ASSESSMENT OF FOREST AND WOODLAND TREATMENT EFFECTS ON WILDLIFE




Written by Alexander Evans, Esmé Cadiente, Rhiley Allbee, and Gabe Kohler from the Forest Stewards Guild for the New Mexico Department of Game and Fish.

2019

Assessment of Forest and Woodland Treatment effects on Wildlife

Contents

Written by Alexander Evans, Esme Cadiente, Rhiley Allbee, and Gabe Kohler from the Forest Stewards Guild for the New Mexico Department of Game and Fish in 2019.

Introduction

The impacts of changes in vegetation on wildlife in New Mexico are evident from a range of perspectives. For example, one can consider a particular vegetation type and the various animals that live there or consider all the different vegetation types a particular taxonomic group uses. Because different users will need the following synthesis organized in different ways, we will present similar information organized around vegetation types, taxonomic groups, and treatment types. Where possible we have highlighted links to the New Mexico State Wildlife Action Plan (NMDGF, 2016) and other useful guiding documents. The recommendations provided should be seen as a starting point and readers are encouraged to delve into the literature cited for the vegetation type, treatment, or wildlife species in question.

Climate Change

The influence of climate change on wildlife species and habitat is ubiquitous in New Mexico and influences planning and implementation of treatments. In the western US, temperatures have increased approximately 1.2°C over the last century (NMDGF, 2016; Spears et al., 2013). At the same time, extreme cold spells during the winter have become less frequent (Wuebbles et al., 2014). In New Mexico specifically, surface air temperatures increased 1°C between 1985 and 2005, and most of this warming occurred in the ten years between 1995-2005 (NMDGF, 2016; Rangwala and Miller, 2010). Geographically, temperature increases have been greatest in the southwestern, central, and northwestern regions of New Mexico, particularly within the Jemez Mountains in the northwest (Enquist and Gori, 2008; NMDGF, 2016). These temperature increases create dramatic, cascading effects in the water-limited state of New Mexico, making it part of the most “climate-challenged” regions of North America (Garfin et al., 2013a).

The warming and drying trends in New Mexico interact with other factors such as insect outbreaks and wildfire. On average, the climate in New Mexico is likely to be warmer and drier by the end of the 21st century than it was during the 20th century, with warmer spring and summer temperatures, reduced snowpack and earlier snowmelts, and longer, drier summer fire seasons (Dominguez et al., 2010; Garfin et al., 2013b; Westerling, 2016).

Changes to the amount and quality of snowpack, and their associated impact to stream flow and water availability during the summer months, have been well documented (Fields et al., 2007; NMDGF, 2016). In particular, the amount of precipitation falling as rain versus snow has substantially increased across the western mountains of the US, having a marked increase on water availability during summer fire seasons. As noted in the New Mexico State Wildlife Action Plan, snow melt and hence peak stream flow are occurring earlier than they did historically (Lundquist et al., 2009; McCabe and Wolock, 2007).

Three lines of evidence predict that these warming and drying conditions are likely to cause increased fire activity: reconstructions of fire and climate in the past (Frechette and Meyer, 2009; Swetnam, 1993), trends over the last few decades (Crockett and Leroy Westerling, 2018; Littell et al., 2009), and predictive models (Keyser and Leroy Westerling, 2017; Keyser and Westerling, 2019). Over the last two decades years, wildfires burned an average of nearly7 million acres annually in the U.S. (NIFC, 2019). Many of the recent wildfires have burned large areas at high severities creating both acute and chronic impacts. Loss of homes and post-fire flooding are examples of the profound impacts wildfire can have on people, communities, and economies. Large, severe wildfires can also completely change ecosystem composition, structure, and function. A recent analysis found that over 12 million acres of former forestland are treeless due to wildfire (Sample, 2017).

As the climate becomes warmer (and in many areas drier), large, severe fires are likely to become more common (An et al., 2015; Bowman et al., 2017; Mitchell et al., 2014; Parks et al., 2018; Westerling and Bryant, 2008). In fact, this pattern is already visible. An examination of wildfires in the western U.S. between 1984 and 2011 showed both the number of large fires and the acreage burned increased significantly (Dennison et al., 2014). Regional studies have documented an increase in burn severity in the southwestern U.S. (Dillon et al., 2011; Singleton et al., 2019a)

Climate models predict altered conditions outside of the range that individual trees or populations may be adapted to, resulting in decreased productivity and greater vulnerability to disturbances such as fire, insects, and pathogens (Fettig et al., 2013). Increased success of bark beetles, for example, is linked to the effect of a warming climate on precipitation regimes (Bentz et al., 2010). Additionally, reduced growth and increased mortality in forests is an associated effect of a warmer, drier climate (Fettig et al., 2019; Van Mantgem and Stephenson, 2007).

The changing climate also affects Southwestern ecosystems' ability to respond to disturbances because decreased moisture availability inhibits tree regeneration. Climate models predict that ponderosa pine regeneration rates after forest disturbances will decrease by up to 8% (Feddema et al., 2013). The effect of climate change on forest regeneration may lead to cover type conversions from a forested state to a shrubland or grassland state. This is of greater concern in low elevation xeric sites rather than higher elevation sites with more mesic conditions (Haffey et al., 2018). Moisture stress in the coming century is a critical consideration for management intervention where maintaining a forested state is a priority (Dodson and Root, 2013).

Piñon-juniper woodlands are expected to experience major declines from climate change related drought and insect-related mortality (Fair et al., 2018; McDowell et al., 2016). Heat and drought drive physiological stress, insect outbreaks, and pest attacks and can result in tree mortality even if precipitation remains the same (Morillas et al., 2017). Furthermore, wildfires are expected to increase in future climate scenarios and may lead to more frequent widespread mortality events or forest type conversion of the piñon-juniper woodlands to shrubland habitats (Moritz et al., 2012; NMDGF, 2016). Pinyon jays are expected to experience the brunt of the decline in the condition of piñon-juniper woodlands because of the close relationship between this bird species and the habitat type (Johnson et al., 2017). Unfortunately, some research suggests that treatments may make these conditions worse (Morillas et al., 2017). Thinning poses a risk to migratory bird populations in piñon-juniper persistent woodlands when combined with drought induced mortality associated with bark beetles (Fair et al., 2018).

As climate change alters availability of moisture throughout New Mexico, some feel that historical conditions of forest structure and composition will no longer be suitable targets for management. Disturbances associated with a changing climate—including, increased drought, insect-outbreak, and longer, drier summer fire seasons—could dramatically change the disturbance regimes of New Mexico’s forests so rapidly that future disturbances will no longer be similar to the disturbances that shaped forest conditions in the past (Millar et al., 2007). Still, others argue that historical conditions provide an important baseline for land management, especially as the forest conditions of the last century shift due to climate change, exotic disease, animal, and plant invasions, and increased fuel loads from fire suppression (Keane et al., 2009). In the Southwest, forest and fire regime restoration fosters resilience the warmer and drier conditions (Fulé, 2008).

Vegetation types

In this report we consider piñon-juniper, ponderosa pine, mixed conifer, spruce-fir, and riparian vegetation types. These types can be linked to the National Vegetation Classification System:

1. Piñon-juniper savannah
    1. G252 Southern Rocky Mountain Juniper Open Woodland Group
    2. G487 Madrean Juniper Open Woodland Group
2. Piñon-juniper woodlands and wooded shrublands
    1. G200 Madrean Pinyon - Juniper Woodland Group
    2. G250 Colorado Plateau Pinyon - Juniper Woodland Group
    3. G252 Southern Rocky Mountain Juniper Open Woodland Group
    4. G253 Southern Rocky Mountain Pinyon - Juniper Woodland Group
3. Ponderosa pine
    1. G228 Southern Rocky Mountain Ponderosa Pine Forest & Woodland Group
    2. G229 Southern Rocky Mountain Ponderosa Pine Open Woodland Group
    3. G213 Central Rocky Mountain Ponderosa Pine Open Woodland Group
4. Mixed conifer – xeric
    1. G226 Southern Rocky Mountain White Fir - Douglas-fir Dry Forest Group
    2. G215 Middle Rocky Mountain Montane Douglas-fir Forest & Woodland Group
5. Mixed conifer – mesic
    1. G225 Rocky Mountain Douglas-fir - White Fir - Blue Spruce Mesic Forest Group
    2. G222 Rocky Mountain Subalpine-Montane Aspen Forest & Woodland Group
6. Spruce-fir
    1. G218 Rocky Mountain Subalpine Moist Spruce - Fir Forest & Woodland Group
    2. G219 Rocky Mountain Subalpine Dry-Mesic Spruce - Fir Forest & Woodland Group
7. Riparian (including bosque)
    1. G147 Great Plains Cottonwood - Green Ash Floodplain Forest Group
    2. G506 Rocky Mountain-Great Basin Montane Riparian & Swamp Forest Group
    3. G797 Western Interior Riparian Forest & Woodland Group
    4. G510 Interior West Ruderal Riparian Forest & Scrub Group

Piñon-Juniper

It is often difficult to differentiate between types of piñon-juniper (*Pinus* spp. and *Juniperus spp.*). Sites dominated by piñon-juniper are diverse in structure and fire history, but in this review, we use four broad categories: savannas, wooded shrublands, open woodlands, and persistent woodlands. These four categories are based on current work by the New Mexico Forest and Watershed Restoration Institute (NMFWRI) on differentiating and managing piñon-juniper vegetation types (New Mexico Forest Restoration Principles Working Group, 2007; Reid, 2019). In turn the, NMFWRI key draws heavily from work by Romme and colleagues (2007, 2009). The three-part delineation that Romme and colleagues uses appears useful in many areas, but does not capture the open woodlands common in northern New Mexico. It is crucial to note that differences between piñon-juniper systems are nuanced and gradations that are not easily captured in strict categories. Moreover, the distribution of piñon and juniper across the landscape can shift over time with the impact of droughts and climatic changes (Redmond et al., 2015; Romme et al., 2009). It can be difficult to determine the specific type of piñon juniper ecosystem where research occurred. Past research often describes sites as ‘piñon-juniper woodlands' without differentiating between open woodlands, wooded shrublands, or persistent woodlands. Managers interested in impacts on particular wildlife species or effects of specific treatments should review those sections as well. The New Mexico State Wildlife Action Plan encourages land management agencies and landowners work together to protect Species of Greatest Conservation Need (SGCN) by maintaining and improving all piñon-juniper ecosystems (NMDGF, 2016).

There are a number of general reference texts for management of piñon-juniper including:

- *Draft Piñon-Juniper Framework* and key (New Mexico Forest Restoration Principles Working Group, 2007; Reid, 2019).
- *Historical and Modern Disturbance Regimes of Piñon-Juniper Vegetation in the Western U.S.* (Romme et al., 2007, 2009)
- *Managing Western Juniper for Wildlife* (Miller, 2001)
- *To masticate or not: Useful tips for treating forest, woodland, and shrubland vegetation* (Jain et al., 2018)
- *Sharing the land with pinyon-juniper birds* (Gillihan, 2006)
- *Do Mechanical Vegetation Treatments of Pinyon-Juniper and Sagebrush Communities Work?* (Jones, 2019)
- *Ecology and management of pinyon-juniper ecosystems in the Bureau of Indian Affairs Southwestern Region* (Waconda, 2008)
- *A Field Guide for Selecting the Most Appropriate Treatment in Sagebrush and Piñon-Juniper Ecosystems in the Great Basin* (Miller et al., 2014)

Piñon-juniper savanna

Savannas dominated by piñon pine and juniper, or just juniper, often occupy low hills or valleys with deep soils, where summer monsoons provide most of the annual precipitation. Romme and colleagues (2009) state that savannas

> *are found where local soils and climate are suitable for both trees and grasses; it is logical that low-severity fires may have maintained low tree densities before disruption of fire regimes following Euro-American settlement.*

The low densities of savannas are maintained by water limitations and frequent fires (Margolis, 2014; Savage et al., 2008). Water limitations are expected to increase under future climate change scenarios, further exacerbating the ecological effects of drought on piñon-juniper savannas (NMDGF, 2016).

Piñon-juniper ecosystems provide habitat for a wide range of species including mule deer (*Odocoileus hemionus*), desert cottontails (*Sylvilagus audubonii*), black-tailed jackrabbits (*Lepus californicus*), 70 species of birds, and many other species (Balda, 1987; Howard et al., 1987). The ecotone and shifting boundaries between savannas and other vegetation types allows for the coexistence of closely related species in piñon-juniper savannas and woodlands (Rodhouse et al., 2010). Since Euro-American settlement of the southwest, more trees have grown into savannas because of unregulated grazing, fire suppression, and other forces (Jacobs, 2011). In general, species richness declines as woody plants encroach on savannas (Ratajczak et al., 2011; Rodhouse et al., 2010). The impact of woody encroachment on wildlife is a reduction in

palatable forage (Kramer et al., 2015), which in turn, has negative impacts on wildlife that rely on forage.
In the 20th century, some piñon-juniper woodlands were chained or cabled (the practice of removing all trees on a site by dragging an anchor chain or cable across a site). Chaining was designed to reduce woody competition and increase grass cover. Less common now, chaining is a dramatic ecosystem change with negative effects on a wide range of wildlife including bobcat (*Lynx rufus*), mountain lion (*Puma concolor*), black bear (*Ursus americanus*), golden-mantled ground squirrel (*Callospermophilus lateralis*), and rock squirrel (*Spermophilus variegatus*) (Gallo et al., 2016).

Species adapted to open savannas can benefit from less heavy-handed treatments that remove canopy and return historic conditions. For example, grassland rodents, as a group, were more abundant on areas where managers removed overstory and slash, and were less abundant where slash was left (Severson, 1986). A recent review of management actions that reduce woodland encroachment in grasslands and shrublands found that the majority (69%) of animal species responses to woodland reduction treatments were non-significant (Bombaci and Pejchar, 2016). Bird responses to complete tree removal by mechanical means were often negative (Bombaci and Pejchar, 2016). Even though many treatments are designed to benefit ungulates, Bombaci and Pejchar's review (2016) found non-significant or negative responses to woodland reduction were common. Of course, the specific impacts vary across species and specific locations. Importantly, Bombaci and Pejchar's review (2016) emphasizes the effects of tree reduction are uncertain for many taxonomic groups. This review does not cover grazing which can have significant implications for site ecology and wildlife (Jones, 2019), but there is a significant body of research on the topic (e.g., Bakker et al., 2010).

Recommendations:

- Managers should work to understand the type of piñon-juniper system they are working in using guides such as those from Romme and colleagues or NMFWRI.
- Reducing tree and shrub encroachment may benefit wildlife by increasing forage, but research suggests many treatments fail to have positive wildlife impacts.
- Chaining or other methods of clearing piñon-juniper stands should be avoided.
- Retain the most productive piñon nut crops trees and the piñons with the largest and/or densest canopies to maximize piñon nut mast crops for wildlife.
- Cutting should be done outside of the migratory bird nesting season to avoid mortality in the short-term of eggs or chicks.
- See the treatment section for recommendations for particular treatments and the wildlife section for recommendations related to specific species.

Piñon-juniper wooded shrublands

Where water availability and fire regimes allow, densities of piñon and juniper trees can increase to form shrublands. These wooded shurblands are located where soils and climate support a well-developed shrub community and tree densities are variable: high during periods of moist climatic conditions and infrequent disturbance (lower during droughts and following disturbance) (Romme et al., 2009; Shinneman and Baker, 2009). Wooded shurblands tend to have winter-dominated precipitation and have deep stored moisture (Romme et al., 2009). While shrublands have higher tree densities than savannas, it can be difficult to differentiate sites that naturally support high tree densities and sites that have increased in density since settlement (Jacobs et al., 2008; Romme et al., 2007). Pre-settlement fires were likely often high severity, killing most or all trees and to top-killing most or all shrubs within the burned area (Baker and Shinneman, 2004; Romme et al., 2009).

It is often difficult to determine the type of piñon-juniper type where a specific study occurred, particularly in wooded shrublands. Hence efforts such as the recent review *Do Mechanical Vegetation Treatments of Pinyon-Juniper and Sagebrush Communities Work?* (Jones, 2019) are particularly useful. Much of the impetus for management of wooded shrublands is directed by the desire to create and maintain habitat for greater sage-grouse (*Centrocercus urophasianus*), reduce wildfire risk, and improve grazing conditions (Boone et al., 2018). However, benefits of treatment must be carefully compared to potential habitat loss. Often the best strategy is to maintain or promote heterogeneity and avoid using a single prescription across large areas (Jones, 2019).

Loss, degradation, or fragmentation of piñon-juniper woodlands from conversion, clearing, firewood cutting, improper grazing practices, and altered fire regimes is a threat to SGCN. One such species, the pinyon jay, is the fastest declining landbird in the intermountain West (Boone et al., 2018; Jones, 2019). However, treatments within pinyon jay nesting areas can have negative impacts, resulting in pinyon jays not nesting in treated areas (Johnson et al., 2018a). Treatments in Colorado had negative impacts on obligate bird species, including mountain chickadee (*Poecile gambeli*), white-breasted nuthatch (*Sitta carolinensis),* Steller's jay (*Cyanocitta stelleri*), and Townsend's solitaire (*Myadestes townsendi*; mature conifer species), Clark's nutcracker (*Nucifraga columbiana*) and western tanager (*Piranga ludoviciana*), northern flicker (*Colaptes auratus*), and pinyon jay (Coop and Magee, 2016; Magee et al., 2019). More generally, the negative effects of mechanical treatments on bird species are widespread and influenced bird populations regardless of whether they were piñon-juniper obligate species or not (Bombaci and Pejchar, 2016). These findings were supported by a follow-up study that found overall bird use to be higher in areas that were untreated versus areas that were treated, which was related to piñon-juniper cover (Jones, 2019). A study of piñon decline found vigor declined in areas of higher tree density and increase in areas of lower density (Johnson et al., 2017). Piñon pines have good nut productivity between 75-160 years, maximum productivity between 160-200 years, and declining productivity after 200+ years (Anderson, 2002). In cases where treatments are prioritized in piñon-juniper wooded shrublands, effort should be made to retain the most productive piñon nut crop trees and the oldest, largest, or densest canopy piñon, to maximize piñon nut mast crops for wildlife, maintain higher basal area trees selected by pinyon jays for nest trees (Johnson et al., 2018a, 2015), and provide nest cover from predation.

A study that investigated thinnings in the ecotone between piñon-juniper woodlands and adjacent sagebrush communities found greater net reduction in woodlands resulted in slight shifts in the bird community toward one with shrubland affinities (Knick et al., 2017). The relationship between structural composition and wildlife use suggests structural complexity is linked to increased bird diversity in juniper woodlands (Jones, 2019). Thinning in wooded shrublands should occur outside the nesting season (e.g., late April through August for gray vireo) if there are species in decline on site (Stake and Garber, 2008).

Recommendations:

- Managers should work to understand the type of piñon-juniper system they are working in using habitat guides (e.g., Romme et al., 2007).
- Care should be taken to avoid short-term negative impacts to nesting bird species by cutting outside of nesting season for piñon-juniper obligate bird species.
- Maintain patches with high trees densities and heterogeneity at the landscape scale.
- See the treatment section for recommendations for particular treatments and the wildlife section for recommendations related to specific species.

Piñon-juniper open woodlands

Open woodlands are usually found on productive upland sites, adjacent to ponderosa pine forests (New Mexico Forest Restoration Principles Working Group, 2007). They tend to form uneven-age stands on the rolling uplands prevalent on the Colorado Plateau in northern Arizona and northern New Mexico. Open woodlands are often a transitional between deep, well-drained soils that support savannas and the shallow, coarse soils that support persistent woodlands. Though more research is needed to understand the historic fire regime in these woodlands, the current hypothesis is that grasses carried frequent, low-severity surface fires, with pockets of denser fuels supporting occasional high-severity patches (New Mexico Forest Restoration Principles Working Group, 2007). In-filling of trees is common in piñon-juniper stands across the Southwest (Jacobs, 2011).

Ungulates, such as deer and elk, find important habitat in piñon-juniper woodlands, particularly for thermal cover and bedding (Bender, 2012). Stands that have become more dense since settlement are likely to have sparse ground cover as trees compete with herbaceous cover (Stoddard et al., 2008). Depending on their condition, piñon-juniper forest types may provide little forage for ungulates such as mule deer while still potentially providing important cover (Bender et al., 2009). To protect ungulate habitat, stand structures that provide both cover and forage must be maintained in mosaic throughout open woodlands.

Many treatments are designed to reduce tree densities and increase herbaceous cover of forage. Thinning has been shown to increase forage, and the combination of thinning or mastication with seeding is particularly effective for increasing native annual forb biomass (Stephens et al., 2016a). For perennial grasses, on the other hand, evidence of treatment effectiveness is highly variable and uncertain (Jones, 2019). For example, some treatments can facilitate non-native species such as cheatgrass (*Bromus tectorum*) (Coop and Magee, 2016). Increasing grass abundance benefits wildlife species that rely on forage. Thinning can also increase forb and shrubs, which can benefit species like deer (Brockway et al., 2002). Deer use has been shown to increase proportionally to the amount of trees removed (Albert et al., 1994). Mule deer preferentially selected areas thinned and burned in ponderosa pine stands with a juniper understory because of increased forage (Horncastle et al., 2013). However, most investigators found that treatments to piñon-juniper woodlands result in either non-significant or negative effects on mule deer and elk (Bombaci and Pejchar, 2016).

Reducing high tree densities appears to benefit small mammals. Total rodent numbers were significantly greater in treated areas (Severson, 1986). However, it is worth noting that individual species and groups responded to treatments differently. For example, piñon mice (*Peromyscus truei*) and rock mice (*P. nasutus*) were more abundant where slash was present and the overstory was relatively intact but grassland rodents, as a group, were more abundant on areas where managers removed overstory and slash (Severson, 1986).

Thinning in piñon-juniper woodlands generally has negative effects on bird populations. A study of mastication and hand-thinning treatments in piñon-juniper woodlands found a substantial reduction in the occupancy of piñon-juniper specialist and conifer obligate bird species, including the Virginia's warbler (*Oreothlypis virginiae*) and gray flycatcher (*Empidonax wrightii*) at the landscape scale, and the pinyon jay at the local scale (Coop and Magee, 2016). The study's authors go on to suggest that less intense thinning followed by surface fuel reductions (e.g., via prescribed fire) and/or crown base height increases (e.g., via pruning) may be best at meeting both fuel reduction and wildlife goals (Coop and Magee, 2016). Still, aligning vegetation management and wildlife goals poses a challenge to routine management. Many thinning prescriptions are designed to reduce juniper densities and preferentially retain piñon even though juniper are preferred nesting sites in some areas of New Mexico (Francis et al., 2011). Furthermore, many responses to thinning treatments are not related to species selection but stand structure. Studies of the gray vireo, for example, show that reductions in canopy cover due to thinning can adversely affect reproduction regardless of the residual tree species composition (NMDGF, 2016; Stake and Garber, 2008). The relatively short time-period between thinning treatments and bird occupancy studies could explain why many birds showed negative responses to thinning, and bird species like the Brewer's sparrow (*Spizella breweri*), that show positive responses to thinning, appear to be anomalies (Jones, 2019).

One study found bats were more reproductively active in piñon-juniper woodlands than ponderosa pine forest, highlighting the importance of woodlands for bats (Chung-MacCoubrey, 2005a). Thinning and prescribed fire have demonstrated positive benefits for bats by increasing herbaceous vegetation, insect habitat, and flight space in the forest (Boyles and Aubrey, 2006; Taylor, 2006). To maintain bat habitat in piñon-juniper woodlands, it is important to preserve as many suitable roost trees as possible. Preferences for roost trees varies and in the Gallina mountains northwest of Magdalena, New Mexico the western long-eared bat preferred junipers with an average diameter of 48 cm while *M. volans* perferred piñon with an average diamter of 27 cm (Chung-MacCoubrey, 2005a, 2005b).

Research on reptiles and insects in southwestern piñon-juniper is sparse. One study using butterflies as an indicator suggests thinning and slash reduction in piñon-juniper had a positive effect, at least initially (Kleintjes et al., 2004).

Recommendations:

- Managers should work to understand the type of piñon-juniper system they are working in using guides such as those from Romme and colleagues or NMFWRI.
- Chaining or other methods of clearing piñon-juniper stands should be avoided.
- Retain the most productive piñon nut crops trees and the piñons with the largest and/or densest canopies to maximize piñon nut mast crops for wildlife.

- Cutting should be done outside of the migratory bird nesting season to avoid mortality in the short-term of eggs or chicks.
- Identify and retain roost trees for bat species.
- See the treatment section for recommendations for particular treatments and the wildlife section for recommendations related to specific species.

Piñon-Juniper persistent woodlands

Piñon-juniper persistent woodlands typically occur on sites with higher moisture availability than savannahs or shrublands. They range from sparsely populated stands of small trees on poor substrates to more dense stands of older trees on more productive sites with canopies composed of either piñon or juniper (Romme et al., 2009). These stands are protected from high severity fire by poor soil conditions that limit the amount and connectivity of grassy fuels able to carry fire through the stand, except in cases of extreme drought or wind (Floyd and Romme, 2012). In general, piñon-juniper stands become more dense and increase in structural complexity as the time since the last fire increases (Huffman et al., 2012). Persistent woodlands have very long fire rotations, often greater than 400 years (Romme et al., 2009). They provide important habitat for a diverse range of obligate bird communities, including many species of conservation concern, such as the pinyon jay (Coop and Magee, 2016). Up to 20 percent of the bird species in piñon-juniper persistent woodlands are obligate species, and many of these are in decline (Paulin et al., 1999). Bat species, such as the western long-eared bat (*Myotis evotis*), find important summer maternity roost habitat within piñon-juniper persistent woodlands (Snider et al., 2013).

In some grasslands and shrubland systems, piñon and juniper regeneration has created dense stands that appear like persistent woodlands. Distinguishing true persistent woodlands and other areas that have more recently become dense is difficult, which has created some controversy about what type of management intervention, if any, is needed in these systems (Romme et al., 2009). Since persistent woodlands provide habitat for so many obligate species, the benefits of treatment must be carefully weighed against the cost of potential habitat loss associated with treatments. In fact, because of the relative scarcity of these stands on the landscape and the long fire-free periods required to develop persistent woodland characteristics, no treatment is often appropriate.

Recommendations

- Managers should work to understand the type of piñon-juniper system they are working in using guides such as (Reid, 2019; Romme et al., 2007).
- The benefits of treatments for grazing and wildfire risk reduction should be carefully weighed against the potential to decrease migratory bird, small mammal, and bat habitat in persistent woodlands. No treatment may be the best option.
- Retain the most productive piñon nut crops trees and the oldest/largest/densest canopy piñons to maximize piñon nut mast crops for wildlife, maintain higher diameter trees selected by pinyon jays for nest trees and provide nest cover from predation.
- Identify and retain roost trees for bat species in persistent woodlands.
- Care should be taken to avoid negative impacts to nesting bird species by cutting outside of nesting season for piñon-juniper obligate bird species.

- Maintain patches with high trees densities and heterogeneity at the landscape scale.
- See the treatment section for recommendations for particular treatments and the wildlife section for recommendations related to specific species.

Ponderosa pine

Historically, Southwestern ponderosa pine (*Pinus ponderosa*) forests were open stands dominated by a single species that experienced low-intensity fire frequently, every 2 to 12 years on average (Hunter et al., 2007). These forests have become much more dense since the late 1800's because of logging, the disruption of natural fire regimes, and livestock grazing followed by favorable climate conditions for tree regeneration (Cooper, 1960; Covington and Moore, 1994; Lynch et al., 2000; Savage and Swetnam, 1990). The change in stand characteristics and 20th century fire suppression strategies mean that wildfires now usually burn at an uncharacteristically high severity instead of the low-intensity fires of the past. The management of these uncharacteristically dense forests and their related wildfire hazard is one of the most important land stewardship issues in the western United States (Noss et al., 2006). There is wide agreement about the need to restore ponderosa pine forest structure and to return fire as the key disturbance for ecosystem structure and function (e.g., Allen et al., 2002). In New Mexico, the Forest Restoration Principles capture the zone of agreement about how most treatments should be implemented in ponderosa pine forests (Bradley, 2009; TNC et al., 2006). Other useful summaries of forest and wildlife management in ponderosa pine include:

- *Guidelines for managing wildlife habitats in southwestern ponderosa pine forests of the United States* (Ffolliott, 1997)
- *A comprehensive guide to fuels treatment practices for ponderosa pine in the Black Hills, Colorado Front Range, and Southwest* (Hunter et al., 2007)
- *Restoring composition and structure in southwestern frequent-fire forests* (Reynolds et al., 2013)

For many species, the habitat value of ponderosa pine forests declined as they became densified and dominated by small trees over the last century. The associated change to forest structure, including a reduction in the prevalence of forest openings, large trees, and snags has reduced habitat value (Dahms and Geils, 1997). Dense stands with little sunlight on the forest floor have impoverished understories with reduced diversity and production of understory species. Yet the herbaceous layer provides essential plant diversity and critical food and habitat for many wildlife species in ponderosa pine forests (Bakker et al., 2010). Long lasting drought conditions have been shown to shift the ecotone between ponderosa pine and piñon-juniper woodland creating more fragmented forest patches and increased soil erosion (NMDGF, 2016). It is yet to be determined what effects this shift in ecotone may have on wildlife habitat over the long term (Flatley and Fulé, 2016; NMDGF, 2016).

Guided by forest restoration principles (Bradley, 2009; TNC et al., 2006), most treatments in ponderosa pine forests focus on removing small diameter trees, restoring a more open structure, and returning fire as a source of natural disturbance. Wildlife adapted to natural conditions (open stands and frequent fire) are likely to benefit from treatments that return or enhance these conditions. For example, small mammals showed an increase in species density and biomass after thinning (Kalies and Covington, 2012). Thinning small trees and promoting the growth of larger ponderosa pine benefits species associated with large diameter trees such as northern goshawk (*Accipiter gentilis*), brown creeper (*Certhia americana*), and many woodpeckers (Hunter et al., 2007). Ponderosa pine restoration treatments showed no effect on home ranges of Abert’s squirrels (*Sciurus aberti*) (Yarborough et al., 2015), though it is important to note that inter-annual abiotic fluctuations can obscure effects (Yarborough et al., 2015). Avian species have also been shown to benefit from restoring a natural fire regime to ponderosa pine forests. For example, the Lewis’s woodpecker (*Melanerpes lewis*) and Grace’s warbler (*Setophaga graciae*), two species identified by The New Mexico State Wildlife Action Plan as SGCNs, are negatively influenced by the loss of their nesting habitat in ponderosa pine from altered fire regimes (NMDGF, 2016).

While there is a zone of agreement around restoration in ponderosa pine, implementing the same prescription everywhere is not optimal for wildlife. Treatment heterogeneity provides important wildlife habitat. For example, Abert’s squirrel, a species that is closely linked with ponderosa pine, shows a preference for denser, untreated patches and therefore retaining some denser canopy areas may benefit them (Loberger et al., 2011). Specifically, winter core habitat areas for Abert’s squirrels should have canopy closure ranging from 55% to 72% to maximize squirrel density and recruitment while meeting other ecological goals (Yarborough et al., 2015). While ungulates benefit from increased forage in open stands or forest gaps, denser stands provide protection and thermal cover (Ffolliott, 1997; Hunter et al., 2007). In the Southwestern U.S., patches of Gambel oak (*Quercus gambelii*) are an important component of productive wildlife habitat, providing browse and acorn mast crops for elk (*Cervus canadensis*), deer, turkey, and many other game and non-game mammals and birds, and also providing cover and nesting structure for wildlife (Reynolds et al., 1970a; Ryniker et al., 2006). Specifically, game and non-game birds have been shown to benefit with increased densities of Gambel oaks in the 12- to 14-inch diameter class (Reynolds et al., 1970b) and non-game bird SGCN’s have shown benefits with increased densities of Gambel oak poles in the 3- to 6-inch diameter range (Jentsch et al., 2008). Virginia's warblers (*Vemivora virginiae*) have shown preference for young, brushy thickets of Gambel oak for foraging and nesting habitat (Lesh, 1999). Snags, stumps, and coarse woody debris add structure to ponderosa pine forests and habitat for a range of species including small mammals such as mice (*Peromyscus* spp.) (Chambers, 1999). Another important consideration during treatment is protecting or increasing the number of snags and downed logs (Chambers et al., 2005; Marcot et al., 2010). Large snags with broken tops, either burned or unburned, are prime habitat for cavity nesting species (Chambers et al., 2005). At the landscape scale, a diversity of structure and stand development phases from forest openings to areas with dense cover support more species than homogeneous landscapes (Hunter et al., 2007). Horncastle and colleagues (2013) also recommend a mosaic approach of treated areas and retention heterogeneously distributed across the landscape .

Recommendations:

- Restoration of ponderosa pine structure and fire as a process benefits a range of wildlife populations at the landscape level.
- Retaining heterogeneity including dense patches of trees, patches of oaks, and other elements is important for wildlife.

- Protect or increase the number of snags and downed logs, while balancing wildfire threat.
- Gambel oaks should be retained, particularly in the 12- to 14-inch diameter range and poles in the 3- to 6-inch diameter range.
- See the treatment section for recommendations for particular treatments and the wildlife section for recommendations related to specific species.

Mixed Conifer

The term "mixed conifer" is difficult to define and has been used for forests along a broad continuum of climatic zones and includes many different assemblages of species (Dieterich, 1983; Evans et al., 2011; Jain et al., 2012). Mixed conifer forests are made up of a range of species with varying structures and spatial patterns. The mixed conifer forests of New Mexico grow at elevations of 8,000 to 10,000 feet and at lower elevations on north-facing slopes and in canyons (Ronco et al. 1983, Dick-Peddie 1993). This is generally above mono-dominant ponderosa pine forests and below colder spruce-fir forests (Evans et al., 2011). As is common in the Southwest, we differentiate in this report between xeric (warm–dry) and mesic (cool–moist) mixed conifer types. Xeric and mesic types are on a continuum and shift from one type to another depending on aspect, elevation, and moisture (Lydersen and North, 2012). A warming climate has had an impact on mixed conifer forests and is likely shifting the distribution of the xeric and mesic types on the landscape (Kane et al., 2014; Kane and Kolb, 2014).

Useful summaries for mixed conifer forests include:

- *A comprehensive guide to fuels treatment practices for mixed conifer forests: California, central and southern Rockies, and the Southwest* (Evans et al., 2011)
- *A comprehensive guide to fuel management practices for dry mixed conifer forests in the northwestern United States* (Jain et al., 2012)
- *Principles and practices for the restoration of ponderosa pine and dry mixed-conifer forests of the Colorado Front Range* (Addington et al., 2018)

Mixed conifer – xeric

The most common species in the xeric mixed conifer forests include ponderosa pine, Douglas-fir (*Pseudotsuga menziesii*), white fir (*Abies concolor*), blue spruce (*Picea pungens*), limber pine (*Pinus flexilis*), southwestern white pine (*Pinus strobiformis*), Gambel oak, and occasionally aspen (*Populus tremuloides*) (Jones, 1974). Altered fire regimes in ponderosa pine forests have allowed fire-intolerant (and shade tolerant) species to flourish, giving them an appearance of mixed conifer forests (Dahms and Geils, 1997; Evans et al., 2011). Historically, wildfires in mixed conifer forests, particularly the mesic, were less frequent and more variable than those of ponderosa pine forests (Touchan et al., 1996). However, xeric mixed conifer experienced low- to moderate-intensity fires frequently, on average every 13 years in the Southwest, though it is important to note that fire return intervals were often twice that (Evans et al., 2011; Huffman et al., 2015; Swetnam and Baisan, 1996). Small patches of high severity fire (where all trees were killed) likely occurred relatively frequently while large patches were uncommon, but may have occurred on centennial scales (Frechette and Meyer, 2009; Fulé et al., 2003; Jenkins et al., 2011). One of the important changes over the 20th century was the increased homogeneity in the structure of mixed conifer forests at the landscape scale. This homogeneity is due in part to historic logging, fire suppression, and livestock grazing that favored densification of conifer forests over the last century. Modern, homogeneous mixed conifer forests allow wildfires to burn across larger areas at uncharacteristically high-severity (Miller et al., 2009; O'Connor et al., 2014; Romme et al., 2003; Singleton et al., 2019b). Fuel treatments have reduced fire severity in mixed conifer forests (Korb et al., 2012; Prichard et al., 2010; Prichard and Kennedy, 2012).

In many ways, xeric mixed conifer is a transitional type between warmer, drier ponderosa pine forests and cooler, wetter mesic mixed conifer forests. Shifts between these vegetation types can be subtle and can occur across short distances, such as a change in aspect. At the landscape scale, xeric and mesic mixed conifer types intermingle and create a mosaic of structures. It is likely that most wildlife takes advantage of habitat and resources across the xeric to mesic spectrum of mixed conifer forests as they are available throughout different times of the day, seasons, or during wetter or drier years.

Mixed conifer – mesic
Mesic mixed conifer forests include many of the same species as mixed conifer forests with the addition of Engelmann spruce (*Picea engelmannii*), subalpine fir (*Abies lasiocarpa*), and corkbark fir (*Abies lasiocarpa var. arizonica*) (Jones, 1974). Mesic mixed conifer forests tend to be on upper elevations or north-facing slopes and to be denser than xeric mixed conifer forests. As discussed above, fire in mesic mixed conifer forests were less frequent but generally of a higher intensity and severity than in xeric stands (Evans et al., 2011; Tepley and Veblen, 2015). The increased tree density in mesic stands, supported by cooler temperatures and more moist conditions, in turn provides more fuel for higher-severity fire (Holden et al., 2009). Still, large stand-replacing fires were rare, in part because of the heterogeneous composition. Aspen groups would have reduced or extinguished some fires burning through heterogeneous mixed conifer stands (Dieterich, 1983). Larger, stand-replacing fires may have occurred in response to regional climate drivers such as the El Niño-Southern Oscillation (ENSO), drought, and multidecadal climate patterns (Brown et al., 2001; Frechette and Meyer, 2009; Yocom-Kent et al., 2015). Climate change effects exacerbate stressors such as drought, insects, and fire occasionally resulting in wide-spread conifer die-off and the ranges of many higher elevation species shrinking to meet suitable climate habitats (Battles et al., 2008; B. J. Bentz et al., 2010; Ganey and Vojta, 2012; NMDGF, 2016). As mixed conifer ranges contract and lower elevation tree species move upslope, wildlife that rely on these habitats may not be able to disperse to their suitable habitats due to restrictions in terrain, fragmentation and individual species responses to climate change (NMDGF, 2016; Rehfeldt et al., 2006).

Mixed conifer forests provide habitat for several important wildlife species, including threatened and endangered species such as the Mexican spotted owl (MSO; *Strix occidentalis lucida*), a SGCN in New Mexico, as well as more common species such as elk and deer. Spotted owls have a particularly large influence on management because of recovery plans and other guidance (USFWS, 1995; Verner et al., 1992). There is a perception that conservation of spotted owls through habitat protection conflicts with fuels treatments (Prather et al., 2008). However, at the landscape scale, the area in which high priority treatments and spotted owl habitat overlap is relatively small—only about one third of the area (Gaines et al., 2010; Prather et al., 2008). One of the reasons for the perception of conflict between habitat and fuels treatment is the spotted owl's requirement for dense forests with high canopy closure and multiple canopy layers (Gaines et al., 2010; USFWS, 1995).. For the MSO, the recovery plan calls for the retention of high canopy density in nesting and roosting habitat called protected activity centers (PACs) (USFWS, 1995). Similarly, the MSO recovery plan recommends management of 25 percent of the landscape for future nesting and roosting habitat; this area should consist of stands that have greater than 130 square feet of basal area per acre and include more than 20 trees per acre that are greater than 18 inches DBH (USFWS, 1995). The recovery plan recommends maintaining the remaining 75 percent of the landscape for foraging habitat, which generally has much lower tree densities and is often managed for uneven-aged structure and retention of large downed logs (USFWS, 1995). As highlighted in the New Mexico State Wildlife Action Plan (NMDGF, 2016), one of the biggest threats to spotted owls is large scale, high severity wildfires and loss of old growth forest (Stephens et al., 2016b). Hence many fuels treatments in mixed conifer and connected ponderosa pine forests are aimed at protecting old growth and mitigating high severity wildfires by re-introducing low severity "good" fire and historic, natural fire regimes. A key element of promoting healthy owl populations is supporting healthy small mammal populations as prey. Management that fosters habitat for small mammals such as sustaining shrub and herbaceous vegetation by thinning small diameter trees, using prescribed fire, and managing grazing pressures can support small mammals and the owls that rely on them for food (Block et al., 2009).

USFS managers have been successful at integrating protections for MSO into other goals such as reducing the risk of high severity wildfire. For example, on the Cibola National Forest, fire managers work to keep fire intensities low and thereby minimize impacts to spotted owl habitat (Evans et al., 2011). On Coconino National Forest, managers seek to maintain healthy MSO populations through a mix of group selection to create open gaps for regeneration, thinnings to reduce tree density in the rest of the stand to encourage growth of larger diameter trees, and retention old trees of species that are more fire resistant in groups of up to four acres in size (Evans et al., 2011). Spotted owls have shown resilience to mixed severity fire (Lee, 2018), and research suggests that controlled burning could be an effective tool in restoring habitat to natural conditions with minimal short-term impact on spotted owls (Bond et al., 2002).

The Jemez Mountains salamander (JMS; *Plethodon neomexicanus*) is another example of a rare endemic animal and SGCN that lives in mixed conifer forests. The majority of this endangered species' habitat is on federally managed lands, including the U.S. Forest Service, Bandelier National Monument, Valles Caldera National Monument, and Los Alamos National Laboratory, though some habitat is located on tribal and private lands (USFWS, 2013). JMS lives under and in fallen logs and old, stabilized talus slopes, especially those with a good covering of damp soil and plant material, which are key habitat elements for the species (Reagan, 1972). A recent study indicates that historical mean fire intervals ranged from 10 to 42 years in JMS habitat and tended to be low severity (Margolis and Malevich, 2016). Studies of wildfire effects on JMS have shown some resilience at the population level (Cummer and Painter, 2007). Forest type conversion from tree to shrub canopy has negative effects on JMS (Cummer and Painter, 2007; NMDGF, 2016).

In contrast to JMS, elk are common in many mixed conifer forests—too common for many forest managers, since they suppress aspen regeneration. Ungulate browse of aspen can be a significant issue in mesic mixed conifer forests (Kota and Bartos, 2010). Thinning that reduces stand densities and counteracts forest encroachment of meadows could increase distribution, quantity, and quality of forage (Halbritter and Bender, 2015), and therefore increase elk use. Research suggests that thinning older stands of mixed conifer forest that are overly dense compared with historical conditions benefit the mammal community through increases in diversity and abundance (Wampler et al., 2008). For example, abundance of the gray-footed chipmunk (*Tamias canipes*) and long-tailed vole (*Microtus longicaudus*) were great in thinned forests and most large mammals were documented in thinned areas (Wampler et al., 2008).

Little research exists on the effect of fire in mixed conifer forests on wildlife. A study of spotted owls in California, Arizona, and New Mexico suggests that wildfires may have little short-term impact on survival, site fidelity, mate fidelity, and reproductive success (Bond et al., 2002). Another study of fire in mixed-conifer forests of California indicates that bats are resilient to landscape-scale fire. Researchers identified that some bat species select burned areas for foraging, which may be because fires reduce understory clutter and increased availability of prey and roosts (Buchalski et al., 2013). Fire can help shift the distribution of coarse woody debris towards historical conditions by decreasing the number of smaller diameter pieces and increasing the number of larger diameter and less decayed pieces, which in turn could benefit wildlife that rely on dead wood (Knapp, 2015). At the same time, research highlights the importance of fire refugia (unburned areas within fire perimeters) for the survival of range of wildlife, particularly where fires are larger, and more severe than they had been historically (Blomdahl et al., 2019). Restoration of natural fire regimes and the forest structure to support natural fire reduces the chances of a large scale, high severity wildfire which can have negative impacts on many species, particularly SGCN (NMDGF, 2016). In addition to using fire as a management tool, other land management practices that restore forest structure and function to provide suitable habitat for SGCN species is recommended in the New Mexico State Wildlife Action Plan (NMDGF, 2016).

Recommendations

- Managers should work to understand the type of mixed conifer forest they are working in and its place the landscape.
- Treatments should foster heterogeneity at the landscape scale and recognize the impact the warming climate is having on distribution of the xeric and mesic types
- Treatments should protect old trees, reduce the likelihood of high severity wildfires, and restore fire regimes.
- See the treatment section for recommendations for particular treatments and the wildlife section for recommendations related to specific species.

Spruce-fir

Dense forests of spruce (*Picea engelmannii*) and fir (*Abies lasiocarpa*) trees occupy the highest elevations in New Mexico. These forests are characterized by cool temperatures and moist conditions (Moir and Ludwig., 1979). Unlike some forests, there is little understory diversity as spruce-fir stands mature (Dye and Moir, 1977). Fires are rare in these forests because of cool, moist conditions. It is not uncommon for spruce-fir forests to remain unaffected by fire for 400 years (Sibold et al., 2006). However, when climatic conditions do allow for fires to burn in spruce-fir forests, they are high-severity, stand replacing. Stand reconstructions have found burn patches as large as 521 ha in spruce-fir forests (Margolis et al., 2011). Fire return is likely linked to climate conditions, stand structure, and composition more than just age (Gass and Robinson, 2007). Because of the long fire return intervals and limited accessibility, spruce-fir forests have been altered less by forest management, grazing, and altered fire regimes than lower elevation forests. However, there is evidence that a changing climate has allowed trees to fill in alpine meadows (Dyer and Moffett, 1999).

Spruce-fir forests can be inhospitable for wildlife because of the cold, snow, and lack of food sources. However, mule deer and elk, when congregated in winter herds, may browse fir (more palatable) and spruce (less palatable) but likely only as a last resort (Alexander, 1987). Red squirrel (*Tamiasciurus hudsonicus)* thrive in the montane, spruce-fir forests of New Mexico. An SGCN that thrives in spruce-fire forests is the Boreal owl (*Aegolius funereus)*, which relies on undisturbed spruce-fir for their habitat. Loss of undisturbed habitat because of timber harvest, insect damage, or large-scale wildfire is negatively impacting the Boreal owl (NMDGF, 2016).

Insects are another major disturbance agent in spruce-fir forests. Spruce beetle (*Dendroctonus rufipennis*) has killed large areas of spruce-fir and warmer, drier conditions may promote extensive spruce beetle outbreaks in the future (O'Connor et al., 2015). Another insect, spruce aphid (*Elatobium abietinum*), has become a new mortality agent in spruce-fir forests (Lynch, 2004). It is worth noting that insect outbreaks in spruce-fir forests do not necessarily increase fire severity (Andrus et al., 2016). A changing climate, however, will make it harder for spruce and fir seedlings to establish after a high severity fire (Harvey et al., 2016). Tree mortality in Colorado Front Range spruce-fir forests has increased even in the absence of lethal bark beetle outbreaks, likely due to moisture stress (Oswald et al., 2016; Smith et al., 2015).

There is little information on management or thinning of spruce-fir forests in the Southwest. In Colorado and farther north, there is a history of group selection, shelterwood, and clearcutting (Alexander, 1987). In the southwest, spruce-fir forests have long been valued more for their provision of water and recreation opportunities than their potential for timber (Reynolds 1962). Because spruce-fir forests have long fire return intervals and are adapted to a high severity fire regime, thinning to reduce fire severity is inappropriate in these forests.

Recommendations

- Though the warming climate is stressing spruce-fir forests, there is little management can do to benefit wildlife in this vegetation type.
- See the treatment section for recommendations for particular treatments and the wildlife section for recommendations related to specific species.

Riparian

In the arid Southwest riparian areas are crucial for wildlife. Riparian systems range from the cottonwood (*Populus spp.*, particularly *P. deltoides wilizeni*) gallery forests of the Rio Grande to small upper elevation streams in mixed conifer or spruce-fir forests. In all these environments, riparian systems provide habitat for increased species diversity and provide irreplaceable resources for a wide range of wildlife. For example, riparian woodlands in Arizona support a greater number and a wider variety of birds than nearby upland ecosystems (Carothers et al., 1974). Declines in native riparian vegetation is linked to declines in breeding and migratory birds (Johnson et al., 2010; McGrath et al., 2009; NMDGF, 2016). Riparian corridors are important for the dispersal and genetic connectivity of amphibians, which was demonstrated in a study on the Rocky Mountain tailed frog (*Ascaphus montanus*) in Idaho (Spear and Storfer, 2010). Rivers and streams provide drinking water and high quality foraging habitat for bats (Taylor, 2006). The importance and interconnections of riparian systems could fill this entire report and are well documented in the Wildlife Action Plan (NMDGF, 2016). Other useful guidance for riparian systems includes:

- *Habitat Restoration and Management of Native and Non-native Trees in Southwestern Riparian Ecosystems* (NMDGF, 2017)
- *A Guide for Planning Riparian Treatments in New Mexico* (NRCS and NMACD, 2007)
- The proceedings from the conference *Desired future conditions for Southwestern riparian ecosystems: Bringing interests and concerns together*
    - A preliminary riparian/wetland vegetation community classification of the Upper and Middle Rio Grande watersheds in New Mexico (Durkin et al., 1995)
    - Avian community composition and habitat importance in the Rio Grande corridor of New Mexico (Leal et al., 1995)

Humans have dramatically altered the hydrology of arid riparian systems through stream diversions, flood control, and groundwater extraction (Stromberg et al., 2013). In the past, the Rio Grande meandered and occasionally filled the floodplain creating a dynamic system (Molles et al., 1998). Dams create barriers to fish movement and release water with substantially less volume and variability than natural flows (Franssen et al., 2007; NMDGF, 2016). Additionally, changes to hydrology, such as a lack of secondary channels, flooding, perched channels, and increasing temperatures, increase the threat of wildfire in bosque systems. The spread of non-native species has compounded human impacts. For example, salt cedar (*Tamarix spp.*) and Russian olive (*Elaeagnus angustifolia*) out-compete native species and choke out native regeneration. Moreover, the changes in species composition, increased tree density, and accumulation of woody debris provide additional increases to the threat of wildfire (Bateman et al., 2008a; Drus, 2013). Human impacts on riparian systems have caused a significant decline in the abundance and variety of birds in these ecosystems (Bock and Block, 2005). Removal of non-native vegetation can benefit native wildlife (Bateman et al., 2008b, 2008a; Mosher and Bateman, 2016), but it is also important to note that non-native vegetation can provide valuable habitat for some wildlife species in areas where conditions are not conducive to native riparian vegetation and non-natives may provide the only available habitat (Katz and Shafroth, 2007; USFWS, 2002; Walker, 2006). Additionally, non-native tree species can provide nesting structure, roosts sites, foraging opportunities, and cover for many wildlife species (NMDGF, 2017). Specifically, at least five New Mexico bird SGCN's breed in salt cedar (NMDGF, 2016), and salt cedar dominated riparian woodlands are part of the critical habitat for the endangered southwestern willow flycatcher (*Empidonax traillii extimus*) and the endangered western yellow-billed cuckoo (*Coccyzus americanus*). Other research provides useful reviews of salt cedar control, water use, wildlife use, and riparian restoration (Hultine et al., 2010; Shafroth et al., 2005).

Given the importance of fire to other vegetation types in New Mexico it is worth discussing its role in riparian systems. In general, fire was less important in riparian systems than connected upland vegetation (Bock and Block, 2005). However, recent research suggests that in montane riparian systems, low-intensity fires may be important for maintaining overall watershed health.

It may exclude an important disturbance from upper elevation riparian and stream habitats (Arkle and Pilliod, 2010). In the Sierra Nevada, research indicates that coniferous riparian forests could be managed for fuel loads and fire return intervals similar to adjacent upland forests (Van de Water and North, 2010). Thinning of conifers and non-riparian shrubs near upper elevation streams could reduce the threat of uncharacteristic wildfire (and hence benefit wildlife that depend on those forests) with little effect on bird density and reproductive success (Stephens and Alexander, 2011). Similar thinnings in New Mexico of conifers near riparian systems appear to have a positive impact on small mammals (Bagne and Finch, 2010). Further research is need in New Mexico to assess the trade-offs between effects of high severity fire and thinning of conifers near riparian habitats. Again, fire is unlikely to be part of habitat restoration in bosque systems in the valley where recent fires have been highly destructive of cottonwood (*Populus spp.*), sycamore (*Platanus spp.*), and other native riparian vegetation (Bock and Bock, 2014).

As the climate warms, riparian systems will become both more important and more threatened (Krosby et al., 2018). Increased prevalence and severity of droughts will negatively affect riparian vegetation and hence wildlife that depend on rivers and streams (Perry et al., 2012). Riparian habitats and the wildlife dependent on them are particularly vulnerable because they are relatively isolated and fragmented (Morgan et al., 1994; NMDGF, 2016). There are opportunities to use the existing flood control system to manage water releases in ways to support and even restore riparian ecosystems (Molles et al., 1998). Restoration of suitable flows and riparian habitat for SGCN is an area of concern and needed conservation action in the Wildlife Action Plan (NMDGF, 2016). The New Mexico Department of Game and Fish has recommended the following guidelines for managing riparian ecosystems within *Habitat Restoration and Management of Native and Non-native Trees in Southwestern Riparian Ecosystems* (NMDGF, 2017):

- Restore native riparian plants following non-native removal or biocontrol and maintain an adequate water supply for native plants.
- Incorporate native drought-tolerant woody and herbaceous species in restoration plantings to address expected changes in climate and low water availability.
- Consider implementing streambank and floodplain modifications (e.g., bank softening, bank lowering) following non-native removal to ensure maintenance of overbank flows, river-floodplain connections, and native plant communities.
- Stage and balance salt cedar removal and native habitat restoration over time to avoid rapid loss of non-native woody riparian habitats for wildlife until alternative native habitats can be developed.
- Protect and sustain existing stands of native riparian vegetation that may serve as important refugia in areas currently or likely to be affected by non-native control efforts.
- At sites where non-native trees are removed from the understory of mature riparian forests, consider planting native trees, shrubs, and herbaceous plants to maintain vertical habitat diversity and ground cover.
- See the treatment section for recommendations for particular treatments and the wildlife section for recommendations related to specific species.

Treatments

This is not a guide to the development of treatment prescriptions because a wide range of factors should be considered in addition to wildlife. However, a number of other guides or syntheses can be helpful in this regard:

- *Short guide for developing CFRP restoration prescriptions* (Savage et al., 2008)
- *A comprehensive guide to fuels treatment practices for ponderosa pine* (Hunter et al., 2007)
- *A comprehensive guide to fuels treatment practices for mixed conifer forests* (Evans et al., 2011)

A central theme to the discussion of treatments is reducing the impact of high severity fire, so it is important to be clear about the effectiveness of treatments in reaching that goal. The scientific consensus on the ability of fuel reduction treatments to change fire behavior has solidified. Modeling provides one avenue for testing the effectiveness of fuel treatments (Collins et al., 2013; Johnson et al., 2011; Loudermilk et al., 2014; Van de Water and North, 2011). Fuel treatments have also been tested by wildfire and proved to reduce severity (Cochrane et al., 2012; Cram et al., 2006; Dailey et al., 2008; Korb et al., 2012; Pollet and Omi, 2002; Prichard et al., 2010; Prichard and Kennedy, 2012; Safford et al., 2012; Stevens-Rumann et al., 2013; Stevens et al., 2014; Wimberly et al., 2009). Importantly, the effects of fuels treatments can reduce wildfire severity even under extreme conditions (Lydersen et al., 2017; Prichard and Kennedy, 2013). While there are important nuances and each wildfire is different, in general, the science is clear: fuel treatments can reduce wildfire severity.

The following sections review a range of treatments and objectives because the specifics of how vegetation is changed are important for understanding wildlife impacts. Managers interested in impacts on particular wildlife species or forest types should review those sections as well.

Thinning

Thinning is a forest treatment made to reduce stand density of trees primarily to improve growth, enhance forest health, remove trees likely to die from competition (Helms, 1998). Thinnings can focus on the understory trees (low thinning), canopy trees (crown thinning), or a mix (free thinning). Often when thinnings remove trees that are smaller than the standard merchantable size in the region they are termed ‘pre-commercial.’ In the Southwest, thinnings are often pre-commercial and instead are designed to restore historic conditions. These are low thinnings which means removing smaller, younger trees (Allen et al., 2002). Reducing the hazard of wildfire through treatments to forest vegetation typically involve one or some combination of reducing tree density through thinning and changes to surface fuels through burning, mastication, or removal. Combined treatments, especially burning and thinning are the gold standard when it comes to reducing the risk of high severity fire. Meta-analyses have found that when compared to burning or thinning alone, combined treatments were most effective at reducing surface fuels, stand density, and the likelihood of crown fire (Fulé et al., 2012; Kalies and Yocom Kent, 2016). Broadcast or pile burning after thinning is important to eliminate activity fuels generated by the forest treatment. If wood debris and other residual biomass generated by the treatment are not disposed of, they largely offset the hazard reduction benefit from opening the canopy (Martinson and Omi, 2013; Van de Water and North, 2011).

The specific prescription for a thinning project is important in how it affects wildlife and prescriptions vary greatly by both land management objectives and forest type. As described in previous sections, dedicated to each forest type, there are useful overview guides for different ecosystems:

- *Historical and Modern Disturbance Regimes of Piñon-Juniper Vegetation in the Western U.S* (Romme et al., 2007, 2009)
- *Managing Western Juniper for Wildlife* (Miller, 2001)
- *A comprehensive guide to fuels treatment practices for ponderosa pine in the Black Hills, Colorado Front Range, and Southwest* (Hunter et al., 2007)
- *Restoring Composition and Structure in Southwestern Frequent-Fire Forests* (Reynolds et al., 2013)
- *A comprehensive guide to fuels treatment practices for mixed conifer forests: California, central and southern Rockies, and the Southwest* (Evans et al., 2011)
- *Ecology, silviculture, and management of the Engelmann spruce-subalpine fir type in the central and southern Rocky Mountains* (Alexander, 1987)
- *A Guide for Planning Riparian Treatments in New Mexico* (NRCS and NMACD, 2007)

Acknowledging the important specifics of thinning prescriptions, forest types, and wildlife species, it is also useful to highlight some general patterns of how thinnings affect wildlife. For example, thinnings open canopies and provide more light and resources to the herbaceous layer. In piñon-juniper savannas, thinning can result in elevated forb and graminoid cover (S. K. Albert et al., 1994; Coop et al., 2017). Evidence from Oregon suggests that thinning overly dense stands can help provide food for wildlife by encouraging higher cover of flowering, fleshy fruit and palatable leaf producing species (Neill and Puettmann, 2013). Thinnings often benefit bats by increasing herbaceous vegetation and insect habitat and flight space in the forest (Taylor, 2006). Thinnings that open gaps and increase shrubs may increase the abundance of nesting birds (Siegel and Desante, 2003).

An extensive meta-analysis of individual studies examined the effects of thinning (removal of small-to-intermediate diameter trees including both thinning and shelterwood treatments) and burning treatment in Southwestern ponderosa pine and mixed conifer forests (Kalies et al., 2010). The researchers found found that:

- small mammals had no response to thinning and burning;
- small mammals had a negative response to overstory removal and wildfire;
- ground-foraging birds and rodents had no strong response to thinning and burning;
- canopy-foraging birds had positive or neutral responses to thinning and burning; and
- canopy-foraging birds had negative responses to overstory removal and wildfire (Kalies et al., 2010).

In sum, the meta-analysis found that in the short-term (≤10 years), small-diameter removal and burning treatments do not negatively impact most of the wildlife species studied (Kalies et al., 2010). Thinning is likely to have the largest positive impact on small mammal populations in areas where tree densities are especially high compared to pre-settlement conditions (Converse et al., 2006). Similarly, thinning in piñon-juniper woodlands increased small mammal abundance, and deer use while songbirds did not show a significant response (Albert et al., 1994). A study in ponderosa pine outside of Santa Fe, New Mexico found the majority of bird species, including those identified as of regional concern, had a positive response to thinning (Bagne and Finch, 2008). It is important to note that chaining or cabling (the practice of removing all trees on a site by dragging an anchor chain or cable across a site) has a detrimental impact on wildlife and should not be considered a thinning (Gallo et al., 2016).

The impact of thinning on coarse woody debris (CWD; an important habitat element for wildlife) depends on the specifics of the site and prescription. Some thinnings increase CWD but others may reduce it (Saud et al., 2018). There are potential tradeoffs between retention of CWD for habitat and fire severity reduction goals. Often these can be moderated by retention of larger pieces of CWD (Evans, 2016). Thinning prescriptions should also address snag retention since they are an important habitat component. In ponderosa pine and mixed-conifer forests of the Southwest, there are fewer snags than is recommended, so snag retention or even creation is an important consideration (Ganey, 1999). Another risk with thinning treatments is the introduction and facilitation of invasive species such as cheatgrass (*Bromus tectorum*) (Havrilla et al., 2017).

In general, thinning should take into account the wider landscape pattern and not just stand-level impacts because effects to wildlife accumulate at the landscape scale (Hutto et al., 1993). There are tradeoffs between increasing habitat heterogeneity where it has been lost (e.g., many mixed conifer forests) and protecting undisturbed habitat for species that prefer interior forest (e.g. red squirrels in spruce-fire forests).

Recommendations

- Treatments that reduce tree densities and surface fuels, especially thinning and burning, effectively reduce the risk of high severity fire.
- Managers should ensure snag and coarse woody debris is retained as a habitat component.
- Thinning prescriptions should take into account the wider landscape pattern and not just stand-level impacts because effects to wildlife accumulate at the landscape scale.
- See the previous section for recommendations related to specific vegetation types and the wildlife section for recommendations related to specific species.

Wildland urban interface (WUI)

The main focus for fuel reduction treatments is protection of lives and property and the people and homes at greatest risk are in the wildland urban interface (WUI) (WFLC, 2014). The WUI is loosely defined as the area where houses or development are in or near wildland vegetation or wildland fuels (Radeloff et al., 2018). As many as 99 million people live in the WUI across 190 million acres of the continental U.S. (Martinuzzi et al., 2015). The WUI continues to grow as more homes are built in areas of high wildfire threat (Radeloff et al., 2018). In New Mexico, homes tend to be built in or near riparian, piñon-juniper, ponderosa pine, and even mixed conifer forests. Treatments in the WUI may use the same tools as thinnings in more remote forests but they place a greater emphasis on ensuring the ability to suppress wildfires and protect homes. This is likely to lead to more dramatic reductions in tree density, removal of surface fuels, thinning of ladder fuels, and even pruning (Cohen, 2000; Evans et al., 2015). Modeled fires show the efficacy of thinning (Ager et al., 2010) and fuel breaks (Bar Massada et al., 2011) in the WUI environment. The Angora Fire of 2007 in El Dorado County, California demonstrated the effectiveness of fuel treatments implemented before the wildfire, which affected fire behavior and helped protect homes (Safford et al., 2009). Similarly, fuel treatments implemented before the 2011 Wallow Fire were able to reduce fire severity (Waltz et al., 2014). Importantly, fuel treatments in the Wallow Fire area gave firefighters opportunities to protect residences during the fire (Bostwick et al., 2011; Kennedy and Johnson, 2014).

The surrounding vegetation, the landscape of forests and grasslands, is important for wildlife in fragmented habitats like the WUI (Watling et al., 2011). However, areas of human development provide vectors and even direct introductions of non-native species (Gavier-Pizarro et al., 2010). Increasing housing density is likely to contribute to the decline of the largest numbers of forest-associated at-risk species (Stein et al., 2010). At the same time, WUI residents may not support timber harvest or other forest management near them (McDonald et al., 2006). Many WUI residents do not support hazard reduction techniques that might protect their homes (Brenkert-Smith et al., 2015; Meldrum et al., 2015). Reasons not to take steps to reduce risk from wildfire, include not wanting to cut trees (a desire to protect amenity values), risk perceptions and knowledge, and economic issues (Collins, 2005). In general, WUI residents are attached to trees and plants around their homes and the wildlife, quiet, privacy, views, and recreational opportunities linked to the wildland (Nelson et al., 2004). It is possible to balance wildfire and risk reduction goals. For example, in piñon-juniper, light thinning followed by surface fuel reductions (e.g., via prescribed fire) and/or crown base height increases (e.g., via pruning) may be best at meeting both fuel reduction and wildlife goals (Coop and Magee, 2016).

Instead of framing the issue in terms of the need to keep fire out of communities, the Cohesive Strategy emphasizes that wildfire is unavoidable and that communities must be fire-adapted (WFLC, 2014). This reframing to acknowledge the coexistence of communities and wildfire is a logical outcome of the expansion of the WUI. The fire-adapted communities concept integrates ongoing efforts to mitigate wildfire hazard in the WUI and acknowledges fire as part of the natural landscape (FAC, 2015). The fire-adapted communities concept may also provide a way for people to acknowledge and coexist with wildlife in the WUI.

Restoration

Restoration focuses on returning ecosystems or habitats to their historic structure and species composition based on the idea that ecosystems are most healthy when they are within the range of conditions to which their component species have adapted (Swanson et al., 1994). Restoration should recognize the inherent variability of ecosystems and be tailored to the specific topographic and vegetation conditions of a project (Floyd and Romme, 2012). Heterogeneity of fire regimes across small spatial scales underscores the need for restoration that is tied to local conditions (Korb et al., 2013). In New Mexico, a set of principles has guided forest restoration in ponderosa pine since 2006 (Bradley, 2009; TNC et al., 2006). These include retention of old or large trees while maintaining structural diversity and resilience. A key element of restoration is the use of reference or benchmark conditions which describe the properly functioning ecosystem (Fulé et al., 1997). Often these reference conditions are framed in terms of the historic range of variability (HRV). Because ecosystems are not static, there is a range of conditions which can still be considered healthy or natural (Morgan et al., 1994). Even as climate change alters basic environmental conditions, ecosystems are more likely to be resilient and resistant when they are within HRV (Fulé, 2008; Keane et al., 2009; Stephens et al., 2010). Though, as the climate warms, shifts in vegetation types may force a reevaluation of restoration goals (Flatley and Fulé, 2016).

Restoration treatments are designed to make a forest stand more resistant and resilient to the types of disturbances that historically shaped the structure and function of the ecosystem. For example, riparian systems restoration may require managed flooding (Molles et al., 1998; Stromberg et al., 2008). Ecological restoration in ponderosa pine forests, on the other hand, generally involves returning low-severity fire to the forest and reducing the probability of high-severity, stand-replacing fires (Hunter et al., 2007). However, not all forest restoration efforts reduce wildfire hazard, e.g., restoration of a spruce-fir forest adapted to a high-severity fire regime would not reduce wildfire hazard (Baker et al., 2007). In another example, fuel treatments in sagebrush can diverge from landscape restoration goals (Jones, 2019).

Since restoration aims to reestablish conditions to which native plants and animals are adapted, it is reasonable to expect positive effects for wildlife. Research from Arizona shows that eight small mammal species benefit from restoration treatments, particularly if habitat elements such as large trees, snags, and woody debris are restored (Kalies et al., 2011). Species in the study were Mogollon vole (*Microtus mogollonensis*), Mexican woodrate (*Neotoma mexicana*), deer mice (*Peromyscus maniculatus),* Abert's squirrel, golden-mantled ground squirrel, rock squirrel (*Otospermophilus variegatus*), gray-collared chipmunk (*Neotamias cinereicollis*), and Botta's pocket gopher (*Thomomys bottae*). Communities of small mammals maintained stability through compensatory dynamics after restoration of ponderosa pine forests (Kalies and Covington, 2012). Restoration treatments have been shown to increase the herbaceous crop (Meddens et al., 2016; Moore et al., 2006) and reintroduction of fire can increase nutrient concentration in forage (Harris and Covington, 1983), both of which can benefit ungulates. Butterflies benefitted from restoration that increased herbaceous cover and common nectar and larval host plants (Kleintjes et al., 2004). Restoration of piñon-juniper stands can increase resilience to wildfire by enhancing levels of understory cover post-burn (Jacobs, 2015).

However, restoration treatments are not universally beneficial for wildlife. A study from California, noted that at the landscape scale forest restoration had little effect on small mammals and songbirds (Stephens et al., 2016b). The effect of forest restoration treatments on Abert's squirrels is uncertain. Some studies show that restoration treatments showed either a positive or unnoticeable effect on home ranges of Abert's squirrels (Yarborough et al., 2015), while others found that certain forest restoration treatments were detrimental to Abert's squirrels at the patch scale and unknown at the landscape scale (Dodd et al., 2006). The preference of Abert's squirrel for dense patches of ponderosa pine with interlocking crowds demonstrates the need for heterogeneity within restoration treatments (Loberger et al., 2011). Other wildlife benefit from heterogeneous stand conditions that reflect the mosaic of disturbances that has shaped forest landscapes through time and space. For example, mule deer benefit from retention of patches of dense bedding and hiding cover in restoration treatments (Germaine et al., 2009). A diverse forest, with patches of various stand densities in proportion with the historical range of variability, accommodates wildlife species with different habitat adaptations (Allen et al., 2002).

Salvage logging

Salvage logging is the practice of harvesting trees after natural disturbances (e.g., wildfires, insect outbreaks, windstorms), typically to recover economic value that may otherwise be lost (Lindenmayer and Noss, 2006). In this assessment, upon referring to salvage logging it is specifically in reference to post-fire logging (though see (Kroll et al., 2012) on salvage after insect outbreaks).

Salvage logging has encountered intense and ongoing academic, political, and public controversy worldwide (Leverkus et al., 2018; Lindenmayer et al., 2017), and particularly within the Western United States (Beschta et al., 2004; DellaSala et al., 2006; Donato et al., 2006). Proponents of salvage logging state that it can diminish the buildup of insect pest populations and reduce fuel loads that therefore decrease the severity of future fires. Whereas opponents argue that it changes plant species composition, reduces plant species richness, causes declines in cavity-nesting bird species' populations, and that the post-fire harvesting operations themselves exacerbate the effects of post-fire erosion (McIver and Starr, 2001). In the southwest, salvage logging is not a frequently utilized treatment due to the relatively low potential timber value of many forestlands. When resilient forest ecosystems are the management goal, the benefits of salvage logging should be carefully evaluated against any negative effects that may occur to early seral forest structure and function, as well as future successional stages (Dunn and Bailey, 2015).

After a wildfire, it is important to retain snags for wildlife (Ritchie et al., 2013). Salvage logging significantly alters post-fire wildlife habitat due to the standard practice of removing the largest diameter snags that have the highest economic value, as well as wildlife value (Chambers et al., 2005; McIver and Starr, 2001). The greatest amount of research has been done regarding the impact to cavity-nesting bird populations. Most cavity-nesting bird populations, such as mountain bluebirds (*Sialia currucoides*), three-toed woodpeckers (*Picoides dorsalis*), and hairy woodpeckers (*Leuconotopicus villosus*), have shown consistent patterns of deceases in abundance and nest densities following salvage logging (Hutto and Gallo, 2006; McIver and Starr, 2001). Conversely, abundances of Lewis' woodpecker increased post salvage logging (McIver and Starr, 2001). If salvage logging is to take place, clumps of straight, large diameter snags that are centrally located in post-fire forests need to be reserved to maintain breeding habitat for many cavity-nesting species (Chambers et al., 2005; Saab et al., 2011, 2009). This would also reduce the probability that insect pest populations may accumulate and infect adjacent green tree stands (McIver and Starr, 2001).

Besides the impact to wildlife habitat, salvage logging can potentially change and exacerbate disturbance dynamics on a post-fire landscape (Leverkus et al., 2018). The construction of road networks can intensify soil erosion, negatively impacting aquatic systems (Karr et al., 2004). Similar to logging in green tree stands, but to a greater degree in post-fire forests, log retrieval systems compact soil and accelerate soil erosion. Impacts are greatest with ground-based skidding, followed by skidding over snow, skyline retrieval, and helicopter retrieval (McIver and Starr, 2001). Within the first few years following salvage logging, there can be a reduction in overall plant species richness as well as vegetation biomass and an increase in exotic plant species introduction (McIver and Starr, 2001). Although 15 years following salvage logging, researchers did not find significant impacts to understory vegetative cover, diversity, or community composition (Peterson and Dodson, 2016).

Post-fire, salvage logging may be seen as a treatment option for reducing fuel loads. Salvage logging has been shown to reduce future surface woody fuels, although the magnitude of this reduction is dependent on a number of factors including: volume and size of wood removed; logging retrieval methods; erosion control; and the amount of coarse woody debris retained for wildlife habitat. Due to these factors, managing for woody fuel reduction may be in competition with other management objectives (Peterson et al., 2015). Additionally, salvage logging does little to change the next stand of trees, in other words live fuel, so the potential for future fires remains (Campbell et al., 2016).

Recommendations:

- The benefits of salvage logging should be carefully evaluated against any negative ecological effects.
- If salvage logging is decided upon as a treatment, harvesting should take place in areas where there is a pre-established road system to minimize soil compaction and erosion
- A harvest retrieval system that causes the least disturbance to soils should be utilized
- Patches of straight, large diameter snags should be reserved in central locations of post-fire forest stands to benefit cavity-nesting birds and reduce the probability of infestation by insect pests.
- See the previous section for recommendations related to specific vegetation types and the wildlife section for recommendations related to specific species.

Lop and scatter

In general, the prescription to ‘lop and scatter’ refers to the process of cutting trees at or near the base, cutting branches off, and then moving the branches away from the stump. Often the prescription will dictate that the resulting slash (small diameter woody material) should not be greater than a specific height (e.g., two feet) above the ground when the work is completed. Since a prescription for lop and scatter can only occur where a thinning is implemented, the effects are intertwined with the effects of thinning.

In piñon-juniper stands, adding slash improves the ecological function of soils, promotes understory establishment, and generally improves degraded conditions (Overby et al., 2000; Stoddard et al., 2008). Lop and scatter treatments have also been shown to reduce erosion (Ashcroft et al., 2017). Any increase in herbaceous production due to lop and scatter treatments would benefit ungulates that rely on this forage. Lop and scatter has been shown to increase grasses, forbs, and shrubs (Brockway et al., 2002). The abundance of woodrats (*Neotoma spp.*) and brush mice (*Peromyscus boylii*) increased in treatment areas with more slash (Severson, 1986). Beyond the Southwest, retention of slash in piles or distributed across the site benefits a range of wildlife including songbirds (Johnson and Landers, 1982), small mammals (Manning and Edge, 2008), even bears (*Ursus americanus*) (Rudis and Tansey, 1995).

Since lop and scatter adds to the surface fuel load it may be a bad prescription in the WUI where an increase in surface fuels can increase risk of wildfire (Farnsworth et al., 2003). Though in a recent study in piñon-juniper, tree boles, branches, and tops scattered after thinning resulted in few changes to surface fuel loading (Huffman et al., 2017).

Lop and scatter should also be evaluated in comparison to potential alterative slash treatments including prescribed burning (addressed in the following section) and mastication. Mastication—the mulching, chipping, or grinding of trees, brush, or slash into small pieces with the intention of providing cover or increasing soil moisture holding capability without inhibiting regeneration—is not a focus for this review. However, it is worth noting that mastication does not remove material, it only rearranges it, and therefore fuel loads after mastication can be very high (Battaglia et al., 2010). The practice can be designed to benefit herbaceous communities (Battaglia et al., 2010) and reduce erosion (Harrison et al., 2016), but if masticated material is too deep there is the potential for reductions in herbaceous species and tree regeneration (Battaglia et al., 2009). Unfortunately, mastication can facilitate the spread and growth of non-native plants such as cheat grass (*Bromus tectorum*) (Coop et al., 2017; Fornwalt et al., 2017). Mastication can increase surface fuel depth and continuity, allowing fires to spread more easily and burn hotter at the soil surface (Harrod et al., 2008; Reiner et al., 2009; Stephens and Moghaddas, 2005a). However, fire behavior in masticated fuels can be unexpected and is an area of active research (Kreye et al., 2014). A useful review of the benefits and cost of mastication is: *To masticate or not: Useful tips for treating forest, woodland, and shrubland vegetation* (Jain et al., 2018).

Recommendations:

- In piñon-juniper stands, lop and scatter should be considered to protect soil, reduce erosion, and promote understory establishment for forage and overall health.
- Lop and scatter may also benefit small mammals and other wildlife
- Care should be taken to balance increased fire risk from lop and scatter treatments with the potential wildlife benefits.
- If possible, lop and scatter should be done outside of migratory bird nesting season.
- See the previous section for recommendations related to specific vegetation types and the wildlife section for recommendations related to specific species.

Prescribed fire

The underlying concept behind the use of prescribed fire (also called controlled burns) is that fire-adapted ecosystems benefit from the reintroduction of fire. Managers implement prescribed fire under specific conditions in order to keep fire severity and intensity within planned parameters. Though fire is a blunt tool, and can burn hotter or kill more vegetation than planned, the majority of controlled burns have impacts similar to pre-settlement fire (Dether and Black, 2006). Since fire is a central process for most vegetation types in New Mexico, wildlife in those habitat areas are adapted to cope with or benefit from the re-introduction of fire (Bock and Block, 2005). As most forests in New Mexico are far denser than they were historically, thinning is usually required before prescribed fire can be utilized as a treatment. Therefore, the wildlife impacts of thinning (as described in the earlier section) are linked to the effects of prescribed fire. Similarly, full restoration includes reintroduction of fire (e.g., Fulé et al., 2002), but not all prescribed fire is restoration. The primary focus of controlled burns can be fuel reduction or wildfire threat reduction.

The synthesis document *Birds and Burns of the Interior West* provides an excellent overview of prescribed fire effects on habitats and populations of birds (Saab et al., 2007). This review noted that birds showed a greater response during the year of the controlled burn than was shown after one year, which highlights the potential for short-lived effects. Resident birds showed more positive responses than migratory birds (Saab et al., 2007). The avian community had high similarity before and after the 2000 Cerro Grande Fire, except in high-severity areas (Kotliar et al., 2007) Since prescribed burns in frequent fire forest types are planned to be low to medium severity this suggests minimal impact on bird communities. Prescribed fire can improve forest habitat for bats by opening canopies, mid-stories, encouraging healthy herbaceous layers to promote insect populations (Boyles and Aubrey, 2006).

As described in the section on thinning, an extensive meta-analysis found:

- small mammals had no response to thinning and burning;
- ground-foraging birds and rodents did not have a strong response to thinning and burning; and
- Canopy-foraging birds had positive or neutral responses to thinning and burning (Kalies et al., 2010).

Fire return intervals and patches can be managed to benefit small mammals (Kelly et al., 2012). Successive burns, multiple years in a row, designed to reduce meadow encroachment may have negative impacts on small mammals (Trottier et al., 1989).

Prescribed fire can encourage healthy grass and increase nutrient concentration in forage (Harris and Covington, 1983; Hunter et al., 2007). When the goal is to increase forage for ungulates in fire adapted ecosystems, prescribed fire in late winter or early spring can increase availability of key nutrients (Bender, 2018). Even repeated controlled burns can benefit plant diversity (Webster and Halpern, 2010). However, where the understory community is sparse with little perennial grass cover, prescribed fire may lead to reduced herbaceous cover (Overby et al., 2000). In Great Basin piñon-juniper woodlands, mule deer use was greater in burned areas than in unburned areas (Stager and Klebenow, 1987). Initial use focused on the edges of the burned area but then expanded to include the entire burned area, this trend continued for decades after the fire (Stager and Klebenow, 1987). Prescribed fire can also increase aspen, an important food source for elk (Higgins et al., 2015).

Prescribed fire has a mixed impact on coarse woody debris and snag densities, both of which are important habitat elements. For example, coarse woody debris helped American toads (*Anaxyrus americanus*) survive prescribed fire in southern Appalachian hardwood forests (Pitt et al., 2013). Depending on the severity and initial conditions of the stand, prescribed fire can reduce the number of snags and the amount of coarse woody debris (Saud et al., 2018; Stephens and Moghaddas, 2005b). Controlled burns can also kill trees, create snags, and increase coarse woody debris (Saab et al., 2007). Because of the mixed effects of prescribed fire, a focus on the retention of snags and coarse woody debris can help maintain wildlife habitat, particularly when fire is first reintroduced into fire-suppressed landscapes (Bagne et al., 2008).

The concept of pryodiversity (diversity of fire severity, season, size, and time since fire) adds important nuance to the interactions of fire and wildlife. Recent research suggests that pryodiversity can foster species diversity, though the interactions can be complex (Kelly et al., 2017). A study in the mixed conifer forests of California's Sierra Nevada showed that fires with a greater heterogeneity of burn severities supported more bird species (Tingley et al., 2016). Retention of unburned patches may support avian diversity (Prowse et al., 2017). Importantly, avian communities diverged over time so the communities in different burn severities became increasingly different during the decade after fire (Tingley et al., 2016). It is important to note that current monitoring of the effects of fire on wildlife may provide insufficient guidance to managers regarding the characteristics of desirable 'mosaics' (e.g. patch size, connectivity or composition of age-since-burnt classes) (Clarke, 2008).

Recommendations:

- Prescribed burns should be scheduled to minimize adverse effects on wildlife, including migratory bird nesting season.
- Attention should be given to retaining and creating snags and coarse woody debris for wildlife habitat.
- See the previous section for recommendations related to specific vegetation types and the wildlife section for recommendations related to specific species.

Wildlife

Each species experiences the effects of treatment differently and many move easily across vegetation types, so a species or group perspective is important. Much of the following information about the impacts of treatment on wildlife is a reiteration of information presented in the previous sections on vegetation type and treatment type. Though, managers interested in impacts on particular forest types or the effects of specific treatments should review those sections as well. More research is needed to understand wildlife response to treatments, as many studies are short term and have not been sufficiently replicated (Clarke, 2008). Moreover, studies in one area are not necessarily transferable to other similar habitats (Johnson and Sadoti, 2019).

Birds

Migratory songbirds

Migratory songbirds are highly mobile and can quickly shift between vegetation types or locations, migrations and life cycles are built on a delicate balance of resources (Stenseth et al., 2015; Visser et al., 2004). Migratory songbirds rely on both montane and riparian habitats in New Mexico (DeLong et al., 2005). There are two guides that provide an overview of birds in New Mexico relative to forest and woodland habitat types:

- *Songbird ecology in southwestern ponderosa pine forests: A literature review* (Block and Finch, 1997).
- *Sharing the land with piñyon-juniper birds* (Gillihan, 2006)

Thinning can often enhance migratory bird habitat. For example, after a fuel reduction treatment in ponderosa pine outside of Santa Fe, New Mexico, the majority of birds, including those identified as of regional concern, had a positive response to thinning (Bagne and Finch, 2008). In Southwestern conifer forests ground-foraging birds did not have a strong response to small-diameter thinning and burning; canopy-foraging birds had positive or neutral responses to small-diameter thinning and burning; and canopy-foraging birds had negative responses to overstory removal (where ≥80% of basal area was removed) and wildfire (Kalies et al., 2010). It's important to recognize that thinning differs from timber harvesting. Thinning reduces fuels quantities and increases habitat heterogeneity, resulting in less impacts on bird communities than commercially oriented timber harvesting.

In piñon-juniper, studies in Arizona and New Mexico have not detected a modified pattern of songbird use in treated compared to untreated plots (S. K. Albert et al., 1994). Bird communities in central western New Mexico and eastern Arizona were highly stable after small reductions in woodland cover (Knick et al., 2017). More drastic treatments have magnified effects. Birds of dense woodland and open woodland habitats used areas cleared of piñon and juniper trees significantly less than more dense untreated areas. Additionally, no grassland or shrubland obligate bird species used the treated areas (Bombaci et al., 2017). Even without any direct human treatment, climate and weather-induced declines in piñon-juniper woodlands can negatively affect birds. Piñon mortality threatens bird communities and diversity and abundance declined significantly in north-central New Mexico after the 2002 piñon bark beetle outbreak (Fair et al., 2018). Bird diversity declined on thinned and unthinned sites after the bark beetle outbreak, though abundance and species richness declined faster in thinned sites (Fair et al., 2018). At the same time, evidence from California suggests that at the landscape scale, efforts to move forests closer to historic conditions has little effect on songbirds (Stephens et al., 2014). Avoiding treatment during the nesting season is likely to benefit songbirds like gray vireo (which nests late April through August) (Stake and Garber, 2008), and is consistent with New Mexico Department of Game and Fish recommendations, federal Executive Orders, and agreements between U.S. Fish and Wildlife Service and resource management agencies to reduce project impacts on nesting birds (for example see). Interpretation and implementation of rules and regulations is not static. For example, the 2017 Memorandum 37050 from the Department of the Interior emphases that Migratory Bird Treaty Act does not prohibit incidental take.

Diversity of fire severity, season, size, and time since fire supports diversity of bird communities (Tingley et al., 2016) (see also Prescribed). Some species may require the structures and conditions that result from high severity fires (Hutto and Patterson, 2016). After prescribed fire in piñon-juniper researchers found only slight changes in bird communities, which appeared to be responding to the post-fire vegetation rather than relative amount of change from pre-treatment (Knick et al., 2014).

High priority, rapidly declining species such as the pinyon jay (*Gymnorhinus cyanocephalus*) may require species-specific mitigation actions. The pinyon jay is one of the landbird declining most rapidly (average rate of –3.6% per year from 1968 to 2015) (Balda, 2002; Boone et al., 2018). Widespread thinning of piñon and juniper trees in nesting areas may negatively affect the pinyon jay (Johnson et al., 2018a), more research is needed to clarify the effects of varying degrees of treatments on pinyon jays (Boone et al., 2018). Even where some pinyon jays avoided treatment areas, they nested adjacent to the treatment (Johnson et al., 2018a). Declines in piñon health affected pinyon jay nesting preferences (Johnson et al., 2017). Based on surveys, post-treatment assessment, and past research Johnson et al. (2018b) offer a number of management recommendations for reducing treatment effects to pinyon jays, including:

1. Carefully identify nesting habitat before treatment;
2. Avoid treatments within and surrounding (600m) nesting colonies; and
3. Leave untreated areas of at least 50 ha within 1 km, to provide alternative colony sites.

Upland Game Birds

Research on upland game birds in New Mexico has focused primarily on, turkey (*Meleagris gallopavo*). There is less information on other species such as the Dusky grouse (*Dendragapus obscurus*) (2018b). Guides to management of turkey include:

- *Management guidelines for Merriam's wild turkeys* (Hoffman et al., 1993)
- *Gould's Wild Turkey (Meleagris gallopavo mexicana) Recovery Plan* (Cardinal and Bulger, 2017)

Wild turkey (*Meleagris gallopavo*) is another example of a bird that declines after piñon-juniper trees are cleared. Removal of trees isolated roost sites and caused a 64% reduction in turkey populations (Scott and Boeker, 1977). Researchers recommend that cleared areas in piñon-juniper woodlands should not be wider than 90 m and strips of cover should be retained as travel lanes to established roost areas (Scott and Boeker, 1977). Another recommendation is to use spring prescribed fires to thin and invigorate dense, decadent stands of brushy manzanita and

Toumey oaks (*Quercus toumeyi*) (Hoffman et al., 1993). In ponderosa pine, turkeys prefer the types of large trees that thinning and restoration foster (Boeker and Scott, 1969). Turkeys feed on Gambel oak acorns and so benefit from having oaks as a component of ponderosa pine forests (Ffolliott, 1997).

Gould’s turkey is a threatened species in New Mexico because of population declines driven by wildfire, lack of water sources, overgrazing, hybridization, habitat loss, and poaching (Cardinal and Bulger, 2017; York and Schemnitz, 2003). Gould's turkey prefer piñon-juniper woodlands and are likely to benefit from treatments that increase abundance of food sources such as piñyon ricegrass (*Piptochaetium fimbriatum*) (York and Schemnitz, 2003).

Large Game Animals

Cover and forage are primary features of native ungulate habitat. Cover provides ungulates with protection from predators, concealment for reproductive activities, and shelter from extreme heat or sunshine in the summer or heat loss in the winter (Bender et al., 2009). Forage is essential to ungulate habitat because it provides a food resource throughout the year, often in the form of grass and leaves. Both cover and forage depend upon the structure and function of vegetation communities.

This review does not cover livestock grazing which can have significant implications for site ecology and wildlife, but there is a significant body of research on the topic (e.g., Bakker et al., 2010). The topic of interactions and competition between native ungulates and cattle is a contentious issue across the western U.S. (Halbritter and Bender, 2015).

Mule deer

Mule deer (*Odocoileus hemionus*) have large home ranges which are likely to include multiple vegetation types. They require forage and browse for nutrition and cover to survive and reproduce. Guides to management of deer include:

- *Guidelines for Management of Habitat for Mule Deer* (Bender, 2012)
- *Habitat Guidelines for Mule Deer: Southwest Deserts Ecoregion* (Heffelfinger et al., 2006)

Thinning appears to increase habitat for deer, both mule deer and white-tail deer, as well as other ungulates. For example, in piñon-juniper forests in Arizona and New Mexico, deer use increased proportionally to the amount of trees removed, down to a basal area density of 7 m^2/ha, in areas where human presence was reduced or excluded(S. K. Albert et al., 1994). Thinning has been shown to increase forage, and the combination of thinning or mastication with seeding is particularly effective for increasing native annual forb biomass (Klebenow, 1965; Stephens et al., 2016a). In mixed-conifer forests, researchers have proposed reducing stand densities and re-opening meadows to increase distribution, quantity, and quality of forage (Halbritter and Bender, 2015). In northeastern New Mexico, thinning conifers via hydraulic mulching or selective cutting resulted in an immediate increase in grass coverage, especially when combined with the use of exclosures (Bender et al., 2013; Kramer et al., 2015). Mule deer use forage plants in their winter range with the greatest density of trees less than those on the more open portions of the range (Klebenow, 1965).

Ungulate exclosures provide an opportunity for herbaceous cover to establish and have been shown to provide eight percent more coverage than areas without exclosures, though drought may decrease these positive increases over time (Kramer et al., 2015). A study of mule deer responses to thinning connected to oil and gas development showed that mule deer have the capacity to shift habitat use in response to environmental modifications, such as decreases in stand density or exclosures (Van Dyke et al., 2012). Prescribed fire is another option for increasing forage for ungulates in fire adapted ecosystems, particularly in late winter or early spring to increase availability of key nutrients (Bender, 2018). Managing forage and habitat availability appears to have the potential to affect the type of response and the degree of habituation by mule deer. In addition to availability of forage, the amount and quality of cover greatly influences mule deer responses to vegetation treatments (Bombaci and Pejchar, 2016).

While ungulates benefit from open stands or forest gaps for forage, denser stands provide protection and thermal cover (Ffolliott, 1997; Hunter et al., 2007). For example, depending on their condition, piñon-juniper forest types may provide little forage for ungulates such as mule deer while still potentially providing important cover (Bender et al., 2009). It should be noted that clearcutting of large blocks which provide winter range can have detrimental impacts on ungulates such as deer (Freedman and Habeck, 1984). A review of studies from the Western US regarding clearing piñon-juniper woodlands found elk and mule deer had non-significant or negative responses to tree removal (Bombaci and Pejchar, 2016).

Efforts to restore mule deer habitat may be most effective when focused on piñon-juniper communities, which provide the most immediate gains in mule deer habitat in north-central New Mexico by allowing increases to forage quantity and quality while maintaining cover attributes (Bender et al., 2009). Grasslands may also provide some gains in mule deer habitat, but these may be more difficult to realize because management must establish both cover and forage in these systems (Bender et al., 2009).

Recommendations

- Reduce stand densities to encourage forage production (moving toward historic range of tree densities)
- Maintain patches of high tree densities for cover
- See the previous sections for recommendations related to specific vegetation types and treatments.

Elk

Elk and mule deer forage requirements overlap, though deer preferences are more limited than elk. To the extent their preferences are similar, the treatment effects discussed previously for mule deer are relevant for elk. Both elk and mule deer are negatively affected by large-scale tree removal in piñon-juniper habitats (Short et al., 1977). Though since elk are more comfortable in large open areas, they may be less affected than deer. Additionally, elk and mule deer utilize ponderosa pine forests that have been logged because of an increase in forage (Patton, 1969). Similarly, both benefit from thinning that increases forage while retaining some cover (Ffolliott, 1997; Horncastle et al., 2013).

Wildfire in mixed conifer forests can benefit elk by providing young, healthy aspen stands. In fact, elk choose burned areas because of the availability of aspen (Halbritter and Bender, 2011). Unlike some other animals covered in this review, elk appear to benefit less from dead trees and downed logs as when they occur at high densities, such as within beetle killed forests, they impede movement and increase energy expenditures (Lamont et al., 2019). While outside the scope of this review, the predator-prey relationship between wolves (*Canis lupus*) and elk can be essential to elk management. In New Mexico, it was previously noted that the density of wolves is too low to have generated a documented effect on elk population sizes (Beschta and Ripple, 2010), but New Mexico Department of Game and Fish have an ongoing research project to determine potential effects of Mexican wolves on Gila elk populations. Another relationship outside the scope of this review that may increase over time is between elk and people. Managing elk in the WUI presents unique challenges (Lee and Miller, 2003).

Bighorn sheep

Bighorn sheep (*Ovis canadensis*) are strongly tied to open habitats as they facilitate predator detection and the visual communication of alarm postures (Smith et al., 1999). Due to this, vegetation management can have substantial impacts on bighorn sheep. Serious declines in bighorn populations have been attributed to vegetation changes in some areas due to fire suppression, human encroachment, and potentially other factors (Krausman et al., 1995). In northeastern Utah, areas treated with clear-cut logging or prescribed burning saw increased bighorn sheep activity compared to untreated areas where activity decreased. A more favorable response was elicited in areas that had been treated with clear-cut logging than those treated with prescribed burning (Smith et al., 1999). An additional study that looked at prescribed fire found that prescribed fire can increase forage and reduce lungworm infections, but can also have negative impacts (Peek et al., 1984).

Small mammals

In general, small mammals benefit from increases in herbaceous cover and diversity, which thinning produces in unnaturally dense forests and woodlands. Small mammal abundance generally increases after thinning in piñon-juniper (Willis and Miller, 1997), though in one study pinon mouse density decreased (S. K. Albert et al., 1994). Small mammals had no response to small-diameter thinning and burning in Southwestern conifer forests in Colorado, although they had a negative response to overstory removal (≥80% basal area removal) and wildfire (Kalies et al., 2010). Even drastic treatments such as chaining of piñon-juniper sites have mixed results for small mammals. While habitat use by golden-mantled ground squirrels and rock squirrels declined on a chained site, chipmunk and mountain cottontail did not (Gallo et al., 2016). Another review highlighted that no small mammal species responded positively to extensive piñon and juniper removal in the Piceance Basin of Colorado (Bombaci et al., 2017). As noted above in the section on lop and scatter treatments, small mammals often benefit from an increase in coarse and fine woody material in forests and woodlands (Severson, 1986).

Restoration in ponderosa benefits small mammals, particularly if key habitat elements (e.g., large trees, snags, woody debris) are protected, enhanced or restored (Kalies et al., 2011). Still, restoration is not universally beneficial for small mammals. Some studies demonstrated that restoration treatments caused either a positive or unnoticeable effect on home ranges of Abert's squirrels (Yarborough et al., 2015), while others found that certain forest restoration treatments were detrimental to Abert's squirrels at the patch scale and unknown at the landscape scale (Dodd et al., 2006).

Research in piñon-juniper suggests similar results: habitat for cottontails in southern New Mexico can be maintained or enhanced during piñon-juniper woodland treatments by preserving some combination of 70-90 down, dead trees and living shrubs per acre (Kundaeli and Reynolds, 1972). Thinning in riparian habitats can benefit small mammals as well (Bagne and Finch, 2010). In higher elevation spruce-fir forests, small mammals are less likely to benefit from thinning. For example, red squirrels (*Tamiasciurus hudsonicus*) tend to be found in areas with canopy cover, high tree densities, and large amounts of dead wood (Doumas et al., 2015). Thinning would reduce these habitat elements and would be likely to negatively affect red squirrel populations (Doumas et al., 2015) .

Carnivores

The home ranges for carnivores are larger than those of small mammals, so the potential impacts of thinning and other treatments are likely to be less pronounced–or at least hard to measure. Of course, the health of carnivore populations depends on the health of prey populations. For example, declines in mule deer populations in Arizona resulted in cougar (*Puma concolor*) predation of bighorn sheep (Kamler et al., 2002).

There are direct effects of treatments on carnivores such as complete removal of piñon and juniper trees which removes important habitat elements for bobcat (*Lynx rufus*), mountain lion (*Puma concolor*), and black bear (Gallo et al., 2016). Cougars in Oregon used riparian forests for travel corridors and showed a preference for proximity to timber harvest sites (Gagliuso, 1991). Gray fox (*Urocyon cinereoargenteus*) and coyotes (*Canis latrans*) showed resilience to a high severity wildfire in central Arizona, which suggests they would also be resilient to the less severe impacts from a prescribed fire (Cunningham et al., 2006).

Black Bear

Of carnivores, black bears are most likely to be adversely affected by treatments due to their use of downed wood and dense thickets of shrubs and smaller trees adjacent to or within mature forests for cover, den sites, and forage (Pilliod et al., 2006). Bears, like other carnivores, use coarse woody debris, large logs, and snags for dens (Rudis and Tansey, 1995), so removing these habitat elements has a negative effect on their population. A study done in central Arizona 1 year after a high-severity burn, showed that black bears had similar population densities pre- and post-fire in both the burned and control areas, but reproductive output was lower within the burned areas suggesting the importance of leaving a mosaic in fire management plans (Cunningham et al., 2003). Black bears showed preference to thinned stands compared to unthinned stands for feeding on sapwood in Montana, likely due to the health and condition of the residual trees in the stand (Pilliod et al., 2006). Yet, black bears typically utilize stands with high stem densities and canopy cover while travelling or resting, presumably for security, and a study in central Arizona found that only 12% of black bear bedding sites were in areas that bad been thinned or selectively logged within the past 20 years (Pilliod et al., 2006). Bears also benefit from mast producing trees like oaks and junipers (Costello et al., 2001), so retaining larger, older mast producing trees would benefit bears and carnivore prey species.

Other Wildlife

There is a wide array of other animals not covered previously in this review. In many cases there is much less information on the effects of treatments on amphibians, reptiles, insects, and other organisms. A review of treatment impacts in piñon-juniper woodlands highlighted that reptiles and the impact of treatments are consistently under-studied in these ecosystems (Bombaci and Pejchar, 2016).

Bats

Forests and woodlands are important for almost all bats species. Their basic habitat requirements are similar across ecosystems: roosting habitat, foraging opportunities, and water sources (Taylor, 2006). Common bats species in ponderosa pine forests include *Myotis volans, Eptesicus fuscus*, *M. evotis*, and *M. occultus* (Morrell et al., 1999). Bats prefer open forest stands and dead trees are particularly important for roosting (Campbell et al., 1996). In general bat foraging activity is concentrated in riparian areas and within the gaps of older, more-diverse forests (Taylor, 2006). Hence thinning or other treatments that create small openings may benefit bats (Taylor, 2006). For example, prescribed fire can open gaps, reduce tree density, promote insect habitat, and create snags for roosting (Boyles and Aubrey, 2006). Bats have been shown to respond positively to burned patches in the Sierra Nevadas in California (Buchalski et al., 2013). Intensive forest management practices in ponderosa pine habitats during summer months may negatively influence bat reproductive success (Morrell et al., 1999).

Piñon-juniper woodlands provide habitat for bats such as western long-eared bat (*Myotis evotis*) (Snider et al., 2013). In fact, one study found bats were more reproductively active in piñon-juniper woodlands than ponderosa pine forest, highlighting the important of woodlands for bats (Chung-MacCoubrey, 2005).

Amphibians

There is relatively little information on the effects of treatments on amphibians. In New Mexico, research has been done that documents Sacramento Mountain salamander (*Aneides hardii*) populations decline where logging activities have occurred. Populations continue to persist, but at lower numbers than in unlogged areas (Borg, 2001; Ramotnik, 1997). Presumably, forest thinning treatments could have similar adverse effects when the forest canopy is opened, causing increased soil temperatures and decreased soil moisture content, but the potential reduction of high severity fire may be beneficial to Sacramento Mountain salamanders long term. In southwestern Washington, forest management had variable impacts on salamander populations; some species declined in abundance after clearcutting and thinning while other did not (Grialou et al., 2000). As noted previously, riparian corridors are important for dispersal and genetic connectivity of amphibians such as the Rocky Mountain tailed frog (Spear and Storfer, 2010). A review of fire effects on amphibians emphasizes that impacts are spatially and temporally variable and incompletely understood (Pilliod et al., 2003). A study of toads in southern Appalachian forests found prescribed fire did not cause direct mortality in part because downed logs provided refugia (Pitt et al., 2013).

Recommendations:

- Treatment activities in Sacramento Mountain salamander occupied habitat to reduce potential for high severity fire should occur outside of their activity season (June-October).
- Decaying logs and coarse woody debris should be maintained at maximum levels possible to provide microhabitat cover
- When thinning, clumps of trees should be retained to maintain canopy cover that conserves lower soil moisture and temperatures as well as higher soil and coarse woody debris moisture content.
- See the previous sections for recommendations related to specific vegetation types and treatments.

Fish

This review does not focus on fish, but it is worth pointing to existing reviews that connect forest management and fish. For example, a 2003 review highlighted the importance of effective pre-fire management activities to address factors that may render fish populations more vulnerable to the effects of fire (e.g. habitat degradation, fragmentation, and nonnative species) (Dunham et al., 2003).

Insects

To truly capture the effects of treatment on insects would require a review of a wide diversity of insect groups and habitats, for which there is currently insufficient data. The New Mexico Department of Game and Fish has not authority over the management of insects, therefore it’s priority and resources for addressing insect response to forest management treatments is extremely limited, at best. Additionally, the diversity of insects makes general statements of impacts difficult. For example, a detailed study of the effects of thinning and burning in Sierra Nevada mixed conifer forests found some groups such as beetles, ants, and spiders showed changes in abundance due to the treatments, but the changes were taxon-specific and showed no general pattern (Apigian et al., 2006). While management decisions that retain a diversity of habitat element such as downed logs and coarse woody material may support insect abundance and diversity, the evidence presently available is insufficient to determine the utility of any specific intervention (Davies et al., 2008). One rare example of research on the impact of treatments on insects indicates that thinning and slash reduction in piñon-juniper increased herbaceous cover and had a positive, initial response by butterflies (Kleintjes et al., 2004). While data related to insect responses to forest management treatments are currently insufficient, it is important to consider that insect populations are necessary to support many bird species, either when rearing young or throughout their entire life cycle. Any impacts to insect populations could have impacts on bird populations as well.

Conclusion

It is difficult to arrive at useful generalities about the effects of vegetation treatments on wildlife in New Mexico. The differences in ecosystems, organisms, and treatment prescriptions overwhelm most simplifications. The conclusions that do cut across categories focus on process and approach. Restoring conditions that are similar to those that existed pre-settlement provides benefits to native wildlife. This includes restoring important processes such as fire and flooding that were the major influence on forest and woodland habitats pre-settlement. Careful consideration of an ecosystem's specific conditions and the specific objectives of a treatment are important regardless of where one is working. For example, clarity about the type of piñon-juniper stand or whether it is a xeric or mesic mixed conifer stand is essential to adequately restore these systems. Similarly, knowing the nesting season bird species is important for reducing negative impacts on wildlife. The importance of heterogeneity also seems to cut across vegetation types and treatments. Retention of patches of dense trees or habitat elements like large downed logs fosters diverse wildlife communities. Another commonality is the need for additional research. Long-term studies and monitoring of the impacts of treatments on wildlife are limited and managers need more information to help inform their work, particularly with regard to climate change.

Vegetation types

The first step in management of areas dominated by piñon-juniper is to understand the type of piñon-juniper system they are working in using habitat guides (e.g., Romme et al., 2007). Other recommendations for piñon-juniper include:

- Thinnings that return woodlands to tree densities within the natural range of variation may benefit wildlife, particularly small mammals and ungulates.
- Care should be taken to avoid negative impacts to nesting bird species by cutting outside of nesting season for piñon-juniper obligate bird species.
- Maintain patches with high trees densities and heterogeneity at the landscape scale.
- Chaining or other methods of clearing piñon-juniper stands should be avoided.
- Retain the most productive piñon nut crops trees and the oldest/largest/densest canopy piñons to maximize piñon nut mast crops for wildlife, maintain higher basal area trees selected by pinyon jays for nest trees and provide nest cover from predation.
- Identify and retain roost trees for bat species.
- The benefits of treatments for grazing, sage-grouse habitat, and wildfire risk reduction should be carefully weighed against the potential to decrease migratory bird, small mammal, and bat habitat in piñon-juniper woodlands

In ponderosa pine forests, restoration of structure and fire as a process benefits most native species. However, wildlife does not benefit from implementing the same prescription everywhere. Treatments should encourage a diversity of structure and stand development phases from forest openings to areas with dense cover. Other recommendations include:

- Retaining heterogeneity including dense patches of trees, patches of oaks, and other elements is important for wildlife.
- Protect or increase the number of snags and downed logs, while balancing the hazard of wildfire.
- Gambel oaks in the 12 to 14-inch diameter range and poles in the 3- to 6-inch diameter range should be retained for game and non-game bird species habitat.

As with piñon-juniper, managers should work to understand the type of mixed conifer forest they are working in and its place on the landscape. Other considerations are similar to those for ponderosa pine forests:

- Treatments should foster heterogeneity at the landscape scale and recognize the impact the warming climate is having on distribution of the xeric and mesic types
- Treatments should protect old trees, reduce the likelihood of high severity wildfires, and restore fire regimes.

Though the warming climate is stressing spruce-fir forests, there is little management can do to benefit wildlife in this vegetation type.

In riparian systems, management needs to address both native and non-native species:

- Restore native riparian plants following non-native removal or biocontrol and maintain an adequate water supply for native plants.
- Incorporate native drought-tolerant woody and herbaceous species in restoration plantings to address expected changes in climate and low water availability.
- Consider implementing streambank and floodplain modifications (e.g., bank softening, bank lowering) following non-native removal to ensure maintenance of overbank flows, river-floodplain connections, and native plant communities.
- Stage and balance salt cedar removal and native habitat restoration over time to avoid rapid loss of non-native woody riparian habitats for wildlife until alternative native habitats can be developed.
- Protect and sustain existing stands of native riparian vegetation that may serve as important refugia in areas currently or likely to be affected by non-native control efforts.
- At sites where non-native trees are removed from the understory of mature riparian forests, consider planting native trees, shrubs, and herbaceous plants to maintain vertical habitat diversity and ground cover.

Treatments

Acknowledging the important specifics of prescriptions, forest types, and wildlife species, it is also useful to highlight some general guidance to help ensure treatments benefit wildlife:

- Treatments that reduce tree densities and surface fuels, especially thinning and burning, effectively reduce the risk of high severity fire.
- Restoration treatments are designed to make a forest stand more resistant and resilient to the types of disturbances that historically shaped the structure and function of the ecosystem and hence are often beneficial for native wildlife.
- Treatments near homes and communities may have to place a greater emphasis on the ability to suppress wildfires than foster wildlife habitat.
- Managers should ensure snag and coarse woody debris is retained as a habitat component.
- Patches of untreated or unburned forest should be retained to provide heterogeneity and habitat.

- Where dense patches or surface fuels are retained on site, such as in a lop and scatter treatments, the increased fire risk must be balanced with the potential wildlife benefits.
- Treatments should be scheduled to minimize adverse effects on wildlife, including migratory bird nesting season.
- Every effort should be made not to introduce non-native species (e.g., cheat grass) when treatments are implemented.
- Treatment prescriptions should take into account the wider landscape pattern and not just stand-level impacts because effects to wildlife accumulate at the landscape scale.

Specific treatments also have elements that may require particular attention. For example, salvage logging requires added focus on erosion because post-fire soils are prone to erosion. Other recommendations for salvage include:

- The benefits of salvage logging should be carefully evaluated against any negative ecological effects.
- If salvage logging is decided upon as a treatment, harvesting should take place in areas where there is a pre-established road system to minimize soil compaction and erosion
- A harvest retrieval system that causes the least disturbance to soils should be utilized

These recommendations are just a starting point for management of New Mexico's forests and woodlands. Stewardship of forests and wildlife requires careful assessment of the specific site and consideration of the impacts of a treatment. Ideally, this report will serve as an introduction to management challenges and a gateway to other useful research.

References

Note: the references in this document are also available in the online database: https://www.mendeley.com/library/community/nmdgf_treatment_effect

Addington, R.N., Aplet, G.H., Battaglia, M.A., Briggs, J.S., Brown, P.M., Cheng, A.S., Dickinson, Y., Feinstein, J.A., Pelz, K.A., Regan, C.M., Thinnes, J., Truex, R., Fornwalt, P.J., Gannon, B., Julian, C.W., Underhill, J.L., Wolk, B., 2018. Principles and Practices for the Restoration of Ponderosa Pine and Dry Mixed-Conifer Forests of the Colorado Front Range. Fort Collins, CO.

Ager, A.A., Vaillant, N.M., Finney, M.A., 2010. A comparison of landscape fuel treatment strategies to mitigate wildland fire risk in the urban interface and preserve old forest structure. For. Ecol. Manage. 259, 1556–1570.

Albert, Luna, N., Chopito, A.L., 1994. Deer, small mammal, and songbird use of thinned Piñón-Juniper plots: preliminary results, in: Desired Future Conditions for Piñon-Juniper Ecosystems. GTR_RM-258. USDA Forest Service, Flagstaff, AZ, pp. 54–64.

Albert, S.K., Luna, N., Chopito, A.L., 1994. Deer, small mammal, and songbird use of thinned piñon-juniper plots: preliminary results, in: Shaw, D., Aldon, E., LoSapio, C. (Eds.), Desired Future Conditions for Piñon-Juniper Ecosystems. Citeseer, Fort Collins, CO, pp. 54–64.

Alexander, R.R., 1987. Ecology, silviculture, and management of the Engelmann spruce-subalpine fir type in the central and southern Rocky Mountains. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, Ft. Collins, CO.

Allen, C.D., Savag, M., Falk, D.A., Suckling, K.F., Swetnam, T.W., Schulke, T., Stacey, P.B., Morgan, P., Hoffman, M., Klingel, J.T., 2002. Ecological Restoration of Southwestern Ponderosa Pine Ecosystems: A Broad Perspective. Ecol. Appl. 12, 1418–1433. https://doi.org/10.2307/3099981

An, H., Gan, J., Cho, S., 2015. Assessing Climate Change Impacts on Wildfire Risk in the United States. Forests 6, 3197.

Anderson, M., 2002. Pinus edulis. In: Fire Effects Information System [WWW Document]. U.S. Dep. Agric. For. Serv. Rocky Mt. Res. Stn. URL https://www.fs.fed.us/database/feis/plants/tree/pinedu/all.html (accessed 4.29.19).

Andrus, R.A., Veblen, T.T., Harvey, B.J., Hart, S.J., 2016. Fire severity unaffected by spruce beetle outbreak in spruce-fir forests in southwestern Colorado. Ecol. Appl. 26, 700–711. https://doi.org/10.1890/15-1121/suppinfo

Apigian, K.O., Dahlsten, D.L., Stephens, S.L., 2006. Fire and fire surrogate treatment effects on leaf litter arthropods in a western Sierra Nevada mixed-conifer forest. For. Ecol. Manage. 221, 110–122.

Apple Snider, E., Cryan, P., Wilson, K., 2013. Roost selection by western long-eared myotis (Myotis evotis) in burned and unburned piñon—juniper woodlands of southwestern Colorado, Journal of Mammalogy. https://doi.org/10.2307/23487451

Arkle, R.S., Pilliod, D.S., 2010. Prescribed fires as ecological surrogates for wildfires: A stream and riparian perspective. For. Ecol. Manage. 259, 893–903.

Ashcroft, N.K., Fernald, A.G., VanLeeuwen, D.M., Baker, T.T., Cibils, A.F., Boren, J.C., 2017. The effects of thinning trees and scattering slash on runoff and sediment yield within dense piñon-juniper woodlands in New Mexico, United States. J. Soil Water Conserv. 72, 122–130. https://doi.org/10.2489/jswc.72.2.122

Bagne, K.E., Finch, D.M., 2010. Response of Small Mammal Populations to Fuel Treatment and Precipitation in a Ponderosa Pine Forest, New Mexico. Restor. Ecol. 18, 409–417. https://doi.org/10.1111/j.1526-100X.2009.00601.x

Bagne, K.E., Finch, D.M., 2008. Small-scale response in an avian community to a large-scale thinning project in the southwestern United States, in: Rich, T. D.; Arizmendi, C.; Demarest, D.; Thompson, C. (Ed.), Tundra to Tropics: Connecting Birds, Habitats and People. USDA Forest Service, McAllen, TX, pp. 669–678.

Bagne, K.E., Purcell, K.L., Rotenberry, J.T., 2008. Prescribed fire, snag population dynamics, and avian nest site selection. For. Ecol. Manage. 255, 99–105. https://doi.org/http://dx.doi.org/10.1016/j.foreco.2007.08.024

Baker, W.L., Shinneman, D.J., 2004. Fire and restoration of piñon-juniper woodlands in the western United States: a review. For. Ecol. Manage. 189, 1–21.

Baker, W.L., Veblen, T.T., Sherriff, R.L., 2007. Fire, fuels and restoration of ponderosa pine–Douglas fir forests in the Rocky Mountains, USA. J. Biogeogr. 34, 251–269. https://doi.org/10.1111/j.1365-2699.2006.01592.x

Bakker, J.D., Rudebusch, F., Moore, M.M., 2010. Effects of Long-Term Livestock Grazing and Habitat on Understory Vegetation. West. North Am. Nat. 70, 334–344. https://doi.org/10.3398/064.070.0306

Balda, R.P., 2002. Pinyon Jay (Gymnorhinus cyanocephalus), in: Rodewald, P.G. (Ed.), The Birds of North America. Cornell Lab of Ornithology, Ithaca, NY.

Balda, R.P., 1987. Avian impacts on pinyon-juniper woodlands, in: Everett, R. (Ed.), Pinyon-Juniper Conference. USDA Forest Service, Reno, NV, pp. 525–533.

Bar Massada, A., Radeloff, V.C., Stewart, S.I., 2011. Allocating fuel breaks to optimally protect structures in the wildland–urban interface. Int. J. Wildl. Fire 20, 59–68. https://doi.org/doi:10.1071/WF09041

Bateman, H., Chung-MacCoubrey, A., Finch, D., Snell, H., Hawksworth, D., 2008a. Impacts of Non-native Plant Removal on Vertebrates along the Middle Rio Grande (New Mexico), Ecological Restoration. https://doi.org/10.3368/er.26.3.193

Bateman, H., Chung-MacCoubrey, A., Snell, H., 2008b. Impact of Non-Native Plant Removal on Lizards in Riparian Habitats in the Southwestern United States, Restoration Ecology. https://doi.org/10.1111/j.1526-100X.2007.00361.x

Battaglia, M., Rhoades, C., Rocca, M.E., Ryan, M.G., 2009. A regional assessment of the ecological effects of chipping and mastication fuels reduction and forest restoration treatments. Joint Fire Science Program, Biose, ID.

Battaglia, M.A., Rocca, M.E., Rhoades, C.C., Ryan, M.G., 2010. Surface fuel loadings within mulching treatments in Colorado coniferous forests. For. Ecol. Manage. 260, 1557–1566. https://doi.org/DOI: 10.1016/j.foreco.2010.08.004

Battles, J., Robards, T., Das, A., Waring, K., Gilless, J., Biging, G., Schurr, F., 2008. Climate change impacts on forest growth and tree mortality: a data-driven modeling study in the mixed-conifer forest of the Sierra Nevada, California. Clim. Change 87, 193–213.

Bender, L.C., 2018. Burning for Big Game. Las Cruces, NM.

Bender, L.C., 2012. Guidelines for Management of Habitat for Mule Deer: Piñon-juniper, Chihuahuan desert, arid grasslands, and associated arid habitat types. NM State University, Cooperative Extension Service, Las Cruces, NM.

Bender, L.C., Boren, J.C., Halbritter, H., Cox, S., 2013. Effects of site characteristics, pinyon-juniper management, and precipitation on habitat quality for mule deer in new mexico.

Human-Wildlife Interact. 7, 47–59.
Bender, L.C., Lomas, L.A., Kamienski, T., 2009. Habitat Effects on Condition of Doe Mule Deer in Arid Mixed Woodland-grassland. Rangel. Ecol. Manag. 60, 277–284. https://doi.org/10.2111/1551-5028(2007)60[277:HEOCOD]2.0.CO;2
Bentz, B.J., Régnière, J., Fettig, C.J., Hansen, E.M., Hayes, J.L., Hicke, J.A., Kelsey, R.G., Negrón, J.F., Seybold, S.J., 2010. Climate Change and Bark Beetles of the Western United States and Canada: Direct and Indirect Effects. Bioscience 60, 602–613. https://doi.org/10.1525/bio.2010.60.8.6
Bentz, Régnière, J., Fettig, C.J., Hansen, E.M., Hayes, J.L., Hicke, J.A., Kelsey, R.G., Negrón, J.F., Seybold, S.J., 2010. Climate Change and Bark Beetles of the Western United States and Canada: Direct and Indirect Effects. Bioscience 60, 602–613. https://doi.org/10.1525/bio.2010.60.8.6
Beschta, R.L., Rhodes, J.J., Boone Kauffman, J., Gresswell, R.E., Wayne Minshall, G., Karr, J.R., Perry, D.A., Richard Hauer, F., Frissell, C.A., 2004. Postfire Management on Forested Public Lands of the Western United States, Conservation Biology.
Beschta, R.L., Ripple, W.J., 2010. Mexican wolves, elk, and aspen in Arizona: Is there a trophic cascade? For. Ecol. Manage. 260, 915–922.
Block, W.M., Finch, D.M., 1997. Songbird ecology in southwestern ponderosa pine forests: a literature review. Fort Collins, CO.
Block, W.M., Ganey, J.L., Scott, P.E., King, R., 2009. Prey ecology of Mexican Spotted Owls in Pine–Oak forests of northern Arizona. J. Wildl. Manage. 69, 618–629. https://doi.org/10.2193/0022-541X(2005)069[0618:PEOMSO]2.0.CO;2
Blomdahl, E.M., Kolden, C.A., Meddens, A.J.H., Lutz, J.A., 2019. The importance of small fire refugia in the central Sierra Nevada, California, USA. For. Ecol. Manage. 432. https://doi.org/10.1016/j.foreco.2018.10.038
Bock, C.E., Block, W.M., 2005. Response of birds to fire in the American southwest, in: Ralph, C. John; Rich, T.D. (Ed.), Bird Conservation Implementation and Integration in the Americas. USDA Forest Service, Pacific Southwest Research Station. PSW-GTR-191, Asilomar, CA, pp. 1093–1099.
Bock, C.E., Bock, J.H., 2014. Effects of wildfire on riparian trees in southeastern Arizona. Southwest. Nat. 59, 568–574. https://doi.org/10.1894/jem-08.1
Boeker, E.L., Scott, V.E., 1969. Roost Tree Characteristics for Merriam’s Turkey. J. Wildl. Manage. 33, 121. https://doi.org/10.2307/3799658
Bombaci, S., Pejchar, L., 2016. Consequences of pinyon and juniper woodland reduction for wildlife in North America. For. Ecol. Manage. 365, 34–50. https://doi.org/http://dx.doi.org/10.1016/j.foreco.2016.01.018
Bombaci, S.P., Gallo, T., Pejchar, L., 2017. Small-Scale Woodland Reduction Practices Have Neutral or Negative Short-Term Effects on Birds and Small Mammals. Rangel. Ecol. Manag. 70, 363–373. https://doi.org/10.1016/j.rama.2016.11.006
Bond, M.L., Gutiérrez, R.J., Franklin, A.B., LaHaye, W.S., May, C.A., Seamans, M.E., 2002. Short-Term Effects of Wildfires on Spotted Owl Survival, Site Fidelity, Mate Fidelity, and Reproductive Success. Wildl. Soc. Bull. 30, 1022–1028.
Boone, J.D., Ammon, E., Johnson, K., 2018. Long-term declines in the Pinyon Jay and management implications for piñon–juniper woodlands, in: Trends and Traditions: Avifaunal Change in Western North America. Western Field Ornithologists, Camarillo, CA, pp. 190–197. https://doi.org/10.21199/SWB3.10.

Borg, C., 2001. Impacts of Timber Harvest on the Endemic Sacramento Mountain Salamander, Aneides hardii (Caudata: Plethodontidae), in the Southern Rocky Mountains of New Mexico, USA.pdf. New Mexico State University.
Bostwick, P., Menakis, J.P., Sexton, T., 2011. How Fuel Treatments Saved Homes from the Wallow Fire. USDA Forest Service, Southwest Region, Albuquerque, NM.
Bowman, D.M.J.S., Williamson, G.J., Abatzoglou, J.T., Kolden, C.A., Cochrane, M.A., Smith, A.M.S., 2017. Human exposure and sensitivity to globally extreme wildfire events. Nat. Ecol. Evol. 1, 58. https://doi.org/10.1038/s41559-016-0058https://www.nature.com/articles/s41559-016-0058#supplementary-information
Boyles, J.G., Aubrey, D.P., 2006. Managing forests with prescribed fire: Implications for a cavity-dwelling bat species. For. Ecol. Manage. 222, 108–115.
Bradley, A., 2009. The New Mexico Forest Restoration Principles: Creating a Common Vision. Ecol. Restor. 27, 22–24.
Brenkert-Smith, H., Meldrum, J.R., Champ, P.A., 2015. Climate change beliefs and hazard mitigation behaviors: homeowners and wildfire risk. Environ. Hazards 14, 341–360. https://doi.org/10.1080/17477891.2015.1080656
Brockway, D.G., Gatewood, R.G., Paris, R.B., 2002. Restoring grassland savannas from degraded pinyon-juniper woodlands: effects of mechanical overstory reduction and slash treatment alternatives. J. Environ. Manage. 64, 179–197.
Brown, P.M., Kaye, M.W., Huckaby, L.S., 2001. Fire history along environmental gradients in the Sacramento Mountains, New Mexico: Influences of local patterns and regional processes. Ecoscience 8, 115–126.
Buchalski, M.R., Fontaine, J.B., Heady III, P.A., Hayes, J.P., Frick, W.F., 2013. Bat Response to Differing Fire Severity in Mixed-Conifer Forest California, USA. PLoS One 8, e57884. https://doi.org/10.1371/journal.pone.0057884
Campbell, J.L., Donato, D.C., Fontaine, J.B., 2016. Effects of post-fire logging on fuel dynamics in a mixed-conifer forest, Oregon, USA: a 10-year assessment. Int. J. Wildl. Fire 25, 646–656. https://doi.org/https://doi.org/10.1071/WF15119
Campbell, L.A., Hallett, J.G., O'Connell, M.A., 1996. Conservation of Bats in Managed Forests: Use of Roosts by Lasionycteris noctivagans. J. Mammal. 77, 976–984. https://doi.org/10.2307/1382778
Cardinal, C.J., Bulger, J.B., 2017. Gould's Wild Turkey (Meleagris gallopavo mexicana) Recovery Plan. Santa Fe, NM.
Carothers, S.W., Johnson, R., Aitchison, S.W., 1974. Population Structure and Social Organization of Southwestern Riparian Birds. Am. Zool. 14, 97–108. https://doi.org/10.1093/icb/14.1.97
Chambers, C., Mast, L., Nystrom, J., 2005. Ponderosa pine snag dynamics and cavity excavation following wildfire in northern Arizona. For. Ecol. Manage. 216, 227–240. https://doi.org/http://dx.doi.org/10.1016/j.foreco.2005.05.033
Chambers, C.L., 1999. Forest management and the dead wood resource in ponderosa pine forests: effects on small mammals, in: Laudenslayer Jr., W.F., Shea, P.J., Valentine, B.E., Weatherspoon, C.P., Lisle, T.E. (Eds.), Symposium on the Ecology and Management of Dead Wood in Western Forests. USDA Forest Service, Pacific Southwest Research Station, Albany, CA, pp. 679–693.
Chung-MacCoubrey, A., 2005a. Use of pinyon-juniper woodlands by bats in New Mexico, Forest Ecology and Management. https://doi.org/10.1016/j.foreco.2004.09.005

Chung-MacCoubrey, A., 2005b. Personal Communication to Jim Stuart, November 14, 2005.
Clarke, M.F., 2008. Catering for the needs of fauna in fire management: science or just wishful thinking? Wildl. Res. 35, 385. https://doi.org/10.1071/WR07137
Cochrane, M.A., Moran, C.J., Wimberly, M.C., Baer, A.D., Finney, M.A., Beckendorf, K.L., Eidenshink, J., Zhu, Z., 2012. Estimation of wildfire size and risk changes due to fuels treatments. Int. J. Wildl. Fire 21, 357–367. https://doi.org/http://dx.doi.org/10.1071/WF11079
Cohen, J.D., 2000. Preventing Disaster: Home Ignitability in the Wildland-Urban Interface. J. For. 98, 15–21.
Collins, B.M., Kramer, H.A., Menning, K., Dillingham, C., Saah, D., Stine, P.A., Stephens, S.L., 2013. Modeling hazardous fire potential within a completed fuel treatment network in the northern Sierra Nevada. For. Ecol. Manage. 310, 156–166. https://doi.org/http://dx.doi.org/10.1016/j.foreco.2013.08.015
Collins, T.W., 2005. Households, forests, and fire hazard vulnerability in the American West: A case study of a California community. Glob. Environ. Chang. Part B Environ. Hazards 6, 23–37. https://doi.org/DOI: 10.1016/j.hazards.2004.12.003
Converse, S.J., White, G.C., Block, W.M., 2006. Small mammal responses to thinning and wildfire in ponderosa pine-dominated forests. J. Wildl. Manage. 70, 1711–1722. https://doi.org/10.2193/0022-541X(2006)70[1711:SMRTTA]2.0.CO;2
Coop, J.D., Grant III, T.A., Magee, P.A., Moore, E.A., 2017. Mastication treatment effects on vegetation and fuels in piñon-juniper woodlands of central Colorado, USA. For. Ecol. Manage. 396, 68–84. https://doi.org/https://doi.org/10.1016/j.foreco.2017.04.007
Coop, J.D., Magee, P.A., 2016. Integrating Fuels Treatments and Ecological Values in Piñon-Juniper Woodlands: Fuels, Vegetation, and Avifauna.
Cooper, C.F., 1960. Changes in vegetation, structure, and growth of southwestern pine forests since white settlement. Ecology 42, 493–499.
Costello, C.M., Jones, D.E., Green-Hammond, K.A., Inman, R., Inman, K., Deitner, B.C., Thompson, R.A., Quigley, H.B., 2001. A study of black bear ecology in New Mexico with models for population dynamics and habitat suitability. Santa Fe, NM.
Covington, W.W., Moore, M.M., 1994. Southwestern ponderosa forest structure: changes since Euro-American settlement. J. For. 92, 39–47.
Cram, D.S., Baker, T.T., Boren, J.C., 2006. Wildland fire effects in silviculturally treated vs. untreated stands on New Mexico and Arizona. U.S. Forest Service, Rocky Mountain Research Station, Fort Collins, CO.
Crockett, J.L., Leroy Westerling, A., 2018. Greater temperature and precipitation extremes intensify Western U.S. droughts, wildfire severity, and sierra Nevada tree mortality. J. Clim. 31. https://doi.org/10.1175/JCLI-D-17-0254.1
Cummer, M.R., Painter, C.W., 2007. Three Case Studies of the Effect of Wildfire on the Jemez Mountains Salamander (Plethodon neomexicanus): Microhabitat Temperatures, Size Distributions, and a Historical Locality Perspective. Southwest. Nat. 52, 26–37. https://doi.org/10.1894/0038-4909(2007)52[26:tcsote]2.0.co;2
Cunningham, S.C., Ballard, W.B., Monroe, L.M., Rabe, M.J., Bristow, K.D., 2003. Black bear habitat use in burned and unburned areas, central Arizona. Wildl. Soc. Bull. 31, 786–792.
Cunningham, S.C., Kirkendall, L., Ballard, W., 2006. Gray fox and coyote abundance and diet responses after a wildfire in central Arizona. West. North Am. Nat. 66, 169–180. https://doi.org/10.3398/1527-0904(2006)66[169:GFACAA]2.0.CO;2

Dahms, C.W., Geils, B.W., 1997. An assessment of forest ecosystem health in the Southwest. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO.
Dailey, S., Fites, J., Reiner, A., Mori, S., 2008. Fire Behavior and Effects in Fuel Treatments and Protected Habitat on the Moonlight Fire. USDA Forest Service, Adaptive Management Service Enterprise Team, Nevada City, CA.
Davies, Z., Tyler, C., Stewart, G., Pullin, A., 2008. Are current management recommendations for saproxylic invertebrates effective? A systematic review. Biodivers. Conserv. 17, 209–234.
DellaSala, D.A., Karr, J.R., Schoennagel, T., Perry, D., Noss, R.F., Lindenmayer, D., Beschta, R., Hutto, R.L., Swanson, M.E., Evans, J., 2006. Post-Fire Logging Debate Ignores Many Issues. Science (80-.). 314, 51b-52b. https://doi.org/10.1126/science.314.5796.51b
DeLong, J.P., Cox, S.W., Cox, N.S., 2005. A comparison of avian use of high- and low-elevation sites during autumn migration in central New Mexico. J. F. Ornithol. 76, 326–333. https://doi.org/10.1648/0273-8570-76.4.326
Dennison, P.E., Brewer, S.C., Arnold, J.D., Moritz, M.A., 2014. Large wildfire trends in the western United States, 1984–2011. Geophys. Res. Lett. 41, 2928–2933.
Dether, D., Black, A., 2006. Learning from escaped prescribed fires-lessons for high reliability. Fire Manag. Today 66, 50–56.
Dieterich, J.H., 1983. Fire history of southwestern mixed-conifer: a case study. J. For. Ecol. Manag. 6, 13–31.
Dillon, G.K., Holden, Z.A., Morgan, P., Crimmins, M.A., Heyerdahl, E.K., Luce, C.H., 2011. Both topography and climate affected forest and woodland burn severity in two regions of the western US, 1984 to 2006. Ecosphere 2, art130. https://doi.org/10.1890/es11-00271.1
Dodd, N.L., Schweinsburg, R.E., Boe, S., 2006. Landscape-Scale Forest Habitat Relationships to Tassel-Eared Squirrel Populations: Implications for Ponderosa Pine Forest Restoration. Restor. Ecol. 14, 537–547. https://doi.org/10.1111/j.1526-100X.2006.00165.x
Dodson, E.K., Root, H.T., 2013. Conifer regeneration following stand-replacing wildfire varies along an elevation gradient in a ponderosa pine forest, Oregon, USA. For. Ecol. Manage. 302, 163–170. https://doi.org/10.1016/J.FORECO.2013.03.050
Dominguez, F., Cañon, J., Valdes, J., 2010. IPCC-AR4 climate simulations for the Southwestern US: the importance of future ENSO projections. Clim. Change 99, 499–514.
Donato, D.C., Fontaine, J.B., Campbell, J.L., Robinson, W.D., Kauffman, J.B., Law, B.E., 2006. Post-wildfire logging hinders regeneration and increases fire risk. Science (80-.). 311, 352. https://doi.org/10.1126/science.1122855
Doumas, S.L., Koprowski, J.L., Noble, W.O., 2015. Landscape-level assessment of Abert’s Squirrel and Red Squirrel in Mixed Conifer Forest. Southwest. Nat. 60, 240–246. https://doi.org/10.1894/SWNAT-D-14-00016.1
Drus, G.M., 2013. Fire ecology of Tamarix, in: Sher, A.A., Quigley, M. (Eds.), Tamarix: A Case Study of Ecological Change in the American West. New York, NY, pp. 240–255.
Dunham, J.B., Young, M.K., Gresswell, R.E., Rieman, B.E., 2003. Effects of fire on fish populations: landscape perspectives on persistence of native fishes and nonnative fish invasions. For. Ecol. Manage. 178, 183–196. https://doi.org/10.1016/S0378-1127(03)00061-6
Dunn, C.J., Bailey, J.D., 2015. Modeling the direct effects of salvage logging on long-term temporal fuel dynamics in dry-mixed conifer forests. For. Ecol. Manage. 341.

https://doi.org/10.1016/j.foreco.2015.01.002
Durkin, P., Muldavin, E., Bradley, M., Carr, S.E., 1995. A preliminary riparian/wetland vegetation community classification of the Upper and Middle Rio Grande watersheds in New Mexico, in: Shaw, D.W., Finch, D.M. (Eds.), Desired Future Conditions for Southwestern Riparian Ecosystems: Bringing Interests and Concerns Together. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station. RM-GTR-272., Albuquerque, NM, pp. 44–57.
Dye, A.J., Moir, W.H., 1977. Spruce-Fir Forest at Its Southern Distribution in the Rocky Mountains, New Mexico. Am. Midl. Nat. 97, 133. https://doi.org/10.2307/2424691
Dyer, J.M., Moffett, K.E., 1999. Meadow Invasion from High-Elevation Spruce-Fir Forest in South-Central New Mexico. Southwest. Nat. 44, 444. https://doi.org/10.2307/3672342
Enquist, C., Gori, D., 2008. Implications of Recent Climate Change on Conservation Priorities in New Mexico CLIMATE CHANGE ECOLOGY & ADAPTATION PROGRAM THE NATURE CONSERVANCY IN NEW MEXICO A CLIMATE CHANGE VULNERABILITY ASSESSMENT FOR BIODIVERSITY IN NEW MEXICO, PART I.
Evans, A., Auerbach, S., Wood Miller, L., Wood, R., Nystrom, K., Loevner, J., Aragon, A., Piccarello, M., Krasilovsky, E., 2015. Evaluating the effectiveness of wildfire mitigation activities in the wildland-urban interface. Forest Stewards Guild, Madison, WI.
Evans, A.M., 2016. Potential ecological consequences of forest biomass harvesting in California. J. Sustain. For. 35, 1–15. https://doi.org/10.1080/10549811.2015.1104254
Evans, A.M., Everett, R., Stephens, S., Youtz, J., 2011. A comprehensive guide to fuels treatment practices for mixed conifer forests: California, central and southern Rockies, and the Southwest. Forest Guild, Santa Fe, NM.
FAC, 2015. Fire Adapted Communities [WWW Document]. URL http://fireadapted.org/
Fair, J.M., Hathcock, C.D., Bartlow, A.W., 2018. Avian communities are decreasing with piñon pine mortality in the southwest. Biol. Conserv. 226, 186–195. https://doi.org/10.1016/j.biocon.2018.06.035
Farnsworth, A., Summerfelt, P., Neary, D.G., Smith, T., 2003. Flagstaff's wildfire fuels treatments: prescriptions for community involvement and a source of bioenergy. Biomass and Bioenergy 24, 269–276. https://doi.org/Doi: 10.1016/s0961-9534(02)00174-5
Feddema, J.J., Mast, J.N., Savage, M., 2013. Modeling high-severity fire, drought and climate change impacts on ponderosa pine regeneration. Ecol. Modell. 253, 56–69. https://doi.org/10.1016/J.ECOLMODEL.2012.12.029
Fettig, C.J., Mortenson, L.A., Bulaon, B.M., Foulk, P.B., 2019. Tree mortality following drought in the central and southern Sierra Nevada, California, U.S. For. Ecol. Manage. 432. https://doi.org/10.1016/j.foreco.2018.09.006
Fettig, C.J., Reid, M.L., Bentz, B.J., Sevanto, S., Spittlehouse, D.L., Wang, T., 2013. Changing Climates, Changing Forests: A Western North American Perspective. J. For. 111, 214–228. https://doi.org/10.5849/jof.12-085
Ffolliott, P.F., 1997. Guidelines for managing wildlife habitats in southwestern ponderosa pine forests of the United States. J. For. Res. 8, 108–110. https://doi.org/10.1007/BF02864979
Fields, C.B., Mortsch, L.D., Brklacich, M., Forbes, D.L., Kovacs, P., 2007. North America. Climate Change 2007: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, UK.
Flatley, W.T., Fulé, P.Z., 2016. Are historical fire regimes compatible with future climate?

Implications for forest restoration. Ecosphere 7. https://doi.org/10.1002/ecs2.1471
Floyd, M.L., Romme, W.H., 2012. Ecological Restoration Priorities and Opportunities in Pinon-Juniper Woodlands. Ecol. Restor. 30, 37–49. https://doi.org/10.3368/er.30.1.37
Fornwalt, P.J., Rocca, M.E., Battaglia, M.A., Rhoades, C.C., Ryan, M.G., 2017. Mulching fuels treatments promote understory plant communities in three Colorado, USA, coniferous forest types. For. Ecol. Manage. 385, 214–224. https://doi.org/http://dx.doi.org/10.1016/j.foreco.2016.11.047
Francis, C.D., Ortega, C.P., Hansen, J., 2011. Importance of juniper to birds nesting in piñon-juniper woodlands in northwest New Mexico. J. Wildl. Manage. 75, 1574–1580. https://doi.org/10.1002/jwmg.200
Franssen, N.R., Gido, K.B., Propst, D.L., 2007. Flow regime affects availability of native and nonnative prey of an endangered predator. Biol. Conserv. 138, 330–340. https://doi.org/10.1016/J.BIOCON.2007.04.028
Frechette, J.D., Meyer, G.A., 2009. Holocene fire-related alluvial-fan deposition and climate in ponderosa pine and mixed-conifer forests, Sacramento Mountains, New Mexico, USA. The Holocene 19, 639–651. https://doi.org/10.1177/0959683609104031
Freedman, J.D., Habeck, J.R., 1984. Fire, logging, and white-tailed deer interrelationships in the Swan Valley, northwestern Montana, in: Lotan, J.E., Brown, J.. (Eds.), Fire's Effects on Wildlife Habitat. USDA Forest Service. INT-GTR-186., Missoula, MT, pp. 23–35.
Fulé, P.Z., 2008. Does It Make Sense to Restore Wildland Fire in Changing Climate? Restor. Ecol. 16, 526–531. https://doi.org/10.1111/j.1526-100X.2008.00489.x
Fulé, P.Z., Covington, W.W., Moore, M.M., 1997. Determining Reference Conditions for Ecosystem Management of Southwestern Ponderosa Pine Forests. Ecol. Appl. 7, 895–908. https://doi.org/doi:10.1890/1051-0761(1997)007[0895:DRCFEM]2.0.CO;2
Fulé, P.Z., Covington, W.W., Smith, H.B., Springer, J.D., Heinlein, T.A., Huisinga, K.D., Moore, M.M., 2002. Testing ecological restoration alternatives: Grand Canyon, Arizona. For. Ecol. Manage. 170, 19–41.
Fulé, P.Z., Crouse, J.E., Heinlein, T.A., Moore, M.M., Covington, W.W., Verkamp, G., 2003. Mixed-severity fire regime in a high-elevation forest of Grand Canyon, Arizona, USA. Landsc. Ecol. 18, 465–486.
Fulé, P.Z., Crouse, J.E., Roccaforte, J.P., Kalies, E.L., 2012. Do thinning and/or burning treatments in western USA ponderosa or Jeffrey pine-dominated forests help restore natural fire behavior? For. Ecol. Manage. 269, 68–81. https://doi.org/10.1016/j.foreco.2011.12.025
Gagliuso, R.A., 1991. Habitat alteration and human disturbance : their impact on cougar habitat utilization in southwest Oregon. Oregon State University.
Gaines, W.L., Harrod, R.J., Dickinson, J., Lyons, A.L., Halupka, K., 2010. Integration of Northern spotted owl habitat and fuels treatments in the eastern Cascades, Washington, USA. For. Ecol. Manage. 260, 2045–2052. https://doi.org/DOI: 10.1016/j.foreco.2010.09.011
Gallo, T., Stinson, L.T., Pejchar, L., 2016. Pinyon-juniper removal has long-term effects on mammals. For. Ecol. Manage. 377, 93–100. https://doi.org/http://dx.doi.org/10.1016/j.foreco.2016.06.029
Ganey, J.L., 1999. Snag density and composition of snag populations on two National Forests in northern Arizona. For. Ecol. Manage. 117, 169–178.
Ganey, J.L., Vojta, S.C., 2012. Rapid increase in log populations in drought-stressed mixed-conifer and ponderosa pine forests in northern Arizona. Open J. For. 2, 59.

Garfin, G., Jardine, A., Merideth, R., Black, M., LeRoy, S., 2013a. Assessment of Climate Change in the Southwest United States. Southwest Climate Alliance and Island Press, Washington, DC.

Garfin, G., Jardine, A., Merideth, R., Black, M., LeRoy, S., 2013b. Assessment of Climate Change in the Southwest United States. Southwest Climate Alliance and Island Press, Washington, DC.

Gass, T.M., Robinson, A.P., 2007. A hierarchical analysis of stand structure, composition, and burn patterns as indicators of stand age in an Engelmann spruce – subalpine fir forest. Can. J. For. Res. 37, 884–894.

Gavier-Pizarro, G.I., Radeloff, V.C., Stewart, S.I., Huebner, C.D., Keuler, N.S., 2010. Housing is positively associated with invasive exotic plant species richness in New England, USA. Ecol. Appl. 20, 1913–1925. https://doi.org/doi:10.1890/09-2168.1

Germaine, S.S., Germaine, H.L., Boe, S.R., 2009. Characteristics of mule deer day-bed and forage sites in current-condition and restoration-treated ponderosa pine forest. Wildl. Soc. Bull. 32, 554–564. https://doi.org/10.2193/0091-7648(2004)32[554:COMDDA]2.0.CO;2

Gillihan, S.W., 2006. Sharing the land with pinyon-juniper birds. Salt Lake City, UT.

Grialou, J.A., West, S.D., Wilkins, R.N., 2000. The Effects of Forest Clearcut Harvesting and Thinning on Terrestrial Salamanders. J. Wildl. Manage. 64, 105. https://doi.org/10.2307/3802979

Haffey, C., Sisk, T.D., Allen, C.D., Thode, A.E., Margolis, E.Q., 2018. Limits to Ponderosa Pine Regeneration Following Large High-Severity Forest Fires in the United States Southwest. Fire Ecol. 14, 143–163.

Halbritter, H., Bender, L.C., 2011. Quality of Habitat Occupied by Elk (Cervus elaphus) in the Southern Sacramento Mountains, New Mexico. Southwest. Nat. 56, 1–8. https://doi.org/10.1894/TAL-14.1

Halbritter, H.H., Bender, L.C., 2015. Herbivory of sympatric elk and cattle on lincoln national forest, south-central New Mexico. For. Ecosyst. 2, 25. https://doi.org/10.1186/s40663-015-0049-0

Harris, G.R., Covington, W.W., 1983. The effect of a prescribed fire on nutrient concentration and standing crop of understory vegetation in ponderosa pine. Can. J. For. Res. 13, 501–507. https://doi.org/10.1139/x83-074

Harrison, N.M., Stubblefield, A.P., Varner, J.M., Knapp, E.E., 2016. Finding balance between fire hazard reduction and erosion control in the Lake Tahoe Basin, California–Nevada. For. Ecol. Manage. 360, 40–51. https://doi.org/http://dx.doi.org/10.1016/j.foreco.2015.10.030

Harrod, R.J., Peterson, D.W., Ottmar, R., 2008. Effects of mechanically generated slash particle size on prescribed fire behavior and subsequent vegetation effects. Joint Fire Science Program, Boise, ID.

Harvey, B.J., Donato, D.C., Turner, M.G., 2016. High and dry: post-fire tree seedling establishment in subalpine forests decreases with post-fire drought and large stand-replacing burn patches. Glob. Ecol. Biogeogr. 25, 655–669. https://doi.org/10.1111/geb.12443

Havrilla, C.A., Faist, A.M., Barger, N.N., 2017. Understory Plant Community Responses to Fuel-Reduction Treatments and Seeding in an Upland Piñon-Juniper Woodland. Rangel. Ecol. Manag. 70, 609–620. https://doi.org/10.1016/j.rama.2017.04.002

Heffelfinger, J.R., Brewer, C., Alcalá-Galván, C.H., Hale, B., Weybright, D.L., Wakeling, B.F., Carpenter, L.H., Dodd., N.L., 2006. Habitat Guidelines for Mule Deer: Southwest Deserts Ecoregion. Boise, ID.

Helms, J.A., 1998. The Dictionary of Forestry. Society of American Foresters, Bethesda, MD.
Higgins, A.M., Waring, K.M., Thode, A.E., 2015. The effects of burn entry and burn severity on ponderosa pine and mixed conifer forests in Grand Canyon National Park. Int. J. Wildl. Fire 24, 495–506. https://doi.org/http://dx.doi.org/10.1071/WF13111
Hoffman, R.W., Shaw, H.G., Rumble, M.A., Wakeling, B.F., Mollohan, C.M., Schemnitz, S.D., Engel-Wilson, R., Hengel, D.A., 1993. Management guidelines for Merriam's wild turkeys. Denver, CO.
Holden, Z.A., Morgan, P., Evans, J.S., 2009. A predictive model of burn severity based on 20-year satellite-inferred burn severity data in a large southwestern US wilderness area. For. Ecol. Manage. 258, 2399–2406.
Horncastle, V.J., Fenner Yarborough, R., Dickson, B.G., Rosenstock, S.S., 2013. Summer habitat use by adult female mule deer in a restoration-treated ponderosa pine forest. Wildl. Soc. Bull. 37, 707–713. https://doi.org/10.1002/wsb.301
Howard, V.W., Jr., Cheap, K.M., Hier, R.H., Thompson, T.G., Dimas, J.A., 1987. Effects of cabling pinyon-juniper on mule deer and lagomorph use, in: Everett, R.L. (Ed.), Pinyon-Juniper Conference. USDA Forest Service, Reno, NV, pp. 552–557.
Huffman, D.W., Crouse, J.E., Walker Chancellor, W., Fulé, P.Z., 2012. Influence of time since fire on pinyon–juniper woodland structure. For. Ecol. Manage. 274, 29–37. https://doi.org/10.1016/j.foreco.2012.02.014
Huffman, D.W., Stoddard, M.T., Springer, J.D., Crouse, J.E., 2017. Understory Responses to Tree Thinning and Seeding Indicate Stability of Degraded Pinyon-Juniper Woodlands. Rangel. Ecol. Manag. 70, 484–492. https://doi.org/10.1016/j.rama.2017.01.008
Huffman, D.W., Zegler, T.J., Fulé, P.Z., 2015. Fire history of a mixed conifer forest on the Mogollon Rim, northern Arizona, USA. Int. J. Wildl. Fire 24, 680–689. https://doi.org/http://dx.doi.org/10.1071/WF14005
Hultine, K.R., Belnap, J., van Riper, C., Ehleringer, J.R., Dennison, P.E., Lee, M.E., Nagler, P.L., Snyder, K.A., Uselman, S.M., West, J.B., 2010. Tamarisk biocontrol in the western United States: ecological and societal implications. Front. Ecol. Environ. 8, 467–474. https://doi.org/10.1890/090031
Hunter, M.E., Shepperd, W.D., Lentile, J.E., Lundquist, J.E., Andreu, M.G., Butler, J.L., Smith, F.W., 2007. A comprehensive guide to fuels treatment practices for ponderosa pine in the Black Hills, Colorado Front Range, and Southwest. USDA Forest Service, Rocky Mountain Research Station, Fort Collins, CO.
Hutto, R.L., Gallo, S.M., 2006. The effects of postfire salvage logging on cavity-nesting birds. Condor 108, 817–831. https://doi.org/10.1650/0010-5422(2006)108[817:teopsl]2.0.co;2
Hutto, R.L., Hejl, S.J., Preston, C.R., Finch, D.M., 1993. Effects of silvicultural treatments on forest birds in the Rocky Mountais: implications and management recomendations, in: Finch, D.M., Stangel, P.W. (Eds.), Status and Management of Neotropical Migratory Birds. USAD Forest Service, Rocky Mountain Research Station. GTR-RM-229, Estes Park, CO, pp. 386–391.
Hutto, R.L., Patterson, D.A., 2016. Positive effects of fire on birds may appear only under narrow combinations of fire severity and time-since-fire. Int. J. Wildl. Fire 25, 1074–1085. https://doi.org/https://doi.org/10.1071/WF15228
Jacobs, B.F., 2015. Restoration of degraded transitional (piñon-juniper) woodland sites improves ecohydrologic condition and primes understory resilience to subsequent disturbance. Ecohydrology 8, 1417–1428. https://doi.org/10.1002/eco.1591

Jacobs, B.F., 2011. Spatial patterns and ecological drivers of historic piñon–juniper woodland expansion in the American southwest. Ecography (Cop.). 34, 1085–1095. https://doi.org/10.1111/j.1600-0587.2011.06614.x
Jacobs, B.F., Romme, W.H., Allen, C.D., 2008. Mapping "old" vs. "young" piñon-juniper stands with a predictive topo-climatic model. Ecol. Appl. 18, 1627–41.
Jain, T., Sikkink, P., Keefe, R., Byrne, J., 2018. To masticate or not: Useful tips for treating forest, woodland, and shrubland vegetation, RMRS-GTR-381. Fort Collins, CO.
Jain, T.B., Battaglia, M.A., Han, H.-S., Graham, R.T., Christopher R. Keyes, Fried, J.S., Sandquist, J.E., 2012. A Comprehensive Guide to Fuel Management Practices for Dry Mixed Conifer Forests in the Northwestern United States. USDA Forest Service, Rocky Mountain Research Station, Fort Collins, CO.
Jenkins, S.E., Hull Sieg, C., Anderson, D.E., Kaufman, D.S., Pearthree, P.A., 2011. Late Holocene geomorphic record of fire in ponderosa pine and mixed-conifer forests, Kendrick Mountain, northern Arizona, USA. Int. J. Wildl. Fire 20, 125–141. https://doi.org/doi:10.1071/WF09093
Jentsch, S., Mannan, R.W., Dickson, B.G., Block, W.M., 2008. Associations Among Breeding Birds and Gambel Oak in Southwestern Ponderosa Pine Forests. J. Wildl. Manage. 72, 994–1000. https://doi.org/10.2193/2007-073
Johnson, A.S., Landers, J.L., 1982. Habitat Relationships of Summer Resident Birds in Slash Pine Flatwoods. J. Wildl. Manage. 46, 416–428.
Johnson, K., Petersen, N., Smith, J., Sadoti, G., 2018a. Piñon-juniper fuels reduction treatment impacts pinyon jay nesting habitat. Glob. Ecol. Conserv. 16. https://doi.org/10.1016/j.gecco.2018.e00487
Johnson, K., Petersen, N., Smith, J., Sadoti, G., 2018b. Surveys for Nesting Pinyon Jays at Rio Grande del Norte National Monument. Albuquerque, NM.
Johnson, K., Sadoti, G., 2019. Model transferability and implications for woodland management: a case study of Pinyon Jay nesting habitat. Avian Conserv. Ecol. 14, art17. https://doi.org/10.5751/ACE-01467-140217
Johnson, K., Sadoti, G., Smith, J., 2017. Weather-induced declines in piñon tree condition and response of a declining bird species. J. Arid Environ. 146, 1–9. https://doi.org/10.1016/j.jaridenv.2017.07.006
Johnson, K., Wickersham, L., Smith, J., Petersen, N., Wickersham, J., 2015. Nest-scale Habitat Use by Pinyon Jay and Gray Vireo in the BLM Farmington Resource Area 2013−2014. Albuquerque, NM.
Johnson, M.C., Kennedy, M.C., Peterson, D.L., 2011. Simulating fuel treatment effects in dry forests of the western United States: testing the principles of a fire-safe forest. Can. J. For. Res. 41, 1018–1030. https://doi.org/doi:10.1139/x11-032
Johnson, M.J., Magill, R.T., III, C. van R., 2010. Yellow-billed cuckoo distribution and habitat associations in Arizona, 1998-1999, in: van Riper III, C., Wakeling, B.F., Sisk, T.D. (Eds.), The Colorado Plateau IV: Integrating Research and Resources Management for Effective Conservation. University of Arizona Press, Tucson, AZ, pp. 197–212.
Jones, A., 2019. Do Mechanical Vegetation Treatments of Pinyon-Juniper and Sagebrush Communities Work? Salt Lake City, UT.
Jones, J.R., 1974. Silviculture of southwestern mixed conifers and aspen: the status of our knowledge. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, Co.

Kalies, E.L., Chambers, C.L., Covington, W.W., 2010. Wildlife responses to thinning and burning treatments in southwestern conifer forests: A meta-analysis. For. Ecol. Manage. 259, 333–342.
Kalies, E.L., Covington, W.W., 2012. Small mammal community maintains stability through compensatory dynamics after restoration of a ponderosa pine forest. Ecosphere 3, art78. https://doi.org/10.1890/ES12-00143.1
Kalies, E.L., Dickson, B.G., Chambers, C.L., Covington, W.W., 2011. Community occupancy responses of small mammals to restoration treatments in ponderosa pine forests, northern Arizona, USA. Ecol. Appl. 22, 204–217. https://doi.org/10.1890/11-0758.1
Kalies, E.L., Yocom Kent, L.L., 2016. Tamm Review: Are fuel treatments effective at achieving ecological and social objectives? A systematic review. For. Ecol. Manage. 375, 84–95. https://doi.org/http://dx.doi.org/10.1016/j.foreco.2016.05.021
Kamler, J.F., Lee, R.M., DeVos, J.C., Ballard, W.B., Whitlaw, H.A., 2002. Survival and Cougar Predation of Translocated Bighorn Sheep in Arizona. J. Wildl. Manage. 66, 1267. https://doi.org/10.2307/3802959
Kane, J.M., Kolb, T.E., 2014. Short- and long-term growth characteristics associated with tree mortality in southwestern mixed-conifer forests. Can. J. For. Res. 44, 1227–1235. https://doi.org/10.1139/cjfr-2014-0186
Kane, J.M., Kolb, T.E., McMillin, J.D., 2014. Stand-scale tree mortality factors differ by site and species following drought in southwestern mixed conifer forests. For. Ecol. Manage. 330, 171–182. https://doi.org/http://dx.doi.org/10.1016/j.foreco.2014.06.042
Karr, J.R., Rhodes, J.J., Minshall, G.W., Hauer, F.R., Beschta, R.L., Frissell, C.A., Perry, D.A., 2004. The Effects of Postfire Salvage Logging on Aquatic Ecosystems in the American West. Bioscience 54, 1029–1033. https://doi.org/10.1641/0006-3568(2004)054[1029:teopsl]2.0.co;2
Katz, G., Shafroth, P., 2007. Biology, Ecology and Management of Elaeagnus angustifolia l. (Russian Olive) in Western North America. Wetlands 23, 5212–5212. https://doi.org/10.1672/0277-5212(2003)023
Keane, R.E., Hessburg, P.F., Landres, P.B., Swanson, F.J., 2009. The use of historical range and variability (HRV) in landscape management. For. Ecol. Manage. 258, 1025–1037. https://doi.org/DOI: 10.1016/j.foreco.2009.05.035
Kelly, L.T., Brotons, L., McCarthy, M.A., 2017. Putting pyrodiversity to work for animal conservation. Conserv. Biol. 31, 952–955. https://doi.org/10.1111/cobi.12861
Kelly, L.T., Nimmo, D.G., Spence-Bailey, L.M., Taylor, R.S., Watson, S.J., Clarke, M.F., Bennett, A.F., 2012. Managing fire mosaics for small mammal conservation: a landscape perspective. J. Appl. Ecol. 49, 412–421. https://doi.org/10.1111/j.1365-2664.2012.02124.x
Kennedy, M.C., Johnson, M.C., 2014. Fuel treatment prescriptions alter spatial patterns of fire severity around the wildland–urban interface during the Wallow Fire, Arizona, USA. For. Ecol. Manage. 318, 122–132. https://doi.org/http://dx.doi.org/10.1016/j.foreco.2014.01.014
Keyser, A., Leroy Westerling, A., 2017. Climate drives inter-annual variability in probability of high severity fire occurrence in the western United States. Environ. Res. Lett. 12. https://doi.org/10.1088/1748-9326/aa6b10
Keyser, A.R., Westerling, A.L.R., 2019. Predicting increasing high severity area burned for three forested regions in the western United States using extreme value theory. For. Ecol. Manage. 432. https://doi.org/10.1016/j.foreco.2018.09.027
Klebenow, D.A., 1965. A Montane Forest Winter Deer Habitat in Western Montana. J. Wildl.

Manage. 29, 27–33.
Kleintjes, P.K., Jacobs, B.F., Fettig, S.M., 2004. Initial response of butterflies to an overstory reduction and slash mulching treatment of a degraded piñon-juniper woodland. Restor. Ecol. 12, 231–238. https://doi.org/10.1111/j.1061-2971.2004.00212.x
Knapp, E.E., 2015. Long-term dead wood changes in a Sierra Nevada mixed conifer forest: Habitat and fire hazard implications. For. Ecol. Manage. 339, 87–95. https://doi.org/http://dx.doi.org/10.1016/j.foreco.2014.12.008
Knick, S.T., Hanser, S.E., Grace, J.B., Hollenbeck, J.P., Leu, M., 2017. Response of bird community structure to habitat management in piñon-juniper woodland-sagebrush ecotones. For. Ecol. Manage. 400, 256–268. https://doi.org/10.1016/j.foreco.2017.06.017
Knick, S.T., Hanser, S.E., Leu, M., 2014. Ecological Scale of Bird Community Response to Piñon-Juniper Removal. Rangel. Ecol. Manag. 67, 553–562. https://doi.org/10.2111/REM-D-13-00023.1
Korb, J.E., Fulé, P.Z., Stoddard, M.T., 2012. Forest restoration in a surface fire-dependent ecosystem: An example from a mixed conifer forest, southwestern Colorado, USA. For. Ecol. Manage. 269, 10–18. https://doi.org/10.1016/j.foreco.2012.01.002
Korb, J.E., Fulé, P.Z., Wu, R., 2013. Variability of warm/dry mixed conifer forests in southwestern Colorado, USA: Implications for ecological restoration. For. Ecol. Manage. 304, 182–191. https://doi.org/http://dx.doi.org/10.1016/j.foreco.2013.04.028
Kotliar, N.B., Kennedy, P.L., Ferree, K., 2007. Avifaunal responses to fire in southwestern montane forests along a burn severity gradient. Ecol. Appl. 17, 491–507. https://doi.org/10.1890/06-0253
Kramer, D.W., Sorensen, G.E., Taylor, C.A., Cox, R.D., Gipson, P.S., Cain, J.W., 2015. Ungulate exclusion, conifer thinning and mule deer forage in northeastern New Mexico. J. Arid Environ. 113, 29–34. https://doi.org/10.1016/J.JARIDENV.2014.09.008
Krausman, P.R., Shaw, W.W., Etchberger, R.C., Harris, L.K., 1995. The decline of bighorn sheep in the Santa Catalina Mountains, Arizona. Biodivers. Manag. madrean Archipel. sky islands Southwest. United States Northwest. Mex. 245–250.
Kreye, J.K., Brewer, N.W., Morgan, P., Varner, J.M., Smith, A.M.S., Hoffman, C.M., Ottmar, R.D., 2014. Fire behavior in masticated fuels: A review. For. Ecol. Manage. 314, 193–207. https://doi.org/http://dx.doi.org/10.1016/j.foreco.2013.11.035
Kroll, A.J., Giovanini, J., Jones, J.E., Arnett, E.B., Altman, B., 2012. Effects of salvage logging of beetle-killed forests on avian species and foraging guild abundance. J. Wildl. Manage. 76, 1188–1196. https://doi.org/10.1002/jwmg.353
Krosby, M., Theobald, D.M., Norheim, R., McRae, B.H., 2018. Identifying riparian climate corridors to inform climate adaptation planning. PLoS One 13, e0205156. https://doi.org/10.1371/journal.pone.0205156
Kundaeli, J.N., Reynolds, H.G., 1972. Desert Cottontail Use of Natural and Modified Pinyon-Juniper Woodland. J. Range Manag. 25, 116. https://doi.org/10.2307/3896798
Lamont, B.G., Monteith, K.L., Merkle, J.A., Mong, T.W., Albeke, S.E., Hayes, M.M., Kauffman, M.J., 2019. Multi-scale habitat selection of elk in response to beetle-killed forest. J. Wildl. Manage. jwmg.21631. https://doi.org/10.1002/jwmg.21631
Leal, D.A., Meyer, R.A., Thompson, B.C., 1995. Avian community composition and habitat importance in the Rio Grande corridor of New Mexico, in: Shaw, D.W., Finch, D.M. (Eds.), Desired Future Conditions for Southwestern Riparian Ecosystems: Bringing Interests and Concerns Together. USDA Forest Service, Rocky Mountain Forest and Range Experiment

Station. RM-GTR-272., Albuquerque, NM, pp. 62–68.
Lee, D.E., 2018. Spotted Owls and forest fire: a systematic review and meta-analysis of the evidence. Ecosphere 9. https://doi.org/10.1002/ecs2.2354
Lee, M.E., Miller, R., 2003. Managing Elk in the Wildland-Urban Interface: Attitudes of Flagstaff, Arizona Residents. Wildl. Soc. Bull. https://doi.org/10.2307/3784372
Lesh, T.D., 1999. Habitat Selection by Selected Breeding Passerine Birds in Pine-Oak Forests of Northern Arizona.
Leverkus, A.B., Lindenmayer, D.B., Thorn, S., Gustafsson, L., 2018. Salvage logging in the world's forests: Interactions between natural disturbance and logging need recognition. Glob. Ecol. Biogeogr. 27, 1140–1154. https://doi.org/10.1111/geb.12772
Lindenmayer, D., Thorn, S., Banks, S., 2017. Please do not disturb ecosystems further. Nat. Ecol. Evol. https://doi.org/10.1038/s41559-016-0031
Lindenmayer, D.B., Noss, R.F., 2006. Society for Conservation Biology Salvage Logging, Ecosystem Processes, and Biodiversity Conservation. Biology (Basel). 20, 949–958. https://doi.org/10.1111/j
Littell, J.S., McKenzie, D., Peterson, D.L., Westerling, A.L., 2009. Climate and wildfire area burned in western U.S. ecoprovinces, 1916–2003. Ecol. Appl. 19, 1003–1021. https://doi.org/10.1890/07-1183.1
Loberger, C.D., Theimer, T.C., Rosenstock, S.S., Wightman, C.S., 2011. Use of restoration-treated ponderosa pine forest by tassel-eared squirrels. J. Mammal. 92, 1021–1027. https://doi.org/10.1644/10-MAMM-A-321.1
Loudermilk, E.L., Stanton, A., Scheller, R.M., Dilts, T.E., Weisberg, P.J., Skinner, C., Yang, J., 2014. Effectiveness of fuel treatments for mitigating wildfire risk and sequestering forest carbon: A case study in the Lake Tahoe Basin. For. Ecol. Manage. 323, 114–125. https://doi.org/http://dx.doi.org/10.1016/j.foreco.2014.03.011
Lundquist, J.D., Dettinger, M.D., Stewart, I.T., Cayan, D.R., 2009. Variability and Trends in Spring Runoff in the Western United States. University of Utah Press, pp. 63–76.
Lydersen, J., North, M., 2012. Topographic Variation in Structure of Mixed-Conifer Forests Under an Active-Fire Regime. Ecosystems 15, 1134–1146. https://doi.org/10.1007/s10021-012-9573-8
Lydersen, J.M., Collins, B.M., Brooks, M.L., Matchett, J.R., Shive, K.L., Povak, N.A., Kane, V.R., Smith, D.F., 2017. Evidence of fuels management and fire weather influencing fire severity in an extreme fire event. Ecol. Appl. 27, 2013–2030. https://doi.org/10.1002/eap.1586
Lynch, A.M., 2004. Fate and characteristics of Picea damaged by Elatobium abietinum (Walker) (Homoptera: Aphididae) in the White Mountains of Arizona. West. North Am. Nat. 64, 7–17.
Lynch, D.L., Romme, W.H., Floyd, M.E., 2000. Forest restoration in southwestern ponderosa pine. J. For. 98, 17–24.
Magee, P.A., Coop, J.D., Ivan, J.S., 2019. Thinning alters avian occupancy in piñon–juniper woodlands. Condor Ornithol. Appl. 121. https://doi.org/10.1093/condor/duy008
Manning, J.A., Edge, W.D., 2008. Small mammal responses to fine woody debris and forest fuel reduction in southwest Oregon. J. Wildl. Manage. 72, 625–632. https://doi.org/10.2193/2005-508
Marcot, B.G., Ohmann, J.L., Mellen-McLean, K.L., Waddell, K.L., 2010. Synthesis of Regional Wildlife and Vegetation Field Studies to Guide Management of Standing and Down Dead

Trees. For. Sci. 56, 391–404.
Margolis, E.Q., 2014. Fire regime shift linked to increased forest density in a piñon–juniper savanna landscape. Int. J. Wildl. Fire 23, 234–245. https://doi.org/http://dx.doi.org/10.1071/WF13053
Margolis, E.Q., Malevich, S.B., 2016. Historical dominance of low-severity fire in dry and wet mixed-conifer forest habitats of the endangered terrestrial Jemez Mountains salamander (Plethodon neomexicanus). For. Ecol. Manage. 375, 12–26. https://doi.org/http://dx.doi.org/10.1016/j.foreco.2016.05.011
Margolis, E.Q., Swetnam, T.W., Allen, C.D., 2011. Historical Stand-Replacing Fire in Upper Montane Forests of the Madrean Sky Islands and Mogollon Plateau, Southwestern USA. J. Assoc. Fire Ecol. 7, 88–107.
Martinson, E.J., Omi, P.N., 2013. Fuel Treatments and Fire Severity: A Meta-Analysis. USDA Forest Service, Rocky Mountain Research Station, Fort Collins, CO.
Martinuzzi, S., Stewart, S.I., Helmers, D.P., Mockrin, M.H., Hammer, R.B., Radeloff, V.C., 2015. The 2010 wildland-urban interface of the conterminous United States. USDA Forest Service, Northern Research Station, Newtown Square, PA.
McCabe, G.J., Wolock, D.M., 2007. Warming may create substantial water supply shortages in the Colorado River basin. Geophys. Res. Lett. 34, L22708. https://doi.org/10.1029/2007GL031764
McDonald, R.I., Motzkin, G., Bank, M.S., Kittredge, D.B., Burk, J., Foster, D.R., 2006. Forest harvesting and land-use conversion over two decades in Massachusetts. For. Ecol. Manage. 227, 31–41.
McDowell, N.G., Williams, A.P., Xu, C., Pockman, W.T., Dickman, L.T., Sevanto, S., Pangle, R., Limousin, J., Plaut, J., Mackay, D.S., Ogee, J., Domec, J.C., Allen, C.D., Fisher, R.A., Jiang, X., Muss, J.D., Breshears, D.D., Rauscher, S.A., Koven, C., 2016. Multi-scale predictions of massive conifer mortality due to chronic temperature rise. Nat. Clim. Chang. 6, 295–300. https://doi.org/10.1038/nclimate2873
McGrath, L.J., van Riper III, C., Fontaine, J.J., 2009. Flower power: tree flowering phenology as a settlement cue for migrating birds. J. Anim. Ecol. 78, 22–30. https://doi.org/10.1111/j.1365-2656.2008.01464.x
McIver, J.D., Starr, L., 2001. A Literature Review on the Environmental Effects of Postfire Logging. West. J. Appl. For. 16, 159–168. https://doi.org/10.1093/wjaf/16.4.159
Meddens, A.J.H., Hicke, J.A., Jacobs, B.F., 2016. Characterizing the Response of Piñon-Juniper Woodlands to Mechanical Restoration Using High-Resolution Satellite Imagery. Rangel. Ecol. Manag. 69, 215–223. https://doi.org/10.1016/j.rama.2015.12.006
Meldrum, J.R., Champ, P.A., Brenkert-Smith, H., Warziniack, T., Barth, C.M., Falk, L.C., 2015. Understanding Gaps Between the Risk Perceptions of Wildland–Urban Interface (WUI) Residents and Wildfire Professionals. Risk Anal. n/a-n/a. https://doi.org/10.1111/risa.12370
Millar, C.I., Stephenson, N.L., Stephens, S.L., 2007. Climate Change and Forests of the Future: Managing in the Face of Uncertainty, Source: Ecological Applications.
Miller, J.D., Safford, H., Crimmins, M., Thode, A., 2009. Quantitative Evidence for Increasing Forest Fire Severity in the Sierra Nevada and Southern Cascade Mountains, California and Nevada, USA. Ecosystems 12, 16–32. https://doi.org/10.1007/s10021-008-9201-9
Miller, R., 2001. Managing Western Juniper for Wildlife, Woodland Fish and Wildlife Series. Washington State University Extension, Pullman, WA.
Miller, R.F., Chambers, J.C., Pellant, M., 2014. A Field Guide for Selecting the Most

Appropriate Treatment in Sagebrush and Piñon-Juniper Ecosystems in the Great Basin Evaluating Resilience to Disturbance and Resistance to Invasive Annual Grasses, and Predicting Vegetation Response. Fort Collins, CO.
Mitchell, R.J., Liu, Y., O'Brien, J.J., Elliott, K.J., Starr, G., Miniat, C.F., Hiers, J.K., 2014. Future climate and fire interactions in the southeastern region of the United States. For. Ecol. Manage. 327, 316–326. https://doi.org/http://dx.doi.org/10.1016/j.foreco.2013.12.003
Molles, M.C., Crawford, C.S., Ellis, L.M., Valett, H.M., Dahm, C.N., 1998. Managed Flooding for Riparian Ecosystem Restoration. Bioscience 48, 749–756. https://doi.org/10.2307/1313337
Moore, M.M., Casey, C.A., Bakker, J.D., Springer, J.D., Fule, P.Z., Covington, W.W., Laughlin, D.C., 2006. Herbaceous vegetation responses (1992-2004) to restoration treatments in a ponderosa pine forest. Rangel. Ecol. Manag. 59, 135–144.
Morgan, P., Aplet, G.H., Haufler, J.B., Humphries, H.C., Moore, M.M., Wilson, W.D., 1994. Historical Range of Variability -- A Useful Tool for Evaluating Ecosystem Change. J. Sustain. For. 2, 87–111.
Morillas, L., Pangle, R.E., Maurer, G.E., Pockman, W.T., McDowell, N., Huang, C.-W., Krofcheck, D.J., Fox, A.M., Sinsabaugh, R.L., Rahn, T.A., Litvak, M.E., 2017. Tree Mortality Decreases Water Availability and Ecosystem Resilience to Drought in Piñon-Juniper Woodlands in the Southwestern U.S. J. Geophys. Res. Biogeosciences 122, 3343–3361. https://doi.org/10.1002/2017JG004095
Moritz, M.A., Parisien, M.-A., Batllori, E., Krawchuk, M.A., Dorn, J. Van, Ganz, D.J., Hayhoe, K., 2012. Climate change and disruptions to global fire activity. Ecosphere 3, 1–22. https://doi.org/10.1890/ES11-00345.1
Morrell, T.E., Rabe, M.J., DeVos, J.C., Green, H., Miller, C.R., 1999. Bats captured in two ponderosa pine habitats in north-central Arizona. Southwest. Nat. https://doi.org/10.2307/3672350
Mosher, K.R., Bateman, H.L., 2016. The effects of riparian restoration following saltcedar (Tamarix spp.) biocontrol on habitat and herpetofauna along a desert stream. Restor. Ecol. 24, 71–80. https://doi.org/10.1111/rec.12273
Neill, A.R., Puettmann, K.J., 2013. Managing for adaptive capacity: thinning improves food availability for wildlife and insect pollinators under climate change conditions. Can. J. For. Res. 43, 428–440. https://doi.org/10.1139/cjfr-2012-0345
Nelson, K.C., Monroe, M.C., Johnson, J.F., Bowers, A., 2004. Living with fire: homeowner assessment of landscape values and defensible space in Minnesota and Florida, USA. Int. J. Wildl. Fire 13, 413–425. https://doi.org/doi:10.1071/WF03067
New Mexico Forest Restoration Principles Working Group, 2007. Draft Piñon-Juniper Framework. Las Vegas, NM.
NIFC, 2019. Total Wildland Fires and Acres. Boise, ID.
NMDGF, 2017. Habitat Restoration and Management of Native and Non-Native Trees in Southwestern Riparian Ecosystems, Habitat Handbook. Santa Fe, NM.
NMDGF, 2016. State Wildlife Action Plan for New Mexico. Santa Fe, NM.
Noss, R.F., Franklin, J.F., Baker, W.L., Moyle, P.B., 2006. Managing fire-prone forests in the western United States. Front. Ecol. 4, 481–487.
NRCS, NMACD, 2007. A Guide for Planning Riparian Treatments in New Mexico. Albuquerque, NM.
O'Connor, C.D., Falk, D.A., Lynch, A.M., Swetnam, T.W., 2014. Fire severity, size, and climate

associations diverge from historical precedent along an ecological gradient in the Pinaleño Mountains, Arizona, USA. For. Ecol. Manage. 329, 264–278. https://doi.org/http://dx.doi.org/10.1016/j.foreco.2014.06.032

O'Connor, C.D., Lynch, A.M., Falk, D.A., Swetnam, T.W., 2015. Post-fire forest dynamics and climate variability affect spatial and temporal properties of spruce beetle outbreaks on a Sky Island mountain range. For. Ecol. Manage. 336. https://doi.org/10.1016/j.foreco.2014.10.021

Oswald, B., Dugan, S., Balice, R., Unger, D., 2016. Overstory Tree Mortality in Ponderosa Pine and Spruce-Fir Ecosystems Following a Drought in Northern New Mexico. Forests 7, 225.

Overby, S.T., Moir, W.H., Robertson, G.T., 2000. Soil and vegetation changes in a pinyon-juniper area in central Arizona after prescribed fire, in: Ffolliott, Peter F. Baker Jr., M.B., Edminster, C.B., Dillon, M.C., Mora, K.L. (Eds.), Land Stewardship in the 21st Century: The Contributions of Watershed Management. Tucson, AZ, pp. 371–374.

Parks, S.A., Holsinger, L.M., Panunto, M.H., Jolly, W.M., Dobrowski, S.Z., Dillon, G.K., 2018. High-severity fire: Evaluating its key drivers and mapping its probability across western US forests. Environ. Res. Lett. 13. https://doi.org/10.1088/1748-9326/aab791

Patton, D.R., 1969. Deer and elk use of a ponderosa pine forest in Arizona before and after timber harvest. Tempe, AZ.

Paulin, K.M., Cook, J.J., Dewey, S.R., 1999. Pinyon-Juniper Woodlands as Sources of Avian Diversity 240–243.

Peek, J.M., Demarchi, D.A., Demarchi, R.A., Stucker, D.E., 1984. Bighorn sheep and fire: seven case studies, in: Lotan, J.E., Brown, J.. (Eds.), Fire's Effects on Wildlife Habitat. USDA Forest Service. INT-GTR-186., Missoula, MT, pp. 36–43.

Perry, L.G., Andersen, D.C., Reynolds, L. V., Nelson, S.M., Shafroth, P.B., 2012. Vulnerability of riparian ecosystems to elevated CO2 and climate change in arid and semiarid western North America. Glob. Chang. Biol. 18, 821–842. https://doi.org/10.1111/j.1365-2486.2011.02588.x

Peterson, D.W., Dodson, E.K., 2016. Post-fire logging produces minimal persistent impacts on understory vegetation in northeastern Oregon, USA. For. Ecol. Manage. 370. https://doi.org/10.1016/j.foreco.2016.04.004

Peterson, D.W., Dodson, E.K., Harrod, R.J., 2015. Post-fire logging reduces surface woody fuels up to four decades following wildfire. For. Ecol. Manage. 338, 84–91. https://doi.org/http://dx.doi.org/10.1016/j.foreco.2014.11.016

Pilliod, D.S., Bull, E.L., Hayes, J.L., Wales, B.C., 2006. Wildlife and invertebrate response to fuel reduction treatments in dry coniferous forests of the Western United States: a synthesis. USDA For. Serv. Fac. Publ. 63. https://doi.org/10.1016/j.foreco.2013.06.058

Pilliod, D.S., Bury, R.B., Hyde, E.J., Pearl, C.A., Corn, P.S., 2003. Fire and amphibians in North America. For. Ecol. Manage. 178, 163–181. https://doi.org/10.1016/S0378-1127(03)00060-4

Pitt, A.L., Tavano, J.J., Baldwin, R.F., Waldrop, T.A., 2013. Effects of fuel reduction treatments on movement and habitat use of American toads in a southern Appalachian hardwood forest. For. Ecol. Manage. 310, 289–299. https://doi.org/http://dx.doi.org/10.1016/j.foreco.2013.08.032

Pollet, J., Omi, P.N., 2002. Effect of thinning and prescribed burning on crown fire severity in ponderosa pine forests. Int. J. Wildl. Fire 11, 1–10.

Prather, J.W., Noss, R.F., Sisk, T.D., 2008. Real versus perceived conflicts between restoration

of ponderosa pine forests and conservation of the Mexican spotted owl. For. Policy Econ. 10, 140–150. https://doi.org/DOI: 10.1016/j.forpol.2007.07.003
Prichard, S.J., Kennedy, M.C., 2013. Fuel treatments and landform modify landscape patterns of burn severity in an extreme fire event. Ecol. Appl. 24, 571–590. https://doi.org/10.1890/13-0343.1
Prichard, S.J., Kennedy, M.C., 2012. Fuel treatment effects on tree mortality following wildfire in dry mixed conifer forests, Washington State, USA. Int. J. Wildl. Fire 21, 1004–1013. https://doi.org/http://dx.doi.org/10.1071/WF11121
Prichard, S.J., Peterson, D.L., Jacobson, K., 2010. Fuel treatments reduce the severity of wildfire effects in dry mixed conifer forest, Washington, USA. Can. J. For. Res. 40, 1615–1626.
Prowse, T.A.A., Collard, S.J., Blackwood, A., O'Connor, P.J., Delean, S., Barnes, M., Cassey, P., Possingham, H.P., 2017. Prescribed burning impacts avian diversity and disadvantages woodland-specialist birds unless long-unburnt habitat is retained. Biol. Conserv. 215, 268–276. https://doi.org/10.1016/j.biocon.2017.09.005
Radeloff, V.C., Helmers, D.P., Kramer, H.A., Mockrin, M.H., Alexandre, P.M., Bar-Massada, A., Butsic, V., Hawbaker, T.J., Martinuzzi, S., Syphard, A.D., Stewart, S.I., 2018. Rapid growth of the US wildland-urban interface raises wildfire risk. Proc. Natl. Acad. Sci. U. S. A. 115, 3314–3319. https://doi.org/10.1073/pnas.1718850115
Ramotnik, C., 1997. Conservation Assessment of the Sacramento Mountain Salamander.pdf. Fort Collins, CO.
Rangwala, I., Miller, J.R., 2010. Twentieth Century Temperature Trends in Colorado's San Juan Mountains. Arctic, Antarct. Alp. Res. 42, 89–97. https://doi.org/10.1657/1938-4246-42.1.89
Ratajczak, Z., Nippert, J.B., Collins, S.L., 2011. Woody encroachment decreases diversity across North American grasslands and savannas. Ecology 93, 697–703. https://doi.org/10.1890/11-1199.1
Reagan, D.P., 1972. Ecology and Distribution of the Jemez Mountains Salamander, Plethodon neomexicanus. Copeia 486–492.
Redmond, M.D., Cobb, N.S., Clifford, M.J., Barger, N.N., 2015. Woodland recovery following drought-induced tree mortality across an environmental stress gradient. Glob. Chang. Biol. 21, 3685–3695. https://doi.org/10.1111/gcb.12976
Rehfeldt, G.E., Crookston, N.L., Warwell, M. V, Evans, J.S., 2006. EMPIRICAL ANALYSES OF PLANT-CLIMATE RELATIONSHIPS FOR THE WESTERN UNITED STATES.
Reid, K., 2019. Key for the Five Piñon-Juniper types. Las Vegas, NM.
Reiner, A.L., Vaillant, N.M., Fites-Kaufman, J., Dailey, S.N., 2009. Mastication and prescribed fire impacts on fuels in a 25-year old ponderosa pine plantation, southern Sierra Nevada. For. Ecol. Manage. 258, 2365–2372.
Reynolds, H.G., Clary, W.P., Ffolliott, P.F., 1970a. Gambel oak for southwestern wildlife. J. For. 68, 545–547.
Reynolds, H.G., Clary, W.P., Ffolliott, P.F., 1970b. Gambel oak for southwestern wildlife. J. For. 68, 545–547.
Reynolds, R.T., Meador, A.J.S., Youtz, J.A., Nicolet, T., Matonis, M.S., Jackson, P.L., DeLorenzo, D.G., Graves, A.D., 2013. Restoring Composition and Structure in Southwestern Frequent-Fire Forests. USDA Forest Service, Rocky Mountain Research Station, Fort Collins, CO.
Ritchie, M.W., Knapp, E.E., Skinner, C.N., 2013. Snag longevity and surface fuel accumulation following post-fire logging in a ponderosa pine dominated forest. For. Ecol. Manage. 287,

113–122. https://doi.org/http://dx.doi.org/10.1016/j.foreco.2012.09.001
Rodhouse, T., Hirnyck, R., Gerald Wright, R., 2010. Habitat Selection of Rodents Along a Pinyon-Juniper Woodland-Savannah Gradient, Journal of Mammalogy. https://doi.org/10.1644/09-MAMM-A-060.1
Romme, W., Allen, C., Bailey, J., Baker, W., Bestelmeyer, B., Brown, P., Eisenhart, K., Floyd-Hanna, L., Huffman, D., Jacobs, B., Miller, R., Muldavin, E., Swetnam, T., Tausch, R., Weisberg, P., 2007. Historical and Modern Disturbance Regimes of Piñon-Juniper Vegetation in the Western U.S. Colorado Forest Restoration Institute and the Nature Conservancy.
Romme, W.H., Allen, C.D., Bailey, J.D., Baker, W.L., Bestelmeyer, B.T., Brown, P.M., Eisenhart, K.S., Floyd, M.L., Huffman, D.W., Jacobs, B.F., Miller, R.F., Muldavin, E.H., Swetnam, T.W., Tausch, R.J., Weisberg, P.J., 2009. Historical and Modern Disturbance Regimes, Stand Structures, and Landscape Dynamics in Piñon–Juniper Vegetation of the Western United States. Rangel. Ecol. Manag. 62, 203–222. https://doi.org/10.2111/08-188R1.1
Romme, W.H., Kaufmann, M., Veblen, T.T., Sherriff, R., Regan, C., 2003. Ecological Effects of the Hayman Fire: Historical (Pre-1860) and Current (1860-2002) Forest and Landscape Structure, in: Graham, R.T. (Ed.), Hayman Fire Case Study. USDA Forest Service, Rocky Mountain Research Station, Fort Collins, CO, pp. 196–203.
Rudis, V.A., Tansey, J.B., 1995. Regional Assessment of Remote Forests and Black Bear Habitat from Forest Resource Surveys. J. Wildl. Manage. 59, 170–180.
Ryniker, K.A., Bush, J.K., Van Auken, O.W., 2006. Structure of Quercus gambelii communities in the Lincoln National Forest, New Mexico, USA. For. Ecol. Manage. 233, 69–77. https://doi.org/10.1016/J.FORECO.2006.06.008
Saab, V., Block, W., Russell, R., Lehmkuhl, J., Bate, L., White, R., 2007. Birds and burns of the interior West: descriptions, habitats, and management in western forests. Portland, OR. https://doi.org/10.2737/PNW-GTR-712
Saab, V.A., Russell, R.E., Dudley, J.G., 2009. Nest-site selection by cavity-nesting birds in relation to postfire salvage logging. For. Ecol. Manage. 257, 151–159. https://doi.org/10.1016/J.FORECO.2008.08.028
Saab, V.A., Russell, R.E., Rotella, J., Dudley, J.G., 2011. Modeling nest survival of cavity-nesting birds in relation to postfire salvage logging. J. Wildl. Manage. 75, 794–804. https://doi.org/10.1002/jwmg.111
Safford, H.D., Schmidt, D.A., Carlson, C.H., 2009. Effects of fuel treatments on fire severity in an area of wildland-urban interface, Angora Fire, Lake Tahoe Basin, California. For. Ecol. Manage. 258, 773–787.
Safford, H.D., Stevens, J.T., Merriam, K., Meyer, M.D., Latimer, A.M., 2012. Fuel treatment effectiveness in California yellow pine and mixed conifer forests. For. Ecol. Manage. 274, 17–28. https://doi.org/10.1016/j.foreco.2012.02.013
Sample, V.A., 2017. Potential for Additional Carbon Sequestration through Regeneration of Nonstocked Forest Land in the United States. J. For. 115, 309–318. https://doi.org/10.5849/jof.2016-005
Saud, P., Cram, D., Smallidge, S., Baker, T., 2018. Coarse Woody Debris Following Silviculture Treatments in Southwest Mixed-Conifer Forest. Forests 9, 347. https://doi.org/10.3390/f9060347
Savage, M., Derr, T., Evans, A., Krasilovsky, E., Smith, K., Carey, H., 2008. Short guide for

developing CFRP restoration prescriptions. New Mexico Forest and Watershed Restoration Institute, Las Vegas, NM.
Savage, M., Swetnam, T.W., 1990. Early 19th-Century Fire Decline Following Sheep Pasturing in a Navajo Ponderosa Pine Forest. Ecology 71, 2374–2378. https://doi.org/10.2307/1938649
Scott, V.E., Boeker, E.L., 1977. Responses of Merriam's Turkey to Pinyon-Juniper Control. J. Range Manag. 30, 220. https://doi.org/10.2307/3897475
Severson, K.E., 1986. Small Mammals in Modified Pinyon-Juniper Woodlands, New Mexico. J. Range Manag. 39, 31. https://doi.org/10.2307/3899682
Shafroth, P.B., Cleverly, J.R., Dudley, T.L., Taylor, J.P., Riper III, C., Weeks, E.P., Stuart, J.N., 2005. Control of Tamarix in the western United States: implications for water salvage, wildlife use, and riparian restoration. Environ. Manage. 35, 231–246. https://doi.org/10.1007/s00267-004-0099-5
Shinneman, D.J., Baker, W.L., 2009. Historical fire and multidecadal drought as context for piñon-juniper woodland restoration in western Colorado. Ecol. Appl. 19, 1231–1245.
Short, H.L., Evans, W., Boeker, E.L., 1977. The Use of Natural and Modified Pinyon Pine-Juniper Woodlands by Deer and Elk. J. Wildl. Manage. 41, 543. https://doi.org/10.2307/3800529
Sibold, J.S., Veblen, T.T., González, M.E., 2006. Spatial and temporal variation in historic fire regimes in subalpine forests across the Colorado Front Range in Rocky Mountain National Park, Colorado, USA. J. Biogeogr. 33, 631–647. https://doi.org/10.1111/j.1365-2699.2005.01404.x
Siegel, R.B., Desante, D.E., 2003. Bird communities in thinned versus unthinned sierran mixed conifer stands. Wilson Bull. 115, 155–165.
Singleton, M.P., Thode, A.E., Sánchez Meador, A.J., Iniguez, J.M., 2019a. Increasing trends in high-severity fire in the southwestern USA from 1984 to 2015. For. Ecol. Manage. 433, 709–719. https://doi.org/10.1016/J.FORECO.2018.11.039
Singleton, M.P., Thode, A.E., Sánchez Meador, A.J., Iniguez, J.M., 2019b. Increasing trends in high-severity fire in the southwestern USA from 1984 to 2015. For. Ecol. Manage. 433, 709–719. https://doi.org/10.1016/J.FORECO.2018.11.039
Smith, J.M., Paritsis, J., Veblen, T.T., Chapman, T.B., 2015. Permanent forest plots show accelerating tree mortality in subalpine forests of the Colorado Front Range from 1982 to 2013. For. Ecol. Manage. 341, 8–17. https://doi.org/http://dx.doi.org/10.1016/j.foreco.2014.12.031
Smith, T.S., Hardin, P.J., Flinders, J.T., 1999. Response of bighorn and sheep to logging prescribed burning. Wildl. Soc. Bull. 27, 840–845.
Spear, S.F., Storfer, A., 2010. Anthropogenic and natural disturbance lead to differing patterns of gene flow in the Rocky Mountain tailed frog, Ascaphus montanus. Biol. Conserv. 143, 778–786. https://doi.org/10.1016/J.BIOCON.2009.12.021
Spears, M., Harrison, A., Sankovich, V., Gabgopadhyay Subhrendu, 2013. Literature Synthesis on Climate Change Implications for Water and Environmental Resources Third Edition. Denver, Colorado.
Stager, D.W., Klebenow, D.A., 1987. Mule deer response to wildfire in Great Basin pinyon-juniper woodland, in: Everett, R.L. (Ed.), Pinyon-Juniper Conference. USDA Forest Service, Reno, NV, NV, pp. 572–579.
Stake, M.M., Garber, G., 2008. Gray vireo monitoring in northwestern and southeastern New

Mexico, in: Gray Vireo Symposium. New Mexico Department of Game and Fish, Conservation Services Division, Santa Fe, NM, pp. 11–13.
Stein, S.M., Carr, M.A., McRoberts, R.E., Mahal, L.G., Comas, S.J., 2010. Threats to at-risk species in America's private forests: a Forests on the Edge report. USDA Forest Service, Northern Research Station, Newtown Square, PA.
Stenseth, N.C., Durant, J.M., Fowler, M.S., Matthysen, E., Adriaensen, F., Jonzen, N., Chan, K.-S., Liu, H., De Laet, J., Sheldon, B.C., Visser, M.E., Dhondt, A.A., 2015. Testing for effects of climate change on competitive relationships and coexistence between two bird species. Proc. R. Soc. B Biol. Sci. 282, 20141958–20141958. https://doi.org/10.1098/rspb.2014.1958
Stephens, J.L., Alexander, J.D., 2011. Effects of fuel reduction on bird density and reproductive success in riparian areas of mixed-conifer forest in southwest Oregon. For. Ecol. Manage. 261, 43–49. https://doi.org/DOI: 10.1016/j.foreco.2010.09.022
Stephens, Johnston, D., Jonas, J., Paschke, M., 2016a. Understory responses to mechanical treatment of pinyon-juniper in northwestern Colorado. Rangel. Ecol. Manag. 69, 351–359. https://doi.org/10.1016/j.rama.2016.06.003
Stephens, Miller, J.D., Collins, B.M., North, M.P., Keane, J.J., Roberts, S.L., 2016b. Wildfire impacts on California spotted owl nesting habitat in the Sierra Nevada. Ecospehere 7.
Stephens, S.L., Bigelow, S.W., Burnett, R.D., Collins, B.M., Gallagher, C. V, Keane, J., Kelt, D.A., North, M.P., Roberts, L.J., Stine, P.A., Van Vuren, D.H., 2014. California Spotted Owl, Songbird, and Small Mammal Responses to Landscape Fuel Treatments. Bioscience 64, 893–906. https://doi.org/10.1093/biosci/biu137
Stephens, S.L., Millar, C.I., Collins, B.M., 2010. Operational approaches to managing forests of the future in Mediterranean regions within a context of changing climates. Environ. Res. Lett. 5, 24003.
Stephens, S.L., Moghaddas, J.J., 2005a. Experimental fuel treatment impacts on forest structure, potential fire behavior, and predicted tree mortality in a California mixed conifer forest. For. Ecol. Manage. 215, 21–36.
Stephens, S.L., Moghaddas, J.J., 2005b. Fuel treatment effects on snags and coarse woody debris in a Sierra Nevada mixed conifer forest. For. Ecol. Manage. 214, 53–64.
Stevens-Rumann, C., Shive, K., Fulé, P., Sieg, C.H., 2013. Pre-wildfire fuel reduction treatments result in more resilient forest structure a decade after wildfire. Int. J. Wildl. Fire 22, 1108–1117. https://doi.org/http://dx.doi.org/10.1071/WF12216
Stevens, J.T., Safford, H.D., Latimer, A.M., 2014. Wildfire-contingent effects of fuel treatments can promote ecological resilience in seasonally dry conifer forests. Can. J. For. Res. 44, 843–854. https://doi.org/10.1139/cjfr-2013-0460
Stoddard, M.T., Huffman, D.W., Alcoze, T.M., Fule, P.Z., 2008. Effects of slash on herbaceous communities in pinyon-juniper woodlands of northern Arizona. Rangel. Ecol. Manag. 61, 485–495.
Stromberg, J.C., McCluney, K.E., Dixon, M.D., Meixner, T., 2013. Dryland Riparian Ecosystems in the American Southwest: Sensitivity and Resilience to Climatic Extremes. Ecosystems 16, 411–415. https://doi.org/10.1007/s10021-012-9606-3
Stromberg, J.C., Rychener, T.J., Dixon, M.D., 2008. Return of Fire to a Free-Flowing Desert River: Effects on Vegetation. Restor. Ecol.
Swanson, F.J., Jones, J.A., Wallin, D.O., Cissel, J.H., 1994. Natural variability - implications for ecosystem management, in: Jensen, M.E., Bourgeron, P.S. (Eds.), Eastside Forest

Ecosystem Health Assessment Vol II: Ecosystem Management Principles and Applications. GTR-PNW-318. USDA Forest Service, Pacific Northwest Research Station.
Swetnam, T.W., 1993. Fire history and climate change in giant sequoia groves. Science (80-.). 262, 885–889.
Swetnam, T.W., Baisan, C.H., 1996. Historical Fire Regime Patterns in the Southwestern United States Since AD 1700, in: Allen, C.D. (Ed.), Fire Effects in Southwestern Fortest : 2nd La Mesa Fire Symposium. USDA Forest Service, Rocky Mountain Research Station, Los Alamos, NM, pp. 11–32.
Taylor, D.A., 2006. Forest management and bats.
Tepley, A.J., Veblen, T.T., 2015. Spatiotemporal fire dynamics in mixed-conifer and aspen forests in the San Juan Mountains of southwestern Colorado, USA. Ecol. Monogr. 85, 583–603. https://doi.org/10.1890/14-1496.1
Tingley, M.W., Ruiz-Gutiérrez, V., Wilkerson, R.L., Howell, C.A., Siegel, R.B., 2016. Pyrodiversity promotes avian diversity over the decade following forest fire. Proc. R. Soc. B Biol. Sci. 283. https://doi.org/10.1098/rspb.2016.1703
TNC, NRCS, BLM, Sierra Club, USFS, NMSF, Forest Guild, CBD, Solutions, R., Public Service of New Meixco, 2006. New Mexico forest restoration principles. Santa Fe, NM.
Touchan, R., Allen, C.D., Swetnam, T.W., 1996. Fire history and climatic patterns in ponderosa pine and mixed-conifer forests of the Jemez Mountains, northern New Mexico, in: Allen, C.D. (Ed.), Fire Effects in Southwestern Forests: Proceedings of the Second La Mesa Fire Symposium. RM-GTR-286. USDA Forest Service, Fort Collins, CO, pp. 33–46.
Trottier, G.C., Carbyn, L.N., Scotter, G.W., 1989. Effects of Prescribed Fire on Small Mammals in Aspen Parkland, in: Proceedings of the North American Prairie Conference. Edmonton, Alberta, pp. 179–182.
USFWS, 2013. Designation of Critical Habitat for the Jemez Mountains Salamander, in: Federal Register. Washington, DC, pp. 69569–69570.
USFWS, 2002. Southwestern willow flycatcher (Empidonax traillii extimus) final recovery plan. Albuquerque, NM.
USFWS, 1995. Recovery plan for the Mexican spotted owl. USDI Fish and Wildlife Service, Albuquerque, NM.
Van de Water, K., North, M., 2011. Stand structure, fuel loads, and fire behavior in riparian and upland forests, Sierra Nevada Mountains, USA; a comparison of current and reconstructed conditions. For. Ecol. Manage. 262, 215–228. https://doi.org/10.1016/j.foreco.2011.03.026
Van de Water, K., North, M., 2010. Fire history of coniferous riparian forests in the Sierra Nevada. For. Ecol. Manage. 260, 384–395.
Van Dyke, F., Fox, A., Harju, S.M., Dzialak, M.R., Hayden-Wing, L.D., Winstead, J.B., 2012. Response of mule deer to habitat modification near natural gas development. Wildl. Biol. Pract. 8, 50–69.
Van Mantgem, P.J., Stephenson, N.L., 2007. Apparent climatically induced increase of tree mortality rates in a temperate forest. Ecol. Lett. 10, 909–916. https://doi.org/10.1111/j.1461-0248.2007.01080.x
Verner, J., McKelvey, K.S., Noon, B.R., Gutierrez, R.J., Gould Jr., G.I., Beck, T.W., 1992. The California spotted owl: a technical assessment of its current status. USDA Forest Service, Pacific Southwest Research Station, Albany, CA.
Visser, M.E., Both, C., Lambrechts, M.M., 2004. Global Climate Change Leads to Mistimed Avian Reproduction, in: Adv. Ecol. Res. pp. 89–110. https://doi.org/10.1016/S0065-

2504(04)35005-1
Waconda, J., 2008. Ecology and management of pinyon-juniper ecosystems in the Bureau of Indian Affairs Southwestern Region, in: Gottfried, G.J., Shaw, J.D., Ford, P.L. (Eds.), Ecology, Management, and Restoration of Pinon-Juniper and Ponderosa Pine Ecosystems. USDA Forest Service, Rocky Mountain Research Station. RMRS-P-51, Albuquerque, NM, pp. 139–142.
Walker, H.A., 2006. Southwestern avian community organization in exotic Tamarix: Current patterns and future needs. Monit. Sci. Technol. Symp. Unifying Knowl. Sustain. West. Hemisphere, Sept. 20-24, 2004, Denver, CO 274–286.
Waltz, A.E.M., Stoddard, M.T., Kalies, E.L., Springer, J.D., Huffman, D.W., Meador, A.S., 2014. Effectiveness of fuel reduction treatments: Assessing metrics of forest resiliency and wildfire severity after the Wallow Fire, AZ. For. Ecol. Manage. 334, 43–52. https://doi.org/http://dx.doi.org/10.1016/j.foreco.2014.08.026
Wampler, C.R., Frey, J.K., VanLeeuwen, D.M., Boren, J.C., Baker, T.T., 2008. Mammals in Mechanically Thinned and Non-Thinned Mixed-Coniferous Forest in the Sacramento Mountains, New Mexico. Southwest. Nat. 53, 431–443. https://doi.org/10.1894/CLG-10.1
Watling, J.I., Nowakowski, A.J., Donnelly, M.A., Orrock, J.L., 2011. Meta-analysis reveals the importance of matrix composition for animals in fragmented habitat. Glob. Ecol. Biogeogr. 20, 209–217. https://doi.org/10.1111/j.1466-8238.2010.00586.x
Webster, K.M., Halpern, C.B., 2010. Long-term vegetation responses to reintroduction and repeated use of fire in mixed-conifer forests of the Sierra Nevada. Ecosphere 1, art9. https://doi.org/doi:10.1890/ES10-00018.1
Westerling, A., Bryant, B., 2008. Climate change and wildfire in California. Clim. Change 87, 231–249.
Westerling, A.L.R., 2016. Increasing western US forest wildfire activity: Sensitivity to changes in the timing of spring. Philos. Trans. R. Soc. B Biol. Sci. 371. https://doi.org/10.1098/rstb.2015.0178
WFLC, 2014. The National Strategy: The Final Phase of the Development of the National Cohesive Wildland Fire Management Strategy. Wildland Fire Executive Council, Washington, DC.
Willis, M.J., Miller, R.F., 1997. Importance of western juniper communities to small mammals, in: Monsen, S., Richards, S., Tausch, R., Miller, R., Goodrich, C. (Eds.), Ecology and Management of Piñon-Juniper Communities within the Interior West. USDA Forest Service, Provo, UT, pp. 210–214.
Wimberly, M.C., Cochrane, M.A., Baer, A.D., Pabst, K., 2009. Assessing fuel treatment effectiveness using satellite imagery and spatial statistics. Ecol. Appl. 19, 1377–1384. https://doi.org/10.1890/08-1685.1
Wuebbles, D., Meehl, G., Hayhoe, K., Karl, T.R., Kunkel, K., Santer, B., Wehner, M., Colle, B., Fischer, E.M., Fu, R., Goodman, A., Janssen, E., Kharin, V., Lee, H., Li, W., Long, L.N., Olsen, S.C., Pan, Z., Seth, A., Sheffield, J., Sun, L., Wuebbles, D., Meehl, G., Hayhoe, K., Karl, T.R., Kunkel, K., Santer, B., Wehner, M., Colle, B., Fischer, E.M., Fu, R., Goodman, A., Janssen, E., Kharin, V., Lee, H., Li, W., Long, L.N., Olsen, S.C., Pan, Z., Seth, A., Sheffield, J., Sun, L., 2014. CMIP5 Climate Model Analyses: Climate Extremes in the United States. Bull. Am. Meteorol. Soc. 95, 571–583. https://doi.org/10.1175/BAMS-D-12-00172.1
Yarborough, R.F., Gist, J.A., Loberger, C.D., Rosenstock, S.S., 2015. Habitat use by Abert's

squirrels (Sciurus aberti) in managed forests. Southwest. Nat. 60, 166–170. https://doi.org/10.1894/jkf-49.1

Yocom-Kent, L.L., Fulé, P.Z., Bunn, W.A., Gdula, E.G., 2015. Historical high-severity fire patches in mixed-conifer forests. Can. J. For. Res. 45, 1587–1596. https://doi.org/10.1139/cjfr-2015-0128

York, D.L., Schemnitz, S.D., 2003. Home range, habitat use, and diet of Gould's turkeys, Peloncillo Mountains, New Mexico. Southwest. Nat. 48, 231–240. https://doi.org/10.1894/0038-4909(2003)048<0231:HRHUAD>2.0.CO;2