | 0.0                       | 0.0     | 0.0         | 0.0 |  |  |
| Woodhouse's scrub-jay              | <i>Aphelocoma woodhouseii</i> | WOSJ                  | 0.5                        | 0.8     | 0.0     | 0.7     | 0.3     | 0.4     | 0.4                       | 0.5     | 0.4         | 0.4 |  |  |
| UNKNOWN FLYCATCHER                 | none                          | UKFLYC                | 0.0                        | 0.0     | 0.0     | 0.0     | 0.0     | 0.3     | 0.0                       | 0.0     | 0.0         | 0.0 |  |  |
| UNKNOWN HUMMINGBIRD                | none                          | UKHUMM                | 0.0                        | 0.0     | 0.0     | 0.0     | 0.0     | 0.0     | 0.3                       | 0.3     | 0.1         | 0.1 |  |  |
| UNKNOWN PASSERINE                  | none                          | UKPASS                | 0.0                        | 0.0     | 0.0     | 0.0     | 0.5     | 0.5     | 0.8                       | 1.0     | 0.4         | 0.4 |  |  |
| UNKNOWN SPARROW                    | none                          | UNKSPAR               | 0.4                        | 0.9     | 0.3     | 1.2     | 0.0     | 0.0     | 0.0                       | 0.0     | 0.3         | 0.3 |  |  |
| UNKNOWN VIREO                      | none                          | UKVIREO               | 0.0                        | 0.0     | 0.0     | 0.0     | 0.5     | 0.5     | 0.8                       | 1.0     | 0.4         | 0.4 |  |  |
|                                    |                               | Column average        | 0.1                        | 0.2     | 0.2     | 0.3     | 0.2     | 0.3     | 0.3                       | 0.4     | 0.3         | 0.3 |  |  |

<sup>1</sup>American Ornithological Union.

treatments during the winter of 2010–2011 (figs. 10a, 10b). Bird abundance on the control plot versus the treatment plot of the PJ1 site was higher during fall 2008, fall 2011, and fall 2015, and at the PJ2 site during fall 2009, spring 2010, fall 2011, and fall 2015. A similar pattern of more birds on treatment plots even before tree thinning also occurred at the two ponderosa pine sites (fig. 10c, 10d). Overall bird abundance at both the pinyon-juniper and the ponderosa pine sites was generally greater in the fall than during the spring breeding season, except for spring 2015, which had the highest bird counts across the four study sites and considerably more birds on the treatment plots than on the control plots (fig. 10).

Abundant birds found at the pinyon-juniper sites over the duration of the study included the American robin, ash-throated flycatcher, black-chinned hummingbird, Bewick's wren, chipping sparrow, juniper titmouse, mountain bluebird, mountain chickadee, mourning dove, northern flicker, pinyon jay, spotted towhee, Townsend's solitaire, western bluebird, western kingbird, western meadowlark, and Woodhouse's scrub-jay (table 2). The abundance of most bird species was not significantly different between the pinyon-juniper control and treatment plots over the 3 years prior to thinning treatments nor over the 5 years following treatments, based on Chi-Square test results. Birds at

the pinyon-juniper sites that did increase significantly on treatment plots relative to control plots after tree thinning, but not before tree thinning, included the black-chinned hummingbird (PJ1 site only,  $p = 0.02$ ), lark sparrow (PJ1 site only,  $p = 0.02$ ), northern flicker (PJ1 site only,  $p = 0.007$ ), western bluebird (PJ2 site only,  $p = 0.0006$ ), and western kingbird (PJ2 site only,  $p = 0.03$ ). No birds at the pinyon-juniper sites declined significantly in response to tree thinning.

Abundant birds found at the ponderosa pine sites over the duration of the study included the American robin, Audubon's warbler, black-capped chickadee, dark-eyed junco, Grace's warbler, hairy woodpecker, mountain chickadee, northern flicker, plumbeous vireo, red-breasted nuthatch, red crossbill, Steller's jay, white-breasted nuthatch, western bluebird, and Woodhouse's scrub-jay (table 3). As at the pinyon-juniper sites, abundance of most bird species was not significantly affected by the thinning treatments. Bird species that were significantly more abundant on the treated plots following thinning treatments, but not before, included the chipping sparrow (PP1 site only,  $p = 0.003$ ), Steller's jay (PP2 site only,  $p = 0.03$ ), and western bluebird (both PP1 and PP2 sites,  $p = 0.003$ ). The brown creeper was the only bird species that showed a significant decrease in abundance after tree thinning treatments at the PP2 site ( $p = 0.04$ ).

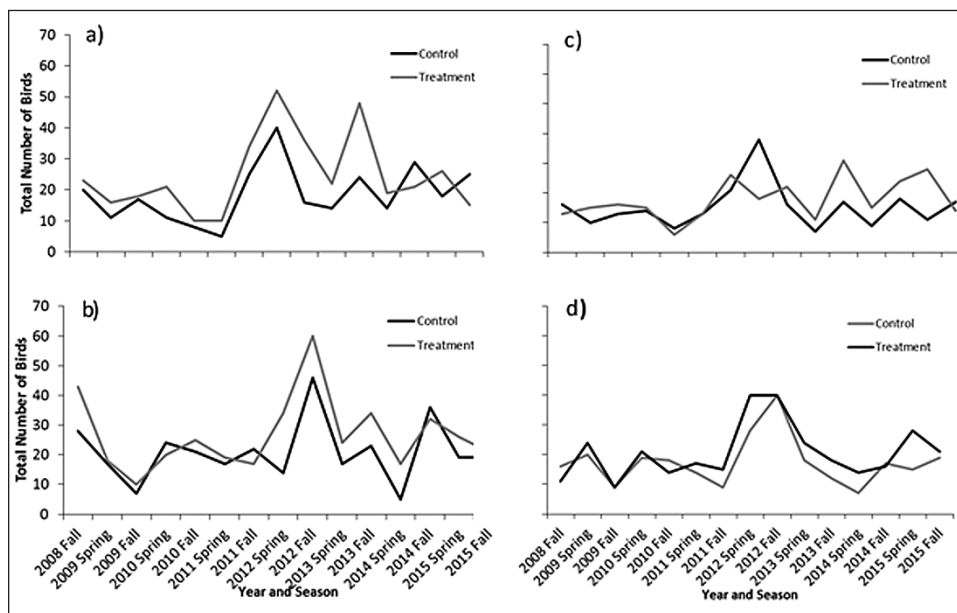


**Figure 10**—Average number of birds per day from 3-day point counts in the spring and fall on (a,b) the two pinyon-juniper (PJ1 and PJ2) and (c,d) the two ponderosa pine (PP1 and PP2) sites, 2010–2015.

Cluster analysis results of similarities in bird species compositions between site locations and between control vs. treatment study plots during the spring breeding seasons showed that in 2010, prior to thinning treatments, study plot bird assemblages grouped together by site location only (fig. 11a). During the fall 2010 migration period, the bird species compositions at the PJ1 and PP1 sites were more similar to each other than to the PJ2 or PP2 sites. In spring 2011 immediately following tree thinning, both spring and fall bird assemblages across study plots at both the pinyon-juniper sites and ponderosa pine sites again grouped together by location, not by treatment or control. But in fall 2011 the two pinyon-juniper control sites grouped together, while both pinyon-juniper treatment plots were very different from all others, including ponderosa pine plots, and the ponderosa pine plots grouped by location, not treatment (fig. 11d). In 2015, 5 years after tree thinning treatments, both spring and fall bird assemblages of control plots at both ponderosa pine sites grouped together, while ponderosa pine treatment plot assemblages were very different; the pinyon-juniper site plots grouped more by site location, not by treatment (fig. 11e,f). These findings suggest that the tree thinning treatments did cause the bird species compositions of ponderosa pine sites to shift from similar species compositions based on location to similar species compositions based on tree thinning treatments and that breeding season bird species assemblages in ponderosa pine responded to thinning more so than did fall migration bird species assemblages. In contrast, bird species compositions of the pinyon-juniper sites showed little consistent response to the thinning treatments in the spring or fall.

## Rodents

Lists of all nocturnal rodent species encountered across all four study sites before tree thinning and after tree thinning, along with their summed counts per 3-night sample period, are presented in table 4 for the pinyon-juniper sites and in table 5 for the ponderosa pine sites. In total, 461 rodents represented by 9 species were recorded from the pinyon-juniper sites and 156 rodents representing 7 species were recorded from the ponderosa pine sites from 2008 through 2015. Diurnal rodents such as chipmunks and squirrels were present at all study sites and several were captured in traps, but were not appropriately sampled with nocturnal live

traps, and are not included in the nocturnal rodent data records.

Numbers of all individual nocturnal rodents over all species recorded across the study sites in both the spring and fall of each year from 2008 through 2015 tended to be considerably higher at the pinyon-juniper sites (about 10 individuals per plot per 3-night sampling period) compared to the ponderosa pine sites (fewer than 5 individuals per plot per 3-night sampling period) (fig. 12). Overall rodent abundance at the pinyon-juniper sites was highest in spring 2009, then declined through 2015, with peaks in abundance in 2011 and again in 2013. At both pinyon-juniper sites, in 2008, 2009, and 2010, rodent abundance fluctuated between being lower and higher on the treatment plots than on the control plots prior to tree thinning. But from 2011 on, rodent abundance was primarily higher on the control plots following tree thinning, especially at the PJ1 site (fig. 12). Rodent abundance at the ponderosa pine sites was similar between control and treatment plots over time, with shifts back and forth between greater abundance at control or treatment plots, but there was no distinct change in rodent abundance that might be attributed to tree thinning treatment effects (fig. 12).

The most abundant and relatively consistent rodent species found at the pinyon-juniper sites was the pinyon mouse (see tables 4 and 5 for scientific names of all rodents mentioned in this section). Other common rodents found at the pinyon-juniper sites included the deer mouse, white-footed mouse, white-throated woodrat, and silky pocket mouse. Rare rodent species found at the pinyon-juniper sites included the Mexican vole, northern grasshopper mouse, plains pocket mouse, and plains woodrat (table 4). The most common rodent species typically found at the ponderosa pine sites was the deer mouse, along with pinyon mouse; brush mouse, Mexican vole, western harvest mouse, white-footed mouse, and white-throated woodrat were recorded rarely (table 5).

The pinyon mouse was not only the most abundant rodent species at the pinyon-juniper sites, but it was also the only species to significantly decline on the treated plots following thinning treatments at both pinyon-juniper sites based on Chi-Square tests (PJ1 site,  $p = 0.002$ ; PJ2 site,  $p < 0.0001$ ). The deer mouse

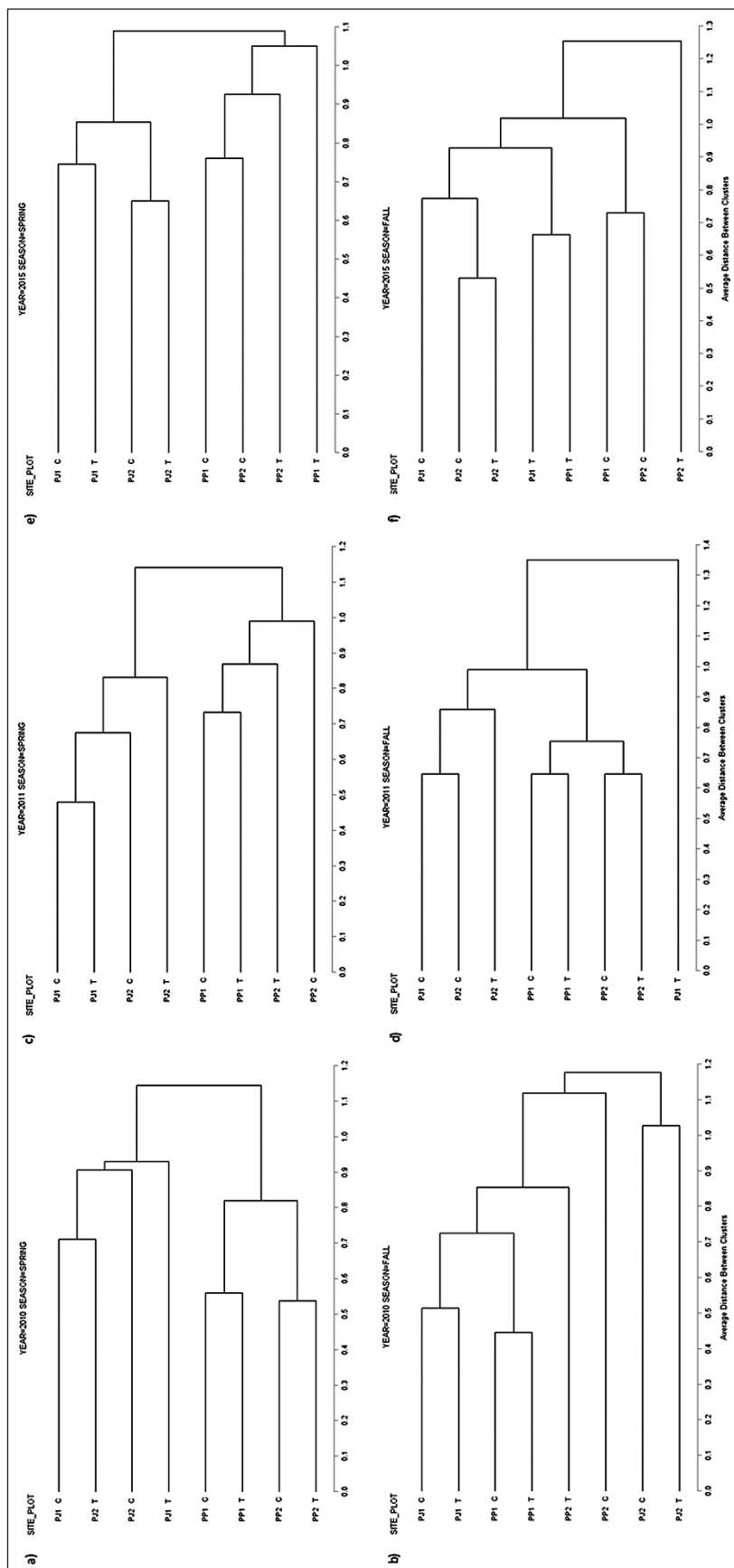


**Figure 11**—Cluster analysis results showing the similarity of sites and paired plots based on similarity of the bird species compositions in (a) spring 2010, (b) fall 2010, (c) spring 2011, (d) fall 2011, (e) spring 2015, and (f) fall 2015.

**Table 4**—Pinyon-juniper rodent species and mean counts of individuals per species averaged over counts per 3-day sample period, averaged over the seasons and years prior to tree thinning from fall 2008 through fall 2010, and averaged over the seasons and years after tree thinning from spring 2011 through fall 2015.

| Nocturnal rodent species on pinyon-juniper sites |                               |              | Before thinning, 2008–2010 |            |             |            | After thinning, 2011–2015 |            |             |            |
|--------------------------------------------------|-------------------------------|--------------|----------------------------|------------|-------------|------------|---------------------------|------------|-------------|------------|
| Common name                                      | Scientific name               | Species code | PJ1                        |            | PJ2         |            | PJ1                       |            | PJ2         |            |
|                                                  |                               |              | Contro<br>l                | Treat<br>d | Contro<br>l | Treat<br>d | Contro<br>l               | Treat<br>d | Contro<br>l | Treat<br>d |
| Deer mouse                                       | <i>Peromyscus maniculatus</i> | PEMA         | 2.0                        | 1.0        | 5.3         | 3.0        | 2.0                       | 1.0        | 1.6         | 4.0        |
| Mexican vole                                     | <i>Micropus mexicanus</i>     | MIME         | 0.0                        | 0.0        | 0.3         | 0.0        | 0.0                       | 0.0        | 0.0         | 0.0        |
| Northern grasshopper mouse                       | <i>Onychomys leucogaster</i>  | ONLE         | 0.0                        | 1.0        | 0.0         | 0.0        | 0.0                       | 0.0        | 0.0         | 0.0        |
| Pinyon mouse                                     | <i>Peromyscus truei</i>       | PETR         | 6.0                        | 8.5        | 4.4         | 6.4        | 7.4                       | 4.0        | 6.1         | 2.8        |
| Plains pocket mouse                              | <i>Perognathus flavescens</i> | PEFL2        | 0.0                        | 0.0        | 0.0         | 0.0        | 0.0                       | 0.0        | 0.0         | 1.0        |
| Plains woodrat                                   | <i>Neotoma micropus</i>       | NEMI         | 0.0                        | 0.0        | 0.0         | 0.0        | 0.0                       | 0.0        | 0.0         | 2.0        |
| Silky pocket mouse                               | <i>Perognathus flavus</i>     | PEFL         | 0.0                        | 0.0        | 1.0         | 1.5        | 1.0                       | 0.0        | 1.0         | 1.0        |
| White-footed mouse                               | <i>Peromyscus leucopus</i>    | PELE         | 5.0                        | 5.5        | 2.0         | 2.0        | 1.0                       | 1.0        | 1.7         | 1.0        |
| White-throated woodrat                           | <i>Neotoma albigula</i>       | NEAL         | 0.0                        | 0.0        | 0.0         | 1.0        | 1.5                       | 2.0        | 1.0         | 1.0        |
| Column average                                   |                               |              | 1.4                        | 1.8        | 1.5         | 1.5        | 1.4                       | 0.9        | 1.3         | 1.4        |

**Table 5**—Ponderosa pine rodent species and mean counts of individuals per species averaged over counts per 3-day sample period, averaged over the seasons and years prior to tree thinning from fall 2008 through fall 2010, and averaged over the seasons and years after tree thinning from spring 2011 through fall 2015.

| Nocturnal rodent species on ponderosa pine (PP) sites |                                  |              | Before thinning, 2008–2010 |            |             |            | After thinning, 2011–2015 |            |             |            |
|-------------------------------------------------------|----------------------------------|--------------|----------------------------|------------|-------------|------------|---------------------------|------------|-------------|------------|
| Common name                                           | Scientific name                  | Species code | PP1                        |            | PP2         |            | PP1                       |            | PP2         |            |
|                                                       |                                  |              | Contro<br>l                | Treat<br>d | Contro<br>l | Treat<br>d | Contro<br>l               | Treat<br>d | Contro<br>l | Treat<br>d |
| Brush mouse                                           | <i>Peromyscus boyleyi</i>        | PEBO         | 0.0                        | 0.0        | 0.0         | 0.0        | 0.0                       | 1.0        | 1.0         | 1.0        |
| Deer mouse                                            | <i>Peromyscus maniculatus</i>    | PEMA         | 5.5                        | 1.3        | 3.0         | 2.7        | 1.5                       | 1.8        | 2.5         | 2.4        |
| Mexican vole                                          | <i>Micropus mexicanus</i>        | MIME         | 0.0                        | 0.0        | 0.0         | 0.0        | 1.0                       | 1.0        | 0.0         | 0.0        |
| Pinyon mouse                                          | <i>Peromyscus truei</i>          | PETR         | 2.5                        | 6.5        | 4.0         | 1.5        | 1.4                       | 1.4        | 2.0         | 0.0        |
| Western harvest mouse                                 | <i>Reithrodontomys megalotis</i> | REME         | 0.0                        | 0.0        | 0.0         | 0.0        | 0.0                       | 0.0        | 1.0         | 1.0        |
| White-footed mouse                                    | <i>Peromyscus leucopus</i>       | PELE         | 2.0                        | 1.0        | 0.0         | 2.0        | 1.0                       | 0.0        | 1.0         | 0.0        |
| White-throated woodrat                                | <i>Neotoma albigula</i>          | NEAL         | 0.0                        | 0.0        | 0.0         | 0.0        | 0.0                       | 0.0        | 1.0         | 1.0        |
| Column average                                        |                                  |              | 1.4                        | 1.3        | 1.0         | 0.9        | 0.7                       | 0.8        | 1.2         | 0.8        |

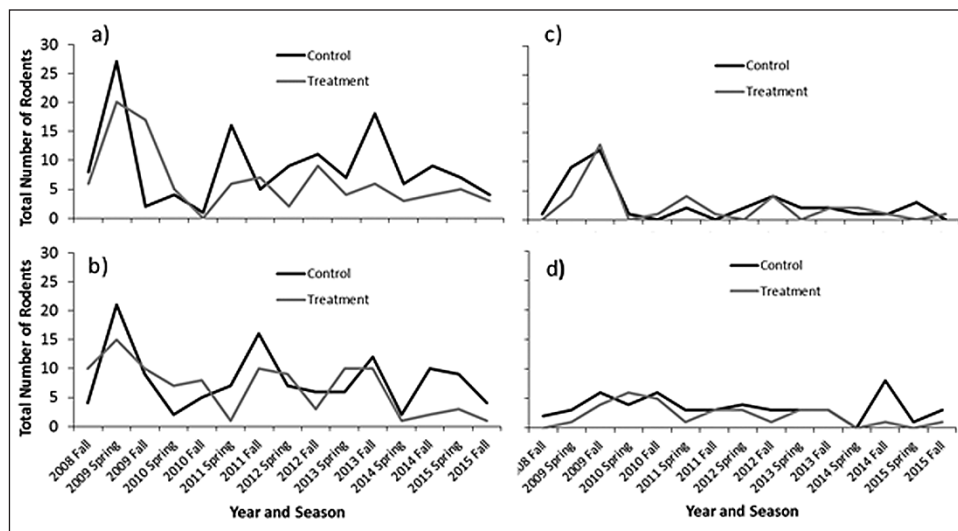


**Figure 12**—Total number of rodents over 3-night live trapping periods in the spring and fall on (a,b) the two pinyon-juniper (PJ1 and PJ2) and (c,d) the two ponderosa pine (PP1 and PP2) sites, 2010–2015.

also significantly declined at the PJ1 site ( $p = 0.04$ ) after thinning treatments, but deer mice were significantly more abundant on the treated plot at the PJ2 site ( $p = 0.02$ ) following thinning treatments. Otherwise, there were no significant differences in rodent numbers between control and treated plots at the pinyon-juniper sites over the pretreatment, or over the posttreatment, periods. Deer mice numbers were significantly lower on the PP2 site treatment plot ( $p = 0.006$ ) compared to the control plot, but only following thinning treatments, and there were no significant differences in rodent numbers between control and treated plots at the ponderosa pine sites over the pretreatment, or over the posttreatment, periods.

Cluster analysis of similarities or differences in rodent species compositions between site locations and between control vs. treatment plots in 2010, 2011, and 2015, showed that in 2010 prior to thinning treatments, rodent assemblages across the pinyon-juniper and the ponderosa pine sites had similar species compositions and that there were no clear groupings by forest type or treatments (fig. 13). In spring 2011 immediately after tree thinning, rodent assemblages at the two pinyon-juniper sites were most similar based on treatment type, not site location, while all ponderosa pine site control and treatment plots were more similar by forest type (fig. 13). In fall 2011, the treatment plots from the two pinyon-juniper sites still were similar in rodent compositions, while the pinyon-juniper control plots

were very different from each other and the treatment plots, and all of the ponderosa pine sites still grouped together (fig. 13). Five years after tree thinning treatments, rodent assemblages no longer grouped by location or by treatment. The PJ1 site control and treatment plots grouped together and with the PP1 control plot, while the PJ2 site plots grouped with the PP2 and PP1 plots in both the spring and fall (fig. 13). These findings show that the thinning treatments changed the species compositions at the pinyon-juniper sites 1 year after thinning, but that pattern had disappeared by 5 years since thinning treatments.

### *Medium-sized to Large Animals*

Over 2,000 animals were recorded from the wildlife camera traps that were operated year-round from 2012 through 2015. However, many of the animals photographed on study plots at the pinyon-juniper sites were domestic livestock, primarily cattle and some horses. Many small animal species also were captured on the wildlife cameras, but were often difficult to identify, so we excluded analysis of small animals such as chipmunks, squirrels, and small birds. We did include wild turkeys (see table 6 for scientific names of wildlife species mentioned in this section) because of their large size and ease of identification in photographs.

Cameras at the ponderosa pine sites recorded more native animals (284 individuals) than cameras at the

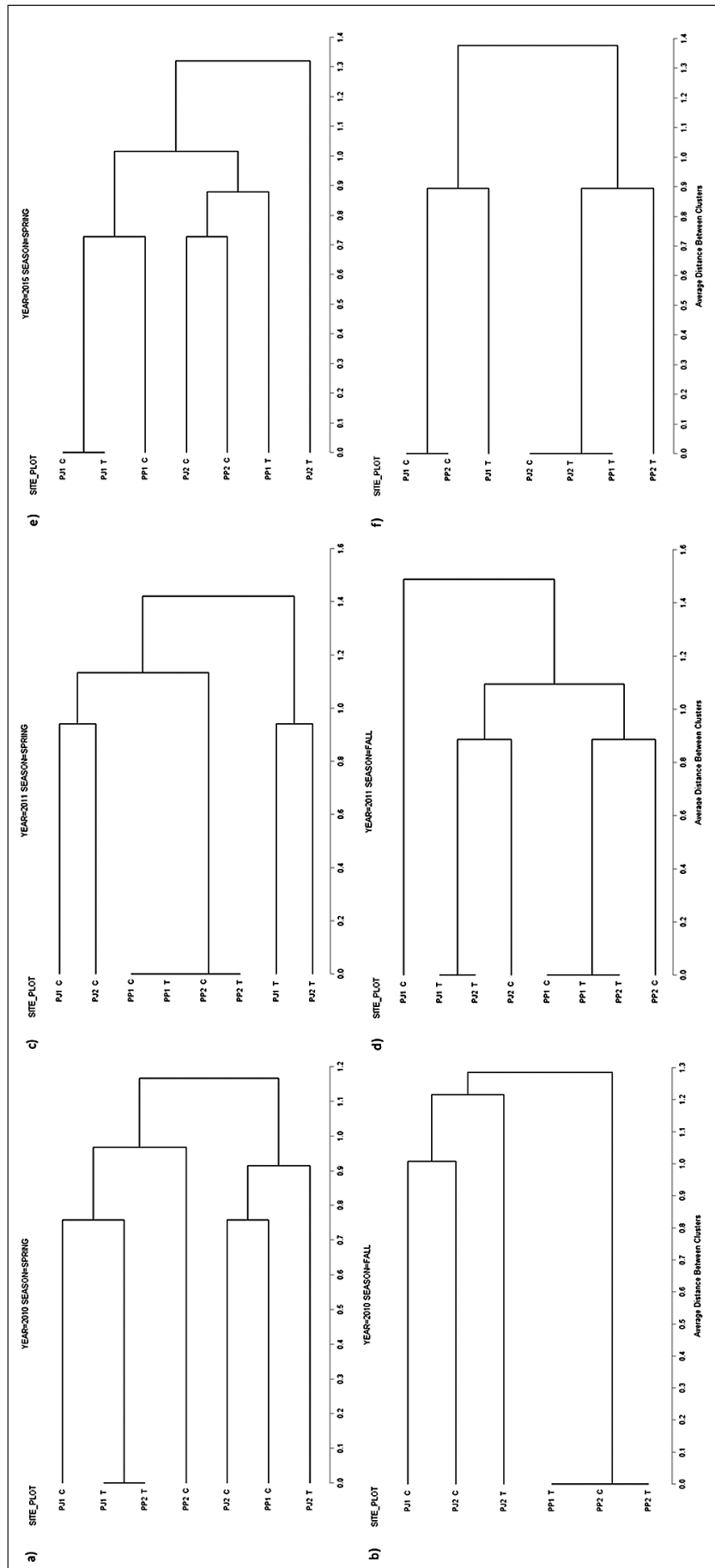


**Figure 13**—Cluster analysis results showing the similarity of sites and paired plots based on similarity of the rodent community species compositions in (a) spring 2010, (b) fall 2010, (c) spring 2011, (d) fall 2011, (e) spring 2015, and (f) fall 2015.

**Table 6**—Wildlife species and counts summed over the 4-year period 2012 through 2015 from camera traps at pinyon-juniper (PJ) and ponderosa pine (PP) sites following tree thinning treatments in winter 2010–2011.

| Native wildlife species |                                 | Pinyon-juniper sites |         |         |         | Ponderosa pine sites |         |         |         | Row sum |
|-------------------------|---------------------------------|----------------------|---------|---------|---------|----------------------|---------|---------|---------|---------|
| Common name             | Scientific name                 | PJ1                  |         | PJ2     |         | PP1                  |         | PP2     |         |         |
|                         |                                 | Control              | Treated | Control | Treated | Control              | Treated | Control | Treated |         |
| American black bear     | <i>Ursus americanus</i>         | 0                    | 0       | 1       | 0       | 4                    | 2       | 4       | 0       | 11      |
| Black-tailed jackrabbit | <i>Lepus californicus</i>       | 9                    | 7       | 17      | 0       | 3                    | 0       | 2       | 0       | 38      |
| Bobcat                  | <i>Lynx rufus</i>               | 0                    | 0       | 0       | 0       | 3                    | 0       | 0       | 0       | 3       |
| Common raccoon          | <i>Procyon lotor</i>            | 0                    | 0       | 0       | 0       | 1                    | 1       | 0       | 0       | 2       |
| Coyote                  | <i>Canis latrans</i>            | 31                   | 6       | 19      | 1       | 4                    | 7       | 4       | 0       | 72      |
| Desert cottontail       | <i>Sylvilagus auduboni</i>      | 0                    | 1       | 3       | 0       | 22                   | 1       | 2       | 0       | 29      |
| Gray fox                | <i>Urocyon cinereoargenteus</i> | 1                    | 0       | 0       | 0       | 1                    | 0       | 7       | 2       | 11      |
| Mule deer               | <i>Odocoileus hemionus</i>      | 1                    | 3       | 5       | 5       | 13                   | 7       | 89      | 40      | 163     |
| North American elk      | <i>Cervus canadensis</i>        | 3                    | 0       | 0       | 0       | 0                    | 0       | 7       | 2       | 12      |
| Wild turkey             | <i>Meleagris gallopavo</i>      | 0                    | 13      | 60      | 33      | 32                   | 22      | 1       | 1       | 162     |
| Column sum              |                                 | 45                   | 30      | 105     | 39      | 83                   | 40      | 116     | 45      | 503     |

pinyon-juniper sites (219 individuals), and many more native animals were recorded on the control plots than on the treatment plots across both the pinyon-juniper and the ponderosa pine sites (fig. 14).

The most abundant animal recorded at the two pinyon-juniper sites was wild turkey, which were recorded only from the treatment plot at the PJ1 site, but were abundant on the control plot at the PJ2 site (table 6). Chi-Square tests for differences in animal numbers between control and treatment plots within each site (table 6) showed that over the posttreatment period from 2012 to 2015, wild turkeys were significantly more abundant on the treated plot than the control plot at the PJ1 site ( $p = 0.0003$ ), but were significantly more abundant on the control plot at the PJ2 site ( $p = 0.005$ ). The coyote was the next most abundant animal species at the pinyon-juniper sites, and coyotes were significantly more abundant on the control plots at both sites compared to the treated plots (both PJ1 and PJ2 sites,  $p < 0.0001$ ). Black-tailed jackrabbits were significantly more abundant on the control plot at the PJ2 site ( $p < 0.0001$ ), but not significantly different

between control and treatment plots at the PJ1 site. Otherwise, there were no significant differences in the numbers of large animals between the control and treatment plots from 2012 through 2015 at either pinyon-juniper site.

Mule deer were the most abundant animal photographed at the ponderosa pine sites, and they were significantly more abundant on the control plot than on the treated plot at the PP2 site ( $p < 0.0001$ ), but not significantly different at the PP1 site. Wild turkeys were common at the PP1 site and rare at the PP2 site, but numbers were not significantly different between control and treatment plots at either site. Coyotes and black bears were significantly more abundant on the control plot at the PP2 site (both  $p = 0.04$ ), but not significantly different between plots at the PP1 site, and desert cottontails were significantly more abundant on the control plot at the PP1 site ( $p < 0.0001$ ), but not different between plots at the PP2 site. Otherwise, there were no significant differences in the numbers of large animals between the control and treatment plots from 2012 through 2015 at either ponderosa pine site.

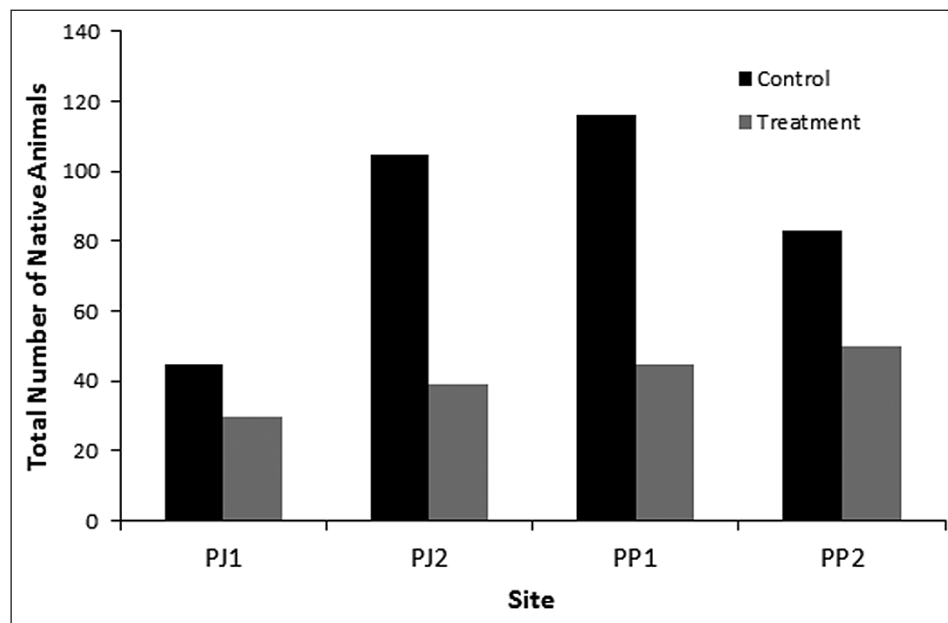


**Figure 14**—Total number of native medium to large mammals and wild turkeys recorded from camera traps on the control and treatment plots at the two pinyon-juniper (PJ1 and PJ2) and the two ponderosa pine (PP1 and PP2) sites following tree thinning treatments, 2012–2015.

Domestic livestock were considerably more abundant at the pinyon-juniper sites from 2012 through 2015 (511 animals) than at the ponderosa pine sites (145 animals); at all sites except the PP1 site where grazing was supposed to be excluded (60 trespass animals), livestock were much more abundant on the treated study plots (fig. 15). Chi-Square tests revealed that significantly more livestock occurred on the treatment plots than on the control plots from 2012 through 2015 at both pinyon-juniper sites ( $p < 0.0001$ , both PJ1 and PJ2 sites), and at one ponderosa pine site ( $p = 0.0001$ , PP1 site), but not at the other ponderosa pine site ( $p = 0.74$ , PP2 site).

## DISCUSSION

The tree thinning treatments accomplished the primary forest restoration objectives of reducing tree stand and fire fuel densities; changing stand structures from a dominance of small and young trees to stands dominated by fewer, older, and larger trees with some small patches of dense trees; and creating more open habitats and increased understory vegetation foliage cover. Responses of wildlife to the thinning treatments were generally neutral or positive for birds and neutral or negative for rodents, and generally negative for large mammals and wild turkey. Additionally, we found that domestic livestock were much more abundant on treated study plots, apparently in response to increased

production of understory vegetation. Three years of prethinning baseline data followed by 5 years of post-treatment data allowed us to understand the short-term (i.e., 5 years) temporal dynamics of tree thinning effects on vegetation and wildlife in both pinyon-juniper and ponderosa pine woodlands.

## Wildlife Habitat

### Trees

The New Mexico State Forestry treatment prescriptions applied in our study were similar to tree thinning prescriptions used across the Southwest in both pinyon-juniper and ponderosa pine stands (Allen et al. 2002, 2008; Fulé et al. 1997; Savage et al. 2008) and were designed to renew ecosystem structure and function and reverse unhealthy forest characteristics. The treatments were successful in reducing stand basal area on both pinyon-juniper and ponderosa pine sites, and in creating small clustered groups of trees of variable diameter, with small-diameter trees removed within the dripline of larger trees. The prescription created more open stands with reduced canopy cover and increased distance between groups of trees in both woodland types. This “thin from below” treatment approach is a common method employed by land managers for restoration (Fulé et al. 2002); by removing smaller understory trees, both old-growth

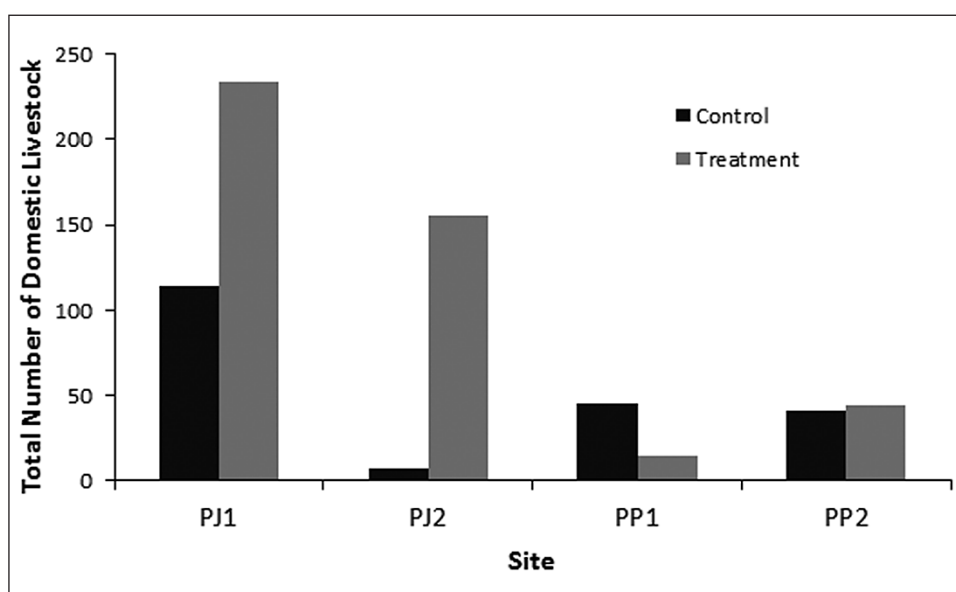


**Figure 15**—Total number of domestic livestock (cattle and horses) recorded from camera traps on the control and treatment plots at the two pinyon-juniper (PJ1 and PJ2) and the two ponderosa pine (PP1 and PP2) sites following tree thinning treatments, 2012–2015.

trees and appropriate diameter class distribution are retained (Kaufman et al. 1998). The resulting stands had more mature individuals and variable openings, a common outcome for most Southwestern forest restoration prescriptions, particularly for ponderosa pine-dominated stands (Wightman and Rosenstock 2008). The more open crown structure impacts the subcanopy environment by removing the canopy's filtering effect on sunlight and precipitation (Naumburg and DeWald 1999), which alters habitat for some animals and understory plant species. The greater proportion of larger, healthier trees is expected to increase tree productivity in the long term as a result of reduced competition, thus improving resiliency to wildfire, drought, insects, and disease (Allen et al. 2010, 2015).

The fuel reduction aspects of the treatment prescriptions were designed to reduce the potential for severe wildfire (Feeney et al. 1998; Keane et al. 2002; Sala et al. 2005). When crown spacing is increased, density is reduced in the lower canopies, and ladder fuels are reduced, a wildfire is likely to burn with lower intensity and thereby cause lower mortality to overstory trees (Agee and Skinner 2005), which in turn reduces loss of habitat for wildlife. Restoration treatments in Southwestern forest types can increase individual tree growth (Feeney et al. 1998; Fulé et al. 2007; Ronco et al. 1985; Skov et al. 2005), decrease tree water stress (Kolb et al. 1998; Skov et al. 2005; Wallin et al. 2004), increase tree defense against bark beetles through increased resin production (Kolb et al. 1998), and increase leaf nitrogen concentration and hence photosynthetic capacity in some cases (Feeney et al. 1998; Wallin et al. 2004; Zausen et al. 2005). These improvements in forest health are expected to have positive effects on wildlife habitat.

Tree thinning also may potentially affect wildlife food sources. Twoneedle pinyon is a masting plant species that produces large crops of cones and seeds sporadically about every 5 to 7 years (Janetski 1999; Zlotin and Parmenter 2008), and juniper berries are produced almost every year but in variable densities (Chambers et al. 1999). Our finding of greater cone production (and possibly larger seeds) on the pinyon trees that remained after treatment, compared to pinyon trees on control plots at one of two sites, indicates that tree thinning may have reduced competition for some

remaining larger trees, allowing for greater reproductive output. We are not aware of any other studies that have addressed tree thinning effects on pinyon cone and seed production, and we recommend that others conduct such studies. If tree thinning does increase pinyon and perhaps ponderosa pine cone and seed production, the implications are important to animal species such as the many birds, rodents, and larger mammals, including bears, coyotes, and foxes, that utilize pine seeds as a food resource.

### *Understory Vegetation*

The strong positive response of herbaceous understory vegetation to tree thinning in our study probably resulted from increased plant growth due to increased soil moisture from reduced tree canopy interception and evaporation of precipitation, less water uptake by tree roots, and perhaps the mulching effects of wood chips. Perennial grasses and forbs that were in place before tree thinning and annual forbs sprouting from seeds after thinning accounted for the increase in herbaceous plant cover following tree removals. Understory vegetation responses to tree thinning treatments in dry forests are thought to be driven by changes in availability of limiting resources, primarily nitrogen and water (Ares et al. 2010; Coomes and Grubb 2000; Miller and Seastedt 2009). Abella et al. (2015) concluded that increased species richness and cover following tree thinning are variable and depend on soil parent material, the specifics of thinning implementation, and the presence or exclusion of livestock grazing.

Other research in pinyon-juniper woodlands has found the same response patterns (Albert et al. 2004; Brockway et al. 2002; Huffman et al. 2013; Jacobs 2015; Matchett et al. 2010; Owen et al. 2009; Ramirez et al. 2008; Stephens et al. 2016). Stephens et al. (2016) concluded that understory vegetation responses to pinyon-juniper thinning depend on the type of treatment used, but that all treatments that reduce tree cover increase understory vegetation cover. Brockway et al. (2002) studied the effects of juniper removal by mechanical mastication (not chipping) on understory vegetation in pinyon-juniper savanna near our study area and compared the effects of removing, piling, and spreading slash. They found that herbaceous

vegetation cover and diversity significantly increased when all three methods were used, but spreading the slash had the most beneficial effect; they recommended mulching with slash over the other treatments.

Jacobs (2015) found total understory cover increased several-fold 3 to 5 years posttreatment. Matchett et al. (2010) evaluated short-term effects of thinning methods on southwestern pinyon-juniper woodlands and found that thinning treatments with slash pile/burn and mastication increased the abundance of herbaceous vegetation; pretreatment tree dominance dictated the strength of the increase. Matchett et al. (2010) concluded that thinning-induced increases in perennial grass cover in areas of high tree dominance were mainly due to an increase in growth of individuals present before the treatment, as opposed to an increase due to the recruitment of new individuals, which corresponds to our findings.

The same findings apply to ponderosa pine woodlands (Abella 2009; Fulé et al. 2001; Jacobs 2015; Laughlin and Fulé 2008; Miller and Seastedt 2009; Moore et al. 2006; Stoddard and McGlone 2008; Stoddard et al. 2011; Thomas and Waring 2015). Korb and Springer (2003) concluded that understory productivity in ponderosa pine forests is inversely related to the density of the overstory trees. Thinning treatments to reduce overstory density have repeatedly been shown to increase understory productivity, particularly when pretreatment stands are dense (Bedunah et al. 1988; Metlen and Fiedler 2006; Moore and Deiter 1992).

Invasion and colonization of treated sites by exotic weeds is a potential negative consequence of tree thinning. As discussed earlier, minimizing soil surface disturbance during treatments and mulching with wood chips or scattered slash are effective ways to reduce exotic weed invasion (Miller and Seastedt 2009; Owen et al. 2009). The tree thinning treatments imposed in our study did not lead to increases in exotic invasive weed species. Many exotic invasive weed species occurred along roads and other disturbed locations across our study area, including prickly Russian thistle, kochia (burning bush) (*Bassia scoparius*), white horehound (*Marrubium vulgare*), and cheatgrass. Over the 8-year study, we found only four exotic invasive weed species at only one treatment plot following tree removals: prickly Russian thistle, common mullein,

redstem stork's bill, and cheatgrass. We attribute the lack of exotic invasive weed increases to very little soil surface disturbance from the manual thinning methods used, along with the mulching effects of wood chips.

The exceptional drought that occurred over our region for 4 years after the removal of trees probably dampened the increases in herbaceous vegetation establishment and production compared to levels expected under average or wetter than average conditions. The study sites also were grazed by livestock, and the foliage canopies of herbaceous plants were reduced by an estimated 20 to 40 percent of what remained to be measured at the end of each annual growing season (based on livestock stocking rates and our observations). Therefore, the significant increase in understory herbaceous cover that we measured was likely to be an underestimate of the plant growth and standing biomass that would have been in place during nondrought conditions and without livestock grazing.

Smith (2011) found that precipitation is a strong determinant of understory response following thinning, concluding that long-term drought can compromise the ability of vegetation to respond to management. Climate influences on understory response have been discussed in many studies, with a general finding of a strong positive correlation between annual precipitation and understory productivity and diversity (Abella and Covington 2004; Bataineh et al. 2006; Fulé et al. 2002; Moore et al. 2006; Sabo et al. 2008). Climate change projections call for continued warming temperatures and intensified future droughts in the region (Llewellyn and Vaddey 2013), so we expect our findings to be indicative of tree thinning effects on vegetation for projected future conditions.

## Wildlife

### Birds

The neutral to positive responses of overall bird abundance to treatments at both the pinyon-juniper sites indicate that the small-scale manual tree thinning treatments had little effect on birds collectively. Some species that prefer more open woodland habitats such as the western bluebird, lark sparrow, northern flicker, and western kingbird increased significantly on

pinyon-juniper thinned locations compared to control locations. Despite severe drought conditions, tree thinning did not negatively affect bird abundance in either the pinyon-juniper or the ponderosa pine woodlands. Only one species, the brown creeper, declined on thinned locations at one ponderosa pine site. No bird species declined in response to thinning at either pinyon-juniper site.

These findings indicate that small-scale WUI-type thinning treatments do not have negative impacts on bird communities in the region, but probably increase bird diversity by adding open and edge habitats. These results are consistent with Bombaci and Pejchar's (2016) findings across a number of thinning studies in pinyon-juniper woodlands. In contrast, studies of bird responses to large-scale mechanical (e.g., chaining, blading, extraction) pinyon-juniper reduction treatments consistently have resulted in bird declines in treated areas (Bombaci and Pejchar 2016). Stephens et al. (2012) found that nonmechanical thinning and low severity fire both had positive effects on many more bird species than did mechanical thinning. Greater bird numbers during the fall migration period when many nonterritorial migrants were moving through the study areas also are consistent with results of other research on tree thinning effects on birds in pinyon-juniper woodlands (Bombaci and Pejchar 2016). Our findings of greater bird abundances and species richness at the pinyon-juniper sites compared to the ponderosa pine sites are consistent with Paulin et al. (1999) in Utah. These results indicate that the more diverse pinyon-juniper habitats support more species of birds than ponderosa pine and emphasize the importance of pinyon-juniper habitat to birds in central New Mexico.

Many studies have found that the removal of small-diameter trees typical of fuel reduction treatments has a neutral to positive effect on bird species (Gaines et al. 2010; Hurteau et al. 2008; Kalies et al. 2009; Manley et al. 2015; Stephens et al. 2012; Verschuyt et al. 2011; White et al. 2013). But studies have revealed that responses are generally species- or guild-specific or vary over time, attributable to the pace of vegetation response of the understory and overstory strata (Yegorova 2013). Kalies et al. (2009) found that, at the guild level, aerial foraging birds benefited from small-diameter tree removal, but had negative responses to large-diameter tree overstory removal, while ground

shrub-foraging birds responded positively to overstory removal. Woodpeckers, however, declined following overstory removals. Gaylord (2014) found that the occurrence probability of bark foragers and seed eaters was more closely associated with annual variability of food resources such as bark beetles, seed production, and composition of tree species. Foliage insectivores, which glean invertebrates from foliage of trees and shrubs, were associated with higher tree cover and fuel reduction that reduced cover of these species.

Hurteau et al. (2008) concluded that treatments to reduce forest fuels had little effect on avian diversity over 4 years, but did affect some aspects of species composition and abundance. Their results suggest that although the small-scale forest treatments they studied may have influenced the avian species present, natural annual variation in bird abundance was a stronger source of variation. Similarly, Szaro and Balda (1986) found that various intensities of forest thinning treatments influenced bird density and species richness, but treatments had a greater influence on community composition. White et al. (2013) used computer simulations to evaluate avian response to conifer forest fuel reduction treatments and found that treatment methods that increased stand structure complexity also increased overall avian species richness. Kalies et al. (2009) similarly found that a mosaic of forest conditions may be the most appropriate technique for providing suitable habitat for a wide range of forest passerines. They suggest that landscape-level forest treatments applied by land managers will have only modest effects on avian species.

The spring breeding season bird species compositions of pinyon-juniper woodlands are known to differ from those of breeding birds of ponderosa pine woodlands (Laudenslayer and Balda 1976; Paulin et al. 1999). Our cluster analysis results of spring breeding season bird assemblages confirmed that our pinyon-juniper and ponderosa pine sites had different bird species assemblages. In addition, bird assemblages at the ponderosa pine sites were affected more by the thinning treatments than those at the pinyon-juniper sites. Why bird species compositions of our pinyon-juniper sites were little affected by tree thinning is not clear, but is indicative of a greater change to the ponderosa pine habitats than to the pinyon-juniper habitats, relative to the different assemblages of birds that use those

respective habitat types. Kalies et al. (2009) reported that tree canopy reduction in ponderosa pine forests negatively affected bird communities, while Bombaci and Pejchar (2016) reported that tree canopy reductions in pinyon-juniper woodlands had little to positive effects on birds.

## Rodents

The weak response of rodent abundances to tree thinning at one pinyon-juniper site and both ponderosa pine sites, along with the significant declines of pinyon mice at both pinyon-juniper sites, and significant declines of deer mice at one pinyon-juniper and one ponderosa pine site, is consistent with previous research on the effects of tree thinning in pinyon-juniper woodlands. Bombaci and Pejchar (2016) reported that most of the studies they reviewed concluded that tree thinning in pinyon-juniper woodlands had largely neutral responses of rodents to tree reductions; the exception was when wood slash was left on the ground after thinning treatments, which caused increases in rodents. Pinyon-juniper woodland rodent species such as pinyon mice, deer mice, and woodrats (*Neotoma* spp.) tended to decline when trees were completely removed, but increased when slash or other wood was left on the ground following thinning treatments (Baker and Frischknecht 1973; Turkowski and Watkins 1976), especially when trees were thinned rather than completely removed (Severson 1986). In contrast, grassland specialist rodent species in pinyon-juniper savanna areas have been found to increase only when trees were completely removed (Severson 1986). Deer mice were the primary species found to increase among those studies when pinyon-juniper was thinned, but only if wood slash was left on the ground (Albert et al. 1995; Sedwick and Ryder 1987; Severson 1986).

Studies of rodent responses to tree thinning in ponderosa pine woodlands also have yielded results similar to ours. Several studies found early and positive responses of small forest-floor mammals to thinning (Converse et al. 2006a,b,c; Muzika et al. 2004; Sullivan et al. 2005; Suzuki and Hayes 2003; Wilson and Carey 2000; Wilson and Forsman 2013; Zwolak 2009). Some researchers suggest that increases in herbaceous and shrub cover following thinning enhance habitat for some rodents (Bagne and Finch 2010; Block et al. 2005; Carey and Johnson 1995; Converse

et al. 2006b). Positive responses have generally been strongest in forests that originally lacked understory cover and shrub components (Wilson and Forsman 2013).

Bagne and Finch (2010) studied rodent responses to tree thinning in ponderosa pine woodlands of the Santa Fe watershed, and deer mice increased in some habitats where wood slash was left on the ground following tree reductions and also increased in response to greater precipitation. Other studies also have attributed positive effects of ponderosa pine tree thinning for rodents to increases in downed woody debris left on the ground after thinning (Converse et al. 2006a,b; Manning and Edge 2004; Suzuki and Hayes 2003) and to increases in herbaceous understory plants following tree reductions (Converse et al. 2006a; Manning and Edge 2004).

Increases in overall rodent habitat heterogeneity resulting from tree reductions have also been reported (Carey and Wilson 2001; Muzika et al. 2004). Stephens et al. (2012) found that rodents tended to respond positively to mechanical thinning and negatively to nonmechanical thinning and low severity fire across a variety of forest types, probably due to residual slash left in place after mechanical thinning. In contrast, McIver et al. (2013) suggested that across a broad spectrum of woodland and forest ecosystems, treatment responses by rodents tended to be subtle or nonexistent, and any changes in rodent abundance that did occur were subtle and transient, lasting only 1 to 3 years.

The Northern Arizona University Ecological Restoration Institute (2010) reviewed literature and examined posttreatment time period effects on rodents in ponderosa pine woodlands and concluded that vegetation density was an important factor for four key rodent species, and that species associated with dense plant cover were the only ones to increase concurrently with increased vegetation density. The presence of slash piles and longer duration of the slash pile presence produced positive occupancy responses from all rodent species. The presence of downed wood or slash is especially important for some species (Chambers 2002; Converse et al. 2006a), particularly deer mice, because the animals use slash habitats for cover, nesting, and food (e.g., more insects). The Northern Arizona University study acknowledged

the importance of downed wood as a habitat feature for some members of the small mammal community, but concluded that the presence of downed wood is less important than overstory and understory vegetation composition and structure. Thinning operations also may open forests, increasing the success of predators hunting small mammals (Gese et al. 1996). The reduction in trees along with the removal of downed woody materials apparently had little to no effect on woodland rodents in our study, especially at the ponderosa pine sites.

The one negative response at the PJ1 site was primarily due to a decline in pinyon mice following tree reduction, as Severson (1986) and Albert et al. (1995) found in southwestern and northwestern New Mexico, respectively. Pinyon mice are specialists of pinyon-juniper woodlands and primarily use the trees as habitat (Albert et al. 1995; Findley et al. 1975; Holbrook 1978). Severson (1986) found that pinyon mice declined when trees were removed, even if slash was left on the ground, while pinyon mice increased where some trees were left and slash remained on the ground. Most other rodents increased in response to slash piles.

Our PJ1 site had a higher density of pinyon trees than the PJ2 site, and the proportionately greater loss of trees at the PJ1 site apparently had a proportionately greater impact on pinyon mice than at the PJ2 site. Another important difference between the two pinyon-juniper sites was that the landowner at the PJ2 site also left some slash in a small drainage on the treated study plot. The slash probably provided habitat for generalist deer mice, which did increase slightly on that plot following the thinning treatment and were captured in traps near the slash. Grassland rodent species such as white-footed mice, silky pocket mice, western harvest mice, and plains woodrats were present in low numbers at both pinyon-juniper sites, and we had more captures of those species, especially at the PJ2 site, following tree thinning treatments. That response of grassland rodent species is consistent with the findings of others (O'Meara et al. 1981; Rodhouse et al. 2010; Severson 1986). Pinyon mice, which are largely arboreal foragers on pinyon trees (Albert et al. 1995; Findley et al. 1975; Holbrook 1978), declined following reductions of standing pinyon trees, while deer mice, which dominated at the ponderosa pine sites, did not respond to reductions in standing ponderosa pine trees or to the spreading of wood chips on the ground.

Higher numbers of rodents at both pinyon-juniper sites and at one ponderosa pine site at the onset of the study, and again toward the end of the study, in contrast with low numbers during the extreme drought period in 2011 through 2013, were probably in response to increased precipitation and plant production (i.e., food resources). Rodents in the Southwest are known to respond positively to the bottom-up effects of increased rainfall and plant production, and populations typically increase fairly rapidly within 1 year of increased plant production (Ernest et al. 2000; Lightfoot et al. 2012). Bagne and Finch (2009) concluded that precipitation interacted with tree thinning treatments to affect deer mice abundance. Little change in rodent species compositions following thinning treatments in cluster analysis corresponds with the associated lack of change in rodent numbers relative to the thinning treatments. Our pinyon-juniper sites were dominated by pinyon mice and the ponderosa sites were dominated by deer mice; the other rodent species were probably at low enough numbers to have little effect on the species similarity values in cluster analysis. Overall, the tree thinning treatments in our study had little effect on rodent abundances or on rodent species compositions, but there is evidence for individual species responses based on an increase or a decrease in their individually preferred habitat resources, especially the decline in pinyon mice after tree thinning.

### *Medium-sized to Large Animals*

The medium-sized to large mammals and wild turkeys that declined in abundance (based on camera trap photograph frequency) following tree thinning treatments may have reacted to a change in habitat structure that created more open stands of trees with fewer trees for protective cover. Mule deer, coyotes, black-tailed jackrabbits, and wild turkey all have preferences for mixes of relatively open habitat, along with vegetation cover for visual protection from predators (Bender et al. 2013; Findley 1987; Findley et al. 1975; Kennedy and Fontaine 2009; New Mexico Department of Game and Fish 2013). Turkeys additionally require surface water sources, trees for roosting, and summer brood areas (New Mexico Department of Game and Fish 2013). Given that camera installation and data collection on control and treatment plots did not begin until 2012, we cannot be certain that the differences in animal abundances between treatment and control

plots did not exist before the treatments, but the high degree of control vs. treatment differences and the consistency of the pattern across all four study sites indicate that lower numbers of native animals on thinned study plots were a response to tree thinning. Another consideration relative to the results of our study was the unaccounted-for effects of the landscape habitats surrounding our study sites. Large wildlife species tend to have larger territories and movement patterns relative to our small study sites, and the surrounding landscape habitats probably affected the behavior of large animals using the smaller study site patches. We could not control for surrounding landscape effects, so those remain unknown but likely factors affecting the animals that did move through our study sites.

Relatively little research has been conducted on tree thinning effects on larger mammals and wild turkey. Albert et al. (1995) found that tree thinning in pinyon-juniper woodlands in northwestern New Mexico increased mule deer use based on fecal pellet counts. Kramer et al. (2014) found that tree thinning and livestock exclusion from savanna, pinyon-juniper, and ponderosa pine study sites in northeastern New Mexico caused an increase in mule deer forage species similar to our findings and concluded that tree thinning was beneficial for short-term improvements in mule deer forage. Watkins et al. (2007) emphasize the importance of woody shrubs as browse forage for mule deer, along with trees and forest edges for cover. Understory vegetation increased following thinning treatments in our study, yet mule deer abundance declined. In contrast, other research has shown increased mule deer abundance in larger-landscape-scale thinning studies in ponderosa pine woodlands (Thomas and Waring 2015). Because our study sites were on private lands with occasional human activity, tree cover may have been especially important to mule deer for visual cover, as Watkins et al. (2007) suggested. The relatively uniform open habitats created by the removal of trees in this study may have resulted in lower numbers of mule deer, coyotes, and other native animals. Watkins et al. (2007) also conclude that domestic livestock damage mule deer habitat by consuming vegetation and altering the vegetation composition and structure. Perhaps the great increase in livestock on our treated plots also had a negative effect on mule deer and other native wildlife.

### *Implications of Pinyon-Juniper Tree Thinning Treatments to Forest Ecology and Management for Wildlife*

One of the objectives of tree thinning treatments for the Watershed Restoration Program was to improve habitat for wildlife. Based on the findings of this study, that objective was partially fulfilled. Animal species that prefer more open habitats became more abundant (e.g., western bluebirds and others), while species that prefer more tree canopy and vertical structure cover declined (e.g., brown creepers, pinyon mice, large mammals, and turkeys). Our findings of larger wildlife species declining in relation to tree thinning indicates that more small dense stands of trees should be left in place to provide cover for such species (Bender et al. 2013; Patton 2011). Animal diversity is well known to be a function of habitat or environmental diversity or heterogeneity (Patton 2011); the more types of habitat present in a given area, the more types of animal species that area will support. Edge effects also are important to many species, so creating edges of dense and sparse trees also increases habitat heterogeneity and species richness (Patton 2011).

Stephens et al. (2012) concluded that there is no single fuel reduction treatment method that will enhance habitat for all wildlife species, and that wildfire fuel reduction projects should aim to create a variety of vegetation successional stages to accommodate a range of habitat specialist wildlife species. In order to increase the abundance and diversity of wildlife relative to this program, we recommend that tree thinning treatments should maximize habitat diversity by retaining even more small patches of dense trees within the project areas than the thinning prescription called for, perhaps of about 0.25 to 0.5 ha in size for species that require some denser vegetation. Tree thinning project plans should consider what types of wildlife species are targeted for management, and then the tree thinning treatments should be planned to create habitat for those particular types of species, along with achieving the needs for wildfire fuel reductions.

In addition to habitat structure and cover, increasing understory vegetation and tree production increases food resources to animals. Understory vegetation cover and production increased significantly

in response to tree thinning treatments, apparently resulting in increases in wildlife that prefer greater understory vegetation and clearly increasing use by domestic livestock. If producing more productive understory vegetation for wildlife is part of a thinning project management plan, then the plan should have provisions for livestock grazing and management of livestock as one of the planned objectives. Our findings showed that livestock use increased dramatically on thinned pinyon-juniper plots. Such a finding is beneficial to livestock producers, but may not be appropriate for supporting wildlife species that also utilize productive understory vegetation.

Field research addressing vegetation and wildlife is affected considerably by weather conditions, before and during the study and for the short term (<10 years) and long term (>10 years). The entire New Mexico region experienced not only an extreme drought and increasing annual temperatures beginning at the onset of this study in 2011 and persisting through 2014 (U.S. Drought Monitor 2016), but also an extreme freeze event in the winter of 2011 (Western Regional Climate Center 2016). In that context, both vegetation and animals that we studied were under an umbrella of environmental stress conditions that probably dampened, or perhaps in some cases enhanced, potential responses of some species and overall biotic diversity and productivity relative to tree thinning treatments. Given that recent human-caused climate change is already affecting vegetation and wildlife (Chen et al. 2011; Fettig et al. 2013; Root et al. 2003), research on forest thinning activities should interpret data in the context of more variable weather conditions along with gradually increasing temperatures and their cumulative effects on ecosystem processes.

This study was conducted on only a subset of greater landscape topography, soils, and woodland types that are found in the region. The findings of this study may not apply to other landscapes, soils, and woodland or forest vegetation types in the region. Additionally, the small number of study sites and paired plots did not provide us with site replication needed for powerful statistical analyses and experimental testing on a broad landscape scale. Bombaci and Pejchar (2016) reviewed what is currently known about tree thinning effects on wildlife in pinyon-juniper woodlands, including

deficiencies in and needs for studies of wildlife responses to tree reductions. They recommend long-term studies (>10 years), large spatial scale studies (>100-ha plots), equal research effort among different treatment types, equal levels of research effort in different geographic regions, and more studies of amphibians, reptiles, and arthropods. However, there also is a need for more studies on smaller scale tree thinning projects such as WUI projects, along with larger scale studies for thinning projects that cover large areas of landscapes. We believe that this type of paired watershed study design is a good approach, as Neary (2016) concluded, but that more such studies are needed with greater replication of sampling locations over the long term (i.e.,  $\geq 10$  years), and broader coverage of landscape and forest or woodland types and tree thinning methods. Such an expansion of research will be expensive, but is the best way to understand the effects of forest tree thinning on wildlife across the Southwest.

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# Using the Assessment, Inventory, and Monitoring Strategy to Measure Treatment Effectiveness in Pinyon-Juniper Woodlands

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It is increasingly important to understand how different management actions affect public lands and renewable resources. In response to a lack of standardized monitoring, the Bureau of Land Management (BLM) developed the Assessment, Inventory, and Monitoring (AIM) strategy (Toevs et al. 2011). The AIM approach is based on five key elements: 1) a standardized set of core and contingent indicators, 2) a statistically valid sampling design, 3) a structured implementation process, 4) electronic data capture, and 5) integration with remote sensing. This strategy can help to more efficiently meet national, regional, and local resource information needs. The Taos Field Office (TFO) of the BLM has used the AIM protocol for 3 successive years to monitor vegetation treatment effectiveness and to inventory wildlife habitat. Here we discuss the implementation of AIM protocols and use AIM data to describe five, two-needle pinyon pine (*Pinus edulis*) and oneseed juniper (*Juniperus monosperma*) treatment areas in and around Taos, New Mexico. The use of AIM data shows us the complex consequences of these management actions and illustrates the advantages of ongoing monitoring at the field office level.

The BLM's AIM protocol includes both core and supplementary methods of data collection; core methods are used on every AIM plot and supplementary methods may be added at the project level to suit any additional data needs. More detailed information on

AIM methods and implementation can be found in Herrick et al. (2009) and Taylor et al. (2014). Core methods include: plot characterization, line point intercept (LPI), vegetation heights, species inventory, canopy gap intercept, and soil stability. Core methods were executed in the present study in addition to belt vegetation measurement transects and tree density measurement (AIM supplemental methods) to characterize ecological trends and treatment effectiveness. Data were collected at a total of 19 treatment plots from five individual treatment areas named A-E (four thinning plus prescribed burning and one with thinning only, table 1). Each treatment area varied in the date of treatment, method of burning/thinning and size. These 19 plots were then compared to 24 AIM plots from untreated sites within respective Ecological Sites (ES). Plots within the same ES share similar soil, vegetation, topography, and climate thus representing areas within the landscape that are likely to respond similarly to disturbances. Both treated and untreated plots consisted of a sample area of 2,827.4 square meters and were distributed across the landscape using a stratified, spatially balanced random sample design as described by Stevens and Olden (2004). The ES were used as strata for both the treated and untreated sample designs. Data collection from treated and untreated areas occurred over a span of three years from 2014-2016 during the peak plant growth seasons for each plot area.

**Table 1**—Description of each treatment area including methods of treatment, year of treatment, and size of treatment area. Each treatment area, A-E, is a geographically distinct area within the Rio Grande del Norte National Monument, within the BLM Taos Field Office, New Mexico.

| <b>Treatment area</b> | <b>Thinning</b> | <b>Pile burn</b> | <b>Broadcast burn</b> | <b>Jackpot burn</b> | <b>Size (acres)</b> |
|-----------------------|-----------------|------------------|-----------------------|---------------------|---------------------|
| A                     | 2005            | 2011             | 2011                  | 2014                | 646                 |
| B                     | 2013            | --               | --                    | 2015                | 191                 |
| C                     | 2013            | --               | --                    | 2014                | 39                  |
| D                     | 2015            | --               | --                    | --                  | 236                 |
| E                     | 2005            | 2008-2104        | --                    | --                  | 231                 |

In order to evaluate treated areas, data were reduced to the subset of indicators that best represented the management objectives of treatments. Data were then summarized within treatment areas and within the ES. These data included: tree density, number and size of canopy gaps, desirable species cover, and soil aggregate stability. Desirable species were defined as those species which are desired and preferred forage for wildlife species and cattle within each relevant Natural Resources Conservation Service (NRCS) Ecological Site Description (ESD). These ESDs included: F048AY015NM (Mixed Conifer), R048AY005NM (Mountain Malpais), R036XA004NM (Gravelly slopes), R036XB007NM (Malpais), and F036XA001NM (PJ-Oak Woodland) (Sylvester et al. 2003a,b; Wright and Bishop 2003). Since there is no published ESD for the PJ-Oak Woodland site, desirable species were defined from the similar ES Mountain Malpais. Data were analyzed using standard statistical methods without adjustment; these include calculating the mean values and standard deviation for tree density, canopy gaps, species cover, and soil stability values.

With the exception of the thinning only area (D) and the thinning and jackpot burn area (C) all other treated areas had a lower mean tree density than non-treatment areas within the same ES (table 2). Tree foliar cover from LPI shows a similar pattern.

Desirable species cover was the indicator that varied the most among ecological sites and treatment areas. Within area A across all ESs, the treated plots had a similar or higher average cover of desirable species when compared to untreated plots within respective ESs (table 3). Similarly, treatment areas B and E had slightly higher average values within the Mixed Conifer site compared to untreated plots. Conversely,

area C had significantly lower average cover of desirable species within the Mountain Malpais site when compared to untreated areas (5.1% compared to 23.3%).

When evaluating canopy gap size classes, almost all treatment areas across all ESs had significantly more large gaps (greater than 200cm) when compared to non-treated areas (table 4). The exception to this was treatment A in the Malpais and Mountain Malpais ESs, which had a similar proportion of large gaps compared to non-treated plots in the same ES. No significant differences in soil aggregate stability were found when comparing untreated and treated areas.

When comparing among treatment areas, area A, treated with thinning, pile, broadcast, and jackpot burning, was the most successful across all AIM indicators. This area achieved a reduction in tree density, had similar or greater average cover of desirable species compared to untreated areas, had similar amounts and distribution of large canopy gaps compared to untreated sites, and showed no signs of reduced soil stability. Despite similar precipitation, soils, and treatment methods this treatment has responded better than the four other treatment areas and provides better wildlife habitat and a more hydrologically stable site. The success of this treatment is most likely a product of the mosaic nature of the multiple treatment methods (a combination of thinning, pile, jackpot, and broadcast burning), the size of the treatment area, and the longer time to recover from treatment as compared to the other areas. Treatment area A underwent a combination of thinning and burning treatments spread out over 646 acres (table 1). This approach allowed a mosaic structure of treated and untreated patches providing cover and forage for wildlife species as well as a sufficient seed bank for recovery.

**Table 2**—Average tree density and standard deviations for each treatment area A-E (Table 1) and non-treated areas summarized by ecological site. An X represents an ecological site that was not found in the respective treatment area. Data were collected between May 2014 and October 2016. Non-treatment data were not collected from the Gravelly ecological site. Number of plots (n) within each treatment area and ecological site are shown below the average values.

| Ecological Site  | Tree Density (adult trees/hectare) |         |                |         |                |         |                |         |                |         |                 |         |
|------------------|------------------------------------|---------|----------------|---------|----------------|---------|----------------|---------|----------------|---------|-----------------|---------|
|                  | A                                  |         |                | B       |                |         | C              |         |                | D       |                 |         |
|                  | Average                            | Std Dev | Average        | Std Dev | Average        | Std Dev | Average        | Std Dev | Average        | Std Dev | Average         | Std Dev |
| Mixed Conifer    | 28.29<br>(n=2)                     | 0.00    | 0.00<br>(n=1)  | n/a     | X              | X       | X              | X       | 36.78<br>(n=5) | 34.16   | 114.36<br>(n=3) | 68.61   |
| Gravelly         | 0.00<br>(n=1)                      | n/a     | X              | X       | X              | X       | X              | X       | X              | X       | X               | X       |
| Malpais          | 0.00<br>(n=2)                      | 0.00    | X              | X       | X              | X       | X              | X       | X              | X       | 2.72<br>(n=13)  | 7.93    |
| Mountain Malpais | 10.61<br>(n=2)                     | 5.00    | 19.45<br>(n=2) | 7.50    | 73.09<br>(n=3) | 20.72   | X              | X       | X              | X       | 60.13<br>(n=3)  | 104.14  |
| PJ-Oak           | X                                  | X       | X              | X       | X              | X       | 74.27<br>(n=1) | n/a     | X              | X       | 38.91<br>(n=5)  | 23.46   |

**Table 3**—Average desirable species cover for each treatment area A-E (Table 1) and non-treated areas summarized by ecological site. An X represents an ecological site that was not found in the respective treatment area. Data were collected between May 2014 and October 2016. No non-treatment data were collected from the Gravelly ecological site. Number of plots (n) within each treatment area and ecological site are shown below the average values.

| Ecological Site  | Average of Desirable Species Cover |                |               |               |               | Untreated       |
|------------------|------------------------------------|----------------|---------------|---------------|---------------|-----------------|
|                  | A                                  | B              | C             | D             | E             |                 |
| Mixed Conifer    | 8.7%<br>(n=2)                      | 4.7%<br>(n=1)  | X             | X             | 3.9%<br>(n=5) | 1.6%<br>(n=3)   |
| Gravelly         | 4.7%<br>(n=1)                      | X              | X             | X             | X             | X               |
| Malpais          | 20.0%<br>(n=2)                     | X              | X             | X             | X             | 14.2%<br>(n=13) |
| Mountain Malpais | 21.7%<br>(n=2)                     | 12.7%<br>(n=2) | 5.1%<br>(n=3) | X             | X             | 23.3%<br>(n=3)  |
| PJ-Oak Woodland  | X                                  | X              | X             | 4.7%<br>(n=1) | X             | 0.46%<br>(n=5)  |

**Table 4**—Average proportion of plots with large canopy gaps (greater than 200cm) for each treatment area A-E (Table 1) and non-treated areas summarized by ecological site. An X represents an ecological site that was not found in the respective treatment area. Data were collected between May 2014 and October 2016. No non-treatment data were collected from the Gravelly ecological site. Number of plots (n) within each treatment area and ecological site are shown below the average values.

| Ecological Site  | % area with canopy gaps >200cm |              |              |              |              | Non-Treatment  |
|------------------|--------------------------------|--------------|--------------|--------------|--------------|----------------|
|                  | A                              | B            | C            | D            | E            |                |
| Mixed Conifer    | 32%<br>(n=2)                   | 40%<br>(n=1) | X            | X            | 33%<br>(n=5) | 21%<br>(n=3)   |
| Gravelly         | 0%<br>(n=1)                    | X            | X            | X            | X            | X              |
| Malpais          | 5%<br>(n=2)                    | X            | X            | X            | X            | 6%<br>(n=13)   |
| Mountain Malpais | 12%<br>(n=2)                   | 20%<br>(n=2) | 33%<br>(n=3) | X            | X            | 13%<br>(n=3)   |
| PJ-Oak Woodland  | X                              | X            | X            | 40%<br>(n=1) | X            | 23.7%<br>(n=5) |

Continued monitoring with the AIM protocols is important because it will allow land managers to understand the recovery of these treatments over time. The utility of these data could be improved by increasing sample sizes within treatment areas to add more power to conclusions drawn between different treatment methods. This is particularly important to BLM management as different treatment methods vary in terms of cost, practicality, and complexity.

These AIM data also support the idea that similar management actions will have varied results across different ecological sites. This is particularly evident in the treatment that was jackpot burned two years post-thinning, [treatment B] in which the same method of thinning and burning produced an increase in desirable species in one ES (Mixed Conifer) and a decrease in desirable species cover in another site (Mountain Malpais).

However, it is assumed that the untreated data provide a reliable proxy for the pretreatment condition. While this is the best available option when evaluating treatment effectiveness, there is still a large range of variability within the data from untreated sites. Moreover, the untreated data set did not cover all the relevant ecological sites that occurred in the treatment areas—the Gravelly site (R036XA004NM) did not have any untreated data. Continued monitoring of sites following treatment has the potential to better inform management in an adaptive framework. Ideally, it would also be possible to collect data using the standard AIM protocols before any management action occurs. Thus, AIM indicators could be used to directly assess pre and post treatment.

AIM provides an effective and standardized approach to assessing outcomes of various management actions within Pinyon-Juniper habitat types. In particular, AIM data provide a thorough assessment of ecosystem attributes that can inform evaluations of wildlife habitat and forage. Wildlife biologists can use the detailed data on understory vegetation cover and heights, species composition, soil surface cover, canopy gaps, and soil stability to accurately evaluate habitat suitability for many wildlife species, including game birds, pinyon-juniper obligate bird species, mule deer, and elk. Furthermore, supplemental indicators can be added to provide more specific data relevant to each wildlife species.

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## Field Trip to Sandia and Manzanita Mountains

On Friday, 14 October, the symposium sponsored a field trip to the Sandia and Manzanita Mountains. Craig Allen, Esteban Muldavin, and Bill Romme led the trip to the Elena Gallegos Picnic Ground, Otero Canyon, and Chamisoso Canyon. The trip focused on the local application of descriptions of broad piñon-juniper (*Pinus edulis*, *Juniperus* spp.) types in relation to historical fire regimes, restoration, and wildlife management (Fig. 1, Romme et al. 2009).

At Elena Gallegos, in the western foothills of the Sandia Mountains, the vegetation exemplified piñon-juniper savanna at the lower end of the elevation-moisture gradient where junipers are dominant (Fig. 2). Trip leaders discussed current and preferred management practices and the role of fire in this summer-precipitation-dominated ecosystem. They noted that because this site had been protected from grazing for several decades, the reintroduction of low intensity surface fire would be desirable to maintain

ecosystem processes, limit unnatural juniper establishment, and enhance wildlife habitat. However, management options were limited because of the location in a wildland-urban interface.

Otero Canyon, at higher elevation on the east side of the Manzanitas, exemplified persistent piñon-juniper woodland toward the upper end of a moisture-elevation gradient where tall piñons dominated over junipers (Fig. 3). Leaders discussed the evidence that natural fire frequency was considerably lower in this type, given limited fine fuels. Accordingly, wildlife habitat management practices would be very different here from that in a savanna, with focus on maintaining (rather than simplifying) complex stand structures and avoiding habitat fragmentation.

The final stop was a treated area in Chamisoso Canyon, at mid-elevation on the east side of the Manzanitas. Based on an adjacent untreated stand,

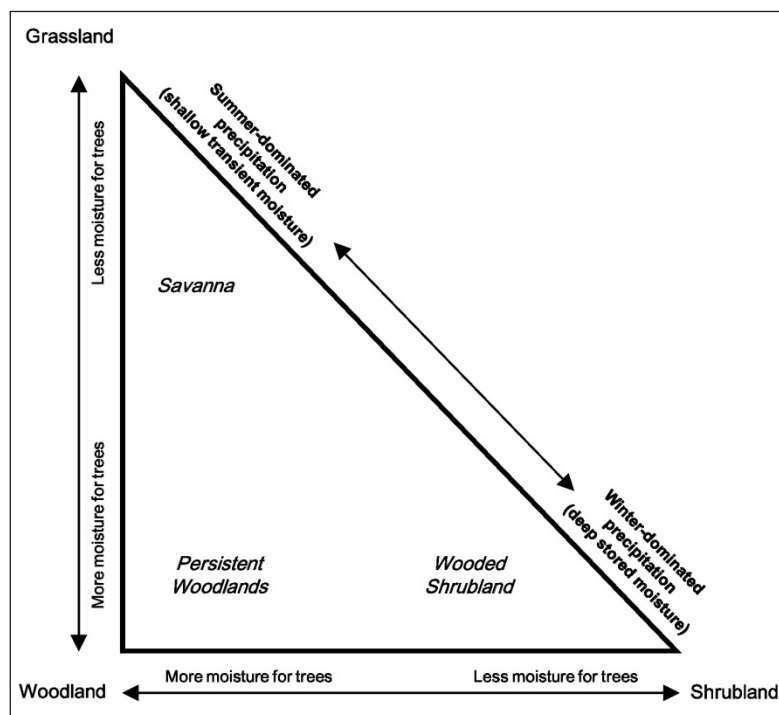


**Figure 1**—A general framework for the distribution of three broad types of piñon and juniper woodlands (Romme et al. 2009).



**Figure 2**—Piñon-juniper savanna at Elena Gallegos Picnic Ground, Sandia Mountain foothills. Photo: Jacqueline Smith.



**Figure 3**—Persistent piñon-juniper woodland, Otero Canyon, east side of Manzanita Mountains. Photo: Jacqueline Smith.

the site had likely been co-dominated by piñons and junipers forming a moderately dense woodland (Fig. 4). At this site, trip leaders discussed the efficacy of the treatment for wildfire reduction and the state of the remaining woodland. Participants suggested that the lack of quantitative monitoring of post-treatment wildlife populations and vegetation made understanding the impacts of the treatment on wildlife uncertain. Potential effects of the treatment on wildlife were discussed, in light of talks presented at the symposium.

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**Figure 4**—Treated piñon-juniper, Chamisoso Canyon, east side of Manzanita Mountains. Photo: Jacqueline Smith.

# Piñon-Juniper Habitats: Status and Management for Wildlife

12-14 October 2016

Sheraton Airport Hotel, Albuquerque, NM

## Wednesday, October 12

### Session 1: Current and Predicted State of Piñon-juniper Habitats, 8:30-12:25 (25 min./talk)

8:00-8:30 AM: Coffee

8:30-8:40 AM: Welcome: *Why a symposium on piñon-juniper?* – Intro and session chair – Marikay Ramsey, BLM NM

8:40-9:05 AM: *P-J is not just one vegetation type: key variation in structure and disturbance dynamics* – Bill Romme

9:10-9:35 AM: *Piñon-juniper woodlands as dynamic ecosystems: Past, present, and future* – Craig D. Allen

9:40-10:05 AM: *Vulnerability of piñon-juniper habitats to climate change in the Southwest* – Jack Triepke

10:05-10:25 AM: Break

10:30-10:55 AM: *Using remote sensing data to map juniper response to drought induced mortality* – Raul Campos-Marquetti and Daniel Ginter

11:00-11:25 AM: *Fire regime shift linked to increased forest density in a piñon-juniper savanna landscape* – Ellis Margolis

11:30-11:55 AM: *Stand structural dynamics and density-dependent tree mortality in drought-affected Great Basin *Pinus monophylla** – Juniperus osteosperma woodlands – Samuel W. Flake and Peter J. Weisberg

12:00-12:25 PM: *Environmental drivers of tree mast production in piñon-juniper-oak woodlands of central New Mexico* – Robert R. Parmenter, Roman I. Zlotin, Orrin B. Myers, and Douglas I. Moore.

12:25-2:00 PM: Lunch

2:00-2:25 PM: The odd couple: mutual facilitation by piñon pines and ectomycorrhizal fungi - Catherine Gehring, Andrew Krohn, Chris Sthultz, and Tom Whitham

### Session 2: Habitat Needs of Piñon-Juniper Wildlife, 2:00-5:45 (25 min./talk)

2:30-2:35 PM: Intro and session chair – Chuck Hayes, NM Department of Game and Fish

2:35-3:25 PM: *Elk, deer, and P-J: Needs, what works, and what doesn't* – Lou Bender (50 minutes)

3:30-3:55 PM: *The importance of P-J to black bears in New Mexico* – Rick Winslow

3:55-4:15 PM: Break

4:20-4:45 PM: *Pinyon Jay habitat use in New Mexico piñon-juniper woodlands* – Kristine Johnson, Jacqueline Smith, Giancarlo Sadoti, and Teri Neville

4:50-5:15 PM: *Habitat use at multiple scales by nesting Gray Vireos in New Mexico* – Lynn Wickersham, Kristine Johnson, Giancarlo Sadoti, Teri Neville, and John Wickersham

5:20-5:45 PM: *Ecology of Montezuma Quail in the Capitan Mountains of New Mexico* - Ryan S. Luna, Elizabeth A. Oaster, Karlee D. Cork, Ryan O'Shaughnessy, Randy L. Howard, Scott P. Lerich, and Louis A. Harveson

5:45-7:15 PM: Social with cash bar

## Thursday, October 13

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### Session 3: Current management activities and relevance for wildlife: goals, practices, and outcomes, 8:30-12:25 (25 min./talk)

**8:00-8:30 AM:** Coffee

**8:30-8:35 AM:** Intro and session chair – Kristine Johnson, Natural Heritage NM, UNM Biology

**8:35-9:00 AM:** *Effects of piñon-juniper woodland thinning on avian communities in the Arkansas River Valley, Colorado* – Pat Magee

**9:05-9:30 AM:** *Identifying and mitigating social-ecological tradeoffs: Vegetation, birds, fuels, and modeled fire behavior in piñon-juniper fuel treatments* – Jonathan Coop

**9:35-10:00 AM:** *Restore NM: Creating partnerships for landscape restoration* – Jeremy Kruger

**10:00-10:20 AM:** Break

**10:30-10:55 AM:** *Using the Assessment, Inventory, and Monitoring strategy to measure treatment effectiveness in the Taos Field Office* – Alexander Laurence-Traynor, Jessa Davis, Jason Karl, and Zoe Davidson

**11:00-11:25 AM:** *The effects of tree thinning on wildlife in piñon-juniper and ponderosa pine woodlands in the Manzano Mountains, New Mexico* – David Lightfoot, Cody Stropiki Victoria Amato, Conor Flynn, and Anne Russell

**11:30-11:55 AM:** *Managing piñon-juniper for multiple values and goals* – Rebecca McLain

**12:00-12:25 PM:** *Tailoring management to the inherent variability of P-J vegetation* – Bill Romme

**12:25-2:30 PM:** Lunch

### Session 4: Breakout Discussion Groups

**2:30-4:30 PM:** Sign up for a discussion group at the registration table.

**4:30 PM:** Wrap-up

### Posters:

*Contraction and expansion zones in western North America piñon-juniper woodlands under projected 21st century climate change* – Jacob R. Gibson, Thomas C. Edwards, Jr., Gretchen G. Moisen, Tracey Frescino, Niklaus Zimmermann, and Achilleas Psomos

*Effects of prescribed fire on Gray Vireo nesting success on the Sevilleta NWR* - G. Manakitivipart, D.C. Barton, K. Granillo, and M.E. Persche

*Surveying and nest monitoring of the Gray Vireo (Vireo vicinior) in the Sevilleta National Wildlife Refuge* - Ben Vizzachero and Sze Wing Yu

## Friday, October 14

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**8:30–3:30 PM:** Field trip to Sandia and Manzano Mountains



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