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 Peichar (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.

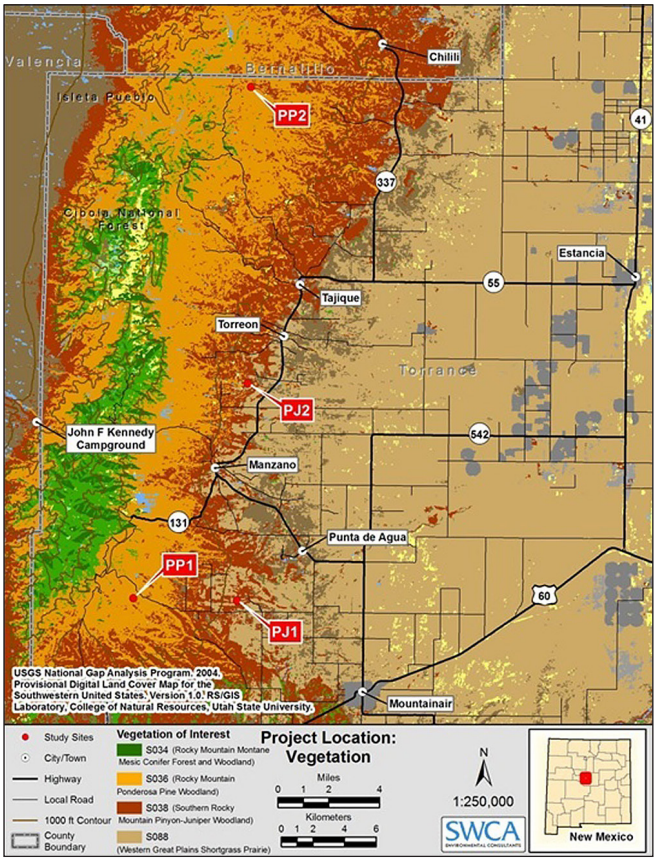

Figure 1—Southwest Regional Gap Analysis Project land cover map of the study region and location of study sites on the eastern slopes of the Manzano Mountains, Torrance County, New Mexico.

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; 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 convectional 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-m² vegetation measurement quadrats at 10-m intervals (fig. 2).

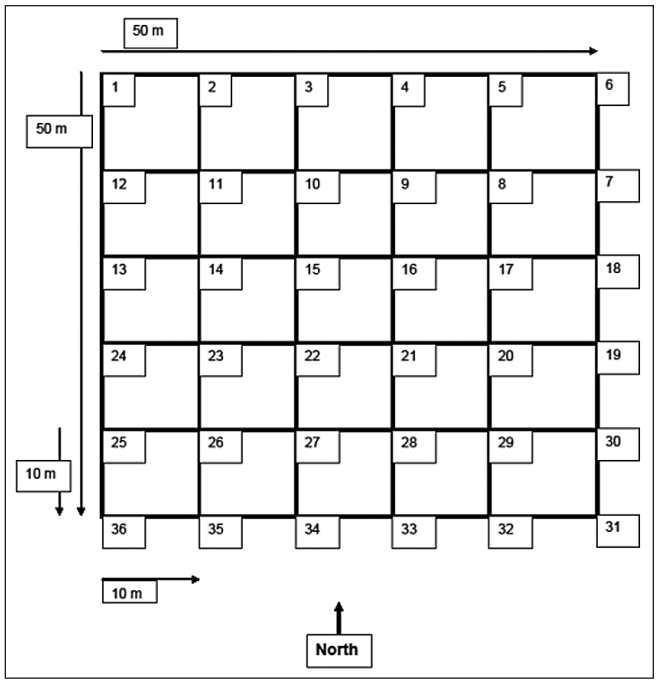

Figure 2—Diagram of a wildlife and vegetation measurement plot. Numbers 1 through 36 are 1-m^2 vegetation quadrat numbers. Components of the diagram are not to scale for ease of viewing.

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 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 masting plant species where individual trees collectively produce large seed crops infrequently (every several years) on an irregular basis (Janetski 1999). Given the uncertainly 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^2 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^2/acre (9.2–13.7 m^2 /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^2/acre (5.6 m^2/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 (≤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.

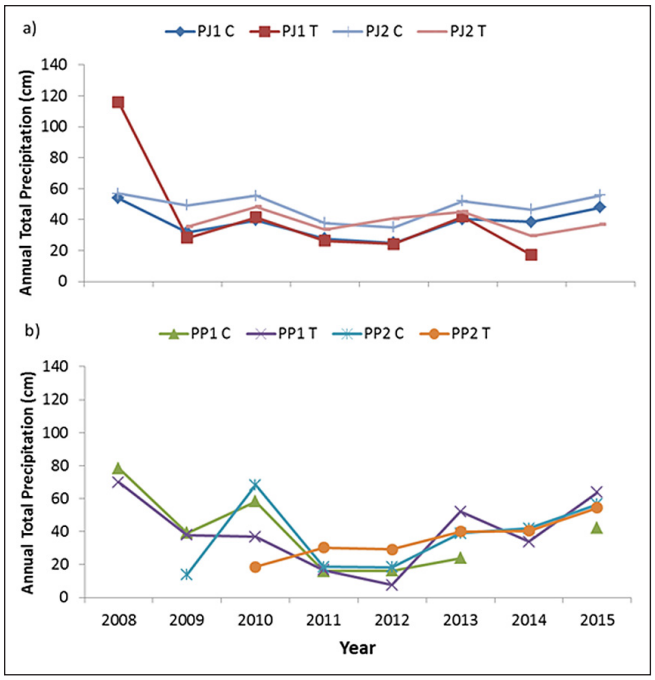

Figure 3—Total annual precipitation values for (a) the two pinyon-juniper (PJ1 and PJ2) and (b) the two ponderosa pine (PP1 and PP2) sites, 2008–2015. Data gaps were due to rain gauge outages, primarily from American black bear damage.

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 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^2/acre (46 m^2 /ha) at the ponderosa pine sites and around 140 ft^2/acre (32 m^2/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^2/acre (19.5 m^2/ha) on the treated ponderosa pine watersheds, while at the pinyon-juniper sites basal area was reduced to around 40 ft^2/acre.

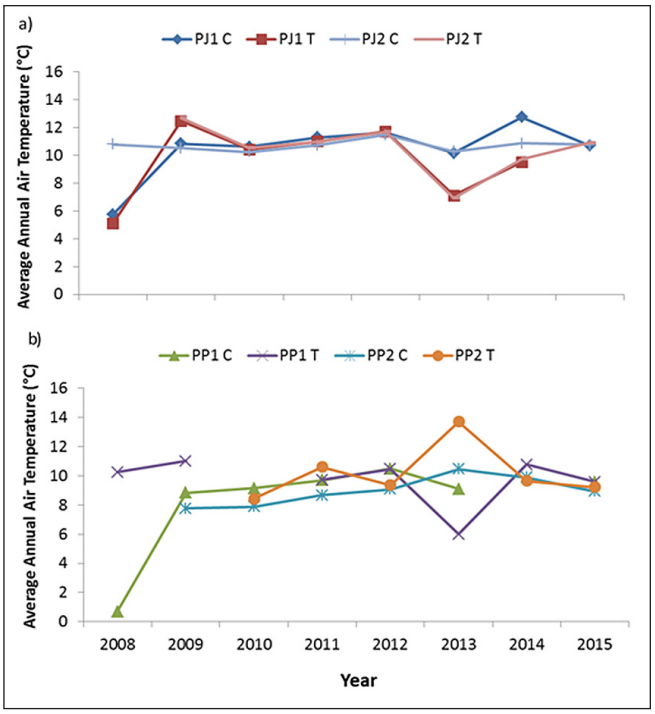

Figure 4—Average annual temperatures for (a) the two pinyon-juniper (PJ1 and PJ2) and (b) the two ponderosa pine (PP1 and PP2) sites, 2008–2015. Data gaps were due to temperature gauge outages, primarily from American black bear damage.

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, $p = 0.001$;

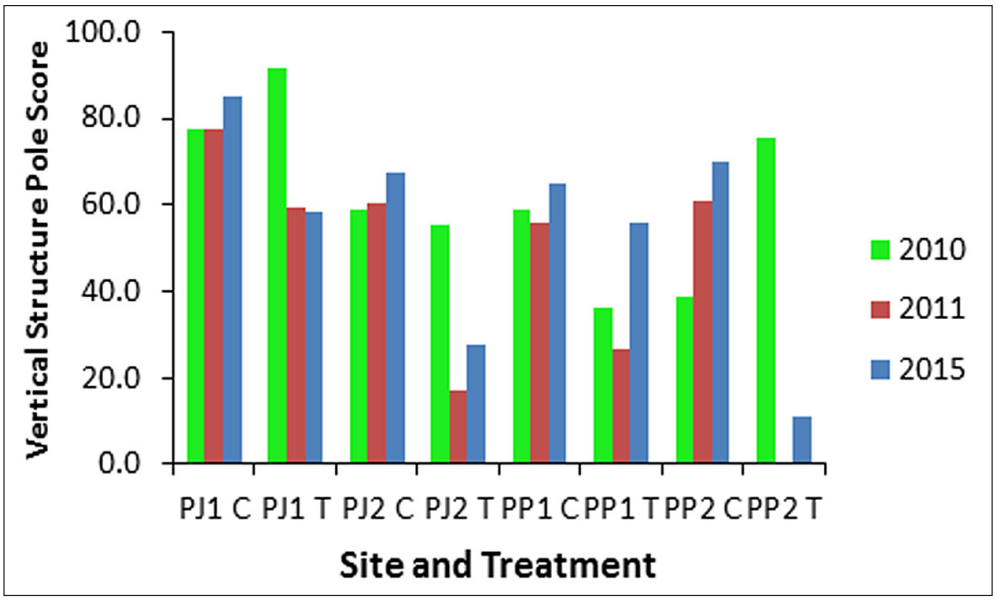

Figure 5—Average (averaged over 11 points per plot) vertical vegetation canopy cover structure 0 to 3 m above the ground surface for the two pinyon-juniper (PJ1 and PJ2) and the two ponderosa pine (PP1 and PP2) sites. C = control; T = treatment.

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

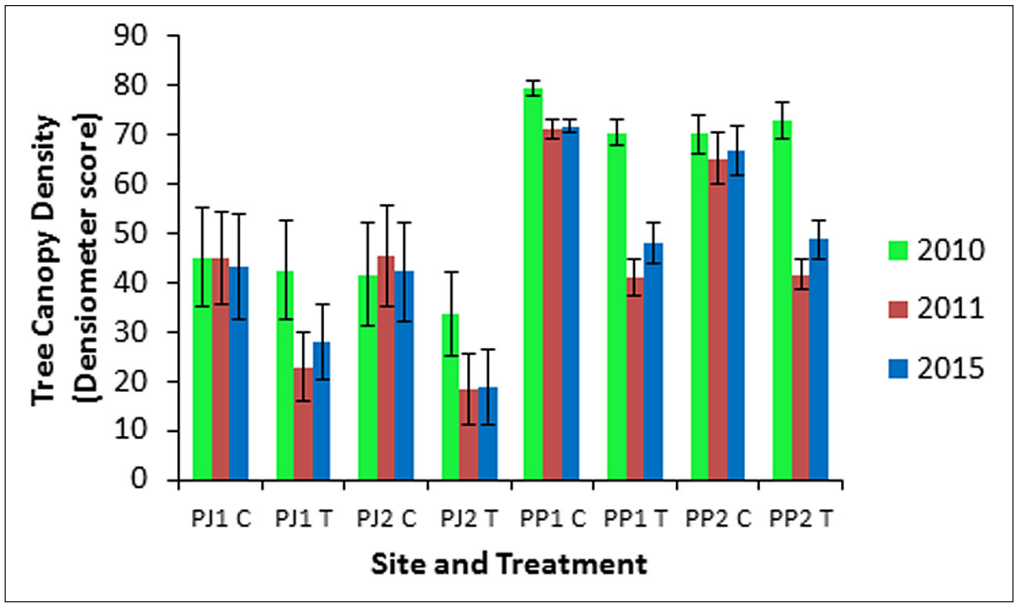

Figure 6—Average densiometer score (averaged over 11 points per plot) for horizontal tree crown canopy. Error bars represent ± one standard error of the mean.

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^2 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.

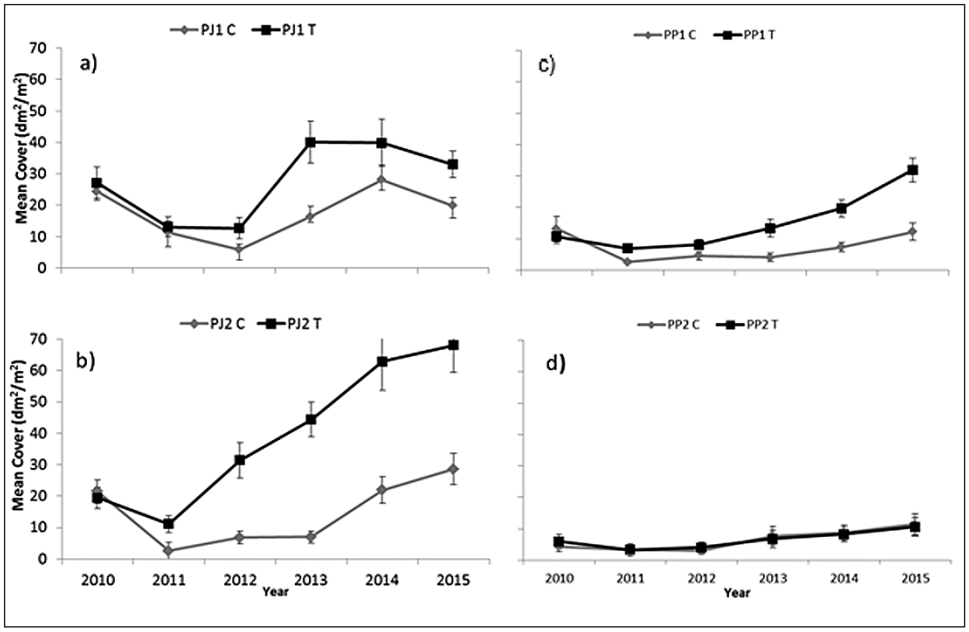

Figure 7—Mean herbaceous plant canopy cover values across all vegetation quadrats on (a,b) the two pinyon-juniper (PJ1 and PJ2) and (c,d) the two ponderosa pine (PP1 and PP2) sites, 2010–2015. C = control; T= treatment. Thinning treatments occurred on the treatment plots between 2010 and 2011. Error bars represent ± 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
		Wood chips	**34.11**	**0.0**	**<0.0001**
	PJ2	All herbs	6.4	11.1	0.2121
		Wood chips	41.6	0.0	<0.0001
	PP1	**All herbs**	**2.1**	**7.0**	**0.0016**
		Wood chips	**33.9**	**0.0**	**<0.0001**
	PP2	All herbs	3.3	3.3	0.9980
		Wood chips	**40.2**	**0.0**	**<0.0001**
2015	PJ1	**All herbs**	**19.8**	**33.0**	**0.0103**
		Wood chips	**0.0**	**22.8**	**<0.0001**
	PJ2	**All herbs**	**28.6**	**68.0**	**0.0002**
		Wood chips	**0.0**	**20.1**	**<0.0001**
	PP1	**All herbs**	**12.2**	**32.0**	**<0.0001**
		Wood chips	**0.0**	**18.0**	**<0.0001**
	PP2	All herbs	3.4	2.9	0.8953
		Wood chips	**0.0**	**15.6**	**<0.0001**

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

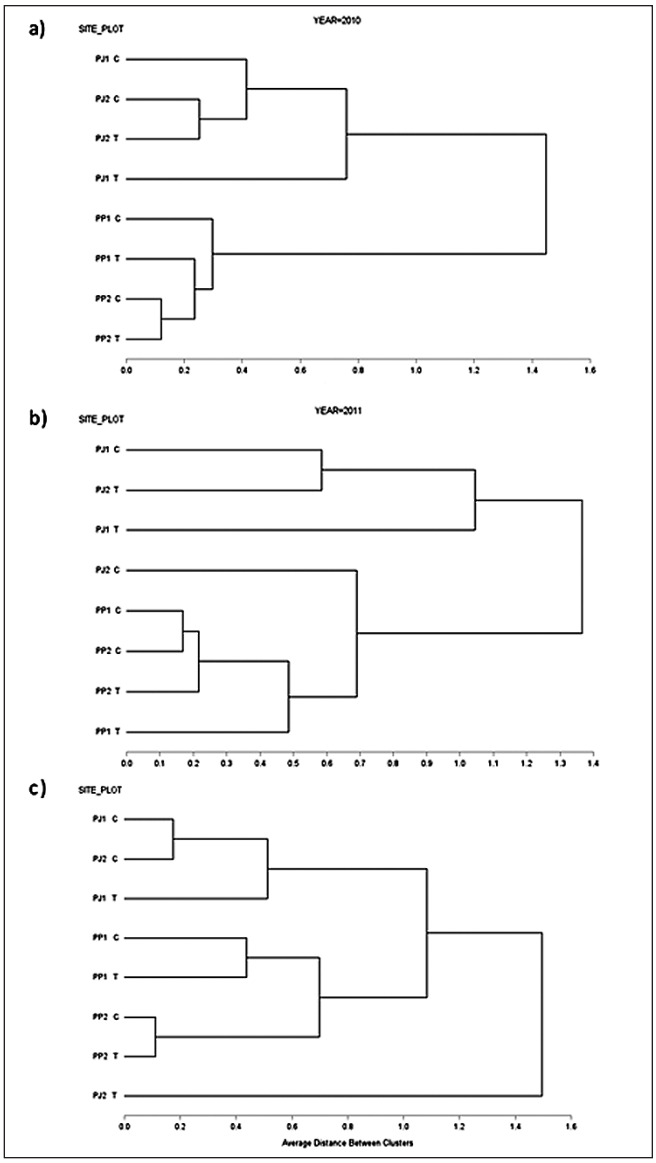

Figure 8—Cluster analysis results showing the similarity of sites and paired plots based on similarity of the herbaceous plant community species compositions in (a) 2010, (b) 2011, and (c) 2015.

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

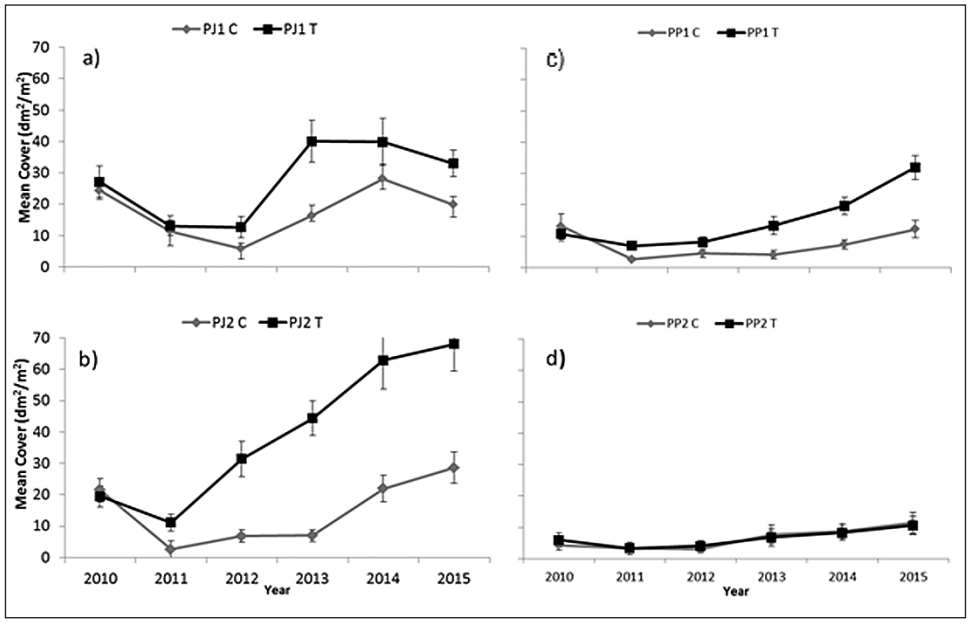

Figure 9—Mean wood chip cover values across all vegetation quadrats on (a,b) the two pinyon-juniper (PJ1 and PJ2) and (c,d) the two ponderosa pine (PP1 and PP2) sites, 2010–2015. Thinning treatments occurred on the treatment (T) plots between 2010 and 2011. Error bars represent ± one standard error of 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[1] code	PJ1		PJ2		PJ1		PJ2		Row average
			Control	Treated	Control	Treated	Control	Treated	Control	Treated	
American crow	*Corvus brachyrhynchos*	AMCR	0.5	0.7	0.9	0.7	0.9	0.7	0.5	0.6	0.7
American goldfinch	*Spinus tristis*	AMGO	0.0	0.3	0.0	0.0	0.0	0.0	0.0	2.0	0.3
American kestrel	*Falco sparverius*	AMKE	0.0	0.0	0.3	0.0	0.0	0.3	0.0	0.3	0.1
American robin	*Turdus migratorius*	AMRO	0.3	0.3	0.5	0.5	0.7	0.5	0.8	0.4	0.5
Ash-throated flycatcher	*Myiarchus cinerascens*	ATFL	0.3	0.3	0.5	0.3	0.4	0.3	0.5	0.3	0.4
Audubon's warbler	*Setophaga coronata auduboni*	AUWA	0.3	0.3	0.3	0.8	0.3	0.3	0.3	0.4	0.4
Barn swallow	*Hirundo rustica*	BARS	0.3	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.1
Black-capped chickadee	*Poecile atricapillus*	BCCH	0.3	0.3	0.3	0.3	0.0	1.0	0.0	0.0	0.3
Black-chinned hummingbird	*Archilochus alexandri*	BCHU	0.0	0.0	0.0	0.0	0.0	0.3	0.5	0.5	0.2
Bell's vireo	*Vireo bellii*	BEVI	0.0	0.0	0.0	0.0	0.3	0.0	0.3	0.0	0.1
Bewick's wren	*Thryomanes bewickii*	BEWR	0.3	0.4	0.7	0.6	0.3	0.7	0.3	0.3	0.4
Blue-gray gnatcatcher	*Polioptila caerulea*	BGGN	0.0	0.0	0.0	0.0	0.0	0.8	0.0	0.0	0.1
Broad-tailed hummingbird	*Selasphorus platycercus*	BTAH	0.3	0.0	0.0	0.3	0.3	0.3	0.0	0.4	0.2
Black-throated gray warbler	*Setophaga nigrescens*	BTYW	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0
Bushtit	*Psaltriparus minimus*	BUSH	0.0	0.3	0.0	0.0	0.6	0.5	0.0	0.0	0.2
Canyon towhee	*Melozone fusca*	CANT	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.0	0.1
Chipping sparrow	*Spizella passerina*	CHSP	0.3	0.4	0.7	0.8	1.3	0.5	0.7	0.6	0.7
Cordilleran flycatcher	*Empidonax occidentalis*	COFL	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.3	0.1
Cooper's hawk	*Accipiter cooperii*	COHA	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.7	0.1
Common raven	*Corvus corax*	CORA	0.3	0.4	0.8	0.6	0.7	0.8	0.6	1.0	0.6
Dark-eyed junco	*Junco hyemalis*	DEJU	0.0	1.0	1.0	0.3	0.0	0.0	0.0	0.0	0.3
Great blue heron	*Ardea herodias*	GBHE	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0
Gray flycatcher	*Empidonax wrightii*	GRFL	0.0	0.0	0.0	0.3	0.0	0.3	0.3	0.0	0.1
Gray vireo	*Vireo vicinior*	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 (PJ) sites			Before thinning, 2008–2010				After thinning, 2011–2015				
Common name	Scientific name	AOU[1] code	PJ1		PJ2		PJ1		PJ2		Row average
			Control	Treated	Control	Treated	Control	Treated	Control	Treated	
Hairy woodpecker	*Picoides villosus*	HAWO	0.0	0.3	0.0	0.3	0.0	0.0	0.0	0.3	0.1
Hermit thrush	*Catharus guttatus*	HETH	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0
House finch	*Haemorhous mexicanus*	HOFI	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.1
Juniper titmouse	*Baeolophus ridgwayi*	JUTI	0.3	0.6	0.7	0.5	0.4	0.6	0.6	0.6	0.5
Lark sparrow	*Chondestes grammacus*	LASP	0.0	0.0	0.0	0.0	0.0	0.7	0.0	0.9	0.2
Ladder-backed woodpecker	*Picoides scalaris*	LBWO	0.0	0.0	0.0	0.3	0.3	0.3	0.0	0.0	0.1
Lesser goldfinch	*Spinus psaltria*	LEGO	0.0	0.0	0.0	0.0	0.3	0.0	0.3	0.5	0.1
Mountain bluebird	*Sialia currucoides*	MOBL	0.0	0.0	1.7	5.0	0.0	0.4	1.0	0.5	1.1
Mountain chickadee	*Poecile gambeli*	MOCH	0.3	0.3	0.5	0.3	0.5	0.4	0.0	0.4	0.4
Mourning dove	*Zenaida macroura*	MODO	0.7	0.9	0.5	0.5	0.3	0.8	0.4	0.9	0.6
Northern flicker	*Colaptes auratus*	NOFL	0.4	0.4	0.5	0.6	0.4	0.4	0.5	0.5	0.5
Northern harrier	*Circus cyaneus*	NOHA	0.3	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.1
Northern mockingbird	*Mimus polyglottos*	NOMO	0.0	0.0	0.3	0.0	0.0	0.0	0.4	0.5	0.1
Northern rough-winged swallow	*Stelgidopteryx serripennis*	NRWS	0.0	0.0	0.0	0.0	1.7	0.5	0.5	0.5	0.4
Orange-crowned warbler	*Oreothlypis celata*	OCWA	0.7	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.1
Pinyon jay	*Gymnorhinus cyanocephalus*	PIJA	0.3	0.3	0.5	0.0	1.7	2.3	2.8	2.3	1.3
Plumbeous vireo	*Vireo plumbeus*	PLVI	0.0	0.0	0.0	0.3	0.3	0.0	0.3	0.4	0.2
Red-breasted nuthatch	*Sitta canadensis*	RBNU	0.0	0.0	0.0	0.3	0.3	0.0	0.0	0.3	0.1
Ruby-crowned kinglet	*Regulus calendula*	RCKI	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0
Red crossbill	*Loxia curvirostra*	RECR	0.0	0.0	0.7	0.0	0.0	0.0	0.0	0.0	0.1
Red-tailed hawk	*Buteo jamaicensis*	RTHA	0.0	0.0	0.0	0.0	0.3	0.3	0.0	0.9	0.2
Rufous hummingbird	*Selasphorus rufus*	RUHU	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Say's phoebe	*Sayornis saya*	SAPH	0.0	0.0	0.0	0.0	0.0	0.3	0.3	0.4	0.1
Savannah sparrow	*Passerculus sandwichensis*	SAVS	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.1
Spotted towhee	*Pipilo maculatus*	SPTO	0.3	0.3	0.6	0.6	0.3	0.3	0.3	0.4	0.4
Sharp-shinned hawk	*Accipiter striatus*	SSHA	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0

(CONT.)

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

[1]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[1] code	PP1		PP2		PP1		PP2		Row average
			Control	Treated	Control	Treated	Control	Treated	Control	Treated	
American crow	*Corvus brachyrhynchos*	AMCR	0.3	0.3	0.0	0.0	0.7	0.3	0.0	0.0	0.2
American robin	*Turdus migratorius*	AMRO	0.4	0.4	0.5	0.3	0.3	0.4	0.5	0.6	0.4
Ash-throated flycatcher	*Myiarchus cinerascens*	ATFL	0.0	0.0	0.4	0.3	0.0	0.0	0.5	0.7	0.2
Audubon's warbler	*Setophaga coronata auduboni*	AUWA	0.0	0.4	0.0	0.0	0.5	0.9	0.4	0.6	0.4
Black-capped chickadee	*Poecile atricapillus*	BCCH	0.3	0.9	0.6	0.6	0.3	0.0	0.0	0.0	0.3
Black-chinned hummingbird	*Archilochus alexandri*	BCHU	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.3	0.1
Blue-gray gnatcatcher	*Polioptila caerulea*	BGGN	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0
Black-headed grosbeak	*Pheucticus melanocephalus*	BHGR	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0
Brown creeper	*Certhia americana*	BRCR	0.0	0.7	0.0	0.0	0.3	0.3	0.3	0.0	0.2
Broad-tailed hummingbird	*Selasphorus platycercus*	BTAH	0.0	0.0	0.3	0.4	0.4	0.3	0.0	0.3	0.2
Bushtit	*Psaltriparus minimus*	BUSH	0.0	0.0	0.3	1.3	0.0	2.3	2.0	1.7	1.0
Chipping sparrow	*Spizella passerina*	CHSP	0.0	0.0	0.0	1.7	0.9	0.6	0.3	0.5	0.5
Cordilleran flycatcher	*Empidonax occidentalis*	COFL	0.0	0.0	0.0	0.0	0.3	0.3	0.0	0.0	0.1
Cooper's hawk	*Accipiter cooperii*	COHA	0.0	0.0	0.3	0.0	0.0	0.0	0.3	0.0	0.1
Common raven	*Corvus corax*	CORA	0.5	0.5	0.5	0.5	0.4	0.3	0.6	0.5	0.5
Dark-eyed junco	*Junco hyemalis*	DEJU	0.3	0.3	1.1	0.4	0.7	0.9	1.0	0.4	0.6
Downy woodpecker	*Picoides pubescens*	DOWO	0.0	0.0	0.0	0.0	0.3	0.0	0.3	0.5	0.1
Evening grosbeak	*Coccothraustes vespertinus*	EVGR	0.0	0.7	0.0	0.0	0.0	0.0	0.0	0.0	0.1
Great horned owl	*Bubo virginianus*	GHOW	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0
Gray flycatcher	*Empidonax wrightii*	GRFL	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.1
Grace's warbler	*Setophaga graciae*	GRWA	0.5	0.5	0.5	0.6	0.5	0.5	0.5	0.5	0.5
Hairy woodpecker	*Picoides villosus*	HAWO	0.4	0.4	0.5	0.3	0.5	0.4	0.3	0.0	0.3
Hermit thrush	*Catharus guttatus*	HETH	0.0	0.0	0.3	0.4	0.3	0.7	0.0	0.0	0.2
Juniper titmouse	*Baeolophus ridgwayi*	JUTI	0.0	0.8	0.0	0.0	0.0	1.4	1.0	0.5	0.5
Ladder-backed woodpecker	*Picoides scalaris*	LBWO	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0
Lesser goldfinch	*Spinus psaltria*	LEGO	0.0	0.0	0.0	0.0	0.0	0.7	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[1] code	PP1		PP2		PP1		PP2		Row average
			Control	Treated	Control	Treated	Control	Treated	Control	Treated	
Lucy's warbler	*Oreothlypis luciae*	LUWA	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.1
Mountain bluebird	*Sialia currucoides*	MOBL	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.7	0.2
Mountain chickadee	*Poecile gambeli*	MOCH	0.4	0.3	0.6	0.7	1.1	0.7	0.8	0.8	0.7
Mourning dove	*Zenaida macroura*	MODO	0.5	0.3	0.3	0.3	0.0	0.0	0.0	0.7	0.3
Northern flicker	*Colaptes auratus*	NOFL	0.3	0.5	0.4	0.5	0.4	0.4	0.3	0.4	0.4
Northern mockingbird	*Mimus polyglottos*	NOMO	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0
Orange-crowned warbler	*Oreothlypis celata*	OCWA	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0
Pinyon jay	*Gymnorhinus cyanocephalus*	PIJA	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.3	0.1
Pine siskin	*Spinus pinus*	PISI	0.5	0.5	0.0	0.0	0.0	0.7	0.9	0.7	0.4
Plumbeous vireo	*Vireo plumbeus*	PLVI	0.4	0.5	0.4	0.5	0.4	0.5	0.5	0.4	0.4
Pygmy nuthatch	*Sitta pygmaea*	PYNU	0.0	0.0	0.0	0.0	0.7	0.0	0.0	0.0	0.1
Red-breasted nuthatch	*Sitta canadensis*	RBNU	0.3	0.7	0.0	0.3	0.3	0.3	0.3	0.5	0.3
Ruby-crowned kinglet	*Regulus calendula*	RCKI	0.3	0.4	0.0	0.7	0.7	0.7	0.0	0.3	0.4
Red crossbill	*Loxia curvirostra*	RECR	0.8	1.2	1.4	1.2	0.7	0.0	0.0	5.0	1.3
Red-tailed hawk	*Buteo jamaicensis*	RTHA	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0
Rufous hummingbird	*Selasphorus rufus*	RUHU	0.0	0.0	0.3	0.3	0.0	0.0	0.0	0.0	0.1
Spotted towhee	*Pipilo maculatus*	SPTO	0.0	0.3	0.3	0.3	0.0	0.3	0.0	0.0	0.2
Sharp-shinned hawk	*Accipiter striatus*	SSHA	0.0	0.0	0.0	0.0	0.3	0.3	0.0	0.3	0.1
Steller's jay	*Cyanocitta stelleri*	STJA	0.4	0.3	0.4	0.4	0.5	0.7	0.5	0.5	0.4
Swainson's hawk	*Buteo swainsoni*	SWHA	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0
Townsend's solitaire	*Myadestes townsendi*	TOSO	0.5	0.0	0.0	0.4	0.0	0.3	0.0	0.3	0.2
Tree swallow	*Tachycineta bicolor*	TRES	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.1
Turkey vulture	*Cathartes aura*	TUVU	0.3	0.7	0.3	0.0	1.7	0.3	0.3	0.0	0.5
Violet-green swallow	*Tachycineta thalassina*	VGSW	0.0	0.0	0.0	0.0	0.0	0.7	0.0	0.6	0.2
White-breasted nuthatch	*Sitta carolinensis*	WBNU	0.4	0.6	0.5	0.4	0.4	0.4	0.5	0.4	0.4
White-crowned sparrow	*Zonotrichia leucophrys*	WCSP	0.0	0.0	0.3	0.3	0.0	0.0	0.0	0.0	0.1
Western bluebird	*Sialia mexicana*	WEBL	0.0	0.0	0.0	0.3	0.8	0.8	1.4	0.7	0.5
Western kingbird	*Tyrannus verticalis*	WEKI	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0
Western tanager	*Piranga ludoviciana*	WETA	0.0	0.0	0.0	0.0	0.0	0.8	0.3	0.0	0.1
Western wood-pewee	*Contopus sordidulus*	WEWP	0.0	0.5	0.0	0.3	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[1] code**	**PP1**		**PP2**		**PP1**		**PP2**		**Row average**
			Control	**Treated**	**Control**	**Treated**	**Control**	**Treated**	**Control**	**Treated**	
Williamson's sapsucker	*Sphyrapicus thyroideus*	WISA	0.0	0.0	0.5	0.0	0.3	0.0	0.0	0.3	0.1
Wild turkey	*Meleagris gallopavo*	WITU	0.0	0.0	0.0	0.0	0.0	0.3	0.3	0.0	0.1
Wilson's warbler	*Cardellina pusilla*	WIWA	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0
Woodhouse's scrub-jay	*Aphelocoma woodhouseii*	WOSJ	0.5	0.8	0.0	0.7	0.3	0.4	0.4	0.5	0.4
UNKNOWN FLYCATCHER	none	UKFLYC	0.0	0.0	0.0	0.0	0.0	0.3	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
UNKNOWN PASSERINE	none	UKPASS	0.0	0.0	0.0	0.0	0.5	0.5	0.8	1.0	0.4
UNKNOWN SPARROW	none	UNKSPAR	0.4	0.9	0.3	1.2	0.0	0.0	0.0	0.0	0.3
UNKNOWN VIREO	none	UKVIREO	0.0	0.0	0.0	0.0	0.5	0.5	0.8	1.0	0.4
		Column average	0.1	0.2	0.2	0.3	0.2	0.3	0.3	0.4	0.3

[1]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 ($p = 0.04$).

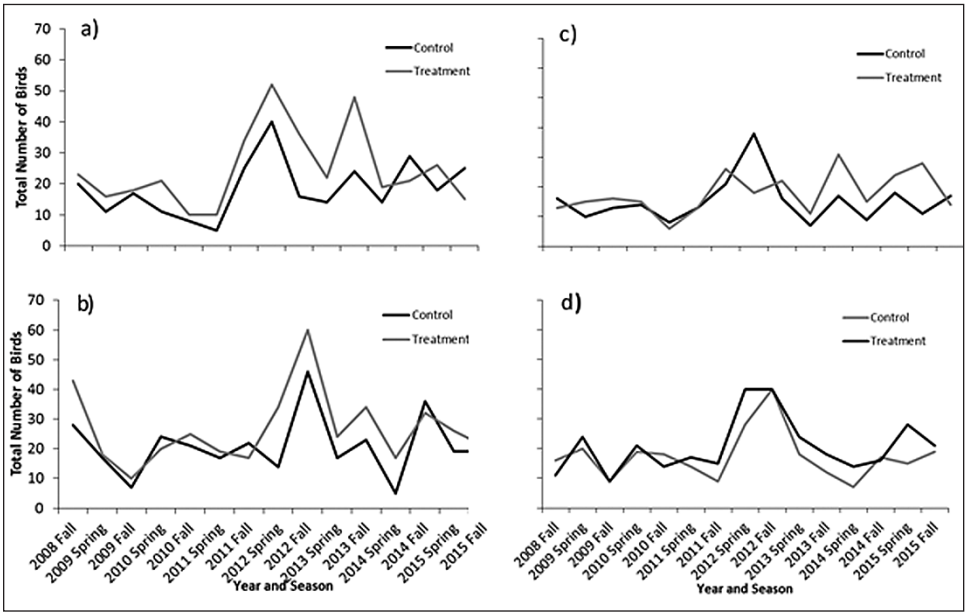

Figure 10—Average number of birds per day from 3-day point counts in the spring and fall on (a,b) the two pinyon-juniper (PJ1 and PJ2) and (c,d) the two ponderosa pine (PP1 and PP2) sites, 2010–2015.

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

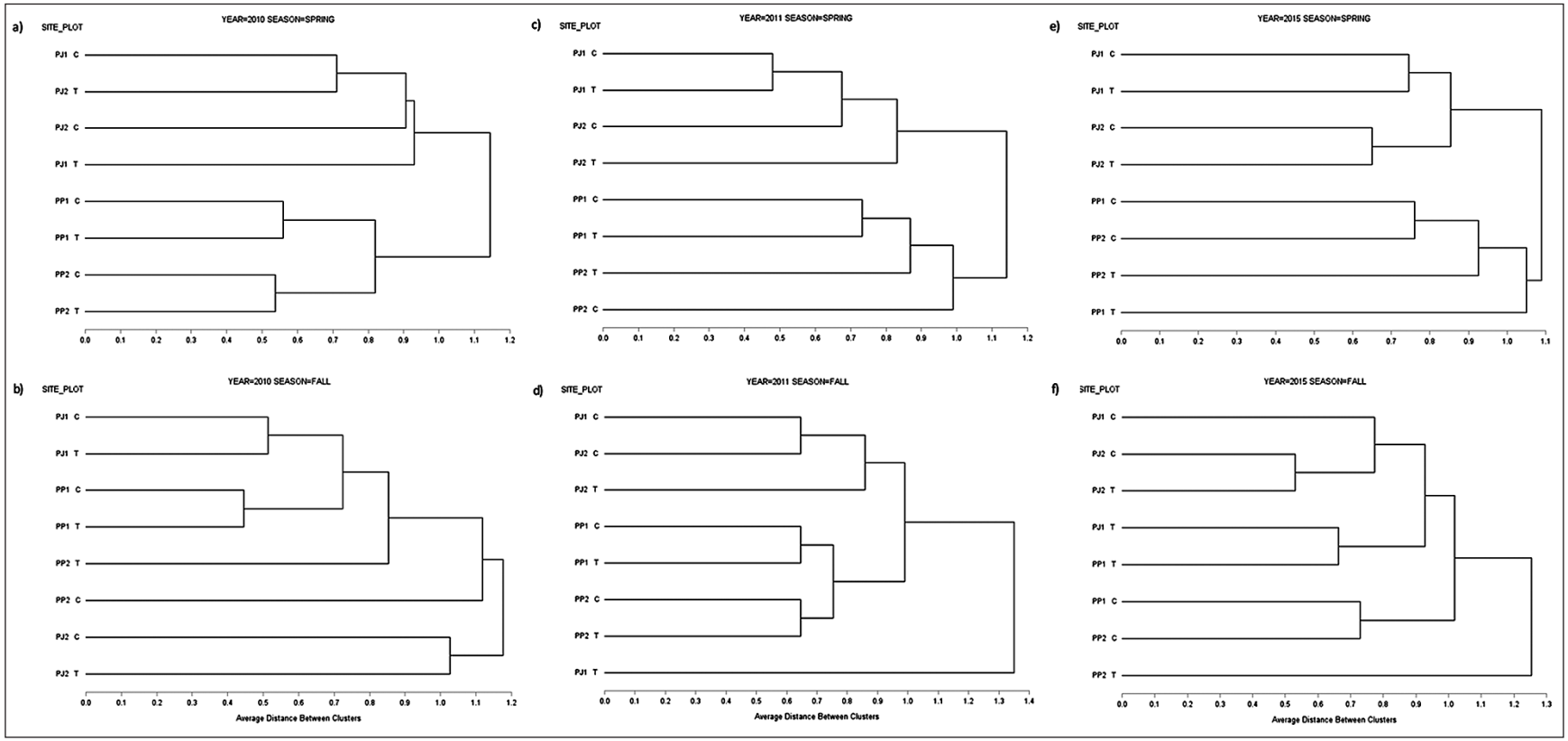

Figure 11—Cluster analysis results showing the similarity of sites and paired plots based on similarity of the bird species compositions in (a) spring 2010, (b) fall 2010, (c) spring 2011, (d) fall 2011, (e) spring 2015, and (f) fall 2015.

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				
			PJ1		PJ2		PJ1		PJ2		Row
Common name	Scientific name	Species code	Control	Treated	Control	Treated	Control	Treated	Control	Treated	average
Deer mouse	*Peromyscus maniculatus*	PEMA	2.0	1.0	5.3	3.0	2.0	1.0	1.6	4.0	2.5
Mexican vole	*Micropus mexicanus*	MIME	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0
Northern grasshopper mouse	*Onychomys leucogaster*	ONLE	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1
Pinyon mouse	*Peromyscus truei*	PETR	6.0	8.5	4.4	6.4	7.4	4.0	6.1	2.8	5.7
Plains pocket mouse	*Perognathus flavescens*	PEFL2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.1
Plains woodrat	*Neotoma micropus*	NEMI	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.0	0.3
Silky pocket mouse	*Perognathus flavus*	PEFL	0.0	0.0	1.0	1.5	1.0	0.0	1.0	1.0	0.7
White-footed mouse	*Peromyscus leucopus*	PELE	5.0	5.5	2.0	2.0	1.0	1.0	1.7	1.0	2.4
White-throated woodrat	*Neotoma albigula*	NEAL	0.0	0.0	0.0	1.0	1.5	2.0	1.0	1.0	0.8
		Column average	1.4	1.8	1.5	1.5	1.4	0.9	1.3	1.4	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				
			PP1		PP2		PP1		PP2		Row
Common name	Scientific name	Species code	Control	Treated	Control	Treated	Control	Treated	Control	Treated	average
Brush mouse	*Peromyscus boylei*	PEBO	0.0	0.0	0.0	0.0	0.0	1.0	1.0	1.0	0.4
Deer mouse	*Peromyscus maniculatus*	PEMA	5.5	1.3	3.0	2.7	1.5	1.8	2.5	2.4	2.6
Mexican vole	*Micropus mexicanus*	MIME	0.0	0.0	0.0	0.0	1.0	1.0	0.0	0.0	0.3
Pinyon mouse	*Peromyscus truei*	PETR	2.5	6.5	4.0	1.5	1.4	1.4	2.0	0.0	2.4
Western harvest mouse	*Reithrodontomys megalotis*	REME	0.0	0.0	0.0	0.0	0.0	0.0	1.0	1.0	0.3
White-footed mouse	*Peromyscus leucopus*	PELE	2.0	1.0	0.0	2.0	1.0	0.0	1.0	0.0	0.9
White-throated woodrat	*Neotoma albigula*	NEAL	0.0	0.0	0.0	0.0	0.0	0.0	1.0	1.0	0.3
		Column average	1.4	1.3	1.0	0.9	0.7	0.8	1.2	0.8	1.0

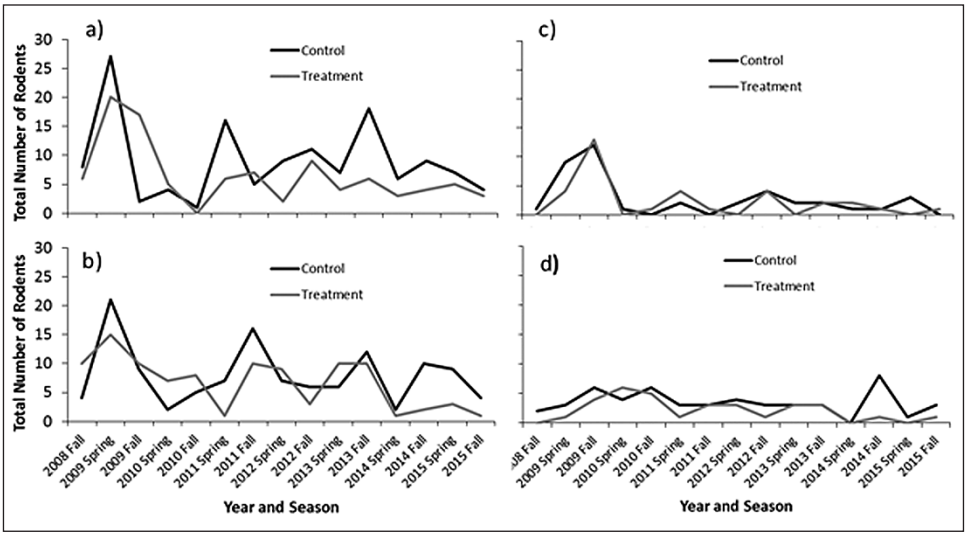

Figure 12—Total number of rodents over 3-night live trapping periods in the spring and fall on (a,b) the two pinyon-juniper (PJ1 and PJ2) and (c,d) the two ponderosa pine (PP1 and PP2) sites, 2010–2015.

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

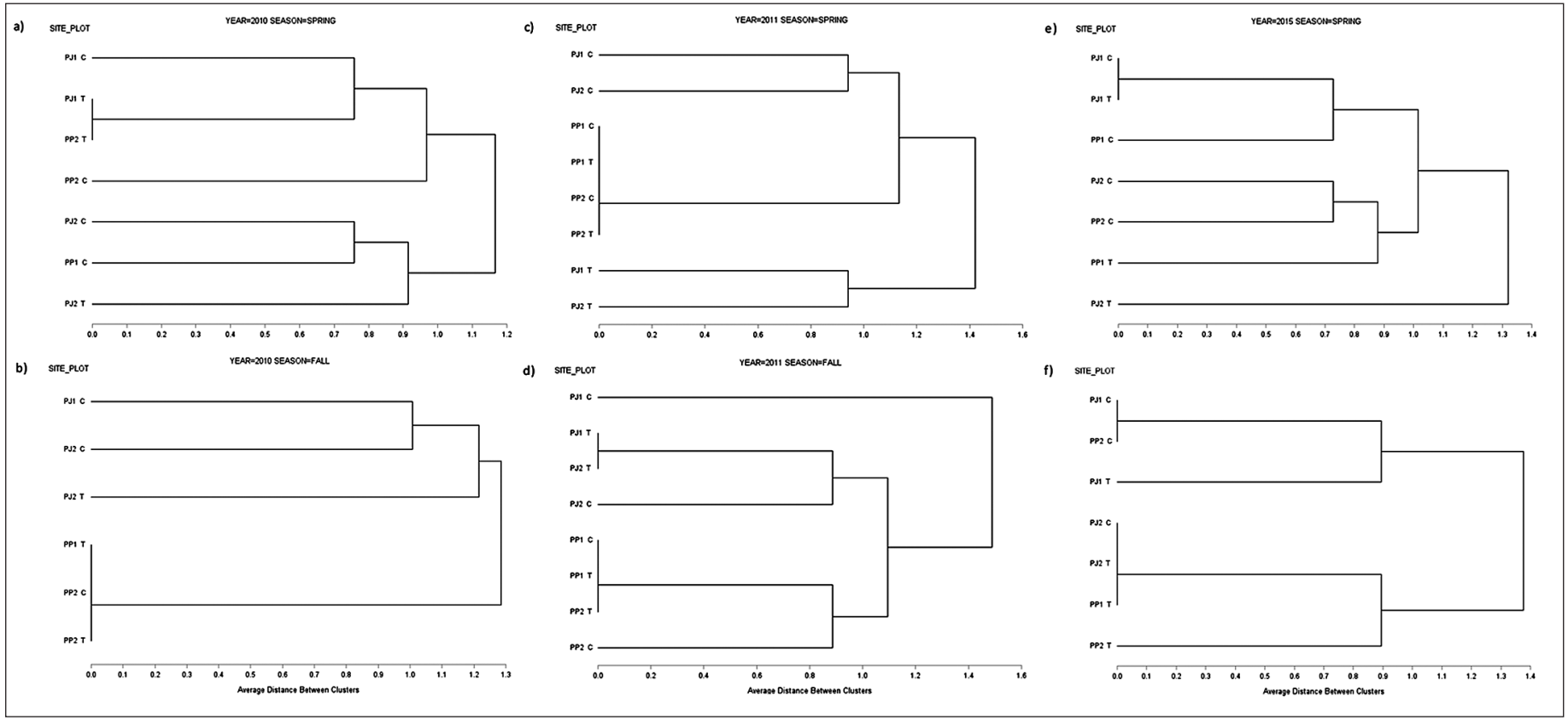

Figure 13—Cluster analysis results showing the similarity of sites and paired plots based on similarity of the rodent community species compositions in (a) spring 2010, (b) fall 2010, (c) spring 2011, (d) fall 2011, (e) spring 2015, and (f) fall 2015.

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	*Ursus americanus*	0	0	1	0	4	2	4	0	11
Black-tailed jackrabbit	*Lepus californicus*	9	7	17	0	3	0	2	0	38
Bobcat	*Lynx rufus*	0	0	0	0	3	0	0	0	3
Common raccoon	*Procyon lotor*	0	0	0	0	1	1	0	0	2
Coyote	*Canis latrans*	31	6	19	1	4	7	4	0	72
Desert cottontail	*Sylvilagus auduboni*	0	1	3	0	22	1	2	0	29
Gray fox	*Urocyon cinereoargenteus*	1	0	0	0	1	0	7	2	11
Mule deer	*Odocoileus hemionus*	1	3	5	5	13	7	89	40	163
North American elk	*Cervus canadensis*	3	0	0	0	0	0	7	2	12
Wild turkey	*Meleagris gallopavo*	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.

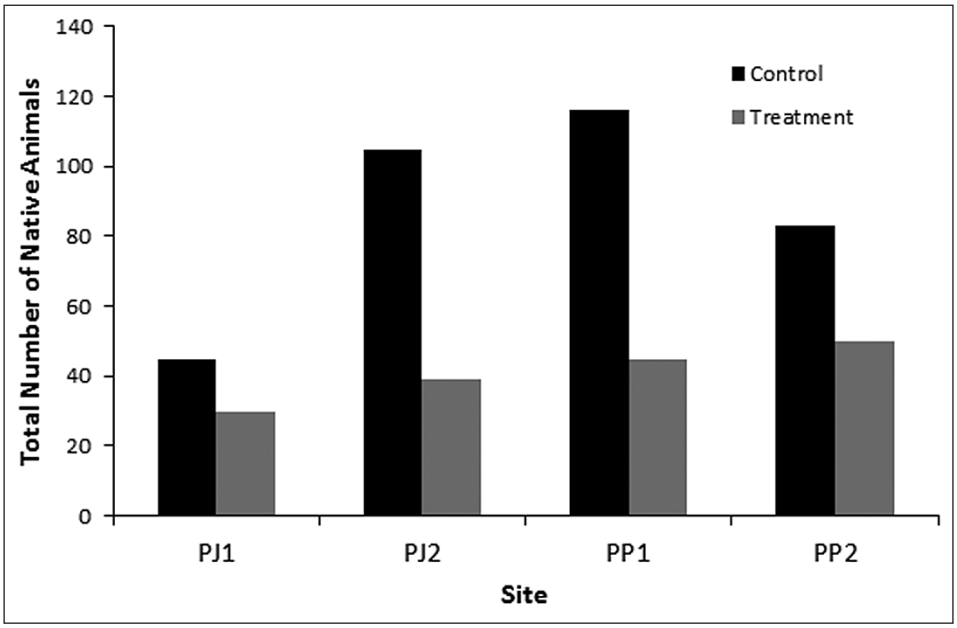

Figure 14—Total number of native medium to large mammals and wild turkeys recorded from camera traps on the control and treatment plots at the two pinyon-juniper (PJ1 and PJ2) and the two ponderosa pine (PP1 and PP2) sites following tree thinning treatments, 2012–2015.

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

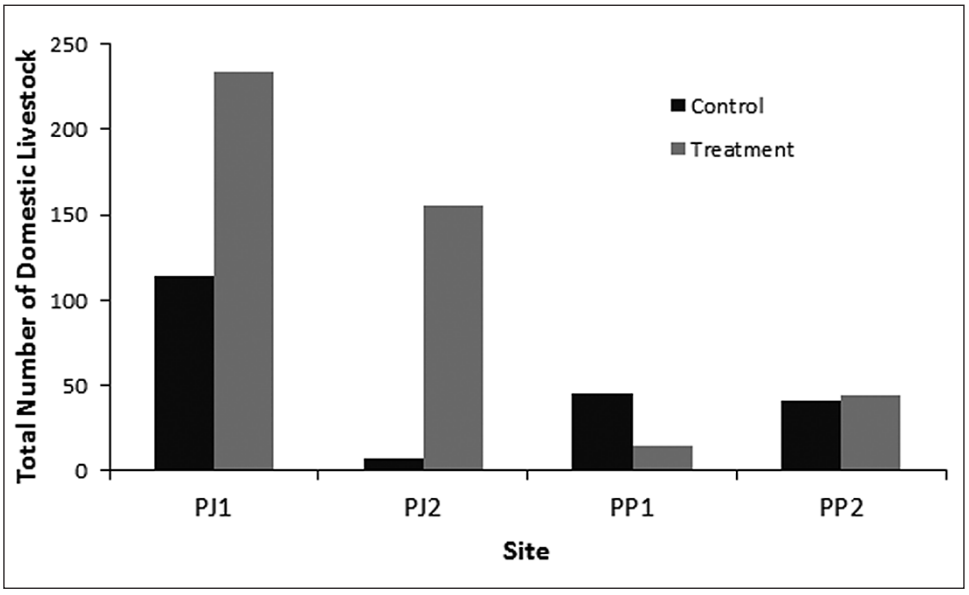

Figure 15—Total number of domestic livestock (cattle and horses) recorded from camera traps on the control and treatment plots at the two pinyon-juniper (PJ1 and PJ2) and the two ponderosa pine (PP1 and PP2) sites following tree thinning treatments, 2012–2015.

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; Verschuyl 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., ≥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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