Section 4. Wildlife and Habitat

Chapter 4.1. Biodiversity and Representative Species in Dry Pine Forests

Brice B. Hanberry and R. Kasten Dumroese[1]

Introduction

As described in Chapter 1.1 (Dumroese, this synthesis, *The Northeastern California Plateaus Bioregion Science Synthesis: Background, Rationale, and Scope*), the Lassen and Modoc National Forests (hereafter the Lassen and the Modoc) share dry pine forestland that was not addressed by two prior science syntheses: *Science Synthesis to Support Socioecological Resilience in the Sierra Nevada and Southern Cascade Range* (hereafter, *Sierra Nevada Science Synthesis*; Long and others 2014) and Synthesis of Science to Inform Land Management within the Northwest Forest Plan Area (hereafter Northwest Forest Plan Science Synthesis; Spies et al. 2018). These previous syntheses concentrated on biodiversity and species, such as fisher (*Martes pennanti*), Pacific marten (*Martes caurina*), and spotted owl (*Strix occidentalis*), associated with dense, closed-canopy forests, for example, late-successional and old-growth Douglas-fir (*Pseudotsuga menziesii*) and western hemlock (*Tsuga heterophylla*) forests.

In contrast to closed-canopy forests, dry pine forests provide a range of open conditions with greater exposure to light and wind, along with a unique fire ecology that historically consisted of frequent low-severity fires. However, similar to closed-canopy forests, dry pine forests have the potential to become old forests because ponderosa pine (*Pinus ponderosa*), the predominant species, can live up to 600 years. In this chapter, we combined an overview of biodiversity with a more detailed examination of representative species of dry ponderosa pine forests in the Southern Cascade Mountains and Modoc Plateau of the Lassen and Modoc. We focused on black-backed woodpecker (*Picoides arcticus*) and flammulated owl (*Psiloscops flammeolus*), which represent examples of Sierra Nevada species of local interest for forest planning. We also included an endangered species listed under the U.S. Federal Endangered Species Act of 1973, the gray wolf (*Canis lupus*) that probably will increase in California. Although gray wolves use a variety of ecosystem types that contain large ungulates, their preferred prey in California likely will be deer and elk (Kovacs et al. 2016), which use herbaceous plants and tree cover present in dry pine forests.

We supplemented limited research specific to the Lassen and the Modoc with information from dry pine forests in other locations. Dry pine forests have wide distributions, and comprehensive species lists or detailed studies from outside the region may not be completely transferable to conditions in Northeastern California. Studies on wolves specific to this region likely will become available within the next 2 decades, after establishment of multiple packs, while some invertebrate species may never be inventoried. Even within a region, many studies may be necessary for a complete synthesis, because studies may produce variable and potentially contrasting results arising from natural ecological variation. Differences in local species assemblages, forest structure (e.g., composition, tree density, large-tree density, snag density, amount of canopy cover), treatments (e.g., fire compared to thinning, severity of thinning and fires, number and timing of applications, measurable effect on vegetation), abundance of nonnative plants and other species, survey design (e.g., number of replications, time since treatment, short-term compared to long-term responses, number of survey years), spatial scale, weather before and during measurements, climate, soils, topography, natural disturbances, management history, metapopulation dynamics, and random chance are a few issues that may influence results.

Fungi, Lichen, Mosses, and Herbaceous Plants

Open pine forests generally contain a gradient of conditions that favor a different suite of fungi and plant species than do closed forests. In contrast to closed forests,

[1]**Brice B. Hanberry** is a research ecologist, U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, 8221 South Highway 16, Rapid City, SD 57702; **R. Kasten Dumroese** is a research plant physiologist, U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station and director, Western Center for Native Plant Conservation and Restoration Science, 1221 South Main Street, Moscow, ID 83843.

Citation: Hanberry, B.B.; Dumroese, R.K. 2020. Biodiversity and representative species in dry pine forests. In: Dumroese, R.K.; Moser, W.K., eds. Northeastern California plateaus bioregion science synthesis. Gen. Tech. Rep. RMRS-GTR-409. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 85–100.

open pine forests are generally drier, less protected, and typically contain less internal structural complexity. Species associated with open pine forests may need to be clarified for the Southern Cascades and Modoc Plateau. Limited information underscores the need for increased published research to assess status of fungi, lichen, and mosses in dry pine forests generally, and specifically in the Lassen and Modoc.

Fungi provide a variety of ecological functions, including maintenance of soil structure and nutrient availability. Fungal fruiting bodies are food sources for many animals, such as flying squirrels and other small mammals, and in turn, animals spread fungus (Lehmkuhl et al. 2004a). Fungal assemblages are unique in dry pine forests, particularly depending on time after fire (Reazin et al. 2016). Some fungi, primarily ascomycetes, fruit after fire during rain events, probably from a spore bank in which some spores require fire to germinate (Claridge et al. 2009). Postfire fungi stabilize soil and capture nitrogen, assisting reestablishment of herbaceous cover after fire (Claridge et al. 2009).

Lichen provide ecological services, such as nitrogen fixation, and are a food source for animals, including ungulates. In general, lichen biodiversity is great in the Southern Cascades and low in the Modoc Plateau, due to differences in moisture and exposure (Jovan 2008). Lichen associations are unique in dry pine forests (Lehmkuhl et al. 2004b), but current levels of research are insufficient to reconcile conflicting information about lichen associations (Lesher et al. 2003b; Lehmkuhl et al. 2004b). Wolf lichen (*Letharia* sp.) is the most common lichen in dry pine forests of the Lassen and the Modoc.

Although mosses are not major components of dry pine forests, rare mosses are most abundant in ponderosa pine-Douglas-fir forests compared to other ecosystems of Northwestern Washington (Heinlen and Vitt 2003). Composition of the moss layer changes in response to fuel-reduction treatments of logging and burning, and pioneers quickly colonize after fire, thereby stabilizing soils (Hardman and McCune 2010). Few studies of liverworts and biological crusts in ponderosa pine forests appear to be available (Williston 2000; see also Chapter 3.4, Warren, this synthesis, *Biological Soil Crusts*).

Dry pine forests include grasses such as Idaho fescue (*Festuca idahoensis*) and bluebunch wheatgrass (*Pseudoroegneria spicata*), along with a variety of flowering herbaceous plants. Herbaceous vegetation provides habitat for animal species, including pollinators, birds, small mammals, and ungulates. Even mistletoe brooms will provide nesting sites for some raptors (Pilliod et al. 2006). Shrub cover may be present, depending on tolerance to fire and fire-return interval. For example, even though antelope bitterbrush (*Purshia tridentata*) is a dominant shrub in dry pine forests, bitterbrush is fire-sensitive and bitterbrush cover was 75 to 92 percent lower after 2 burns during an 11-year interval (Busse and Riegel 2009).

Dry pine forests vary in tree density, providing different structure for fungi, plants, and wildlife. These forests can be open and contain an understory of herbaceous vegetation due to light availability and limited woody plant development and tree debris. Coarse woody debris and other thick organic layers, while important to many species, block herbaceous plant establishment, similar to garden mulches. In addition, light, heat, or smoke are required by some species to break dormancy, particularly legumes, poppies, and phlox (Bigelow and Manley 2009). Frequent, low-severity fire in pine forests removes woody vegetation and forest-floor debris, reducing competition and mechanical barriers for herbaceous plant establishment, while in turn, herbaceous vegetation spreads surface fire (fig. 4.1.1).

Figure 4.1.1—Low-intensity surface fires, such as this one near Coyote Springs on the Lassen, encourage development of herbaceous plants by reducing competition from woody shrubs and by developing forest-floor conditions that favor establishment of grasses and wildflowers (photo by Debbie Mayer, Forest Service).

Land use produces different effects in dry pine forests. Overgrazing tends to convert native bunchgrasses to invasive annual grasses and perennial forbs and reduce native flowering plant and insect abundance (Johnson et al. 1994; Niwa et al. 2001; Wisdom et al. 2000). In contrast, fire suppression leads to transition from herbaceous to woody plants. Across the western portions of the United States and Canada, including 15 studies in the Sierra Nevada, prescribed fire and tree-cutting generally increased disturbance-dependent native species in the short-term and total understory abundance in the long term (Abella and Springer 2015). Nonetheless, disturbance provides an opportunity for spread of invasive species.

Preserving substrates that already contain visible fungi, lichen, bryophytes, and biological crusts will help maintain diversity (Lehmkuhl et al. 2004b). Denser patches within open forests have reduced exposure to light and wind, which generally is more favorable to these taxa. Management to retain large trees and aggregated tree distributions will provide a range of more protected conditions within open forests.

Invertebrates

Every ecosystem and disturbance will result in a unique assemblage of invertebrates, and open pine forests and prescribed fire are important for maintaining local populations of species in the landscape. Fundamental biological or ecological information is needed for most recorded species or taxonomic groups; in particular, information about segmented worms, snails, and slugs is sparse (James 2000). Invertebrates comprise the greatest amount of animal biomass and provide a variety of ecosystem services.

Most research on arthropods focuses on harmful insects, such as defoliators or those that can penetrate bark of live trees, primarily woodborers (Coleoptera: Buprestidae, Cerambycidae) and bark beetles (Coleoptera: Scolytidae), or stream invertebrates rather than terrestrial biodiversity. Nonetheless, a variety of insect predators, including spiders, ants, true bugs, lacewings, snakeflies, beetles, flies, and wasps can control forest pests (Niwa et al. 2001). Mammals and birds use insects as a food source. Small vertebrates, such as birds and bats, depend on insects as a protein source for their young during spring, while a variety of species consume fruits and seeds following pollination. Trees killed by insects may be more attractive to birds than artificially created snags (Farris and Zack 2005).

Species composition changes among ecosystems and species richness and abundance also may vary (Niwa et al. 2001; Stephens and Wagner 2006). The composition and abundance of arthropod communities differ between ponderosa pine and white fir (*Abies concolor*) fallen logs and stumps in Northeastern California (Koenigs et al. 2002). Ground beetle species richness was greater in thinned and burned ponderosa pine forests than in unmanaged or thinned-only stands in Northern Arizona, likely due to greater understory plant abundance; however, wildfire-burned stands had greatest species richness (Villa-Castillo and Wagner 2002). During later sampling at the same location, species richness of ground and darkling beetles was greatest after wildfire (Chen et al. 2006). Thinning, burning, and wildfire produced different assemblages of carabids and tenebrionids; six genera were indicators of burning, but generally varied in prominence by year (Chen et al. 2006). Additionally, ant functional groups varied among the treatments, and particularly were different in the wildfire stands (Stephens and Wagner 2006). Beetle community composition changed after thinning and burning in mixed-conifer forests in the Sierra Nevada, but remained diverse and abundant with the addition of rare species (Apigian et al. 2006).

Short-term (less than 2 years) studies after disturbance may detect temporary richness and abundance decreases that will no longer be present in longer-term studies after vegetation and assemblages change; nonetheless, some species will respond positively and rapidly. For example, 1 year after prescribed fire, following 1 year after silvicultural disturbance, overall mite abundance declined in burned plots, although some species increased in abundance in the Black Mountain Experimental Forest on the Lassen (Camann et al. 2008). Two years after prescribed fire in the Black Mountain Experimental Forest, mite abundance was much greater in burned plots, while spider, beetle, and springtail abundance remained lower in burned plots (Gillette et al. 2008). Six spider species or species groups were indicator groups for burns; postburn specialists generally were active mobile hunters such as cursorial spiders (Gnaphosidae), wolf spiders (Lycosidae), and crab spiders (Thomisidae) (Gillette et al. 2008).

Pollinators include bees, wasps, and ants (Hymenoptera), butterflies and moths (Lepidoptera), flies (Diptera), and some beetles (Coleoptera). Pollinators depend on flowering plant abundance and diversity, which are present in open pine forests. Native bees primarily are solitary; bees that nest in wood rely on cavities made by boring beetles (Niwa et al. 2001). In Arizona, butterfly species richness and abundance were greater in thinned and burned ponderosa pine-oak stands than in untreated plots (Waltz and Covington 2004). Egg and larval densities of a defoliator, pandora moth (*Coloradia pandora*), were the same in thinned and untreated ponderosa pine stands in Central Oregon, but adult emergence and egg hatch occurred 7 to 10 days earlier in thinned stands (Ross 1995). Greater butterfly species richness occurred in forest burns, fuel breaks, and riparian burns than in unburned controls in Yosemite National Park and the Rogue River National Forest (Huntzinger 2003). Regardless of food resources, light, moisture, and heat availability may be preferred for basking by butterflies (Waltz and Covington 2004) (fig. 4.1.2), and similarly may influence aggregations of bark arthropods (Rall 2006).

Open forests that contain a diverse floral community and tree structural features, including cavities, support a diverse arthropod assemblage. In the short term, disturbances will cause mortality and reduce resources such as food, cover, and nest substrates. Prescribed burns and silvicultural disturbances can be timed to encourage flowering plant diversity and to occur when insects are mobile and not during larval or pupal stages, if possible, to limit mortality (Nyoka 2010). Insecticides to control insect outbreaks and exotic invasive insects are other research and management concerns (Niwa et al. 2001). Overgrazing may reduce plant species richness, and thus insect species richness (Niwa et al. 2001). Exotic invasive plants also will reduce native insect species richness, although some species provide floral resources for pollinators (Niwa et al. 2001). Best-management practices have been developed for pollinators (USDA and USDOI 2015).

Invertebrates are ideal for research due to representation by many species with large populations along with short generation times, high dispersal rates, and rapid population growth that produce fast response to change. Insect outbreaks and consequent treatments may provide unique research opportunities, particularly to study reproduction. Pollinator decreases have become an emphasis for current and future research, both to understand poorly studied basic ecology, population dynamics, and foraging preferences and to identify conditions that are causing declines.

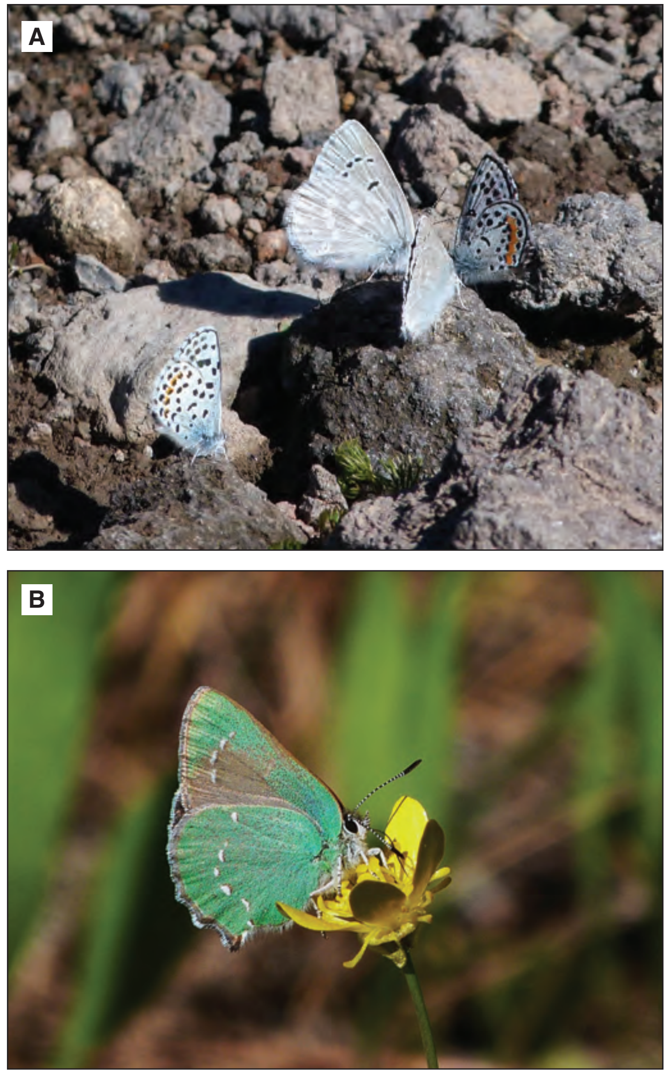

Figure 4.1.2—Butterflies, including these (A) diminutive common “blues” (*Lycaenidae*), prefer habitat that provides them moist, warm, and sunny locations. Such habitats are often associated with riparian zones and proper management can encourage suitable habitat for a wide range of flora and fauna, including (B) this green Sheridan’s Hairstreak (*Callophrys sheridanii*) (photos by Tony Kurz, used with permission).

Vertebrates

Many vertebrate species take advantage of food resources provided by open pine forests, consistently selecting large-diameter trees and snags, while some species use nearby cover from denser patches of trees (Bunnel 2013; Pilliod et al. 2006). Open pine forests with large-diameter trees and an herbaceous understory support different communities than closed conifer forests or pine forests with denser, small-diameter trees and non-pine ingrowth filling under the canopy (Bock and Block 2005; Finch et al. 1997; George et al. 2005; Wisdom et al. 2000). Western forests are dominated by conifers, and thus, food resources from diverse understory vegetation likely provide for a greater number of species than an understory of tree regeneration. Understory presence of non-coniferous vegetation with seeds and fruits in open forests is a valuable resource for consumers, such as arthropods, small mammals, birds, and ungulates, which in turn support predators. Open midstories allow maneuverability for birds to prey on small mammals or insects present in the herbaceous understory of open forests. Aerial insectivores, such as olive-sided flycatchers (*Contopus cooperi*) and western bluebirds (*Sialia mexicana*), and some bats, such as hoary bats (*Lasiurus cinereus*), use openings for foraging (Wisdom et al. 2000).

Large-diameter trees, large snags, and rotten holes in live large trees provide habitat for a variety of vertebrates. About 25 to 50 percent of vertebrate species may use snags for nesting, resting, and foraging (Hutto 2006). Stands with large pines, in some cases with open midstories, are preferred foraging sites for raptors, while adjacent denser stands, again with large trees, supply nesting and roosting sites (Palladini 2007; see section on flammulated owls below).Woodpeckers are strong primary excavators that supply cavity sites for nests, dens, and roosts, primarily for birds and bats (Bunnell 2013) (fig. 4.1.3). Woodpeckers select large-diameter trees that are greater than necessary for cavities, indicating tree age at which rot develops (Bunnell 2013). Additionally, sapwood decay, probably increased by beetle and woodpecker activity, may be necessary for snag selection by woodpeckers; ponderosa pine often is a preferred tree species for excavation, in part

Figure 4.1.3—Woodpeckers, such as this male black-backed woodpecker, are primary excavators of cavities in living (but having stem rot) and dead trees. After use by the woodpeckers, these cavities are used by a variety of species, including mountain bluebirds (*Sialia currucoides*) and California bats (*Myotis californicus*) (photo by Martin Tarby, used with permission).

due to containing a relatively large volume of sapwood (Farris and Zack 2005).

Reptiles use exposed environmental conditions of open dry pine forests, with cover from rocks and scree or logs, and some species particularly prefer postfire conditions (Germaine and Germaine 2003; Pilliod et al. 2006; Wisdom et al. 2000). For amphibians, moist conditions from aggregated trees, coarse woody debris, tree litter, or embedded wetlands are necessary in open forests (Pilliod et al. 2006). As with other taxa, reptiles and amphibians associated with open forests may experience mortality and short-term declines with burns and thinning (Pilliod et al. 2006).

Some mammal species also prefer open forests, large-diameter trees, and herbaceous vegetation that supports abundant insect and ungulate populations. Bats tend to select large-diameter trees for roosting, near a dense clump of large-diameter trees (Lacki and Baker 2007). If large trees and snags are available, then treatments to open up forests maybe be positive for bats, such as the hoary bat, by providing greater abundance of insects and space to maneuver (Pilliod et al. 2006; Wisdom et al. 2000). Eight small-mammal species trapped in ponderosa pine forests in Arizona responded positively to different stand elements (i.e., large trees and snags particularly, varying canopy cover, woody debris) present in restoration treatments (Kalies et al. 2012). Total abundance of small mammals generally increases after any type of fuel reduction (Converse et al. 2006). Some large carnivores forage in open forests, den in large hollows, and avoid human-caused mortality by using roadless areas (Wisdom et al. 2000; Witmer et al. 1998).

Coarse woody debris also is used by some vertebrates, but conversely, tree debris will act as a barrier to establishment of herbaceous vegetation, which also is a valuable resource to vertebrates. The amount of coarse woody debris in dry pine forests will be less than in closed forests because frequent fire will consume dry fuels. However, some patchy distribution of tree debris may remain or develop over time.

Even though some species remain present in dense mixed-conifer forests, thinning and prescribed burning of ponderosa pine and dry mixed-conifer forests benefit passerine birds and small mammals (Fontaine and Kennedy 2012; Kalies et al. 2010). High-severity fire produced strong positive and negative effects, indicating the importance of fire to bird species (Fontaine and Kennedy 2012). Rall (2006) connected increases in light availability and presence of large trees with abundance of arthropods and bark-gleaning bird species in Northern California dry pine forests. White-headed woodpecker (*Picoides albolarvatus*), brown creeper (*Certhia americana*), and white-breasted nuthatch (*Sitta carolinensis*), species that select large live trees for foraging, generally increased in abundance along with abundance of preferred bark arthropods in stands treated by thinning and prescribed fire (Rall 2006). If large trees and snags are retained, aerial, ground, and bark insectivores, granivores, and ground, tree, and cavity nesters generally favor stands that have been burned (Finch et al. 1997; Pilliod et al. 2006), but guild responses may vary (e.g., Gaines et al. 2007). Limited road densities and fuelwood restrictions will help reduce snag removal (Wisdom et al. 2000).

Black-backed Woodpecker

The distribution of black-backed woodpeckers generally encompasses coniferous forests of the Western United States, excluding the Central and Southern Rocky Mountains. In California, black-backed woodpeckers are residents in the Warner Mountains of Modoc and Lassen Counties, along with the Siskiyou and Sierra-Cascade Mountains (Tremblay et al. 2016). Black-backed woodpeckers migrate short distances, but periodically move from their resident distribution to follow beetle outbreaks in recently burned forests (fig. 4.1.4). Black-backed woodpeckers primarily eat larvae of wood-boring beetles (Cerambycidae and Buprestidae), engraver beetles (Curculionidae), and mountain pine beetles (*Dendroctonus ponderosae*) in dead or dying trees after severe fires (Tremblay et al. 2016). Black-backed woodpeckers typically arrive within a year after a severe fire and remain for up to 10 years, with densities peaking about 2 to 5 years after fire and coinciding with beetle life cycles (Tremblay et al. 2016). Young birds colonize burned locations and generally reside there for their lifespans (Siegel et al. 2015). Black-backed woodpeckers also will use unburned forests with dense patches of snags due to beetle outbreaks, albeit at lower densities than in burned forests (Fogg et al. 2014).

Snags provide foraging, nesting, and roosting sites for black-backed woodpeckers (fig. 4.1.5). Black-backed woodpeckers are associated with high densities of fire-created large-diameter snags, primarily ponderosa pine, Jeffrey pine (*Pinus jeffreyi*), Douglas-fir, lodgepole pine

Figure 4.1.4—Black-backed woodpeckers, like this foraging male, consume a variety of insects associated with trees, especially bark beetles and wood borers (photo by Martin Tarby, used with permission).

(*Pinus contorta*), and true firs (*Abies* species) (Tremblay et al. 2016). Most black-backed woodpecker detections in the Sierra Nevada and Cascades occur in forests of 87 or more square feet per acre (20 m^2/ha) basal area of snags of 9 inches (23 cm) diameter or larger at breast height that burned within the previous 7 years (Odion and Hanson 2013). In Oregon and California, black-backed woodpeckers selected larger-diameter trees (about 16 inches [40 cm] in diameter) for nesting (Tremblay et al. 2016). In general, black-backed woodpeckers have a home range of 247 to 988 acres (100 to 400 ha) (Siegel et al. 2013).

Limited snag creation and retention due to fire suppression, salvage logging after fire, and snag removal have decreased habitat, abundance, and nesting success of black-backed woodpeckers in California (Hanson and

Figure 4.1.5—Snags, especially those of large-diameter trees, provide important foraging, nesting, and roosting sites for a variety of animals, including black-backed woodpeckers (photo by Eric E. Knapp, Forest Service).

North 2008). Earth Island Institute et al. (2012) estimated that fewer than 1,000 black-backed woodpecker pairs may remain in California and the Eastern Oregon Cascades. Black-backed woodpeckers were proposed as a candidate species under the Endangered Species Act. In 2017, the U.S. Department of the Interior, U.S. Fish and Wildlife Service, determined the Oregon–Cascades/California population was not a listable entity under the Act as it did not meet the criteria under the Distinct Population Segment Policy (USFWS 2017). Black-backed woodpeckers are a management indicator species of snags in burned National Forests in the Sierra Nevada.

In summary, black-blacked woodpecker populations may shift in space and increase quickly following fire; therefore, studies that examine populations over large spatial and temporal scales to determine trends would be valuable. Attributes to be examined should include variation in snag density, diameter, and basal area necessary to maintain woodpecker populations.

Figure 4.1.6—Flammulated owls prefer forests that have large-diameter trees with cavities in which they can breed, and open canopy structure that facilitates their hunting of anthropods, especially large insects (photo by Nigel Voaden, used with permission).

Flammulated Owl

The flammulated owl has a wide geographic range throughout mid-elevation montane zones of the Western United States, including the Cascades and Sierra Nevada. Flammulated owls eat insects, particularly nocturnal arthropods, including owlet and geometrid moths (Noctuidae and Geometridae), crickets and grasshoppers (Orthoptera), and beetles (Coleoptera) (Linkhart and McCallum 2013). Where winter temperatures limit arthropod abundance, flammulated owls are long-distance neotropical migrants to Mexico and perhaps Central America (Linkhart and McCallum 2013). Flammulated owls select primarily open ponderosa pine forests or similar open structure in pine, fir, or aspen forests, often with an oak component, in cool, semiarid climates (Linkhart and McCallum 2013) maintained by low-severity surface fires. Open forests allow light onto the forest floor, supporting an herbaceous understory and associated arthropods that owls hunt during night. Owls roost in denser foliage around large live trees (e.g., diameter of 21 inches [55 cm] or greater) and singing males also frequently use large trees (mean of 17.5 inches [45 cm]) (Linkhart and McCallum 2013). The flammulated owl nests in woodpecker-created cavities (fig. 4.1.6), which tend to be in larger-diameter snags (mean 21 inches [54 cm]) (Bunnell 2013). In Colorado, during a long-term study, productivity and years of territory occupancy by breeding pairs were greater in open stands of large-diameter ponderosa pine or Douglas-fir, compared to dense, small-diameter stands of Douglas-fir (Linkhart and Reynolds 2006). Flammulated owls were not present after removal of snags in Arizona (Franzreb and Ohmart 1978) and removal of mature ponderosa pines and snags in California (Marshall 1988).

While flammulated owls may be abundant, information about population dynamics is limited to areas in Colorado and New Mexico (Linkhart and McCallum 2013). Flammulated owls have a low annual rate of reproduction, with delayed breeding by males for at least 1 year, small clutch sizes, and lack of second clutches (Linkhart and McCallum 2013). Although flammulated owls do use nest boxes, these owls may not nest in boxes as readily as other small owls (Linkhart and McCallum 2013).

Research needs include population estimates to closely monitor demography because populations will have difficulty recovering from declines in survivorship and productivity given low reproduction rates. Pollinator declines, due to a range of factors including insecticides, may affect prey availability, particularly of moths. Climate change may produce phenological mismatches between peak abundance of prey on the breeding grounds and long-distance migration of flammulated owls. Further study may be needed to determine optimal retention of larger-diameter trees and snag.

Gray Wolf

Generally extirpated from the United States by the 1930s, gray wolves re-colonized and bred in Northern Montana around Glacier National Park in 1986 (Ream et al. 1989), and the small population grew steadily. Additionally, wolves were reintroduced to Yellowstone National Park and Central Idaho during 1995–1996. In the Northern Rocky Mountains and Pacific Northwest, current range includes parts of Idaho, Montana, Wyoming, Oregon, and Washington, with an estimated population of at least 1,900 wolves (USFWS et al. 2016). These populations increased quickly, resulting in transfer to State management and legalization of hunting. The western population remains stable, with most mortality related to human causes (USFWS et al. 2016; Witmer et al. 1998). About 200 wolves were estimated in Oregon and Washington during 2015.

The legal status of gray wolves in the United States is complex and changing, affecting management (Kovacs et al. 2016). Gray wolves acquired Federal protection under the Endangered Species Preservation Act of 1966, which was renewed in 1973 under the Endangered Species Act (Creel and Rotella 2010). The current status of wolves varies by location and date. Stable and expanding populations have been down-listed from endangered to threatened or delisted as demographic criteria are met. Delisting generally results in legal appeal because when legal authority for wolf management switches from the U.S. Fish and Wildlife Service, State agencies may allow public hunting and predator control, which may exceed desired quotas and encourage poaching (Creel and Rotella 2010).

Wolves in California remain federally endangered. The last known wolf in California before extirpation was killed in Lassen County in 1924 (Kovacs et al. 2016). During 2011, a male wolf (OR-7) left Northeastern Oregon and entered California. A breeding pair occurred in Siskiyou County during 2015, (Kovacs et al. 2016) and in Lassen County during 2017 (fig. 4.1.7) and 2018 data from the California Department of Fish and Wildlife (https://www.wildlife.ca.gov/conservation/mammals/gray-wolf; accessed August 2018) showed known resident wolf territories included a portion of Southwestern Lassen County and Northcentral Plumas County (fig. 4.1.8). As populations of gray wolves stabilize and expand north of California, dispersing individuals and establishment of wolf packs and breeding pairs are anticipated to become more common in California. Historical gray wolf range included the Southern Cascades, Modoc Plateau, the Sierra Nevada, the western slope of the Sierra Nevada foothills and mountains, and the Klamath Mountains (Kovacs et al. 2016). Despite proximity to dispersing wolves, the Lassen and Modoc may provide less favorable habitat than other Sierra Nevada and Southern Cascade national forests, based on various predictors including land ownership, human density, and prey density (Antonelli et al. 2016). Nonetheless, lands administered by the U.S. Department of Agriculture, Forest Service and the U.S. Department of the Interior, Bureau of Land Management provide an opportunity for management at larger landscapes to support wolves. Limited road access will reduce human access and human-caused mortality, while a combination of forage and cover for deer and elk also will provide suitable wolf habitat. Management to reverse the process of fire exclusion, which allowed replacement of herbaceous vegetation and fire-tolerant shrubs with trees, and managed use of vegetation by livestock will increase forage for native ungulates. Open pine forests, mountain meadows and aspens, and oak woodlands are vegetation types that provide a combination of forage and cover.

Dispersing wolves in California prompted listing of the gray wolf as endangered under the California Endangered Species Act (CESA). Under both Federal and State ESA, “take” of a wolf is prohibited except under limited circumstances. California Department of Fish and Wildlife and stakeholders also developed a gray wolf plan, which does not propose demographic criteria to warrant delisting, given minimal science to base decisions of a sustainable population in California (Kovacs et al. 2016). In other Western States, wolves appear to be established as a resident species after 4 breeding pairs were confirmed for 2 successive years, while 8 breeding pairs for 2 successive years is another stability threshold (Kovacs et al. 2016). In addition to conserving a biologically sustainable populations of wolves, goals include managing native ungulate populations, minimizing livestock losses, and communicating science-based information to the public (Kovacs et al. 2016). Under a cooperative agreement with the U.S. Fish and Wildlife Service, California Department of Fish and Wildlife is the lead agency for investigating wolf reports, including livestock depredations, and capturing wolves to monitor activity, (Kovacs et al. 2016). California Department of Fish and Wildlife will work cooperatively with U.S. Fish and Wildlife Service to implement non-lethal wolf control actions and provide relevant information to reduce potential conflicts with livestock producers and other landowners in the local area (Kovacs et al. 2016). The U.S. Fish and Wildlife Service

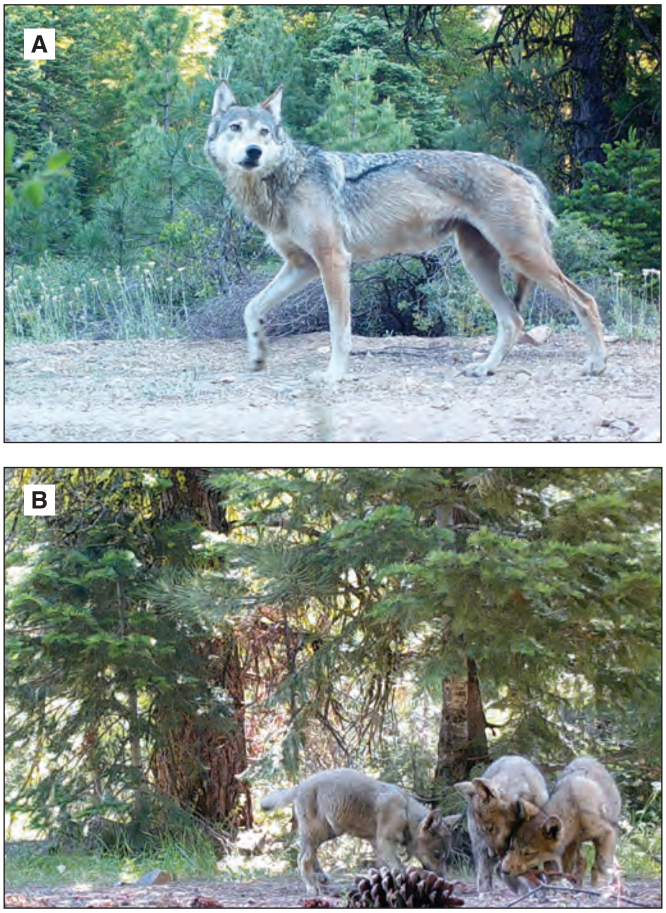

Figure 4.1.7—Trail camera photos of (A) a female gray wolf in Lassen County in 2017 and (B) three of her pups (photos by Tom Rickman, Forest Service).

Figure 4.1.8—Gray wolves continue to expand their range across the Western United States, including Northeastern California, where they are known to have resident territories as of May 2018 (adapted from "Wolf Activity Map" https://www.wildlife.ca.gov/conservation/mammals/gray-wolf; accessed August 2018).

will authorize take of federally listed species and investigate cases that involve unauthorized take of a federally listed species.

Ungulates generally make up the bulk of wolf diets, and wolves may affect ungulate population densities and behaviors. Where ungulate densities are high, wolves may alleviate ungulate impacts on rangelands and wetlands by controlling ungulate populations, thus preventing excessive browsing of vegetation (Leopold et al. 1947). Predation of elk appears to have allowed recovery of depleted deciduous woody plants in riparian areas of Yellowstone National Park (Beschta and Ripple 2016), which benefits aquatic invertebrates and fish, insects, bats, birds, and mammal species, such as beaver (Beschta and Ripple 2016). In areas where prey populations are at or near carrying capacity, wolf predation may be compensatory, a replacement for starvation or disease. However, compared to other States, California has fewer elk, which have not recovered historical abundance and range since near extirpation, and a long-term declining trend in the deer population, due to factors such as reduced habitat and forage and perhaps extreme weather (Kovacs et al. 2016). Wolves may reduce small, increasing populations of elk and decreased deer populations, reversing progress to support and re-establish elk and deer populations (Kovacs et al. 2016). Wolves also will prey on wild horses (Webb et al. 2008). As wolves become established, scientific information will become available about wolf-ungulate interactions in California.

Wolves also may affect abundance of subordinate predators. Coyotes particularly overlap ecologically with wolves, and coyotes typically increase without wolf presence and decrease with wolf presence (Kovacs et al. 2016). Reduced coyote populations, in turn, release populations of smaller animals, such as foxes. Wolf presence may help sensitive species and federally designated threatened and endangered vertebrate species, including the Sierra Nevada red fox (*Vulpes nulpes necator*).

Wolves appear to rarely interact directly with humans. Wolf attacks may be due to disease; starvation; defense of territory, den sites, and pups; habituation; food guarding; or when trapped (Kovacs et al. 2016). Domestic dogs away from human presence are vulnerable to attack (Kovacs et al. 2016). Strategies to avoid wolf and domestic dog conflicts are similar to those to avoid conflict with other wildlife species. Wolves regularly encounter domesticated animals, which may result in conflict with humans. Wolves, including the Lassen pack, already have targeted livestock as prey in California. The Cascade Range and Modoc Plateau areas contain relatively high beef cattle and sheep densities. Livestock owners and hunters may experience economic or other losses from damage to livestock or hunting dogs and perhaps reduced ungulate hunting opportunities, and may be uninterested in practicing non-lethal measures compared to lethal control to coexist with wolves.

Nonetheless, scientists representing the American Society of Mammalogists summarized the need for non-lethal management, according to the best current ecological, social, and ethical scholarship in a 2017 special issue of the *Journal of Mammalogy*. Bergstrom (2017) stated that from the "perspectives of ecology, wildlife biology and management, social science, ethics, and law and policy showing that non-lethal methods of preventing depredation of livestock by large carnivores may be more effective, more defensible on ecological, legal, and wildlife-policy grounds, and more tolerated by society than lethal methods…" Lethal control of other species, such as coyote, has been unsuccessful overall at preventing livestock depredation (Bergstrom 2017). Indeed, predator removal may even increase livestock depredation by increasing the number of subadults who have fewer skills and may need to take greater risks than adults. When locally successful, loss of a predator may release prey populations, resulting in vegetation suppression and overuse. Nonselective lethal control methods, such as traps and poison baits, often kill unintended species, including those that may be threatened and endangered or candidate species (Bergstrom 2017), and pose a risk to young children and pets. In addition, allowing lethal management encourages a culture of poaching (Chapron and Treves 2016). To avoid lethal wolf control by private citizens, governmental agencies and nongovernmental organizations provide assurance to livestock owners through compensation funds or tax credits for proven and probable livestock depredation by wolves. Private and State agencies paid $504,000 in compensation during 2015 (USFWS et al. 2016). In contrast, domesticated but free-roaming dogs cost $5 million in livestock depradation in Texas alone (Pimentel et al. 2005).

Non-lethal control strategies can be effective at many scales with use of "integrated predator management," such as herd management and deterrents (Smith et al. 2014; Stone et al. 2016, 2017). Many guides are available with recommendations including: increase human (i.e., range riders) or guardian presence, particularly during vulnerable times such as calving and presence of young calves; condense calving season and confine cattle during calving and while calves are young; provide nighttime corrals; reduce food attractants such as carcasses and sick livestock and afterbirth; implement "mob" grazing at higher stocking with frequent pasture changes; mix age classes to add presence of experienced animals to naïve yearlings and include individuals of more vigilant breeds; electrify or place flagging (i.e., fladry) on fencing or portable barriers; place bells on livestock; and employ shifting scare tactics such as varying siren and light arrays (with effectiveness rating by Smith et al. 2014). Deterrents are best used in combination to prevent wolf habituation, and proactively instead of reactively after problems occur (Smith et al. 2014). Stone et al. (2017) applied a variety of non-lethal deterrents and animal husbandry techniques, deployed depending on location and season, to protect sheep on public grazing lands in Idaho for 7 years. Sheep depredation losses to wolves were 0.02 percent of the total number of sheep present in protected areas, whereas losses were 3.5 times greater in nonprotected areas where wolves were lethally controlled.

Smith et al. (2014) and Stone et al. (2016) suggest that learning how to coexist with wolves is an active process that requires problem-solving techniques, adaptive management, and knowledge of how resource use by wolves in space and time overlaps with location and timing of husbandry practices. Separation of livestock from wolf core areas during periods of intense activity is one of the leading factors in reducing wolf depredations (Smith et al. 2014). Certain locations and seasons increase risk, such as proximity to active elk grounds and during calving season; switching grazing schedules by location and season may be necessary to avoid associated wolf activity (Smith et al. 2014).

Monitoring of animal numbers is critical to quickly recognize a problem and adjust techniques to resolve wolf-livestock conflicts before conflicts escalate (Stone et al. 2016). When done frequently, records of interactions and related wolf observations in concert with evaluation of livestock protection strategies can inform managers to refine techniques and prevent a few wolves or wolf packs from becoming chronic livestock hunters (Smith et al. 2014; Stone et al. 2016). Northern California livestock producers may be interested in first trying the coexistence strategies of attractant removal and range riders (Antonelli et al. 2016).

Wolf management in California will depend on current legal and population status, scientific information specific to California, cooperation among Federal and California State agencies, and communication and cooperation with the public, particularly invested stakeholders, about non-lethal techniques to reduce predation of livestock. Wolf management also includes interactions with agencies outside California that have managed dispersing and then breeding wolves and which share management for wolves in the Western United States. Outreach efforts to prepare for coexisting with wolf presence include personal contact with grazing permittees, neighboring ranchers, and hunters to provide assistance with non-lethal deterrents and strategies, formal presentations to community and stakeholder groups, interviews with news media outlets, and webpages supplying information about wolf management (e.g., see studies listed by Smith et al. 2014). Similarly, collaboration among Federal and State agencies, livestock producers, county extension services and agricultural commissioners, nongovernmental organizations, and other stakeholders may help minimize wolf-livestock conflicts and resolve conflicts between ranchers and wolf advocates. For example, wolf management advisory panels, comprised of different stakeholders, can clarify contentious viewpoints and facilitate coexistence solutions (Mazur and Asah 2013). Also, Stone et al. (2016) suggest written agreements that clearly define expected roles and responsibilities, along with systems to evaluate effectiveness of non-lethal methods to reduce wolf-livestock interactions under varying situations. Federal, State, and tribal agencies spent about $6.43 million on wolf management, outreach, monitoring, and research during 2015 (U.S. Fish and Wildlife Service et al. 2016).

Summary

Dry pine forests are extensive and have not been as intensively studied for biodiversity as closed-canopy forest ecosystems. Most research has focused on the role of fire on tree composition and fuels, whereas very little research has been published on associated lichen, fungi, mosses, forbs, grasses, and arthropods, and even most vertebrate research is limited. Research in dry pine forests consequently is more similar in stage to the previous Northwest Forest Plan Science Synthesis, in which Marcot and Molina (2006) concluded that information was needed on basic occurrence, distribution, and ecology of rare and poorly known species of late-successional and old-growth Douglas-fir and western hemlock forests. We anticipate that bat and pollinator research will increase during the next decade, given emphasis on reversing declines. Information gaps in published research likely indicate a need for greater monitoring and development of survey designs for some taxa.

Nonetheless, a starting point and major task that will benefit biodiversity of dry pine forests includes managing a range of densities and large-diameter trees at landscape scales. While no conditions will benefit all species, dry pine forests support a subset of species that use forests with a relatively open midstory and a range of overstory canopy, which provide a full spectrum of conditions between grasslands and closed forests. Thinning from below, with large-tree retention, to variable densities in combination with prescribed fire will develop structure, function, composition, and connectivity of open pine forests.

Fire is important to control tree regeneration and has maintained open pine forests for thousands of years. Low-severity fire was an historical disturbance in dry pine forests. Additionally, black-backed woodpeckers, in particular, use large-diameter trees damaged by more severe fires. Species of dry pine forests will be able to tolerate fire as part of the open forest ecosystem, although spring burns will produce greatest mortality of young birds. Burning and mechanical treatments timed around breeding seasons, if possible, will minimize mortality. Although management by burning and thinning may have short-term consequences measured in animal mortality, there are greater consequences to losing open forest conditions. Invasive species may spread, if they are not monitored and controlled, due to stand entry.

In addition to open forest structure, common elements are required by many species of dry pine forests: (1) large trees and snags, including living trees with rotten areas for nests and dens; (2) understory vegetation of grasses and flowering plants, which provide fruits and seeds; (3) and a range of tree density, including aggregated trees and large patches of dense trees for escape cover, nesting and resting, and wetter or more protected conditions. Retention of live and dead trees larger than the median diameter at which wood rot commonly develops is important for many species (Bunnell 2013). Black-backed woodpeckers specifically may require high densities of large-diameter trees at landscape scales to provide high densities of large-diameter snags after high-severity fire. Management activities to minimize removal of snags, especially large-diameter snags, may be warranted

(Wisdom et al. 2000). Coarse woody debris, while valuable, should not be as great a focus as development of understory herbaceous vegetation.

Due to the large land bases of the Lassen and Modoc, these National Forests contribute to landscape ecosystem management, which is important to other Federal and State agencies, particularly for wolf management. Because elk herds are newly established and deer are declining in California, wolf predation may be an additional cause of mortality. Management of dry pine forests to reduce tree densities and encourage understory herbaceous vegetation, and some fire-tolerant shrub cover, will help support ungulate populations, which in turn will help support wolf populations. In addition to wolf-ungulate interactions, wolves prey on domestic animals, as do dogs that cause millions of dollars in livestock losses annually (Pimental et al. 2005). Wolf-livestock conflicts can be minimized proactively through coexistence strategies including non-lethal deterrence methods. Wolves are reflective of most large carnivores, which decline in numbers or range due to interactions with humans. To reverse these trends, management needs include maintaining large land areas with limited road (i.e., human) access and new approaches to reduce conflicts with humans (Witmer et al. 1998). If not already in place, plans and programs to (1) manage and monitor wolves, including identifying core areas with limited road access, (2) monitor ungulate response to wolves, (3) educate visitors and hunters about recreating with wolves, (4) assist and form agreements with grazing permittees for non-lethal coexistence practices, (5) develop and evaluate new coexistence strategies, and (6) coordinate with other agencies and organization will help the process of wolf reestablishment in California.

References

Abella, S.R.; Springer, J.D. 2015. Effects of tree cutting and fire on understory vegetation in mixed conifer forests. Forest Ecology and Management. 335: 281–299.

Antonelli S; Boysen, K; Piechowski, C; [et al.]. 2016. An analysis of wolf-livestock conflict hotspots and conflict reduction strategies in northern California. Santa Barbara, CA: University of California: Bren School of Environmental Science and Management. 116 p.

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. Forest Ecology and Management. 221: 110–122.

Bergstrom, B. 2017. Carnivore conservation: shifting the paradigm from control to coexistence. Journal of Mammalogy. 98: 1–6.

Beschta, R.L.; Ripple W.J. 2016. Riparian vegetation recovery in Yellowstone: the first two decades after wolf reintroduction. Biological Conservation. 198: 93–103.

Bigelow, S.W.; Manley, P.N. 2009. Vegetation response to fuels management in the Lake Tahoe Basin. In: Effects of fuels management in the Tahoe Basin: a scientific literature review. Davis, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station and the Tahoe Science Consortium. https://www.fs.fed.us/psw/partnerships/tahoescience/fuel_management.shtml (3 Dec. 2017).

Bock, C.; Block, W.M. 2005. Fire and birds in the southwestern United States. In: Saab, V.; Powell, H., eds. Fire and avian ecology in North America. Studies in Avian Biology 30. Camarillo, CA: Cooper Ornithological Society: 14–32.

Bond, M.L.; Siegel, R.B.; Craig, D.L., eds. 2012. A conservation strategy for the black-backed woodpecker (*Picoides arcticus*) in California. Version 1.0. Point Reyes Station, CA: The Institute for Bird Populations and California Partners in Flight. 131 p.

Bunnell, F.L. 2013. Sustaining cavity-using species: patterns of cavity use and implications to forest management. ISRN Forestry. 2013: 457698.

Camann, M.A.; Gillette, N.E.; Lamoncha, K.L.; [et al.]. 2008. Response of forest soil Acari to prescribed fire following stand structure manipulation in the southern Cascade Range. Canadian Journal of Forest Research. 38: 956–968.

Chapron, G.; Treves, A. 2016. Blood does not buy goodwill: allowing culling increases poaching of a large carnivore. Proceedings of the Royal Society B: Biological Sciences. 283: 20152939.

Chen, Z.: Grady, K.; Stephens, S.; [et al.]. 2006. Fuel reduction treatment and wildfire influence on carabid and tenebrionid community assemblages in the ponderosa pine forest of northern Arizona, USA. Forest Ecology and Management. 225: 168–177.

Claridge, A.W.; Trappe, J.M.; Hansen, K. 2009. Do fungi have a role as soil stabilizers and remediators after forest fire? Forest Ecology and Management. 257: 1063–1069.

Converse, S.J.; White, G.C.; Farris, K.L.; [et al.]. 2006. Small mammals and forest fuel reduction: national-scale responses to fire and fire surrogates. Ecological Applications. 16: 1717–1729.

Creel, S.; Rotella, J.J. 2010. Meta-analysis of relationships between human offtake, total mortality and population dynamics of gray wolves (*Canis lupus*). PloS ONE. 5: e12918.

Earth Island Institute; the Center for Biological Diversity; the Blue Mountains Biodiversity Project; and the Biodiversity Conservation Alliance. 2012. Petition to list the black-backed woodpecker (*Picoides arcticus*) as threatened or endangered under the federal endangered species act. https://www.biologicaldiversity.org/species/birds/black-backed_woodpecker/pdfs/BBWO_FESA_Petition.pdf (16 Apr. 2020).

Farris K.L.; Zack, S. 2005. Woodpecker-snag interactions: an overview of current knowledge in ponderosa pine systems. In: Ritchie, M.W.; Maguire, D.A.; Youngblood, A., tech. coords. Proceedings of the symposium on ponderosa pine: issues, trends, and management. Gen. Tech. Rep PSW-GTR-198. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 183–195.

Finch, D.; Ganey, J.L.; Yong, W.; [et al.]. 1997. Effects and interactions of fire, logging, and grazing. In: Block, W.M.; Finch, D.M., tech. eds. Songbird ecology in southwestern ponderosa pine forests: a literature review. Gen. Tech. Rep. RM-292. Fort Collins, CO: U.S. Department. of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 103–136.

Fogg, A.M.; Roberts, L.J.; Burnett, R.D. 2014. Occurrence patterns of black-backed woodpeckers in green forest of the Sierra Nevada Mountains, California, USA. Avian Conservation and Ecology. 9: 3.

Fontaine, J.B.; Kennedy, P.L. 2012. Meta-analysis of avian and small-mammal response to fire severity and fire surrogate treatments in U.S. fire-prone forests. Ecological Applications. 22: 1547–1561.

Franzreb, K.E.; Ohmart, R.D. 1978. The effects of timber harvesting on breeding birds in a mixed-coniferous forest. Condor. 80: 431–441.

Gaines, W.L.; Haggard, M.; Lehmkuhl, J.F.; [et al.]. 2007. Short-term response of land birds to ponderosa pine restoration. Restoration Ecology. 15: 670–678.

George, T.; Zack, S.; Laudenslayer, W.F. Jr. 2005. A comparison of bird species composition and abundance between late- and mid-seral ponderosa pine forests. In: Ritchie, M.W.; Maguire, D.A.; Youngblood, A., tech. coords. Proceedings of the symposium on ponderosa pine: issues, trends, and management. Gen. Tech. Rep. PSW-GTR-198. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 159–169.

Germaine, S.S.; Germaine, H.L. 2003. Lizard distributions and reproductive success in a ponderosa pine forest. Journal of Herpetology. 37: 645–652.

Gillette, N.E.; Vetter, R.S.; Mori, S.R.; [et al.]. 2008. Response of ground-dwelling spider assemblages to prescribed fire following stand structure manipulation in the southern Cascade Range. Canadian Journal of Forest Research. 38: 969–980.

Hanson C.T., North, M.P. 2008. Postfire woodpecker foraging in salvage-logged and unlogged forests of the Sierra Nevada. Condor. 110: 777-782.

Hardman, A.; McCune, B. 2010. Bryoid layer response to soil disturbance by fuel reduction treatments in a dry conifer forest. Bryologist. 113: 235–245.

Heinlen, E.R.; Vitt, D.H. 2003. Patterns of rarity in mosses of the Okanogan Highlands of Washington State: an emerging coarse filter approach to rare moss conservation. Bryologist. 106: 34–52.

Huntzinger, M. 2003. Effects of fire management practices on butterfly diversity in the forested western United States. Biological Conservation. 113: 1–12.

Hutto, R.L. 2006. Toward meaningful snag-management guidelines for postfire salvage logging in North American conifer forests. Conservation Biology. 20: 984–993.

James, S. 2000. Earthworms (Annelida: Oligochaeta) of the Columbia River basin assessment area. Gen. Tech. Rep. PNW-GTR-491. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 13 p.

Johnson, C.G.; Clausnitzer, R.R.; Mehringer, P.J.; [et al.]. 1994. Biotic and abiotic processes of eastside ecosystems: the effects of management on plant and community ecology, and on stand and landscape vegetation dynamics. Gen. Tech. Rep. PNW-GTR-322. Portland, OR: U.S. Department of Agriculture, Forest Service, Northwest Research Station. 76 p.

Jovan, S. 2008. Lichen bioindication of biodiversity, air quality, and climate: baseline results from monitoring in Washington, Oregon, and California. Gen. Tech. Rep. PNW-GTR-737. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 115 p.

Kalies, E.L.; Chambers, C.L.; Covington, W.W. 2010. Wildlife responses to thinning and burning treatments in southwestern conifer forests: a meta-analysis. Forest Ecology and Management. 259: 333–342.

Kalies, E.L.; Dickson, B.G.; Hambers, C.L.; [et al.]. 2012. Community occupancy responses of small mammals to restoration treatments in ponderosa pine forests, northern Arizona, USA. Ecological Applications. 22: 204–217.

Koenigs, E.; Shea, P.J.; Borys, R.; [et al.]. 2002. An investigation of the insect fauna associated with coarse woody debris of *Pinus ponderosa* and *Abies concolor* in northeastern California. In: Laudenslayer, W.J.; Shea, P.J.; Valentine, B.E.; [et al.], eds. Proceedings of the symposium on the ecology and management of dead wood in western forests. Gen. Tech. Rep. PSW-GTR-181. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 97–110.

Kovacs, K.E.; Converse, K.E.; Stopher, M.C. 2016. Conservation plan for gray wolves in California. Sacramento, CA: California Department of Fish and Wildlife. https://www.wildlife.ca.gov/conservation/mammals/gray-wolf (3 Dec. 2017).

Lacki, M.J.; Baker, M.D. 2007. Day roosts of female fringed myotis (*Myotis thysanodes*) in xeric forests of the Pacific Northwest. Journal of Mammalogy. 88: 967–973.

Lehmkuhl, J.F.; Gould, L.E.; Cazares, E.; [et al.]. 2004a. Truffle abundance and mycophagy by northern flying squirrels in eastern Washington forests. Forest Ecology and Management. 200: 49–65.

Lehmkuhl, J.F. 2004b. Epiphytic lichen diversity and biomass in low-elevation forests of the eastern Washington Cascade Range, USA. Forest Ecology and Management. 187: 381–392.

Leopold, A.; Sowls, L.K.; Spencer, D.L. 1947. A survey of over-populated deer ranges in the United States. Journal of Wildlife Management. 11: 162–177.

Lesher, R.D.; Derr, C.C.; Geiser, L.H. 2003. Natural history and management considerations for Northwest Forest Plan Survey and Manage lichens. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 218 p.

Linkhart, B.D.; McCallum, D.A. 2013. Flammulated owl (*Psiloscops flammeolus*). In: Rodewald, P.G., ed. The birds of North America. Ithaca, NY: Cornell Lab of Ornithology.

Linkhart, B.D.; Reynolds, R.T. 2006. Lifetime reproduction of flammulated owls in Colorado. Journal of Raptor Research. 40: 29–37.

Long, J.W.; Quinn-Davidson, L.N.; Skinner, C.N., eds. 2014. Science synthesis to support socioecological resilience in the Sierra Nevada and southern Cascade Range. Gen. Tech. Rep. PSW-GTR-247. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 712 p.

Marcot, B.G.; Molina, R. 2006. Conservation of other species associated with older forest conditions. In: Haynes, R.W.; Bormann, B.T.; Lee, D.C.; [et al.], eds. Northwest forest plan—the first 10 years (1994–2003): synthesis of monitoring and research results. Gen. Tech. Rep. PNW-GTR-651. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 145–292.

Marshall, J.T. 1988. Birds lost from a giant sequoia forest during 50 years. Condor. 90: 359–372.

Mazur, K.E.; Asah, S.T. 2013. Clarifying standpoints in the gray wolf recovery conflict: procuring management and policy forethought. Biological Conservation. 167: 79–89.

Niwa, C.G.; Sandquist R.E.; Crawford R.; Frest[et al.]. 2001. Invertebrates of the Columbia River basin assessment area. Gen. Tech. Rep. PNW-GTR-512. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 84 p.

Nyoka, S. 2010. Can restoration management improve habitat for insect pollinators in ponderosa pine forests of the American Southwest? Ecological Restoration. 28: 280–290.

Odion D.C.; Hanson C.T. 2013. Projecting impacts of fire management on a biodiversity indicator in the Sierra Nevada and Cascades, USA: The black-backed woodpecker. Open Forest Science Journal. 6: 14–23.

Palladini, M. 2007. The response of Cooper's hawks to experimental ponderosa pine forest restoration treatments in an adaptive management area. Arcata, CA: Humboldt State University. M.S. thesis.

Pilliod, D.; 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. Gen. Tech. Rep. RMRS-GTR-173 Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Pimentel, D.; Zuniga, R.; Morrison, D. 2005. Update on the environmental and economic costs associated with alien-invasive species in the United States. Ecological Economics. 52(3): 273–288.

Rall, C.J. 2006. The response of bark-gleaning birds and their prey to thinning and prescribed fire in eastside pine forests in northern California. Arcata, CA: Humboldt State University. M.S. thesis.

Ream, R.R.; Fairchild, M.W.; Boyd, D.K.; [et al.]. 1989. First wolf den in western U.S. in recent history. Northwestern Naturalist. 70: 39-40.

Reazin, C.; Morris, S.; Smith, J.E.; [et al.]. 2016. Fires of differing intensities rapidly select distinct soil fungal communities in a Northwest U.S. ponderosa pine forest ecosystem. Forest Ecology and Management. 377: 118–127.

Ross, D.W. 1995. Short-term impacts of thinning ponderosa pine on pandora moth densities, pupal weights, and phenology. Western Journal of Applied Forestry. 10: 91–94.

Siegel, R.B.; Tingley M.W.; Wilkerson, R.L.; [et al.]. 2013. Assessing home range size and habitat needs of black-backed woodpeckers in California: report for the 2011 and 2012 field seasons. Point Reyes Station, CA: The Institute for Bird Populations and U.S. Department of Agriculture, Forest Service, Pacific Southwest Region. 107 p.

Siegel, R.B.; Tingley M.W.; Wilkerson, R.L.; [et al.]. 2015. Age structure of black-backed woodpecker populations in burned forests. The Auk. 133: 69–78.

Smith, L.; Hutchinson, J.; DeNesti, L. 2014. Wolf-livestock non-lethal conflict avoidance: a review of the literature. Western Wildlife Outreach. https://wdfw.wa.gov/sites/default/files/2019-02/wolf_livestock_conflict_avoidance_literature_review_11_2014_final_submitted_version.pdf (3 Dec. 2017).

Spies, T.A.; Stine, P.A.; Gravenmier, R.; [et al.], tech. coords. 2018. Synthesis of science to inform land management within the Northwest Forest Plan area. Gen. Tech. Rep. PNW-GTR-966. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 1020 p. 3 vol.

Stephens, S.S.; Wagner, M.R. 2006. Using ground foraging ant (Hymenoptera: Formicidae) functional groups as bioindicators of forest health in northern Arizona ponderosa pine forests. Environmental Entomology. 35: 937–949.

Stone, S.A.; Breck, S.W.; Timberlake, J.; [et al.]. 2017. Adaptive use of non-lethal strategies for minimizing wolf–sheep conflict in Idaho. Journal of Mammalogy. 98: 33–44.

Stone, S.A.; Edge, E.; Fascione N.; [et al.]. 2016. Livestock and wolves: a guide to non-lethal tools and methods to reduce conflicts. Washington, DC: Defenders of Wildlife.

Tremblay, J.A.; Dixon, R.D.; Saab V.A.; [et al.]. 2016. Black-backed woodpecker (*Picoides arcticus*). In: Rodewald, P.G., ed. The birds of North America. Ithaca, NY: Cornell Lab of Ornithology.

U.S. Department of Agriculture and U.S. Department of the Interior [USDA and USDOI] 2015. Pollinator-friendly best management practices for federal lands. https://www.fs.fed.us/wildflowers/pollinators/BMPs/documents/PollinatorFriendlyBMPsFederalLands05152015.pdf (27 Nov. 2017).

U.S. Fish and Wildlife Service [USFWS]. 2017. Endangered and threatened wildlife and plants; 12-month findings on petitions to list 25 species as endangered or threatened species. Federal Register. 82: 46618–46645.

U.S. Fish and Wildlife Service, Idaho Department of Fish and Game, Montana Fish, Wildlife & Parks, Wyoming Game and Fish Department, Nez Perce Tribe, National Park Service, Blackfeet Nation, Confederated Salish and Kootenai Tribes, Wind River Tribes, Confederated Colville Tribes, Spokane Tribe of Indians, Washington Department of Fish and Wildlife, Oregon Department of Fish and Wildlife, Utah Department of Natural Resources, and U.S. Department of Agriculture,Wildlife Services. [USFWS et al.]. 2016. Northern Rocky Mountain Wolf Recovery Program 2015 Interagency Annual Report. Helena, MT: U.S. Fish and Wildlife Services, Ecological Services.

Villa-Castillo, J.; Wagner, M.R. 2002. Ground beetle (Coleoptera: Carabidae) species assemblage as an indicator of forest condition in northern Arizona ponderosa pine forests. Environmental Entomology. 31: 242–252.

Waltz, A.E.M.; Covington, W.W. 2004. Ecological restoration treatments increase butterfly richness and abundance: mechanisms of response. Restoration Ecology. 12: 85–96.

Webb, N.F.; Hebblewhite, M.; Merrill, E.H. 2008. Statistical methods for identifying wolf kill sites using global positioning system locations. Journal of Wildlife Management. 72: 798–807.

Williston, P. 2000. Floristics and distribution patterns of lichens and bryophytes in microbiotic crusts of British Columbia's ponderosa pine forests. In: Darling, L.M., ed. Proceedings of biology and management of species and habitats at risk. Kamloops, BC: British Columbia Ministry of Environment, Lands and Parks and University College of the Cariboo: 769–774.

Wisdom, M.; Holthausen, R.S.; Wales, B.C., [et al.]. 2000. Source habitats for terrestrial vertebrates of focus in the interior Columbia basin: broadscale trends and management implications—vol. 1. Gen. Tech. Rep. PNW-GTR-485. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 175 p.

Witmer, G.W.; Martin, S.K.; Sayler, R.D. 1998. Forest carnivore conservation and management in the Interior Columbia Basin: Issues and environmental correlates. Gen. Tech. Rep. PNW-GTR-420. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 51 p.