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Abstract

The *Northeastern California Plateaus Bioregion Science Synthesis* reviews literature relevant to the ecology and management of the Great Basin ecosystems and dry pine forests of the Lassen and Modoc National Forests. Critical factors on these national forests are reduced water availability—expected to become more challenging as levels and patterns of precipitation and temperature change under climate variability—coupled with a high proportion of rangeland and open woodland whose vegetation community is influenced by grazing of livestock and wild animal populations. Conifer encroachment of rangelands and the densification of woodlands, a result of fire suppression, impact wildlife communities that rely on open woodlands and other habitats characterized by having overstories of low density. Sagebrush habitat, in particular, is threatened by fragmentation and conversion. Socioeconomic changes in the region include a transition in the economic base from extraction to that of consumption of amenity values, and the resulting fragmentation of landownership. The local human population is expected to continue its trend of decline, but increased pressure by recreationists from nearby expanding urban areas is forcing land managers to consider increasingly complex situations or actions integrating social, ecological, and economic factors. Indigenous peoples are assuming a greater role in the management of their lands. Finally, disturbance patterns, such as nonhistorical fire frequency and intensity levels, novel combinations of climate patterns, and the pervasive pressure of nonnative invasive species could result in future ecosystems different than those today, presenting additional managerial challenges. This synthesis is intended to serve as a science-based foundation that supports management of Northeastern California forests, woodlands, and rangelands.

Keywords
Lassen National Forest, Modoc National Forest, Northeastern California, forest planning, community engagement, socioeconomic resilience, ponderosa pine, western juniper, sagebrush rangeland, wildfire, wildlife, ecosystem restoration, climate change, disturbances

Cover photos (clockwise from bottom left): Ponderosa pine (photo by Ken Sandusky, Forest Service). Sagebrush rangeland (photo by Dawn M. Davis, used with permission). Sheridan's Hairstreak butterfly (photo by Tony Kurz, used with permission). Sage thrasher (photo by Tony Kurz, used with permission). Fall colors in the Granger Canyon, Modoc National Forest (photo by Ken Sandusky, Forest Service).

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Foreword

We are pleased to introduce the *Northeastern California Plateaus Bioregion Science Synthesis*, the last in a series of three science syntheses produced collaboratively by Forest Service research scientists and resource managers to support forest plan revisions in the Pacific Northwest and Pacific Southwest Regions. These three publications synthesize the best available scientific information relevant to the National Forests of both Regions. Development of these syntheses provided a strategic opportunity for the Forest Service to leverage strengths across two different arms of the organization, National Forest System (NFS) and Research and Development. As a science-based organization, the Forest Service is committed to using best available science to inform our actions, and these syntheses were created in advance of revising our forest plans to ensure we have a strong scientific foundation. Key topics addressed by these syntheses were identified by the public and by Forest Service resource managers and scientists and refined through public engagement and participation. The final documents are a result of meaningful input from the public and resource managers and serve as testament to the collaborative process.

The *Northeastern California Plateaus Bioregion Science Synthesis* addresses the mostly dry ecosystems and species assemblages found in Northeastern California, including unique pine forests and sagebrush rangelands; it also discusses changing demographics in the context of socioeconomic resilience. In doing so, it fills a gap not met by the other two science syntheses and completes the scientific picture that will inform modernization of our forest plans in the West.

We invite you to read and enjoy the *Northeastern California Plateaus Bioregion Science Synthesis.* As we continue the process of revising our forest plans, we ask that you remain engaged with us. Working together, we can ensure the plans provide for ecosystem and resource sustainability, meet the need for forest restoration and conservation, watershed protection, and species diversity and conservation, and provide a sustainable flow of benefits, services, and uses of the National Forest System that deliver jobs and contribute to the economic and social sustainability of communities.

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Executive Summary

The *Northeastern California Plateaus Bioregion Science Synthesis* is organized into six sections (each with one or more chapters) corresponding to five major topic areas identified by the staffs of the Lassen National Forest and Modoc National Forest (hereafter the Lassen and Modoc) and refined through a facilitated public workshop held in Susanville, CA, in December 2016. The overall objective of this synthesis is to address the unique habitats of the Lassen and Modoc not addressed, or not fully addressed, by other science syntheses that have relevance to the Lassen and the Modoc, and, together with those other syntheses, provide a science-based foundation for preparation of revised, independent forest plans for these two forests.

Section One: Introduction

Chapter 1.1, *The Northeastern California Plateaus Bioregion Science Synthesis: Background, Rationale, and Scope,* briefly discusses the forest plan process and why a science synthesis is the first step. This chapter also describes two other science syntheses that are relevant to the Lassen and Modoc and details the coverage afforded by all three science syntheses.

- In addition to this *Northeast California Plateaus Bioregion Science Synthesis,* the *Science Synthesis to Support Socioecological Resilience in the Sierra Nevada and Southern Cascade Range* and the *Synthesis of Science to Inform Land Management Within the Northwest Forest Plan Area* provide a science-based foundation for developing forest plans for the Lassen and Modoc.
- Forest staffs and the public agreed that this synthesis should focus on dry-site ponderosa pine (*Pinus ponderosa*), western juniper (*Juniperus occidentalis*), and sagebrush (*Artemisia* species) ecosystems to augment science synthesized in the other two syntheses.

Section Two: Forestland

Chapter 2.1, *Understanding and Managing the Dry Conifer Forests of Northeastern California,* begins with a broad discussion about the potential impacts of drought on Western forestlands, and then focuses in more depth on different forest management tenets. The chapter discusses ponderosa and Jeffrey (*Pinus jeffreyi*) pine ecology, history, and how those forests can benefit from silviculture, and concludes with discussion about the ecology and management of juniper-dominated forestlands.

- Water supply, or the lack of it, is the major driver in defining ecosystems in the Interior West and, specifically, Northeastern California.
- Western juniper, the only *Juniperus* species in Northeastern California, shares many characteristics with other juniper species in the Great Basin and research on these species can offer insight into what the future might hold for the Lassen and Modoc.
- Restoration guided by pre-European settlement structure and composition can create forests better able to withstand fluctuations in climate.
- While novel combinations of future climate and past management history may occur, historical evidence can still guide managers as they seek to manage such sites.

Section Three: Rangeland

Chapter 3.1, *Perceptions and History of Rangeland,* describes how rangeland management has been perceived in the Western United States.

- Government policy promoted grazing on rangeland throughout the West.
- Grazing has been an important factor in the historical economic and social development of the Interior West and the Lassen and Modoc in particular.

Chapter 3.2, *Rangeland in Northeastern California,* focuses on the interactions of climate, grazing, and carbon storage on rangelands; the response of native plant communities, especially those dominated by annual invasive grasses, to grazing; meeting rangeland management objectives; and restoring sagebrush ecosystems.

- A high proportion of Northeastern California is rangeland.
- Sixty percent of the Modoc and 30 percent of the Lassen are classified as rangeland suitable for grazing.
- Wild horse (*Equus ferus*) populations can influence rangeland plant composition and biomass.
- Feral and domestic animal concentrations increase soil compaction and impact soil water retention and runoff.
- Reversing conifer encroachment has important effects on greater sage-grouse (*Centrocercus urophasianus*) and other wildlife species.

Chapter 3.3, *Weeds, Wheels, Fire, and Juniper: Threats to Sagebrush Steppe,* reviews threats (and management responses) to the sagebrush ecosystem, including invasive weeds, vehicles, fire, and conifer encroachment.

- Numerous invasive plants have displaced native plant communities throughout the rangeland of Northeastern California.
- Cheatgrass (*Bromus tectorum*) is the most damaging invasive plant in the Great Basin Region, including Northeastern California.
- Fire can control conifer encroachment but also can accelerate cheatgrass dominance where the invasive species is present.

Chapter 3.4, *Biological Soil Crusts,* examines the ecology, threats, and restoration of the microorganisms associated with biological soil crusts.

- Biological soil crusts (BSCs) commonly occur in arid environments.
- BSCs are a commonly neglected aspect of management.
- BSCs improve soil water retention and nutrient capture and retention.
- BSCs are easily damaged by disturbance, such as vehicle travel, and are difficult to restore artificially.
- Natural recovery of BSCs occurs via aerial deposition and depends on the severity and spatial distribution of the initial disturbance.

Section Four: Habitat and Wildlife

Chapter 4.1, *Biodiversity and Representative Species in Dry Pine Forests,* examines the biodiversity of dry pine forests, from fungi to herbaceous plants to invertebrates to three representative species of this habitat: black-backed woodpecker (*Picoides arcticus*), flammulated owl (*Psiloscops flammeolus*), and gray wolf (*Canis lupus*).

- Dry pine forests provide a range of open and closed structure and a history of frequent low-intensity fires, conditions that benefit specific species, including the three mentioned above. Variations in disturbance patterns may impact the dry pine forest community; overgrazing risks accelerating the spread of invasive plants and fire suppression risks reducing open forest structure.
- Black-backed woodpeckers are found in the Lassen and Modoc. They follow beetle outbreaks after fires and forage on such sites for up to 10 years.
- 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 arthropods.
- Populations of gray wolves are expanding and may soon affect local ungulate populations and behavior. Strategies to prevent livestock depredation are critical before conflicts develop.

Chapter 4.2, *Aquatic Ecosystems, Vernal Pools, and Other Unique Wetlands,* focuses on the role and importance of aquatic ecosystems, including lakes, vernal pools, fens, and swales to biodiversity, especially native trout and rare plants.

- Unlike the Pacific Northwest and the Sierra Nevada, the Modoc Plateau contains a greater number of ephemeral and intermittent streams and wetlands, and many closed hydrologic systems.

- Land and water-use activities have negatively impacted several species of fish and two varieties of trout.
- Fens, bogs, and vernal pools are sensitive to grazing and mechanical disturbance.

Chapter 4.3, *Sagebrush Rangelands and Greater Sage-grouse in Northeastern California,* begins with a broad look at sagebrush rangelands before focusing on greater sage-grouse; this bird's health is considered by some as an indicator of overall sagebrush ecosystem health.

- Sagebrush habitat in Northeastern California is imperiled by fragmentation, degradation, and conversion to other uses, compounded by changing climate, grazing, and conifer encroachment.
- Between the early 1950s and the late 1980s, sagebrush populations declined 60 percent on the Lassen and 86 percent on the Modoc.
- Once-abundant populations of greater sage-grouse supported a hunting season, but now populations have declined precipitously in sagebrush habitat in Northeastern California.
- Conifer encroachment is a serious influence on sagebrush extent and quality. Juniper distribution in the Interior West has increased 3 to 10 times and abundance has increased 10 times.

Section Five: Society

Chapter 5.1, *An Introduction to Social, Economic, and Ecological Factors in Natural Resource Management of Northeastern California Public Lands*, provides an overview of the section.

- Stressors on the forest ecosystems on the Lassen and Modoc will come from demographic trends and the demands on ecosystem services from within Northeastern California and throughout the State.
- Measures of adaptation may be learned and practiced through community engagement and respect for multiple cultures, particularly those of tribes.

Chapter 5.2, *Demographic Trends in Northeastern California,* focuses on current rural demographics in Lassen and Modoc Counties and their expected changes, the impacts of prisons on local communities, and the decisionmaking process managing natural resources.

- Lassen and Modoc Counties have been declining in population in recent decades and this decline is expected to continue. Nearby urban areas, such as Reno, are expected to grow in area and population.
- The average parcel size of rural landowners is decreasing, portending potential impacts on landscape ecological processes due to fragmentation.
- The presence of prisons affects the economic development potential and the political power of Lassen and Modoc Counties.

Chapter 5.3, *Ecosystem Services and Public Land Management,* examines the social benefits provided by ecosystems, the economic benefits of these services, and societal inputs into resource management.

- The changes in land use, economic activity, and social relations that accompany the shift from industrial extraction to amenity value consumption now require public lands managers to simultaneously manage for ecological, economic, and social concerns.
- Social demand for more conservation activities reflects the tension between increasing social value for open space and declining ecosystem health.
- As more Californians adopt lifestyle values centered on amenity and investment, private land has shifted from relatively few farmers and ranchers with large tracts of land to much smaller parcels owned by relatively more landowners.
- A shift in community attitudes in recent years is changing from valuing fire suppression less to valuing fire management more.

Chapter 5.4, *Community Engagement in the Decisionmaking Process for Public Land Management in Northeastern California*, focuses on community engagement and how communities can participate in natural resource management and use that engagement to resolve conflict.

- Historically, Northeastern California has had a strong timber industry that has declined in recent times. Another extractive process, agricultural production, is still a predominant driver of the local economy.
- Over the years, decisionmaking authority concerning natural resource management policy and practice has shifted from a centralized model to a more participative one.
- Complex land resource issues and the diverse social values make it challenging to engage the local community into the decisionmaking process as it pertains to local ecosystem services.
- Subsistence gathering has an economic aspect and strong cultural connection that can influence public response to management activities.

Chapter 5.5, *Integrating Tribes and Culture Into Public Land Management,* investigates how tribes value place, interact with managers, and use fire management.

- Native American land management practices were historically not incorporated into public agency land management decisionmaking, but that is changing as agencies become more receptive to traditional environmental knowledge and indigenous land management practices.
- Environmental values can move beyond solely considering the consumptive aspects of land management and can integrate detailed knowledge of regional ecological conditions with a conservation ethos, especially when considering Native American cultural traditions of places and landscapes.
- Identity of place can be degraded by land management or resource allocation practices that fail to consider the importance of a particular site to a community.
- Traditional environmental knowledge judges the success of conservation efforts as much by the outcome as by the extent of community participation and the input from networks of localized knowledge.

Section Six: Responding to Disturbances

Chapter 6.1, *Ecological Disturbance in the Context of a Changing Climate: Implications for Land Management in Northeastern California,* takes a broad look at various disturbance factors that are currently, or may in the future, affect these ecosystems, how a changing climate interacts with those factors, and what possible management techniques could be considered to mitigate disturbances.

- For Northeastern California, most climate models show rising temperatures, but precipitation may increase, remain steady, or slightly decrease. Regardless, precipitation is expected to follow the Mediterranean pattern of most precipitation falling in the winter months. With warmer temperatures, less precipitation will fall as snow.
- Historically, the Great Basin has experienced frequent droughts, some quite lengthy, yet for Northeastern California the vegetation composition has remained relatively stable for at least the past 4,300 years.
- The distribution and range of sagebrush is closely tied to climate. Models based on climate and hydrology predict areas that the amount of area in Northeastern California that is appropriate for sagebrush will increase and then decrease in the future. Cheatgrass will continue to change sagebrush communities.
- A range of studies have been conducted examining modeled future climates and impacts on wildlife species. For example, pika (*Ochotona princeps*) have come to be a model species for climate change in montane systems. Pika have shown resilience with regards to invasive species as well as wildfire.
- The choice of seed source is for ecological restoration projects is complicated by climate change. It is unclear how effective the current California Seed Zone Map will be given modeled future climates. Additional research is needed; several new tools are available to assess how appropriate a seed sources is for a given planting site.

Table of Contents

Section 1. Introduction

Chapter 1.1. The Northeastern California Plateaus Bioregion Science Synthesis: Background, Rationale, and Scope

R. Kasten Dumroese[1]

Introduction

Situated in Northeastern California, the Lassen National Forest spans 1.2 million acres (485,625 ha) and the Modoc National Forest another 1.65 million acres (667,730 ha), mainly in Lassen and Modoc Counties, but also across portions of Butte, Plumas, Shasta, Siskiyou, and Tehama counties. A unique blend of geology, topography, elevations from 2,000 to 14,000 feet (610 to 4,270 m), and climate have fostered immense biodiversity within this area. On the western edge of these national forests, the Cascade Range, characterized by steep changes in elevation and a Mediterranean climate (cool, wet winters and warm, dry summers), draws out the precipitation from Pacific storms, leaving a rain shadow to the east. The result is a rich vegetation gradient of mixed-conifer forest, dry pine forests, oak savannahs, juniper woodlands, and sagebrush steppe. In addition, fire, promoted by the Mediterranean climate, further stimulates a mosaic of vegetation across the landscape that supports a wide variety of fauna. For more detailed descriptions of this area, see Gonzales and Hoshi (2015a, b), Riegel et al. (2006), and Skinner and Taylor (2006).

The Cascade Range, characterized by basalt parent material, runs from southern British Columbia to just south of Lassen Peak in Northern California, where it transitions to the granite parent material of the Sierra Nevada (fig. 1.1.1). In Northern California, the Cascade Range receives abundant rain and snow, and supports rich, mixed-conifer forests that supply valuable forest products and serve as the source of appreciable surface water. The forests of the Cascade Range are home to several animal species of concern, such as the northern spotted owl (*Strix occidentalis caurina*) and Pacific marten (*Martes caurina*) (Gonzales and Hoshi 2015a).

The Modoc Plateau, itself comprised of multiple plateaus such as Devil's Garden, is a broad transition zone from the basalt parent material of the Cascade Range to the sedimentary/alluvial parent materials of the Great Basin (fig. 1.1.1; Fuller et al. 2015). The Modoc Plateau began forming about a million years ago, when a long series of gentle eruptions of smooth lava began flowing across the landscape (see Peacock 1931). Because of its transitional geologic state and location in the Cascade Range rain shadow, much of the Modoc Plateau is exemplified by ecosystems typical of the Great Basin (sagebrush rangeland, shrub steppe, and juniper woodlands). These ecosystems provide important habitat for wildlife species that require sagebrush (*Artemisia* species), such as greater sage-grouse (*Centrocercus urophasianus*), pygmy rabbit

Figure 1.1.1—The Lassen and Modoc National Forests are in Northeastern California, where the Cascade Range, the Sierra Nevada, and the Modoc Plateau intersect. Here the watershed of the Pit River forms the headwaters of the Sacramento River, one of the most important waterways in the State.

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Figure 1.1.2—The wet meadows, vernal pools, and other wetlands of Northeastern California provide critical habitat for resident and migratory birds, such as these sandhill cranes visiting Pine Creek Valley on the Lassen National Forest (photo by Emmett Richards, Forest Service).

(*Brachylagus idahoensis*), pronghorn (*Antilocapra americana*), and sagebrush sparrow (*Artemisiospiza nevadensis*). Other wildlife, such as mule deer (*Odocoileus hemionus*) and elk (*Cervus canadensis*), also make use of this habitat. Vernal pools (temporary, seasonal wetlands) support rare plants and animals and the biodiversity of California's vernal pools is noteworthy (e.g., King et al. 1996; Simovich 1998). Vernal pools help connect wetland habitat along the Pacific Flyway and are thus important for migrating birds such as sandhill cranes (*Grus canadensis*), ducks, and geese (fig. 1.1.2).

The Modoc Plateau gives rise to the 3-million-acre (1.2-million-ha) Pit River watershed that is critical to wildlife and human sustainability (fig. 1.1.1). The upper reaches of the watershed flow from the Warner Mountains, forming the headwaters of the Sacramento River that supplies about 20 percent of the water to the Sacramento Basin, which in turns irrigates about 2.1 million acres (850,000 ha) of agricultural crops (California Department of Water Resources 1998; Gonzales and Hoshi 2015a). It also supports many endemic and threatened aquatic species (Gonzales and Hoshi 2015a; Moyle et al. 2011).

The Federal Government manages about 60 percent of the Modoc Plateau, with about one-third managed by the U.S. Department of Agriculture, Forest Service. Contained within the footprint of the Lassen and Modoc National Forests are 6 wilderness areas covering about 257,000 acres (104,000 ha). The Caribou, South Warner, and Thousand Lakes wildernesses are managed by the Forest Service, Ishi Wilderness is managed jointly by the Forest Service and the U.S. Department of the Interior, Bureau of Land Management and Lassen Volcanic Wilderness is managed by the U.S. Department of the Interior, National Park Service. Also within the footprint are the Lava Beds National Monument (46,000 acres [18,615 ha]) and Lassen National Park (106,000 acres [42,900 ha]), both managed by the National Park Service.

Planning for the Future

A land management plan or "forest plan" guides how the Forest Service manages the associated public lands and natural resources for a period of 15 to 20 years. Following mandates and provisions of the 1982 Planning Rule, the forest plan for the Modoc National Forest (hereafter, the Modoc) was completed in 1991, while the forest plan for the Lassen National Forest (hereafter, the Lassen) was completed a year later. Thus, it is time for both forests to revise their existing forest plans to meet the legal requirements of the National Forest Management Act of 1976 and incorporate changes in law, regulation, and policy. The plan revisions will be guided by the 2012 Planning Rule (https://www.fs.usda.gov/planningrule), which requires public and tribal input throughout a multi-step process that embraces the fact that ecological, social, and economic objectives are interrelated. Because conditions have changed since the original forest plans were written and because new science is available, the first step is to prepare a science synthesis, guided by input from the public, tribes, and forest staffs (fig. 1.1.3). Although the Lassen and Modoc will each prepare an independent forest plan, these neighboring forests have worked together on this science synthesis because they share management of unique geology and ecosystems.

What Is a Science Synthesis?

Scientists are continually learning and gaining new understanding of the natural processes that affect ecosystems, how humans influence ecosystems, and the ways that society values what ecosystems provide. Since the last forest plans for the Lassen and Modoc were written, much new information has been discovered. Combining these discoveries into a single document, commonly referred to as a "synthesis," requires reviewing the best available scientific information. The goal of the science synthesis is to combine the findings from the full body of relevant science within a topic area (defined and refined by input from the public, tribes, and forest staffs) into a current, concise, comprehensive, and coherent

Figure 1.1.3—Forest plans are revised through a multi-step process that encourages public participation. The first step is Pre-Assessment, which includes development of a science synthesis.

overview that can be more readily interpreted and used by forest staffs and stakeholders during the forest plan process. In turn, the science synthesis will be followed by additional opportunities for public input as the forest plans develop (fig. 1.1.3). While science syntheses focus mainly on broad, widely accepted and applicable concepts, this synthesis also sought to incorporate research specifically conducted on or near the Lassen and Modoc.

Best Available Scientific Information

In *A Citizens' Guide to National Forest Planning*, the Forest Service explains its definition of best available scientific information (see textbox 1.1.1). In evaluating science for inclusion in the synthesis, the Science Team (comprised of seven Forest Service scientists representing the Rocky Mountain and Pacific Southwest Research Stations) that prepared this synthesis followed an assessment hierarchy similar to that used in recent science syntheses prepared in support of forest planning in California, Oregon, and Washington (Long et al. 2014; Spies et al. 2018). Peer-reviewed (refereed) journal publications were the primary sources of information because these publications are critically scrutinized by independent reviewers (referees), usually anonymously, to ensure that the study methods were well-developed, appropriate, and with reasonable assumptions; data were correctly analyzed; results were valid; the discussion and conclusions were logical and supported by the data collected, any information gaps and/or inconsistencies were addressed, and the work is placed in the proper context within the body of knowledge; and that the references included the most recent, relevant, refereed work as well as older, foundational studies. The Science Team also examined publications having undergone peer review but published in other formats, such as government publications (e.g., Forest Service General Technical Reports), conference proceedings, proceedings from professional organizations, and university theses and dissertations. Occasionally, unpublished government reports were included too. During the initial public forum and comment period (December 2016), participants suggested nearly 60 publications to be considered in addition to publications identified using Internet search tools, such as Google Scholar and Web of Science, and the Forest Service database, Treesearch (https://treesearch.fs.fed.us). More than 1,100 publications were reviewed in the process.

Textbox 1.1.1—"The 2012 Planning Rule requires the use of the best available scientific information to inform planning and plan decisions. Science is a dynamic process that builds knowledge and reduces uncertainty by testing predictions; scientific information can be considered the expanding body of knowledge developed through the scientific process. Scientific information comes in many forms, including social, economic, and ecological information. Scientific information comes from many sources—for example, from peer-reviewed articles, scientific assessments, expert opinion, and data in the form of monitoring results. It also comes from information gathered during public involvement efforts and traditional ecological knowledge. What is the 'best available scientific information'? Generally, it is high-quality information that results from well-developed and appropriate methods, draws logical conclusions based on reasonable assumptions, explains information gaps and inconsistencies, has been appropriately peer-reviewed, is placed in the proper context within the body of knowledge, and cites references. Not all information, however, needs to meet all of these characteristics to be considered best available scientific information. At a minimum, scientific information needs to be available, accurate, reliable, and relevant. 'Available' means that the Forest Service does not need to create new scientific information and conduct new research, but simply should use information that currently exists. Finally, one of the fundamentals to effective use of scientific information is transparency in how it is used. The 2012 Planning Rule requires the Forest Service to document and summarize how the universe of best available scientific information was identified and how it informed the planning process."

From: A citizens' guide to national forest planning. Prepared by the Federal Advisory Committee on Implementation of the 2012 Land Management Planning Rule. Version 1.0. Washington, DC: U.S. Department of Agriculture, Forest Service, Washington Office. September 2016. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fseprd520670.pdf.

Northeastern California Plateaus Bioregion Science Synthesis

The goal of the *Northeastern California Plateaus Bioregion Science Synthesis* (hereafter, Plateaus Science Synthesis) is to address the unique niches of the Lassen and Modoc not addressed, or fully addressed, by two previously completed science syntheses that have relevance to the Lassen and

Modoc. The first relevant synthesis is the *Science Synthesis to Support Socioecological Resilience in the Sierra Nevada and Southern Cascade Range* (hereafter, Sierra Nevada Science Synthesis) published by Long et al. (2014), which covered the nine national forests in California associated with the Sierra Nevada, including the Lassen and Modoc (see textbox 1.1.2). This comprehensive, two-volume publication addressed the “forested mountains” (p. iv), primarily the “conifer-dominated forest ecosystems” (p. 4) of the Sierra Nevada, the Southern Cascades, and the Modoc Plateau. The second is the *Synthesis of Science to Inform Land Management within the Northwest Forest Plan Area* (hereafter, simply Northwest Forest Plan Science Synthesis; Spies et al. 2018). Portions of this synthesis are applicable because the Lassen and Modoc, according to the 2018 data used in the Pacific Southwest Region of the Forest Service for planning, have approximately 45,000 and 51,000 acres (18,210 and 20,640 ha; 3 and 3.8 percent of the total landbase), respectively, included in the Northwest Forest Plan Area. Moreover, the ecoregions (see textbox 1.1.3) associated with these designated acres (Level III Ecoregion 4) cover a much larger area of the Lassen (a total of 585,000 acres [236,740 ha]).

Why Is Another Science Synthesis Needed?

Most commonly, national forests are supported in their efforts to revise forest plans by a single science synthesis. Because of their unique location on the landscape, portions of the Lassen and Modoc were already addressed by the Northwest Forest Plan and Sierra Nevada science syntheses. Thus, during the initial stages of development, the Plateaus Science Synthesis was referred to as the “Great Basin Science Synthesis,” reflecting the known need to address sagebrush rangeland (characteristic of the Great Basin) not covered by the Northwest Forest Plan and Sierra Nevada science syntheses. Early input from forest staffs, tribes, local governments, and the general public showed, however, that some forest landscapes that fell within the defined scopes of the earlier syntheses were insufficiently addressed and amply unique to require special attention, thus pushing the scope of this effort beyond the Great Basin definition (see *Scope* below). Some other topics not included in the previous syntheses but deemed necessary in order for the Lassen and Modoc to revise their forest plans included juniper forestland, wild horses, greater sage-grouse, and the effects of prison populations on local demographics. Because plateaus are common landscape features of the Lassen and Modoc, this effort was renamed the *Northeastern California Plateaus Bioregion Science Synthesis* to more uniquely describe the land base covered. Therefore, the Lassen and Modoc will each establish their revised forest plans drawing from all three science syntheses as necessary and appropriate for different portions of their landscapes (fig. 1.1.4).

Textbox 1.1.2—The ecological portion of Sierra Nevada Science Synthesis drew heavily from Forest Service General Technical Reports PSW-GTR-220 (North et al. 2009) and PSW-GTR-237 (North 2012) that included substantial discussion about forest types occurring in the Southern Cascades and the Sierra Nevada and their fauna (e.g., fisher, Pacific marten, northern spotted owl). The Sierra Nevada Science Synthesis also made a substantial effort to address social issues. The effects of grazing on wet meadows and their restoration was the focus of Chapter 6.3 (Long and Pope 2014), and grazing on national forests in California was described in Chapter 9.5 (Charnley and Long 2014).

Textbox 1.1.3—Ecoregions are areas where ecosystems (a biological community of interacting organisms and their physical environment) are relatively similar. This synthesis uses ecoregions, developed by Omernik (1987) and widely used by many Federal agencies, to provide a common point of reference because maps and descriptions for California ecoregions (Griffith et al. 2016) are readily available on the Internet (https://www.epa.gov/eco-research/ecoregion-download-files-state-region-9#pane-04 or https://dx.doi.org/10.3133/ofr20161021). Whereas Level 1 Ecoregions are the broadest, most general descriptions of the landscape, Level IV Ecoregions provide the most detailed descriptions of particular portions of the landscape. See Appendix 1.1 for descriptions of Level IV Ecoregions used in this synthesis.

Scope

To integrate all three syntheses and note where the Plateaus Science Synthesis provides novel information beyond the Northwest Forest Plan and Sierra Nevada science syntheses, this synthesis divides the Lassen and Modoc into four, necessarily broad, vegetation zones: (1) Southern Cascades Forestland; (2) Sierra Nevada Forestland; (3) Dry Pine Forestland; and (4) Juniper Forestland, Sagebrush Rangeland, and Shrubland (fig. 1.1.5).

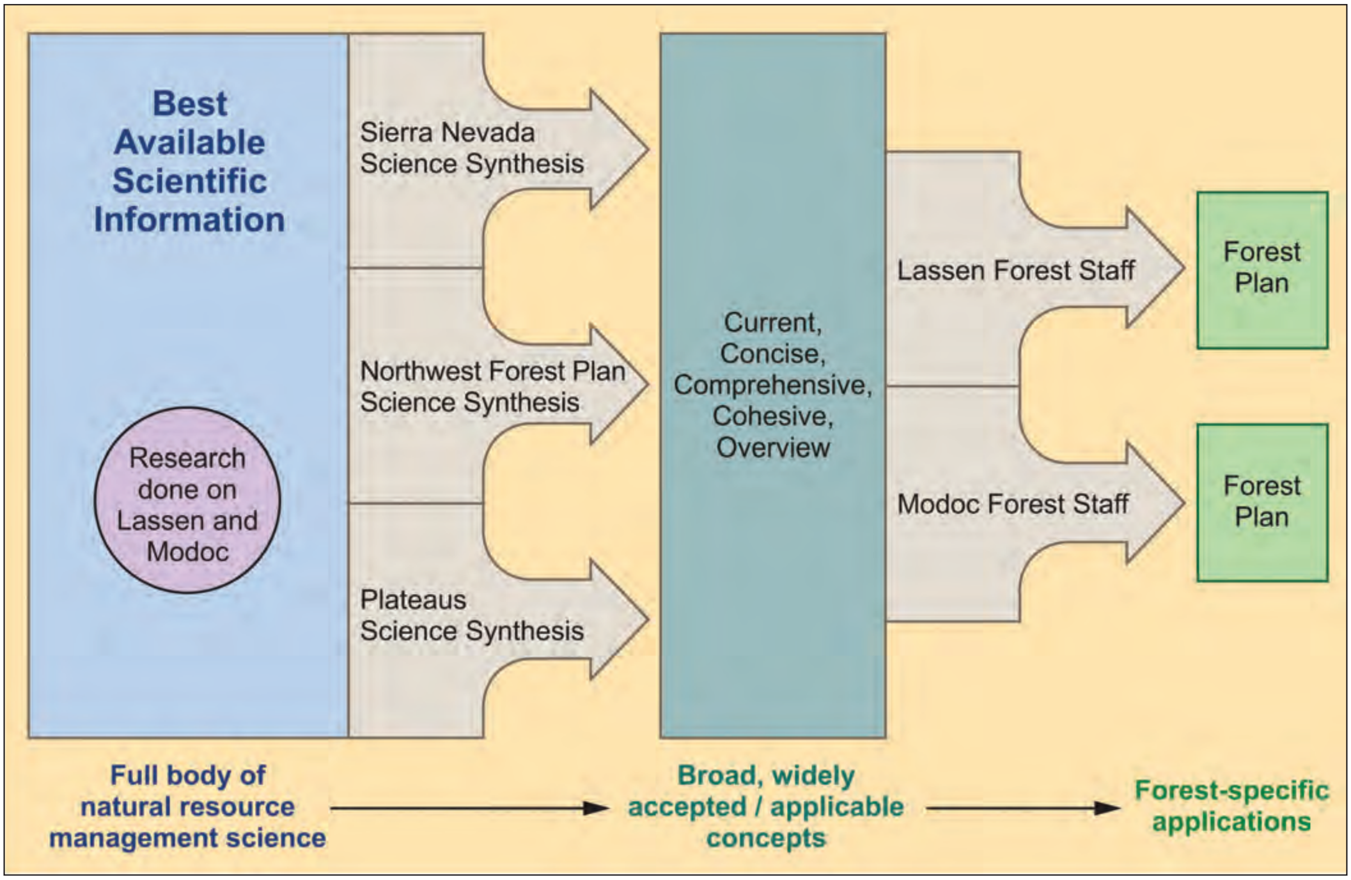

Figure 1.1.4—A science synthesis searches the best available scientific information and condenses it into a current, concise, comprehensive, coherent overview that staffs of national forests can then use to develop forest plans for their respective forests. For this synthesis, the science team reviewed the best available scientific information for topics identified by staffs of the Lassen and Modoc, tribes, local governments, and the general public, with specific focus on finding any relevant research conducted on or near the Lassen and the Modoc. This synthesis, along with the other two science synthesis that have relevance to the Lassen and Modoc (the Sierra Nevada and Northwest Forest Plan science syntheses), primarily deliver the broad, widely accepted and applied concepts that provide a solid science-based foundation that the forest staffs can use as a starting point in the process to revise their specific forest plans. The forest plans are more focused on the pragmatic specifics of the Lassen and the Modoc.

Southern Cascades Forestland

Both the Northwest Forest Plan Science Synthesis and the Sierra Nevada Science Synthesis discuss the forested landscapes of the "Southern Cascades." The Southern Cascades occur within the Level III Ecoregion 4 (Cascades) and are mainly represented by four Level IV Ecoregions (see textboxes 1.1.3, 1.1.4, 1.1.5 and Appendix 1.1 for descriptions of Level IV Ecoregions used in this synthesis). The Southern Cascades forest communities move downslope from alpine zones through high-elevation zones of mountain hemlock, lodgepole pine, white fir, and Shasta red fir to mid-elevation forests of western hemlock, western redcedar, incense cedar, white fir, Shasta red fir, and Jeffrey pine to lower-elevation forests of either ponderosa and Jeffrey pines, or ponderosa pine and Douglas-fir mingled with California black and canyon live oaks. Because of extensive coverage in the previous syntheses, these forests are not discussed in the Plateaus Science Synthesis. The exception is Level IV Ecoregion 4g (California Cascades Eastside Conifer Forest), which is included in the Dry Pine Forestland category in the Plateaus Science Synthesis (figs. 1.1.5 and 1.1.6).

Sierra Nevada Forestland

The Sierra Nevada Science Synthesis generally placed the forests of the Lassen and Modoc in two categories: Sierra Nevada forests and the forests of the Modoc Plateau. The Sierra Nevada forests are represented by Level III Ecoregion 5 (Sierra Nevada) with four Level IV Ecoregions (see textbox 1.1.5) that generally pertain

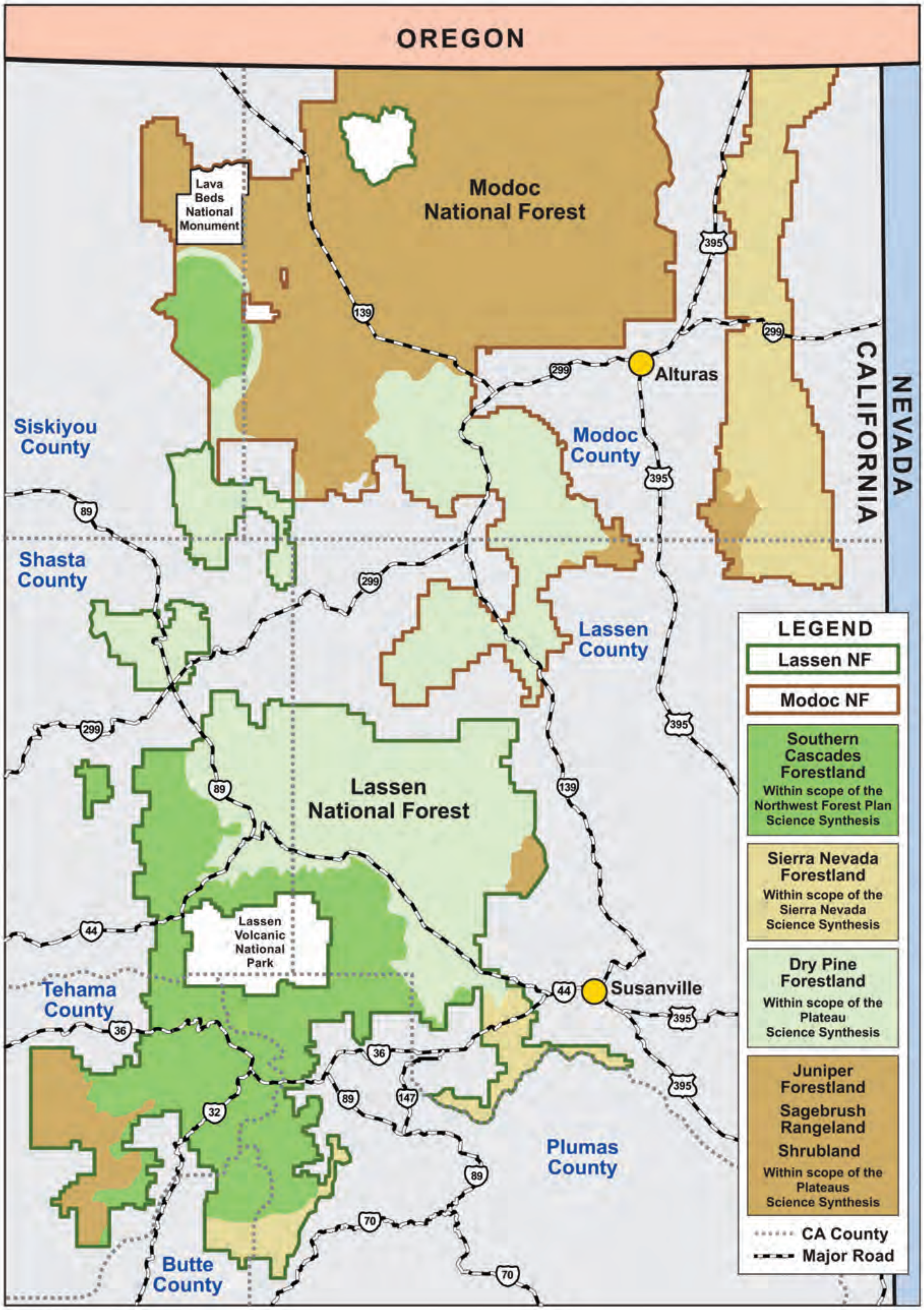

Figure 1.1.5—The Plateaus Science Synthesis delineates the Lassen and Modoc into four broad vegetation zones: (1) Southern Cascades Forestland; (2) Sierra Nevada Forestland; (3) Dry Pine Forestland; and (4) Juniper Forestland, Sagebrush Rangeland, and Shrubland. The Southern Cascades Forestland and the Sierra Nevada Forestland were covered in previous syntheses; thus, the focus of the Plateaus Science Synthesis is on Dry Pine Forestland and Juniper Forestland, Sagebrush Rangeland, and Shrubland.

Textbox 1.1.4—Southern Cascades forest landscapes included in the Northwest Forest Plan and Sierra Nevada Science Synthesis generally represent these ecoregions:

Level III Ecoregion 4—Cascades

- Level IV Ecoregions:
  - *4e: High Southern Cascades Montane Forest*
  - *4f: Low Southern Cascades Mixed Conifer Forest Service*
  - *4g: California Cascades Eastside Conifer Forest Service*
  - *4h: Southern Cascades Foothills*

See Appendix 1.1 for descriptions of these Level IV Ecoregions.

Textbox 1.1.5—Sierra Nevada forest landscapes included in the Sierra Nevada Science Synthesis generally represent these ecoregions:

Level III Ecoregion 5—Sierra Nevada

- Level IV Ecoregions:
  - *5c: Northern Sierra Upper Montane Forests*
  - *5d: Northern Sierra Mid-Montane Forests*
  - *5e: Northern Sierra Lower Montane Forests*
  - *5f: Northern Sierra Mixed Conifer-Pine Forests*

Level III Ecoregion 9—Eastern Cascades Slopes and Foothills

- Level IV Ecoregions:
  - *9m: Warner Mountains*
  - *9n: High-Elevation Warner Mountains*
  - *9q: Adin/Horsehead Mountains Forest and Woodland*

See Appendix 1.1 for descriptions of these Level IV Ecoregions.

to higher-elevation forests of Shasta red and white firs, Jeffrey and sugar pines, incense cedar and quaking aspen; mid-elevation forests of white fir, Douglas-fir, Jeffrey pine, black oak, and tanoak; and lower-elevation forests of Douglas-fir, ponderosa pine, canyon live, interior live, and black oaks, and tanoak. Because of extensive coverage in the previous syntheses, these forests are not discussed in the Plateaus Science Synthesis. The Sierra Nevada Science Synthesis also discusses forests on the Modoc Plateau; these forests are included in Level III Ecoregion 9 (Eastern Cascades Slopes and Foothills) and are mostly represented by the higher-elevation forests of the Warner Mountains (Level IV Ecoregions 9m, 9n) and the lower elevation conifer forests (predominantly Level IV Ecoregion 9q). These forests are chiefly lower-elevation communities of ponderosa and Jeffrey pines and white fir transitioning to forests of ponderosa, Jeffrey, and Washoe pines, quaking aspen, and white fir on the Warner Mountains. Although the forest staffs believed that the forests of the Warner Mountains (Ecoregions 9m and 9n) were sufficiently addressed by the Sierra Nevada Science Synthesis, they also believed that the ponderosa and Jeffrey pine forests on the remainder of the Modoc Plateau (Ecoregion 9q) should be included as part of the dry pine forestland covered in the Plateaus Science Synthesis (figs. 1.1.5 and 1.1.6).

Dry Pine Forestland

In this synthesis, dry pine forestland is defined as the drier portions of the Southern Cascades and the forested portions of the Modoc Plateau excluding juniper-dominated stands (figs. 1.1.5 and 1.1.6). This forestland is mainly ponderosa and Jeffrey pines.

Juniper Forestland, Sagebrush Rangeland, Shrubland

This broad vegetation zone embraces those ecosystems more often associated with the Great Basin and not covered by the Northwest Forest Plan and Sierra Nevada science syntheses (figs. 1.1.5 and 1.1.6). Western juniper is the predominant conifer, with rare occurrence of singleleaf piñon.

Organization of the Plateaus Science Synthesis

The Plateaus Science Synthesis is organized into six sections (each with one or more chapters) corresponding to major topic areas defined through the initial public workshop. The first section is this **Introduction**. The second section, **Forestland**, has a single chapter (Chapter 2.1, Moser, this synthesis, *Understanding and Managing*

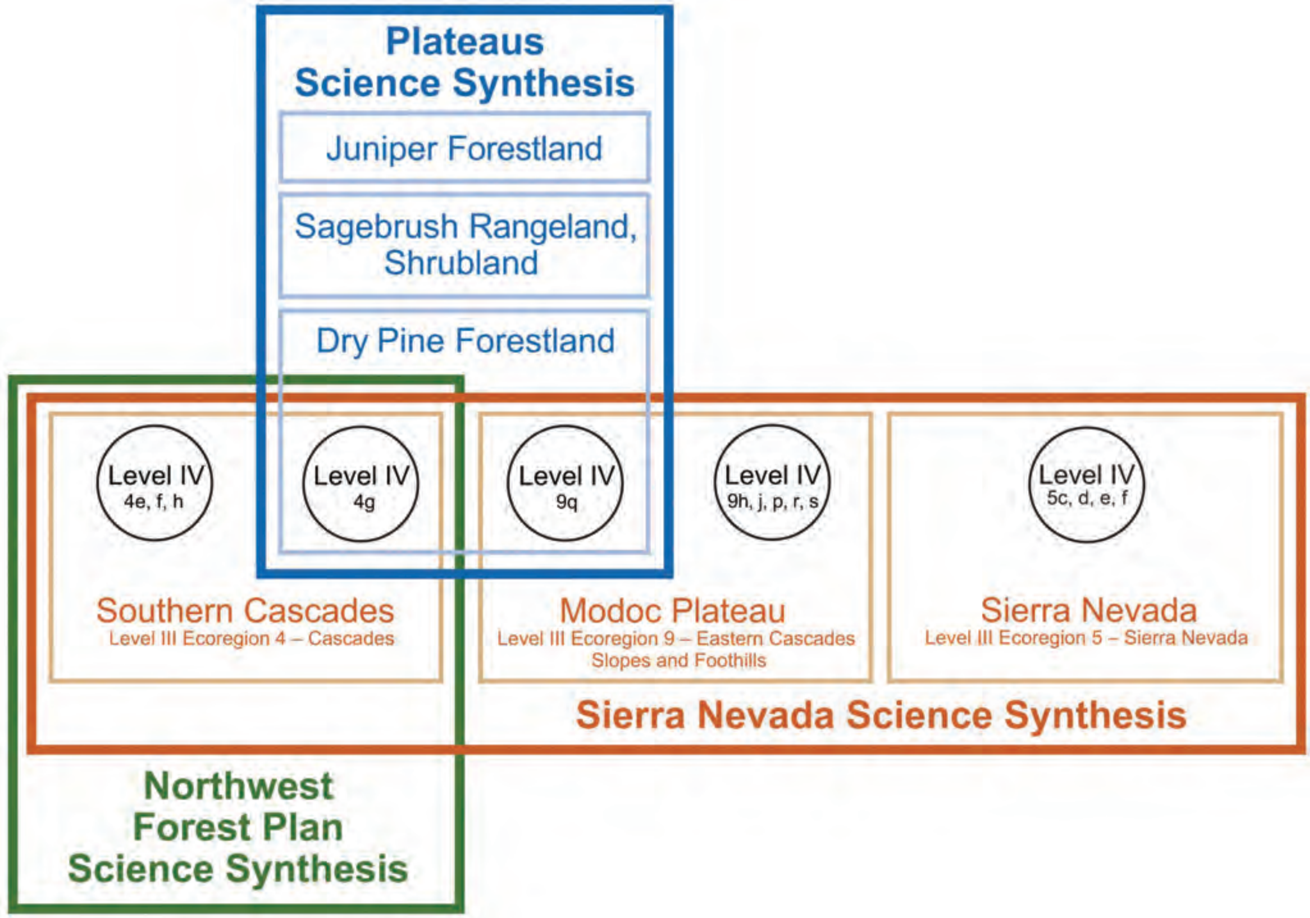

Figure 1.1.6—This Venn diagram shows the scope of three science syntheses as they pertain to the Lassen and Modoc National Forests. The Southern Cascades, which occur in both the Lassen and the Modoc, were part of the discussion within the Northwest Forest Plan Science Synthesis. The Sierra Nevada Science Synthesis also discussed the Southern Cascades as well as forests of the Sierra Nevada and the non-juniper-dominated forests of the Modoc Plateau. The Dry Pine Forestland discussion in the Plateaus Science Synthesis augments discussion about the Southern Cascades in the Northwest Forest Plan and Sierra Nevada science syntheses, particularly that of the California Cascades Eastside Conifer Forest (Level IV Ecoregion 4g). The Dry Pine Forestland also includes discussion about forested landscapes on the Modoc Plateau discussed in the Sierra Nevada Science Synthesis, notably those dominated by Jeffrey and ponderosa pines with some white fir (Level IV Ecoregion 9q; see textbox 1.1.6 for scientific names of tree species used in this chapter). Because neither of the previous syntheses focused on either juniper-dominated landscapes or sagebrush rangelands, these ecosystems are a focus of the Plateaus Science Synthesis.

the Dry Conifer Forests of Northeastern California) devoted to understanding and managing the dry conifer forests of the Lassen and Modoc. This chapter begins with a broad discussion about the potential impacts of drought on Western forestlands, and then focuses in more depth on different forest management tenets. The chapter discusses ponderosa and Jeffrey pine ecology, history, and how those forests can benefit from silviculture, and concludes with discussion about the ecology and management of juniper-dominated forestlands.

The third section, **Rangeland**, consists of four chapters. The first chapter (Chapter 3.1, Warren, this synthesis, *Perceptions and History of Rangeland*) briefly describes how rangeland management has been perceived in the Western United States. The second chapter focuses on the interactions of climate change, grazing, and carbon storage on rangelands; the response of native plant communities, especially those dominated by annual invasive grasses, to grazing; meeting rangeland management objectives; and restoring sagebrush ecosystems (Chapter 3.2, Dumroese, this synthesis, *Rangeland in Northeastern California*).

Textbox 1.1.6—Tree species mentioned in this chapter.

California black oak—*Quercus kelloggii*

Canyon live oak—*Quercus chrysolepis*

Douglas-fir—*Pseudotsuga menziesii*

Incense cedar—*Calocedrus decurrens*

Interior live oak—*Quercus wislizeni*

Jeffrey pine—*Pinus jeffreyi*

Lodgepole pine—*Pinus contorta*

Mountain hemlock—*Tsuga mertensiana*

Ponderosa pine—*Pinus ponderosa*

Quaking aspen—*Populus tremuloides*

Shasta red fir—*Abies magnifica*

Singleleaf piñon—*Pinus monophylla*

Sugar pine—*Pinus lambertiana*

Tanoak—*Notholithocarpus densiflorus*

Washoe pine—*Pinus ponderosa* var. *washoensis*

Western hemlock—*Tsuga heterophylla*

Western juniper—*Juniperus occidentalis*

Western redcedar—*Thuja occidentalis*

White fir—*Abies concolor*

The third chapter (Chapter 3.3, Padgett, this synthesis, *Weeds, Wheels, Fire, and Juniper: Threats to Sagebrush Steppe*) reviews threats (and management responses) to the sagebrush ecosystem, including invasive weeds, vehicles, fire, and conifer encroachment. Chapter 3.4 (Warren, this synthesis, *Biological Soil Crusts*) examines the ecology, threats, and restoration of the microorganisms associated with biological soil crusts.

The fourth section, **Habitat and Wildlife**, consists of three chapters. The first chapter (Chapter 4.1, Hanberry and Dumroese, this synthesis, *Biodiversity and Representative Species in Dry Pine Forests*) examines the biodiversity of dry pine forests, from fungi to herbaceous plants to invertebrates to three representative species of this habitat: black-backed woodpecker (*Picoides arcticus*), flammulated owl (*Psiloscops flammeolus*), and gray wolf (*Canis lupus*). Chapter 4.2 (Padgett, this synthesis, *Aquatic Ecosystems, Vernal Pools, and Other Unique Wetlands)* focuses on the role and importance of aquatic ecosystems, including lakes, vernal pools, fens, and swales to biodiversity, especially native trout and rare plants. The final chapter (Chapter 4.3, Dumroese, this synthesis, *Sagebrush Rangelands and Greater Sage-grouse in Northeastern California*) begins with a broad look at sagebrush rangelands and then focus on greater sage-grouse because some view the health of this species as an indicator of overall sagebrush ecosystem health.

Society is the topic of the fifth section. Given that the major focal point of new forests plans is management of National Forest System lands so that they are ecologically sustainable and contribute to social and economic sustainability, this section addresses the subject in five chapters. The first chapter provides an overview of the section (Chapter 5.1, Flores, this synthesis, *An Introduction to Social, Economic, and Ecological Factors in Natural Resource Management of Northeastern California Public Lands*). The second chapter (Chapter 5.2, Flores and Russell, this synthesis, *Demographic Trends in Northeastern California*) focuses on current rural demographics in Lassen and Modoc Counties and their expected changes, the impacts of prisons on local communities, and the decisionmaking process managing natural resources. The third chapter (Chapter 5.3, Flores and Haire, this synthesis, *Ecosystem Services and Public Land Management*) examines the social benefits provided by ecosystems, the economic benefits of these services, and societal inputs into resource management. Community engagement is the focus of the fourth chapter (Chapter 5.4, Flores and Stone, this synthesis, *Community Engagement in the Decisionmaking Process for Public Land Management in Northeastern California*), and hones in on how communities can participate in natural resource management and use that engagement to resolve conflict. How tribes value place, interact with managers, and use fire management is the theme of the fifth chapter (Chapter 5.5, Flores and Russell, this synthesis, *Integrating Tribes and Culture Into Public Land Management*).

Responding to Disturbances is the final section. Its single chapter (Chapter 6.1, Wright, this synthesis, *Ecological Disturbance in the Context of a Changing Climate: Implications for Land Management in Northeastern California*) takes a broad look at various factors that are currently, or may in the future, affect these ecosystems, how a changing climate interacts with those disturbances, and what possible management techniques could be considered to mitigate disturbances.

References

California Department of Water Resources. 1998. Executive Summary Chapter 4: Urban, agricultural, and environmental water use. In: The California water plan update. Sacramento, CA: California Department of Water Resources, Planning and Local Assistance. Bulletin 160-98. 17 p. https://www.water.ca.gov/waterplan/docs/previous/b160-98/esch4.pdf (3 Oct. 2017).

Charnley, S.; Long, J. 2014. Managing forest products for community benefit. In: Long. J.W.; Quinn-Davidson, L.N.; Skinner, C.N., eds. 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: 629-661. Chapter 9.5.

Fuller, M.; Brown, S.; Wills, C.; [et al.], eds. 2015. Modoc Plateau Geomorphic Province. In: Fuller, M.; Brown, S.; Wills, C.; [et al.], eds. Geological Gems of California, GeoGem Note 26. California Geological Survey Special Report 230 under Interagency Agreement C01718011 with California State Parks.

Gonzales, A.G.; Hoshi, J., eds. 2015a. Cascades and Modoc Plateau Province. In: California state wildlife action plan, 2015 update: a conservation legacy for Californians. Sacramento, CA: California Department of Fish and Wildlife with assistance from Ascent Environmental Inc. 51 p. Chap. 5.2, Vol. 1. With assistance from Ascent Environmental Inc. https://www.wildlife.ca.gov/SWAP/Final (2 Oct. 2017).

Gonzales, A.G.; Hoshi, J., eds. 2015b. Central Valley and Sierra Nevada Province. In: California state wildlife action plan, 2015 update: a conservation legacy for Californians. Sacramento, CA: California Department of Fish and Wildlife with assistance from Ascent Environmental Inc. 93 p. Chap. 5.4, Vol. 1. https://www.wildlife.ca.gov/SWAP/Final (2 Oct. 2017).

Griffith, G.E.; Omernik, J.M.; Smith, D.W.; [et al.]. 2016. Ecoregions of California (poster): U.S. Geological Survey Open-File Report 2016-1021, with map, scale 1:1,100,000. Reston, VA: U.S. Department of the Interior, U.S. Geological Survey. https://dx.doi.org/10.3133/ofr20161021 (21 Aug. 2019).

King, J.L.; Simovich, M.A.; Brusca, R.C. 1996. Species richness, endemism and ecology of crustacean assemblages in northern California vernal pools. Hydrobiologia. 328(2): 85–116.

Long, J.W.; Pope, K.L. 2014. Wet meadows. In: Long, J.W.; Quinn-Davidson, L.N.; Skinner, C.N., eds. 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: 341–372. Chap. 6.3.

Long, J.W.; Quinn-Davidson, L.N.; Skinner, C.N. 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.

Moyle, P.B.; Katz, J.V.E.; Quiñones, R.M. 2011. Rapid decline of California's native inland fishes: a status assessment. Biological Conservation. 144: 2414–2423.

North, M. 2012. Managing Sierra Nevada forests. Gen. Tech. Rep. PSW-GTR-237. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 184 p.

North, M.; Stine, P.; O'Hara, K.; [et al.]. 2009. An ecosystem management strategy for Sierran mixed-conifer forests. Gen. Tech. Rep. PSW-GTR-220 (2nd printing, with addendum). Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 49 p.

Omernik, J.M. 1987. Ecoregions of the conterminous United States. Map (scale 1:7,500,000). Annals of the Association of American Geographers. 77(1): 118–125.

Peacock, M.A. 1931. The Modoc lava field, Northern California. Geographical Review. 21: 259–275.

Riegel, G.M.; Miller, R.F.; Skinner, C.N.; [et al.]. 2006. Northeastern plateaus bioregion. In: Sugihara, N.G.; van Wagtendonk, J.W.; Fites-Kaufman, J.; [et al.], eds. Fire in California's ecosystems. Berkeley, CA: University of California Press: 225–263. Chap. 11.

Simovich, M.A. 1998. Crustacean biodiversity and endemism in California's ephemeral wetlands. In: Witham, C.W.; Bauder, E.T.; Belk, D.; [et al.], eds. Ecology, conservation, and management of vernal pool ecosystems–proceedings from a 1996 Conference. Sacramento, CA: California Native Plant Society: 107–118.

Skinner, C.N.; Taylor, A.H. 2006. Southern Cascades bioregion. In: Sugihara, N.G.; van Wagtendonk, J.W.; Fites-Kaufman, J.; [et al.], eds. Fire in California's ecosystems. Berkeley, CA: University of California Press: 195–224. Chap. 10.

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.

Appendix 1.1—Descriptions (Griffith et al. 2016) of the Main Level IV Ecoregions Found Within the Broader Level III Ecoregions Occurring Within the Lassen and Modoc National Forests.

Level III Ecoregion 4—Cascades

4e The **High Southern Cascades Montane Forest Ecoregion** is an undulating, volcanic plateau containing isolated buttes, cones, and peaks. Some parts of the region are glaciated. The terrain often is less dissected than that of Ecoregion 4f. Elevations of Ecoregion 4e are generally intermediate to elevations in the Low Southern Cascades Mixed Conifer Forest (4f) and the Cascade Subalpine/Alpine (4d) ecoregions. In California, elevations of the ecoregion are mostly 5,500 to 8,500 feet (1,676–2,591 m), but are lower in Oregon at 4,000 to 8,200 feet (1,219–2,499 m). Cryic soils support mixed-coniferous forests dominated by mountain hemlock, lodgepole pine, and, in Oregon, some Pacific silver fir; the soils are colder than the mesic and frigid soils of the Low Southern Cascades Mixed Conifer Forest Ecoregion (4f). White fir and Shasta red fir also occur in the ecoregion, with some grand fir in Oregon. This region has a longer summer drought and more intermittent streams than the Cascade Crest Montane Forest Ecoregion (4c) to the north in Oregon and Washington.

4f The **Low Southern Cascades Mixed Conifer Forest Ecoregion** generally is lower in elevation and less rugged than the more highly dissected Western Cascades Montane Highlands (4b) to the north in Oregon. Although still mostly a mesic mixed-conifer region, the climate is drier than in the Western Cascades of Oregon (Ecoregions 4a and 4b), and the vegetation reflects this. Western hemlock and western redcedar, indicator species of Ecoregions 4a and 4b, decrease in abundance southward in this ecoregion in Oregon and are replaced by Sierra Nevada species, such as incense cedar, white fir, Shasta red fir, and Jeffrey pine that tolerate prolonged summer drought. In California, shrubs such as manzanita and ceanothus are common. Curlleaf mountain-mahogany, big sagebrush, and antelope bitterbrush occur as well, with their dispersion centers in the Great Basin regions farther east. River and stream discharge is significantly less than in systems to the north. Soil temperature regimes are mesic and frigid, and the soil moisture regime is xeric. Elevations in the California part of this ecoregion are about 3,000 to 7,600 feet (914–2,316 m).

4g The **California Cascades Eastside Conifer Forest Ecoregion** is drier than the other California Cascades regions. It is dominated by ponderosa pine and, in some areas where conditions are harsher, Jeffrey pine. In lower, drier areas, the region blends into the western juniper and sagebrush fields more typical of adjacent Ecoregion 9. The region wraps around to the western side (that is, the Mount Shasta foothills), as similar dry conditions exist from the rain shadow cast by the Klamath Mountains to the west. Elevations range from 3,000 to 7,100 feet (914–2,164 m).

4h The **Southern Cascades Foothills Ecoregion** of volcanic hills and plateaus is mostly in the 2,000- to 4,000-foot (610–1,219 m) elevation range, stretching from the town of Paradise in the south to the Pit River in the north. It contains dry-mesic mixed-conifer forest and lower montane black oak-conifer forest and woodland. Ponderosa pine is abundant along with some Douglas-fir, and, at higher elevations, white fir. Hardwoods typically are black oak and canyon live oak. Soil temperature regimes are mostly mesic with some frigid, and soil moisture regimes are xeric.

Level III Ecoregion 5—Sierra Nevada

5c The **Northern Sierra Upper Montane Forests Ecoregion** ranges in elevation mostly from 6,000 to 8,000 feet (1,829–2,438 m), and its forests have a mix of conifers, including red fir, white fir, Jeffrey pine, sugar pine, incense cedar, and some lodgepole pine. Intermixed are areas of quaking aspen groves. Some montane chaparral also occurs in areas of harsh exposure, repeated fires, and clear cuts. Geology types are mostly Mesozoic granitic rocks and Tertiary volcanics, although in the north these occur along with some areas of slate, sandstone, metavolcanics, and metasedimentary rocks. Soil temperature regimes are mostly frigid, with some cryic. Soil moisture regimes are mostly xeric, but are udic in areas where snow persists through spring.

5d In contrast to the volcanic and granitic rocks of similarly zoned Ecoregion 5g to the south, in the **Northern Sierra Mid-Montane Forests Ecoregion,** metamorphic rocks are abundant, with Paleozoic metasedimentary argillite, phyllite, and quartzite, and some metavolcanics, as well as Mesozoic peridotite and serpentine. Elevations range mostly from 3,000 to 6,000 feet (914–1,829 m), with some lower west-end canyon elevations. This higher-elevation mixed-conifer forest has more white fir and Douglas-fir and less ponderosa pine than in the lower-elevation Ecoregion 5h to the south. Jeffrey pine occurs on ultramafic rocks and some drier areas to the east. Black oak and tanoaks are common hardwoods, along with canyon live oak. Soil temperature regimes are mostly mesic and soil moisture regimes are mostly xeric.

5e Generally lower in elevation than adjacent Ecoregion 5d, the **Northern Sierra Lower Montane Forests Ecoregion** has a mix of montane hardwood, montane hardwood-conifer, and mixed-conifer forests. Elevations range mostly from 2,000 to 4,000 feet (610–1,219 m), with a few higher areas. Ecoregion 5e has less ponderosa pine than Ecoregions 5h and 5n to the south. Douglas-fir is a more widespread conifer, and hardwoods include canyon live oak, interior live oak, black oak, and tanoak. Annual precipitation is somewhat higher than in Ecoregion 5h immediately to the south. Geology is a complex mix of Mesozoic granitic rocks, Jurassic to Triassic metavolcanics, and some Mesozoic to Paleozoic metasedimentary and ultramafic rocks.

5f The **Northeastern Sierra Mixed Conifer-Pine Forests Ecoregion** includes many of the drier eastside forests of the Northern Sierra Nevada that occur north of Bridgeport, in the Lake Tahoe area, and to the northern extent of the Sierra near Susanville. These are mid-elevation dry forests, typically between 5,000 and 8,000 feet (1,524–2,438 m), with a diverse mix of conifers, such as Jeffrey, ponderosa, and sugar pines; incense cedar; and California white fir. The understory can include sagebrush, antelope bitterbrush, and a fire-maintained chaparral component of snowbrush and manzanita. Soil temperature regimes are frigid and soil moisture regimes are mostly xeric.

Level III Ecoregion 6—Central California Foothills and Coastal Mountains

6a The **Tuscan Flows Ecoregion** is a gently southwest-sloping plateau with some steep canyons and a few steep volcanic cones. Although the region is geologically related to the southwestern end of the Cascades Ecoregion (4), it has ecosystem similarities to the Sierra Nevada foothills part of Ecoregion 6. Blue oak woodlands, annual grasslands, and foothill pine occur.

Level III Ecoregion 9—Eastern Cascades Slopes and Foothills

9g The **Klamath/Goose Lake Basins Ecoregion** covers river floodplains, terraces, and lake basins. Various wildrye, bluegrass, hairgrass, sedge, and rush species once covered the basins, but most of the wet meadows and wetlands have been drained for agriculture. Sagebrush and bunchgrass occur on most of the upland areas. Several marshland wildlife refuges here are critical to preserving the regional biodiversity, particularly for at-risk bird and fish species. In California, Butte Valley also is included in the ecoregion. Although the Butte Valley area differs somewhat from the Lower Klamath and Tule Lake Basins, it also has pasture and cropland.

9h The **Fremont Pine/Fir Forest Ecoregion** occurs mostly in Oregon, with a small area west of Goose Lake in California. It contains mid-elevation mountains and high plateaus that rarely exceed timberline. Closed-canopy forests contrast with the savanna of the Klamath Juniper Woodland/Devil's Garden Ecoregion (9j). Ponderosa pine is widespread, but white fir, sugar pine, lodgepole pine, and incense cedar also grow at elevations greater than 6,500 feet (1,981 m) and on northern slopes. Residual soils are common in contrast to ecoregions farther north in Oregon where residual soils have been buried by pumice and ash. Ecoregion 9h has a high density of lakes and reservoirs.

9j The **Klamath Juniper Woodland/Devil's Garden Ecoregion** is composed of undulating hills, benches, and escarpments covered with a mosaic of rangeland and woodland. Western juniper grows on shallow, rocky soils with an understory of low sagebrush, mountain big sagebrush, bitterbrush, and bunchgrasses. Other shrubland/grasslands include shrub species uncommon in eastern Oregon, such as woolly mule-ears, Klamath plum, and birchleaf mountain-mahogany. The diverse shrublands provide important wildlife habitat. Reservoirs dot the landscape and are important to lowland irrigation. Soil temperature regimes in the California part of Ecoregion 9j are mesic, whereas soil temperatures in the Oregon part are mesic and frigid.

The Devil's Garden place name appears on topographic maps in both States, 60 miles (97 km) apart.

9m The **Warner Mountains Ecoregion** comprises the low to mid-elevations of the Warner Mountains. In contrast to the mesic soils of the surrounding lower ecoregions, soil temperature regimes here are frigid. Soil moisture regimes are xeric. Vegetation includes big sagebrush, low sagebrush, perennial bunchgrasses, and western juniper at low elevations. At higher elevations, ponderosa pine, Jeffrey pine, Washoe pine, aspen, and white fir are common. Streams on the western side of this fault-block mountain drain to Goose Lake or the Pit River, and streams on the eastern side, which are much shorter, drain to Surprise Valley in Ecoregion 80.

9n The **High-Elevation Warner Mountains Ecoregio**n is a subalpine/alpine area that ranges in elevation from about 7,800 to 9,892 feet (2,377–3,015 m) at Eagle Peak. This high-elevation zone contains aspen and lodgepole pine, with whitebark pine as the primary subalpine and timberline tree above about 8,500 feet (2,591 m). Drought-tolerant alpine cushion plants also occur. Soil temperature regimes are cryic. Annual precipitation is about 28–32 inches (711–813 mm) with deep winter snow.

9o The small **Likely Tableland Ecoregion** is a gently sloping footslope west of the Warner Mountains (9m). Relief is only about 100–300 feet (30.5–91 m). Elevations range from 4,500 to 5,500 feet (1,372–1,676 m). The sagebrush and grassland of Ecoregion 9o contrast with the adjacent higher relief hills and mountains of Ecoregion 9p that have abundant piñon-juniper woodland and some pine. Annual precipitation is only 12–15 inches (305–381 mm). The Quaternary andesite and basalt are younger than the Tertiary volcanics of Ecoregion 9p. Some perennial streams cross the region from the Warner Mountains to the Pit River. Soil temperature regimes are mesic and moisture regimes are aridic and xeric.

9p The **Modoc/Lassen Juniper-Shrub Hills and Mountains Ecoregion** is a faulted and eroded volcanic plateau with many volcanic hills and mountains. Soil temperature regimes are mesic and frigid. Soil moisture regimes are mostly xeric and aridic. Vegetation is mainly western juniper, big sagebrush, and perennial bunchgrasses. On the higher-elevation mountains, however, some small areas of Jeffrey pine, ponderosa pine, and white fir occur. There are few streams and most of those are dry during summer. Streams that flow from the ecoregion drain to either the Pit River, the Madeline Plains, or in the south, to the Honey Lake Valley.

9q The **Adin/Horsehead Mountains Forest and Woodland Ecoregion** consists of mountains to the north, east, and south of Big Valley. Elevations range from about 5,000 to 7,036 feet (1,524–2,145 m). Ecoregion 9q has more pine and mixed-conifer forest than the drier Ecoregion 9p to the east or the adjacent lower Ecoregion 9r. Soil temperature regimes are frigid and mesic and soil moisture regimes are mostly xeric. Vegetation is a mix of Jeffrey pine, ponderosa pine, and some white fir, and at low elevations, western juniper, big sagebrush, birchleaf mountain-mahogany, and other deciduous shrubs. There are a few streams and springs, but almost no lakes.

9r The **Adin/Dixie Low Hills Ecoregion** consists of hills and lava plateaus to the west of Ecoregion 9q that are lower in elevation (4,000–6,000 feet; 1,219–1,829 m) with less relief. The vegetation is mostly big sagebrush, low sagebrush, and scattered junipers, and generally lacks the pine of the nearby mountain ecoregions (9q, 4g). Soil temperature regimes are mesic and soil moisture regimes are xeric. Some streams cross the ecoregion to the Pit River, and several small shallow reservoirs occur here. Ranching and livestock grazing are predominant land uses.

9s The **Modoc Lava Flows and Buttes Ecoregion** is a volcanic plateau surrounding the Medicine Lake Highlands of the Cascades Ecoregion (4). It is lower and drier than those highlands, with more juniper and pine. Soil temperature regimes are mesic and soil moisture regimes are aridic and xeric. Vegetation includes western juniper, big sagebrush, and native perennial grassland. Water drains down through joints in the basalt rock to the groundwater reservoir, limiting overland flow of water and development of stream channels on the plateau.

Reference

Griffith, G.E.; Omernik, J.M.; Smith, D.W.; [et al.], 2016. Ecoregions of California (poster): U.S. Geological Survey Open-File Report 2016–1021, with map, scale 1:1,100,000. Reston, VA: U.S. Department of the Interior, U.S. Geological Survey. http://dx.doi.org/10.3133/ofr20161021 (21 Aug. 2019).

Section 2. Forestland

Chapter 2.1. Understanding and Managing the Dry Conifer Forests of Northeastern California

W. Keith Moser[1]

Introduction

The volcanic soils of the Southern Cascades and the Modoc Plateau support a diverse assemblage of conifer species. On the western slopes of the Southern Cascades, Douglas-fir (*Pseudotsuga menziesii*), ponderosa (*Pinus ponderosa*) and lodgepole (*Pinus contorta*) pines, and white (*Abies concolor*) and Shasta red firs (*Abies magnifica*) are common, whereas ponderosa and Jeffrey (*Pinus jeffreyi*) pines inhabit the drier east slopes. Eastern slopes and valleys commonly contain big sagebrush (*Artemisia* species) and western juniper (*Juniperus occidentalis*) habitats. The Great Basin, with its high desert plant assemblages of sagebrush- and shrub-steppe and piñon-juniper woodlands extends onto the Modoc Plateau in Northeastern California, where the piñon pine (as *Pinus monophylla*) component is rarely represented. These communities are shaped by patterns of fire frequency and intensity, successional dynamics and community structure, and soil type and quality (Gonzales and Hoshi 2015). As discussed in Chapter 1.1 (Dumroese, this synthesis, *The Northeastern California Plateaus Bioregion Science Synthesis: Background, Rationale, and Scope*), forest staffs and the public commented that portions of the forest and woodland ecosystems of the Lassen and Modoc National Forests (hereafter, the Lassen and Modoc) required attention beyond what was provided by two other syntheses that included these forests: the *Science Synthesis to Support Socioecological Resilience in the Sierra Nevada and Southern Cascade Range* (hereafter, Sierra Nevada Science Synthesis) published by Long et al. (2014) and the *Synthesis of Science to Inform Land Management Within the Northwest Forest Plan Area* (hereafter, Northwest Forest Plan Science Synthesis) published by Spies et al. (2018). These two valuable syntheses serve both to geographically frame the region that will be discussed in this *Northeastern California Plateaus Bioregion Science Synthesis* (hereafter, Plateaus Science Synthesis) and provide an excellent template for this author to adapt to the needs of the Lassen and Modoc.

National forests provide many different benefits to the Nation, including biodiversity, recreation (fig. 2.1.1.) and economic values. Often these benefits can be achieved through compatible management strategies; in other instances, priorities must be identified. This chapter does not seek to prioritize any benefits or actions as that process is more properly dealt with in the forest plan. However, no benefits will be provided in the future if forested ecosystems are no longer vigorous and productive. Given the diverse land use histories and diverse expectations about future climatic scenarios, this chapter will focus on what is known about the science of restoration in ecosystems in the Great Basin side of the Lassen and Modoc National Forests (see textbox 2.1.1 and see the discussion on restoration found later in this chapter). This discussion is often framed using examples, where appropriate, from similar ecosystems elsewhere in the

Figure 2.1.1—Fall colors in the Granger Canyon, Modoc National Forest. Given the importance of recreation on our national forests, managing scenic diversity is an important component of resource decision making (photo by Ken Sandusky, Forest Service).

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Great Basin with the goal of enhancing these ecosystems to be resistant to those disturbances they can resist and be resilient in the face of disturbances they cannot.

Based on the input during the public scoping process and from staffs of the Lassen and the Modoc, this chapter focuses on three issues in the context of dry-site ponderosa/Jeffrey pine and juniper woodlands:

- Pre-European settlement conditions and how they might guide current management,
- Fire history and its impact on current and future structure and composition, and
- In the face of changing climate, wildfire patterns, and local needs, how to increase resilience and resistance (see *Resistance and Resilience* below for definitions).

Textbox 2.1.1

- **Restoration** is the "resetting the ecological clock"–the returning of an ecosystem to a composition and structure that predates the disturbance.
- **Rehabilitation** repairs damage but does not focus on creating the former state.
- **Reallocation** may mend damage but pushes the ecosystem to an alternative land use or system. Millar et al. (2007) call this last action a "response," the transitioning of an ecosystem from the present undesirable or unsustainable state to new conditions.
- All three of these processes can involve **Repair**, which covers a variety of remediative actions, whether focused on the species, landscape, or ecosystem level.
- Hobbs and Cramer (2008), North et al. (2009b), and others presented a more detailed elaboration of restoration processes.

Topics Covered Under the Previous Syntheses

The Northwest Forest Plan Science Synthesis and the Sierra Nevada Science Synthesis have extensive coverage of important forest types that occur on the Lassen and the Modoc (fig. 2.1.2). Specifically, the forest types defined by Level IV Ecoregions 4e (High Southern Cascades Montane Forest), 4f (Low South Cascades Mixed-Conifer Forest), 4g. (California Cascades Eastside Conifer Forest), and 4h (Southern Cascades Foothills) were covered under the Northwest and Sierra Nevada science syntheses (see Appendix 1.1 in Chapter 1.1, Dumroese, this synthesis, *The Northeastern California Plateaus Bioregion Science Synthesis: Background, Rationale, and Scope* for full ecoregion descriptions). In addition, the Sierra Nevada Science Synthesis discussed several other forested ecoregions, especially those of Level III Ecoregion 4 (Sierra Nevada). Thus, the focus of this synthesis is largely on the Level IV Ecoregions that the public and forest staffs believed needed additional attention: eastside mixed-conifer forests (4g, 9q; mainly mixtures of ponderosa and Jeffrey pines) and juniper woodlands (9j) (fig. 2.1.3).

The Effect of Drought on Structure and Composition in Dry Forests

Any discussion of Great Basin ecosystems must acknowledge the limited annual precipitation coupled with hot summer temperatures that results in high rates of evapotranspiration and subsequent plant stress (Houghton et al. 1975). Water availability is one limit to the growing space of a tree or stand (Oliver and Larson 1996) and a combination of individual tree, species, and stand characteristics determines the landscape response to drought events. Fluctuations in climatic variables, including precipitation, place demands upon natural ecosystems; ecosystems are subject to even greater demands if they have been influenced by past management actions, including livestock grazing and fire suppression. Weather variability can create conditions that set up forests for a fall. For example, the climate of the Sierra Nevada since 1850 has been warmer and wetter than between 1650 and 1850 (Stine 1996, cited in Raumann and Cablk 2008), favoring conifer establishment and expansion. Even short-term fluctuations can result in severe disturbance events. For example, in Florida the above-average precipitation in winter 1997-98 resulted in a flush of ground vegetation growth that was all the more susceptible to the subsequent spring-summer drought, resulting in the most severe wildfire season in the State's history (Butry et al. 2001).

Temperature exacerbates any decrease in precipitation. A key element of any discussion of drought-influenced mortality in the Western United States is the expectation of higher temperatures during this century compared to the climate normal. Drought stress is a function of the high drying power of air (vapor pressure deficit or VPD) as well as lower available soil moisture. Vapor pressure deficit increases with increasing temperature and reduced relative humidity (Bradford and Bell 2017; Pallardy 2008). Average temperatures in the region are projected to increase by 3.2 to 4.3 °F (1.7 to 2.2 °C) by 2070 compared

Figure 2.1.2—Mount Shasta from Happy Camp on the Modoc National Forest. Northeastern California is situated at the intersection of the Southern Cascades, the northernmost Sierras, and the western arm of the Great Basin (photo by Chris Bielecki, Forest Service).

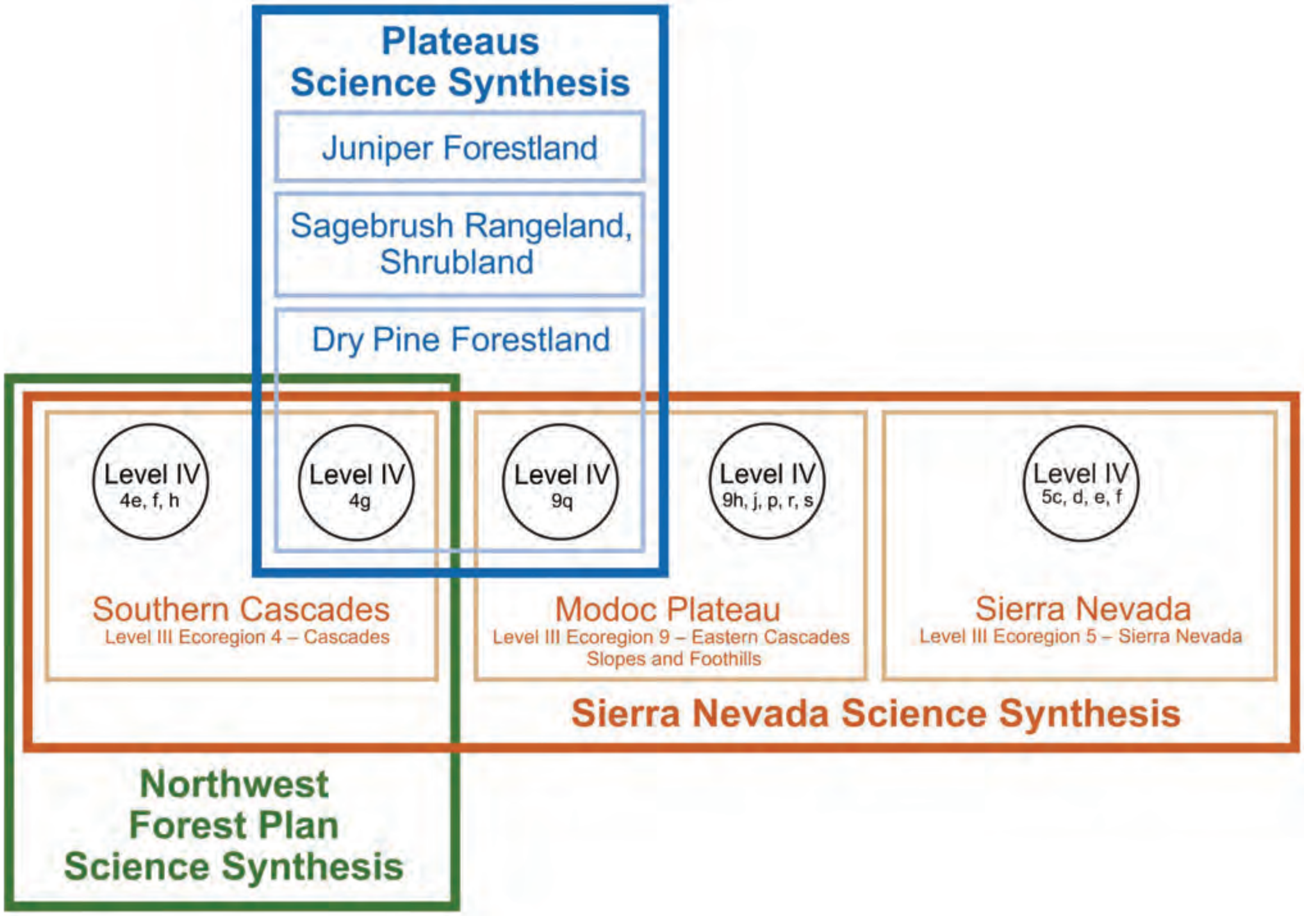

Figure 2.1.3—Ecosystem delineations used in the Northwest Forest Plan, Sierra Nevada, and Plateaus science syntheses. Because portions of the Northwest Forest Plan and Sierra Nevada Science Synthesis adequately covered the Lassen and Modoc National Forests, the Plateaus Science Synthesis focuses mainly on Dry Pine Forestland and Juniper Forestland, Sagebrush Rangeland, and Shrubland.

to the historical baseline of 1979–2009, and average precipitation is projected to decrease by 2 inches (5 cm) by 2100 ((Gonzales and Hoshi 2015; PRBO Conservation Science 2011).

Species Shift

A study of large trees and seedlings on U.S. Department of Agriculture, Forest Service, Forest Inventory and Analysis (FIA) plots found that climatic influences resulted in contractions and shifts of species, more so for montane species than subalpine (Bell et al. 2014). Twenty-nine percent of the contractions were movements away from warmer climates, perhaps reflecting the influence of increased winter precipitation. Although greater winter precipitation can aid the growth of large trees, it generally does not help with seedling establishment and growth, which are much more dependent on summer season precipitation due to the reduction of winter soil moisture before the growing season temperatures arrive (Pearson 1920).

Projections of future changing climate suggest that many variables will be altered, as has happened during climate events of the past. Some of these will be changes in absolute and seasonal precipitation and temperature, variability, and extremes. Depending upon the temperature variations, some of the changes in vegetation may be quite dramatic. Disturbances such as severe fire may accelerate these changes that might otherwise lag the temperature differences (Nolan et al. 2018). Projected temperatures may particularly affect vegetation composition and structure. The resulting vegetation communities may be quite novel in their makeup. Given the rapidity of projected change, some of the vegetation states may be quite temporary, with some projecting ecosystem stability being reached only in the 22nd century (Nolan et al. 2018).

Despite the potential for long-term increased temperature and drought to eliminate a species or even an ecosystem, particularly at the edge of its range (Bradley 2010; Miller and Wigand 1994), a more probable outcome is a modification of the composition and structure of landscapes currently forested (Clark et al. 2016). Drought-influenced mortality occurs more rapidly under higher temperatures (Vankat 2013). But projected environmental conditions, especially drought, are difficult to model because of the novelty of future climate scenarios and the complexity of interactions among climate, land use history, and disturbance.

In the Western United States, widespread drought coincides with a decline in tree growth (Clark et al. 2016), although not all drought events necessarily result in mortality (Van Gunst et al. 2016). Despite ponderosa pine's superior stomatal control and resistance to cavitation (the formation of embolisms within the plant's water-conducting tissues that cause hydraulic failure) (Maherali and DeLucia 2000), dry forest ecosystems can be susceptible to further droughty conditions (Bradford and Bell 2017), either through cavitation or carbon starvation, as the frequently closed stomata do not permit enough carbon uptake to maintain the system (North et al. 2009a). High temperatures during the growing season combined with reduced precipitation in the winter and early spring can explain much of the reduced growth rates of conifers in the Southwestern United States (Williams et al. 2013).

Competition for soil moisture can intensify the nominal effects of a drought. A comparative study of ponderosa pine in Arizona and South Dakota and red pine (*Pinus resinosa*) in Minnesota found that trees in low-density treatments displayed higher resistance and resilience to drought than trees in higher density treatments (Bottero et al. 2016). That study reported a negative relationship between growth and resistance and resilience to drought and tree population. Thinnings that reduce density can increase resistance and/or resilience to drought, at least for younger or smaller trees (D'Amato et al. 2013). Bradford and Bell (2017) found that tree mortality was positively related to temperature extremes and negatively related to winter and spring precipitation. Without the influence of other climatic factors, most of the species being studied displayed a strong relationship between mortality and density. Actions that target timber yields or fuels reduction, such as density reduction or prescribed fires, can also result in reduced drought vulnerability as fewer surviving trees are competing for a fixed amount of soil moisture. Multi-aged management can also create diverse size and age classes and therefore distribute risk (Clark et al. 2016; O'Hara 2002). But density reduction and similar measures may produce undesirable results, such as increased transpiration from larger trees with larger leaf/sapwood ratios (Clark et al. 2016) and temporary higher respiration loads of the remaining trees in newly thinned stands (Pallardy 2008) that can cause tree stress. In most piñon-juniper ecosystems, severe drought can result in the removal (by death) of water-demanding piñon. Thinning systems can certainly emulate this process and seek to favor more drought-tolerant species such as western juniper (Clark et al. 2016).

Researchers have a substantial understanding of the effects of drought on individual trees but less of an understanding about drought effects on communities or landscapes. Higher levels of carbon dioxide can increase water-use efficiency in drier areas, spurring further increases in the extent of woody encroachment into grasslands (Bradley 2010). Individuals in the canopy that might respond positively to favorable growing conditions are affected as well by neighbors that might also respond to those growing conditions (Clark et al. 2016). Any of several paradoxical outcomes may ensue. Faster growing trees might reach resource limitations more quickly, resulting in competition-induced mortality (Assmann 1970; Clark et al. 2016). Conversely, drought can slow growth and therefore reduce competition-induced mortality. The reduced vigor, however, can increase vulnerability to forest health issues, such as declines or even direct insect or disease attack (Manion 1991), as many pathogens can tolerate water stress better than the host trees (Clark et al. 2016).

Eastside Ponderosa Pine Forests[1]

Pre-European Settlement Conditions and How They Might Guide Current Management

Ponderosa pine occurs throughout the Western United States, with its greatest extent in Northeastern California (5 million acres, or 2.1 million ha) and the Northwestern Inland West (Graham and Jain 2005). Two varieties exist: the Rocky Mountain variety (*Pinus ponderosa* var. *scopulorum*), and the Pacific variety (*Pinus ponderosa* var. *ponderosa*), which ranges from British Columbia to California and Northwest Nevada (USDA NRCS 2017). It is generally believed that prior to Euro-American settlement (hereafter, simply "presettlement") ponderosa pine forests throughout the species' range were composed of large trees randomly distributed in an open, "park-like" stand of clumps at a large scale with few seedlings and saplings below the canopy (Sudworth 1900). The stands were believed to be uneven-aged, with occasional large individual trees 400 to 600 years old, which made them popular as witness trees in the region (Youngblood et al. 2004). Small gaps of even-aged cohorts were common (Crotteau and Ritchie 2014; Safford and Stevens 2017) although the frequent low- and moderate-severity events that historically occurred left more of a fine-grained pattern on the landscape that is difficult to assess years later (Safford and Stevens 2017). The trees were large. One traveler in the area of the Lassen National Forest observed a stand of pines that had trees more than 10 feet (3 m) in diameter and up to 200 feet (61 m) tall (Reed and Gaines 1949; Safford and Stevens 2017). Van Hooser and Keegan (1988) found ponderosa pines in California exceeding 6 feet (183 cm) in diameter. While many might picture a balanced uneven-aged forest to have a reverse-J-shaped diameter distribution (de Liocourt 1898), Safford and Stevens (2017) point out that this is the distribution for an undisturbed forest. Historical studies found a flat or hump-shaped diameter distribution in old-growth forests with frequent low-severity fires (Lydersen and North 2012; North et al. 2007; Oliver 2001) and others as cited in Safford and Stevens (2017). Oliver (2001) demonstrated prescriptions for converting a J-shaped diameter distribution into a hump-shaped diameter distribution using fire. Taylor (2010) observed similar results from fire on studies on Beaver Creek and Devil's Pinery on the Lassen National Forest.

Ponderosa pine is an intermittent seeder (Bailey and Covington 2002; Savage and Swetnam 1990; White 1985). The study of old-growth stands by Youngblood et al. (2004) found that seed crops occurred about every 4 to 5 years with at least one seed crop every decade since 1850. Taylor (2010) found adequate seed crops occurring every 2 to 3 years. Due to above-normal precipitation from the middle of the 19th century through the second decade of the 20th century (Graumlich 1987; Taylor 2010), a higher proportion than normal of the seedfall successfully germinated (Garfin and Hughes 1996; Youngblood et al. 2004). As ponderosa pines often establish in gaps created by the death of one or more trees in the overstory, ponderosa pine forests start out as clumped. As they age, within-species competition thins them to a wider and more random pattern across the stand (Youngblood et al. 2004).

The species has evolved to survive in a fire-maintained ecosystem. Trees as small as 2 inches (5 cm) in diameter breast height (d.b.h.) can withstand the heat from most surface fires thanks to thick, insulating bark (Graham and Jain 2005). Bailey and Covington (2002) found 10-foot-tall (3-m) saplings with 4 inches (10 cm) of root collar diameter are fire resistant.

Although historical ponderosa pine forests may have been multi-aged, they did not necessarily have a multi-layered

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[1] A comprehensive summary of ponderosa pine forests in Northern California can be found in Safford, H.D.; Stevens, J.T. 2017. Natural range of variation for yellow pine and mixed-conifer forests in the Sierra Nevada, southern Cascades, and Modoc and Inyo National Forests, California, USA. Gen. Tech. Rep. PSW-GTR-256. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 229 p

canopy. O'Hara et al. (1996) referred to such a structure as "old forest single stratum" (Youngblood et al. 2004). Munger (1917) observed ponderosa pine stands containing 12 to 40 trees of 12-inch (30.5 cm) diameter or larger per acre (30 to 99 trees per hectare [tph]) with very few seedlings or saplings in the understory (Fitzgerald 2005). On a protected old-growth site on the Blacks Mountain Experimental Forest in the Lassen, Youngblood et al. (2004) measured an overall density of 20 ± 1.4 trees per acre (50 ± 3.5 tph) with a mean diameter of about 24 inches (about 60 cm), with the distribution range from 10 to 49 inches (25 to 125 cm) d.b.h. These results were comparable to a study of an old-growth stand in Northern Arizona of 26 ± 2.8 trees per acre (65 ± 7 tph) (Covington et al. 1997), where fire has historically been the key disturbance (Moore et al. 1999).

Ponderosa Pine and Spacing

The mosaic of open space and grouping that Youngblood et al. (2004) observed most likely affords some resistance to fire by breaking up the canopy and reducing the vulnerability to crown fires (Fitzgerald 2005). Youngblood et al. (2004) found that no more than 3 to 15 percent of presettlement old-growth ponderosa pine forests in the Eastside Cascades of Oregon and California still existed. Their study of three old-growth forests in eastern Oregon and Northeastern California found that on two of the forests, at small scales, overstory trees were randomly distributed, and at larger scales, the trees were clumped, with the maximum radius of the clump being 70 to 80 feet (22 to 24 m). Trees in the upper canopy class that were aged were almost without exception larger than 14 inches (36 cm) diameter and older than 100 years (Youngblood et al. 2004).

Vegetative Complexes of Ponderosa Pine

Eastside Ponderosa Pine and Associates

Associates of ponderosa pine in California include western juniper, quaking aspen (*Populus tremuloides*), Jeffrey pine, lodgepole pine, Douglas-fir, incense cedar (*Calocedrus decurrens*), California black oak (*Quercus kelloggii*), and white fir (Oliver and Ryker 1990). In Northeastern California, ponderosa pine and Jeffrey pine forests grow on deep, well-drained soils. On dry sites south of Alturas, CA, ponderosa and Jeffrey pine are found together and the two species are often managed in the same manner. Ponderosa pine is the only native yellow pine north of Alturas (fig. 2.1.4). Jeffrey pine is generally found on lower-productivity sites than ponderosa pine. The lower productivity in Jeffrey pine stands probably resulted in longer fire-return intervals than on sites with ponderosa pine (Taylor 2000) and others as cited in Riegel et al. (2006a). On slightly moister sites, such as the west slope of the Sierras and the east side of the Cascades, ponderosa pine becomes more of a mid-seral species with Douglas-fir becoming a late-seral species. On sites where white fir is the potential vegetation type, disturbance, particularly fire, will keep the forest cover in ponderosa pine (Graham and Jain 2005).

Figure 2.1.4—Ponderosa pine is the dominant tree species on the dry forested areas of much of the Lassen and Modoc National Forests (photo by Ken Sandusky, Forest Service).

Stand Structure of Ponderosa Pine

Non-lethal fire caused by humans and lightning were instrumental in maintaining ponderosa pine forests in the Western United States (Graham and Jain 2005). Frequent dry-lightning days coincided with the season when the fine fuels were the driest. Frequent low fires killed small stems by cambial scorch, thus performing a thinning from below (Youngblood et al. 2004). Although these fires

Figure 2.1.5—The clumpy nature of ponderosa pine regeneration is apparent on the Lassen National Forest (photo by Bonnie Lou Millar, Forest Service).

that maintained ponderosa pine forests usually caused little mortality, fire intensity varied across the landscape. Intensive burning of patches followed by ponderosa pine regeneration would create a mosaic of even-aged patches. Variation of fire intensity in a patch would create uneven-aged patches (Taylor 2010), albeit often resulting in a fine-grained landscape (Safford and Stevens 2017). The clumpy nature of ponderosa pine regeneration (fig. 2.1.5) resulted from these heterogeneous effects of wildfire (Moore et al. 1999; Pearson 1950; Sánchez Meador et al. 2009; Youngblood et al. 2004). Regeneration processes were not the only process or function influenced by spatial patterns; others include tree mortality, snow accumulation and melt, wind patterns, and fire behavior (Larson et al. 2012). Even in even-aged groups, differentiation might occur due to phenotypic variation (Taylor 2010).

Influence of Large Ponderosa Pine on Forest Structure

In the dry forest ecosystems of the Pacific Northwest, large ponderosa pines are the principal drivers of forest structure and function (Franklin and Johnson 2012). Even putatively mixed-conifer forests were dominated by ponderosa pine at low densities. The total large tree (greater than 21 inches or 53 cm) basal area (of all species) has declined by more than 50 percent and the proportion of these large trees compared to the total number of trees is only 20 percent of the historical numbers (Hagmann et al. 2013), and are at much greater risk of catastrophic fire now than before. Fire suppression has dramatically increased lodgepole pine density in Central Oregon since the 19th century. Besides the obvious implications for the susceptibility to fire-induced and density-dependent mortality, another impact on stand dynamics is being noted, as ponderosa and lodgepole pines in these dense stands are taking longer to grow to a given height (Hagmann et al. 2013; Shuffield 2010).

Fire History and Its Impact on Current and Future Structure and Composition

Presettlement fire frequency throughout the ponderosa pine range was less than 40 years (Agee 1993). In the southern edge of its range, presettlement fires were a mixture of low- and high-intensity fires (Fulé et al. 1997). Historical frequency of low-intensity fire ranged from 4 to 11 years in the Eastside Cascades (Youngblood et al. 2004 and citations therein), a median interval of 7 years (mean of 11 years) on dry ponderosa pine sites in California (Van de Water and Safford 2011), 5 to 29 years on the west side of the Northern Sierras and 3 to 8 years on east side (Raumann and Cablk 2008; Taylor 2004). Fire frequency of around 12 years in old-growth ponderosa pine forests in the Ishii Wilderness in Northeastern California was comparable to ponderosa pine forests on the east side of the Southern Cascades and to mixed-conifer (but predominantly ponderosa pine) stands in the Klamath and Sierra mountain ranges (Taylor 2010). Although declines in fire frequency are likely due to the cessation of Native American burning, widespread grazing on public lands that reduced fine fuels, and reduced ignitions after the California Gold Rush, the main driver of reduced fire effects across the landscape was the State and Federal mandates to suppress all fires (Taylor 2010). Subsequently, ponderosa pine sites on the South Lake Tahoe Basin indicated no evidence of fires for 62 years (Raumann and Cablk 2008) and in Northern Arizona the post-exclusion interval was 120 years, much longer than the historical fire-return interval of 2 to 8 years (up to 15 years for larger fires) (Moore et al. 1999).

Historically, lightning was the main source of ignitions in ponderosa pine forests and continues to play a significant role (Fitzgerald 2005; Hagmann et al. 2013), whereas the role of Native Americans in burning was probably low in these regions (Youngblood et al. 2004). A belt from Northern California through Oregon and Idaho and into Northern Montana contains high numbers of lightning strikes and the resulting fires (Schmidt et al. 2002). Although it is difficult to determine the extent of fires in presettlement times, the prevailing opinion is that burn areas were small. In Northern California, for example, the average area of burn was 850 acres (350 ha), although 16 fires between 1627 and 1992 were believed to be larger than 1,200 acres (500 ha) (Taylor and Skinner 1998). Another analysis determined that some fires ranged up to thousands of hectares in size in Northeastern California (Norman and Taylor 2003).

Fire frequency in ponderosa pine forests throughout the West depended upon elevation and their associated vegetation. At drier, lower elevations, frequent surface fires occurred in these forests. At higher elevations on moist sites, where ponderosa pine was frequently associated with species such as Douglas-fir, white fir, or lodgepole pine, the fire-return interval was longer and the fire intensity ranged from surface fires to stand-replacing fires (Fitzgerald 2005).

Seasonality influences fire behavior and fire effects. In Northern California, 93 percent of all fires occurred between the dry midsummer and early fall (Taylor and Skinner 2003). In contrast, 40 percent of presettlement fires in southwestern ponderosa pine in Arizona occurred in the spring and the remainder were summer fires (Fitzgerald 2005; Fulé et al. 1997).

Mean presettlement fire-return intervals in ponderosa pine forests ranged from 2 to 50 years (table 2.1.1). The Northern California study by Taylor and Skinner (1998) also observed that the postsettlement fire-return intervals (22 years) were longer than during the settlement (12 years) or presettlement (14 years) eras. Furthermore, they found that the interval was shorter on south- and west-facing slopes, a pattern also observed by some (Heyerdahl et al. 2001) and not by others (Everett et al. 2000).

The ability to survive a fire may be either species- or individual tree-based. Species-based attributes include fire-induced flowering, seed dispersal from serotinous cones, such as those on lodgepole pine, or persistent seed buried in the soil, such as those from chokecherry (*Prunus virginiana*). Individual tree-based attributes include thick bark, such as that on ponderosa pine and California black oak, sprouting from the stump, such as shortleaf pine (*Pinus echinata*) and various oak (*Quercus*) species, or

Table 2.1.1—Presettlement fire-return intervals in ponderosa pine ecosystems.

Area	Interval (years)	Source
West wide	2–47	Fitzgerald (2005)
Northern California	21	Norman and Taylor (2003)
Northern California, Lassen National Forest	11	Norman and Taylor (2005)
Central Oregon	30–50	Volland (1963)
Eastern Washington	7	Everett et al. (2000)
Northern Arizona	2–15 (median of 4)	Fulé et al. (1997)
Southern Cascades	12	Taylor (2010)

Figure 2.1.6—To reduce fuels on the Modoc National Forest, woody residues after harvesting are piled, dried, and frequently burned during late fall or winter, when surrounding snow cover reduces the chance of the fire escaping (photo by Ken Sandusky, Forest Service).

having grass-stage needles that protect the apical bud, such as longleaf pine (*Pinus palustris*). These attributes are necessary for trees to persist in a fire-mediated environment (Grace and Platt 1995; Kauffman 1990). Miller (2000) concluded that ponderosa pine is one of the most fire-resistant species in the Western United States and this resistance was believed to increase as the tree matured. Multiple characteristics contribute to improving the chances for the individual pine trees to survive a fire: thick bark that sloughs off when on fire to minimize heat transfer; a deep rooting habit; and an open crown structure that allows for dissipation of heat and reduces the potential for crown scorch (Brown and Wu 2005; Cooper 1961; Fitzgerald 2005; Moore et al. 1999).

Fire, Density, and Tree Size

In late-seral interior ponderosa pine stands, large 200- to 500-year-old trees are an important constituent for their contribution to forest structure and habitat (Ritchie et al. 2008). Despite their prominence in the stand, however, dense understory vegetation can negatively influence their growth and survival. Ritchie et al. (2008) evaluated a study (Oliver 2000) on Blacks Mountain Experimental Forest that compared an unburned, unharvested, 65-year-old stand with neighboring treated stands. On the experimental forest in the 1990s, plots first measured in 1934 were treated to create low-diversity (removing all large overstory trees) and high-diversity (thinning from below) structures that were then subjected to burn and no-burn treatments (Ritchie et al. 2008). Untreated stands displayed a dramatic increase in density of trees less than 12 inches (30 cm) diameter between 1934 and 1999. The proportion of the total basal area in pine declined by 12 to 20 percent in that time, mainly replaced by white fir and incense cedar. Thinned and burned stands (similar to those in fig. 2.1.6) adjacent to the untreated stand had fewer large trees at risk and lower rates of mortality (Ritchie et al. 2008). Such treatments are, however, not without risk. Careful thinning can accelerate growth of old pine trees (Kolb et al. 2007), but prescribed burning on a site with a long history of fire exclusion may result in secondary mortality of the older trees (Crotteau and Ritchie 2014; Ritchie et al. 2008). Delayed effects upon the entire forest ecosystem can occur, too. Not only does fire remove snags, but snags from fire-killed trees deteriorate faster than trees killed by other causes, such as insects (Laudenslayer 2002).

Land Use Practices and Their Impact on Fire

Livestock grazing and timber harvesting were widely practiced in ponderosa pine forests by the end of the 19th century (Graham and Jain 2005; Madany and West 1983; Rummell 1951). In 1880 in Northeastern California, 45,000 grazing animals were reported; this number increased threefold by 1909. This intensive grazing pattern reduced fine fuels and native grasses and palatable shrubs; consequently, annual grasses and unpalatable trees and shrubs (e.g., sagebrush and western juniper) became established (Laudenslayer et al. 1989; Riegel et al. 2006). In comparison, grazing in the Southwestern United States reduced grass competition on ponderosa pine sites and the cessation of grazing, coupled with favorable seed crops and climate, resulted in dense stands of ponderosa pine regeneration (Moore et al. 1999; Pearson 1950; Savage and Swetnam 1990). Postsettlement, heavy grazing and fire suppression changed the fire regime in ponderosa pine forests, allowing regeneration to proceed without barrier and fuels to accumulate to much-greater-than-historical levels. Furthermore, the high-grading of large ponderosa pine, Douglas-fir, and western larch (*Larix occidentalis*) throughout the range of the pine resulted in greater densities of thin-barked, shade-tolerant species in the understory, creating ladder fuels and forest stands more prone to damage from insects and drought (Fitzgerald 2005).

Fire Suppression and Forest Types

Fire suppression resulted in more pronounced effects in forests subject to frequent low-intensity fires, such as pine forests, than in more moist forests. For example, in a study of the South Tahoe Lake Basin, Raumann and Cablk (2008) found stands subject to more than 6 decades of fire suppression to be much denser than those under natural fire regimes. Although forest stands up to 9,800 feet (3,000 m) displayed increased density, about three-fourths of the densification occurred below 7,500 feet (2,300 m), which was the upper limit of Jeffrey pine-white fir forests (Raumann and Cablk 2008). The authors found an increase in forest extent from 1940 through 1969 due to regeneration after harvest. After 1969, they determined that most forest expansion in undisturbed areas was succession.

Climate, Elevation, and Forest Types

Relationships exist among biophysical site characteristics, vegetation associations, and resulting disturbance frequencies and responses. Following the Life Zone Concept (Merriam 1898), which defined ranges of vegetation types in the Southwestern United States, Stephens et al. (2015) found that the most important variables for defining stands in the Southern Sierras were actual evapotranspiration, elevation, and aspect. Each ecosystem has a unique response to natural and human-caused disturbance. In a study of a mixed-conifer forest in the Lake Tahoe Basin, Van Gunst et al. (2016) determined that density-dependent mortality occurred more on lower-elevation forests with drier climates than on forests at higher elevations. Further, they found that density-dependent mortality declined as density increased in these mid- to upper-elevation forests. Separating density from drought, they surmised that lower-elevation forests may experience less drought-related mortality as the pine-dominants (Jeffrey and sugar pine [*Pinus lambertiana*]) are more drought-tolerant than other species that may occur on the site. They attribute this drought resistance to greater stomatal control and resistance to cavitation, as was observed in ponderosa pine by Maherali and DeLucia (2000). They did find greater mortality on north-facing slopes, regardless of density, suggesting that these communities did not typically face droughty situations and were thus more susceptible to drought events when they did occur.

Changes in Ponderosa Pine Forests

Grazing, Fire Suppression, and Climate Change

Where fire, grazing, and timber harvesting have stopped on moist sites (where the potential vegetation is white fir), ponderosa pine is being succeeded by Douglas-fir and white fir (Graham and Jain 2005) (see fig. 2.1.7). In some places this succession is occurring so rapidly that the normal progression from ponderosa pine to Douglas-fir of 400 years is being reduced to 50 years because of the lack of non-lethal disturbance (Graham and Jain 2005). Several factors influenced the change in pine regeneration, the most notable being the change in grazing practices (from sheep to cattle), suppression of natural, heretofore low-intensity fires by the Forest Service in 1905, and climate (Norman and Taylor 2005), although the climate influences are not as definitive as one might expect, as Norman and Taylor (2005) referred to evidence that cooler, moister springs and drier summers aided successful regeneration

Figure 2.1.7—Young ponderosa pine in the Pine Creek Valley of the Lassen National Forest (photo by Ken Sandusky, Forest Service). Fire, grazing, and available soil moisture are some of the influences that minimize pine regeneration in such open parks.

of Jeffrey and ponderosa pines. (See Chapter 6.1, Wright, this synthesis, *Ecological Disturbance in the Context of a Changing Climate: Implications for Land Management in Northeastern California* for more discussion about climate-induced changes to fire.) Paradoxically, the change in disturbance patterns created an environment where the more infrequent but more severe disturbances also negatively impacted forest regeneration. Severe wildfires occurring after decades of fire suppression can result in the lack of pine trees as a seed source and what little regeneration that does occur is mainly shade-tolerant, fire-intolerant species (Welch et al. 2016).

Youngblood et al. (2004) attribute the decline of Eastside Cascades old-growth ponderosa pine to fire suppression, livestock grazing, selective logging of old trees, and road building. Managers have responded by trying to restore disturbance frequency and stability by thinning, underburning, and other fuels-reduction treatments (Fiedler et al. 1996; Youngblood et al. 2004).

Juniper Woodlands

Piñon-Juniper Ecosystems

In the Western United States, piñon-juniper ecosystems cover 100 million acres (40 million ha). Several pine and juniper species grow together, or separately, across the piñon-juniper complex. In the Great Basin, the principal species are *Pinus monophylla/Juniperus osteosperma* (singleleaf piñon/Utah juniper) but continuing westward into Northwestern Nevada and neighboring California, the principal species is western juniper (Romme et al. 2009) with low abundance of piñon. Juniper woodland has greatly increased in both spatial distribution and density (Miller et al. 2008); however, historical evidence indicates that piñon-juniper increased and decreased in extent over the previous millennia (Miller and Wigand 1994).

Western juniper is the only representative of the Great Basin piñon-juniper complex in Northeastern California. It is typically the only conifer on the site, except along an ecotone with ponderosa pine. Western juniper began migrating northward from the Lake Lahontan basin 10,000 years before present (BP) (Tausch 1999) and reached its present-day range in Northeastern California and Southeastern Oregon between 7,000 and 4,000 years BP (Mehringer and Wigand 1987, as cited in Riegel et al. 2006a). Before Euro-American settlement in the mid-19th century, juniper occurred mainly on rocky surfaces or sandy sites with little vegetation (Miller and Rose 1995). In this region, old juniper stands are primarily found on shallow soils with heavy clays, and are generally open (Riegel et al. 2006). On sites with low fuels, western juniper can reach 1,000 years of age (Waichler et al. 2001). Sites near Devil's Garden in the Modoc contain old-growth juniper (200 to 500 years old) with some individuals as old as 700 years. The wide spacing of the trees implies that there were few stand-replacing fires. Disturbance limited western juniper to low-productivity sites before Euro-American settlement. But the species can grow in almost any type of soil. Since settlement, western juniper has established on more productive sites (Adams 1975; Burkhardt and Tisdale 1969, 1976; Miller and Rose 1995, 1999). This expansion is a result of an increase in fire-return interval, grazing, and higher-than-average annual precipitation as paleoecological evidence suggests that the presettlement fire-return interval kept the species off these sites.

In California in 1989, the total area of western juniper and piñon-juniper woodlands stands with at least 10 percent cover in tree crowns was estimated at 2.4 million acres (1 million ha) (Bolsinger 1989), with 1.1 million acres (0.4 million ha) being on national forests. This total did not include woodlands with less than 10 percent cover or located on other public land or private land, or scattered trees. Before the arrival of Euro-Americans, western juniper was primarily restricted to areas with poor soils, which reduced the amount of fine fuels able to carry fires (Burkhardt and Tisdale 1969; Miller and Rose 1995). Favored by Euro-American settlers for grazing of cows and sheep due to the proximity to inhabited areas, the land cleared of trees to improve forage has been offset over the years by the afforestation of rangeland due to fire suppression; thus, the overall quantity of land in these woodland types has changed little since 1945 (Bolsinger 1989). Recent trends suggest that afforestation is decreasing, as reflected in the increasingly large proportion of western juniper and piñon-juniper stands greater than 100 years old (73 percent and 59 percent, respectively). Bolsinger (1989) observed that stands less than 50 years old make up only 12 percent of western juniper and 9 percent of piñon-juniper stands. Singleleaf piñon and western juniper together account for 93 percent of total wood volume in such woodlands. Grass cover is higher in western juniper stands than in piñon-juniper stands (Bolsinger 1989).

While in Northeastern California the western juniper woodlands (fig. 2.1.8) are a continuation of the extensive piñon-juniper woodlands of the Great Basin (Young and Evans 1981), piñon becomes scarce. Of the 60,000 acres (24,280 ha) of juniper habitat on the Lassen, less than 1 percent is piñon-juniper (Bolsinger 1989) and the author found no piñon-juniper on the 470,000 acres of (190,200 ha) juniper habitat on the Modoc. Juniper woodlands are largely found in Klamath Juniper Woodland/Devil's Garden (Level IV Ecoregion 9j), Modoc/Lassen Juniper Scrub (9p), Adin/Dixie Low Hills (9r), and Modoc Lava Flows and Buttes (9s) ecotypes (see Appendix 1.1 in Chapter 1.1, Dumroese, this synthesis, *The Northeastern California Plateaus Bioregion Science Synthesis: Background, Rationale, and Scope* for ecoregion descriptions). On the Lassen, western juniper occurs in big and low sagebrush communities (table 2.1.2).

Presettlement Conditions and How They Might Guide Current Management

Except for western juniper, common species of juniper are all associated with piñon (Miller and Wigand 1994). Not surprisingly given their wide distribution across the Western United States, most research has focused on piñon-juniper systems. Across these different juniper-dominated ecosystems, some patterns concerning juniper emerge:

- Mammals and birds spread seed more than 100 meters.
- Climate and disturbances affect the rate of infill between early-established trees.
- Understory floral diversity progressively decreases as juniper overstory increases.

When available, this chapter draws on research from pure juniper woodlands but also notes work within piñon-juniper woodlands because the similar stand dynamics, disturbance patterns, and climate responses are relevant to management questions on the Lassen and Modoc. Furthermore, some climate projections propose that the future weather will be hotter and drier (Clark et al. 2016), suggesting that future Northeastern California woodlands might behave in a manner more like those of the Central Great Basin and the constituent juniper species.

Figure 2.1.8—Western juniper, like most *Juniperus* species, can form closed-canopy, woodland, and savanna landscapes, depending upon site productivity, precipitation and disturbance (fire) interval. The Warner Mountains viewed from the Devil's Garden, Modoc National Forest (photo by John Cichoski, Forest Service).

Table 2.1.2—Summary of western juniper/sagebrush vegetation communities in Lassen County, CA (Young and Evans 1981).

	Western juniper/big sagebrush	Western juniper/low sagebrush
Soils	Deep, 10–32 inches (25–80 cm), heavy clay loam	Biscuit and swale, mounds 4–8 inches (10–20 cm), wet in spring and baked hard in summer
Trees per acre (trees per hectare; tph)	60 (150 tph) 10 (25 tph) after burns 40 (98 tph) since 1900	11 (28 tph) 2 (5 tph) since 1900
Canopy cover	40 to 60 percent	–
Age (year established)	84 percent established between 1880 and 1920; oldest 1855	Oldest 1600

Western Juniper

In the late Pleistocene, the more drought-tolerant juniper species were 310 to 400 miles (500 to 640 km) further north and 3,280 to 4,920 feet (1,000 to 1,500 m) lower in elevation than they are now. Western juniper (probably var. *australis*) grew in Kings Canyon, CA, at the glacial maximum. It was in the area of Lake Lahontan, NV, about 12,000 years BP and arrived in Northeastern California and Southeastern Oregon 5,000 to 8,000 years later (Miller and Wigand 1994).

The range of western juniper has greatly increased since the 1880s (Eddleman 1986). In Northeastern California, the peak years of establishment range from the late 1800s to the early 1900s (Young and Evans 1981), and in Southern Oregon, the early 1900s (Adams 1975). In the sagebrush steppe, presettlement juniper was primarily in shrub form on shallow, rocky soils, where there was little fine fuel. Most of the juniper woodlands on the landscape today became established as annual precipitation increased and fire frequency decreased after the late 1800s (Riegel et al. 2006). Little older western juniper was left on the land because mortality of juniper trees due to fire is much higher for trees less than 50 years old than for the older cohort (Riegel et al. 2006). Because of management and altered disturbance histories on the sites, western juniper stands can become quite dense. In Central Oregon, Eddleman (1986) found juniper densities averaging 412 trees per acre (1,018 tph), with a range of 335 to 450 trees per acre (830 to 1,120 tph), although he stated that his results were higher than other contemporaneous studies. Even 50 years ago, Burkhardt and Tisdale (1969) reported that some seral stands of western juniper had densities of more than 800 trees per acre (2,000 tph).

Western juniper is less dependent on precipitation than juniper communities to the west and south, although prolonged wet or dry spells can influence tree cover and growth. For example, relatively high winter precipitation in Southeastern Oregon in the late 1800s and early 1900s resulted in extensive recruitment of western juniper (Romme et al. 2009). Paradoxically, the western juniper that benefited from increased precipitation does not, in turn, improve water retention or soil moisture potential on the land. Tree canopy can intercept precipitation. In California, western juniper woodland cover of 40 percent intercepted 15 to 20 percent of precipitation (Evans 1988). Miller and Wigand (1994) found that heavy juniper cover results in reduced ground flora cover, which in turn results in damaged hydrologic process due to less infiltration and more runoff and erosion.

Western juniper primarily invades big sagebrush communities and to some extent, low sagebrush communities (Burkhardt and Tisdale 1969, 1976; Young and Evans 1981). In Central Oregon, Eddleman (1986) determined that woody plants, whether sagebrush or juniper, act as nurse plants for western juniper seedlings, similar to what Phillips (1909) observed for piñon pine (*Pinus edulis*), very likely due to reduced soil surface temperature and increased shading, and perhaps higher availability of moisture and nutrients.

In a study in Southeastern Oregon, Adams (1975) documented the interactions between western juniper and associated shrub species. Western juniper increased from less than 59 percent of crown area in 1929 to 73 percent in 1966. Shrub crown cover declined 36 percent during the same period even though total plant density declined by only 2 percent. The author attributed the disparity to the replacement of large-crowned bitterbrush (*Purshia tridentata*) and big sagebrush (86 percent of shrub density in 1920s, only 33 percent in the late 1960s) by narrow-crowned green rabbitbrush (*Ericameria teretifolia*). In Northeastern California, Loft (1998) estimated that in a

40-year period (1957–1997) the overstory of pine and juniper increased by more than 400 percent on bitterbrush range. Adams (1975) suggested that a certain synchronicity existed between juniper and the associated vegetation, with few shrub plants being established in years where juniper had low growth ring indices.

Young and Evans (1981) intended to examine the cause of juniper establishment (i.e., wildfire suppression, grazing, or climate change), but no ungrazed areas were available to test a grazing hypothesis, the relatively short period of time of invasion and establishment, and the dense stand's individual tree growth rate would make it difficult to explore climate influences. They pointed out that trees less than 50 years old were very susceptible to fire damage, suggesting that perhaps fire suppression played a role. Both locations were "fireproof"—the western juniper/big sagebrush site because it was so dense that no ground vegetation grew under the canopy (Bruner and Klebenow 1979) and the western juniper/low sagebrush site because of heavy grazing of the native grasses (Young and Evans 1981). They observed that only about 10 percent of the trees in the western juniper/big sagebrush stand bore fruit, with the result that the seed source in western juniper/big sagebrush stand usually came from trees in the western juniper/low sagebrush stand.

Juniper Overstory vs. Forage Understory

There is an inverse relationship between overstory cover and understory plant cover (Tausch and Tueller 1990). The strong interaction between overstory tree canopy density and understory forage productivity is often exacerbated by human activities. Drier sites on shallow soils on south-facing slopes show greatest western juniper impact on plant community. Miller and Wigand (1994) noted that desirable browse in some bitterbrush ranges was shaded out by a 400 percent increase in overstory pines and junipers. Fire suppression and grazing created the conditions that allowed juniper woodlands to move from their low-productivity sites onto rangeland (Burkhardt and Tisdale 1969; Miller and Rose 1995; Miller and Wigand 1994).

Clearing sites of western juniper to restore or create rangeland for grazing is a common practice in the Interior West (fig. 2.1.9). To test a site's ability to respond to efforts to improve ecosystem resilience, in 1991 sites in a western juniper woodland in Southeastern Oregon were harvested to evaluate the understory response (Bates et al. 2000). The site was at full occupancy and most of the co-occurring sagebrush shrubs were dead. Juniper canopy cover was 24 percent of the area and most of the remainder was bare ground, subject to erosion. Dry conditions region-wide limited treatment response the following year. In 1993, however, an almost order-of-magnitude difference in understory biomass was found on the cut plots compared to the wooded plots. Canopy cover of herbaceous species increased by a factor of 3 in the interspaces. Besides the obvious competition for water through their extensive root systems, Bates et al. (2000) also concluded that juniper interfered with nitrogen uptake of other plants as an additional competitive advantage. The authors noted that the postdisturbance ground vegetation community was connected with pre-disturbance composition (fig. 2.1.10). The expansion of herbaceous cover in the open areas was also a significant benefit in reducing erosion.

Juniper Woodlands in the West

Throughout juniper's range, three types of juniper woodlands exist: persistent woodland, wooded shrubland, and savanna (Romme et al. 2009; Vankat 2013). In persistent woodland, soils, climate, and disturbance patterns favor juniper; the fire regime consists of infrequent but high-severity crown fires. Persistent juniper woodlands were minimally affected by fire-suppression efforts as 20th-century fire management practices did not usually target such ecosystems (Vankat 2013). The dynamics of persistent woodlands are driven more by climate, disease, and insects than by fire. Less is known about wooded shrubland and savanna (Romme et al. 2009).

In juniper savannas, soils favor juniper, but fire is more frequent. In wooded shrubland, which have a shrub layer beneath an open tree canopy (Vankat 2013), soils support woody plants. But species vary between deep soils and thin rocky sites, particularly where winter precipitation constitutes a higher proportion of the total annual precipitation. Periodic wildfire may result in oscillating woodland conditions (West and Van Pelt 1986). Figure 2.1.11 displays how periodic wildfires might cause the ecosystem to alternate between woodlands and savannas.

In moist periods, the number of trees will increase, whereas in dry periods, the number of trees will decrease (Romme et al. 2009; Tausch et al. 1981). Savannas have grass and forb understory under an open tree canopy, generally on sites with moderately deep soils. In the Southwestern United States, juniper is found at low elevations adjacent to grassland ecosystems, but can also be found at higher elevations (Vankat 2013).

Figure 2.1.9—The varied nature of structure in western juniper stands provides varied habitat for native and introduced species, including these wild horses up Pencil Road Trail on the Modoc National Forest (photo by Ken Sandusky, Forest Service).

Patterns of Expansion

Cycles of high and low precipitation cause corresponding responses of piñon-juniper establishment and mortality (Chambers et al. 2013; Swetnam and Betancourt 1998). Juniper and associated woodlands expands across the landscape in one of three ways: (1) infill—in existing woodlands, the tree density within the stand expands by growth of the existing trees and by establishment of new trees; (2) expansion—trees become established in formerly treeless lands, such as sagebrush or other shrubland; and (3) succession—the successional pathway following stand-replacing fire progresses from annual herbaceous species to forbs and perennial grasses and then, after several decades, to shrub cover (fig. 2.1.11). Tree seedlings then follow, and woodland forms in 100 to 300 years (Vankat 2013). Huffman et al. (2012) also found a relationship between juniper presence and time since fire. Seedling survival in woodlands requires an enhancement of the site to reduce the environmental stresses and successfully establish. This addition may take the form of physical cover or nurse plants, even other juniper. Juniper species will establish in the interspaces, and then will expand if drought kills the competing components (Vankat 2013).

Figure 2.1.10—The influence of juniper on forage and the potential for restoration depends upon the density and length of tenure of juniper, such as this open-grown western juniper on the Lassen National Forest, and the mix of ground flora that existed prior to afforestation (photo by KC Pasero, Forest Service).

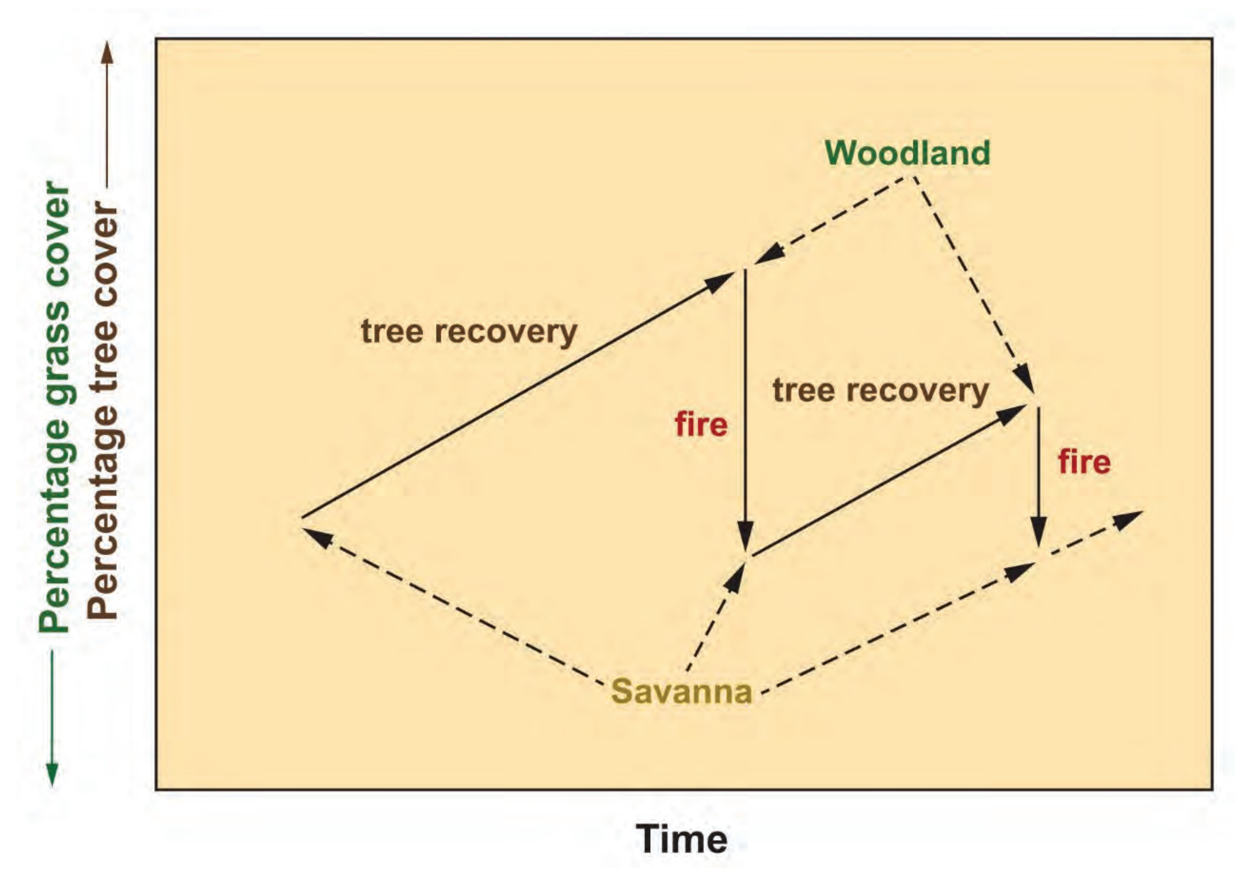

Figure 2.1.11—One model of periodic wildfires causing intercommunity cycles between juniper woodland and savanna (adapted from West and Van Pelt 1986).

The number of trees increased rapidly in the Great Basin during the 19th century (Tausch et al. 1981). A study of various juniper ecosystems in Oregon, Idaho, Nevada, and Utah (which included western and Utah juniper) found that prior to settlement, woodlands had slow rates of establishment and low densities. Piñon and juniper trees established at a higher rate in the late 1800s than they had in the several centuries before, beginning after 1860 in Utah and Nevada and after 1850 in Idaho. These species increased gradually later in the 19th century, then more rapidly in the early 20th century. The rate of establishment then declined later in the 20th century (Miller et al. 2008).

The historical increase in tree density occurred at roughly the same time across the Great Basin. Scientists have suggested several possible causes, including warmer temperatures from 1850 through 1960, wetter weather between 1850 and 1900, fire suppression, and increased cattle grazing and timber harvesting for mining (Chambers et al. 2013; Tausch and Hood 2007). Climatic conditions were milder and wetter in the latter part of the 19th century and the first half of the 20th century than they were in the latter half of the 20th century (Antevs and Wright 1938; Graumlich 1987; Miller et al. 2008). Apparently, this increase in tree number was successful, as the authors go on to suggest that the decline in recent establishment may be due to the lack of suitable sites unoccupied by trees.

Tree density in juniper woodlands of the Great Basin has also increased during the last 150 years (Romme et al. 2009), with the current density 4 to 10 times greater than the presettlement conditions (Tausch and Hood 2007). Infill has occurred locally in persistent woodlands, but infill in this type was not so widespread across the Great Basin region. Infill has occurred much more in wooded shrubland and savannas, where elimination of fine fuels due to timbering and grazing resulted in changes in the fire regime (Romme et al. 2009; Tausch et al. 1981). Grazing increased tree density by reducing fine fuels and competing vegetation in mixed-conifer forests, but had less of an effect on the density of piñon-juniper and juniper woodlands. Forty percent of the trees in the Great Basin were established less than 150 years ago (Tausch et al. 1981).

Grazing Impact in Piñon-Juniper Woodlands

Grazing had little influence on tree cover in persistent woodlands. In wooded shrubland and savanna, grazing resulted in increased tree cover and different ground vegetation species as previously dominant herbaceous species were eliminated, for example with western juniper in Oregon (Miller and Rose 1995). Subsequently, considerable attempts have been made to improve grazing on wooded shrublands and savannas by using mechanical treatments such as chaining or cutting (Vankat 2013). Elimination of shrub and grass species was not inevitable, however. Cottam and Stewart (1940) found that sites in Southern Utah that had deep soils, grasses, and sagebrush could withstand Utah juniper invasion, but on poorer sites in the foothills, junipers replaced these species. The piñon-juniper communities are very susceptible to severe wildfires. Old-growth woodlands on presettlement landscapes were usually located on sites with limited fuel loads, often with thin soils or steep slopes, or both, where disturbance was not frequent (Tausch and Hood 2007; Tausch et al. 1981; West et al. 1998). In contrast, very productive sites on deeper soils generate greater fine-fuel loads and thus experience more frequent fires. Old-growth woodlands could also be protected from severe fire by the frequently burned sagebrush communities that surrounded them.

Drought

Juniper is more resistant to drought and is affected by fewer biotic agents than piñon (Vankat 2013). Juniper exhibits low natural mortality in the Great Basin (Adams 1975). According to Romme et al. (2009), while historical fires did not thin piñon-juniper woodlands from below, stand-replacing fire was the main source of juniper mortality. Recovery from such severe fires could take hundreds of years (Romme et al. 2009), creating a multi-century fire-return interval.

Spread of Piñon-Juniper Into Sagebrush and Grasslands and Fire Effects

Disturbance characteristics, especially the frequency and intensity of fire, influence spatial arrangement of woodlands in a landscape of sagebrush and the distribution of tree sizes within these woodlands (Tausch and Hood 2007). Throughout the Great Basin, juniper woodlands are expanding into adjacent sagebrush ecosystems and increasing in density (Bradley 2010; Burkhardt and Tisdale 1976; Miller and Rose 1999; Tausch et al. 1981). For example, one study in Central Nevada determined that 20 percent of the ecotonal area between sagebrush ecosystems and piñon-juniper woodlands showed juniper woodland expansion during a 20-year (1986–2005) period (Bradley and Fleishman 2008b). Although sagebrush shrublands are adapted to many different temperature conditions (Bradley 2010), piñon-juniper woodlands can compete successfully for water at greater depths (Leffler and Caldwell 2005 as cited in Bradley 2010). The expansion of piñon-juniper woodlands is largely the result of cattle and sheep grazing that resulted in the removal of most ground flora that might compete with piñon-juniper (Bradley and Fleishman 2008a; Miller and Rose 1995; but see Vankat 2013), concurrent suppression of wildfires (Bradley and Fleishman 2008a), and late 19th century/early 20th century wet conditions that greatly facilitated establishment and growth of trees (Antevs and Wright 1938; Tausch and Hood 2007).

The pattern of fuel loading and fire behavior changed from fine-fuel dominated open woodlands to one where the tree species crowded out the grasses. Fuel continuity across a landscape increases as the tree biomass increases, thus increasing the possibility of severe, stand-replacing fires. Postfire composition and structure depend upon fire severity, the prefire composition, and the amount of tree dominance, but severe crown fires increase the probability that post-burn sites will be devoid of woody vegetation (Miller and Tausch 2001).

Fires were less likely to reset the ecological succession of any preexisting ground flora on the site, however (Tausch and West 1988). In a comparison of burned and unburned piñon–juniper woodlands in Nevada and California, Koniak (1985) concluded that species that became dominant in mid-successional and later-successional stages were present at early stages; tree species that were eliminated by burning were the only exception. Early-successional species, which bear seeds that survive fire or have buds at root crowns, take advantage of increased availability of water, light, and nutrients (Tausch and West 1988). Annual and perennial forbs tend to dominate early-successional sites, shrubs and annual grasses the mid-successional sites, and shrubs, annual grasses, and trees the late-successional sites (Koniak 1985). Root-sprouting shrubs and forbs were found more on northerly and easterly aspects, annual forbs on southerly and westerly aspects, and seed-germinated shrubs on northerly, easterly, and westerly aspects (Tausch and West 1988). As the tree species begin to dominate the site, fewer resources are available to understory species and their percentage cover declines (Koniak 1985; Tausch and Hood 2007).

Approaches to Forest Management

Ecological restoration focuses on reestablishing "the structure, function and integrity of indigenous ecosystems" qualified by "to the extent possible" (SER 1993) (fig. 2.1.12). Higgs (1997) refers to ecological fidelity as an endpoint, where restored plant structure and composition provide functional success and durability. He also refers to "mutually beneficial human-wildland interactions," suggesting that true sustainability accepts that human beings are part of the landscape. Given that the ability to adapt to disturbance is an indicator of a healthy ecosystem, successful ecological restoration should seek to restore resilience and adaptive capacity (North et al. 2009b).

Overview

Ecological Forest Management

Silviculture is the art and science of manipulating forest composition and structure to achieve goals (Helms 1998). Silviculture is not an end, but rather a tool or series of tools intended to meet varied objectives, including timber production, wildlife habitat, or even "naturalness" (O'Hara 2016). O'Hara identifies current forest management trends that advertise being "natural" by purporting to mimic natural processes or minimize disturbance effects, often incorporating labels such as "nature" or "ecological" to broadcast their bona fides. Yet, forestry has historically focused on managing stands to attain objectives that could not otherwise be achieved by natural processes (O'Hara 2002, 2016). Proponents of "nature-emulating silviculture" fail to recognize that the cumulative effect of changing climate, historical management and mismanagement, and invasive plants, insects, and diseases are creating novel ecosystems that require novel management practices (O'Hara 2016). For example, in their study of historic (1911) and current conditions of a series of forests in Southcentral Oregon, Hagmann et al. (2013) observed that current and projected climates are hotter and drier than the period when the inventoried trees established and grew. While ecosystem responses to past disturbances and management activities can inform managers (Moore et al. 1999; Youngblood et al. 2004), managing to recreate an environment that will never occur again is likely to

Figure 2.1.12—Restoring forest and woodland ecosystems, such as these on the top of the Warner Mountains in the Modoc National Forest, requires an understanding of "the structure, function, and integrity of indigenous ecosystems" (SER 1993), but also some expectation of potential future disturbances, both long- and short-term (photo by Chris Bielecki, Forest Service).

be unsuccessful (Millar et al. 2007) and such historical responses would not necessarily predict what happens after future management actions (O'Hara 2016).

Restoration

Planning for future forests is challenging for today's managers as the future combination of environment, structure, climate conditions, and other influences will produce novel environments (Clark et al. 2016). The stands today are often no longer at their optimal condition due to years of fire suppression and inconsistent management goals and activities (Graham and Jain 2005). Nonetheless, we can obtain guidance from current and past conditions (Heyerdahl et al. 2001) and management practices that achieve composition and structure goals now can serve future managers with similar goals (Helms and Tappeiner 1996; O'Hara 2016).

North et al. (2009b) cite a definition of ecological restoration from the Forest Service Manual (2020.5): "The process of assisting the recovery of resilience and adaptive capacity of ecosystems that have been degraded, damaged, or destroyed. Restoration focuses on establishing the composition, structure, pattern, and ecological processes necessary to make terrestrial and aquatic ecosystems sustainable, resilient, and healthy under current and future conditions." Ecological restoration has been described as the process of repairing damage caused by humans to the diversity and dynamics of indigenous ecosystems (Jackson et al. 1995). Practitioners of restoration have traditionally viewed their goal as the recreation of some past ecological state (Hobbs and Cramer 2008), although differences exist in defining success as some combination of restoring structure, function, diversity, and socioeconomic impacts (Wortley et al. 2013). Sometimes realism demands that managers repair the most egregious damage ("rehabilitation"), but not necessarily return to some historic state, or moving toward an alternative land use ("reallocation") (Aronson et al. 1993; Hobbs and Cramer 2008). A measure of success is the evaluation of an ecosystem's or community's "key ecological attribute," a characteristic of a community's biology that, when present, define the community as "healthy" and if absent or changed, presage the degradation or loss of the community sometime in the future (Gonzales and Hoshi 2015). North et al. (2009b) argue that restoration should focus on enhancing resilience, the ability to respond to disturbances in a way that maintains the ecological function of a particular ecosystem. This focus melds the drive for rehabilitation with the perspectives of those (O'Hara 2016) who contend that future novel situations require novel management.

Human activities dramatically changed these forests. As will be explained later in the specific forest-type sections, today's forests in Northeastern California and throughout the Great Basin are denser, with a higher proportion of basal area in smaller trees and more dominated by shade-tolerant species. Existing dry forests have permanently reduced capacity to withstand stressors without undergoing significant change (Noss et al. 2006). Current efforts emphasize restoring "processes that shape systems rather than any particular structure or composition of the past" (Hagmann et al. 2013; Millar et al. 2007; Stephens et al. 2010).

Many authors advocate the recreation of spatial heterogeneity that was produced by variable fire intensities, which not only enhances ecosystem values but also reduces the susceptibility to severe disturbance (Crotteau and Ritchie 2014; Hagmann et al. 2013; Larson et al. 2012). This heterogeneity existed on small and large scales and restoration efforts are expected to mimic them. Larsen (2012) defined two scales of heterogeneity: global, which encompasses a stand or even a forest, and local, which is at the scale of tree clump, open gaps, and single trees. Those advocating restoration often fall into one of two camps: "structural"—those who emphasize first restoring historical stand structure and composition through mechanical thinning, and "functional"—those who prioritize use of ecological processes, such as fire, for restoration (North et al. 2007; Stephenson 1999). Neither of these two philosophies fits all sites, although in the specific case of restoring fire to long-unburned sites, many researchers have recommended structural restoration (Covington et al. 1997; Fiedler et al. 1996; Knapp et al. 2017; Moore et al. 2004; North et al. 2007), so long as the process element is instituted in a timely fashion.

Resistance and Resilience

Millar et al. (2007) argue that management for persistence of a species necessitates taking up a broad ecoregion-based view and not be wedded to a particular location, composition, or population level. This practice gives managers flexibility to target achievable goals. Resistance and resilience are two options in the adaptive strategy of assisting ecosystems to accommodate changes due to climate and its resulting influences on disturbance patterns.

Understanding the definitions of resistance and resilience is essential for crafting management objectives and the practices to achieve them. With expectations of changing climates impacting future forest vigor, composition, and structure (Clark et al. 2016), maintaining or enhancing resistance and resilience is considered an important goal (Millar et al. 2007). A successful outcome assumes that the forest ecosystems exposed to disturbances will continue to provide ecological goods and services (DeRose and Long 2014; Puettmann 2011).

Desiring to make the forest more resistant to climate change is easier said than done, given the complications of projecting climate outcomes, the potential effects of said outcomes on the vegetation community, and the administrative and legal requirements for any management actions on public lands. Faced with a multitude of moving parts, silviculturists can focus on what they do best, which is manipulating composition and structure (DeRose and Long 2014).

Differentiating resistance and resilience revolves around timing or perspective: resistance is the ability of an ecosystem to withstand or influence the disturbance ("before" or "during") and resilience is the ability to recover from the disturbance ("after"). Just like any management action, managers should define the scale of disturbances—stands or landscapes—and develop metrics for analysis accordingly. Defining such a disturbance includes recognizing that, depending upon the severity, both resistance and resilience are impacted (DeRose and Long 2014) and that management actions can enhance resistance and resilience simultaneously (Puettmann 2011).

Several definitions emphasize this difference in perspective. Resistance "(to fire) is the ability of community to remain unchanged when challenged by disturbances" (DeRose and Long 2014; Grimm and Wissel 1997); "forestall(s) impacts and protect(s) highly valued resources" (Millar et al. 2007); is the "capacity of an ecosystem to retain its fundamental structure, processes, and functioning despite stresses, disturbances, or invasive species" (Chambers et al. 2013); and "[s]tand resistance to disturbance as the influence of structure and composition on the severity of the disturbance" (DeRose and Long 2014).

Resilience "is the ability of a forest to survive a wildfire relatively intact," which was usually the case in ponderosa pine ecosystems prior to settlement (Fitzgerald 2005); is "the capacity of a system to absorb disturbance and reorganize while undergoing change so as to still retain essentially the same function, structure, identity and feedback" (DeRose and Long 2014; Walker et al. 2004); seeks to "improve the capacity of ecosystems to return to desired conditions after disturbances" (Millar et al. 2007); and is "the capacity of an ecosystem to regain its fundamental structure, processes, and functioning when subjected to stress and disturbances" (Chambers et al. 2013). DeRose and Long (2014) define resilience as "the influence of a particular disturbance on structure and composition" and state that in order to devise effective management practices, managers must define measurable attributes that are associated with their desired goals.

Management Activities That Can Promote Restoration

Resistance to fire is enhanced by fuels reduction, lower stand densities, and removing ladder fuels. Other management activities include thinning the stand to increase crown spacing and retaining large trees (Moore et al. 1999). DeRose and Long (2014) referred to strategic placement of area treatments (SPLATS) where 20 percent of the area treated can slow fire by 60 percent.

The multiple uncertainties regarding climate, insect and disease attack, and forest response can be overwhelming. Breaking down the analysis into discrete stages, focusing first on the vegetation influences, followed by attention to disturbance and disturbance influences, and finally on vegetation structure and composition in the context of a specific time horizon may be a prudent approach (DeRose and Long 2014).

Managers are usually expected to deal with immediate and long-term concerns at the same time. While there might be some event that triggers public and management concerns, such as preventing a wildfire, managers often approach the treatment, in this case thinning, with the intent to serve long-term goals, such as increasing a forest stand's ability to respond to potential future drought (DeRose and Long 2014; Lévesque et al. 2014). Silviculture only "buys time" as trees grow and enter a vulnerable stage again. Timely treatments increase resistance but economic and political barriers impede these timely treatments (DeRose and Long 2014).

In their chapter on maintaining and restoring ecosystems in Southern Nevada, Chambers et al. (2013) summarize the range of management actions available to managers seeking to enhance ecosystem resistance and resilience. The authors emphasize a "realistic approach" basing their goals on the current rather than historical potential of ecosystems to support a given set of ecological conditions. They recognized that these vegetation classifications are fluid, as large-scale tree mortality can cause ecotones to shift and entire vegetation communities and regional disturbance patterns to change. While their recommendations are most appropriate for the Southern Nevada region, their characterization of ecosystems that are similar in character to those of the Lassen and the Modoc bears repeating below (table 2.1.3). It should be noted that Chambers et al. (2013) emphasize resistance strategies, which they term "prevention," to reduce disturbances such as fire and invasive species.

Forests with a long history of fire suppression are a prominent landscape feature of the Western United States. In this situation, the disturbance that DeRose and Long (2014) would identify as a management goal is to reduce the impact of catastrophic wildfire and the strategy to adopt would be resistance. Looking to restore more open stand structures consistent with historic disturbance patterns, Fiedler et al. (1996) advocated a resistance strategy of combining thinning with prescribed burning to reduce susceptibility to wildfire, and a resilience strategy of manipulating growing space to favor tree regeneration and control competition from thin-barked, shade-tolerant species. The authors recognize the different opportunities and constraints of the two treatments—the specificity and ability to remove large trees of harvesting, the efficiency in removing small trees, especially firs, the reduction in fuels and stimulation of seedbeds of prescribed burning—and advocate the judicious combination of both practices.

Table 2.1.3—Resilience and resistance characteristics of the major ecosystem types in Southern Nevada and guidelines for appropriate management actions (excerpted from table 7.2, Chambers et al. 2013).

Ecosystem	Resistance and resilience	Guidelines for appropriate management actions
Mixed-conifer	**Resilience**—Moderate to high. Relatively high precipitation, long growing seasons, and moderate growth and establishment rates. Potential to migrate upslope with climate warming. **Resistance**—Moderate to low. Multiple nonnative invaders adapted to environmental conditions; competition with invaders from established native plants can be high.	**Protection**—Control inappropriate recreational activities and overgrazing; detect and eradicate invasive species. **Prevention**—Warranted to decrease fuel loads, restore understory composition, and decrease invasion. Potential for wildland fire use and prescribed fire where risk of large or high-severity fire is low and fire spread can be controlled, and for tree thinning followed by surface fire or pile burning in wildland urban interface and areas with higher fuel loads. **Restoration**—Warranted following surface disturbance or in areas with insufficient fire-tolerant understory species for site recovery after fire. Seed burial (drilling) or transplanting natives adapted to local site conditions and climate warming preferred.
Piñon and juniper	**Resilience**—Moderate. Moderate precipitation, long growing seasons, moderate to slow growth and establishment. Potential for die-off at lower elevations with climate warming. **Resistance**—Low. Many nonnative invaders adapted to environmental conditions; competition from established shrubs and herbaceous species dependent on site productivity and ecological condition.	**Protection**—Control inappropriate recreational activities and overgrazing; detect and eradicate invasive species; suppress fires at lower elevations and that threaten ecosystem integrity. **Prevention**—Warranted to decrease fuel loads, restore understory composition, and decrease invasion. Focus is on mesic sites in early to intermediate stages of tree expansion, and in moderate to high ecological condition. Potential for wildland fire use and prescribed fire on productive sites at high elevation; mechanical treatments more appropriate on sites with low productivity. **Restoration**—Warranted following surface disturbance and in areas with insufficient fire-tolerant understory species for site recovery after fire. Seed burial (drilling) or transplanting natives adapted to local site conditions and climate warming preferred.

Ponderosa Pine

Restoring Ecosystem Function

To restore ecosystem structure and function, Youngblood et al. (2004) recommended that managers use: (1) reference conditions (such as the ones in their study) as guidelines, but recognize that variability occurs across all situations and thus treatments must be implemented accordingly; (2) keep existing live and dead old-growth ponderosa pines; (3) create and maintain clumps (or at least random distributions) of trees; (4) slowly reintroduce fire to control fuels and create both spatial and vertical heterogeneity; and (5) retain coarse woody debris. The interaction between the spatial patterns in structure and composition of dry forests and fire resistance means that restoring the one improves the other (Hagmann et al. 2013).

Improve Fire Resilience

Focusing more specifically on improving fire resilience in ponderosa pine forests, (Fitzgerald 2005) repeated the principles of: (1) reducing surface fuels, (2) removing ladder fuels, (3) retaining large trees that are more fire resistant, and (4) increasing the size and randomness of spacing between tree crowns, as put forth by Agee (2002).

Reintroducing Fire

Reintroducing fire into long-unburned areas could result in higher-than-desired mortality due to the accumulated surface and ladder fuels; many researchers (Covington et al. 1997; Fiedler et al. 1996; Fitzgerald 2005; Mast et al. 1999; Moore et al. 1999) recommend prior nonfire mechanical treatments, including thinning (fig. 2.1.13). Subsequent fires would then be less likely to escape and managers would be able to control the fuel loads in a safer manner.

Opening up the canopy in such a fashion is likely to change the microclimate of the stand. The resulting drier forest floors and increased within-stand wind speeds could potentially lead to increased fire intensity and spread (Fitzgerald 2005; Weatherspoon 1996), an outcome that may seem to contradict the goal of reducing severe fires. Fitzgerald (2005) points out that once the heavy fuel loads are reduced, such fire patterns can result in a presettlement-like forest structure (fig. 2.1.14).

Figure 2.1.13—Researchers have recommended preburn mechanical fuels treatments, such as this one within a riparian zone on the Modoc National Forest, before reintroducing fire into long-unburned areas. Balancing streamside fuels reduction with riparian zone protection from erosion and high temperatures is a challenging task (photo by James Brogan, Forest Service).

Figure 2.1.14—Prescribed fire can assist in reducing ground-level and ladder fuels, reduce stand density, create snags, such as this one next to Blue Lake on the Modoc National Forest, and restore those forest functions facilitated by fire (photo by John Cichoski, Forest Service).

Benefits of Silviculture

Thinning vs. Fire vs. Mortality

Even without catastrophic fires, untreated, dense stands face other risks. In a long-term study on the Blacks Mountain Experimental Forest (Crotteau and Ritchie 2014), two treatments (high and low structural diversity) were implemented on 12 stands with primarily ponderosa and Jeffrey pines in the overstory. Subsequent to harvest, half of the stands were burned and the treatments were compared to neighboring long-untreated (65 years) stands in a Research Natural Area (RNA). The authors observed that the RNA (burn, no-thin) had a higher mortality rate among large trees (24+ inches/60+ cm in diameter), resulting in a projection of few large trees into the future. Many of those that would remain were deemed to be at high risk to fire and beetle attack, casting their future into doubt as well. Thinned stands displayed less overstory mortality and an increase in overstory structure as ingrowth moved into the canopy. Thinned and burned stands experienced some delayed mortality due to the amount of elapsed time between the thinning and prescribed burning (but still much less than the untreated stands), but the fuels reductions were expected to improve the resilience of the stands to future fires (Crotteau and Ritchie 2014).

Benefits of Prescribed Fire

Prescribed burning can help lower stand density, reduce fuel loads, and generally restore ecosystem function to historically fire-maintained forests. Fires in 1990 and 1994 in the Ishi Wilderness in the Southern Cascades of California promoted ponderosa pine regeneration and demonstrated that even the infrequent use of fire can create presettlement structure and composition (Taylor 2010). Like most pine stands treated with prescribed fire (e.g., Boyer 1987; Crotteau and Ritchie 2014), a negative post-burn effect resulting in reduced stand growth can occur. Nonetheless, the effect of the treatments (burn and thin) decreases in influence over time. Crotteau and Ritchie (2014) suggest that these results support the claim that diverse, multiuse stands can produce as much as even-aged stands. Where fire is not available as a management tool, even logged stands may be more resilient than unburned, unlogged stands that have not been treated for more than 100 years.

Benefits of Thinning

Thinning, by reducing competition and resultant individual tree stress, is often recommended as a tool to restore dense stands to a more resilient state (Daniel et al. 1979; Nyland 2016; Smith et al. 1997) (fig. 2.1.15), including those stands that have been unburned and otherwise unmanaged (Covington et al. 1997). The remaining trees can then take advantage of the released growing space. Examples of this strategy are found throughout the region. Stands with high basal area and tree density, factors identified as negatively influencing diameter growth, are highly prone to lethal beetle attacks. Thinning of young eastside ponderosa pine stands in the 1960s and 1970s reduced the basal area to a level where remaining trees experienced less stress and were subsequently less vulnerable to insect attack (Fiddler et al. 1989). Conversely, researchers have related increased radial growth in thinned stands to lower beetle-caused

Figure 2.1.15—This view north of Parker Creek looking south toward the South Warner Wilderness, Modoc National Forest, is an example of the open nature of ponderosa pine and western juniper woodlands (photo by Ken Sandusky, Forest Service).

mortality levels in ponderosa pine stands in Oregon and South Dakota and higher levels of resin flow from wounds in Arizona (Kolb et al. 2007). Large trees may be at risk of mortality even without reintroduction of fire, and thinning may increase growing space per tree and increase survival of the large trees (Crotteau and Ritchie 2014). Kolb et al. (2007) concluded that "judicious thinning" could improve the health of residual large trees.

Forest Health Benefits

Forests with high density force trees to compete with each other for light, water, and nutrients (Oliver and Larson 1996) and thus have fewer resources available to defend against insects and disease. Accordingly, management that results in large increases in radial stem growth rate of ponderosa pine can also increase resistance to bark beetle attacks (Kolb et al. 2007). In Northern New Mexico, a study of managed second-growth ponderosa pine stands found that the thinned stands displayed characteristics of greater resilience, such as maintaining growth during drought and recovering after drought, and greater mean tree size and diameter growth, than stands that had not been treated (Thomas and Waring 2015). The authors concluded that even faced with severe drought stress, trees in thinned stands still had enough photosynthate to allocate to diameter growth. These stands with lower density and higher individual tree size were also believed to be more resistant to attack by insects, such as bark beetles. See Chapter 6.1, Wright, this synthesis, *Ecological Disturbance in the Context of a Changing Climate: Implications for Land Management in Northeastern California* for more discussion about bark beetles.

Importance of Large Trees

Restoration treatments often target retaining and developing large-diameter, fire-resistant pines, usually by thinning from below or broadcast burning, or a combination thereof, to recreate perceived historical stand structure and promote floral and faunal diversity (Crotteau and Ritchie 2014; Youngblood et al. 2004, and see Chapter 4.1, Hanberry and Dumroese, this synthesis, *Biodiversity and Representative Species in Dry Pine Forests*). Although the focus on large trees may underestimate the ecological relevance of smaller trees, whose presence can be hard to reconstruct (Fulé et al. 1997; Hagmann et al. 2013; Larson et al. 2012; Mast et al. 1999; North et al. 2007; but see Moore et al. 2004), large old pines provide the structure (or "backbone") of ponderosa pine and mixed-conifer forests (Franklin and Johnson 2012).

North et al. (2007) advise managers desiring to restore a presettlement forest structure after a period of suppression not to retain too many small trees. Some management recommendations require that all prefire suppression trees be retained in order to recreate historical structure. However, these trees grew up in a suppression environment, and retaining all of them may result in a denser forest than would have occurred under a normal fire regime (Abella et al. 2006; Larson et al. 2012).

Juniper Woodlands

Historically, piñon-juniper ecosystems have not been consistently or sustainably managed. More emphasis has been placed on removing woodland overstory to create or improve forage for livestock (Gottfried and Severson 1994; Tausch 1999). Thus, not a great deal of information is available for developing silvicultural prescriptions specific to the type. Ellenwood (1994) recommended two-step shelterwood and single-tree selection treatments which retained tree cover and sustained productivity. It is challenging, however, to manage this type economically as the market for small-diameter products is limited (Gottfried 2004). The selection method can reduce stand density while maintaining multiple ages and horizontal and vertical diversity. Managers can also retain trees in openings, but over the long term must decide whether the ultimate purpose of the openings is forage or tree regeneration. Openings intended for tree regeneration could temporarily benefit livestock and wild ungulates.

It would require large decreases in overstory to benefit livestock forage, but individual species of ground vegetation may respond differently (Gottfried and Severson 1994).

Restoration of juniper ecosystems to sagebrush benefit from a landscape-level focus (Tausch and Hood 2007), and include short-term and long-term components. In the near term, the principal risk to sagebrush ecosystems is from land use, whereas long-term risks might be equally derived from land use and climate change [effects] (Bradley 2010). Once trees are established, removal by harvesting, chaining, or fire is the principal tactic. Prescribed fire can be used to control tree establishment; however, the outcome depends upon the composition and structure present before the burning. Plant response after removal of western juniper depends on the seed pools in the soil, initial plant communities, soils, size of openings, and later management activities (Miller and Wigand 1994).

In their west-wide study of piñon-juniper ecosystems, Miller et al. (2008) recommended proactive management to reduce piñon-juniper woodlands over at least 50 percent of the study area. Failure to do so will result in the continued decrease in the historic sagebrush landscape. If the sagebrush component is still present, prescribed burning is more likely to be successful. Chaining can result in a 4- to 7-fold increase in biomass of herbaceous species, but the site could revert to a tree-dominated state within 25 years.

The choice of method requires calculating tradeoffs, especially in areas associated with greater sage-grouse (see Chapter 4.3, Dumroese, this synthesis, S*agebrush Rangelands and Greater Sage-grouse in Northeastern California*). Thinning will reduce basal area with less subsequent erosion, but chaining exposes more mineral soil to regeneration and is considerably cheaper. Seeding is necessary on sites with few understory plants or a limited seedbank; such sites are vulnerable to entry by invasive weeds or grasses (Tausch and Hood 2007). A vigorous juniper regeneration cohort can prove problematic in such a situation, however. In his study in Central Oregon, Eddleman (1986) found a large number of western juniper trees located under existing sagebrush or large juniper trees in the overstory. Mechanical clearing by itself would only serve to release the juniper regeneration. Management activities, such as post-treatment burning, may be necessary to prevent reversion to the previous woodland state.

References

Abella, S.R.; Fulé, P.Z.; Covington, W.W. 2006. Diameter caps for thinning southwestern ponderosa pine forests: viewpoints, effects, and tradeoffs. Journal of Forestry. 104: 407–414.

Adams, A. 1975. A brief history of juniper and shrub populations in southern Oregon. Wildlife Research Report 6. Corvallis, OR: Oregon State Wildlife Commission. 33 p.

Agee, J.K. 1993. Fire ecology of Pacific Northwest forests. Washington, DC: Island Press. 55 p.

Agee, J.K. 2002. Fire behavior and fire-resilient forests. In: Proceedings, Fire in Oregon's forests: risks, effects, and treatment options; Bend, OR; 2002 October 22–23. Portland, OR: Oregon Forest Resources Institute: 119–126.

Antevs, E.; Wright, J.K. 1938. Rainfall and tree growth in the Great Basin. Carnegie Institution of Washington Publ. 469; American Geographical Society Special Publ. 21. Washington, DC: Carnegie Institution of Washington and American Geographical Society of New York. 97 p.

Aronson, J.; Floret, C.; Floc'h, E.; [et al.]. 1993. Restoration and rehabilitation of degraded ecosystems in arid and semi-arid lands. I. A view from the South. Restoration Ecology. 1: 8–17.

Assmann, E. 1970. The principles of forest yield studies: studies in the organic production, structure, increment and yield of forest stands. Oxford, UK: Pergamon Press. 520 p.

Bailey, J.D.; Covington, W.W. 2002. Evaluating ponderosa pine regeneration rates following ecological restoration treatments in northern Arizona, USA. Forest Ecology and Management. 155: 271–278.

Bates, J.D.; Miller, R.F.; Svejcar, T.J. 2000. Understory dynamics in cut and uncut western juniper woodlands. Journal of Range Management. 53: 119–126.

Bell, D.M.; Bradford, J.B.; Lauenroth, W.K. 2014. Early indicators of change: divergent climate envelopes between tree life stages imply range shifts in the western United States. Global Ecology and Biogeography. 23: 168–180.

Bolsinger, C.L. 1989. California's western juniper and pinyon-juniper woodlands: area, stand characteristics, wood volume, and fenceposts. Resour. Bull. PNW-RB-166. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 37 p.

Bottero, A.; D'Amato, A.W.; Palik, B.J.; [et al.]. 2016. Density-dependent vulnerability of forest ecosystems to drought. Journal of Applied Ecology. 54: 1605–1614.

Boyer, W.D. 1987. Volume growth loss: a hidden cost of periodic prescribed burning in longleaf pine? Southern Journal of Applied Forestry. 11: 154–157.

Bradford, J.B.; Bell, D.M. 2017. A window of opportunity for climate-change adaptation: easing tree mortality by reducing forest basal area. Frontiers in Ecology and the Environment. 15: 11–17.

Bradley, B.A. 2010. Assessing ecosystem threats from global and regional change: hierarchical modeling of risk to sagebrush ecosystems from climate change, land use, and invasive species in Nevada, USA. Ecography. 33: 198–208.

Bradley, B.A.; Fleishman, E. 2008a. Relationships between expanding pinyon-juniper cover and topography in the central Great Basin, Nevada. Journal of Biogeography. 35: 951–964.

Bradley, B.A.; Fleishman, E.J. 2008b. Can remote sensing of land cover improve species distribution modelling? Journal of Biogeography. 35: 1158–1159.

Brown, P.M.; Wu, R. 2005. Climate and disturbance forcing of episodic tree recruitment in a southwestern ponderosa pine landscape. Ecology. 86: 3030–3038.

Bruner, A.D.; Klebenow, D.A. 1979. Predicting success of prescribed fires in pinyon-juniper woodlands in Nevada. Res. Pap. INT-RP-219. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 12 p.

Burkhardt, J.W.; Tisdale, E.W. 1969. Nature and successional status of western juniper vegetation in Idaho. Journal of Range Management. 22: 264–270.

Burkhardt, J.W.; Tisdale, E.W. 1976. Causes of juniper invasion in southwestern Idaho. Ecology. 57: 472–484.

Butry, D.T.; Mercer, E.; Prestemon, J.P.; [et al.]. 2001. What is the price of catastrophic wildfire? Journal of Forestry. 99: 9–17.

Chambers, J.C.; Pendleton, B.K.; Sada, D.W.; [et al.]. 2013. Maintaining and restoring sustainable ecosystems in southern Nevada. In: Chambers, J.C.; Brooks, M.L.; Pendleton, B.K.; [et al.], eds. The Southern Nevada agency partnership science and research synthesis: science to support land management in southern Nevada. Gen. Tech. Rep. RMRS-GTR-304. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 125–154. Chap. 7.

Clark, J.S.; Iverson, L.; Woodall, C.W.; [et al.]. 2016. The impacts of increasing drought on forest dynamics, structure, and biodiversity in the United States. Global Change Biology. 22: 2329–2352.

Cooper, C.F. 1961. Pattern in ponderosa pine forests. Ecology. 42: 493–499.

Cottam, W.P.; Stewart, G. 1940. Plant succession as a result of grazing and of meadow desiccation by erosion since settlement in 1862. Journal of Forestry. 38: 613–626.

Covington, W.W.; Fulé, P.Z.; Moore, M.M.; [et al.]. 1997. Restoring ecosystem health in ponderosa pine forests of the Southwest. Journal of Forestry. 95: 23–29.

Crotteau, J.S.; Ritchie, M.W. 2014. Long-term stand growth of interior ponderosa pine stands in response to structural modifications and burning treatments in northeastern California. Journal of Forestry. 112: 412–423.

D'Amato, A.W.; Bradford, J.B.; Fraver, S.; [et al.]. 2013. Effects of thinning on drought vulnerability and climate response in north temperate forest ecosystems. Ecological Applications. 23: 1735–1742.

Daniel, T.W.; Helms, J.A.; Baker, F.S. 1979. Principles of silviculture. 2nd ed. New York, NY: McGraw-Hill Book Co. 500 p.

de Liocourt, F. 1898. De l'aménagement des sapinières [On the amelioration of fir forests]. Bulletin Trimestiel, Société Forestière de Franche-Comté et Belfort. 6: 396–409.

DeRose, R.J.; Long, J.N. 2014. Resistance and resilience: a conceptual framework for silviculture. Forest Science. 60: 1205–1212.

Eddleman, L.E. 1986. Establishment and stand development of western juniper in central Oregon. In: Everett, R.L., comp. Proceedings—pinyon-juniper conference. Gen. Tech. Rep. INT-GTR-215. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 255–259.

Ellenwood, J.R. 1994. Silvicultural systems for pinyon-juniper. In: Shaw, D.W.; Aldon, E.F.; LoSapio, C., tech coords. Desired future conditions for piñon-juniper ecosystems; proceedings of the symposium. Gen. Tech. Rep. RM-GTR-258. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 203–208.

Evans, R.A. 1988. Management of pinyon-juniper woodlands. Gen. Tech. Rep. INT-GTR-249. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 34 p.

Everett, R.L.; Schellhaas, R.; Keenum, D.; [et al.]. 2000. Fire history in the ponderosa pine/Douglas-fir forests on the east slope of the Washington Cascades. Forest Ecology and Management. 129: 207–225.

Fiddler, G.O.; Fiddler, T.A.; Hart, D.R.; [et al.]. 1989. Thinning decreases mortality and increases growth of ponderosa pine in northeastern California. Res. Pap. PSW-RP-194. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 12 p.

Fiedler, C.E.; Arno, S.F.; Harrington, M.G. 1996. Flexible silvicultural and prescribed burning approaches for improving health of ponderosa pine forests. In: Covington, W.W.; Wagner, P.K., tech. coords. Conference on adaptive ecosystem restoration and management: restoration of Cordilleran conifer landscapes of North America. Gen. Tech. Rep. RM-GTR-278. Fort Collins: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 69–74.

Fitzgerald, S.A. 2005. Fire ecology of ponderosa pine and the rebuilding of fire-resilient ponderosa pine ecosystems. 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: 197–225.

Franklin, J.F.; Johnson, K.N. 2012. A restoration framework for federal forests in the Pacific Northwest. Journal of Forestry. 110: 429–439.

Fulé, P.Z.; Covington, W.W.; Moore, M.M. 1997. Determining reference conditions for ecosystem management of southwestern ponderosa pine forests. Ecological Applications. 7: 895–908.

Garfin, G.; Hughes, M. 1996. Eastern Oregon divisional precipitation and Palmer drought severity index from tree-rings. Unpubl. final report to the U.S. Department of Agriculture, Forest Service. Tucson, AZ: Laboratory of Tree-Ring Research. University of Arizona Cooperative Agreement PNW 90–174.

Gonzales, A.G.; Hoshi, J., eds. 2015. Cascades and Modoc Plateau Province. California state wildlife action plan, 2015 update: a conservation legacy for Californians. Vol. 1 Sacramento, CA: California Department of Fish and Wildlife with assistance from Ascent Environmental Inc. 51 p. Chap. 5.2. https://www.wildlife.ca.gov/SWAP/Final (2 Oct. 2017).

Gottfried, G.J. 2004. Silvics and silviculture in the southwestern pinyon-juniper woodlands. In: Shepperd, W.D.; Eskew, L.G., compilers. Silviculture in special places: proceedings of the National Silviculture Workshop. Proceedings RMRS-P-34. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 64–79.

Gottfried, G.J.; Severson, K.E. 1994. Managing pinyon-juniper woodlands. Rangelands. 16: 234–236.

Grace, S.L.; Platt, W.J. 1995. Effects of adult tree density and fire on the demography of pregrass stage juvenile longleaf pine (*Pinus palustris* Mill.). Journal of Ecology. 83: 75–86.

Graham, R.T.; Jain, T.B. 2005. Ponderosa pine ecosystems. 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: 1–32.

Graumlich, L.J. 1987. Precipitation variation in the Pacific Northwest (1675–1975) as reconstructed from tree rings. Annals of the Association of American Geographers. 77: 19–29.

Grimm, V.; Wissel, C. 1997. Babel, or the ecological stability discussions: an inventory and analysis of terminology and a guide for avoiding confusion. Oecologia. 109: 323–334.

Hagmann, R.K.; Franklin, J.F.; Johnson, K.N. 2013. Historical structure and composition of ponderosa pine and mixed-conifer forests in south-central Oregon. Forest Ecology and Management. 304: 492–504.

Helms, J.A. 1998. The dictionary of forestry. Bethesda, MD: Society of American Foresters. 210 p.

Helms, J.A.; Tappeiner, J.C. 1996. Silviculture in the Sierra. In: Erman, D.C., gen. ed. and the SNEP Science Team. Sierra Nevada Ecosystem Project: final report to Congress. Vol. IV: Final report to Congress. Davis, CA: University of California, Davis, Centers for Water and Wildland Resources: 439–476.

Heyerdahl, E.K.; Brubaker, L.B.; Agee, J.K. 2001. Spatial controls of historical fire regimes: a multiscale example from the interior west, USA. Ecology. 82: 660–678.

Higgs, E.S. 1997. What is good ecological restoration? Conservation Biology. 11: 338–348.

Hobbs, R.J.; Cramer, V.A. 2008. Restoration ecology: interventionist approaches for restoring and maintaining ecosystem function in the face of rapid environmental change. Annual Review of Environment and Resources. 33: 39–61.

Houghton, J.G.; Sakamoto, C.M.; Gifford, R.O. 1975. Nevada's weather and climate. Reno, NV: Nevada Bureau of Mines and Geology; University of Nevada, Mackay School of Mines. 84 p.

Huffman, D.W.; Crouse, J.E.; Chancellor, W.W.; Fulé, P.Z. 2012. Influence of time since fire on pinyon-juniper woodland structure. Forest Ecology and Management. 274: 29–37.

Jackson, L.L.; Lopoukhine, N.; Hillyard, D. 1995. Ecological restoration: a definition and comments. Restoration Ecology. 3: 71–75.

Kauffman, J.B. 1990. Ecological relationships of vegetation and fire in Pacific Northwest forests. In: Walstad, J.D.; Radosevich, S.R.; Sandberg, D.V., eds. Natural and prescribed fire in Pacific Northwest forests. Corvallis, OR: Oregon State University Press: 39–52.

Knapp, E.E.; Lydersen, J.M.; North, M.P.; [et al.]. 2017. Efficacy of variable density thinning and prescribed fire for restoring forest heterogeneity to mixed-conifer forest in the central Sierra Nevada, CA. Forest Ecology and Management. 406: 228–241.

Kolb, T.; Agee, J.; Fulé, P.; [et al.]. 2007. Perpetuating old ponderosa pine. Forest Ecology and Management. 249: 141–157.

Koniak, S. 1985. Succession in pinyon-juniper woodlands following wildfire in the Great Basin. The Great Basin Naturalist. 45: 556–566.

Larson, A.J.; Stover, K.C.; Keyes, C.R. 2012. Effects of restoration thinning on spatial heterogeneity in mixed-conifer forest. Canadian Journal of Forest Research. 42: 1505–1517.

Laudenslayer, W.F., Jr. 2002. Effects of prescribed fire on live trees and snags in eastside pine forests in California. In: In: Sugihara, N.G.; Morales, M.E.; Morales, T.J., eds. Proceedings of the symposium, fire in California ecosystems: integrating ecology, prevention and management. Miscellaneous Publication 1. Eugene, OR: Association for Fire Ecology: 256–262.

Laudenslayer, W.F., Jr.; Darr, H.H.; Smith, S. 1989. Historical effects of forest management practices on eastside pine communities in northeastern California. In: Tecle, A.; Covington, W.W.; Hamre, R.H., tech. coords. Multiresource management of ponderosa pine forests Gen. Tech. Rep. RM-185. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 26–34.

Leffler, A.J.; Caldwell, M.M. 2005. Shifts in depth of water extraction and photosynthetic capacity inferred from stable isotope proxies across an ecotone of *Juniperus osteosperma* (Utah juniper) and *Artemisia tridentata* (big sagebrush). Journal of Ecology. 93: 783–793.

Lévesque, M.; Siegwolf, R.; Saurer, M.; [et al.]. 2014. Increased water-use efficiency does not lead to enhanced tree growth under xeric and mesic conditions. New Phytologist. 203: 94–109.

Loft, E.R. 1998. Economic contribution of deer, pronghorn antelope, and sage grouse hunting to northeastern California and implications to the overall "value" of wildlife. California Wildlife Conservation Bulletin 11. Sacramento, CA: California Department of Fish and Game. 42 p.

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.

Lydersen, J.; North, M. 2012. Topographic variation in structure of mixed-conifer forests under an active-fire regime. Ecosystems. 15: 1134–1146.

Madany, M.H.; West, N.E. 1983. Livestock grazing-fire regime interactions within montane forests of Zion National Park, Utah. Ecology. 64: 661–667.

Maherali, H.; DeLucia, E.H. 2000. Xylem conductivity and vulnerability to cavitation of ponderosa pine growing in contrasting climates. Tree Physiology. 20: 859–867.

Manion, P.D. 1991. Tree disease concepts. Englewood Cliffs, NJ: Prentice-Hall, Inc. 399 p.

Mast, J.N.; Fulé, P.Z.; Moore, M.M.; [et al.]. 1999. Restoration of presettlement age structure of an Arizona ponderosa pine forest. Ecological Applications. 9: 228–239.

Mehringer, P.J.J.; Wigand, P.E. 1987. Western juniper in the Holocene. In: Everett, R.L., comp. Proceedings - pinyon-juniper conference. Gen. Tech. Rep. INT-GTR-215. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 109–119.

Merriam, C.H. 1898. Life zones and crop zones of the United States. Biological Survey Bulletin. 10. Washington, DC: U.S. Department of Agriculture. 79 p.

Millar, C.I.; Stephenson, N.L.; Stephens, S.L. 2007. Climate change and forests of the future: managing in the face of uncertainty. Ecological Applications. 17: 2145–2151.

Miller, M. 2000. Fire autecology. In: Brown, J.K.; Smith, J.K., eds. Wildland fire in ecosystems: effects of fire on flora. Gen. Tech. Rep. RMRS-GTR-42-volume 2. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 9–34.

Miller, R.F.; Rose, J.A., 1995. Historic expansion of *Juniperus occidentalis* (western juniper) in southeastern Oregon. The Great Basin Naturalist. 55: 37–45.

Miller, R.F.; Rose, J.A. 1999. Fire history and western juniper encroachment in sagebrush steppe. Journal of Range Management. 55: 550–559.

Miller, R.F.; Tausch, R.J. 2001. The role of fire in pinyon and juniper woodlands: a descriptive analysis. In: Galley, K.E.M.; Wilson, T.P.; Sugihara, N.G., eds. Fire conference 2000: The first national congress on fire ecology, prevention, and management. Tall Timbers Research Station Misc. Publ. 13. Tallahassee, FL: Tall Timbers Research Station: 15–30.

Miller, R.F.; Tausch, R.J.; McArthur, E.D.; [et al.]. 2008. Age structure and expansion of piñon-juniper woodlands: a regional perspective in the Intermountain West. Res. Pap. RMRS-RP-69. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 15 p.

Miller, R.F.; Wigand, P.E. 1994. Holocene changes in semiarid pinyon-juniper woodlands. BioScience. 44: 465–474.

Moore, M.M.; Covington, W.W.; Fulé, P.Z. 1999. Reference conditions and ecological restoration: a southwestern ponderosa pine perspective. Ecological Applications. 9: 1266–1277.

Moore, M.M.; Huffman, D.W.; Fulé, P.Z.; [et al.]. 2004. Comparison of historical and contemporary forest structure and composition on permanent plots in southwestern ponderosa pine forests. Forest Science. 50: 162–176.

Munger, T.T. 1917. Western yellow pine in Oregon. Bull. 418. Washington, DC: U.S. Department of Agriculture, Forest Service. 57 p.

Nolan, C.; Overpeck, J.T.; Allen, J.R.; [et al.]. 2018. Past and future global transformation of terrestrial ecosystems under climate change. Science. 361: 920–923.

Norman, S.P.; Taylor, A.H. 2003. Tropical and north Pacific teleconnections influence fire regimes in pine-dominated forests of north-eastern California, USA. Journal of Biogeography. 30: 1081–1092.

Norman, S.P.; Taylor, A.H. 2005. Pine forest expansion along a forest-meadow ecotone in northeastern California, USA. Forest Ecology and Management. 215: 51–68.

North, M.; Hurteau, M.; Innes, J. 2009a. Fire suppression and fuels treatment effects on mixed-conifer carbon stocks and emissions. Ecological Applications. 19: 1385–1396.

North, M.; Innes, J.; Zald, H. 2007. Comparison of thinning and prescribed fire restoration treatments to Sierran mixed-conifer historic conditions. Canadian Journal of Forest Research. 37: 331–342.

North, M.; Stine, P.; O'Hara, K.; Zielinski, W.; [et al.]. 2009b. An ecosystem management strategy for Sierran mixed-conifer forests. Gen. Tech. Rep. PSW-GTR-220 (2nd printing, with addendum). Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 49 p.

Noss, R.F.F.; Franklin, J.F.; Baker, W.L.; [et al.]. 2006. Managing fire-prone forests of the western United States. Frontiers in Ecology and the Environment. 4: 481–487.

Nyland, R.D. 2016. Silviculture: concepts and applications. 3rd ed. Long Grove, IL: Waveland Press. 680 p.

O'Hara, K.L. 2002. The historical development of uneven-aged silviculture in North America. Forestry. 75: 339–346.

O'Hara, K.L. 2015. What is close-to-nature silviculture in a changing world? Forestry: An International Journal of Forest Research. 89(1): 1–6.

O'Hara, K.L. 2016. What is close-to-nature silviculture in a changing world? Forestry. 89: 1–6.

O'Hara, K.L.; Latham, P.A.; Hessburg, P.; [et al.]. 1996. Technical commentary: a structural classification for inland northwest forest vegetation. Western Journal of Applied Forestry. 11: 97–102.

Oliver, C.D.; Larson, B.C. 1996. Forest stand dynamics: update edition. New York, NY: John Wiley and Sons. 520 p.

Oliver, W.W. 2000. Ecological research at the Blacks Mountain Experimental Forest in northeastern California. Gen. Tech. Rep. PSW-GTR-179. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 66 p.

Oliver, W.W. 2001. Can we create and sustain late successional attributes in interior ponderosa pine stands? Large-scale ecological research studies in northeastern California. In: Vance, R.K.; Edminster, C.B.; Covington, W.W.; [et al.], eds. Ponderosa pine ecosystems restoration and conservation: steps toward stewardship. Proceedings RMRS-P-22. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 99–103.

Oliver, W.W.; Ryker, R.A. 1990. *Pinus ponderosa* Dougl. ex Laws. ponderosa pine. In: Burns, R.M.; Honkala, B.H., tech. coords. Silvics of North America. 1: Conifers. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture, Forest Service: 413.

Pallardy, S.G. 2008. Physiology of woody plants. 3rd ed. Burlington, MA: Academic Press. 464 p.

Pearson, G. 1920. Factors controlling the distribution of forest types, part II. Ecology. 1: 289–308.

Pearson, G.A. 1950. Management of ponderosa pine in the Southwest as developed by research and experimental practice. Agriculture Monograph 6. Washington, DC: U.S. Department of Agriculture, Forest Service. 218 p.

Phillips, F.J. 1909. A study of piñon pine. Botanical Gazette. 48: 216–223.

PRBO Conservation Science. 2011. Projected effects of climate change in California: ecoregional summaries emphasizing consequences for wildlife, version 1.0. Petaluma, CA: PRBO Conservation Science. 68 p.

Puettmann, K.J. 2011. Silvicultural challenges and options in the context of global change: "simple" fixes and opportunities for new management approaches. Journal of Forestry. 109: 321–331.

Raumann, C.G.; Cablk, M.E. 2008. Change in the forested and developed landscape of the Lake Tahoe basin, California and Nevada, USA, 1940–2002. Forest Ecology and Management. 255: 3424–3439.

Reed, G.; Gaines, R. 1949. Gold rush: the journals, drawings and other papers of J. Goldsborough Bruff, April 2, 1849–July 20, 1851. New York: Columbia University Press. 794 p.

Riegel, G.M.; Miller, R.F.; Skinner, C.N.; [et al.]. 2006. Northeastern plateaus bioregion. In: Sugihara, N.; van Wagtendonk, J.; Fites-Kaufmann, J.; [et al.], eds. Fire in California's ecosystems. 1st ed. Oakland, CA: University of California Press: 225–263.

Ritchie, M.W.; Wing, B.M.; Hamilton, T.A. 2008. Stability of the large tree component in treated and untreated late-seral interior ponderosa pine stands. Canadian Journal of Forest Research. 38: 919–923.

Romme, W.H.; Allen, C.D.; Bailey, J.D.; [et al.]. 2009. Historical and modern disturbance regimes, stand structures, and landscape dynamics in piñon-juniper vegetation of the western United States. Rangeland Ecology and Management. 62: 203–222.

Rummell, R.S. 1951. Some effects of livestock grazing on ponderosa pine forest and range in central Washington. Ecology. 32: 594–607.

Safford, H.D.; Stevens, J.T. 2017. Natural range of variation for yellow pine and mixed-conifer forests in the Sierra Nevada, southern Cascades, and Modoc and Inyo National Forests, California, USA. Gen. Tech. Rep. PSW-GTR-256. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 229 p.

Sánchez Meador, A.; Moore, M.; Bakker, J.; [et al.]. 2009. 108 years of change in spatial pattern following selective harvest of a *Pinus ponderosa* stand in northern Arizona, USA. Journal of Vegetation Science. 20, 79–90.

Savage, M.; Swetnam, T.W. 1990. Early 19th-century fire decline following sheep pasturing in a Navajo ponderosa pine forest. Ecology. 71: 2374–2378.

Schmidt, K.M.; Menakis, J.P.; Hardy, C.C.; [et al.]. 2002. Development of coarse-scale spatial data for wildland fire and fuel management. Gen. Tech. Rep. RMRS-GTR-87. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 41 p.

Shuffield, C.D. 2010. Overstory composition and stand structure shifts within inter-mixed ponderosa pine and lodgepole pine stands of the south-central Oregon pumice zone. Corvallis, OR: Oregon State University. 138 p. M.S. thesis.

Smith, D.M.; Larson, B.C.; Kelty, M.J.; [et al.]. 1997. The practice of silviculture: applied forest ecology. 9th ed. New York, NY: John Wiley and Sons. 537 p.

Society for Ecological Restoration [SER]. 1993. Mission statement, Society for Ecological Restoration. Restoration Ecology. 1: 206–207.

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.L.; Lydersen, J.M.; Collins, B.M.; [et al.]. 2015. Historical and current landscape-scale ponderosa pine and mixed conifer forest structure in the Southern Sierra Nevada. Ecosphere. 6: 1–63.

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. Environmental Research Letters. 5: 024003.

Stephenson, N.L. 1999. Reference conditions for giant sequoia forest restoration: structure, process, and precision. Ecological Applications. 9: 1253–1265.

Stine, P. 1996. Climate, 1650–1850. In: Erman, D.C., gen. ed. and the SNEP Science Team. Sierra Nevada Ecosystem Project. Vol. II: Assessments and scientific basis for management options. Davis, CA: University of California, Davis, Centers for Water and Wildland Resources: 25–30.

Sudworth, G.B. 1900. Stanislaus and Lake Tahoe forest reserves, California, and adjacent territory. In: Annual reports of the Department of the Interior, 21st annual report of the U.S. Geological Survey, part 5. Washington, DC: U.S. Government Printing Office: 505–561.

Swetnam, T.W.; Betancourt, J.L. 1998. Mesoscale disturbance and ecological response to decadal climatic variability in the American Southwest. Journal of Climate. 11: 3128–3147.

Tausch, R.J. 1999. Historic pinyon and juniper woodland development. In: Monsen, S.B.; Stevens, R., compilers. Proceedings: ecology and management of pinyon-juniper communities within the Interior West Proc. RMRS-P-9. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research: 12–19.

Tausch, R.J.; Hood, S. 2007. Chapter 4: Pinyon/juniper woodlands. In: Hood, S.M.; Miller, M., eds. Fire ecology and management of the major ecosystems of southern Utah. Gen. Tech. Rep. RMRS-GTR-202. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 57–71.

Tausch, R.J.; Tueller, P.T. 1990. Foliage biomass and cover relationships between tree- and shrub-dominated communities in pinyon-juniper woodlands. The Great Basin Naturalist. 50: 121–134.

Tausch, R.J.; West, N.E. 1988. Differential establishment of pinyon and juniper following fire. American Midland Naturalist. 119: 174–184.

Tausch, R.J.; West, N.E.; Nabi, A. 1981. Tree age and dominance patterns in Great Basin pinyon-juniper woodlands. Journal of Range Management. 34: 259–264.

Taylor, A.H. 2000. Fire regimes and forest changes in mid and upper montane forests of the southern Cascades, Lassen Volcanic National Park, California, USA. Journal of Biogeography. 27: 87–104.

Taylor, A.H. 2004. Identifying forest reference conditions on early cut-over lands, Lake Tahoe Basin, USA. Ecological Applications. 14: 1903–1920.

Taylor, A.H. 2010. Fire disturbance and forest structure in an old-growth *Pinus ponderosa* forest, southern Cascades, USA. Journal of Vegetation Science. 21: 561–572.

Taylor, A.H.; Skinner, C.N. 1998. Fire history and landscape dynamics in a late-successional reserve, Klamath Mountains, California, USA. Forest Ecology and Management. 111: 285–301.

Taylor, A.H.; Skinner, C.N. 2003. Spatial patterns and controls on historical fire regimes and forest structure in the Klamath Mountains. Ecological Applications. 13: 704–719.

Thomas, Z.; Waring, K.M. 2015. Enhancing resiliency and restoring ecological attributes in second-growth ponderosa pine stands in northern New Mexico, USA. Forest Science. 61: 93–104.

U.S. Department of Agriculture, Natural Resources Conservation Service [USDA NRCS]. 2017. The PLANTS Database. Greensboro, NC: U.S. Department of Agriculture, Natural Resources Conservation Service, National Plants Data Team. https://plants.sc.egov.usda.gov/java/ (1 Sep. 2019).

Van de Water, K.M.; Safford, H.D. 2011. A summary of fire frequency estimates for California vegetation before Euro-American settlement. Fire Ecology. 7: 26–58.

Van Gunst, K.J.; Weisberg, P.J.; Yang, J.; [et al.]. 2016. Do denser forests have greater risk of tree mortality: a remote sensing analysis of density-dependent forest mortality. Forest Ecology and Management. 359: 19–32.

Van Hooser, D.; Keegan, C.E. 1988. Distributions and volumes of ponderosa pine forests. In: Baumgartner, D.M.; Lotan, J.E., eds. Symposium proceedings, Ponderosa pine—the species and its management. Pullman, WA: Washington State University, Cooperative Extension: 1–6.

Vankat, J. 2013. Vegetation dynamics on the mountains and plateaus of the American Southwest. Plant and Vegetation, vol. 8. Dordrecht, the Netherlands and New York: Springer Science & Business Media. 461 p.

Volland, L.A. 1963. Phytosociology of the ponderosa pine type on pumice soils in the Upper Williamson River Basin, Klamath County, Oregon. Corvallis, OR: Oregon State University. 166 p. M.S. thesis.

Waichler, W.S.; Miller, R.F.; Doescher, P.S. 2001. Community characteristics of old-growth western juniper woodlands. Journal of Range Management. 54: 518–527.

Walker, B.; Holling, C.S.; Carpenter, S.; [et al.]. 2004. Resilience, adaptability and transformability in social-ecological systems. Ecology and Society. 9: 5.

Weatherspoon, C.P., 1996. Fire-silviculture relationships in Sierra forests. In: Erman, D.C., gen. ed. and the SNEP Science Team. Sierra Nevada Ecosystem Project: final report to Congress. Vol II: Assessments and scientific basis for management options. Davis, CA: University of California, Centers for Water and Wildland Resources: 1167–1176.

Welch, K.R.; Safford, H.D.; Young, T.P. 2016. Predicting conifer establishment post wildfire in mixed conifer forests of the North American Mediterranean-climate zone. Ecosphere. 7: e01609.

West, N.E.; Tausch, R.J.; Tueller, P.T. 1998. A management-oriented classification of pinyon-juniper woodlands of the Great Basin. Gen. Tech. Rep. RMRS-GTR-12. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 42 p.

West, N.E.; Van Pelt, N.S. 1986. Successional patterns in pinyon-juniper woodlands. In: Everett, R.L., comp. Proceedings—pinyon-juniper conference. Gen. Tech. Rep. INT-GTR-215. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 43–52.

White, A.S. 1985. Presettlement regeneration patterns in a southwestern ponderosa pine stand. Ecology. 66: 589–594.

Williams, A.P.; Allen, C.D.; Macalady, A.K.; [et al.]. 2013. Temperature as a potent driver of regional forest drought stress and tree mortality. Nature Climate Change. 3: 292–297.

Wortley, L.; Hero, J.M.; Howes, M. 2013. Evaluating ecological restoration success: a review of the literature. Restoration Ecology. 21: 537–543.

Young, J.A.; Evans, R.A. 1981. Demography and fire history of a western juniper stand. Journal of Range Management. 34, 501–506.

Youngblood, A.; Max, T.; Coe, K. 2004. Stand structure in eastside old-growth ponderosa pine forests of Oregon and northern California. Forest Ecology and Management. 199: 191–217.

Section 3. Rangeland

Chapter 3.1. Perceptions and History of Rangeland

Steven D. Warren[1]

Historical Perceptions of Rangeland

When asked today to define range or rangeland, most people respond with something such as "open lands used for grazing by livestock." That perception, however common, is incorrect. Range or rangeland is a type of land, not a type of use. While livestock grazing is, indeed, a common use of rangeland, it is by no means the only one. The definition of rangeland is variable and has changed over the years. The 1872 poem and the later classic western folk song by Brewster Higley, *Home on The Range,* predated the massive expansion of livestock in the Western United States and makes no mention of livestock; it refers only to buffalo, deer, and antelope. A century later, even after the explosion of the livestock population, the published definition of rangelands was "...those areas of the world, which by reason of physical limitations—low and erratic precipitation, rough topography, poor drainage, or cold temperatures—are unsuited to cultivation and which are a source of forage for free-ranging native and domestic animals, as well as a source of wood products, water and wildlife" (Stoddart et al. 1975). This definition makes it clear that the presence of domestic livestock, however common, is not necessary for land to be classified as rangeland. Twenty years later, the definition, published again without inclusion of domestic livestock, was "a type of land that supports different vegetation types including shrublands such as deserts and chaparral, grasslands, steppes, woodlands, temporary treeless areas in forests, and wherever dry, sandy, rocky, saline or wet soils, and steep topography preclude the growing of commercial farm and timber crops" (Heady and Child 1994). A decade later, the definition was "all areas of the world that are not barren deserts, farmed, or covered by bare soil, rock, ice, or concrete" (Holechek et al. 2004). The presence of, and use by, domestic livestock for grazing is not, and never has been, a requisite part of the definition of rangeland. However, because grazing by domestic livestock has been a common and easily recognized use of many rangelands for more than a century, people unfamiliar with the history of rangelands typically equate the two. Thus, a good working definition of "range" may be "[a]ll lands, except for urban, agricultural or densely forested lands, that support predominantly native or naturalized vegetation capable of sustaining native or domestic grazing and/or browsing ungulates, whether or not those animals are present."

History of Grazing in the Western United States

Grazing domestic livestock in the Western United States has occurred for nearly 500 years, since the days of the Spanish explorer Coronado (Holechek et al. 2004; Reyes Castañeda et al. 1904). Cattle arrived in San Diego in 1769, but it was the discovery of gold in California in 1849 that brought more than 300,000 people (Larson-Praplan 2014), and with them, a demand for red meat that was largely met by sheep (Beck and Haase 1989) and later by beef (Jelenik 1999). Demand for beef changed the grazing paradigm in California from small herds to nearly a million head by 1860 (Larson-Praplan 2014). Cattle numbers remained limited in much of the Western United States until the end of the Civil War in 1865, which triggered rapid expansion of people and the associated livestock industry across the West, facilitated by railroads and favorable Federal policies toward homesteading (e.g., the Homestead Act of 1862) and grazing. The number of range cattle and sheep boomed between 1880 and 1890 (Stewart 1936) with most grazing unregulated and on public land. Land not claimed under the Homestead Act of 1862 was considered part of the public domain and was available for use by almost anyone for almost any legal activity; this became the "open range" (Larson-Praplan 2014). Ranchers were content to use the "public domain" because it was free and not susceptible to taxation. Because nobody owned the land, nobody was responsible to maintain it, and public domain lands were severely abused and overgrazed. A consequence of the public domain policy is often referred to as an example of the "Tragedy of the Commons" (Hardin 1968).

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In 1891, the Forest Reserve Act withdrew most unsettled tracts of forested land from the public domain and put it under the administration of what would become the U.S. Department of Agriculture, Forest Service for timber production, while some adjacent and intermixed parcels of rangeland were allocated for grazing. This continues to create no small degree of confusion, as the U.S. Grazing Service (now the U.S. Department of the Interior, Bureau of Land Management) became responsible for managing most rangeland and some lands that contain forest.

To encourage continued settlement of marginal lands, Congress passed the Enlarged Homestead Act of 1909 granting 320 acres to homesteaders for farming (fig. 3.1.1). In 1916, Congress passed the Stockraising Homestead Act granting 640 acres to ranchers if they raised at least 50 cows, although the General Accounting Office reported that on much of the remaining public domain land, due to climate and/or the degraded condition of the land, it would require 160 acres to support a single cow for one month.

To reverse a trend of declining range condition, the Department of the Interior began granting grazing permits in 1898 to try to limit the number of livestock on federal land under its jurisdiction, and the Forest Service, established in 1905, initiated a system of grazing allotments for rangeland under its jurisdiction. Between 1910 and 1920, a series of laws were established on Forest Service lands to further regulate grazing. As the laws were established and enforced, the condition of grazed rangelands gradually improved. In the 1920s, the discipline of range management evolved and several universities established departments of range management to train landowners and managers in ecological principles related to the management of rangelands subjected to grazing.

In 1934, Congress passed the Taylor Grazing Act that withdrew all remaining land from the public domain and placed it under the jurisdiction of what would become the Bureau of Land Management (Hurlburt 1935) in

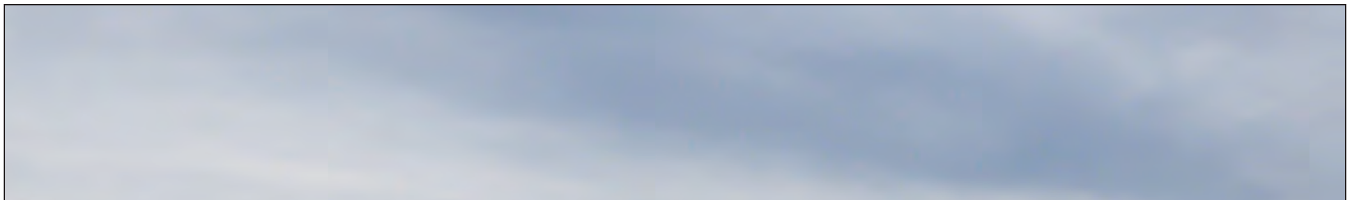

Figure 3.1.1—Vestiges of early homesteading and ranching can still be seen on the Lassen and Modoc National Forests (photo by KC Pasero, Forest Service).

order to "...stop injury to the public lands by preventing overgrazing and soil deterioration." Under the Taylor Grazing Act, ranchers could obtain long-term leases on public land for the purpose of livestock grazing. Stocking rates were established locally by the Bureau of Land Management. *The Western Range*, by Secretary of Agriculture, H.A. Wallace, and Chief of the Forest Service, F.A. Silcox, was published in 1936. Wallace and Silcox (1936), and a chapter authored by Clapp (1936), indicated that rangelands occupied approximately 728 million acres (295 million ha), or 40 percent of the total land area of the coterminous United States. In 1948, the American Society of Range Management (now known as The Society for Range Management) was established as a professional organization dedicated to fostering proper management of rangelands (Howery 2015). Many changes have been implemented, including severe reductions in stocking rates on many permits. While some lands continue to be severely overgrazed, the slow but generalized trend is toward improved rangeland condition (Mitchell 2000).

References

Beck, W.A.; Haase, Y.D. 1989. Historical atlas of the American west. Norman, OK: University of Oklahoma Press.

Clapp, E.H. 1936. The major range problems and their solution: a resumé. In: Wallace, H.A.; Silcox, F.A., eds. The Western Range. Washington, DC: U.S. Government Printing Office: 1–69.

Hardin, G. 1968. The tragedy of the commons. Science. 162:1243–1248.

Heady, H.F.; Child, R.D. 1994. Rangeland ecology and management. Boulder, CO: Westview Press. 522 p.

Holechek, J.L.; Pieper, R.D.; Herbel, C.H. 2004. Range management: principles and practices, 5th ed. Upper Saddle River, NJ: Pearson–Prentice Hall.

Howery, J.D. 2015. A brief history of how the Society for Range Management was founded. Rangelands. 37(1): 20–25.

Hurlburt, V. 1935. The Taylor Grazing Act. The Journal of Land & Public Utility Economics. 11: 203–206.

Jelinek, L.J. 1999. "Property of every kind": ranching and farming during the Gold-Rush era. California History. 77(4): 233–249.

Larson-Praplan, S. 2014. History of rangeland management in California. Rangelands. 36: 11–17.

Mitchell, J.E. 2000. Rangeland resource trends in the United States: A technical document supporting the 2000 USDA Forest Service RPA Assessment. Gen. Tech. Rep. RMRS-GTR-68. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 84 p.

Reyes Castañeda, P.R.; Vásquez de Coronado, F.; de Mendoza, A.; [et al.]. 1904. The journey of Coronado: 1540–1542; from the City of Mexico to the Grand Cañon of the Colorado and the Buffalo Plains of Texas, Kansas, and Nebraska, as told by himself and his followers. New York: A.S. Barnes & Company.

Stewart, G. 1936. History of range use. In: Wallace, H.A.; Silcox, F.A., eds. The Western Range. Washington, DC: U.S. Government Printing Office: 119–133.

Stoddart, L.A.; Smith, A.D.; Box, T.W. 1975. Range Management, 3rd ed. New York: McGraw-Hill Book Company.

Wallace, H.A.; Silcox, F.A. 1936. The western range. Washington, DC: U.S. Government Printing Office. 620 p.

Chapter 3.2. Rangeland in Northeastern California

R. Kasten Dumroese[1]

Introduction

Estimates of rangeland in the United States vary widely depending on the definition used, but, in general about one-third of the area of the United States (511 to 662 million acres [207 to 268 million ha] out of 2 billion acres [800 million ha] in the coterminous United States) is considered rangeland (Reeves and Mitchell 2011; USDA 2016; USDOI 2013). Rangeland should not be confused with grazing land that includes rangeland, pastureland, forestland, or any other land with potential for providing forage for wild or domestic ungulates (Society for Range Management 1998, and see Chapter 3.1, Warren, this synthesis, *Perceptions and History of Rangeland*). The U.S. Department of the Interior manages grazing on about 200 million acres (81 million ha), of which its Bureau of Land Management (BLM) oversees grazing on about 155 million acres (63 million ha) (USDOI 2013), although only about 135 million acres (55 million ha) of the BLM-managed ground are considered rangelands (Reeves and Mitchell 2012). Similarly, although the U.S. Department of Agriculture, Forest Service manages grazing on 95 million acres (38 million ha) (USDA 2016), it only manages about 50 million acres (20 million ha) of rangeland (Reeves and Mitchell 2011).

Forest Service rangeland is managed to sustain its health, diversity, and productivity and to meet society's needs now and in the future (USDA 2015; see USDA 2012 for an overview of the Nation's rangeland status). Rangeland productivity is often associated with forage for livestock. In the 17 Western States, BLM-managed forage accounted for about 1.6 percent of the total livestock receipts (USDOI 2013). Although the number of domestic animals permitted to graze on public lands continues its 100-year decline (USDOI 2013), grazing remains an important employment sector in the Sierra Nevada (Stewart 1996) and plays a role in sustaining viable ranching operations that maintain large tracts of open space (Huntsinger et al. 2010).

Rangeland constitutes a large proportion of the Lassen and Modoc National Forests (hereafter the Lassen and the Modoc). Of the Modoc's 1.6 million acres (647,000 ha), about 1 million (405,000 ha) is considered rangeland, of which 90 percent is suitable for livestock grazing (USDA and USDOI 2007). Of that, about 20 percent (320,000 acres [130,000 ha]) is covered with sagebrush (*Artemisia* spp.) ecosystem (USDA 1991). On the Lassen, about 410,000 acres (166,000 ha) of the forest's 1.2 million acres (486,000 ha) is suitable for grazing. Rangelands include about 27,500 acres (11,130 ha) of sagebrush, 20,000 acres (8100 ha) of antelope bitterbrush (*Purshia tridentata*), and 16,000 acres (6500 ha) grasslands (USDA 1992). Native ungulates on this rangeland include mule deer (*Odocoileus hemionus*) and pronghorn (*Antilocapra americana*). Historically, bison (*Bison bison*) may have also grazed in Northeastern California (Merriam 1926 cited in Norman and Taylor 2005). Both forests have a long tradition of cattle and sheep grazing (see Brown 1945 and Norman and Taylor 2005) that continues through grazing allotments administered through a permit system (fig. 3.2.1). Wild horses also graze these rangelands. Recent, successful breeding packs of gray wolf (*Canis lupus*) in Siskiyou and Lassen Counties suggest these carnivores may become a factor in the management of native ungulates, livestock, and wild horses (see Chapter 4.1, Hanberry and Dumroese, this synthesis, *Biodiversity and Representative Species in Dry Pine Forests*).

Grazing in the West, in California, and on the Lassen and the Modoc, is an important topic because changes in grazing management can have profound effects on plant community composition, fire occurrence and effects, and ranch income and market value (Lewandrowski and Ingram 2002), *The Science Synthesis to Support Socioecological Resilience in the Sierra Nevada and Southern Cascade Range* (hereafter, Sierra Nevada Science Synthesis) published by Long et al. (2014) presented significant discussion about grazing. In particular is Chapter 6.3—*Wet Meadows* (Long and Pope 2014). Despite that seemingly narrow title, the authors note that grazing "involves a complex interplay of social and ecological factors." They also provide a broad overview of grazing strategies applicable to diverse landscapes that includes the socioecological perspective and identify opportunities where disturbance through grazing can potentially help land managers meet desired ecological

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Figure 3.2.1—A long tradition of grazing cattle and sheep on the Lassen and Modoc continues through a permit system that ensures proper management of public land (photo by Ken Sandusky, Forest Service).

conditions. In addition, they discuss grazing management in riparian areas and grazing effects on water quality, soils, nutrients, vegetation, amphibians, and bumblebees. Causes and restoration of channel incision are also presented. Information on monitoring and evaluating management is provided. The authors also mention the deficiencies of published grazing research and conclude with discussion about research needs.

Chapter 9.5 (*Managing Forest Products for Community Benefit*) in the Sierra Nevada Science Synthesis provides a broad and in-depth discussion about balancing ecological goals with local community well-being (Charnley and Long 2014), and Section 5 (Society) in this *Northeastern California Plateaus Bioregion Science Synthesis* (particularly Chapter 5.4, Flores and Stone, this synthesis, *Community Engagement in the Decisionmaking Process for Public Land Management in Northeastern California*) focuses on public input to decision making and in solving conflict associated with resource management. This is critical because management occurs within the social context, no single approach will be universally satisfactory, and often societal needs supersede biophysical factors (see Stanturf et al. 2014). Charnley and Long (2014) also discuss how public land grazing supports conservation of private rangelands by ensuring that a critical baseline of resources is available to the industry.

While most of the grazing concepts and concerns included in the Sierra Nevada Science Synthesis have direct applicability to dry pine forestland and sagebrush rangeland, this synthesis builds upon that effort by focusing more explicitly on the effects of grazing within the Lassen and Modoc, particularly sagebrush rangeland. The goal of this chapter is to review the interactions of climate change, grazing, and carbon storage; the response of native plant communities to grazing; and grazing effects on invasive annual grasses. For additional discussion on grazing, see Chapter 3.1, Warren, this synthesis, *Perceptions and History of Rangeland;* Chapter 3.3, Padgett, this synthesis, *Weeds, Wheels, Fire, and Juniper: Threats to Sagebrush Steppe;* Chapter 4.2, Padgett, this synthesis, *Aquatic Ecosystems, Vernal Pools, and Other Unique Wetlands;* and Chapter 4.3, Dumroese, this synthesis, *Sagebrush Rangelands and Greater Sage-grouse in Northeastern California.*

Interactions of Climate Change, Grazing, and Carbon Storage

During the next century, changes in climate are expected to impact the rangelands of the Lassen and the Modoc. While most models show climate will warm, debate continues about how precipitation may be affected. Some models show drier conditions (Cayan et al. 2008; Polley et al. 2013) whereas others show an increase in precipitation (Allen and Luptowitz 2017; Reeves et al. 2014). As climate changes, weather will likely have more year-to-year variability and more frequent, extreme events, including drought (see Stanturf et al. 2014 and references therein), which will impact Western rangelands (IPCC 2007).

On one hand, a climate shift toward warmer temperatures with less precipitation in Northern California may push perennial systems, especially in concert with grazing, toward an annual-dominated system because they have less resistance (i.e., ability to withstand disturbance) and resilience (i.e., ability to respond to disturbance) than annual systems when faced with drought (Ruppert et al. 2015). On the other hand, a climate shift toward warmer temperatures and more precipitation would increase net primary production (additions of carbon through photosynthesis less carbon loss through respiration). Either change would affect levels of soil organic carbon (SOC), an important issue in the climate change conversation.

In general and on an area basis, rangeland soils store less carbon than other temperate, terrestrial ecosystems (Tanentzap and Coomes 2012), but given their global abundance, the sequestration potential is significant. Grazed lands have potential to sequester about 20 percent of the annual carbon dioxide released (Follett and Reed 2010). Soils comprise the largest pool of terrestrial organic carbon (Jobbágy and Jackson 2000); more carbon is typically stored belowground than in aboveground biomass where it is also less susceptible to loss by disturbance (e.g., fire). In addition, biological soil crusts also contribute to the carbon pool (Elbert et al. 2012; see Chapter 3.4, Warren, this synthesis, *Biological Soil Crusts*). This carbon promotes infiltration of precipitation and a soil's ability to hold water, cycle nutrients, and thereby improve plant growth. On rangelands, these traits provide societal and ecological benefits (e.g., clean water, clean air, erosion control, grazing income, recreation, wildlife, biodiversity, etc.) (Follett and Reed 2010). Thus, research to understand SOC and the factors that influence it, particularly on rangelands, is increasing.

Often, literature devoted to assessing global carbon sequestration potential focuses on "grazing land" that includes, for example, former cropland converted to pasture. On such sites, increases in SOC can be dramatic and skew estimates of carbon sequestration potential, further compounded by differences in other assumptions used in the models (for example, extent of area). On rangelands, vegetation type (grass versus shrub), plant growth characteristics (root mass and dynamics), grazer type (livestock versus wild ungulates), and climate can all influence carbon sequestration (Tanentzap and Coomes 2012). In their meta-analysis, McSherry and Ritch (2013) found that soil texture, precipitation, grass type, grazing intensity, study duration, and soil sampling depth explained most of the variation observed in SOC studies. Overall, they note that increasing grazing intensity on grassland sites dominated by C3 (cool-season) species decreased SOC but concluded that grazer effects on SOC are context-specific, especially because SOC accumulation is influenced by type of grazer; rangelands grazed by wildlife are predicted to annually add three times the SOC compared to those used by livestock (Tanentzap and Coomes 2012).

SOC is a function of net primary production, particularly the carbon allocated to the belowground portions of plants (Jobbágy and Jackson 2000). In their review, Piñeiro et al. (2010) noted that grazed rangelands receiving less than 16 inches (400 mm) of precipitation had larger amounts of roots than nongrazed sites, perhaps a function of changing allocation patterns stimulated by grazing or changes in plant community composition. Moreover, Evans et al. (2012) found that on an arid (10.5 inches [270 mm]) bunchgrass-dominated rangeland, SOC was similar under moderately grazed and nongrazed sites.

Grazing and Native Plant Community Response

While climo-geographic variables (e.g., soil, climate, geography, etc.) and plant adaptedness (e.g., genetics) determine plant communities across the landscape, grazing is implicated as the primary force that alters that plant community composition (Twidwell et al. 2013), and grazing can interact with other factors as well. These changes may be perceived to be negative, neutral, or positive, depending on the context. In addition to species composition, grazing may also affect the ability of a plant community to withstand disturbance (i.e., resistance) induced by other factors, such as drought, and the community's potential to return to the pre-disturbance condition (i.e., resilience).

Broadly at the global scale, when facing climate-induced drought, grazed perennial systems have slightly less resistance and resilience compared to nongrazed systems. Because the effects of drought and grazing are additive, grazed perennial systems frequently exposed to severe drought may be more likely to shift toward annual-dominated systems (Ruppert et al. 2015).

In California, native bunchgrass sites transitioned to communities of nonnative species as the amount of grazing increased (Stein et al. 2016). Once nonnative species invade, grazing of Mediterranean grasslands in California enhanced abundance of exotic forbs already present on the site but did not foster invasion by new exotic forbs, whereas the opposite occurred for exotic grasses. Conversely, the abundance of native forbs remained relatively stable but native forb diversity increased, whereas effects on native grasses were mixed, with this effect becoming more consequential on drier sites (Stahlheber and D'Antonio 2013). For Mediterranean California grasslands, winter and early spring grazing yielded the most consistent increases in native plant cover and diversity (Stahlheber and D'Antonio 2013), which could likely be maintained (especially the herbaceous component; Eldridge et al. 2013) but not enhanced (Stein et al. 2016) with low-intensity grazing.

These examples highlight the ongoing challenge for range managers and scientists. That is, understanding dynamic grazing—plant community changes to inform management decisions to achieve desired states of vegetation on the landscape. For nearly 3 decades, state-and-transition models (Laycock 1991; Westoby et al. 1989) have provided a framework for describing and analyzing a variety of vegetation dynamics. It is a useful tool for identifying stable (or current) vegetation states and subsequent thresholds driven by grazing, invasive species, fire, etc., or their combinations, where that stable vegetation state changes (see the review by Cingolani et al. 2005). The flexibility of this model allows range managers to apply it to a wide variety of field situations, including sagebrush systems (e.g., Allen-Díaz and Bartolome 1998), and be alerted to sustained processes (alone or in combination) that produce undesired states (Bestelmeyer et al. 2004). With this knowledge, focused, site-specific management tools can be developed to guide grazing activities (and foster improved communication among land managers, ranchers, and the general public) in a way as to tailor site-specific prescriptions to reach desired ecological conditions (e.g., Cagney et al. 2010); the desired ecological condition is often considered to be the existing condition before excessive disturbance occurred. When manipulating grazing to obtain a desired plant community composition is proposed, modifying grazing prescriptions may (Porensky et al. 2016) or may not (Stein et al. 2016) result in the desired condition.

Wild Horses

Management of wild horses (*Equus ferus*) and burros (*Equus asinus*) is a contentious and emotionally charged topic (fig. 3.2.2). The 1971 Wild Free-Roaming Horses and Burros Act requires management of these species to maintain their presence on Western rangelands while ensuring that wildlands are sustainably managed. This is a challenge for land managers. First, research has found that wild horse populations can increase up to 20 percent each year and the 1971 Act stipulates removing animals in order to maintain desired ecological conditions (see Garrott and Oli 2013; Natural Research Council 2013). Second, the costs to maintain captured horses increased $55 million between 2000 and 2012, accounting for 60 percent of the entire wild horse and burro management program (Garrott and Oli 2013). Third, divergent social perspectives have made finding consensus about management practices difficult (see Chapter 5.3, Flores and Haire, this synthesis, *Ecosystem Services and Public Land Management*).

On one hand, Beever and Aldridge (2011) note the effects of wild horses on ecosystems, as is the case with all herbivores, will vary depending on a variety of factors, such as elevation, wild horse population, season, and duration of use. On the other hand, the presence of unmanaged or poorly managed wild horses in sagebrush rangelands can lower the cover of grass, shrub, and overall plant cover, which can have serious implications for greater sage-grouse (*Centrocercus urophasianus;* see Chapter 4.3, Dumroese, this synthesis, *Sagebrush Rangelands and Greater Sage-grouse in Northeastern California*); increase the abundance of annual invasive grasses such as cheatgrass (*Bromus tectorum*); and compact the soil to reduce water infiltration and the activity of ants, which provide important ecological services including roles in water infiltration and carbon sequestration (Beever 2003; Beever and Aldridge 2011; Beever and Herrick 2006). Ants also are an important food source for greater sage-grouse, especially chicks (Ersch 2009). Ants play a key role in maintaining rangeland health through, for example,

Figure 3.2.2—Wild horses on the Modoc National Forest are managed through the Devil's Garden Plateau Wild Horse Territory Management Plan (Jeffers 2013) (photo by Ken Sandusky, Forest Service).

their modification of soil chemical and physical properties, seed dispersal activities, and disturbance effects on plant diversity (see Carlisle et al. 2018). Davies et al. (2014) note that impacts from wild horses can be intensive even at low populations because, unlike domestic livestock whose grazing is more intensively managed, wild horse grazing is largely unmanaged and occurs year-round. Thus, wild horse grazing can greatly increase the overall impact from nonnative grazers on the landscape (see also Beever and Brussard 2000). Moreover, this effect may be exacerbated during drought (Beever and Aldridge 2011).

In response to the contentious nature of wild horse and burro management in the West, the National Research Council (2013) released a nearly 400-page review of the science related to sustainable management. The nine-chapter report describes the issues; discusses population processes, size, growth rates, and fertility management; examines genetic diversity and population models; provides framework for establishing and adjusting appropriate management levels; and discusses necessary social considerations for managing horses and burros. Currently, wild horses on the Modoc are managed under the Garden Plateau Wild Horse Territory Management Plan (Jeffers 2013), which outlines management for the next 15 to 20 years on about 225,000 acres (91,000 ha) of the Modoc. The plan, in cooperation with the BLM describes desired future conditions, population control, improvement projects, and habitat monitoring. Still, management remains contentious.

Grazing Effects on Invasive Annual Grasses

The concept of using livestock to reduce fuel loads and subsequent fire frequency and severity, particularly on sites with invasive annual grasses, has been discussed for decades, and was the focus of a recent comprehensive review (Strand et al. 2014; see textbox 3.2.1). On one hand, the authors note that invasion by annual exotic grasses can occur with or without grazing, but high-intensity grazing encourages invasion by reducing competition from desired native vegetation. On the other hand, properly timed grazing has been shown to suppress the invaders, but results vary and are often contradictory because of site-specific characteristics and the timing and intensity of grazing (Chambers et al. 2014).

Textbox 3.2.1—Key findings of Strand et al. (2014).

- High-severity grazing (i.e., greater than 50 percent utilization), especially in the spring during initiation of bolting of perennial grasses, can suppress competition from native herbaceous plants and cause soil disturbance that can favor annual invasive grasses, including cheatgrass.
- Livestock grazing at low/moderate severity (i.e., less than 50 percent utilization) generally has little influence on the cover of perennial grasses and forbs.
- A window of opportunity may exist for targeted grazing to reduce annual grasses before perennial grasses initiate bolting or during dormancy of perennial grasses.
- Livestock grazing can reduce the standing crop of perennial and annual grasses to levels that can reduce fuel loads, fire ignition potential, and spread.
- Grazing after perennial grasses produce seed and enter a dormant state can reduce the residual biomass left on the site, thereby decreasing the fire hazard the following spring and summer.
- Economic analyses reveal that fuel treatments in sagebrush ecosystems have the highest benefit/cost ratio when the perennial grasses comprise the dominant vegetation, i.e. prior to annual grass invasion and shrub dominance.

In Northern Nevada, cattle grazing in May targeting a cheatgrass-dominated rangeland (50 to 60 percent cover) effectively reduced cheatgrass biomass (80 to 90 percent removal achieved in a 2-day period) and subsequent intensity of prescribed fire the following fall. When grazed again the next spring, cheatgrass was reduced to a level that prevented prescribed fire from carrying (Diamond et al. 2009).

In their review of Western rangelands, Strand et al. (2014) concluded that grazing cheatgrass in early spring prior to active growth of native perennial grasses was the best opportunity to reduce cheatgrass. Smith et al. (2012) point out that to be successful, initiation of grazing must be based on the growth stage of the annual grass, not simply a calendar date. Because early season grazing targets removal of green seed heads, Mosely and Roselle (2006) further suggest pulsed grazing events because grazed cheatgrass can regrow seed heads. Moreover, such winter and early spring grazing of interior California grasslands yielded increases in native plant cover and diversity (Stahlheber and D'Antonio 2013). Similarly, in Central California, short-term, high-intensity grazing by sheep conducted just prior to inflorescences reduced medusahead (*Taeniatherum caput-medusae,* syn: *Elymus caput-medusae*) cover substantially and increased forb cover and diversity (DiTomaso et al. 2008).

The presence of greater sage-grouse, however, complicates implementation of spring grazing strategies, especially those with multiple re-entries, as the best time to apply grazing to reduce annual grasses coincides with brood rearing, when maximum herbaceous cover is desired to reduce nest predation (see Chapter 4.3, Dumroese, this synthesis, *Sagebrush Rangelands and Greater Sage-grouse in Northeastern California*). Promptly removing grazers after a single, targeted, short-duration event before resumption of growth of dormant, desired vegetation may be a way to reduce annual grasses and maintain vigor (Strand et al. 2014) and/or increase cover and diversity (Stahlheber and D'Antonio 2013) of native plants beneficial to greater sage-grouse.

Davison (1996) suggests that intensive grazing be employed to reduce fire danger on sites now dominated by annual invasive grasses with little to no desired perennial vegetation. This may also improve the spring nutritional status of livestock, defer grazing on adjacent perennial rangelands, lower fire suppression costs, and protect adjacent, more pristine rangeland. Frost and Launchbaugh (2003) provide a pragmatic approach to developing and implementing grazing plans with an emphasis on weed management, and the Smith et al. (2012) management guide illustrates annual grass phenological stages to target.

Grazing Effects on Soil Properties

In their review, Drewry et al. (2008) note that grazing animals alter soil properties, which can have direct and indirect effects on subsequent plant growth. Changes in soil properties are a consequence of hoof activity. While greater treading by horses, cows, and sheep associated with increasing levels of stocking may damage plants, disrupt the soil surface, and be readily observed by land managers, changes in soil properties are less visible but may be more important. Of the soil properties investigated, bulk density is perhaps the most important. Grazing animals apply appreciable vertical force onto the soil, especially when walking (Greenwood and McKenzie 2001), and this force compresses the soil and thereby increases bulk density.

Grazing-induced increases in bulk density as a result of treading are consistently reported (Drewry et al. 2008; Evans et al. 2012; Greenwood and McKenzie 2001; Tate et al. 2004). The size of the grazing animal also influences the impact; a single cow imparted a greater change in soil bulk density than did three deer or six sheep (Cournane et al. 2011). Areas with higher incidence of treading, such as where animals congregate (i.e., water sources, shade) or travel repeatedly (i.e., along fence lines) have higher bulk densities compared to other grazed areas (Greenwood and McKenzie 2001; Tate et al. 2004). High stocking rates with long treading intervals on wet soils, and the subsequent increases in bulk density are associated with declines in pasture productivity (Drewry et al. 2008).

Increases in soil bulk density are associated with decreases in the abundance of macropores. Macropores promote soil aeration and water infiltration and decrease resistance to root growth. These factors affect plant growth, microbial and invertebrate communities, and accumulation of SOC. The impacts on soil bulk density by treading of grazing animals is greater when soils are wet (i.e., winter and spring) (fig. 3.2.3) and more pronounced on finer-textured soils (i.e., those with more silt and clay relative to sand) (Cournane et al. 2011; Drewry et al. 2008; Greenwood and McKenzie 2001). In a long-term (30-year) grazing study on a semiarid grassland with a loam soil, grazed plots had higher soil bulk density than nongrazed plots, and spring grazing (wetter soil) at moderate stocking (45 to 50 percent consumption of available forage) increased soil bulk density in the top 6 inches (15 cm) of the soil profile more than fall grazing (drier soil) (Evans et al. 2012).

Research by Tate et al. (2004) on long-term plots on the San Joaquin Experimental Range in the Sierra Nevada found that bulk densities varied by canopy cover type, with soils under an open grassland having greater bulk density than soils residing under landscapes having 30 percent tree (either pine [*Pinus*] or oak [*Quercus*]) or shrub (ceanothus [*Ceanothus*]) canopies. Eldridge et al. (2015) concluded that shrubs help moderate grazing impacts by: (1) restricting access of grazing animals to soil under their canopies, which reduces changes to bulk density; (2) improving water infiltration through lower bulk density and increased litter cover, the latter fostering decomposition and soil aggregation; and (3) protecting the biological soil crust (see Chapter 3.4, Warren, this synthesis, *Biological Soil Crusts*).

Grazing intensity also affects bulk density. Increasing the amount of residual dry matter (native annual grass) on a Sierra Nevada site to more than 980 pounds per acre (1,100 kg per ha) by reducing grazing intensity decreased bulk density about 10 percent compared to sites having less than 400 pounds per acre (450 kg ha) (Tate et al. 2004). On these same sites, characterized by coarse-textured soils, absence of livestock grazing allowed soil bulk density to return to the same level as that found on sites where grazing was excluded for more than 26 years. It is likely that recovery time of rangeland soil bulk densities will depend on soil texture, with finer-textured soils requiring more time.

Meeting Grazing Management Objectives

While much of the literature focuses on the ecological aspects of grazing, more emphasis is finally being placed on better understanding the socioecological underpinnings of grazing and their relevance, given the strong and centuries-long influence of human activity (including politics) on grazing of rangelands (Bennett et al. 2013; Thevenon et al. 2010). This is critical, as Briske et al. (2011) conclude that while purposeful rotation of livestock can achieve diverse management goals, a robust science literature shows it may not necessarily provide specific ecological endpoints. The authors contend that this is because experiments have intentionally excluded the human management component, including manager objectives, experience, and decision making. Given this shortcoming but the need to sustain the rangeland ecosystem, Follett and Reed (2010) conclude that development and implementation of necessary and effective management plans are more likely to succeed if done at more local levels with active engagement from all stakeholders. In Chapter 5.1 (Flores, this synthesis, *An Introduction to Social, Economic, and Ecological Factors in Natural Resource Management of Northeastern California Public Lands*) it is noted that decision making for ecosystems requires balance between the complexity of the ecosystem and the various ecosystem benefits provided to a broad palette of stakeholders. Successful collaborations promote dialog and afford the community opportunity to address management challenges and solutions, strengthen local livelihoods, utilize ecosystem service, and sustain the ecosystem. While Flores approaches the topic of community engagement in the decisionmaking process from a forest management perspective, the discussion and tenets are applicable to any

Figure 3.2.3—Livestock grazing can change soil properties, especially when soils are wet (photo by Ken Sandusky, Forest Service).

ecosystem, including rangelands. As Eldridge et al. (2013) conclude: ***"Ultimately, however, the prevailing land use is likely to depend on social systems and human decisions, and how society reconciles competing valuations of ecosystem services related to soil carbon, grazing and wildlife habitat."***

Regardless, meeting management objectives will involve movement of livestock across the landscape at densities that allow the ecological system to thrive. Although Bailey and Brown (2011) suggest that attentive and timely adjustments to herd size and location at the landscape scale is "more likely to be effective in maintaining or improving rangeland health than fencing..." a system of fencing is typically used to constrain livestock movement to focus grazing and avoid overconsumption of preferred plants or grazing areas at crucial times, such as during sage-grouse nesting (see Chapter 4.3, Dumroese, this synthesis, *Sagebrush Rangelands and Greater Sage-grouse in Northeastern California*) or during the critical late-summer period in riparian zones (see Long and Pope 2014), rather than periodic deferments of these areas throughout the season (Bailey and Brown 2011). Such enclosures could be used to provide short-term, concentrated grazing on invasive annual grasses (see above). While fencing may be useful, it comes with drawbacks as well. Different types of fencing can affect movement of wild ungulates differently (e.g., Gates et al. 2012; Karhu and Anderson 2006; and references therein), and even impact greater sage-grouse mortality (see the grazing impact section in Chapter 4.3, Dumroese, this synthesis, *Sagebrush Rangelands and Greater Sage-grouse in Northeastern California*).

Monitoring grazed rangelands is essential to meeting multiple-use objectives because it documents ecological changes on the rangeland that can be used to adjust management. A recent look at Bureau of Land Management allotments found that just less than two-thirds of them had been monitored. Of those not meeting standards, only a third had a full suite of monitoring data (Veblen et al. 2014). The authors noted that their independent data acquisition, along with conversations with rangeland experts, revealed that monitoring ground cover needed more emphasis as a grazing-related metric (table 3.2.1). Indeed, Fynn et al. (2017) conclude that the previous year's effects on grazing (which would be known with monitoring) can have a profound implication for current-year management, because of the lag effect among

Table 3.2.1—Results of informal conversations with Federal (n = 20) and university (n = 20) rangeland science experts on how best to prioritize monitoring of rangeland condition and livestock impacts. Experts were presented with a hypothetical monitoring scenario (table 3 in Veblen et al. 2014).

Monitoring priority	Federal (%)	University (%)
Cover	55	70
Bare ground	25	15
Gap	5	5
Production	10	10
Frequency	5	0
Density	10	10
Utilization	35	25
Cattle and/or wildlife condition	5	10
Soils	25	10
Reference areas or ecological site	30	40
Photos	30	15
Remote sensing	30	35
Identification of at-risk areas	25	15

years on rangeland recovery. They note this lag of recovery will vary more as climatic variability continues to become more extreme, and thus suggest a more conservative approach to determining stocking rates in order to maintain greater heterogeneity on the landscape. Otherwise, a mismatch between stocking levels and carried-forward forage levels can result in overstocking that overrides all other management initiatives and objectives. Tillman et al. (2006) conclude that maintaining biodiversity (another aspect of greater heterogeneity) promotes ecosystem stability. Thus, maintaining greater diversity and heterogeneity can maintain the productivity and stability of livestock populations (Fynn 2012; Hobbs et al. 2008) as well as the stability of wildlife, especially greater sage-grouse, other obligate sagebrush fauna, and mule deer. Finally, increasing plant species richness (diversity) positively increases abundance of other flora and fauna, including pollinators, while decreasing the abundance and diversity of invading plant species (Scherber et al. 2010).

Restoration

Social Aspects

Undoubtedly, restoration of rangelands involves a social component. A discussion (from a forestry perspective) about the necessity of community engagement in meeting management (including restoration) goals is included in Chapter 5.1 (Flores, this synthesis, *An Introduction to Social, Economic, and Ecological Factors in Natural Resource Management of Northeastern California Public Lands*). That discussion is readily applicable to sagebrush rangelands as well, and critical to building the relationships and trust needed to garner greater support for restoration activities. Gordon et al. (2014) add to this discussion by noting specific public views on various rangeland restoration practices based on surveys. They found that the general public was more accepting of prescribed fire, grazing, felling, and mowing treatments than those that included herbicides and chaining. These practices were also more acceptable to people who expressed greater concerns that inaction was unacceptable. Even so, with the exception of livestock grazing to reduce fine fuels, none of these restoration treatments were embraced by more than half of the respondents. Interestingly, simply providing more information about the merits of these techniques was unlikely to gain more support, rather, land managers building trust with the public is the better avenue for implementing any of these practices more widely. As part of this trust, the process used to make management decisions has a strong impact on how stakeholders view the decisions and subsequent implementation (Shinder et al. 2002), and this process requires that all parties participate in give-and-take discussion toward eventually understanding the rationale for the treatment and have input into the potential tradeoffs and outcomes associated with it.

Landscape to Local

A current emphasis of sagebrush rangeland restoration is improving habitat for greater sage-grouse. Some conservationists consider greater sage-grouse an “umbrella species” for sagebrush ecosystems. The assumption with this management philosophy is that other sagebrush-obligate species of concern (as well as other flora and fauna associated with the sagebrush biome) will simultaneously benefit when the sagebrush ecosystem is managed and/or restored for greater sage-grouse (Rowland et al. 2006). For example, 85 percent of the restoration associated with conifer removal to improve greater sage-grouse habitat across the Western United States also coincided with moderate to high levels of the sagebrush-obligate Brewer’s sparrow (*Spizella breweri*) (Donnelly et al. 2017). Similarly, in Wyoming, 50 to 90 percent of the restoration work for greater sage-grouse overlapped migration, stopover, and wintering areas of mule deer (Copeland et al. 2014). Conifer removal to restore greater sage-grouse habitat increased butterfly species richness and abundance at most sites (McIver and Macke 2014). And, these treatments were modeled to increase potential forage 37 percent and ranch income 15 percent (McClain 2012; see Chapter 4.3, Dumroese, this synthesis, *Sagebrush Rangelands and Greater Sage-grouse in Northeastern California*). Thus, considering rangeland restoration through the lens of sage-grouse restoration has merit.

Several recent research efforts have concentrated on prioritizing restoration and quantifying its effectiveness at the landscape level. These efforts have included mapping tree cover across the range of greater sage-grouse (Falkowski et al. 2017), determining frameworks for removing encroaching conifers (Reinhardt et al. 2017), and assessing effectiveness of seeding treatments after wildfire (Arkle et al. 2014).

Beginning restoration with an eye toward landscape function is prudent. Fuhlendorf et al. (2017) contend that restoration and conservation should focus on landscape processes such as fragmentation, conifer encroachment, and habitat conversion. Otherwise, reliance on local management efforts that enable populations to persist will be for naught because such activities do not create suitable habitat at a scale necessary for the species to thrive. Even so, incremental, local management efforts, if done in sufficient quantities across the landscape can achieve desired landscape-level results. For example, some recent work is demonstrating that landscape-level restoration for greater sage-grouse is improving rangeland health. In Southeast Oregon (and just north of the Modoc), removing conifers from about 20 percent of an 84,000 acre (34,000 ha) area during an 8-year period in increments of 42 to 6,200 acres (17 to 2,500 ha) resulted in improvements in annual female and nest survival (6.6 and 18.8 percent, respectively) and an estimated 25 percent increase in overall population growth compared to the nontreated control areas (Severson et al. 2017). Conifer removal is also known to increase songbird (Donnelly et al. 2017; Holmes et al. 2017) and forb (Bates et al. 2017) abundance and soil water availability (Roundy et al. 2014), and yield longer seasonal streamflow (Kormos et al. 2017).

The goal of sagebrush rangeland restoration is enhancing the resistance and resilience of the system to current and future disturbances so that ecosystem services and function continue. During the last decade appreciable research and effort has been made to link prioritization of restoration with frameworks to guide the science-based resistance and resilience. In particular, Chambers et al. (2017) describe five indicators of resistance and resilience for sagebrush rangelands, the objectives associated with those indicators, and strategies for achieving those objectives (table 3.2.2). Specific guides to science-based restoration practices have been developed to support the achievement of resistance and resilience in rangeland restoration (table 3.2.3).

Summary

The Lassen and Modoc manage extensive rangeland and grazing land and support a long tradition of livestock grazing. Changes in climate are predicted to impact these ecosystems, but uncertainty remains a challenge for land managers. While rangelands store less carbon than many other types of ecosystems, rangeland SOC is a significant contributor to global carbon sequestration. Livestock grazing affects rangelands. Notably, grazing may affect the ability of rangeland to withstand disturbance (i.e., resistance) induced by drought and other factors, and the potential of rangeland to return to its pre-disturbance condition (i.e., resilience). Grazed rangelands with a high abundance of perennial species have slightly less resistance and resilience compared to nongrazed systems. Grazing can also influence the occurrence and abundance of plants species, which has implications for managing fire frequency and intensity and providing habitat for obligate sagebrush species such as greater sage-grouse. Managing grazing intensity and the level of residual dry matter that remains on the site can ensure soil physical properties conducive to productivity and water infiltration continue. Wild horses, which are often managed less intensively than livestock, present a particularly vexing challenge to land managers, highlighting the fact that societal values and the human decisionmaking process drive management decisions and community engagement is essential in devising successful management plans. Monitoring is necessary to ensure management objectives are achieved, and to determine when restoration is required. Maintaining the ecosystem services provided by rangelands, i.e., rangeland function, may be more appropriate than attempting to restore to an historic reference condition. Recent efforts have provided science-based, practical approaches, frameworks, and guidelines for restoring sagebrush-dominated rangelands.

Table 3.2.2—Five indicators of resistance and resilience in sagebrush rangelands, restoration objectives associated with those traits, and strategies for achieving desired levels of resilience and resistance. From Chambers et al. (2017).

Indicator of resistance and resilience	Restoration objective	Strategies for achievement
Extent and connectivity of sagebrush ecosystems	Minimize fragmentation to maintain large landscape availability and connectivity for sage-grouse and other sagebrush dependent species.	Secure conservation easements to prevent conversion to tillage agriculture, housing developments, etc., and maintain existing connectivity. Develop appropriate public land use plans and policies to protect sagebrush habitat and prevent fragmentation. Manage conifer expansion to maintain connectivity among populations and facilitate seasonal movement. Suppress fires in targeted areas where altered fire regimes (due to invasive annual grasses, conifer expansion, climate change, or their interactions) are resulting in fire sizes and severities outside of the historical range of variability*, increasing landscape fragmentation, and impeding dispersal, establishment, and persistence of native plants and animals.
Functionally diverse plant communities	Maintain or restore key structural and functional groups including native perennial grasses, forbs, and shrubs and biological crusts to promote biogeochemical cycling and hydrologic and geomorphic processes, promote successional processes, and reduce invasion probabilities.	Manage grazing to maintain soil and hydrologic functioning and capacity of native perennial herbaceous species, especially perennial grasses, to effectively compete with invasive plant species. Reduce conifer expansion to prevent high-severity fires and maintain native perennial herbaceous species that can stabilize geomorphic and hydrologic processes and minimize invasions. Restore disturbed areas with functionally diverse mixtures of native perennial herbaceous species and shrubs with capacity to persist and stabilize ecosystem processes under altered disturbance regimes and in a warming environment.
Introduction and spread of nonnative invasive plant species	Decrease the risk of nonnative invasive plant species introduction, establishment, and spread to reduce competition with native perennial species and prevent transitions to undesirable alternative states.	Limit anthropogenic activities that facilitate invasion processes including surface disturbances, altered nutrient dynamics, and invasion corridors. Use early detection and rapid response for emerging invasive species of concern to prevent invasion and spread. Manage livestock grazing to promote native perennial grasses and forbs that compete effectively with invasive plants. Actively manage invasive plant infestations using integrated management approaches such as chemical treatment of invasives and seeding of native perennials.
Wildfire regimes outside of the historical range of variability	Reduce the risk of wildfires outside of the historical range of variability to prevent large-scale landscape fragmentation and/or rapid ecosystem conversion to undesirable alternative states.	Reduce fuel loads to: (1) decrease fire size and severity and maintain landscape connectivity, (2) decrease competitive suppression of native perennial grasses and forbs by woody species, and thus (3) lower the longer-term risk of dominance by invasive annual grasses and other invaders. Suppress fires in low- to moderate-resistance and resilience sagebrush-dominated areas to prevent conversion to invasive annual grass states and thus maintain ecosystem connectivity, ecological processes, and ecosystem services. Suppress fires adjacent to or within recently restored ecosystems to promote recovery and increase capacity to absorb future change. Use fuel breaks in carefully targeted locations along existing roads where they can aid fire-suppression efforts and have minimal effects on ecosystem processes.

(Continued)

Table 3.2.2—(Continued).

Indicator of resistance and resilience	Restoration objective	Strategies for achievement
Ecosystem recovery toward desired states following disturbance	Restore and maintain ecosystem processes and functional attributes following disturbance that are consistent with current and projected environmental conditions and allow ecosystems to absorb change.	Assess postdisturbance conditions and avoid seeding where sufficient native perennial herbaceous species exist to promote successional processes, stabilize hydrologic and geomorphic processes, and make conditions conducive to recruitment of sagebrush. Consider seeding or transplanting sagebrush species adapted to site conditions following large and severe wildfires that decrease recruitment probabilities to increase the rate of recovery and decrease fragmentation. In areas with depleted native perennials, use species and ecotypes for seeding and outplanting that are adapted to site conditions and to a warmer and drier climate where projections indicate long-term climate change. Avoid seeding introduced forage species that outcompete natives.

*Historical range of variability and natural range of variability are essentially the same.

Table 3.2.3—Science-based restoration guides to implement local activities toward achieving resistance and resilience on sagebrush rangelands.

Reference	Description
Pyke et al. 2015a	Concepts for understanding and applying restoration
Pyke et al. 2015b	Landscape-level restoration decisionmaking
Pyke et al. 2017	Site level restoration decisionmaking
Miller et al. 2014	Selecting the most appropriate treatments with respect to invasive annual grasses
Miller et al. 2015	Rapidly assessing post-wildfire recovery potential
Maestas and Campbell 2014	Using soil temperature and moisture regimes to predict potential ecosystem resistance and resilience
Chambers 2016	Compilation of 14 fact sheets with “how-to” descriptions for a variety of restoration activities

References

Allen, R.J.; Luptowitz, R. 2017. El Nino-like teleconnection increases California precipitation in response to warming. Nature Communications 8: 16055. https://www.nature.com/articles/ncomms16055.

Allen-Díaz, B.; Bartolome, J.W. 1998. Sagebrush–grass vegetation dynamics: comparing classical and state-transition models. Ecological Applications. 8(3): 795–804.

Arkle, R.S.; Pilliod, D.S.; Hanser, S.E., [et al.]. 2014. Quantifying restoration effectiveness using multi-scale habitat models: implications for sage-grouse in the Great Basin. Ecosphere. 5: 1–32.

Bailey, D.W.; Brown, J.R. 2011. Rotational grazing systems and livestock grazing behavior in shrub-dominated semi-arid and arid rangelands. Rangeland Ecology and Management. 64: 1–9.

Bates, J.D.; Davies, K.W.; Hulet, A.; [et al.]. 2017. Sage grouse groceries: forb response to piñon-juniper treatments. Rangeland Ecology and Management. 70(1): 106–115.

Beever, E.A. 2003. Management implications of the ecology of free-roaming horses in semi-arid ecosystems of the western United States. Wildlife Society Bulletin. 31: 887–895.

Beever, E.A., Aldridge, C.L. 2011. Influences of free-roaming equids on sagebrush ecosystems, with a focus on greater sage-grouse. In: Knick, S.T., Connelly, J.W., eds. Greater sage-grouse: ecology and conservation of a landscape species and its habitats. Studies in Avian Biology 38. Berkeley, CA: University of California Press for Cooper Ornithological Society: 273–290.

Beever, E.A.; Brussard, P.F. 2000. Examining ecological consequences of feral horse grazing using exclosures. Western North American Naturalist. 60(3): 236–254.

Beever, E.A.; Herrick, J.E. 2006. Effects of feral horses in Great Basin landscapes on soils and ants: direct and indirect mechanisms. Journal of Arid Environments. 66: 96–112.

Bennett, J.E.; Salomon, M.; Letty, B.; [et al.]. 2013. Aligning policy with the socio-ecological dynamics of rangeland commons. African Journal of Range and Forage Science. 30(1–2): iii–ix

Bestelmeyer, B.T.; Herrick, J.E.; Brown, J.R.; [et al.]. 2004. Land management in the American Southwest: a state-and-transition approach to ecosystem complexity. Environmental Management. 34(1): 38–51.

Briske, D.D.; Sayre, N.F.; Huntsinger, L.; [et al.]. 2011. Origin, persistence, and resolution of the rotational grazing debate: integrating human dimensions into rangeland research. Rangeland Ecology and Management. 64(4): 325–334.

Brown, W.S., Jr. 1945. History of the Modoc National Forest. San Francisco, CA: U.S. Department of Agriculture, Forest Service.

Cagney, C.; Bainter, E.; Budd, B.; [et al.]. 2010. Grazing influence, objective development, and management in Wyoming's greater sage-grouse habitat with emphasis on nesting and early brood rearing. B-1203. Laramie, WY: University of Wyoming, Cooperative Extension Service. 57 p

Carlisle, J.D.; Stewart, D.R.; Chalfoun, A.D. 2018. An invertebrate ecosystem engineer under the umbrella of sage-grouse conservation. Western North American Naturalist. 77(4): 450–463.

Cayan, D.R.; Maurer, E.P.; Dettinger, M.D.; [et al.]. 2008. Climate change scenarios for the California region. Climatic Change. 87: S21–S42.

Chambers, J.C., ed. 2016. Information and tools to restore and conserve Great Basin ecosystems. Great Basin Factsheet Series 2016. Reno, NV: Great Basin Fire Science Exchange. 79 p.

Chambers, J.C.; Bradley, B.A.; Brown, C.S.; [et al.]. 2014. Resilience to stress and disturbance and resistance to *Bromus tectorum* invasion in cold desert shrublands of western North America. Ecosystems. 17: 360–375.

Chambers, J.C.; Maestas, J.D.; Pyke, D.A.; [et al.]. 2017. Using resilience and resistance concepts to manage persistent threats to sagebrush ecosystems and greater sage-grouse. Rangeland Ecology and Management. 70: 149–164.

Charnley, S.; Long, J.W. 2014. Managing forest products for community benefit. In: 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: 629–662.

Cingolani, A.M.; Noy-Meir, I.; Díaz, S. 2005. Grazing effects on rangeland diversity: a synthesis of contemporary models. Ecological Applications. 15(2): 757–773.

Copeland, H.E.; Sawyer, H.; Monteith, K.L.; [et al.]. 2014. Conserving migratory mule deer through the umbrella of sage-grouse. Ecosphere. 5: 117.

Cournane, F.C.; McDowell, R.; Littlejohn, R.; [et al.]. 2011. Effects of cattle, sheep and deer grazing on soil physical quality and losses of phosphorus and suspended sediment losses in surface runoff. Agriculture, Ecosystems and Environment. 140: 264–272.

Davies, K.W.; Collins, G.; Boyd, C.S. 2014. Effects of feral free-roaming horses on semi-arid rangeland ecosystems: an example from the sagebrush steppe. Ecosphere. 5: 127.

Davison, J. 1996. Livestock grazing in wildland fuel management programs. Rangelands. 18(6): 242–245.

Diamond, J.M.; Call, C.A.; Devoe, N. 2009. Effects of targeted cattle grazing on fire behavior of cheatgrass-dominated rangeland in the northern Great Basin, USA. International Journal of Wildland Fire. 18: 944–950.

DiTomaso, J.M.; Kyser, G.B.; George, M.R.; [et al.]. 2008. Control of medusahead (*Taeniatherum caput-medusae*) using timely sheep grazing. Invasive Plant Science and Management. 1: 241–247.

Donnelly, J.P.; Tack, J.D.; Doherty, K.E.; [et al.]. 2017. Extending conifer removal and landscape protection strategies from sage-grouse to songbirds, a range-wide assessment. Rangeland Ecology and Management. 70: 95–105.

Drewry, J.J.; Cameron, K.C.; Buchan, G.D. 2008. Pasture yield and soil physical property responses to soil compaction from treading and grazing—a review. Australian Journal of Soil Research. 46: 237–256.

Elbert, W.; Weber, B.; Burrows, S.; [et al.]. 2012. Contribution of cryptogamic covers to the global cycles of carbon and nitrogen. Nature Geoscience. 5: 459–462.

Eldridge, D.J.; Beecham, G.; Grace, J.B. 2015. Do shrubs reduce the adverse effects of grazing on soil properties? Ecohydrology. 8: 1503–1513.

Eldridge, D.J.; Soliveres, S.; Bowker, M.A.; [et al.]. 2013. Grazing dampens the positive effects of shrub encroachment on ecosystem functions in a semi-arid woodland. Journal of Applied Ecology. 50: 1028–1038.

Ersch, E. 2009. Plant community characteristics on insect abundance: implications on sage-grouse brood rearing habits. Corvallis, OR: Oregon State University. 109 p. M.S. thesis.

Evans, C.R.W.; Krzic, M.; Broersma, K.; [et al.]. 2012. Long-term grazing effects on grassland soil properties in southern British Columbia. Canadian Journal of Soil Science. 92: 685–693.

Falkowski, M.J.; Evans, J.S.; Naugle, D.E.; [et al.]. 2017. Mapping tree canopy cover in support of proactive prairie grouse conservation in Western North America. Rangeland Ecology and Management 70: 15–24.

Follett, R.F.; Reed, D.A. 2010. Soil carbon sequestration in grazing lands: societal benefits and policy implications. Rangeland Ecology and Management 63: 4–15.

Frost, R.A.; Launchbaugh, K.L. 2003. Prescription grazing for rangeland weed management. Society for Range Management. 25(6): 43–47.

Fuhlendorf, S.D.; Hovick, T.J.; Elmore, R.D.; [et al.]. 2017. A hierarchical perspective to woody plant encroachment for conservation of prairie-chickens. Rangeland Ecology and Management. 70: 9–14.

Fynn, R.W.S. 2012. Functional resource heterogeneity increases livestock and rangeland productivity. Rangeland Ecology and Management. 65: 319–329.

Fynn, R.W.S.; Kirkman, K.P.; Dames, R. 2017. Optimal grazing management strategies: evaluating key concepts. African Journal of Range and Forage Science. 34(2): 87–98.

Garrott, R.A.; Oli, M.K. 2013. A critical crossroad for BLM's Wild Horse Program. Science. 341(6148): 847-848.

Gates, C.C.; Jones, P.; Suitor, M.; Jakes, A.; [et al.]. 2012. The influence of land use and fences on habitat effectiveness, movements and distribution of pronghorn in the grasslands of North America. In: Somers, M.J.; Hayward, M.W., eds. Fencing for conservation: restriction of evolutionary potential or a riposte to threatening processes? Berlin/Heidelberg: Springer Science+Business Media, LLC: 277–294. Chap. 15.

Gordon, R.; Brunson, M.W.; Shindler, B. 2014. Acceptance, acceptability, and trust for sagebrush restoration options in the Great Basin: a longitudinal perspective. Rangeland Ecology and Management. 67(5): 573–583.

Greenwood, K.L.; McKenzie, B.M. 2001. Grazing effects on soil physical properties and the consequences for pastures: a review. Australian Journal of Experimental Agriculture. 41: 1231–1250.

Hobbs, N.T.; Galvin, K.A.; Stokes, C.J.; [et al.]. 2008. Fragmentation of rangelands: implications for humans, animals and landscapes. Global Environmental Change. 18: 776–785.

Holmes, A.L.; Maestas, J.D.; Naugle, D. 2017. Bird responses to removal of western juniper in sagebrush-steppe. Rangeland Ecology and Management. 70: 87–94.

Huntsinger, L.; Forero, L.C.; Sulak, A. 2010. Transhumance and pastoralist resilience in the western United States. Pastoralism. 1(1): 9–36.

International Panel on Climate Change [IPPC]. 2007. Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment. Report of the Intergovernmental Panel on Climate Change. Solomon, S.; Qin, D.; Manning, M.; [et al.], eds. Cambridge, UK and New York: Cambridge University Press. 996 p.

Jeffers, R.G. 2013. Devil's Garden Plateau wild horse territory management plan. Alturas, CA: U.S. Department of Agriculture, Forest Service, Modoc National Forest Devil's Garden and Doublehead Ranger Districts. 28 p.

Jobbágy, E.G.; Jackson, R.B. 2000. The vertical distribution of soil organic carbon and its relation to climate and vegetation. Ecological Applications. 10: 423–436.

Karhu, R.R.; Anderson, S.H. 2006. The effect of high-tensile electric fence designs on big-game and livestock movements. Wildlife Society Bulletin. 34(2): 293–299.

Kormos, P.R.; Marks, D.; Pierson, F.B.; [et al.]. 2017. Ecosystem water availability in juniper versus sagebrush snow-dominated rangelands. Rangeland Ecology and Management. 70(1): 116–128.

Laycock, W.A. 1991. Stable states and thresholds of range condition on North American rangelands: a viewpoint. Journal of Range Management. 44: 427–433.

Lewandrowski, J.; Ingram, K. 2002. Restricting grazing on federal lands in the West to protect threatened and endangered species: ranch and livestock sector impacts. Review of Agricultural Economics. 24(1): 78–107.

Long, J.W; Pope, K.W. 2014. Wet meadows. In: Long, J.W.; Quinn-Davidson, L.N.; Skinner, C.N., eds. 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: 341–372. Chap. 6.3.

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.

Maestas, J.D.; Campbell, S.B. 2014. Mapping potential ecosystem resilience and resistance across sage-grouse range using soil temperature and moisture regimes. Fact Sheet. Sage Grouse Initiative, https://www.sagegrouseinitiative.com (16 Apr. 2020).

McClain, A. 2012. Ranch level economic impacts of western juniper (*Juniperus occidentalis*) encroachment on sagebrush steppe ecosystems in Owyhee County, Idaho. Moscow: University of Idaho, Moscow. M.S. thesis.

McIver, J.; Macke, E. 2014. Short-term butterfly response to sagebrush steppe restoration treatments. Rangeland Ecology and Management. 67(5): 539–552.

McSherry, M.E.; Ritch, M.E. 2013. Effects of grazing on grassland soil carbon: a global review. Global Change Biology. 19(5): 1347–1357.

Mosley, J.C.; Roselle, L. 2006. Targeted livestock grazing to suppress invasive annual grasses. In: Launchbaugh, K., ed. Targeted grazing: a natural approach to vegetation management and landscape enhancement. Denver, CO: American Sheep Industry Association: 68–77.

Miller, R.F.; Chambers, J.C.; Pellant, M. 2014. A field guide to selecting the most appropriate treatments in sagebrush and pinyon-juniper ecosystems in the Great Basin: evaluating resilience to disturbance and resistance to invasive annual grasses and predicting vegetation response. Gen. Tech. Rep. RMRS-GTR-322.Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 76 p.

Miller R.F.; Chambers, J.C.; Pellant, M. 2015. A field guide for rapid assessment of post-wildfire recovery potential in sagebrush and piñon-juniper ecosystems in the Great Basin: Evaluating resilience to disturbance and resistance to invasive annual grasses and predicting vegetation response. Gen. Tech. Rep. RMRS-GTR-338. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 70 p.

National Research Council. 2013. Using science to improve the BLM wild horse and burro program: a way forward. Washington, DC: The National Academies Press. https://doi.org/10.17226/13511 (9 Dec. 2019).

Norman, S.P.; Taylor, A.H. 2005. Pine forest expansion along a forest-meadow ecotone in northeastern California, USA. Forest Ecology and Management. 215(1-3): 51–68.

Piñeiro, G.; Paruelo, J.M.; Oesterheld, M.; [et al.]. 2010. Pathways of grazing effects on soil organic carbon and nitrogen. Rangeland Ecology and Management. 63(1): 109–119.

Polley, H.W.; Briske, D.D.; Morgan, J.A.; [et al.]. 2013. Climate change and North American rangelands: trends, projections, and implications. Rangeland Ecology and Management. 66(5): 493–511.

Porensky, L.M.; Mueller, K.E.; Augustine, D.J.; [et al.]. 2016. Thresholds and gradients in a semi-arid grassland: long-term grazing treatments induce slow, continuous and reversible vegetation change. Journal of Applied Ecology. 53(4): 1013–1022.

Pyke, D.A.; Chambers, J.C.; Pellant, M.; [et al.]. 2015a. Restoration handbook for sagebrush steppe ecosystems with emphasis on greater sage-grouse habitat—Part 1. Concepts for understanding and applying restoration: U.S. Geological Survey Circular 1416. Reston, VA: U.S. Department of the Interior, U.S. Geological Survey. 44 p.

Pyke, D.A.; Chambers, J.C.; Pellant, M.; [et al.]. 2017. Restoration handbook for sagebrush steppe ecosystems with emphasis on greater sage-grouse habitat—Part 3. Site level restoration decisions: U.S. Geological Survey Circular 1426. Reston, VA: U.S. Department of the Interior, U.S. Geological Survey. 62 p.

Pyke, D.A.; Knick, S.T.; Chambers, J.C.; [et al.]. 2015b. Restoration handbook for sagebrush steppe ecosystems with emphasis on greater sage-grouse habitat—Part 2. Landscape level restoration decisions: U.S. Geological Survey Circular 1418. Reston, VA: U.S. Department of the Interior, U.S. Geological Survey. 21 p.

Reeves, M.C.; Mitchell, J.E. 2011. Extent of conterminous U.S. rangelands: quantifying implications of differing agency perspectives. Rangeland Ecology and Management. 64(6): 585–595.

Reeves, M.C.; Mitchell, J.E. 2012. A synoptic review of U.S. rangelands: a technical document supporting the Forest Service 2010 RPA Assessment. Gen. Tech. Rep. RMRS-GTR-288. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 128 p.

Reeves, M.C.; Moreno, A.L.; Bagne, K.E.; [et al.]. 2014. Estimating climate change effects on net primary production of rangelands in the United States. Climatic Change. 126 (3–4): 429–442.

Reinhardt, J.R.; Naugle, D.E.; Maestas, J.D.; [et al.]. 2017. Next-generation restoration for sage-grouse: a framework for visualizing local conifer cuts within a landscape context. Ecosphere. 8(7): e01888.

Roundy, B.A.; Young, K.; Cline, N.; [et al.]. 2014. Piñon-juniper reduction increases soil water availability of the resource growth pool. Rangeland Ecology and Management. 67: 495–505.

Rowland, M.M.; Wisdom, M.J.; Suring, L.H.; [et al.]. 2006. Greater sage-grouse as an umbrella species for sagebrush-associated vertebrates. Biological Conservation. 129: 323–335.

Ruppert, J.C.; Harmoney, K.; Henkin, Z.; [et al.]. 2015. Quantifying drylands' drought resistance and recovery: the importance of drought intensity, dominant life history and grazing regime. Global Change Biology. 21(3): 1258–1270.

Scherber, C.; Eisenhauer, N.; Tscharntke, T. 2010. Bottom-up effects of plant diversity on multitrophic interactions in a biodiversity experiment. Nature. 468: 553–556.

Severson, J.P.; Hagen, C.A.; Tack, J.D.; [et al.]. 2017. Better living through conifer removal: a demographic analysis of sage-grouse vital rates. PLoS ONE. 12(3): e0174347.

Shindler, B.A.; Brunson, M.W.; Stankey, G.H. 2002. Social acceptability of forest conditions and management practices: a problem analysis. Gen. Tech. Rep. PNW-GTR-537. Portland, OR, USA: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 73 p.

Smith, B.; Sheley, R.; Svejcar, T.J. 2012. Grazing invasive annual grasses: the green and brown guide. Burns, OR: U.S. Department of Agriculture, Agricultural Research Service. 44 p.

Stahlheber, K.A.; D'Antonio, C.M. 2013. Using livestock to manage plant composition: a meta-analysis of grazing in California Mediterranean grasslands. Biological Conservation. 157: 300–308.

Society for Range Management. 1998. A glossary of terms used in range management. Denver, CO: Society for Range Management. 20 p.

Stanturf, J.A.; Palik, B.J., Dumroese, R.K. 2014. Contemporary forest restoration: a review emphasizing function. Forest Ecology and Management. 331: 292–323.

Stein, C.; Harpole, W.S.; Suding, K.N. 2016. Transitions and invasion along a grazing gradient in experimental California grasslands. Ecology. 97(9): 2319–2330.

Stewart, W.C. 1996. Economic assessment of the ecosystem. In: Sierra Nevada ecosystem project: final report to Congress. Vol. III: Assessments, commissioned reports, and background information. Report No. 38. Davis, CA: Centers for Water and Wildland Resources, University of California–Davis: 973–1064. Chap. 23.

Strand, E.K.; Launchbaugh, K.L.; Limb, R.F.; [et al.]. 2014. Livestock grazing effects on fuel loads for wildland fire in sagebrush dominated ecosystems. Journal of Rangeland Applications. 1: 35–57.

Tanentzap, J.A.; Coomes, D.A. 2012. Carbon storage in terrestrial ecosystems: do browsing and grazing herbivores matter? Biological Reviews. 87(1): 72–94.

Tate, K.W.; Dudley, D.M.; McDougald, N.K.; [et al.]. 2004. Effect of canopy and grazing on soil bulk density. Journal of Range Management. 57: 411–417.

Thevenon, F.; Williamson, D.; Bard, E.; [et al.]. 2010. Combining charcoal and elemental black carbon analysis in sedimentary archives: implications for past fire regimes, the pyrogenic carbon cycle, and the human-climate interactions. Global and Planetary Change. 72: 381–389.

Tillman, D.; Reich, P.B.; Knops, J.M.H. 2006. Biodiversity and ecosystem stability in a decade-long grassland experiment. Nature. 441: 629–632.

Twidwell, D.; Allred, B.W.; Fuhlendorf, S.D. 2013. National-scale assessment of ecological content in the world's largest land management framework. Ecosphere 4: 94.

U.S. Department of Agriculture, Forest Service [USDA]. 1991. Affected environment. In: Modoc National Forest—final environmental impact statement, land and resource management plan. Alturas, CA: Pacific Southwest Region, Modoc National Forest: 3–19. Chap. 3.

U.S. Department of Agriculture, Forest Service [USDA]. 1992. Summary of the analysis of the management situation. In: Land and resource management plan—Lassen National Forest. Lassen National Forest, CA: 3–31. Chap. 3.

U.S. Department of Agriculture, Forest Service [USDA]. 2012. Future of America's forest and rangelands: Forest Service 2010 Resources Planning Act assessment. Gen. Tech. Rep. WO-87. Washington, DC. 198 p.

U.S. Department of Agriculture, Forest Service [USDA]. 2015. USDA Forest Service strategic plan: FY 2015–2020. FS-1045. Washington, DC: U.S. Department of Agriculture, Forest Service. 53 p.

U.S. Department of Agriculture, Forest Service [USDA]. 2016. Grazing statistical summary FY 2015. Washington, DC: U.S. Department of Agriculture, Forest Service, Range Management. 104 p.

U.S. Department of Agriculture, Forest Service; U.S. Department of the Interior, Bureau of Land Management [USDA and USDOI]. 2007. Existing condition for livestock grazing. In: Sage steppe ecosystem restoration strategy draft environmental impact study. R5-MB-151. Alturas, CA: Modoc National Forest and Alturas Field Office. p 72. Sec. 3.3.2.

U.S. Department of the Interior [USDOI]. 2013. Forage and livestock grazing. U.S. Department of the Interior economic report FY 2012. Chap. 8. https://www.doi.gov/sites/doi.gov/files/migrated/ppa/upload/Chapter-8-FY2012-Econ-Report.pdf (12 Dec. 2019).

Veblen, K.E.; Pyke, D.A.; Aldridge, C.L.; [et al.]. 2014. Monitoring of livestock grazing effects on Bureau of Land Management land. Rangeland Ecology and Management 67(1): 68–77.

Westoby, M.; Walker, B.; Noy-Meir, I. 1989. Opportunistic management for rangelands not at equilibrium. Journal of Range Management. 42: 266–274.

Chapter 3.3. Weeds, Wheels, Fire, and Juniper: Threats to Sagebrush Steppe

Pamela E. Padgett[1]

Introduction

The Great Basin can be defined floristically by plant communities dominated by species of sagebrush (*Artemisia*) and saltbush (*Atriplex*) in its southern portions and in its northern portions by sagebrush steppe and woodlands dominated by juniper (*Juniperus*). By this definition, nearly 7.4 million acres (3 million ha) of Great Basin sagebrush steppe exists in the coterminous United States. It can also be defined hydrologically as the area in the Western United States that is internally drained; in other words, with a few exceptions, precipitation does not ultimately flow to the oceans, but remains in the basin (USGS 2016). The hydrologic definition is somewhat smaller in area, but important for restoration purposes (Svejcar et al. 2017). Studies clearly show that the sagebrush steppe has been in a continued state of change for many years. Portions of the Lassen and Modoc National Forests (hereafter the Lassen and the Modoc) occur in the northern portion of the Great Basin, which contains the unique Modoc Plateau subregion.

Geologic changes since the Pleistocene (about 11,700 years before present) have led to a drying-out of the area from an area of extensive wetlands and marshes to the semi-desert it is today. Beginning in the 1850s, human perturbations had significant impacts of plant community structure. But even before the gold rush of the 1850s, indigenous peoples manipulated the landscape through fire to increase food supplies and thwart enemies (McAdoo et al. 2013). The Modoc Plateau was not a particularly rich source of gold (https://www.fs.usda.gov/detail/modoc/learning/history-culture/?cid=stelprdb5310687), although the mountains to the west and south were quite productive. With dwindling forest resources near active mines, even distant mines had profound effects on woodland resources in the Great Basin. Wood was needed to fuel the mills (heating furnaces and creating charcoal) and to provide timbers to build and support the mining structures. Woodcutters were traveling more than 50 miles (80 km) to acquire the necessary trees (Morris and Rowe 2014). It was the demand for food and fresh meat to feed the booming mining towns that really opened up the sagebrush steppe for settlement (Svejcar 2015). In 1862 when the Homestead Act was signed, 160 acres (65 ha) was given to any man who could prove after 5 years that he had "improved" the land. Improvements required proof of cultivation and construction of a dwelling. Because a sustainable cattle and sheep operation was not feasible on 160 acres, the use of public, unpatented land, was extensive and on a first come, first feed basis (Morris and Rowe 2014). In 1909, the Enlarged Homestead Act increased the acres allotted to 320 acres (129 ha) and in 1916, to 640 acres (259 ha), in part as recognition that 160 acres was insufficient for livestock operations (Svejcar 2015). Although some knew, and argued, that even 640 acres was insufficient for a profitable livestock operation, and ranchers would still need open rangeland to graze their herds. One of the requirements of the 1909 Act was an increase in acres cultivated; 20 acres had to be under cultivation by the second year, and 40 acres (16 ha) from the third until the fifth and final year of the contract. This ushered in the establishment of dryland wheat cultivation, a project that the U.S. Department of Agriculture had been working on for some time (Gates 1968). And many believe it was the introduction of dryland wheat that brought cheatgrass (*Bromus tectorum*) to the Great Basin. Homesteading also increased pressure on what little woodlands were left for construction of dwellings and fences (Morris and Rowe 2014).

Homesteading and livestock ranching went through a series of booms and busts, harsh winters and unrelenting droughts in the late 1880s. By the 1890s, ranchers were rethinking 100-percent dependence on open range and began planning for cultivated hay to be used as winter feed, further expanding tillage in the Great Basin. Morris and Rowe (2014) argue that the disturbances caused by cropping exceed those caused by livestock. Management on unclaimed Federal land did not happen until 1934, when the Taylor Grazing Act was enacted. Until then, it was first come, first serve on the public lands and many acres near homesteads became clear examples of "the

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tragedy of the commons." It has been argued that what we see today on the sagebrush steppe is more a relic of the early part of the last century and less the effects of today's management. In any event, past events leave us with today's challenges in returning the sagebrush steppe to its unique ecological function.

The vast acreage of sagebrush steppe occupying the Lassen and Modoc is one of the key features that sets these national forests apart from the other national forests occupying the Sierra and Cascade ecoregions. By and large, the greatest use of these lands for human benefit is in livestock grazing. Grazing has changed natural processes and functions of the sagebrush steppe, creating both intended and unintended consequences. Some of these consequences are impacting grazing use itself. The Lassen and the Modoc completed extensive literature and resource reviews in 2010 when they each developed a Travel Management Plan (USDA 2010a, b). In addition, the Modoc's Environmental Impact Statement (EIS) for management of the sagebrush steppe ecosystem was finalized in April 2008 (USDA et al. 2008). And parts of both forests are covered by the *Science Synthesis to Support Socioecological Resilience in the Sierra Nevada and Southern Cascade Range* (hereafter, Sierra Nevada Science Synthesis) (Long et al. 2014) and the *Synthesis of Science to Inform Land Management within the Northwest Forest Plan Area* (hereafter Northwest Forest Plan Science Synthesis) (Spies et al. 2018). Because neither of these syntheses addresses the sagebrush steppe and during the last few decades ecosystem changes that reduce biodiversity and habitat suitability for several species have become a serious management problem, this chapter focuses on primary threats to the sagebrush steppe:

- Invasive weeds and loss of native grasses, forbs, and shrubs
- Surface disturbances from vehicle use
- Fire and changes to fire behavior
- Invasion by conifers.

Invasive Plant Species

Although invasive species may contribute to overall species richness in the short term, in the long run, they often cause significant decline, or even local extinction, of native plants through competition for nutrients, light, and water (Dukes and Mooney 2004) (fig. 3.3.1), as well as changes in ecosystem structure and function that can modify habitat suitability for many organisms. For example, when perennial pepperweed (*Lepidium latifolium*) invades riparian areas, it out-competes willows and cottonwood seedlings. Without these native trees, birds lose nesting sites, insects lose natural predators, and many carnivores lose a food source (Young et al. 1995).

Invasive plant species reproduce and spread rapidly. However, it often takes a disturbance event such as fire, extensive vehicle and foot traffic (including firefighting equipment), flooding, or excessive use by animals (native or wild) for invasive exotic plant species to gain a foothold. Nevertheless, because a nonnative species cannot expand its range unless it is already present on the site, early detection and rapid response to movement and introduction of seeds, rooting stems and roots pieces, or other propagules is the most important step in reducing the spread of noxious weeds (USDA 2013).

This chapter has a focus on "weeds," but the definitions for weeds used in the literature can be conflicting and confusing, and the definitions of some terms even overlap. To simplify the discussion, the term "invasive plant" is used in this chapter as defined by Presidential Executive Order 13112: "Invasive species" means an alien species whose introduction does or is likely to cause economic or environmental harm or harm to human health a non-native species (before European settlement) within the ecosystem considered and whose introduction causes or is likely to cause economic or environmental harm (Federal Register 1999; USDA NRCS n.d.).

Numerous invasive plants, like perennial pepperweed, cheatgrass, medusahead (*Taeniatherum caput-medusae,* syn: *Elymus caput-medusae*), Dyer's woad (*Isatis tinctoria*), and various nonnative thistles have displaced native plants and altered local plant communities. Northeastern California has the highest number of species listed by the California Department of Food and Agriculture (CDFA) as noxious weeds in the State. Many weeds come into California from the Great Basin, so management strategies need to consider the regional landscape. Preventing the spread of invasive species through education and early detection are important to maintaining healthy ecosystems. Many of the conservation actions described below address prevention, early detection, and rapid response to new invasive plants to prevent them from becoming widespread. Distribution maps and summary reports for invasive plants, as well as regional strategic plans for prioritized invasive plant species, can be found on the CalWeedMapper website

Figure 3.3.1—Invasive musk thistles (*Carduus nutans*) are pretty but can quickly overrun native plant communities (Leslie J. Mehrhoff, University of Connecticut, Bugwood.org and inset photo by Joseph M. DiTomaso, Bugwood.org).

(https://calweedmapper.cal-ipc.org). Some of the invasive species affecting the province are discussed below (California Department of Fish and Wildlife 2015).

Lassen Invasive Plants

Invasive plants such as cheatgrass and mullein (*Verbascum thapsus*) are not usually tracked on the Lassen, and inventories of species such as medusahead and yellow starthistle (*Centaurea solstitialis*) are known to be incomplete (USDA 2010a). Aside from these four plant species, the Lassen internal invasive plant inventory, which serves to hone in on the most troublesome invasive plants referred to as "noxious," comprises the best available information on invasive plant distributions. According to the 2010 Travel Management Plan, the inventory is updated annually as new occurrences are found and infestations are mapped or remapped using Global Positioning System (GPS) technologies. Targeted invasive plant surveys are conducted annually in conjunction with sensitive plant surveys. They are also identified and recorded during project work. The total area infested by invasive plants on the Lassen was estimated at more than 7,000 acres (2,833 ha) in 2010, though the actual figure is likely considerably higher. The 2010 Travel Management Plan (USDA 2010a) analysis highlighted the strong association between invasive plant infestations and the current network of roads and routes open to motorized vehicle travel.

Modoc Invasive Plants

Seventeen invasive species were considered in the Travel Management Plant (USDA 2010b) analysis (table 3.3.1), but all invasive plant species identified on the forest are of concern with regard to their potential to spread and threaten native ecosystems. The Modoc, however, has prioritized invasive plant infestations for tracking based upon the aggressiveness of the species, the degree of regional concern, and feasibility of control. From the Travel Management Plan:

> *While some species listed in statewide inventories are not identified as a high priority for control efforts and are not specifically addressed in this analysis (i.e., cheatgrass, bull thistle, Russian thistle, medusahead), it remains a priority to prevent the further spread of these species via management activities. However, control of all known infestations of these lower-priority*

> *species is not currently feasible and they are likely to persist throughout the life of this project. A weed occurrence refers to a relatively discreet group of individuals, separated from the next nearest group of the same species by at least ¼ mile. Many of the weed occurrences are immediately adjacent to existing travel routes, due to the disturbed habitat available along the road edges, and the vehicles acting as vectors for weed seeds or other propagules.* (USDA 2010b, p. 200)

While the Lassen and Modoc Travel Management Plans have some overlap in the invasive species they discuss, the Modoc plan lists eight species with a CDFA rating of "A." That rating means those species are of critical concern, subject to quarantine, eradication efforts by the State of California, and in some cases limited entry by the public into infested areas.

Weeds of Greatest Concern

Cheatgrass is perhaps the most serious invasive plant species in terms of habitat degradation for all of the Great Basin bioregion. It has infested more than 100 million acres (404,686 ha) in the Western United States (Mosley et al. 1999). By 1936, "cheatgrass lands" had become a genuine vegetation-type descriptor. Cheatgrass is highly adaptable. The typical germination pattern is a flush of seedlings in the early winter, which enables the plants to build strong root system before going semi-dormant (Young et al. 1969), however it will continue to germinate throughout the spring and summer under favorable conditions. Cheatgrass can germinate and grow under colder temperatures (Aguirre and Johnson 1991), grow faster (Concilio et al. 2015), and extract nutrients more quickly from the soil compared to native Great Basin grasses (Leffler et al. 2011; Monaco et al. 2003). At the

Table 3.3.1—Modoc National Forest noxious weed inventory. California Department of Food and Agriculture (CDFA) ratings are based on the economic threats to crops and ecosystems. An "A" rating is a serious threat requiring rapid quarantine (where appropriate) and eradication efforts. "B" and "C" ratings are systematically less serious, but still invasive plants that are capable of ecosystem harm. The California Invasive Plant Council (Cal-IPC) uses a similar system to evaluate nonnative invasive plants, but places more emphasis on natural ecosystems (table 3-69 from USDA 2010a).

Species	Common name	CDFA rating	Cal-IPC rating	Number of occurrences	Gross acres
Cardaria chalapensis	Lens-podded whitetop	B	moderate	4	9.0
Cardaria draba	Heart-podded hoarycress	B	moderate	1	0.4
Cardaria pubescens	Hairy whitetop	B	limited	2	0.2
Carduus acanthoides	Plumeless thistle	A	limited	1	0.1
Carduus nutans	Musk thistle	A	moderate	12	6.9
Centaurea diffusa	Diffuse knapweed	A	moderate	12	10.6
Centaurea stoebe ssp. *micranthos*	Spotted knapweed	A	high	13	5.1
Centaurea solstitialis	Yellow starthistle	C	high	10	2.3
Centaurea virgata ssp. *squarrosa*	Squarrose knapweed	A	moderate	5	0.2
Cirsium arvense	Canada thistle	B	moderate	34	11.9
Crupina vulgaris	Common crupina	A	limited	1	745.2
Hypericum perforatum	Klamathweed	C	moderate	8	8.8
Isatis tinctoria	Dyer's woad	B	moderate	62	6,069.9
Lepidium latifolium	Tall whitetop	B	high	1	0.1
Linaria dalmatica ssp. *dalmatica*	Dalmatian toadflax	A	moderate	12	974.7
Onopordum acanthium	Scotch thistle	A	high	333	86.5
Salvia aethiopis	Mediterranean sage	B	limited	27	11.6
Total				**539**	**7,941.2**

end of the life cycle, the dried foliage stays attached to the roots, flattening to the soil surface and creating a thatch layer that serves as both a protective mulch for the next year's crop and a barrier to germination of dicot seedlings (Stewart and Hull 1949). With time, the buildup of organic matter changes the characteristic of the soils, decreasing the edaphic suitability for native species, while favoring the spread of annual grasslands (Blank and Morgan 2012; Rimer and Evans 2006).

Once established, cheatgrass reduces rangeland forage quality for livestock (Evans and Young 1984; Hafferkamp et al. 2001). Cheatgrass can also increase economic losses when animals are injured by the spikey awns stuck in ears and eyes or have their fleece contaminated (Mealor et al. 2013) (fig. 3.3.2). The forage and habitat quality are equally poor for wildlife (Aldridge et al. 2008; Knapp 1996; Ostoja and Schupp 2009), but until recently, little incentive or funding was available for improving habitat diversity absent production agriculture. Recently, however, the relationship between cheatgrass infestations and habitat loss for greater sage-grouse (*Centrocercus urophasianus*) has increased the urgency for cheatgrass control in ecosystems (Johnson et al. 2011). Multi-State efforts are underway to improve habitat conditions for the greater sage-grouse to prevent its listing under the Endangered Species Act (USDOI 2015b; USFWS 2015; see Chapter 3.4, Dumroese, in this synthesis, *Sagebrush Rangelands and Greater Sage-grouse in Northeastern California*).

Figure 3.3.2—(A) Cheatgrass flower head (photo by Matt Lavin, University of Montana, used with permission). (B) The awns mature into prickly barbs that embed in fur and clothing, increasing distribution (photo by Pamela E. Padgett, Forest Service).

Cheatgrass originates from the Mediterranean region in Europe. Like most invasive weeds, once introduced, it had few natural pests or pathogens. Fires only enhance the competitive character of cheatgrass. Timed grazing has shown promise in reducing cheatgrass (Diamond et al. 2009). But grazing is only successful in the early season, when the blades are palatable, and it requires careful planning and control as overgrazing will contribute to cheatgrass spread (Mealor et al. 2013; Tzankova and Concilio 2015; see Chapter 3.2, Dumroese, in this synthesis, *Rangeland in Northeastern California*). In small areas, hand removal prior to seed production is often successful in greatly reducing the population, but the approach is not appropriate to large open wildlands (Concilio 2013). Several herbicide treatments specific for annual grasses and broad spectrum are available. Imazapic, a grass-specific pre- and post-emergent herbicide most commonly used in Idaho, Wyoming, and Nevada for cheatgrass control is not available in California. Other grass-specific herbicides are expensive, controversial, and difficult to use across large landscapes (Tzankova and Concilio 2015). Although a broader array of herbicides targeting broadleaf weeds that spare monocots is available, unintended consequences of their use, particularly in shrublands, can be dire. One study looked at the longevity of the effects of the broadleaf herbicide picloram in an effort to restore native grasslands. The aim was to test the hypothesis that temporary reductions of weedy forbs would allow native grasses to gain and foothold and out compete subsequent infestations. The results showed that within 4 years and certainly by 16 years, the returning weeds were well established and the native grasses had reduced abundance (Rinella et al. 2009).

Cheatgrass's Achilles heel of low seed durability makes the elimination of mature plants before they produce seeds a potential option to greatly reduce populations. Two types of biocides seem to hold some promise for long-term control: a fungal pathogen, *Pyrenophora semeniperda* developed by Dr. Susan Meyer (U.S. Department of Agriculture, Forest

Service, Rocky Mountain Research Station; Meyer et al. 2007), and a bacterial pathogen *Pseudomonas flourescens* isolated and developed by Dr. Ann Kennedy (Ibekwe et al. 2010). Both inhibit seedling germination and have been shown to be effective in greenhouse studies and field studies. *Pseudomonas flourescens* is a widely dispersed bacteria found in nearly every soil type. *Pyrenophora semeniperda* is also widely dispersed, but oddly not found in *B. tectorum's* native habitat.

Yellow starthistle is a CDFA C-rated pest, which means that it is a medium to low threat to agriculture or the ecosystems. A member of the Asteraceae family, this winter annual establishes during fall and winter and flowers the next year. This species reproduces primarily by seeds, persists at high population densities, and is associated with disturbance such as grazing, fire, and road construction. The seeds of yellow starthistle may persist in the soil for up to 10 years (Zouhar 2002). This species is widespread on the western, low-elevation portions of the Lassen. Small infestations are usually treated by hand pulling. No economically effective chemical treatments for larger infestations are available that do not harm desirable plant species.

Yellow starthistle occurs throughout the arid and semiarid regions of the West. It is highly adaptable and can rapidly take over landscapes, particularly after disturbances, creating dense stands and decreasing biodiversity. It is unpalatable, even toxic, to livestock and provides very poor habitat for wildlife. *The Field Guide for Managing Yellow Starthistle in the Southwest* (USDA 2014a) has compiled the most recent integrated pest management approaches for controlling this weed. Early detection and eradication of small patches is the best approach, as highly infested areas may take 3 or more years to clear. Because starthistle reproduces only by seeds. Control methods should be focused on removing the plants before they set seeds. Prescribed fire and grazing have been evaluated as possible control mechanisms. Neither is particularly effective alone. Often the plants do not produce enough dry fuel to carry a fire of the intensity needed to kill seeds in the spring and early summer, before the plants start blooming. On one hand, grazing by horses and cattle is not particularly effective because once the plants start flowering, the spines on the flower heads become unpalatable, and even dangerous; thus, grazing using most livestock must be carefully managed. Goats, on the other hand, have been effective if managed for short, intensive grazing with frequent moving. Both chemical and biological control agents are available (refer to the most recent California registered pesticide website: www.cdpr.ca.gov/docs/label/labelque.htm). The biological controls work by reducing seed production (Wilson et al. 2003). Because starthistle can produce millions of seeds per acre, however, the effectiveness of control can take many years in large infestation. Herbicides are most effective when applied at the early growth stages. The most effective strategies combine herbicides with grazing and/or fire.

Pepperweed is a CDFA B-rated weed, which means that it is a medium to high threat to agriculture or ecosystems, but is fairly limited in distribution. It is a perennial forb in the family Brassicaceae. This species is a high priority for control on the Lassen, as it has the potential to severely degrade riparian sites by crowding out native vegetation. Though most of these occurrences consist of fewer than 25 stems, perennial pepperweed has been difficult to eradicate due to this species' ability to form new shoots from buds on lateral, creeping roots (DiTomaso et al. 2013).

Perennial pepperweed is a forb that usually reproduces vegetatively rather than by seeds. Among its adaptations is "salt pumping"—the ability to absorb ions (particularly sodium and magnesium) from deep in the soil profile and release them at the soil surface, effectively creating a saline soil layer on the surface, thus reducing the ability of native plants to germinate and repopulate (Renz and Blank 2004). Pepperweed occurs in every county in California and every Western State, even extending into New England. It tends to be a more serious pest in riparian and seasonally wet areas. Individuals can grow to be 6 feet (2 meters) tall. Like all invasive plants, infestation of pepperweed crowds out native plant species and reduces fauna biodiversity. Several guidebooks and fact sheets for control have been published in the last few years, including the *Forest Service Field Guide for Managing Perennial Pepperweed in the Southwest* (USDA 2014b). Most mechanical methods, such as mowing and discing, are not recommended because pepperweed has an extensive root system that allows mowed plants to quickly resprout, and root segments as small as 1 inch generated by plowing can survive long periods of desiccation and quickly grow into new plants when moisture becomes available. Fire is also known to increase pepperweed infestations, although both fire and mowing can be used to remove top growth prior to chemical treatments. Grazing has been tested as a control mechanism with some success, particularly the use of sheep and goats. And in areas where chemical control may be undesirable, such as vernal pools (Vollmar

Consulting AECOM 2009). However, the plants become unpalatable once flower heads are formed. Some evidence suggests this plant is toxic to horses when consumed in large quantities (Young et al. 1995). Interestingly, dodder (*Cuscuta subinclusa*), preferentially colonizes pepperweed and reduces seed weight and germination of pepperweed by 27 and 42 percent, respectively (Benner and Parker 2004). Dodder itself can, however, become a serious invasive pest.

Oxeye daisy (*Leucanthemum vulgare*) is not rated by CDFA, but is inventoried on the Lassen. This species, in the family Asteraceae, may reproduce vegetatively from shoots that develop from buds on lateral roots in addition to seeds. It was introduced as an ornamental and is still sold commercially in seed packets (Cal-IPC 2018). It is well adapted to many environments from open fields to woodland and can be a significant problem in riparian corridors. Oxeye daisy is a prolific seeder, and seeds remain viable for many years in the soil. Available information indicates that priority for treatment is given to new, small infestations that may be successfully decreased or eradicated with repeated manual treatments.

Medusahead is a CDFA C-rated noxious grass in the family *Poaceae*. This species is highly competitive and may form monotypic stands where it occurs. This grass is unpalatable to livestock and produces a prolific amount of seeds annually. Successful suppression usually involves some combination of herbicide, fire, and reseeding with other grass species (Archer 2001). As with yellow starthistle, inventories within this area are incomplete, and the more than 2,000 acres (809 ha) that this species is known to occupy within the project area is likely a significant underestimate. When infestation cannot be effectively treated with manual control strategies, no economically effective chemical treatments are available for larger infestations that do not harm desirable plant species.

Medusahead has a similar life history to cheatgrass. It, however, is a more recent introduction. It germinates in the fall and winter, growing strong root systems before the shoots expand in the spring. Like cheatgrass it is an annual; once the seeds ripen, the shoots die, leaving a dense thatch that can choke out germination of native plants and provide a fuel layer for fire (fig. 3.3.3). Mowing, discing, grazing, and prescribed fire can be effective means of control. As with most annual invasive plants, conducting control measures before the plants set seeds and shatter is critical to successful eradication. Unlike cheatgrass, no biological controls have been found, although efforts to find and develop them continue.

Figure 3.3.3—Medusahead litter with emerging seedlings (photo by Thomas Getts, University of California Agriculture and Natural Resources, used with permission).

Scotch thistle (*Onopordum acanthium*) is a CDFA-A-rated weed. Originally introduced in the late 1900s as a horticultural ornamental, it is still grown in gardens today. Scotch thistle is typically a biannual, but individuals can persist for several years. During the first year, plants present as low-growing rosettes. In the second year, the stems grow to their full height and the plants flower, creating seeds for expansion of populations. Severe infestations can form tall, dense, impenetrable stands, especially in fertile soils. Like most weeds, it gets a foothold in disturbed areas, but can rapidly spread into natural areas, especially into particularly fertile soils. Chemical control of this thistle is difficult because of its ability to germinate nearly year-round, requiring multiple herbicide applications. Herbicides are effective on first-year seedlings, but once the stem begins to elongate, chemical control loses its effectiveness. Research has demonstrated certain requirements for Scotch thistle seed germination, providing some possible management strategies that may reduce expansion of populations. The achene coat must be leached prior to germination due to a water-soluble inhibitor on the seed surface. Seed germination is much higher when seed/soil contact is maximized, and seeds require light to germinate.

Knapweeds (*Centaurea* species) are CDFA-A rated. Diffuse (*C. diffusa*), sparrose (*C. virgata*), and spotted (*C. stoebe* ssp. *micranthos*) are the most common. Drought and fire resistant, knapweeds produce allelopathic effects and are highly competitive with other plants, often displacing desired vegetation. Knapweeds are now found in all United States and much of Canada. *Centaurea* is a large genus comprised of about 500 species, none native to California. Most species are highly prolific in disturbed areas and once infested, can be very difficult to eradicate. Like nearly all invasive plants, eradication of knapweeds requires time and a carefully planned multifaceted management approach (DiTomaso et al. 2013). Spotted knapweed is particularly invasive, as it reproduces not only by seed but also vegetatively from lateral roots. New plants can develop at about 1.25-inch (3-cm) intervals along the lateral roots, expanding populations peripherally. Diffuse knapweeds are often spread by a "tumbleweed strategy." At maturity, the stems separate from roots and the entire plant is tumbled around by the wind, dispersing seeds over potentially long distances. Once established, eradication of all knapweeds is challenging. Most species have stout taproots that readily resprout unless entirely removed. In ecologically sensitive areas, eradication by hand removal is possible, but may take two or three treatments per year for multiple years. Control and management require an integrated approach. Herbicides can reduce seedling numbers, but knapweeds are prolific seeds, and germination can occur throughout the year when conditions are favorable. Grazing may be helpful in the early season, but soil disturbance from hooves can provide ideal seedbeds. Fire generally is not very effective unless the fire intensity is high and heat penetrates well into the soil profile to kill seeds and roots. Several herbicides are effective in controlling knapweeds. Application timing is critical to the success; most are effective during the early stages of growth. Montana has introduced 13 insect species for control of spotted and diffuse knapweeds with good results (Duncan et al. 2017), and several projects are underway in California (CDFA 2018).

Vehicles

Vehicles can be disturbance sources, damaging native plants and allowing invasive plant populations to expand (fig. 3.3.4). They can also be vectors for invasive plants, serving as a transport mechanism for moving invasive species seeds and other propagules into pristine areas, resulting in new infestations. Vehicles are generally interpreted as motorized personal conveyances, but bicycles, construction equipment, and even aircraft can unknowingly carry noxious hitchhikers into the back country. There is, unfortunately, very little experimental or scientific data supporting the somewhat intuitive notion that vehicles are vectors for seed dispersal. Observations, however, frequently show that weedy infestations are densest near trails and roads (Usher 1988). One small study counted the number of seeds found on visitors (mostly shoes) to a park in South Africa. The 68 participants fell into three categories: hikers, cyclists, and dog walkers. Dog walkers (but not the dogs) were found to carry the most nonnative seeds, followed by hikers, the shoes of cyclists, then dogs. None of the bike tires carried seeds (Bouchard et al. 2015). A modeling study testing relative importance of potential seed vectors was conducted by Brancatelli and Zalba (2018). The study used several variables including the physical characteristics of seeds that effect transport, potential volume of seeds any one vector could transport, and control and impact of the particular species. Cargo carried into a protected site was found to have the highest potential for introduction of alien plant species, followed by vehicles.

Figure 3.3.4—Cheatgrass has been observed invading along roads in arid shrubland environments. It is often the first plant to germinate in early spring, which gives it a competitive advantage over later-germinating native plant seedlings (photo by Pamela E. Padgett, Forest Service).

The Travel Management Plans completed by the Lassen and the Modoc in 2010 did a thorough job of analyzing the issue of standard vehicle travel and damage to sagebrush steppe. Once the preferred alternatives were adopted, off-road traffic, and even use of graveled roads, is generally prohibited by the Travel Management Plans, although exceptions are made for ranchers and hunters whose legislatively permitted activities require access to remote areas. It is expected that the reduction in off-road activity and the reduction in road access in general will reduce physical disturbance to soil surfaces, and thus reduce the opportunities for existing weed populations to expand (see also *Impacts of Energy Development and Vehicles* in Chapter 4.3, Dumroese, this synthesis, *Sagebrush Rangelands and Greater Sage-grouse in Northeastern California*). The next step is to ensure that vehicles (including bicycles and aircraft) are weed-free prior to entering the back country, as recommended by regional guidelines. The *USDA Forest Service Guide to Noxious Weed Prevention Practices* (USDA 2001) has basic guidelines in managing equipment going in and out of the field, and although a bit dated, should be standard operating procedures for all staff, contractors, ranchers, and recreationists.

Fire

Fire is a natural component of the sagebrush steppe (fig. 3.3.5). The history and current status of fire in the sagebrush steppe has been well described by Ellsworth and Kauffman (2017) and Riegel et al. (2006). Empirical data regarding fire-return intervals before settlement in sagebrush steppe is limited, but estimates of 15 to 25 years before human activities are typical (Miller and Rose 1995). However, natural fire-return intervals are influenced by moisture gradient. In dryer areas, such as south-facing slopes where evapotranspiration is high and overall vegetation productivity is low, juniper (*Juniperus*) trees older than 50 years are common. While on the more productive adjacent slopes where evapotranspiration is lower due to lower solar radiation, older junipers are usually absent, but vegetation cover is denser. Fire ecologists use this relationship between moisture, vegetation density, and fire behavior on physically adjacent landscapes to deduce fire-return intervals absent human influences. The reasoning follows: because junipers are more resistant as they age (50 years being a commonly

Figure 3.3.5—Fire is a natural component of the sagebrush ecosystem and is one of the most common tools used to restore them. Prescribed fire is used to replace wildfires that would naturally keep sagebrush stands from becoming invaded by conifers that reduce the perennial grass and forb components (photo by Kenneth O. Fulgham, Regents of the University of California, used with permission).

noted age), the absence of older trees suggest that fire-return intervals on productive soils, pre-European settlement, may have been 50 years or less. In contrast, less-productive areas with lower fuels loads may not experience crown-killing fires for 100 years or more (Riegel et al. 2006; Rimer and Evans 2006). See Chapter 2.1 (Moser, this synthesis, *Understanding and Managing the Dry Conifer Forests of Northeastern California*) for additional discussion.

In modern times, the changes in fuel loading, particularly invasions of weedy grasses, is arguably the most serious problem in increased fire rates. Dried foliage is easily ignited and often provides a continuous mat of flammable fuel that accelerates fire spread (Stewart and Hull 1949). After fire, native plants are slow to recruit and grow, allowing cheatgrass, among others, to dominate the landscape (Stewart and Hull 1949). The presence of cheatgrass and other annual grasses has changed the fire regimes in many areas (Brooks et al. 2004; D'Antonio and Vitousek 1992). Lightning strikes are frequent in the sagebrush steppe (van Wagtendonk and Cayan 2008). Under pristine conditions, a lightning strike may initiate a fire, but with little understory fuel, the fire is restricted to a small area, as is consistent with the patchy nature of shrublands and woodlands. When grasses occupy the understory, a lightning strike can become a conflagration as fire spreads from shrub and tree patches on corridors of grass tinder. Thus, cheatgrass can increase both the frequency and extent of fire, with high associated costs for public land managers (Borman 2000; National Interagency Fire Center 2013).

The recovery of native shrubs following fire depends on several variables (Ellsworth and Kauffman 2017), among them, the general health of the individual and the age. Antelope bitterbrush (*Purshia tridentata*) and curlleaf mahogany (*Cercocarpus ledifolius*), important browse for wildlife, rarely resprout when younger than 5 years or older than 20 to 40 years (Martin and Driver 1983). Most of the sagebrush species are highly susceptible to fires (Hanna and Fulgham 2015). Except for silver sagebrush (*Artemisia cana*), regrowth after fire is seed-dependent. After large high-intensity fires, recolonization by sagebrush can be slow if few seeds are left unburned and mature plants with viable seeds are far away.

In 2015, the Secretary of the U.S. Department of the Interior (DOI) issued *Secretarial Order 3336 – The Initial Report* (USDOI 2015a) that highlighted the need to change the way fires were managed on rangelands, particularly on rangelands occupied, or once occupied, by sage-grouse (*Centrocercus* spp.). The order required actions by many DOI agencies and required that DOI work cooperatively and collaboratively with other Federal agencies, States and tribes, and stakeholders to develop an "enhanced fire prevention, suppression, and restoration strategy."

Conifer Encroachment

Changes in fire regime and grazing have contributed to extensive conifer encroachment into the sagebrush ecosystems (fig. 3.3.6), reducing habitat for greater sage-grouse (*Centrocercus urophasianus*) and grazing opportunity for livestock (Burkhardt and Tisdale 1969; Miller and Wigand 1994; Miller and Rose 1995). Throughout the Great Basin, this encroachment is often a mixture of juniper and Jeffrey and ponderosa pines (yellow pines; *Pinus jeffreyi* and *P. ponderosa*) (see Chapter 2.1, Moser, this synthesis, *Understanding and Managing the Dry Conifer Forests of Northeastern California*, for a robust discussion on juniper woodlands in the West, their ecology and dynamics, and management). This chronic, relentless encroachment, and its impacts on wildlife, have been observed for decades. Loft (1998) writes *"Northeastern California has recently been identified as a focus area for deer habitat management efforts on public lands in California where the objective is to improve habitat conditions (Loft et al. 1998). Deer populations and deer habitat conditions have declined significantly in recent decades. Deer populations in the area were estimated at 160,000, 130,000, and 35,000 in 1949, 1992, and 1996, respectively (Longhurst et al. 1952, Loft et al. 1998). Since 1957, overstory canopy of juniper and pine has increased by over 400 percent on some key bitterbrush ranges, thereby crowding and shading out of desirable browse (CDFG unpubl. data 1998)."*

The scale of encroachment puts pressure on limited resources. Greater sage-grouse are particularly sensitive to conifers. Data suggests that increases in conifer cover as small as 4 percent eliminate breeding leks from once active areas (Severson et al. 2017). Other than an increase in the available literature supporting the need for conifer removal in support of expanding greater sage-grouse habitat, no new papers contain substantial new methods for management of conifer woodlands. See Chapter 3.2 (Dumroese, this synthesis, *Rangeland in Northeastern California*) for restoration techniques of sagebrush rangelands, including conifer removal.

Figure 3.3.6—Western junipers (*Juniperus occidentalis*) are native and an important component of the Great Basin plant community, but they can also be invasive, replacing shrublands with woodlands. Juniper encroachment is a problem for greater sage-grouse habitat (stock photo purchased from alamy.com).

References

Aguirre, L.; Johnson, D.A. 1991. Influence of temperature and cheatgrass competition on seedling development of two bunchgrasses. Journal of Range Management. 44: 347–354.

Aldridge, C.L.; Nielsen, S.E.; Beyer, H.L.; [et al.]. 2008. Range-wide patterns of greater sage-grouse persistence. Diversity and Distributions. 14: 983–994.

Archer, A.J. 2001. *Taeniatherum caput-medusae.* In: Fire Effects Information System [Database]. Missoula, MT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory. https://www.feis-crs.org/feis/ (11 Feb. 2020).

Benner, D.; Parker, V.T. 2004. Impact of a native parasitic plant, *Cuscuta subinclusa,* on the progress of an invasive species, *Lepidium latifolium,* in a wetland community. Ecological Society of America Annual Meeting Abstracts. 89: 41.

Blank, R.S.; Morgan, T. 2012. Suppression of *Bromus tectorum* L. by established perennial grasses: potential mechanisms: part I. Applied and Environmental Soil Science. 2012: 632172. 9 p.

Borman, M.M. 2000. The Great Basin: healing the land. Boise, ID: U.S. Department of the Interior, Bureau of Land Management. 36 p.

Bouchard, E.H.; Little, L.E.; Miller, C.M.L.; [et al.]. 2015. Undeclared baggage: do tourists act as vectors for seed dispersal in fynbos protected areas? Koedoe. 57: 1323. 9 p.

Brancatelli, G.I.E.; Zalba, S.M. 2018. Vector analysis: a tool for preventing the introduction of invasive alien species into protected areas. Nature Conservation. 24: 43–63.

Brooks, M.L.; D'Antonia, C.M.; Richardson, D.M.; [et al.]. 2004. Effects of invasive alien plants on fire regime. BioScience. 54: 677–688.

Burkhardt, J.W.; Tisdale, E. 1969. Nature and successional status of western juniper vegetation in Idaho. Journal of Range Management. 264–270.

California Department of Fish and Wildlife. 2015. California State Wildlife Action Plan. https://www.wildlife.ca.gov/SWAP/Final (14 Aug. 2018).

California Department of Food and Agriculture [CDFA]. 2018. California Department of Agriculture Projects and Resources. https://www.cdfa.ca.gov/plant/ipc/biocontrol/84weed_proj-summ.htm (14 Aug. 2018).

California Invasive Plant Council [Cal-IPC]. 2018. California Invasive Plant Control. https://www.cal-ipc.org/resources/library/publications/ipcw/report59/ (27 July 2018).

Concilio, A.L. 2013. Effectiveness and cost of downy brome (*Bromus tectorum*) control at high elevation. Invasive Plant Science and Management. 6: 502–511.

Concilio, A.; Vargas, T.; Cheng, W. 2015. Effect of an invasive annual grass on rhizosphere priming in invaded and uninvaded soils. Plant and Soil 393: 245–257.

D'Antonio, C.M.; Vitousek, P.M. 1992. Biological invasions by exotic grasses, the grass-fire cycle, and global change. Annual Review of Ecology and Systematics. 23: 63–87.

Diamond, J.M.; Call, C.A.; Devoe. N. 2009. Effects of targeted cattle grazing on fire behavior of cheatgrass-dominated rangeland in the northern Great Basin, USA. International Journal of Wildland Fire. 18: 944–950.

DiTomaso, J.M.; Kyser, G.B. 2013, Weed control in natural areas in the western United States. Davis, CA: University of California, Weed Research and Information Center. 544 p.

Dukes, J.S.; Mooney, H.A. 2004. Disruption of ecosystem processes in western North America by invasive species. Revista Chilena de Historia Natural. 77: 411–437.

Duncan, C.; Story, J.; Sheley, R. 2017. Biology, ecology and management of Montana knapweeds. Bozeman, MT: Montana State University Extension. 19 p.

Ellsworth, L.M.; Kauffman, J.B. 2017. Plant community response to prescribed fire varies by pre-fire condition and season of burn in mountain big sagebrush ecosystems. Journal of Arid Environments. 144: 74–80.

Evans, R.A.; Young, J.A. 1984. Microsite requirements of downy brome (*Bromus tectorum*) infestation and control on sagebrush rangelands. Weed Science. 32: 13–17.

Federal Register. 1999. Executive Order 13112 of February 3, 1999: Invasive Species. Federal Register. 64(25): 6183–6186.

Gates, P.W. 1968. History of public land law development. Washington, DC: United States Land Law Review Commission. 828 p.

Hafferkamp, M.R.; Heitschmidt, R.K.; Grings, E.E.; [et al.]. 2001. Suppression of annual bromes impacts rangeland: vegetation responses. Journal of Range Management. 54: 656–662.

Hanna, S.K.; Fulgham, K.O. 2015. Post-fire vegetation dynamics of a sagebrush steppe community change significantly over time. California Agriculture. 69: 36–42.

Ibekwe, A.M.; Kennedy, A.C.; Stubbs, T.L. 2010. An assessment of environmental conditions for control of downy brome by *Pseudomonas fluorescens* D7. International Journal of Environmental Technology and Management. 12: 27–46

Johnson, D.H.; Holloran, M.J.; Connelly, J.W.; [et al.]. 2011. Influences of environmental and anthropogenic features on greater sage-grouse populations, 1997–2007. In: Knick, S.T.; Connelly, J.W., eds. Greater sage-grouse: Ecology and conservation of a landscape species and its habitats. Studies in Avian Biology 38. Berkeley, CA: University of California Press for Cooper Ornithological Society: 407–450.

Knapp, P.A. 1996. Cheatgass (*Bromus tectorum*) dominance in the Great Basin Desert: history, influences, and persistence to human activities. Global Environmental Change. 6: 37–52.

Leffler, J.A.; Monaco, T.A.; James, J.J. 2011. Morphological and physiological traits account for similar nitrate uptake by crested wheatgrass and cheatgrass. Natural Resources and Environmental Issues. 17: 10.

Loft, E.R. 1998. Economic contribution of deer, pronghorn antelope, and sage grouse hunting to northeastern California and implications to the overall "value" of wildlife. California Wildlife Conservation Bulletin 11. Sacramento, CA: California Department of Fish and Game. 42 p.

Long, J.W.; Quinn-Davidson, L.N.; Skinner, C.N. 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.

Martin, R.E.; Driver, C.H. 1983. Factors affecting antelope bitterbrush reestablishment following fire. In: Tiedemann, A.R.; Johnson, D.L., compilers. Research and management of bitterbrush and cliffrose in western North America. Gen. Tech. Rep. INT-152. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermounain Research Station: 266–279.

McAdoo, J.K.; Schultz, B.W.; Swanson, S.R. 2013. Aboriginal precedent for active management of sagebrush-perennial grass communities in the Great Basin. Rangeland Ecological Management. 66: 241–253.

Mealor, B.A.; Mealor, R.D.; Kelley, W.K.; [et al.]. 2013. Cheatgrass management handbook—managing an invasive annual grass in the Rocky Mountain region. Laramie, WY and Fort Collins, CO: University of Wyoming and Colorado State University. 136 p.

Meyer, S.E.; Quinney, D.; Nelson, D.L.; [et al.]. 2007. Impact of the pathogen *Pyrenophora semeniperda* on *Bromus tectorum* seedbank dynamics in North American cold deserts. Weed Research. 47: 54–62.

Miller, R.F.; Rose, J.A, 1995. Historic expansion of *Juniperus occidentalis* (western juniper) in southeastern Oregon. The Great Basin Naturalist. 55(1): 37-45.

Miller, R.F.; Wigand, P.E., 1994. Holocene changes in semiarid pinyon-juniper woodlands. BioScience. 44: 465–474.

Monaco, T.A.; Johnson, D.A.; Norton, J.M.; [et al.]. 2003. Contrasting responses of intermontain west grasses to soil nitrogen. Journal of Range Management. 56: 282–290.

Morris, L.R.; Rowe, R.J. 2014. Historical land use and altered habitats in the Great Basin. Journal of Mammology. 95: 1144–1156.

Mosley, J.C.; Bunting, S.C.; Manoukian, M.E. 1999. Cheatgrass. In: Sheley R.L.; Petroff, J.K., eds. Biology and management of noxious rangeland weeds. Corvallis, OR: Oregon State University Press: 175–188.

National Interagency Fire Center. 2013. 2013 National report on state and agency fires and acres burned. https://www.nifc.gov/fireInfo/fireInfo_stats_YTD2013.html (26 Jan. 2018).

Ostoja, S.M.; Schupp, E.W. 2009. Conversion of sagebrush shrublands to exotic annual grasslands negatively impacts small mammal communities. Diversity and Distributions. 15: 863–870.

Renz, M.J.; Blank, R.R. 2004. Influence of perennial pepperweed (*Lepidium latifolium*) biology and plant/soil relationships on management and restoration. Weed Technology. 18: 1359–1363.

Riegel, G.M.; Miller, R.F.; Skinner, C.N.; [et al.]. 2006. Northeastern plateaus bioregion. In: Sugihara, N.G.; van Wagtendonk, J.W.; Fites-Kaufman, J.; Shaffer, K.E.; Thode, A.E., eds. Fire in California's ecosystems. Berkeley, CA: University of California Press: 225–263.

Rimer R.L.; Evans, R.D. 2006. Invasion of downy brome (*Bromus tectorum* L.) causes rapid changes in the nitrogen cycle. American Midlands Naturalist. 6: 252–258.

Rinella, M.J.; Maxwell, B.E.; Fay, P.K.; [et al.]. 2009. Control effort exacerbates invasive-species problem. Ecological Applications. 19: 155–162.

Severson, J.P.; Hagen, C.A.; Tack, J.D.; [et al.]. 2017. Better living through conifer removal: a demographic analysis of sage-grouse vital rates. PLoS ONE. 12: e0174347.

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.

Stewart, G.; Hull, A.C. 1949. Cheatgrass (*Bromus tectorus* L.)—an ecological intruder in southern Idaho. Ecology. 30: 58–74.

Svejcar, L.N.; Kildisheva, O.A. 2017. The age of restoration: challenges presented by dryland systems. Plant Ecology. 218: 1–6.

Svejcar, T. 2015. The northern Great Basin: A region of continual change. Rangelands. 37: 114–118.

Svejcar, T.; Boyd, C.; Davies, K.; [et al.]. 2017. Challenges and limitations to native species restoration in the Great Basin, USA. Plant Ecology. 218: 81–94.

Tzankova, Z.; Concilio, A. 2015. Controlling an invasive plant at the edge of its range: towards a broader understanding of management feasibility. Biological Invasions. 17: 507–527.

U.S. Department of Agriculture, Forest Service [USDA]. 2001. USDA Forest Service guide to noxious weed prevention practices. Version 1.0, 5 Jul 2001. 25 p. https://www.fs.fed.us/invasivespecies/documents/FS_WeedBMP_2001.pdf (17 April 2020).

U.S. Department of Agriculture, Forest Service; U.S. Department of the Interior, Bureau of Land Management; Modoc County, CA [USDA et al]. 2008. Sage steppe ecosystem restoration strategy: Final environmental impact statement. R5-MB-161. Alturas, CA: Modoc National Forest, Bureau of Land Management Alturas Field Office, and Modoc County. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5315306.pdf (18 April 2020).

U.S. Department of Agriculture, Forest Service [USDA]. 2010a. Motorized travel management—Final environmental impact statement—Lassen National Forest. R5-MB-207. Susanville, CA: Lassen National Forest. 636 p.

U.S. Department of Agriculture, Forest Service [USDA]. 2010b. Motorized travel management—Final environmental impact statement—Modoc National Forest. R5-MB-202. Alturas, CA: Modoc National Forest. 639 p.

U.S. Department of Agriculture, Forest Service [USDA]. 2013. Forest Service national strategic framework for invasive species management. FS-1017. Washington, DC: USDA Forest Service. 35 p.

U.S. Department of Agriculture, Forest Service [USDA]. 2014a. Field guide for managing yellow starthistle in the Southwest. TP-R3-16-07. Albuquerque, NM: Southwestern Region. 8 p.

U.S. Department of Agriculture, Forest Service [USDA]. 2014b. Field guide for managing perennial pepperweed in the Southwest. TP-R3-16-23. Albuquerque, NM: Southwestern Region. 7 p.

U.S. Department of Agriculture; Natural Resources Conservation Service [USDA NRCS]. n.d. Native, invasive, and other plant-related definitions. https://www.nrcs.usda.gov/wps/portal/nrcs/detail/ct/technical/ecoscience/invasive/?cid=nrcs142p2_011124 (24 Jan. 2018).

U.S. Department of the Interior [USDOI]. 2015a. SO 3336—The initial report. A strategic plan for addressing rangeland fire prevention, management, and restoration in 2015. https://www.forestsandrangelands.gov/rangeland/documents/SO3336-TheInitial%20Report_20150310.pdf (26 Jan. 2018).

U.S. Department of the Interior [USDOI]. 2015b. Historic conservation campaign protects greater sage-grouse. https://www.doi.gov/pressreleases/historic-conservation-campaign-protects-greater-sage-grouse (29 Sep. 2017).

U.S. Department of the Interior, U.S. Geological Survey [USGS]. 2016. The Great Basin and Columbia Plateau. https://greatbasin.wr.usgs.gov/ (24 Jan. 2018).

U.S. Fish and Wildlife Service [USFWS]. 2015. Greater sage-grouse (Centrocercus urophasianus). https://ecos.fws.gov/speciesProfile/profile/speciesProfile.action?spcode=B06W (9 Sep. 2015).

Usher, M.B. 1988. Biological invasions of nature reserves: A search for generalisations. Biological Conservation. 44: 119–135.

van Wagtendonk, J.W.; Cayan, D. 2008. Spatial patterns of lightning strikes and fires in California. Fire Ecology. 4(1): 34–56.

Vollmar Consulting AECOM. 2009. Effectiveness of small vernal pool preserves. Prepared for the Placer Land Trust. Berkeley, CA: Vollmar Consulting. 87 p.

Wilson, L.M.; Jette, C.; Connett, J.; [et al.]. 2003. Biology and biological control of yellow starthistle, 2nd ed. FHTET-1998-17. Morgantown, WV: Forest Health Technology Enterprise Team: 11–47.

Young, J.A.; Evans, R.A.; Eckert, R.E., Jr. 1969. Population dynamics of downy brome. Weed Science. 17: 20–26.

Young, J.A.; Turner, C.E.; James, L.F. 1995. Perennial pepperweed. Rangelands. 17: 121–123.

Zouhar, K. 2002. Centaurea solstitialis. In: Fire Effects Information System. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory. https://www.fs.fed.us/database/feis/plants/forb/censol/all.html (20 April 2020).

Chapter 3.4. Biological Soil Crusts

Steven D. Warren[1]

Introduction

Biological soil crusts (BSCs) develop when various combinations of a vast array of bacteria, cyanobacteria, fungi, lichens, terrestrial algae, and mosses occupy the surface and upper few millimeters of the soil (Warren et al. 2019b). Historically, BSCs have been referred to as cryptobiotic, cryptogamic, microbiotic, microfloral, microphytic, and organogenic crusts. They can be present in a wide range of ecological, successional, and climatic conditions when and where disturbance and/or aridity have resulted in opportunities for colonization of exposed bare soil surfaces. BSCs are most prevalent, however, in arid and semiarid ecosystems, such as those of Northeastern California, where vascular plant cover and diversity are perennially low, leaving large areas available for colonization by some combination of the organismal groups mentioned above.

Ecological Roles of, and Threats to, BSCs

The ecological roles of BSCs are many and varied and include: (1) the redistribution of precipitated rainwater; (2) the capture, collection, and use of airborne and soil nutrients; (3) interaction with vascular plants; and (4) soil stabilization (Belnap and Lange 2001; Warren 1995; Weber et al. 2016).

Hydrology

The scientific literature is replete with apparent contradictions regarding role of BSCs relative to their effects on soil hydrology. Many authors have concluded that BSCs enhance infiltration, while others have concluded that the presence of a crust diminishes it. An extensive review by Warren (2001) revealed that much of the variation can be attributed to soil texture. Where the sand content of the soil exceeds about 80 percent, and the soil is not frost-heaved, BSCs generally reduce water infiltration compared to soil without crust. This seems to be attributable to soil porosity. Porosity of sandy soil may be reduced as BSCs accumulate finer soil particles or when polysaccharide exudates, rooting structures, or cyanobacterial filaments clog the soil pores, thus impeding the free passage of water. Where sand content is less than about 80 percent, BSC organisms contribute to soil aggregation, and, thus, higher effective porosity and higher rates of infiltration than in similar soil without biological crusts.

Research has shown BSCs impede evaporation and help retain moisture in the soil (Booth 1941). This may be accomplished as the BSCs create a surface seal or mulch layer (Booth 1941; Fritsch 1922), increase organic matter content of the soil (Metting 1981), or the polysaccharide exudates of cyanobacteria absorb moisture (Belnap and Gardner 1993).

Despite the overall importance of BSCs, and the well-documented effects of disturbance on these communities, restoring degraded habitats has received proportionately little attention (Bowker 2007). Reflection on the broader scope of BSC restoration can improve our perspective of how to effectively manage important dryland regions, in addition to directing future research.

Nutrient Cycling

The presence of BSCs is positively correlated to the presence and abundance of many micro- and macronutrients in the soil. Numerous studies documented the fixation of atmospheric nitrogen by BSCs dominated by cyanobacteria or lichens possessing cyanobacterial symbionts. Vascular plants in the Great Basin take up 9 to 11 pounds of nitrogen per ac (10 to 12 kg per ha) (West and Skujiņš 1977). Of this nitrogen requirement, precipitation provides 0.9 to 5.4 pounds per ac (1 to 6 kg per ha) per year (Schlesinger 1991; West 1978). Heterogeneous nitrogen fixers contribute a fraction of that need (Reichert et al. 1978; Steyn and Delwiche 1970). The balance is attributable to BSCs.

Much of the nitrogen fixed by BSCs is retained in the surface few centimeters of the soil (Fletcher and Martin 1948) where it is available to vascular plants. The nitrogen content of soils with BSCs may be up to 7 times higher than in similar soils without BSCs (Shields 1957). Other essential elements are also accumulated in surface soils with BSCs (Harper and Pendleton 1993; Loope and Gifford 1972; Kleiner and Harper 1972). While this can be attributed, in part, to the accumulation of fine soil

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particles by the rough surface of crusted soil (Kleiner and Harper 1972), other factors also play a role. BSC organisms accelerate weathering of rocks, thus speeding the genesis of soil and adding important minerals (Metting 1991). It has also been demonstrated that cyanobacteria exude polysaccharides that contain chelating agents that concentrate essential nutrients (Lange 1974, 1976). In addition, negatively charged clay particles may be bound to, or incorporated into, the polysaccharide exudates of some cyanobacteria, thus attracting and binding with positively charged essential elements (Belnap and Gardner 1993).

On sites in Colorado and Utah, BSCs were found to increase plant uptake of many nutrients, especially for herbaceous plants, results similar to other field and greenhouse studies under a variety of different soil conditions (Harper and Belnap 2001). This improved nutrient availability may occur because BSCs typically accumulate finer soil particles important for holding and exchanging nutrients, many accumulate or fix nitrogen from the atmosphere, and release it to the soil environment for use by vascular plants and other organisms, and BSCs might modify the soil carbon-to-nitrogen ratio that promotes decomposition and release of nutrients. Research also indicates that later-successional BSCs have rates of photosynthesis 3 to 4 times higher than early-successional BSCs (Grote et al. 2010); this carbon accumulation may have implications on polymer secretion.

Enhancement of Vascular Plant Germination and Survival

BSCs frequently improve infiltration of rainwater into the soil. Dark-colored BSCs absorb more solar radiation than soil without a crust, raising the temperature near the surface by as much as 9 °F (5 °C) (Harper and Marble 1988). Elevated soil temperatures, when coupled with adequate soil moisture, may accelerate the germination and initial growth of vascular plants (Harper and Pendleton 1993). The effects of BSCs on seed germination are mixed. Greenhouse studies either show no difference in germination (Godínez-Alvarez et al. 2012; Pando-Moreno et al. 2014; Serpe et al. 2006) or reduced germination (Song et al. 2017) on BSC-covered soil. Mendoza-Aguilar et al. (2014) found no germination effect on field sites. Some of these effects may be attributable to the type of crust tested; Serpe et al. (2006) in a laboratory setting, found that germination of native Western United States grasses, as well as cheatgrass (*Bromus tectorum*), was about the same on bare soil or one with a tall-moss crust, but significantly reduced when the BSC was dominated by short mosses.

Soil Stabilization

BSCs contribute to soil stability in at least three ways. First, the presence of vegetative structures such as larger lichens and mosses tends to dissipate the kinetic energy of wind, raindrops, and overland flow of water, thus reducing the susceptibility of soil erosion. Second, even where larger structures are absent, many BSCs are characterized by an uneven micro-topography that decreases the velocity of both wind and water. Third, BSCs contribute to mechanical and chemical aggregation of soil particles. Moss rhizoids, fungal hyphae, and filamentous cyanobacteria and algae often form a dense fibrous mesh that holds soil particles in place (Belnap and Gardner 1993; Fletcher and Martin 1948; Schulten 1985; Tisdall and Oades 1982). The extracellular polysaccharide exudates of some BSC organisms, particularly cyanobacteria, form glue-like bonds that prevent detachment.

McCalla (1946) first showed that soil aggregates formed by association with mosses, algae, and fungi were more resistant to disintegration by falling raindrops than aggregates formed without the benefit of BSC organisms. Osborn (1952) subsequently determined that the presence of a cyanobacteria-dominated crust on a deteriorated rangeland significantly reduced detachment and splash erosion by raindrops. Greater rainfall intensity was required to initiate splash erosion on moss- and lichen-dominated crusts than cyanobacterial crusts or bare ground.

Effects of Disturbance

BSCs can be disturbed and often killed by a variety of disturbances. These may include livestock trampling (Warren and Eldridge 2001), human trampling (Cole 1990), off-road vehicular traffic (Webb et al. 1988; Wilshire 1983), fire (Johansen 2001), military training (Warren 2014), competition for resources from invasive species (Dettweiler-Robinson et al. 2013), and several other disturbance vectors.

Most physical disturbance leads to increased soil erosion (Booth 1941; Brotherson and Rushforth 1983; Loope and Gifford 1972), and loss of ecological function when compared to intact crusts. Belnap (2002) found that four passes of a vehicle reduced the ability of BSCs to fix nitrogen from the atmosphere, especially on coarse-textured soils. This reduction affects other organisms, too, because BSC-fixed nitrogen is readily used by other plants and microorganisms (Belnap 2002). Grazing can reduce soil nitrogen levels by up to 50 percent (Jeffries 1989) and plants growing on grazed land have lower nitrogen concentrations (Evans and Belnap 1999), reflecting the reduced nitrogen fixation and sharing by the BSC. Fire can also degrade or destroy BSCs, thus having an immediate effect on nitrogen fixation (Johansen 2001); fire disturbance is greater under burned woody plants and less so under herbaceous material, a reflection of differing fire intensities. Manipulating fire intensity through prescribed burning can reduce impacts to BSCs (Warren et al. 2015). Johansen (2001) noted, however, that if a BSC is damaged and cheatgrass invades, BSCs may never regain their pre-disturbance levels. Recent work by Dettweiler-Robinson et al. (2013) reveals that lichen cover, in both early and late-successional stages, is negatively affected by cheatgrass. Rosentreter et al. (2001) discuss monitoring and management of BSCs to reduce degradation.

Artificial Restoration of BSCs

It may seem intuitive to attempt to restore BSCs by inoculating disturbed sites with crust organisms, but such applications have been relatively rare. Several researchers have harvested BSCs from the field in an attempt to use them as inocula for restoration sites (Belnap 1993; Bu et al. 2014; Maestre et al. 2006). This work was generally done under controlled conditions and yielded limited success. In general, the best results, which provided a modicum of success, required very moist substrates. Salvaging BSC from construction sites has also been examined (Chiquoine et al. 2016), as has transplanting soil cores with intact BSCs (Cole et al. 2010). In most of the aforementioned cases, inoculation hastened recovery of BSC organisms, particularly in controlled laboratory settings, with some recovery also in field studies. While the results were promising, the destruction of BSCs in one area to provide inoculants for another area is counterproductive in the context of large-scale arid land reclamation. Use of salvaged crusts from construction sites is promising for limited areas (Chiquoine et al. 2016). It is unlikely that providing sufficient supplemental water for successful large-scale reclamation in arid environments will be feasible.

To avoid destroying BSCs for use as inocula, some research has investigated the potential for laboratory-grown BSC amendments for use in inoculating disturbed areas (Zhao et al. 2016). For example, Buttars et al. (1998) incorporated laboratory-grown cyanobacteria into alginate pellets. These pellets, once crushed to allow release of the organisms and applied to moistened soil in the laboratory, resulted in significant increases in cyanobacterial biomass and frequency and nitrogen fixation. Incorporation of cyanobacteria into starch pellets was not successful due to poor survival during the pelletization process (Howard and Warren 1998). Kubeková et al. (2003) grew cyanobacteria and immobilized it on hemp cloth. Laboratory trials indicated improved growth compared to alginate pellets, but in four of five field trials, no significant crust recovery occurred. When cyanobacterial inoculants have been applied to the soil surface, rather than incorporated into the surface layer of the soil, mortality has been high. Laboratory-grown moss protonema, the earliest stage of growth, transplanted into the sands of the Gurbantunggut Desert of China has seen some success when supplemented with liquid growth media (Xu et al. 2008). The addition of laboratory-grown cyanobacteria to polyvinyl alcohol and a liquid soil tackifier appeared to accelerate the formation of a biocrust in a laboratory setting (Park et al. 2017).

Although some degree of success has been noted, large-scale field trials have yet to be attempted, and successful laboratory production and growth across BSC components is not universal. Given the general lack of success of artificial techniques to restore the BSC component, the levels of water required, and the per-acre costs, it is reasonable to question whether these approaches merit further consideration in arid areas except in critical situations where cost is not a constraint.

Aerobiology and Natural Recovery of BSCs

BSCs are found in almost all environments, justifying the question as to how crust organisms became so spatially and climatically dispersed in the first place and if that same process is still operating. In general, as post-disturbance succession takes place, the initial colonizers that stabilize the surface tend to be large filamentous cyanobacteria (Belnap and Eldridge 2001). They are followed by smaller cyanobacteria and green algae, which are followed, in turn, by small lichens. Where climatic conditions permit, larger lichens and mosses appear in later-successional communities. The distribution and successful establishment of these organisms is governed both by historical and contemporary factors (Leavitt and Lumbsch 2016).

BSCs are naturally dispersed by wind (Warren et al. 2019a, b). Aerially dispersed microorganisms were first observed by Darwin (1846). Meier and Lindbergh (1935) collected airborne organisms during a flight over the Arctic. Shortly thereafter, the field of aerobiology was established (Benninghoff 1991) and large numbers of microorganisms, including bacteria, fungi, terrestrial algae, and bryophytes, have been documented throughout the Earth's atmosphere (Després et al. 2012; Genitsaris et al. 2011; Schlichting 1969; Sharma et al. 2007; Tesson et al. 2016). Some of these organisms may have multiple generations while in the atmosphere, such that the atmosphere becomes a truly aerial habitat (Womack et al. 2010). Not surprisingly many of the species documented in the atmosphere are also common in BSC communities and may be deposited almost anywhere, including bare soil (Harding et al. 2011; Kvíderová 2012; Marshall and Chalmers 1997; Takeuchi 2013; Vonnahme et al. 2016), buildings (Samad and Adhikary 2008; Sethi et al. 2012), stone monuments (Macedo et al. 2009; Tomaselli et al. 2000), and plant surfaces (McGorum et al. 2015; Sethi et al. 2012). In addition to algae and cyanobacteria, other BSC components, including asexual reproductive lichen fragments, soredia, and/or lichen-forming fungal spores (Heinken 1999; Leavitt and Lumbsch 2016; Tormo et al. 2001), as well as spores, gametophyte fragments, and specialized asexual diaspores of bryophytes (Laaka-Lindberg et al. 2003; Stark 2003) can also be dispersed by wind.

BSC organisms can achieve airborne status when strong non-convective horizontal winds blowing over unconsolidated soil surfaces pick up large quantities of soil and the associated organisms. Strong dust storms occur in North America (McLeman et al. 2014), Alaska (Nickling 1978), China (Wang et al. 2004), Australia (Ekström et al. 2004), Africa (Prospero and Mayor-Bacero 2013), and the Middle East (Almuhanna 2015). On a smaller, but more common scale, dust may be lifted into the atmosphere by strong vertical vortices or "dust devils" (Horton et al. 2016; Metzger et al. 2011). Once airborne, dust particles and the BSC organisms that often accompany them, are subject to a variety of forces, including trade winds and the jet stream, that carry them, often rapidly, between hemispheres, continents, and climatic zones (Doherty et al. 2008; Griffin et al. 2002; Kellogg and Griffin 2006; Lee et al. 2006; Prospero 1999; Prospero and Lamb 2003; Uno et al. 2009). Many of these windborne BSC propagules can survive long periods of desiccation (Holzinger and Karsten 2013; Rajeev et al. 2013). Thus, given the forces mixing the atmosphere, the likelihood for BSC propagules to be present in it, and BSC resistance to desiccation, little doubt exists that organisms originating from almost any given location have the potential to be deposited anywhere on Earth (Barberán et al. 2014; Herbold et al. 2014; Jungblut et al. 2010), as evidenced by similarity of BSC species in the northern and southern polar regions, Iceland, and extreme southern Chile (Galloway and Aptroot 1995; Jungblut et al. 2012; Piñeiro et al. 2012).

Thus, natural recovery of BSC is expected due to windborne deposition. Time estimates for natural recovery of BSCs following disturbance have varied widely depending on the nature, periodicity, extent, and spatial and temporal distribution of the disturbance, and soil and climatic conditions. Five years following one-time human trampling, Cole (1990) noted a nearly complete recovery of visible BSC cover, although the complex pinnacled surface micro-topography attributable to many crusts had not recovered to pre-disturbance levels. Anderson et al. (1982) estimated that 14 to 18 years were adequate for recovery of a BSC following exclusion of livestock grazing in the cool Great Basin. In contrast, little evidence of recovery was observed during the first 10 years following cessation of grazing at another Great Basin location (Jeffries and Klopatek 1987). Recovery lagged 20 years following burning of a shrub community in the transition zone between the Great Basin and Mojave Deserts in Southwestern Utah (Callison et al. 1985). Belnap (1993) estimated that full recovery of BSCs in the Great Basin, including visual as well as functional characteristics, could require as many as 30 to 40 years for the cyanobacterial component, 45 to 85 years for lichens, and 250 years for mosses. In the Sonoran Desert, after 56 years, a cyanobacterial crust degraded by military exercises had not recovered to levels typical of adjacent undisturbed areas (Kade and Warren 2002). And, in the Mojave Desert full recovery of the cyanobacterial component of the BSC from disturbance caused by military vehicles was estimated to require up to 85 to 120 years (Belnap and Warren 2002). Hilty et al. (2004) found that active restoration of burned sagebrush steppe through direct seeding increased natural recruitment of BSC compared to passive restoration, especially when grazing was temporarily suspended.

References

Almuhanna, E.A. 2015. Dustfall associated with dust storms in the Al-Ahsa Oasis of Saudi Arabia. Open Journal of Air Pollution. 4: 65–75.

Anderson, D.C.; Harper, K.T.; Rushforth, S.R. 1982. Recovery of cryptogamic soil crusts from grazing on Utah winter ranges. Journal of Range Management. 35: 355–359.

Barberán, A.; Henley, J.; Fierer, N.; [et al.]. 2014. Structure, inter-annual recurrence, and global-scale connectivity of airborne microbial communities. Science of the Total Environment. 487: 187–195.

Belnap, J. 1993. Recovery rates of cryptobiotic crusts: inoculant use and assessment methods. Great Basin Naturalist. 53: 89–95.

Belnap, J. 2002. Impacts of off-road vehicles on nitrogen cycles in biological soil crusts: resistance in different U.S. deserts. Journal of Arid Environments. 52: 155–165.

Belnap, J.; Eldridge, D. 2001. Disturbance and recovery of biological soil crusts. In: Belnap, J.; Lange, O.L., eds. Biological soil crusts: structure, function and management. Berlin, Germany: Springer-Verlag: 363–383.

Belnap, J.; Gardner, J.S. 1993. Soil microstructure in soils of the Colorado Plateau: the role of the cyanobacterium *Microcoleus vaginatus*. Great Basin Naturalist. 53: 40–47.

Belnap, J.; Lange, O.L., eds. 2001. Biological soil crusts: structure, function and management. Berlin, Germany: Springer-Verlag.

Belnap, J.; Warren, S.D. 2002. Patton's tracks in the Mojave Desert, USA: an ecological legacy. Arid Land Research and Management. 16: 245–258.

Benninghoff, W.J. 1991. Aerobiology and its significance in biogeography and ecology. Grana. 30: 9–15.

Booth, W.E. 1941. Algae as pioneers in plant succession and their importance in soil erosion control. Ecology. 22: 38–45.

Bowker, M.A. 2007. Biological soil crust rehabilitation in theory and practice: An underexploited opportunity. Restoration Ecology. 15: 13–23.

Brotherson, J.D.; Rushforth, S.R. 1983. Influence of cryptogamic crusts on moisture relationships of soils in Navajo National Monument, Arizona. Great Basin Naturalist. 43: 73–78.

Bu, C.; Wu, S.; Yang, Y.; Zheng, M. 2014. Identification of factors influencing the restoration of cyanobacteria-dominated biological soil crusts. PLoS ONE. 9: e90049.

Buttars, S.M.; St. Clair, L.L.; Johansen, J.R.; [et al.]. 1998. Pelletized cyanobacterial soil amendment: laboratory testing for survival, escapability, and nitrogen fixation. Arid Soil Research and Rehabilitation. 12: 165–178.

Callison, J.; Brotherson, J.D.; Bowns, J.E. 1985. The effects of fire on the blackbrush [*Coleogyne ramosissima*] community of southwestern Utah. Journal of Range Management. 38: 535–538.

Chiquoine, L.P.; Arbella, S.R.; Bowker, M.A. 2016. Rapidly restoring biological soil crusts and ecosystem functions in a severely disturbed desert ecosystem. Ecological Applications. 26: 1260–1272.

Cole, C.; Stark, L.R.; Bonine, M.L.; [et al.]. 2010. Transplant survivorship of bryophyte soil crusts in the Mojave Desert. Restoration Ecology. 18: 198–205.

Cole, D.N. 1990. Trampling disturbance and recovery of cryptogamic soil crusts in Grand Canyon National Park. The Great Basin Naturalist. 50: 321-325.

Darwin, C. 1846. An account of the fine dust which often falls on vessels in the Atlantic Ocean. Quarterly Journal of the Geological Society of London. 2: 26–30.

Després, V.R.; Huffman, J.A.; Burrows, S.M.; [et al.]. 2012. Primary biological aerosol particles in the atmosphere: a review. Tellus B: Chemical and Physical Meteorology. 64: 11598.

Dettweiler-Robinson, E.; Bakker, J.D.; Grace, J.B. 2013. Controls of biological soil crust cover and composition shift with succession in sagebrush shrub-steppe. Journal of Arid Environments. 94: 96–104.

Doherty, O.M.; Riemer, N.; Hameed, S. 2008. Saharan mineral dust transport into the Caribbean: Observed atmospheric controls and trends. Journal of Geophysical Research. 113: D07211.

Ekström, M.; McTainsh, G.H.; Chappell, A. 2004. Australian dust storms: temporal trends and relationships with synoptic pressure distributions 1960-00. International Journal of Climatology. 24: 1581–1599.

Evans, R.D.; Belnap, J. 1999. Long-term consequences of disturbance on nitrogen cycling in an arid grassland. Ecology. 80: 150–160.

Fletcher, J.E.; Martin, W.P. 1948. Some effects of algae and molds in the rain-crust of desert soils. Ecology. 29: 95–100.

Fritsch, F.E. 1922.The terrestrial algae. Journal of Ecology. 10: 220–236.

Galloway, D.J.; Aptroot, A. 1995. Bipolar lichens: a review. Cryptogamic Botany. 5: 184–191.

Genitsaris, S.; Kormas, K.A.; Moustaka-Gouni, M. 2011. Airborne algae and cyanobacteria: occurrence and related health effects. Frontiers in Bioscience. 3: 772–787.

Godínez-Alvarez, H.; Morín, C.; Rivera-Aguilar, V. 2012. Germination, survival and growth of three vascular plants on biological soil crusts from a Mexican tropical desert. Plant Biology. 14: 157–162.

Griffin, D.W.; Kellogg, C.A.; Garrison, V.H.; [et al.]. 2002. The global transport of dust: An intercontinental river of dust, microorganisms and toxic chemicals flows through the Earth's atmosphere. American Scientist. 90: 228–235.

Grote, E.E.; Belnap, J.; Housman, D.C.; Spark, J.P. 2010. Carbon exchange in biological soil crust communities under differential temperatures and soil water contents: implications for global change. Global Change Biology. 16: 2763–2774.

Harding, T.; Jungblut, A.D.; Lovejoy, C.; [et al.]. 2011. Microbes in high arctic snow and implications for the cold biosphere. Applied and Environmental Microbiology. 77: 3234–3243.

Harper, K.T.; Belnap, J. 2001. The influence of biological soil crusts on mineral uptake by associated vascular plants. Journal of Arid Environments 47: 347–357.

Harper, K.T.; Marble, J.R. 1988. A role for nonvascular plants in management of arid and semiarid rangelands. In: Tueller, P.T., ed. Vegetation science applications for rangeland analysis and management. Dordrecht: Kluwer Academic Publishers: 135–169.

Harper, K.T.; Pendleton, R.L. 1993. Cyanobacteria and cyanolichens: can they enhance availability of essential minerals for higher plants? Great Basin Naturalist. 53: 59–72.

Heinken, T. 1999. Dispersal patterns of terricolous lichens by thallus fragments. The Lichenologist. 31: 603–612.

Herbold, C.W.; Lee, C.K.; McDonald, I.R.; [et al.]. 2014. Evidence of global-scale aeolian dispersal and endemism in isolated geothermal microbial communities of Antarctica. Nature Communications. 5: 3875.

Hilty, J.H.; Eldridge, D.J.; Rosentreter, R.; [et al.]. 2004. Recovery of biological soil crusts following wildfire in Idaho. Journal of Range Management. 57: 89–96.

Holzinger, A.; Karsten, U. 2013. Desiccation stress and tolerance in green algae: consequences for ultrastructure, physiological, and molecular mechanisms. Frontiers in Plant Science. 4: 327.

Horton, W.; Miura, H.; Onishchenko, O.; Couede, L.; [et al.]. 2016. Dust devil dynamics. Journal of Geophysical Research: Atmospheres. 121: 7197–7214.

Howard, G.L.; Warren, S.D. 1998. The incorporation of cyanobacteria into starch pellets and determination of escapability rates for use in land rehabilitation. USACERL Special Report 98/56. Champaign, IL: U.S. Army Corp of Engineers, Construction Engineering Research Laboratory. 26 p.

Jeffries, D.L. 1989. The vegetation, soil, and cryptogamic crusts of blackbrush communities in the Kaiparowits Basin. Tempe, Arizona: Arizona State University. 157 p. Ph.D. dissertation.

Jeffries, D.L.; Klopatek, J.M. 1987. Effects of grazing on the vegetation of the blackbrush association. Journal of Range Management. 40: 390–392.

Johansen, J.R. 2001. Impacts of fire on biological soil crusts. In: Belnap, J.; Lange, O.L., eds. Biological soil crusts: structure, function, and management. Berlin, Germany: Springer-Verlag: 386–397.

Jungblut, A.D.; Lovejoy, C.; Vincent, W.F. 2010. Global distribution of cyanobacterial ecotypes in the cold biosphere. The ISME Journal 4: 191–202.

Jungblut, A.D.; Vincent, W.F.; Lovejoy, C. 2012. Eukaryotes in Arctic and Antarctic cyanobacterial mats. FEMS Microbial Ecology. 82: 416–428.

Kade, A.; Warren, S.D. 2002. Soil and plant recovery after historic military disturbances in the Sonoran Desert, USA. Arid Land Research and Management. 16: 231–243.

Kellogg, C.A.; Griffin, D.W. 2006. Aerobiology and the global transport of desert dust. Trends in Ecology and Evolution. 21: 638–644.

Kleiner, E.F.; Harper, K.T. 1972. Environment and community organization in grasslands of Canyonlands National Park. Ecology. 53: 299–309.

Kubeková, K.; Johansen, J.R.; Warren, S.D.; [et al.]. 2003. Development of immobilized cyanobacterial amendments for reclamation of microbiotic soil crusts. Algological Studies. 109: 341–362.

Kvíderová, J. 2012. Research on cryosestic communities in Svalbard: the snow algae of temporary snowfields in Petuniabukta, Central Svalbard. Czech Polar Reports. 2: 8–19.

Laaka-Lindberg, S.; Korpelainen, H.; Pohjamo, M. 2003. Dispersal of asexual propagules in bryophytes. The Journal of Hattori Botanical Laboratories. 93: 319–330.

Lange, W. 1974. Chelating agents and blue-green algae. Canadian Journal of Microbiology. 20: 1311–1321.

Lange, W. 1976. Speculations on a possible essential function of the gelatinous sheath of blue-green algae. Canadian Journal of Microbiology. 22: 1181-1185.

Leavitt, S.D., Lumbsch, H.T. 2016. Ecological biogeography of lichen-forming fungi. In: Druzhinina, I.S.; Kubicek, C.P., eds. Environmental and microbial relationships. Cham, Switzerland: Springer International Publishing: 15–37.

Lee, H.N.; Igarashi, Y.; Chiba, M.; [et al.]. 2006. Global model simulation of the transport of Asian and Saharan dust: total deposition of dust mass in Japan. Water, Air, and Soil Pollution. 169: 137–166.

Loope, W.L.; Gifford, G.F. 1972. Influence of soil microfloral crust on select properties of soils under pinyon-juniper in southeastern Utah. Journal of Soil and Water Conservation. 27: 164–167.

Macedo, M.F.; Miller, A.Z.; Dionísio, A.; [et al.]. 2009. Biodiversity of cyanobacteria and green algae on monuments in the Mediterranean Basin: an overview. Microbiology. 155: 3476–3490.

Maestre, F.T.; Martín, N.; Díez, B.; [et al.]. 2006. Watering, fertilization, and slurry inoculation promote recovery of biological crust function in degraded soils. Microbial Ecology. 52: 365–377.

Marshall, W.A.; Chalmers, M.O. 1997. Airborne dispersal of Antarctic terrestrial algae and cyanobacteria. Ecography. 20: 585–594.

McCalla, T.M. 1946. Influence of some microbial groups on stabilizing soil structure against falling water drops. Soil Science Society of America Proceedings. 11: 260–263.

McGorum, B.C.; Pirie, R.S.; Glendinning, L.; [et al.]. 2015. Grazing livestock are exposed to terrestrial cyanobacteria. Veterinary Research. 46: 16.

McLeman, R.A.; Dupre, J.; Ford, L.B.; [et al.]. 2014. What we learned from the Dust Bowl: lessons in science, policy, and adaptation. Population and Environment. 35: 417–440.

Meier, F.C.; Lindbergh, C.A. 1935. Collecting microorganisms from the Arctic atmosphere: with field notes and material. The Scientific Monthly. 40: 5–20.

Mendoza-Aguilar, D.O.; Cortina, J.; Pando-Moreno, M. 2014. Biological soil crust influence on germination and rooting of two key species in a *Stipa tenacissima* steppe. Plant and Soil. 375: 267–274.

Metting, B. 1981. The systematics and ecology of soil algae. Botanical Review. 47: 195-312.

Metting, B. 1991. Biological soil features of semiarid lands and deserts. In: Skujiņš, J., ed. Semiarid lands and deserts. New York: Marcel Dekker: 257–293.

Metzger, S.M.; Balme, M.R.; Towner, M.C.; [et al.]. 2011. *In situ* measurements of pload and transport in dust devils. Icarus. 214: 766–772.

Nickling, W.G. 1978. Eolian sediment transport during dust storms: Slims River Valley, Yukon Territory. Canadian Journal of Earth Sciences. 15: 1069–1084.

Osborn, B. 1952. Range soil conditions influence water intake. Journal of Soil and Water Conservation. 7: 128–132.

Pando-Moreno, M.; Molina, V.; Jurado, E.; [et al.]. 2014. Effect of biological soil crusts on the germination of three plant species under laboratory conditions. Botanical Sciences. 92: 273–279.

Park, C-H.; Li, X.R.; Zhao, Y.; [et al.]. 2017. Rapid development of cyanobacterial crust in the field for combating desertification. PLoS ONE. 12(6): e0179903.

Piñeiro, R.; Popp, M.; Hassel, K.; [et al.]. 2012. Circumarctic dispersal and long-distance colonization of South America: the moss genus Cinclidium. Journal of Biogeography. 39: 2041–2051.

Prospero, J.M. 1999. Long-range transport of mineral dust in the global atmosphere: Impact of African dust on the environment of the southeastern United States. Proceedings of the National Academy of Science USA. 96: 3396–3403.

Prospero, J.M.; Lamb, P.J. 2003. African droughts and dust transport to the Caribbean: climate change implications. Science. 302: 1024–1027.

Prospero, J.M.; Mayor-Bracero, O.L. 2013. Understanding the transport and impact of African dust on the Caribbean Basin. Bulletin of the American Meteorological Society. September 2013: 1329–1337.

Rajeev, L.; Nunes da Rocha, U.; Klitgord, N.; [et al.]. 2013. Dynamic cyanobacterial response to hydration and dehydration in a desert biological soil crust. International Society for Microbial Ecology Journal. 7: 2178–2191.

Reichert, R.C.; Skujiņš, J.; Sorenson, D.; [et al.]. 1978. Nitrogen fixation by lichens and free-living microorganisms in deserts. In: West, N.E.; Skujiņš, J., eds. Nitrogen in desert ecosystems. Stroudsburg, PA: Dowden, Hutchinson and Ross: 20–30.

Rosentreter, R.; Eldridge, D.D.; Kaltenecker. 2001. Monitoring and management of biological soil crusts. In: Belnap, J.; Lange, O.L., eds. Biological soil crust: structure, function, and management. Berlin, Germany: Springer-Verlag: 457–468. https://www.ntc.blm.gov/krc/uploads/643/2001%20Monitoring%20and%20management%20of%20biological%20soil%20crusts--handout.pdf (9 Dec. 2019).

Samad, L.K.; Adhikary, S.P. 2008. Diversity of micro-algae and cyanobacteria on building facades and monuments in India. Algae. 23: 91–114.

Schlesinger, W.H. 1991. Biogeochemistry: an analysis of global change. New York: Academic Press.

Schlichting, H.E. 1969. The importance of airborne algae and protozoa. Journal of the Air Pollution Control Association. 19: 946–951.

Schulten, J.A. 1985. Soil aggregation by cryptogams of a sand prairie. American Journal of Botany. 72: 1657–1661.

Serpe, M.D.; Orm, J.M.; Barkes, T.; [et al.]. 2006. Germination and seed water status of four grasses on moss-dominated biological soil crusts from arid lands. Plant Ecology. 2006: 163–178.

Sethi, S.K.; Samad, L.K.; Adhikary, S.P. 2012. Cyanobacteria and micro-algae in biological crusts on soil and sub-aerial habitats of eastern and northeastern region of India. Phycos. 42: 1–9.

Sharma, N.K.; Rai, A.K.; Singh, S.; [et al.]. 2007. Airborne algae: Their present status and relevance. Journal of Phycology. 43: 615–627.

Song, G.; Li, X.; Hui, R. 2017. Effect of biological soil crusts on seed germination and growth of an exotic and two native plant species in an arid ecosystem. PLoS ONE 12(10): e0185839.

Stark, L.R. 2003. Mosses in the desert. Fremontia. 31: 26–33.

St. Clair, L.L.; Johansen, J.R.; Webb, B.L. 1986. Rapid stabilization of fire-disturbed sites using a soil crust slurry: inoculation studies. Reclamation and Rehabilitation Research. 4: 261–269.

Steyn, P.L.; Delwiche, C.C. 1970. Nitrogen fixation by nonsymbiotic micro-organisms in some California soils. Environmental Science and Technology. 4: 1122–1128.

Takeuchi, N. 2013. Seasonal and altitudinal variations in snow algal communities on an Alaskan glacier Gulkana glacier in the Alaska Range. Environmental Research Letters. 8: 035002.

Tesson, S.V.; Skjøth, C.A.; Šanti-Temkiv, T.; [et al.]. 2016. Airborne microalgae: Insights, opportunities, and challenges. Applied and Environmental Microbiology. 82: 1978–1991.

Tisdall, J.M.; Oades, J.M. 1982. Organic matter and water-stable aggregates in soils. Journal of Soil Science. 33: 141–163.

Tomaselli, L.; Lamenti, G.; Bosco, M.; [et al.]. 2000. Biodiversity of photosynthetic micro-organisms dwelling on stone monuments. International Biodeterioration and Biodegradation. 46: 251–258.

Tormo, R.; Recio, D.; Silva, I.; [et al.]. 2001. A quantitative investigation of airborne algae and lichen soredia obtained from pollen traps in southwest Spain. European Journal of Phycology. 36: 385–390.

Uno, I.; Eguchi, K.; Yumimoto, K.; [et al.]. 2009. Asian dust transported one full circuit around the globe. Nature Geoscience. 2: 557–560.

Vonnahme, T.R.; Devetter, M.; Źárský, J.D.; [et al.]. 2016. Controls on microalgal community structures in cryoconite holes upon high-Arctic glaciers, Svalbard. Biogeosciences. 13: 659–674.

Wang, X.; Dong, Z.; Zhang, J. Liu, L. 2004. Modern dust storms in China: an overview. Journal of Arid Environments. 58: 559–574.

Warren, S.D. 1995. Ecological role of microphytic soil crusts in arid environments. In: Allsopp, D.; Caldwell, R.R.; Hawksworth, D.L., eds. Microbial diversity and function. Wellingford, UK: CAB International. 199–209.

Warren, S.D. 2001. Synopsis: influence of biological soil crusts on arid land hydrology and soil stability. In: Belnap, J.; Lange O.L., eds. Biological soil crusts: Structure, function, and management. Berlin, Germany: Springer-Verlag: 349–360.

Warren, S.D. 2014. Role of biological soil crusts in desert hydrology and geomorphology: implications for military training operations. Reviews in Engineering Geology. 22: 177–186

Warren, S.D.; Eldridge, D.J. 2001. Biological soil crusts and livestock in arid ecosystems: are they compatible? In: Belnap J.; Lange O.L., eds. Biological soil crusts: structure, function and management. Berlin, Germany: Springer-Verlag: 401–415.

Warren, S.D.; St. Clair, L.L.; Johansen, J.R.; [et al.]. 2015. Biological soil crust response to late season prescribed fire in a Great Basin juniper woodland. Rangeland Ecology and Management. 68: 241–247.

Warren, S.D.; St. Clair, L.L.; Leavitt, S.D. 2019a. Aerobiology and passive restoration of biological soil crusts. Aerobiologia. 35: 45–56.

Warren, S.D.; St. Clair, L.L., Stark, L.R.; [et al.]. 2019b. Reproduction and dispersal of biological soil crust organisms. Frontiers in Ecology and Evolution. 7: 344.

Webb, R.H.; Steiger, J.W.; Newman, E.B. 1988. The response of vegetation to disturbance in Death Valley National Monument, California. U.S. Geological Survey Bulletin 1793. Reston, VA: Department of the Interior, U.S. Geological Survey.

Weber, B.; Büdel, B.; Belnap, J., eds. 2016. Biological soil crusts: an organizing principle in drylands. Cham, Switzerland: Springer International Publishing.

West, N.E. 1978. Physical inputs of nitrogen to desert ecosystems. In: West, N.E.; Skujiņš. J., eds. Nitrogen in desert ecosystems. Stroudsburg, PA: Dowden, Hutchinson and Ross. 165-170

West, N.E. and Skujiņš, J. 1977. The nitrogen cycle in North American cold-winter semi-desert ecosystems. Oecologia Plantarum. 12: 45–53.

Wilshire, H.G. 1983. The impact of vehicles on desert stabilizers. In: Webb, R.H.; Wilshire, H.G., eds. Environmental effects of off-road vehicles. New York: Springer-Verlag New York: 31–50.

Womack, A.M.; Bohannan, B.J.M.; Green, J.L. 2010. Biodiversity and biogeography of the atmosphere. Transactions of the Royal Society. 365: 3645–3653.

Xu, S.; Yin, C.; He, M.; [et al.]. 2008. A technology for rapid reconstruction of moss-dominated soil crusts. Environmental Engineering Science. 25: 1129–1137.

Zhao, Y.; Bowker, M.A.; Zhang, Y.; [et al.]. 2016. Enhanced recovery of biological soil crusts after disturbance. In: Weber, B.; Büdel, B.; Belnap, J., eds. Biological soil crusts: an organizing principle in drylands. Cham, Switzerland: Springer International Publishing: 499–523.

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,

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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.

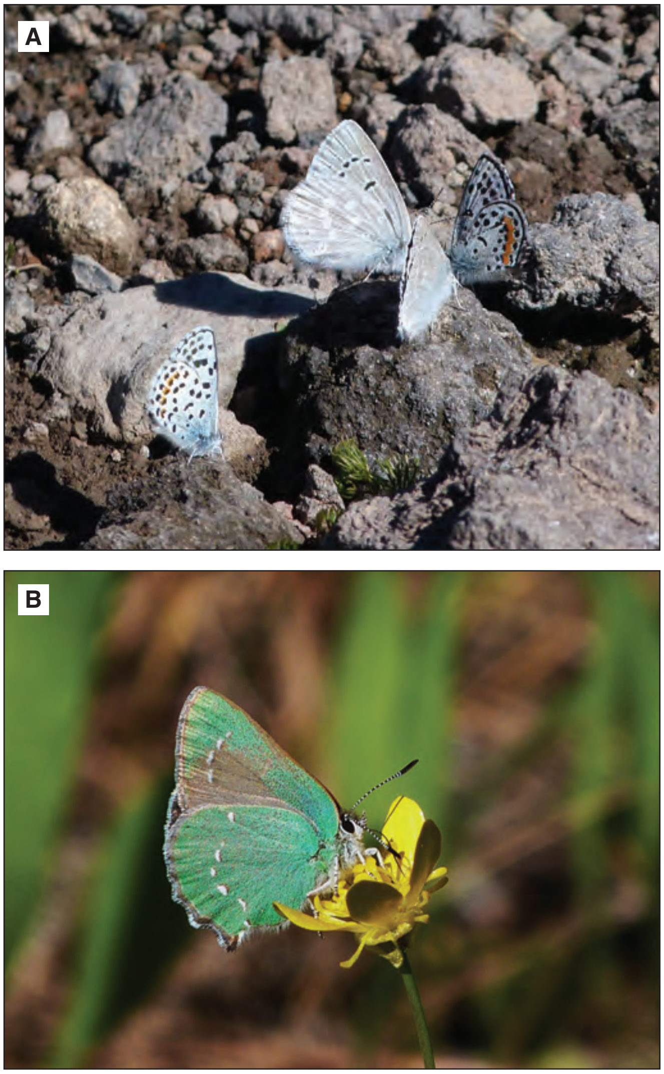

Figure 4.1.2—Butterflies, including these (A) diminutive common "blues" (*Lycaenidae*), prefer habitat that provides them moist, warm, and sunny locations. Such habitats are often associated with riparian zones and proper management can encourage suitable habitat for a wide range of flora and fauna, 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 due to containing a relatively large volume of sapwood (Farris and Zack 2005).

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).

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).

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).

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 (*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.

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).

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).

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

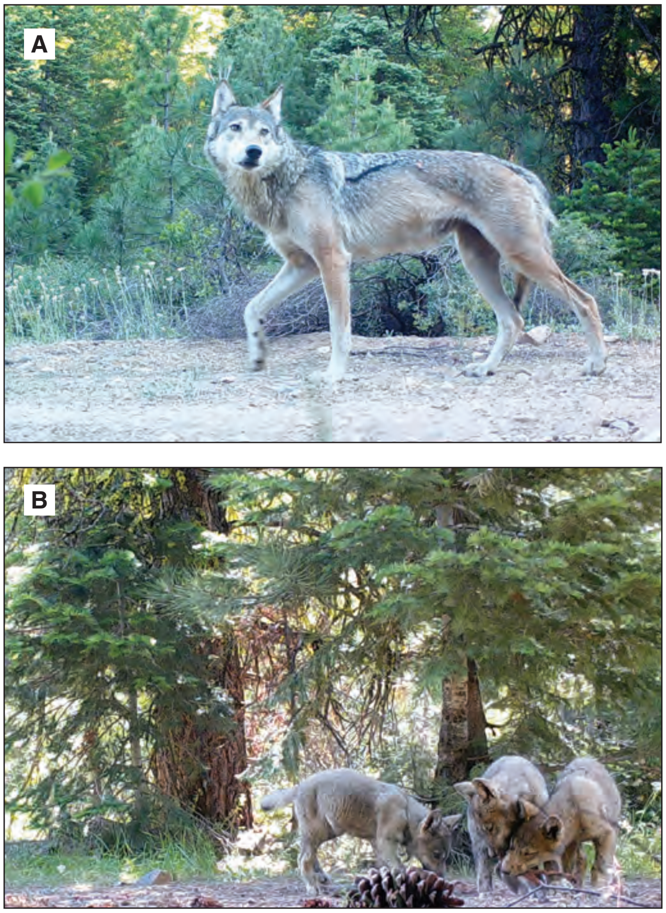

Figure 4.1.7—Trail camera photos of (A) a female gray wolf in Lassen County in 2017 and (B) three of her pups (photos by Tom Rickman, Forest Service).

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.

Chapter 4.2. Aquatic Ecosystems, Vernal Pools, and Other Unique Wetlands

Pamela E. Padgett[1]

Introduction

Part of the rationale for the *Northeastern California Plateaus Bioregion Science Synthesis* is to more fully address portions of the Lassen and Modoc National Forests (hereafter the Lassen and the Modoc) representative of the Great Basin ecosystem (see Chapter 1.1, Dumroese, this synthesis, *The Northeastern California Plateaus Bioregion Science Synthesis: Background, Rationale, and Scope*). Although the Great Basin can be defined floristically, it can also be defined hydrologically as the area in the Western United States that is internally drained; in other words, precipitation does not ultimately flow to the oceans, but remains in the basin. Geologic changes since the Pleistocene (about 11,700 years before present) have led to a drying-out of the region. What was an area of extensive wetlands and marshes then is semi-desert today (Currey 1990). The recent hydrologic patterns have created unique aquatic habitats, especially vernal pools.

The wide annual fluctuation in the presence of water creates unique habitat conditions that drive unique adaptations of the flora and fauna. Many plants and animals are able to survive long dry periods, which makes them resilient to the potential changes in precipitation in the future. But many species also exist under a delicate balance of physiologic responses to environmental change (Steward et al. 2012). Trampling by livestock, vehicle passage, and trails created by foot traffic destroy refugia for quiescent and hibernating species. Vernal pools have been drained, or in some cases flooded to provide water for livestock (USFWS 2005). Fens have been drained or rechanneled, and intermittent stream channels have been used as irrigation ditches ultimately reducing, or eliminating, viable habitat for uniquely adapted creatures (Larned et al. 2010; Roche et al. 2012). While vernal pools have been fairly well described in the literature, much less is known about vernal wetlands and ecology and the importance of intermittent streams.

Other Relevant Science Syntheses

Portions of the water resources encompassed by the Lassen and the Modoc in Northeastern California have already been the focus of two recent science syntheses. The first is the *Science Synthesis to Support Socioecological Resilience in the Sierra Nevada and Southern Cascade Range* (Long et al. 2014), which has an entire section (Section 6) focused on water resources and aquatic ecosystems in the national forests of the Sierra Nevada. Chapter 6.1 – *Watershed and Stream Ecosystems* (Hunsaker et al. 2014) discusses factors that affect water quality and quantity. Chapter 6.2 – *Forested Riparian Areas* (Hunsaker and Long 2014) takes a broad look at these systems and in particular how they are affected by fire and grazing. Chapter 6.3 – *Wet Meadows* (Long and Pope 2014), focuses on grazing within herbaceous wetlands having fine-textured soils and shallow groundwater tables in summer. Chapter 6.4 – *Lakes: Recent Research and Restoration Strategies* (Pope and Long 2014) discusses the high social value of these resources, stressors on them, and ways to assess resilience and resistance; this discussion is pertinent to lakes within and adjacent to the Lassen and Modoc, such as Clear Lake (fig. 4.2.1). And, Chapter 9.5 – *Managing Forest Products for Community Benefit* (Charnley and Long 2014) includes substantial discussion about grazing in riparian areas. The second synthesis with relevance to the Lassen and the Modoc is the *Synthesis of Science to Inform Land Management within the Northwest Forest Plan Area* (Spies et al. 2018), which, in Chapter 7, lays out an aquatic conservation strategy to protect biodiversity, especially fish (Reeves et al. 2018).

Most of the information contained in those five chapters directly applies to the Modoc Plateau and the Warner Mountains of Northeastern California, including the predictions for precipitation changes due to climate change. Where the Modoc Plateau departs from the Pacific Northwest and most of the Sierra landscape is in the greater number of ephemeral and intermittent streams and wetlands and many closed-basin hydrologic systems. Much of the area is more akin to the Eastern Sierra than the Western or Northwestern parts of the range. Alkali lakes and playas (dry lakes) are a significant component of the Modoc Plateau aquatic ecosystems (Snyder 1962). "Wet meadows" are more likely to be ephemeral, where standing water is seasonal, only evident during some part of the year, or vernal wet meadows where subsurface water creates a temporary wet meadow ecosystem during the wet season, but dries out during the summer.

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Lakes, Streams, and Fish

The surface drainage on the Modoc Plateau is highly variable. It ranges from highly porous volcanic rock with rapid infiltration of precipitation and snowmelt to areas where soils are highly impermeable and intermittent pools and marshes are notable features. Seeps are common and large springs are numerous (fig. 4.2.2.A), even in areas of light precipitation. The principal streams draining the Modoc Plateau are the Pit River and its tributaries, and the headwater tributaries of the Klamath River (fig. 4.2.2.B). On the Modoc, two ranger districts, the Doublehead and the Devil's Garden, generally lack hydrologic connectivity of their stream networks. There are few, if any, perennial streams, but a relatively large number of seasonally flowing streams that drain into reservoirs, stock ponds, or other depressions in the ground (swales) and act to recharge the groundwater table. Streams on the western slopes of the Warner Mountains drain to Goose Lake or the Pit River (fig. 4.2.2.C). Streams on the eastern side are much shorter, draining into Surprise Valley, and most of these streams have no outlet. Goose Lake once drained into the north fork of the Pit River, but the lake level in modern times has been below the outlet. Interestingly, Goose Lake has a history of drying up completely, as recorded in 1852, 1926, 1929–34, and in 1992. A series of fens, meadows, and other special aquatic features occur up and down both sides of the Warner Mountain Range (Sikes et al. 2013) (fig. 4.2.2.D).

Of the nearly 2,000 miles (3,219 km) of total stream length on the Lassen, nearly two-thirds (1,442 miles [2,321 km]) are intermittent streams that do not sustain year-round flows (USDA 2010a). Lakes and streams on the Lassen are equally diverse. Eagle Lake, the second largest natural lake entirely within California, is a closed basin that lies near the junction of the three provinces, the Cascades, the Sierra Nevada, and the Great Basin. Lakes Almanor and Britton are two large reservoirs in the Feather River and Pit River

Figure 4.2.1—Like many water bodies in the Lassen and Modoc, Clear Lake within the Clear Lake National Wildlife Refuge, is a manmade reservoir originally created for water management and irrigation. Now it is a prime recreation destination (photo by Modoc National Forest).

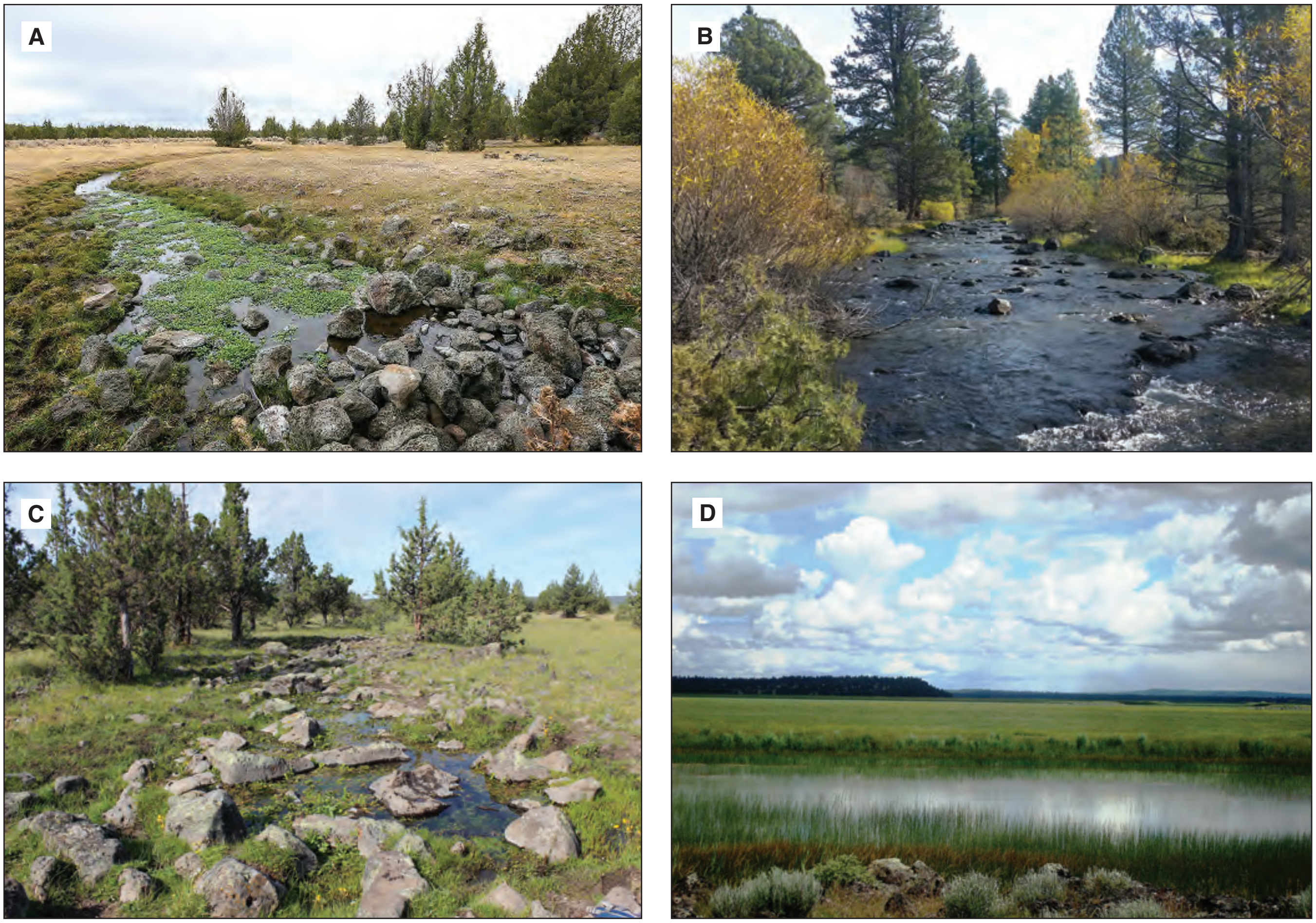

Figure 4.2.2—The Modoc Plateau has many unique aquatic features. Springs and fens are fed by ground water and support plants that may occur nowhere else. They are also magnets for wildlife. (A) Warm Valley spring (photo by Shaun Hunger, used with permission). (B) South Fork Pit River on the Modoc National Forest flows to the main-stem Pit River, which is an important segment of the Sacramento River watershed (photo by Ken Sandusky, Forest Service). (C) Goose Lake Creek (photo by Laura Snell, UC Cooperative Extension, used with permission) where the rocky landscapes keep the banks intact from livestock, wild horses, and wildlife. (D) Fairchild Swamp, a permanently wet ecosystem in the Modoc National Forest, is, like other wetlands, an important water resource for a variety of flora and fauna (photo by John Cichoski, Forest Service).

watersheds, respectively. In addition, numerous small alpine and pothole lakes dot the landscape, many located in the Caribou and Thousand Lakes Wildernesses. Lands east of the Cascades are relatively dry and drain eastward through two main streams, Pine Creek (to Eagle Lake) and the Susan River (to Honey Lake). The dry lands of the Modoc Plateau to the north drain westward through the Pit River, a tributary of the Sacramento River. In these drier areas, there are natural seasonal flowing streams and ponds as well as developed stock ponds, wildlife ponds, and small reservoirs designed to capture limited stream flow and snow runoff. Many of these ponds and reservoirs dry up each season as water slowly evaporates or percolates down through the porous substrate. The west side of the Lassen is much wetter and has many streams that flow to the Sacramento River. These include Battle Creek, Antelope Creek, Mill Creek, Deer Creek, Butte Creek, and the North Fork of the Feather River (fig. 4.2.3).

Several fish species are endemic to the Lassen and the Modoc and two are listed as threatened or endangered species: shortnose sucker (*Chasmistes brevirostris*) and Lost River sucker (*Deltistes luxatus*). These are described in the Modoc Travel Management Plan (USDA 2010b). Additionally, two subspecies of rainbow trout—one of each forest—are of special concern (see textbox 4.2.1). The first, the Eagle Lake rainbow trout (*Oncorhynchus mykiss aquilarum*), is endemic to the Eagle Lake watershed, residing in the lake and primarily Pine Creek. The second is the Goose Lake trout (*Oncorhynchus mykiss* pop 6), which

Figure 4.2.3—Mill Creek flows from the southern slopes of Lassen Peak to the Sacramento River and supports the highest-elevation spawning areas for spring-run Chinook salmon in California (photo by John Cichoski, Forest Service).

is endemic to Goose Lake and its major tributaries (Lassen and Willow Creeks in California and the extensive Thomas Creek system and Crane Creek in Oregon), as well as to smaller streams such as Cottonwood Creek in California and several small streams in Oregon (fig. 4.2.4).

Textbox 4.2.1

"Native rainbow trout east of the Cascades are commonly called 'redband trout' (*Oncorhynchus mykiss* ssp.). Redband trout are a primitive form of rainbow trout and are an evolutionary intermediate between ancestral 'cutthroat'-like species and coastal rainbow trout. Redband trout are described as inland populations of *O. mykiss,* with few morphological and meristic characters distinguishing them from coastal rainbow trout. Although there is no consensus on the classification of redband trout east of the Cascades, there is some agreement that at least two broad groups exist in Oregon: the Interior Columbia Basin redband trout and the Oregon Great Basin redband trout. In addition, redband trout in the upper Klamath Basin (e.g., Sprague and Williamson Rivers) represent a third evolutionary group within Oregon. Populations of redband trout in the Great Basin have been isolated for thousands of years and therefore evolved distinct genetic lineages (ancestries)." (USFWS 2009)

Figure 4.2.4—Goose Lake trout, an endemic species to its namesake and the Lassen and Willow Creeks on the Modoc National Forest, is a Forest Service Sensitive Species. While ongoing management efforts to improve water quality aid the species, changes in land use and implementation of water-use policies challenge this species' status (photo by Ken Sandusky, Forest Service).

Given the isolation, both strains of rainbow trout are now considered unique subpopulations and are threatened by implementation of changes in land use and water-use policy. The Goose Lake is on the northern border with Oregon, thus conservation strategies are shared between California and Oregon (see textbox 4.2.2). The lake is a closed basin, therefore it continues to increase in alkalinity and salinity. The redband trout native to Goose Lake has evolved specialized adaptations to cope with alkalinity and salinity. Conservation is largely driven by a two-State, cooperative management group, the Goose Lake Watershed Council. A Conservation Strategy was prepared in 1996, and that document (Heck et al. 2008) continues to guide conservation efforts by maintaining or improving water quality and quantity while striving to maintain a balance between competing uses such as agriculture and recreation. It also calls for protecting and increasing habitat conditions for the trout to provide spawning, resting, and foraging habitat.

The Eagle Lake redband trout are long-lived relative to other rainbows, and are considered among the hardiest of the subspecies, well-adapted to the harsher conditions of an inland lake. The population within the lake is relatively robust, but has only one remaining spawning stream—Pine Creek. The most recent report on the status of the Eagle Lake redband is in the *State of the Salmonids: Status of California's Emblematic Fishes* (Moyle et al. 2017), commissioned by California Trout. The report calls the fishery unsustainable largely because of inconsistent access to Pine Creek by spawning trout (fig. 4.2.5), and

Textbox 4.2.2

"Since 1995, conditions for Goose Lake Redband trout in California have steadily improved because large sections of Lassen Creek and other streams have increased protection from grazing due to changes in [Forest Service] allotments and otherwise been restored. These conservation measures have likely improved habitat conditions, which can benefit runs of lake fish to re-establish themselves when hydrologic conditions are favorable. Presumably, headwater populations have increased as well, thanks to better management. Recent habitat improvements in Oregon actually led to an expansion of the distribution of the species from 1995 to 2007, according to ODFW surveys (Scheerer et al. 2010)." Moyle et al. (2017)

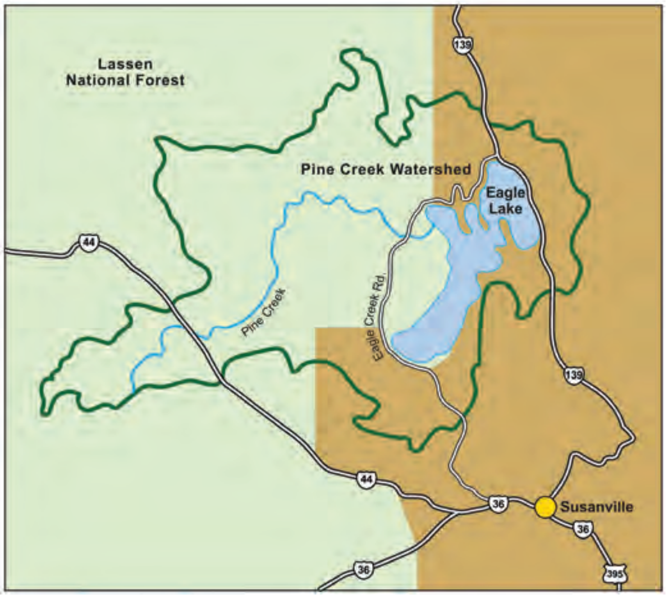

Figure 4.2.5—The Pine Creek Watershed provided critical habitat to sustain Eagle Lake redband trout, but inconsistent access to Pine Creek now requires this species to be reared in hatcheries (adapted from California Department of Fish and Wildlife 2019).

the need to rear redbands in hatcheries. As of 2017, most fish that move up into Pine Creek to spawn are trapped at a weir and artificially spawned. Eggs are hatched and fry are reared at several hatcheries in California. While this has maintained the adequate populations in the lake, without human intervention the long-term success of the subspecies is in doubt because the species is not self-sustaining. While few fish species that have been introduced over the years are able to withstand the alkalinity of the lake, the introduction of brook trout to Pine Creek in the 1930s and ’40s has greatly limited the redband’s reproductive success. Redband trout do not compete well with the Eastcoast native, and mature brook trout are major predators of redband fry (Moyle et al. 2017).

Vernal Pools and Other Unique Aquatic Habitat

Vernal pools are seasonal wetlands that do not follow the classic geologic successional pattern where ponds accumulate organic material over time to become wetlands that further fill in to become to meadows and eventually forests. Vernal pools rarely have water sources other than precipitation, therefore no surface water inlets or outlets (fig. 4.2.6). They are formed over unusually impermeable soils and rocks which enables precipitation to accumulate during the rainy season and evaporate as the temperatures warm. However, many of the natural processes that lead to permanent intermittent pools are not completely understood (Norwick 1991).

Vernal wet meadows have similar subsoils, but seasonal water remains below the surface. While intermittent pools and wetlands are not unique to the Lassen and the Modoc, the number and the importance of these distinctive habitat types across the landscape does require a thoughtful examination of the significance to management decisions. An estimated 80 to 90 percent of the vernal pools in the Central Valley of California have been lost to various agricultural activities such as drainage, ditching, and subsoil disturbances (Holland 1998). On the Modoc, Holland (2006) mapped 660 vernal pools, of which more than 30 percent had some indication of hydrologic disturbance. Much of knowledge related to vernal pools has been gathered from the Central Valley complexes. While it is likely that the geomorphic processes are similar between valley and upland, or montane vernal pools, Bovee et al. (2018) suggests that there may be significant differences in the biological components and processes. Once lost, constructing functioning vernal pools de novo or restoring degraded pools is challenging (Lamers et al. 2015; Schlatter et al. 2016). Department of the Interior, U.S. Fish and Wildlife Service, signed a recovery plan for vernal pools (USFWS 2005) for California and Southern Oregon, although progress has been slow and the success rates low (Schlatter et al. 2016). Additional discussion about vernal pools from a climate change perspective can be found in

Figure 4.2.6—Vernal pools, such as this one on the west side of the Warner Mountains, occur where impermeable soil conditions allow shallow pools of water to occur, although they may dry out during summer. These unique wetlands host a variety of plants and animals, some of which depend on these features (photo Gary Nafis, used with permission).

Chapter 6.1, Wright, this synthesis, *Ecological Disturbance in the Context of a Changing Climate: Implications for Land Management in Northeastern California*.

"Swales" are a form of intermittent wetland. They may connect pools or stand alone and are generally of shorter inundation. They differ from vernal pools in that after filling from winter precipitation, the surface water slowly infiltrates into the underlying substrate rather than evaporating, as is typical of vernal features. Swales may be natural or manmade. Many infiltration ponds constructed for groundwater recharge are constructed swales.

Unlike vernal pools and swales, "fens" are wet year-round (fig. 4.2.7). They are aquatic features that are fed by groundwater seeps. Like bogs, they accumulate peat over hundreds to thousands of years. They differ from bogs, however, in that they are groundwater-dependent rather than relying on precipitation. Bogs do not occur in regions with pronounced dry seasons, such as California (Lamers et al. 2015; Sikes et al. 2013). Fens are among the most sensitive habitat types in the Sierra Nevada Ecoregion (Sikes et al. 2013). They often contain unique assemblages of plant species and those assemblages can vary widely from fen to fen, making vegetation classification difficult (Patterson and Cooper 2007). The Warner Mountains on the Modoc are particularly rich in fens and the rare plants associated with them (Sikes et al. 2013; USDA FS 2010b, p. 138). Holland (2006) mapped 132 fens on the Modoc. Sixty fens have been identified by ground surveys on the Lassen, which represents approximately 80 percent of the total estimated in the Travel Management Plan (USDA 2010a). The Forest Service Pacific Southwest Region, in collaboration with the California Native Plant Society and many others, published the *Fen Conservation and Vegetation Assessment in the National Forests of the Sierra Nevada and Adjacent Mountains, California* in 2010 and revised in 2013 (Sikes et al. 2013). Fens are generally small geographic features (less than a hectare) and often part of larger meadow complexes. They are difficult to identify from satellite or aerial technologies (Holland

Figure 4.2.7—Fens, such as this one on the Antelope Plains of the Modoc National Forest, are important aquatic features for a variety of flora and fauna, but are also the most sensitive to degradation. Fens are supported by groundwater and often host unique plant communities. While this image shows open water, most fens are not easily detected, as water is often obscured by accumulation of peat (photo by John Cichoski, Forest Service).

2006), and can be missed, or mis-identified during land surveys (Sikes et al. 2013). Therefore, distribution, natural history, and ecology of these features is largely unknown (Chimner et al. 2010). Because they are a year-round water source in a dry climate, they are attractive to wildlife, livestock, and people. By definition, fens require decades, if not centuries, to accumulate their characteristic peat base. Hooves, tire tracks, and modification of hydraulic patterns can easily degrade fens and the associated flora and fauna. Similar to vernal water bodies, once damaged, fen restoration is difficult (Lamers et al. 2015).

Research is scant on intermittent or ephemeral streams in the Great Basin. These water features are a significant component of the aquatic resources on the Lassen and the Modoc. According to the 2011 Perennial Stream assessment (SWAMP 2011) 78 percent of the streams on the Modoc Plateau are nonperennial, and as noted earlier in this chapter, the Lassen has 1,442 miles of intermittent streams as compared to 558 miles of perennial streams (USDA 2010a). Projections for climate change suggest that the miles of intermittent streams are likely to increase with time (Larned et al. 2010; Datry et al. 2016). Similar to vernal features, the ecology of intermittent streams requires organisms to survive periods of desiccation and periods of inundation. They can also provide seasonal connectivity between more permanent aquatic features (Leigh et al. 2016). The confusion in legal classification of intermittent streams has contributed to the paucity of data and information. They do not generally fall within the network of aquatic resource rules and regulations. And while threatened, endangered, or endemic species may be associated with them, very little has been published or recorded about the flora and fauna of nonperennial waterways (Leigh et al. 2016). What has been reported regarding intermittent streams in the United States is derived from research in the Southwest. While many of the general principles undoubtedly apply, little is known about species richness for systems outside the Southwest, and the unique survival mechanics of organisms subjected to the combination of hot summer temperatures and cold winters typical of the Great Basin.

All of these hydrologic features are highlighted by unique flora and fauna, often existing nowhere else but in a single pool, or series of pools, stream reach, or montane fen (fig. 4.2.8). In vernal pools and wetlands, the flora is dominated by low-growing species of annual grasses and forbs adapted to germination and early growth under water, and completing their life cycle during the dry period. These organisms have evolved mechanisms that allow them to remain dormant under very harsh conditions for several years if necessary, as precipitation volume varies widely from year to year, and vernal pools and wetlands may remain dry for a few years in a row before a wet winter refills the basin. These small aquatic features often contain plant and animal species that occur nowhere else, and many of which have been preserved in place for hundreds of thousands of years (Norwick 1991). In vernal pools on the Lassen, slender Orcutt (*Orcuttia tenuis*), known to occur in 22 locations, and Green's tuctoria (*Tuctoria greenei*), found in only one location, are federally listed (threatened and endangered, respectively). These two grasses also occur on the Modoc in vernal pools and vernally wet areas in the Devil's Garden Ranger District, and within ponderosa pine forests of the Big Valley Ranger District and parts of the Doublehead Ranger District. Along with these two grass species, five other plant species are listed as species of conservation concern (see definition above) in vernal pools, swales, and wetlands: disappearing monkeyflower (*Erythranthe inflatula;* syn = *Mimulus evanescens*), playa yellow phacelia (*Phacelia inundata*), Boggs Lake hedgehyssop (*Gratiola heterosepala*), profuseflower mesamint (*Pogogyne floribunda*), Red Bluff

Figure 4.2.8—Many unusual plants, such as this sundew (*Drosera anglica*) growing on the Lassen National Forest, are obligate wetland plants (photo by Craig Odegard, Forest Service).

dwarf rush (*Juncus leiospermus* var. *leiospermus*), woolly meadowfoam (*Limnanthes floccosa* ssp. *bellingeriana*), and Newberry's cinquefoil (*Potentilla newberryi*).

The threats to special aquatic habitat are two-fold: (1) surface disturbances from unmanaged livestock and wild horse grazing (Merriam et al. 2016), and off-road vehicle use (a particular problem near Redding and in forested areas of the Modoc Plateau); and (2) changing the hydrology by installing drain tile in swales and fens, or year-round flooding of vernal pools and wetlands, or using intermittent streams as irrigation ditches. It is important to note that while invasive plants have a significant impact on vernal features in other parts of the State, particularly in the Central Valley (Faist and Beals 2017), they comprise a small fraction of the plant cover in the vernal pools of the Modoc.

The effects of cattle grazing on vernal pools and wetlands is complicated (Bovee et al. 2018; Merriam et al. 2016; see also Chapter 6.1, Wright, this synthesis, *Ecological Disturbance in the Context of a Changing Climate: Implications for Land Management in Northeastern California*). While excessive trampling is destructive to some native plants, grazing has been used a means for reducing invasive weed populations, for example (Faist and Beals 2017). However, Bovee et al. (2018) studying grazing effects in the montane vernal pools of the Modoc Plateau found that grazing had stronger effects on vernal pool specialists than on the plants considered "habitat generalists" and favored annuals over perennials (fig. 4.2.9). Much of the ongoing research on conservation of vernal pools has been focused on preservation and restoration of the endangered endemic species, slender Orcutt. Although it has been debatable whether cattle consume slender Orcutt, Merriam et al. (2016) and Bovee et al. (2018) concluded that it was unpalatable and was not eaten even when no other grasses were available. When herds are well managed by monitoring hoofprint coverage and vegetation consumption, they can effectively reduce nondesirable weedy plants and reduce litter accumulation that alters hydraulic characteristics (Marty 2015). However, that may be at the expense of some native perennial species (Merriam et al. 2016).

The effects of grazing in fens and wet meadows are similar to those found for vernal features. However, because fens do not have a dry season, the effects may be of longer duration and more intense. Understanding the effects of cattle on fens is also complicated by the tendency of fens to be incorporated into larger meadow complexes, and the likelihood that many fens may have been lost during the early settlement years (van Diggelen et al. 2006). In many cases, the nature of "pristine" fens is difficult to establish, so the effects of grazing are studied by exclusion treatments. Merriam et al. (2017) in a study on the Plumas National Forest, demonstrated that exclusion of cattle for 8 years greatly changed the plant community composition, favoring tall, high-nutrient-demand graminoids. Much of the change in plant community structure was related to litter and biomass accumulation. Grazing can be effective in reducing the negative consequences of litter accumulation (Middleton et al. 2006).

The Lassen and the Modoc adopted Travel Management Plans (USDA 2010a, b) in 2010 designed to greatly reduce off-road vehicle traffic. Using dry riverbeds and arroyos as roads is very common in the Southwest, and most likely in the sagebrush steppe as well (Levic et al. 2008), although little was noted in any published literature.

Changes in hydrology, draining, channeling, and flooding is mostly a relic of past management activities, and does not seem to be part of current management activities. It appears that the Lassen and the Modoc are engaged in repairing past "improvements," often attempting to identify what the original natural aquatic feature was (USDA and USDOI 2008).

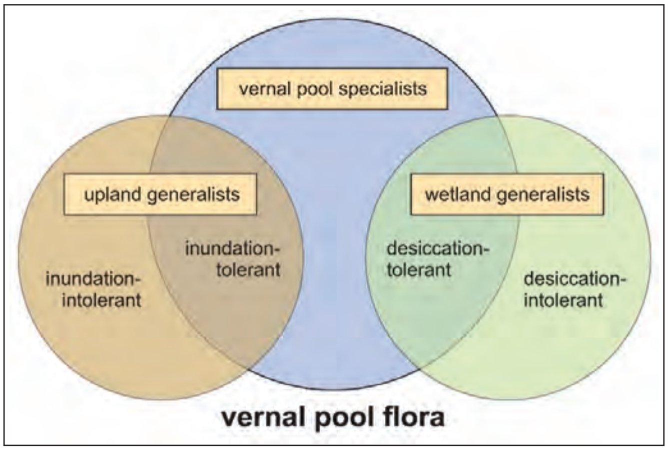

Figure 4.2.9—In vernal pools, plant species can be either endemic (specialists), requiring the vernal pool habitat, or able to grow in vernal pools as well as in other habitats (generalists). Vernal pool specialists are adapted to long periods of complete flooding as well as long periods of desiccation. When the hydrology of a vernal pool is altered, the specialists no longer have suitable habitat and decline, allowing the generalists to take over the site (modified from Bovee et al. 2018).

As noted earlier, restoration of vernal pools and vernally wet areas, in particular, is difficult. Restoration requires the manipulation of the physical, chemical, and biological characteristics of a site with the goal of returning the natural and historic functions to a former or degraded vernal pool (USFWS 2005). Vernal pool restoration may include diverting excess surface runoff (e.g., from agriculture, roads, or other urban hardscapes), reconstructing the characteristic depth from the overlying soil surface to the impermeable layer beneath (e.g., removing silt accumulation from agricultural use or repairing damage due to off-road vehicle use), managing grazing, and/or removing competing species.

The creation of vernal pools, where they had not existed before, is attempted as a means of habitat exchange. Recent literature (Schlatter 2016) indicates, however, that success for construction de novo is very limited. For this reason, preservation must be the fundamental strategy in maintaining vernal pool ecosystems within the planning area (USFWS 2005).

Fens fall into the same restoration category as vernal pools and wetlands—very difficult to accomplish. In order to be labeled a fen, the peat must reach a certain depth, a process that takes centuries. Once peat is removed, the fen is no longer a fen and the organisms reliant on that unique habitat will disappear as well. As presented above with vernal habitat, conservation is the best course of action.

Swales and intermittent streams can be restored and reconstructed. Manmade swales are often constructed in modern times as part of groundwater recharge stations. For example, the U.S. Environmental Protection Agency published a fact sheet explaining to homeowners how to make vegetated swales for storm water infiltration (US EPA 1999). One of the major issues with intermittent streams that have been flooded continuously is the invasion of nonnative species and the loss of native species. The ability to restore the original flora and fauna is poorly researched, but efforts are underway, globally, to improve the understanding of the ecology of intermittent streams (Datry et al. 2016).

References

Bovee, K.M.; Merriam, K.E.; Gosejohan, M.C. 2018. Livestock grazing affects vernal pool specialists more than habitat generalists in montane vernal pools. Applied Vegetation Science. 21: 12–20.

California Department of Fish and Wildlife. 2019. Eagle Lake rainbow trout. https://www.wildlife.ca.gov/Conservation/Fishes/Eagle-Lake-Rainbow-Trout (28 Jan. 2019).

Charnley, S.; Long, J. 2014. Managing forest products for community benefit. In: Long. J.W.; Quinn-Davidson, L.N.; Skinner, C.N., eds. 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: 629–661. Chapter 9.5.

Chimner, R.A.; Lemly, J.M.; Cooper, D.J. 2010. Mountain fen distribution, types and restoration priorities, San Juan Mountains, Colorado, U.S.A. Wetlands. 30: 763–771.

Curry, D.R. 1990. Quaternary palaeolakes in the evolution of semidesert basins, with special emphasis on Lake Bonneville and the Great Basin, USA. Palaeogeography, Palaeoclimatology, Palaeoecology. 76: 189–214.

Datry, T.; Corti, R.; Foulquier, A.; [et al.]. 2016. One for all, all for one: A global river research network. EOS Earth & Space Science News. 97(15): 13–15.

Faist, A.M.; Beals, S.C. 2017. Invasive plant feedbacks promote alternative states in California vernal pools. Restoration Ecology. 26: 255–263.

Heck, M.P.; Scheerer, P.D.; Gunckel, S.L.; [et al.]. 2008. Status and distribution of native fishes in the Goose Lake Basin. Information Report 2008-02. Salem, OR: Oregon Department of Fish and Wildlife. 34p

Holland, R.F. 1998. Great Valley vernal pool distribution, photorevised 1996. In: Witham, C.W.; Bauder E.T.; Belk, D.; [et al.], eds. Ecology, conservation, and management of vernal pool ecosystems—Proceedings from a 1996 Conference. Sacramento, CA: California Native Plant Society: 71–75.

Holland, R.F. 2006. Mapping vernal pools and fens in the Modoc National Forest: a report prepared for the Modoc National Forest. ftp://ftp.dfg.ca.gov/BDB/GIS/BIOS/Public_Datasets/900_999/ds949.zip (15 Jul. 2019).

Hunsaker, C.T.; Long, J.W. 2014. Forested riparian areas. In: Long, J.W.; Quinn-Davidson, L.N.; Skinner, C.N., eds. 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: 323–340. Chap. 6.2.

Hunsaker, C.T.; Long, J.W. Herbst, D.B. 2014. Watershed and stream ecosystems. In: Long, J.W.; Quinn-Davidson, L.N.; Skinner, C.N., eds. 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: 265–322. Chap. 6.1.

Lamers, L.P.M.; Vile, M.A.; Grootjans, A.P.; [et al.]. 2015. Ecological restoration of rich fens in Europe and North America: from trial and error to an evidence-based approach. Biological Reviews. 90: 182–203.

Larned, S.T.; Datry, T.; Arscott, D.B.; [et al.]. 2010. Emerging concepts in temporary-river ecology. Freshwater Biology. 55: 717–738.

Leigh, C.; Boulton, A.J.; Courtwright, J.L.; [et al.]. 2016. Ecological research and management of intermittent rivers: An historical review and future directions. Freshwater Ecology. 61: 1181–1199.

Levick, L.; Goodrich, D.C.; Fernandez, M.; [et al.]. 2008. The ecological and hydrological significance of ephemeral and intermittent streams in the arid and semi-arid American Southwest. EPA/600/R-08/134; ARS/233046. Washington, DC: U.S. Environmental Protection Agency and U.S. Department of Agriculture, Agricultural Research Service, Southwest Watershed Research Center. 116 p.

Long, J.W.; Pope, K.L. 2014. Wet meadows. In: Long, J.W.; Quinn-Davidson, L.N.; Skinner, C.N., eds. 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: 341–372. Chap. 6.3.

Long, J.W.; Quinn-Davidson, L.N.; Skinner, C.N. 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.

Marty, J.T. 2015. Loss of biodiversity and hydrologic functioning seasonal wetlands persists over 10 years of livestock grazing removal. Restoration Ecology. 23: 548–554.

Merriam, K.E.; Markwith, S.H.; Coppoletta, M. 2017. Livestock exclusion alters plant species composition in fen meadows. Applied Vegetation Science. 21: 3–11.

Merriam, K.E.; Gosejohan, M.C.; Weisberg, P.J.; [et al.]. 2016. Livestock use has mixed effects on slender Orcutt grass in northeastern California vernal pools. Rangeland Ecology and Management. 60: 185–194.

Middleton, B.A.; Bettina, H.; van Diggelen, R. 2006. Biodiversity management of fens and fen meadows by grazing, cutting, and burning. Applied Vegetation Science. 9: 307–316.

Moyle, P.B.; Lusardi, R.A.; Samuel P.J.; [et al.]. 2017. State of the salmonids: status of California's emblematic fishes 2017. Davis, CA: University of California, Center for Watershed Sciences and California Trout. 579 p.

Norwick, S.A. 1991. Vernal pools and other seasonal bodies of standing water. Freemontia. 7: 8–19.

Patterson, L.; Cooper, D.J. 2007. The use of hydrologic and ecological indicators for the restoration of drainage ditches and water diversions in a mountain fen, Cascade Range, California. Wetlands. 27: 290–304.

Pope, K.L.; Long, J.W. 2014. Lakes: recent research and restoration strategies. In: Long, J.W.; Quinn-Davidson, L.N.; Skinner, C.N., eds. 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: 373–390. Chap. 6.4.

Reeves, G.H.; Olson, D.H.; Wondzell, S.M.; [et al.]. 2018. The aquatic conservation strategy of the Northwest Forest Plan—a review of the relevant science after 23 years. In: Spies, T.A.; Stine, P.A.; Gravenmier, R.; [et al.], tech. coords. 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, Pacific Northwest Research Station: 461–624.

Roche, L.M.; Kromschroeder, L.; Atwill, E.R.; [et al.]. 2012. Water quality conditions associated with cattle grazing and recreation on national forest lands. PLoS ONE. 8(6): e68127.

Schlatter, K.J.; Faist, A.M.; Collinge, S.K. 2016. Using performance standards to guide vernal pool restoration and adaptive management. Restoration Ecology. 24: 146–152.

Sikes, K.; Cooper, D.; Weis, S.; [et al.]. 2013. Fen conservation and vegetation assessment in the national forests of the Sierra Nevada and adjacent mountains, California (rev. public ver. 2, 2013). Sacramento, CA: California Native Plant Society. 314 p.

Snyder, C.T. 1962. A hydrological classification of valleys in the Great Basin, western United States. Hydrological Sciences Journal. 7: 53–59.

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.

Steward A.L.; von Schiller, D.; Tockner, K.; [et al.]. 2012. When the river runs dry: human and ecological values of dry riverbeds. Ecological Environment. 10: 202–209.

Surface Water Ambient Monitoring Program [SWAMP]. 2011. Extent of California's perennial and non-perennial stream. Sacramento, CA: State Water Resources Control Board. https://www.waterboards.ca.gov (18 Apr. 2020).

U.S. Department of Agriculture, Forest Service [USDA]. 2010a. Motorized Travel Management – Final Environmental Impact Statement – Lassen National Forest. R5-MB-207. Susanville, CA. 677 p.

U.S. Department of Agriculture, Forest Service [USDA]. 2010b. Motorized travel management final environmental impact statement—Modoc National Forest. Alturas, CA.

U.S. Department of Agriculture, Forest Service and U.S. Department of the Interior, Bureau of Land Management [USDA and USDOI] 2008. Sage steppe ecosystem restoration strategy: final environmental impact statement—Modoc National Forest. R5-MB-161. Alturas, CA. 16 p.

U.S. Environmental Protection Agency [US EPA]. 1999. Storm water technology fact sheet. Vegetated swales. 832-F-99-006. Washington, DC. 7 p.

U.S. Fish and Wildlife Service [USFWS]. 2005. Recovery plan for vernal pool ecosystems of California and Southern Oregon. Region 1, Portland, OR.

U.S. Fish and Wildlife Service [USFWS]. 2009. Species fact sheet. Great Basin redband trout (*Oncorhynchus mykiss gibbsi*). Oregon Fish and Wildlife Office. https://www.fws.gov/oregonfwo/articles.cfm?id=149489435/ (2 Oct. 2017).

van Diggelen, R.; Middleton, B.; Bakker, J.; [et al.]. 2006. Fens and floodplains of the temperate zone: present status, threats, conservation and restoration. Applied Vegetation Science 9: 157–162.

Chapter 4.3. Sagebrush Rangelands and Greater Sage-grouse in Northeastern California

R. Kasten Dumroese[1]

Sagebrush Rangelands

Sagebrush (*Artemisia* species) habitat, an intricate, species-rich mosaic of different sagebrush species and a remarkably diverse assemblage of grasses, forbs, and other shrubs, once covered about 170 million acres (69 million ha) across the Western United States (fig. 4.3.1). Noss et al. (1995) note that sagebrush habitat is an imperiled ecosystem because of its degradation, fragmentation, or removal by humans (Connelly et al. 2011), including conversion to agriculture (Leonard et al. 2000) compounded by other factors such as invasion by nonnative annual grasses, encroachment by junipers and piñon pines, improper grazing, and climate change (Davies et al. 2011) that interact in complex ways (see Finch et al. 2015). Additional discussion about climate impacts on sagebrush rangelands can be found in Chapter 6.1 (Wright, this synthesis, *Ecological Disturbance in the Context of a Changing Climate: Implications for Land Management in Northeastern California*).

On the Lassen and Modoc National Forests, sagebrush rangelands provide habitat for obligate species (they only live in sagebrush ecosystems) and facultative species (they use sagebrush ecosystems as well as other ecosystems). Sagebrush-obligate species include pygmy rabbit (*Brachylagus idahoensis*), sagebrush vole (*Lemmiscus curtatus*), pronghorn (*Antilocapra americana*), sagebrush sparrow (*Artemisiospiza nevadensis*), sage thrasher (*Oreoscoptes montanus*), and northern sagebrush lizard (*Sceloporus graciosus graciosus*) (fig. 4.3.2). Noteworthy facultative species are sandhill crane (*Grus canadensis*), elk (*Cervus canadensis*), and mule deer (*Odocoileus hemionus*). Sagebrush rangelands also host a wide variety of invertebrates, including pollinating insects and monarch butterflies (*Danaus plexippus*). Many of these animals are on the California list of Species of Greatest Conservation Need (Gonzales and Hoshi 2015). The pygmy rabbit, once rare in Lassen and Modoc Counties, now appears to be locally extinct (Larrucea and Brussard 2008).

Some conservationists have suggested that greater sage-grouse (*Centrocercus urophasianus*) serve as an "umbrella species" for conservation and restoration of sagebrush ecosystems because their status is an indicator of overall sagebrush ecosystem health. The assumption with this management philosophy is that other sagebrush-obligate species of concern will simultaneously benefit when the sagebrush ecosystem is managed for greater sage-grouse (Rowland et al. 2006). Carlisle et al. (2018) note, however, that while this may be appropriate at the landscape level, some treatments applied at the local level to benefit sage-grouse (e.g., enhance forb abundance by removing sagebrush) may reduce abundance of other sagebrush-obligate species at that local level.

Further, when managers are considering conservation and restoration efforts, they should look for opportunities where projects can meet multiple objectives simultaneously (see Dumroese et al. 2016). Indeed, managing sagebrush rangelands toward a rich mosaic of sagebrush, other sagebrush steppe shrubs, forbs, grasses, and wetland plants can enhance habitat for greater sage-grouse and other wildlife. For example, Copeland et al. (2014) documented that conservation measures for greater sage-grouse overlapped with migration corridors (about 70 percent), stopover locations (about 75 percent), and wintering areas (about 50 to 90 percent) of mule deer in Wyoming, yielding benefits to the herd. Such conservation efforts could also have strong monetary returns. For example, Horney (2010) notes that the decline in greater sage-grouse populations due to increases in juniper canopy cover coincides with loss of habitat and a dramatic decrease in the population of the local mule deer in Northeastern California, from 45,000 animals in the 1950s (160,000 according to Longhurst et al. 1952) to about 4,000 animals in 2008. Loft (1998) calculated, using hunter surveys, that between 1987 and 1997 deer hunting revenue dropped precipitously in Lassen ($5.4 million to $830,000) and Modoc ($4.7 million to $550,000) Counties.

The Decline of Greater Sage-grouse

An iconic symbol of sagebrush rangeland is the greater sage-grouse, and not surprisingly, populations of greater sage-grouse are declining in concert with the loss of

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Figure 4.3.1—Excellent sagebrush habitat in Lassen County supports species that only reside in sagebrush, as well as other species for which sagebrush can be an optional habitat (photo by Dawn M. Davis, used with permission).

quality sagebrush rangeland habitat across its range (see Chapter 3.2, Dumroese, this synthesis, *Rangeland in Northeastern California* for more discussion). Between 1965 and 1985 the population declined 70 percent across the range (Connelly et al. 2004) and the annual rate of decline of male birds (used as an estimate of population size) from 1965 through 2015 was about 2 percent, whereas the loss of leks was about 8 percent annually (Nielsen et al. 2015). This decline prompted the U.S. Department of the Interior, U.S. Fish and Wildlife Service (USFWS) to determine if the greater sage-grouse required protection under the Endangered Species Act of 1973; they concluded in 2005 that listing was not warranted. Challenged in court, the USFWS again reviewed status, and based on implementation of "science-based regulatory mechanisms in Federal and State plants [that] have substantially reduced risks to more than 90 percent of the

Figure 4.3.2—The northern sagebrush lizard requires sagebrush habitat for its livelihood (photo by Tony Kurz, used with permission).

species' modeled breeding habitats..." (https://www.fws.gov/greatersagegrouse/status.php) deemed listing was not warranted in 2015 (Federal Register 2015), but indicated the status of greater sage-grouse will be re-evaluated in 2020 (USDOI 2015). Despite these mechanisms and focused attention across public and private land ownerships, Garton et al. (2015) note little short-term success in stabilizing declining greater sage-grouse populations.

The same decline in greater sage-grouse populations is occurring in Northeastern California. Historically, greater sage-grouse were plentiful on the Modoc National Forest (hereafter the Modoc), despite this area being on the periphery of the species' range. Brown (1945) recounts seeing lines of hunters along the roads several miles in length easily shooting the low-flying, large birds, resulting in near extirpation of the species. A ban on hunting followed by revised bag limits was showing benefits by 1945, and Brown (1945) was optimistic about the species future, especially in the Devil's Garden area of the Modoc. Hunting was reinstated in the early 1950s, but from then until the early 1980s the estimated harvest of greater sage-grouse declined nearly 60 percent on the Lassen National Forest (hereafter the Lassen) and 86 percent on the Modoc (Horney 2010). The number of active leks (sparsely vegetated sites where males perform elaborate strutting to attract and mate with females) in the Devil's Garden Population Management Unit dropped from a high of 29 in 2000 to a single lek for the years 2001 to 2007, and on the Clear Lake National Wildlife Refuge that lies entirely within the Modoc, the number of male birds dropped from a high of about 60 in 1990 to an average of about 8 for the years 1999 to 2009 (Horney 2010).

Landscape Requirements

At the landscape level, modeling by Arkle et al. (2014) concluded that the probability of greater sage-grouse occupying a site was greatest when the cover of low (dwarf) sagebrush (*A. arbuscula, A. nova,* or *A. tripartita*) was 10 to 20 percent and that of Wyoming big sagebrush (*A. tridentata wyomingensis*) was 10 to 15 percent. Greater sage-grouse have been shown to prefer browsing on dwarf sagebrush, which has a lower monoterpene level than that Wyoming big sagebrush and requires less metabolic energy to digest, even when dwarf sagebrush is less abundant (Frye et al. 2013). While the taller stature of Wyoming big sagebrush does, however, provide superior nesting locations and escape cover, and can be associated with greater cover of forbs and grasses, greater sage-grouse readily and successfully use dwarf sagebrush habitat for nesting (Musil 2011).

Spatial Requirements and Annual Movements

Greater sage-grouse are, in general, wanderers, using vast areas of sagebrush habitat for courting, cover, and food in often complicated movements (Eng and Schladweiler 1972). For example, the annual migration of one greater sage-grouse population covers about 150 miles (240 km), and birds may roam 100 miles (160 km) in just 18 days (Smith 2013). In Northeastern California, birds may annually roam (straight-line distance) nearly 19 miles (30 km) (Davis et al. 2014). Such large distances generally correspond to large home ranges. Many studies have examined the home range of greater sage-grouse (e.g., Bruce et al. 2011; Davis et al. 2014; Hagen 1999; Leonard et al. 2000; Schroeder et al. 1999; Stonehouse et al. 2015), but different methodologies, numbers of observations, sites, elevations, seasons, quality of habitat, study objectives, and so on hamper the ability to compare results beyond general trends.

Traditionally, populations of greater sage-grouse have been classified as either migratory or nonmigratory (resident). Migratory populations move more than 10 km between distinct seasonal ranges (i.e., breeding, summer, autumn, winter) to complete their life histories, whereas the seasonal movement of resident populations overlaps within the same area (Connelly et al. 2000). Smith (2013) found that greater sage-grouse migrated about 150 miles (240 km) from winter ranges in Montana to summer ranges in Saskatchewan, whereas Leonard et al. (2000) noted annual migrations in Southeastern Idaho of about 68 miles (110 km). In Lassen County in Northeastern California, Davis et al. (2014) studied birds from 4 lek complexes (13 leks total) and found they moved an average of about 17 miles (27 km), although birds from one complex appeared to be more resident, never moving more than 6 miles (10 km) between seasonal ranges. Fedy et al. (2012), however, challenge this notion of migration. In Wyoming, they found that within populations, birds use different strategies, with some migrating long distances (more than 31 miles [50 km]) while others remained relatively sedentary. Across a range of Wyoming sites (populations), about 40 percent of each population had little inter-seasonal movement, and thus classifying a population as migratory or not may be inappropriate, and counterproductive to conservation, given that individual birds display different strategies to fulfill their annual life history. It is noteworthy that Davis et al. (2014) also noted high variability in the annual movement of individual greater sage-grouse in Lassen County.

Davis and others (2014) examined habitat requirements and annual movement of greater sage-grouse within the population occurring on the Buffalo-Skedaddle Population Management Unit, adjacent to the Lassen. Using the minimum convex polygon (MCP) approach, which uses connected straight lines to define a perimeter enclosing all sightings of the species, they estimated the home range for the population was about 840,150 acres (340,000 ha). This simple method often overestimates home range size because the perimeter, in an effort to encompass all sightings, may include large areas not actually used, or used infrequently, by the species (Boulanger and White 1990). Given this drawback, Davis and others (2014) also determined the annual 95 percent fixed kernel home range size (the area where the birds were observed 95 percent of the time) was 185,300 acres (75,000 ha) and the 50 percent core-area size, where birds spent 50 percent of their time, was 20,000 acres (8,100 ha). Although Davis et al. (2014) caution that these values may underestimate home range size, these values fall within the results (6,600 to 276,400 ha [16,310 to 683,000 acres]) observed by others (Connelly et al. 1988, 2004; Leonard et al. 2000). Further, Davis et al. (2014) found that the winter home range (95 percent fixed kernel) for the Buffalo-Skedaddle population was only about 13,350 acres (5,400 ha) whereas the average home range (95 percent fixed kernel) for the remainder of the year was about 117,375 acres (47,620 ha). Population of greater sage-grouse in Southeastern Idaho (Leonard et al. 2000) and Central Montana (Wallestad 1975) also showed small winter ranges. Davis et al. (2014) found that the home range size for individual birds in Lassen County was highly variable (1,235 to 176,430 acres [500 to 71,300 ha]).

Within the home range of the population, individual greater sage-grouse can travel long distances and have different home ranges depending on sex, age, and lek of capture. In Northeastern California, Davis et al. (2014) found that females had a larger, average home range (28,660 acres, range = 12,355 to 176,185 acres [11,600 ha, range = 5,000 to 71,300 ha]) than did adult males (14,580 acres, range = 1,235 to 44,480 acres [5,900 ha, range = 5,000 to 18,000 ha), the latter being more sedentary than yearling males (26,200 acres [10,600 ha). Within home ranges, wet meadows are an important habitat; see Chapter 6.1, Wright, this synthesis, *Ecological Disturbance in the Context of a Changing Climate: Implications for Land Management in Northeastern California*, for more discussion.

While understanding average home range size for populations of greater sage-grouse is important, Fedy et al. (2012) urge caution in focusing conservation solely on core areas used by greater sage-grouse because of the long distances birds travel during a year in search of seasonal habitat requirements (that is, food, cover). The risk of focusing solely on core area is that such an approach may not adequately include all of the annual needs for the species (Aldridge and Boyce 2007; Doherty et al. 2010; Fedy et al. 2012).

Gene Flow and Augmentation

A concern with ever-increasing fragmentation of sagebrush rangelands and declining greater sage-grouse populations, especially at the periphery of the species' range, is that genetic diversity will decline as increasingly smaller populations become progressively isolated; this can lead to local extinction (Crist et al. 2017; Frankham 2005). Wisdom et al. (2011) concluded that greater sage-grouse in Northeastern California and on the periphery of the species' range were at greater risk for extinction than populations more centrally located with the species' range. Loss of genetic diversity can begin a chronic decline in the population because of a reduction in fitness, less disease resistance, and an impaired ability to react to disturbances (see Davis et al. 2015).

Recent attention in Lassen County has looked at habitat requirements of greater sage-grouse (Davis et al. 2014) and whether or not declining populations on this extreme western end of the range were genetically stable (Davis 2012; Davis et al. 2015). Sampling birds from 13 leks across 4 lek complexes, Davis et al. (2015) found genetic diversity was similar to that of populations with the species' core range and without differentiation among the leks (all leks had diverse genetics). Davis et al. (2015) suggest that the sage-grouse in Northeastern California are maintaining these high levels of genetic diversity by breeding among adjacent leks and/or from sage-grouse populations in Northern Nevada; such inter-population gene flow is the most likely scenario for greater sage-grouse (Oyler-McCance et al. 2005). Indeed, recent work (Cross et al. 2018; Row et al. 2018) notes that greater sage-grouse in Northeastern California have genetic connectivity with birds in Oregon and potential connectivity with birds as far away as Southwestern Idaho. Davis et al. (2015) caution, however, that gene flow disrupted by fragmentation, resulting in reduced genetic diversity, may not become apparent for many generations; thus, Northeastern California birds may already be isolated from Nevada but the isolation is too soon to be detected. Finally, they note that within leks, breeding males and females were mostly unrelated and most likely it is the females moving long distances that aids gene flow and maintenance of genetic diversity.

For very small, isolated populations, genetic diversity could be augmented through introduction of greater sage-grouse from distant populations. In a review of 56 attempted translocations, Reese and Connelly (1997) found rather dismal results (less than 10 percent success). Attempts in Utah (Duvuvuei 2013; Gruber 2012), including one with long-term data (Baxter et al. 2008, 2009, 2013) have proven more successful, with acceptable survival and reproduction when placed in suitable habitat. On the Clear Lake National Wildlife Refuge located within the Modoc, Bell and George (2012) report good success with translocated greater sage-grouse. Hens moved during the breeding season and released at a lek quickly integrated into the local population and survival was similar to resident hens. Augmented with 59 hens, the number of males observed on the refuge lek increased from 5 in 2005 to 16 in 2011. Another approach, experimental in nature and somewhat controversial (see Wyoming Wildlife Federation 2017), is collection of eggs from wild nests, controlled rearing, and return of chicks to surrogate brooding hens (Thompson et al. 2015). The perceived advantages are: (1) reduced predation of eggs, (2) that "donor" hens may re-nest thus mitigating the potential effects on brood production within source populations, and (3) breeding age females are not moved to new locations; translocated birds, in addition to reducing the donor population, typically have lower survival than resident birds (Baxter et al. 2013; Duvuvuei 2013; Gruber 2012).

Seasonal Habitat Requirements

Leks

In early spring, male greater sage-grouse congregate on leks to court females. The lek itself generally occurs where sagebrush cover is minimal (less than 10 percent), such as open meadows, sparsely vegetated ridges, and even agricultural fields (Connelly et al. 2004; Ellis et al. 1989) with generally flat and relatively smooth surfaces (Knick et al. 2013). Modeling the minimum ecological requirements for greater sage-grouse leks in the western portion of their range (including sites in Northeastern California) and using a 3.1-mile (5-km) radius, Knick et al. (2013) found that leks were present when:

- sagebrush-dominated landscape covered 79 percent of the area, compared to 28 percent of the historic leks no longer occupied;
- conifer forest covered less than 1 percent of the area; leks were absent when conifer cover was 40 percent or greater;
- surrounded by, on average, greater than 40 percent landscape cover of sagebrush;
- surrounded by, on average, less than 10 percent agriculture; leks were absent when agriculture exceeded 25 percent;
- densities of roads, powerlines, pipelines, and communication towers were low.

Nest-Site Habitat

A number of studies across the range of greater sage-grouse, including one from Lassen County, report the average distance females move from their lek of capture to their initial nest-site ranges from 1.3 miles (2.1 km) to 4.8 miles (7.8 km), with most studies reporting an average of about 3 miles (4.7 km), and a few studies showing a range of 0.1 to 19 miles (200 m to 30 km) (table 4.3.1). Lyon and Anderson (2003) note, however, that when leks are disturbed (in their study, well pads or roads), the distance from lek of capture to initial nest site nearly doubled, with 91 percent of the nests occurring within 1.9 miles (3 km) of the lek of capture compared to just 26 percent when leks were disturbed. Measuring the distance from lek of captured to females' initial nest sites in Northeastern California, Davis et al. (2014) found that 39 percent of the nests occurred within 1.9 miles (3 km) of the lek of capture, and 73 percent were within 3.1 miles (5 km), results similar to Holloran and Anderson (2005). Davis et al. (2014) also noted a success rate of 56 percent for nests within 3 miles of the lek; the success rate for nests beyond 3 miles was similar. Because of lek locations, some females may move long distances from their lek of capture to nest, with that nest location being much closer to a different lek. Females, especially successful females, show strong fidelity to nest sites (Davis et al. 2015; Fischer et al. 1993).

Table 4.3.1—Average distance female greater sage-grouse travel from leks to nesting sites.

Location	Author(s)	Average km (miles)	Range km (miles)
Northwestern Wyoming	Lyon and Anderson 2003	2.1 (1.3)	--
Southeastern Idaho	Fischer 1994	3.4 (2.1)	--
Colorado	Peterson 1980 (fide Schroeder et al. 1999)	4.0 (2.5)	--
Southeastern Idaho	Wakkinen et al. 1992	4.6 (2.9)	--
Southeastern Alberta	Aldridge and Brigham 2001	4.7 (2.9)	0.4 to 16 (0.2 to 9.9)
Northeastern California	Davis et al. 2014	4.7 (2.9)	--
Western / Central Wyoming	Holloran and Anderson 2005	4.7 (2.9)	0.2 to 27 (0.1 to 16.8)
Northcentral Montana	Moynahan et al. 2007	4.8 (3.0)	0.5 to 30 (0.3 to 18.6)
Southeastern Montana / Southwestern North Dakota	Herman-Brunson et al. 2007	4.9 (3.0)	--
Wyoming	Goebel 1980 (fide Schroeder et al. 1999)	6.2 (3.8)	--
Central Washington	Schroeder et al. 1999	7.8 (4.8)	--

Greater sage-grouse build their nests on the ground (fig. 4.3.3). Nests are predated by a variety of animals, some attracted by scent, for example, skunks (*Mephitidae*) and American badger (*Taxidea taxus*) and some by vision, for example, common raven (*Corvus corax*) and black-billed magpie (*Pica pica*). Coyotes (*Canis latrans*) do not appear to be important predators (Mezquida et al. 2006). Much research has been conducted to describe habitat features that reduce nest predation. Features such as grass height and cover; shrub species, height, and cover; and understory vegetation and cover have been examined, with studies showing positive, negative, and neutral results across these features at the nest level—these discrepancies may be due to differences in the local predator communities (see Coates and Delehanty 2010) and greater sage-grouse subsequently selecting nest sites in response to local predator pressure (Conover et al. 2010), with a preference toward greater concealment from visual (avian) predators (Connelly et al. 2004; Doherty et al. 2010; Kirol et al. 2012). At a landscape scale, recent work found that nesting locations had lower densities of avian predators than did random locations (Dinkins et al. 2012).

Of the predators, common ravens are receiving much attention because their populations are increasing dramatically across the Western United States (see Coates et al. 2014) and more common ravens increase predation levels (Coates and Delehanty 2010). The increase in common raven populations is thought to be associated with anthropogenic disturbances that favor this species, such as provision of additional food sources (e.g., landfills) and electric / communication infrastructure that provides hunting perches and nesting sites (see Coates and Delehanty 2010; Coates et al. 2014; Dzialak et al. 2011; Harju et al. 2018; and references therein). In Wyoming, for every 0.6 miles (1 km) a nest was initiated closer to an overhead transmission line, the risk of nest and brood failure increased 12 and 38 percent, respectively (LeBeau et al. 2014). Knick et al. (2013) found that historic leks no longer active have much greater densities of powerlines and communication towers.

Figure 4.3.3—A female greater sage-grouse on a nest in Lassen County (photo by Dawn M. Davis, used with permission).

Data from a study in Elko County, Nevada, suggests that good nesting habitat has about 20 to 30 percent sagebrush cover and 40 percent or greater total shrub cover (Coates et al. 2010), similar to landscape-scale findings of Knick et al. (2013) and management guidelines proposed by Connelly et al. (2000). In addition to sagebrush canopy cover, grass height has long been identified as important; grasses are taller at nest sites compared to random locations (Hagen et al. 2007) and the occurrence of tall grass can, however, help mitigate decreased cover of sagebrush by obscuring nest sites (e.g., Davis et al. 2014; Kaczor et al. 2011). More recent research is, however, casting some doubt on the value of grass height as a factor in nest predation.

Height of grass and other herbaceous material changes rapidly during spring and this change in phenology coincides with greater sage-grouse nesting. Gibson et al. (2016b) asserts that collecting data without noting this transient nature in phenology may yield incorrect interpretations about vegetation height and brood success. Thus, recent work by Smith et al. (2018a), using data from previous studies in Idaho, Wyoming, and Montana that showed successful nests were associated with taller grass but adjusted for height for when nests were observed, found no difference in grass height between predated and nonpredated nests. Nonetheless, they do show that female greater sage-grouse preferred locating nests where grass was 4 to 8 inches (10 to 20 cm) tall. Gibson et al. (2016a), working in Central Nevada, found that female greater sage-grouse selected for nest sites associated with: (1) higher amounts of sagebrush and nonsagebrush cover heterogeneous in age (not dominated by uniform, tall shrubs); (2) more abundant, taller, and species-diverse forbs; and (3) taller live and residual grasses. This led Smith el al. (2018b) to conclude that the relationship between grass height and nest concealment is likely overstated, and thus strict guidelines for grass height should instead allow local managers flexibility (see Chapter 1 in Stiver et al. 2015), especially given that grass height showed more variability among growing seasons than among ranches following different grazing management plans.

In Lassen County on sites occupied by greater sage-grouse, sagebrush canopy cover is only about 10 percent (Davis et al. 2014), on the low end of the range for sagebrush communities (12 to 43 percent; Connelly et al. 2000). On this site, Davis (2012) found that grass height at the nest bowl significantly, and positively, influenced distribution of nest sites, whereas live sagebrush and shrub canopy cover, percent cover of grasses and bare ground, and nest bowl shrub and forb height were not significant (fig. 4.3.4). About a third of the nests were located under big sagebrush (*Artemisia tridentata*), with others found beneath silver sagebrush (*A. cana*), little sagebrush (*A. arbuscular*), antelope bitterbrush (*Purshia tridentata*), greasewood (*Sarcobatus vermiculatus*), Douglas rabbitbrush (*Chrysothamnus viscidiflorus*), horsebrush (*Tetradymia* species), crested wheatgrass (*Agropyron cristatum*), bluebunch wheatgrass (*Pseudoroegneria spicata*), and basin wildrye (*Leymus cinereus*). Nest success was about 56 percent, similar to rates observed across the range of greater sage-grouse. Another study in Lassen County on a degraded site found that 59 percent of nests were under big sagebrush (Popham and Gutiérrez 2003) and that successful nests were further from the lek (2.2 vs. 1.2 miles [3.6 vs. 2.0 km]), occurred under taller shrubs (25 vs. 19 inches [65 vs. 50 cm]), and the height of visual obstruction was greater (15 vs. 13 inches [40 vs. 32 cm]).

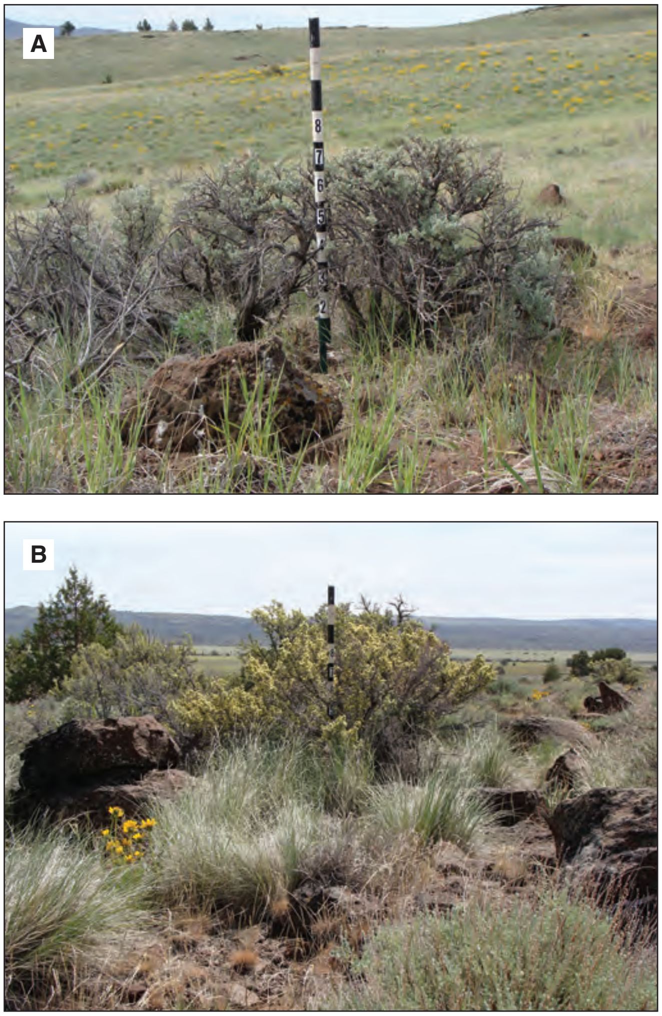

Figure 4.3.4—Examples of (A) unsuccessful and (B) successful greater sage-grouse nest sites in Lassen County (photo by Dawn M. Davis, used with permission).

Dietary Requirements

Spring

For pre-nesting hens, a variety of annual and perennial forbs constitute, by weight, about 18 to 50 percent of their diet (Barnett and Crawford 1994). In the Columbia Basin and Great Basin, *Lomatium* species are a significant portion of the diet of hens and chicks (Barnett 1992; Barnett and Crawford 1994; Ersch 2009); many other genera, including *Agoseris, Collomia, Crepis,* and *Phlox* are also consumed (see Dumroese et al. 2015, 2016). A large variety of annual invertebrates comprise 52 to 60 percent of the diet of very young chicks, with forb consumption increasing with age (Klebenow and Gray 1968; Peterson 1970). During the first week post-hatch in Southeastern Oregon, chicks consume ants (Hymenoptera: Formicidae), darkling beetles (Coleoptera: Tenebrionidae), scarab beetles (Coleoptera: Scarabaeidae), and various caterpillars (Lepidoptera) (Ersch 2009). *Ericameria* and *Chrysothamnus* (rabbitbrush) support more caterpillars, a good protein source, during the spring than do mountain big sagebrush (*Artemisia tridentata* ssp. *vaseyana*) communities (Ersch 2009). Broods move to more open sagebrush canopy cover because, as canopy cover of sagebrush decreases, the abundance of grasses and forbs increases (Olson and Whitson 2002) as does the abundance of invertebrates hosted by the herbaceous plants (fig. 4.3.5).

Summer

During summer, greater sage-grouse search for forage areas rich in succulent forbs, including agricultural fields, sagebrush uplands, and moist drainages (Braun et al. 2005). Forbs comprise 50 percent or more of the juvenile and adult summer diets (Barnett 1992; Barnett and Crawford 1994; Connelly et al. 2000; Ersch 2009; Gregg et al. 2008; Klebenow and Gray 1968; Trueblood

Figure 4.3.5—During the spring, greater sage-grouse hens and chicks move to areas with more open sagebrush canopy because forbs and other herbaceous plants are more plentiful, such as this rangeland near Observation Point in eastern Lassen County. Greater sage-grouse consume a variety of forbs directly, and forbs also indirectly support greater sage-grouse by hosting numerous invertebrates that are essential to the diet of developing chicks (photo by Dawn M. Davis, used with permission).

1954; Wallestad et al. 1975). The suite of plants consumed during summer changes, reflecting species availability as the season progresses. Summer forbs can include milkvetch (*Astragalus*), lily (*Calochortus*), lupine (*Lupinus*), and aster (*Symphyotrichum*) (see Dumroese et al. 2015). Tall sagebrush (more than 15 inches [40 cm]) with adequate canopy cover (10 to 25 percent) are required for resting and escape (Braun et al. 2005).

Autumn

As native vegetation continues to dry out, greater sage-grouse move toward northerly aspects having more moisture to continue to support native forbs late into the season and consumption of sagebrush becomes more common (Braun et al. 2005). Autumn forbs can include buckwheat (*Eriogonum*), goldenrod (*Solidago*), and asters (*Symphyotrichum*) (see Dumroese et al. 2015). Birds, now gathering into larger flocks, begin to congregate in denser, taller stands of sagebrush (Braun et al. 2005).

Winter

Weather and snow conditions have a large impact on habitat use by greater sage-grouse. Flocks prefer areas with dense sagebrush (for cover) and where sagebrush extends 8 to 12 inches (20 to 30 cm) or more above the snow because more than 99 percent of their diet is sagebrush. Birds may also use windswept ridges and southerly aspects (Braun et al. 2005).

Impacts of Fire

Prescribed fire on Wyoming big sagebrush sites in Southeastern Idaho and Eastern Oregon yielded no benefit to greater sage-grouse in either the short- (1 to 3 year) or long- (up to 14 years) term for a variety of nesting and brooding metrics, even when 60 percent of the sagebrush was removed (Beck et al. 2009; Fischer et al. 1996; Rhodes et al. 2010). Similar results were found for wild- and prescribed-fire in mountain big sagebrush habitats in Southeastern Idaho (Nelle et al. 2000). Moreover, because of the slow recovery of sagebrush to a stature that supports nesting sites, prescribed fire may negatively impact nesting habitat (Nelle et al. 2000). In an examination of 211 plots across the Great Basin, Arkle et al. (2014) found few burned areas conducive to breeding (less than 10 percent), brood rearing (less than 6 percent), or use during critical winter months (less than 10 percent) because of a limited sagebrush overstory. Modeling by Pedersen et al. (2003) suggests that light, infrequent fires on sites in Southeastern Idaho, where mountain big sagebrush was the predominate sagebrush species, have no effect on greater sage-grouse populations, whereas severe, frequent fires do, and that a combination of fire and sheep grazing on the season following fire would significantly decrease greater sage-grouse populations. Models by Coates et al. (2015) for the Great Basin, however, suggest that burning sagebrush near leks has a dampening effect on population growth, and in concert with the projected rates of burning and sagebrush recovery for the next 30 years, they predict steady and significant declines of greater sage-grouse in the Great Basin.

On sites within the Hart Mountain National Antelope Refuge, just northeast of the Modoc, and dominated by Wyoming big sagebrush, Pyle and Crawford (1996) found that spring prescribed fires generated more severe fires, that fires in both seasons greatly reduced shrub cover (similar to other studies), and that burning increased total forb cover and diversity. Unfortunately, response of greater sage-grouse to these changes in habitat was not measured. Similarly, Davis and Crawford (2014), working in Northwestern Nevada on the Sheldon National Wildlife Refuge, found that wildfire readily changed the mountain big sagebrush community structure from shrub-dominated to one dominated by native grasses and forbs 2 to 3 years post-burn. A decade later, little difference was noted in total forb cover, but sufficient shrub canopy had regenerated to provide suitable nesting cover. Moreover, abundance of arthropods, and in particular, ants, was unaffected by fire.

Wildfire in native rangeland can foster the invasion of invasive annual grasses, such as cheatgrass (*Bromus tectorum*). This starts a pernicious fire cycle, where the annual grasses burn more frequently than surrounding native rangeland (fig. 4.3.6). Each subsequent fire removes more of the perennial vegetation, chronically converting the shrub-bunchgrass communities required by greater sage-grouse into annual grasslands (D'Antonio and Vitousek 1992; Davison 1996). In Northwestern Nevada, however, Davis and Crawford (2014) report little change in the abundance of cheatgrass following wildfire on a mountain big sagebrush site where livestock grazing has been excluded since 1994.

Impacts of Energy Development and Vehicles

A robust literature concerning the impact of energy development on the sagebrush ecosystem has developed

Figure 4.3.6—Wildfire in sagebrush rangeland can destroy critical habitat required for obligate sagebrush species, as well as allow invasive species, such as cheatgrass, to become established. Often, invasion by cheatgrass changes the natural fire regime, leading to the establishment of vast monocultures of cheatgrass and loss of native flora and fauna diversity (photo by Amanda Shoaf, Forest Service).

during the past 2 decades, with particular focus on greater sage-grouse. Much of this work was done in Wyoming with natural gas production, but because common denominators of energy development, whether it be extracting oil, gas, or hot water (geothermal) from below ground, or operating windmills, are the need for a pad to operate from and supporting infrastructure (e.g., roads). Thus, the literature from natural gas extraction in Wyoming is pertinent to the Lassen and Modoc.

Energy development and transmission in sagebrush rangelands supporting greater sage-grouse has detrimental effects on the populations. Much of the recent energy development is in more pristine sagebrush communities supporting the greatest densities of greater sage-grouse and other sagebrush-obligate species (Bergquist et al. 2007; Crawford et al. 2004; Gilbert and Chalfou 2011; Kaiser 2006; Knick et al. 2003). Greater sage-grouse are affected by energy development and transmission because these activities fragment habitat, increase noise, benefit predators, and foster spread of nonnative, invasive plants. Waste-water holding ponds have potential to facilitate spread of West Nile virus (Schrag et al. 2011), which can infect and rapidly kill greater sage-grouse, and has been detected across the West, including California (USGS 2006).

When well pads are installed near leks (0.25 to 0.5 miles [0.4 to 0.8 km]), the number of male birds in attendance during the breeding season drops dramatically (35 to 91 percent; Blickley et al. 2012a; Harju et al. 2010; Walker et al. 2007); the decrease in males also increases with increasing well-pad density (Harju et al. 2010). This decline can be acute (occurring during the first year; Walker et al. 2007) or more chronic (a delay in the decrease of males between the onset of development and measurable effects on leks; Harju et al. 2010). In one study, male attendance at leks dropped 29 percent when exposed to noise associated with natural gas drilling and 73 percent with noise from roadways (Blickley et al. 2012a) and males that remained on leks exposed to noise had higher levels of stress hormones (Blickley et al. 2012b). Using 30 years of data from Wyoming, Hess and Beck (2012) found unoccupied leks had 10 times more oil and gas wells in a 0.6-mile (1-km) radius than did occupied leks, and the probability of a lek persisting dropped below 50 percent when well density was more than 2 per 247 acres (100 ha). Females were found to move further from leks disturbed by well-pad and road activity before initiating nests, nearly twice as far as females in areas without development (Lyon and Anderson 2003). During winter, greater sage-grouse are less likely to occupy habitat near wells (Doherty et al. 2008), even if that habitat is suitable and regardless of the activity levels at the wells (Matthew et al. 2015).

Wind energy development can affect greater sage-grouse, too. Zimmerling et al. (2013) estimate that every turbine is associated with 3 acres (1.2 ha) of habitat loss, and because each turbine requires a pad, the same issues observed with natural gas pads pertain, although traffic associated with turbines is much less than that for natural gas (4 vehicle visits per year for operation and maintenance versus 1,285 vehicle visits, respectively; see LeBeau et al. 2014 and Sawyer et al. 2009). This reduced disturbance may have less impact on the birds. However, work with open-country, gallinaceous species similar to greater sage-grouse (lesser prairie-chickens, *Tympanuchus pallidicinctus,* and greater prairie-chickens, *T. cupido*) found that birds avoided crossing under transmission lines, suggested that tall structures, including turbines, may inhibit use of suitable habitat near wind energy farms (Pruett et al. 2009). More recent work by LeBeau et al. (2017) suggests that habitat disturbance caused by wind farm infrastructure is more important than turbine height. They modeled that the resulting 2 percent habitat disturbance within 0.75 miles (1.2 km) of a string of 10 turbines caused by the actual turbine footprint and access road would decrease the selection of that habitat for brood-rearing and summer habitat by 60 percent.

As discussed above, electrical energy transmission lines provide hunting perches and nesting sites for avian predators of greater sage-grouse, and in particular, common ravens. Coates et al. (2014) found that the swath of habitat affected by common ravens using a transmission line supported by 65-foot (20-m) tall poles could be as much as 2.8 miles (4.5 km) wide, and LeBeau et al. (2014) found that nest and brood failure decreased with increasing distance from transmission lines.

In addition to the detrimental effects of motorized vehicle noise described above, vehicles can serve as vectors in the spread nonnative plant species. The magnitude of impact is influenced by level of road improvement, soil type, aspect, native vegetation type (i.e., forest or grassland), and the nonnative species (Gelbard and Belnap 2003; Gelbard and Harrison 2003; Hansen and Clevenger 2005). The disturbed edges of roadways allow nonnative plant establishment. Although roadways in forests can facilitate spread of nonnative plants (e.g., Mortensen 2009), grasslands appear more susceptible to invasion (Hansen and Clevenger 2005). In semiarid grasslands, shrublands, and woodlands of Southern Utah, Gelbard and Belnap (2003) found that road type (paved, improved gravel surface, graded without gravel, four-wheel-drive track) was important, mainly because the zone of roadside disturbance increased with increasing development. The general trend was more weeds and less native plants in these disturbed zones. For example, the coverage of cheatgrass increased threefold when comparing four-wheel-drive tracks to paved roadways. This effect persisted past the disturbed zone; the number of nonnative species and their abundance 164 feet (50 m) outward from the edge of roadside disturbance was greater, and the richness of native species lower, along paved roads compared to four-wheel-drive tracks.

As described earlier, greater sage-grouse may act as “umbrella species” for other sagebrush-obligate species and species that can also thrive in the sagebrush ecosystem. Thus, it is likely that energy development may also have detrimental effects on other wildlife (Knick et al. 2003). Indeed, mule deer avoid natural gas wells, especially during winter (Sawyer et al. 2009) and Gilbert and Chalfoun (2011) noted well density decreased abundance of Brewer’s sparrow (*Spizella breweri*), sage sparrow, and vesper sparrow (*Pooecetes gramineus*), but for horned larks (*Eremophila alpestris*), a species that prefers bare ground with little or no vegetation, their abundance increased. Sage thrashers (*Oreoscoptes montanus*), a sagebrush-obligate species, showed no response to energy development (fig. 4.3.7).

Impacts of Grazing

In a review of the literature, Beck and Mitchell (2000) found that livestock grazing can have direct and indirect impacts on sage-grouse habitat, and that these impacts can be positive or negative. Direct positive effects were associated with light grazing and rest-rotation grazing that encouraged forbs. Direct negative effects were associated with overgrazing that reduced forbs and high herd densities that physically disrupted nests. In Idaho, greater sage-grouse formed a new lek where domestic sheep had cleared an area. Indirect negative effects were associated with conversion of sagebrush to pure grass forage and introduction of weeds because these lead to the loss of forb diversity and abundance. Recent work documented 25 years of habitat management on a study site and found that applying small-acreage sagebrush treatments that reduced shrub cover to release the forb understory sustained grazing and increased the number of birds counted on leks (Dahlgren et al. 2015). The authors note, however, that there is a limit to this conversion and the cumulative removal of sagebrush over time can negatively affect the availability of winter cover, resulting in winter mortality.

Determining and implementing a grazing management plan in concert with managing wildlife habitat requires an approach that combines plant community dynamics with habitat requirements with livestock use and their interactions (Vavra 2006). Detailed grazing management practices have been developed to foster compatibility between livestock and greater sage-grouse; these plans focus on annual management of forage to meet native plant growth requirements in order to either maintain a healthy sagebrush site or encourage progression to a more desired state (e.g., Cagney et al. 2010). Grazing has the most influence during nesting and early brood-rearing (Boyd et al. 2014); during this period maximum herbaceous cover and height is desired to decrease predation of nests and chicks. Similarly, Monroe et al. (2017) found that early grazing caused population decline, whereas sage-grouse populations responded positively to that same level of grazing after peak vegetation productivity. Moderate grazing can maintain habitat, sustained heavy grazing reduces forb abundance, and targeted grazing can decrease fine-fuel accumulation where annual invasive grasses are a concern. Pedersen et al. (2003), modeling fire and sheep grazing in mountain big sagebrush in Southeastern Idaho, found that any level of grazing decreased greater sage-grouse populations, and that grazing following burning could reduce populations by 14 percent (light grazing) to 40 percent (heavy grazing).

Figure 4.3.7—Bird species react differently to disturbance caused by roads and well development. While abundance of Brewer's and sage sparrows decreased, abundance of sage thrashers (shown here) showed no response to energy development (photo by Tony Kurz, used with permission).

An indirect effect of grazing is greater sage-grouse mortality caused by collisions with fences; in Idaho about 0.4 strikes occurred per km (0.6 miles) of fence (Stevens et al. 2012a). For California, Stevens et al. (2013) predict that more than 10 percent of the area within 1.8 miles (3 km) of a lek could have more than 1 collision during a season. Prioritizing mitigation would probably affect 6 to 14 percent of the landscape (Stevens et al. 2013) and should focus on flat terrain having more than 0.6 miles (1 km) of fence per 247 acres (100 ha) within 1.2 miles (2 km) of active leks (Stevens et al. 2012a, 2013). Fences with wooden posts spaced less than 13 feet (4 m) apart reduced collisions (Stevens et al. 2012a) and placing reflective markers on fences reduced collisions about 57 to 83 percent (Stevens et al. 2012b; Van Lanen et al. 2017).

Impacts of Conifer Encroachment

As alluded to elsewhere (Moser, this synthesis, *Understanding and Managing the Dry Conifer Forests of Northeastern California*; Padgett, this synthesis, *Weeds, Wheels, Fire, and Juniper: Threats to Sagebrush Steppe*; earlier portions of this chapter) encroachment of juniper into new areas of the sagebrush steppe, as well as infill (increasing canopy closure of previously spare stands of trees) has ecological implications. Miller et al. (2001, 2005, 2008) report a 3- to 10-fold increase in distribution and a 10-fold increase in abundance, bringing the coverage of these species in the interior Western United States to more than 18 million ha (refer to the *Juniper Woodlands* section in Chapter 2.1, Moser, this synthesis, *Understanding and Managing the Dry Conifer Forests of Northeastern California* for more information on potential causes for this expansion). The transition of sites dominated by sagebrush, perennial grasses, and forbs to a piñon-juniper forestland follows a three-stage trajectory (Miller et al. 2005, 2013). Initially, few trees are established (phase 1, less than 10 percent cover). As the sagebrush canopy becomes codominant with the trees (phase 2, 10 to 30 percent cover), the herbaceous understory declines and once the trees dominate (phase 3, more than 30 percent), little to no herbaceous understory remains. Transition from sagebrush steppe to a conifer-dominated landscape further fragments the sagebrush steppe ecosystem and pressures wildlife populations. For example, earlier in this chapter the precipitous decline of mule deer with increasing canopy of conifers in bitterbrush-dominated stands in Northeastern California was noted, a byproduct of the loss of herbaceous browse material. Similarly, conifer expansion is also thought to be detrimental to greater sage-grouse because the tree canopy decreases the herbaceous understory critical to sage-grouse brood success and the trees provide perches for avian predators of eggs (e.g., common ravens) and adult birds (e.g., hawks) (see Coates et al. 2017 and *Nest-Site Habitat* above). In Central Oregon during the breeding season, 75 percent of the greater sage-grouse were found in stands of *Artemisia arbuscula* (low sagebrush) having less than 5 percent juniper cover (Freese et al. 2016) and Baruch-Mordo et al. (2013) found that lek activity ceased with a juniper canopy cover of as little as 4 percent. More recent work by Coates et al. (2017) noted greater sage-grouse avoided sites with canopy cover of 1.5 to 2 percent. Mapping by Falkowski et al. (2017) indicate nearly 40 percent of the greater sage-grouse range in California has more than the 4 percent threshold identified by Baruch-Mordo et al. (2013).

Recent science is demonstrating, however, that greater sage-grouse respond positively to conifer removal (fig. 4.3.8). In Northern Utah, Sandford et al. (2017) found that nest and brood success was greater for hens using habitat nearer areas where mechanical removal had occurred. Similarly, Severson et al. (2017) found that within a large, mostly mechanically treated area (34,000 ha [84,000 acres]) in Southeast Oregon just north of the Modoc (removal treatments from 2007 through 2014 with minimal fire; a portion of the nontreated, experiment control area was in Modoc County), removing conifers increased annual female and nest survival 6.6 and 18.8 percent, respectively, and they estimated a 25 percent increase in overall population growth compared to the nontreated

Figure 4.3.8—Removing juniper from sagebrush rangeland can benefit greater sage-grouse and other sagebrush-obligate species. Treating junipers while their densities are low and judiciously burning the slash to protect the remaining sagebrush allows forbs to rebound in abundance and retention of the woody canopy structure preferred by nesting greater sage-grouse (photo by Amanda Shoaf, Forest Service).

control areas. For the Severson et al. (2017) effort, individual stand-level treatments were 42 to 6,200 acres (17 to 2,500 ha) in size and cumulatively covered about 20 percent of the treated area (about 16,000 ac [6,500 ha]).

Although the abundance of perennial forbs was not consistently enhanced by a variety of conifer removal treatments, increases of 1.5- to 6-fold were observed (Bates et al. 2017). They conclude that mechanical removal and low disturbance fuel-reduction conifer treatments (essentially restricting burning to felled trees), especially in phase 1 and 2 conditions, may be best for greater sage-grouse in the short term. Compared to broadcast prescribed fires, these treatments yield a similar forb response while maintaining the critical shrub canopy required by the birds. Moreover, on phase 3 sites in Southeastern Oregon, conifer removal treatments followed by burning promoted cheatgrass; these sites had 4 to 16 times more cheatgrass than that found on phase 2 sites (Bates et al. 2013). Unfortunately, these treatments fail to remove young conifer seedlings and the seed bank, so follow-up treatments will be required. Boyd et al. (2017) note that prescribed, broadcast fire controls conifer regrowth about twice as long as mechanical felling, but reduction of the shrub canopy and the long-time horizon for it to regrow to a stature that supports greater sage-grouse (in excess of 20 to 30 years) may be prohibitive when quality habitat is needed to sustain the sage-grouse populations. Thus, they suggest restricting broadcast prescribed fire to late phase 2 and phase 3 areas where the forb/sagebrush component has already been so degraded that it currently fails to support greater sage-grouse. Doherty et al. (2018) used simulations to predict potential outcomes of juniper removal and suggest such an approach can assist land managers and stakeholders in better understanding biological returns on investment.

Removing all conifers, especially pre-European settlement junipers (i.e., those older than 150 years), may not be prudent as these trees have other functional traits. For example, these legacy trees can be important roost sites for bats. Trees with at least one cavity and proximity to water were more favored (Anthony 2016). Males preferred home ranges with low juniper canopy cover (less than 10 percent), whereas lactating females used woodlands with up to 20 percent canopy cover.

Severson et al. (2017) conclude that conifer removal may seem expensive but proactive management, that is, targeting areas where conifer encroachment is just beginning (phase 1 sites), can yield positive benefits to overall ecosystem health (see Davies et al. 2011) as well as to ranch-level income (e.g., McClain 2012). They note that a $9 million investment on phase 1 sites having greater sage-grouse would, for example, treat all of Oregon. McClain (2012) modeled that for Southwestern Idaho sites, juniper encroachment from phase 1 to phase 2 would reduce livestock forage 37 percent and ranch income by 15 percent. Farzan et al. (2015) in modeling conifer removal scenarios in Lassen and Modoc Counties notes, however, that derived benefits will depend on goal prioritization at a landscape scale; prioritizing forage production provided little benefit to greater sage-grouse but targeting treatments to benefit the birds also derived forage benefits. See Chapter 3.2 (Dumroese, this synthesis, *Rangeland in Northeastern California*) for more information about prioritizing and restoring sagebrush steppe.

References

Aldridge, C.L.; Boyce, M.S. 2007. Linking occurrence and fitness to persistence: habitat-based approach for endangered greater sage-grouse. Ecological Applications. 17: 508– 526.

Aldridge, C.L.; Brigham, R.M. 2001. Nesting and reproductive activities of greater sage-grouse in a declining northern fringe population. The Condor. 103: 537–543.

Anthony, C.R. 2016. Resource selection and space use of western long-eared myotis (*Myotis evotis*) in a western juniper (*Juniperus occidentalis*) woodland of Central Oregon. Corvallis, OR: Oregon State University. 102 p. M.S. thesis.

Arkle, R.S.; Pilliod, D.S.; Hanser, S.E.; [et al.]. 2014. Quantifying restoration effectiveness using multi-scale habitat models: implications for sage-grouse in the Great Basin. Ecosphere. 5: 1–32.

Barnett, J.K. 1992. Diet and nutrition of female sage grouse during the pre-laying period. Corvallis, OR: Oregon State University. 46 p. M.S. thesis.

Barnett, J.K.; Crawford, J.A. 1994. Pre-laying nutrition of sage grouse hens in Oregon. Journal of Range Management. 47: 114–118.

Baruch-Mordo, S.; Evans, J.S.; Severson, J.P.; [et al.]. 2013. Saving sage-grouse from the trees: a proactive solution to reducing a key threat to a candidate species. Biological Conservation. 167: 233–241.

Bates, J.D.; Davies, K.W.; Hulet, A.; [et al.]. 2017. Sage grouse groceries: forb response to piñon-juniper treatments. Rangeland Ecology and Management. 70(1): 106–115.

Bates, J.D.; Sharp, R.N.; Davies, K.W. 2013. Sagebrush steppe recovery after fire varies by development phase of Juniperus occidentalis woodland. International Journal of Wildland Fire. 23: 117-130.

Baxter, R.J.; Flinders, J.T.; Mitchell, D.L. 2008. Survival, movements, and reproduction of translocated greater sage-grouse in Strawberry Valley, Utah. Journal of Wildlife Management. 72: 179–186.

Baxter, R.J.; Flinders, J.T.; Whiting, D.G.; [et al.]. 2009. Factors affecting nest-site selection and nest success of translocated greater sage grouse. Wildlife Research. 36: 479–487.

Baxter, R.J.; Larsen, R.T.; Flinders, J.T. 2013. Survival of resident and translocated greater sage-grouse in Strawberry Valley, Utah: a 13-year study. Journal of Wildlife Management. 77: 802–811.

Beck, J.L.; Connelly, J.W.; Reese, K.P. 2009 Recovery of greater sage-grouse habitat features in Wyoming big sagebrush following prescribed fire. Restoration Ecology. 17: 393–403.

Beck, J.L.; Mitchell, D.L. 2000. Influences of livestock grazing on sage grouse habitat. Wildlife Society Bulletin. 28(4): 993–1002.

Bell, C.B.; George, T.L. 2012. Survival of translocated greater sage-grouse hens in northeastern California. Western North American Naturalist. 72(3): 369–376.

Bergquist, E.; Evangelista, P.; Stohlgren, T.J.; [et al.]. 2007. Invasive species and coal bed methane development in the Powder River Basin, Wyoming. Environmental Monitoring and Assessment. 128: 381–394.

Blickley. J.L.; Blackwood, D.; Patricelli, G.L. 2012a. Experimental evidence for the effects of chronic anthropogenic noise on abundance of greater sage-grouse at leks. Conservation Biology. 26: 461–471.

Blickley, J.L.; Word, K.R.; Krakauer, A.H.; [et al.]. 2012b. Experimental chronic noise is related to elevated fecal corticosteroid metabolites in lekking male greater sage-grouse (*Centrocercus urophasianus*). PLoS ONE. 7(11): e50462.

Boulanger, J.G.; White, G.C. 1990. A comparison of home-range estimators using Monte Carlo simulation. Journal of Wildlife Management. 54: 310–315.

Boyd, C.S.; Beck, J.L.; Tanaka, J.A. 2014. Livestock grazing and sage-grouse habitat: impacts and opportunities. Journal of Rangeland Applications. 1: 58–77.

Boyd, C.S.; Kerby, J.D.; Svejcar, T.J.; [et al.]. 2017. The sage-grouse habitat mortgage: effective conifer management in space and time. Rangeland Ecology and Management. 70(1): 141–148.

Braun, C.E.; Connelly, J.W.; Schroeder, M.A. 2005. Seasonal habitat requirements of sage-grouse: spring, summer, fall and winter. In: Shaw, N.L.; Pellant, M.; Monsen, S.B., compilers. Sage-grouse habitat restoration symposium proceedings. RMRS-P-38. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 38–42

Brown, W.S., Jr. 1945. History of the Modoc National Forest. San Francisco, CA: U.S. Department of Agriculture, Forest Service.

Bruce, J.R.; Robinson, W.D.; Petersen, S.L.; [et al.]. 2011. Greater sage-grouse movements and habitat use during winter in central Oregon. Western North America Naturalist. 71: 418–424.

Cagney, J.; Bainter, E.; Budd, B.; [et al.]. 2010. Grazing influence, objective development, and management in Wyoming's greater sage-grouse habitat with emphasis on nesting and early brood rearing. B-1203. Laramie, WY: University of Wyoming Cooperative Extension Service. 57 p. https://www.wyowildlife.com/Upload/File/Wyo%20Grazing%202010.pdf (9 Dec 2019).

Carlisle, J.D.; Chalfoun, A.D.; Smith, K.T.; [et al.]. 2018. Nontarget effects on songbirds from habitat manipulation for greater sage-grouse: implications for the umbrella species concept. Condor. 120: 439–455.

Coates, P.S.; Delehanty, D.J. 2010. Nest predation of greater sage-grouse in relation to microhabitat factors and predators. Journal of Wildlife Management. 74: 240–248.

Coates, P.S.; Howe, K.B.; Casazza, M.L.; [et al.]. 2014. Common raven occurrence in relation to energy transmission line corridors transiting human-altered sagebrush steppe. Journal of Arid Environments. 111: 68–78.

Coates, P.S.; Ricca, M.A.; Prochazka, B.G.; [et al.]. 2015. Long-term effects of wildfire on greater sage-grouse—integrating population and ecosystem concepts for management in the Great Basin. Open-File Report 2015–1165. Reston, VA: U.S. Geological Survey. 42 p.

Coates, P.S.; Prochazka, B.G.; Ricca, M.A.; [et al.]. 2017. Pinyon and juniper encroachment into sagebrush ecosystems impacts distribution and survival of greater sage-grouse. Rangeland Ecology and Management. 70(1): 25–38.

Connelly, J.W.; Browers, H.W.; Gates, R.J. 1988. Seasonal movements of sage grouse in southeastern Idaho. Journal of Wildlife Management. 52: 116–122.

Connelly, J.W.; Knick, S.T.; Braun, C.E.; [et al.]. 2011. Conservation of greater sage-grouse: a synthesis of current trends and future management. In: Knick, S.T.; Connelly, J.W., eds. Greater sage-grouse: ecology and conservation of a landscape species and its habitats. Studies in Avian Biology 38. Berkeley, CA: University of California Press for Cooper Ornithological Society: 549–653.

Connelly, J.W.; Knick, S.T.; Schroeder, M.A.; [et al.]. 2004. Conservation assessment of greater sage-grouse and sagebrush habitats. Cheyenne, WY: Western Association of Fish and Wildlife Agencies. 610 p. https://www.ndow.org/wild/conservation/sg/resources/greate_sg_cons_assessment.pdf (20 Aug. 2018).

Connelly, J.W.; Shroeder, M.A.; Sands, A.R.; [et al.]. 2000. Guidelines to manage sage grouse and their habitats. Wildlife Society Bulletin 28: 967–985.

Conover, M.R.; Borgo, J.S.; Dritz, R.E.; [et al.]. 2010. Greater sage-grouse select nest sites to avoid visual predators but not olfactory predators. The Condor. 112(2): 331–336.

Copeland, H.E.; Sawyer, H.; Monteith, K.L.; [et al.]. 2014. Conserving migratory mule deer through the umbrella of sage-grouse. Ecosphere. 5: 117. 16 p.

Crawford, J.A.; Olson, R.A.; West, N.E.; [et al.]. 2004. Ecology and management of sage-grouse and sage-grouse habitat. Rangeland Ecology and Management. 57: 2–19.

Crist, M.R.; Knick, S.T.; Hanser, S.E.l. 2017. Range-wide connectivity of priority areas for greater sage-grouse: implications for long-term conservation from graph theory. The Condor. 119: 44–57.

Cross, T.D.; Schwartz, M.K.; Naugle, D.E.; [et al.]. 2018. The genetic network of greater sage-grouse: range-wide identification of keystone hubs of connectivity. Ecology and Evolution. 8: 5394–5412.

Dahlgren, D.K.; Larsen, R.T.; Danvir, R.; [et al.]. 2015. Greater sage-grouse and range management: insights from a 25-year case study in Utah and Wyoming. Rangeland Ecology and Management. 68: 375–382.

D'Antonio, C.M.; Vitousek, P.M. 1992. Biological invasions by exotic grasses, the grass-fire cycle, and global change. Annual Review of Ecology and Systematics. 23: 63–87.

Davies, K.W.; Boyd, C.S.; Beck, J.L.; [et al.]. 2011. Saving the sagebrush sea: an ecosystem conservation plan for big sagebrush plant communities. Biological Conservation. 144: 2573–2584.

Davis, D.M. 2012. Population structure of greater sage-grouse in northeastern California: implications for conservation in a declining peripheral population. Moscow, ID: University of Idaho. 241 p. Ph.D. dissertation.

Davis, D.M.; Crawford, J.A. 2014. Case study: short-term response of greater sage-grouse habitats to wildfire in mountain sagebrush communities. Wildlife Society Bulletin. 39(1): 129–137.

Davis, D.M.; Reese, K.P.; Gardner, S.C. 2014. Demography, reproductive ecology, and variation in survival of greater sage-grouse in northeastern California. Journal of Wildlife Management. 78: 1343–1355.

Davis, D.M.; Reese, K.P.; Gardner, S.C.; [et al.]. 2015. Genetic structure of greater sage-grouse (*Centrocercus urophasianus*) in a declining, peripheral population. The Condor. 117: 530–544.

Davison, J. 1996. Livestock grazing in wildland fuel management programs. Rangelands. 18(6): 242–245.

Dinkins, J.B.; Conover, M.R.; Kirol, C.P.; [et al.]. 2012. Greater sage-grouse (*Centrocercus urophasianus*) select nest sites and brood sites away from avian predators. The Auk 129: 600–610.

Doherty, K.E.; Hennig, J.D.; Dinkins, J.B.; [et al.]. 2018. Understanding biological effectiveness before scaling up range-wide restoration investments for Gunnison sage-grouse. Ecosphere. 9: e02144.

Doherty, K.E.; Naugle, D.E.; Walker, B.L.; [et al.]. 2008. Greater sage-grouse winter habitat selection and energy development. Journal of Wildlife Management. 72: 187–195.

Doherty, K.E.; Naugle, D.E.; Walker, B.L. 2010. Greater sage-grouse nesting habitat: the importance of managing at multiple scales. Journal of Wildlife Management. 74: 1544–1553.

Dumroese, R.K.; Luna, T.; Pinto, J.R.; [et al.]. 2016. Forbs: foundation for monarch butterflies, other pollinators, and greater sage-grouse in the western United States. Natural Areas Journal. 36: 499–511.

Dumroese, R.K.; Luna, T.; Richardson, B.A.; [et al.]. 2015. Conserving and restoring habitat for greater sage-grouse and other sagebrush-obligate wildlife: the crucial link of forbs and sagebrush diversity. Native Plants Journal. 16: 276–299.

Duvuvuei, O.V. 2013. Vital rates, population trends, and habitat-use patterns of a translocated greater sage-grouse population: implications for future translocations. Logan, UT: Utah State University. 168 p. M.S. thesis.

Dzialak, M.R.; Olson, C.V.; Harju, S.M. [et al.]. 2011. Identifying and prioritizing greater sage-grouse nesting and brood-rearing habitat for conservation in human-modified landscapes. PLoS ONE. 6(10): e26273.

Ellis, K.L.; Parrish, J.L.; Murphy, J.R.; [et al.]. 1989. Habitat use by breeding male sage grouse: a management approach. Great Basin Naturalist. 49: 404–407.

Eng, R.L.; Schladweiler, P. 1972. Sage grouse winter movements and habitat use in central Montana. Journal of Wildlife Management. 36: 141–146.

Ersch E. 2009. Plant community characteristics on insect abundance: implications on sage-grouse brood rearing habitats. Corvallis, OR: Oregon State University. 109 p. M.S. thesis.

Falkowski, M.J.; Evans, J.S.; Naugle, D.E.; [et al.]. 2017. Mapping tree canopy cover in support of proactive prairie grouse conservation in western North America. Rangeland Ecology and Management. 70:15–24.

Farzan, S.; Young, D.J.N.; Dedrick, A.G.; [et al.]. 2015. Western juniper management: assessing strategies for improving greater sage-grouse habitat and rangeland productivity. Environmental Management. 56(3): 675–683.

Federal Register. 2015. Endangered and threatened wildlife and plants; 12-month finding on a petition to list greater sage-grouse (*Centrocercus urophasianus*) as an endangered or threatened species. Proposed rule. Federal Register. 80(191): 59857–59942.

Fedy, B.C.; Aldridge, C.L.; Doherty, K.E.; [et al.]. 2012. Interseasonal movements of greater sage-grouse, migratory behavior, and an assessment of the core regions concept in Wyoming. Journal of Wildlife Management. 76: 1062–1071.

Finch, D.; Boyce, D.; Chambers, J.; [et al.]. 2015. Sage-grouse conservation science strategy 2015–2020. Washington. DC: U.S. Department of Agriculture, Forest Service. 39 p.

Fischer, R.A. 1994. The effects of prescribed fire on the ecology of migratory sage grouse in southeastern Idaho. Moscow, ID: University of Idaho. Ph.D. dissertation.

Fischer, R.A.; Apa, A.D.; Wakkinen, W.L.; [et al.]. 1993. Nesting-area fidelity of sage grouse in southeastern Idaho. The Condor. 95: 1038–1041.

Fischer, R.A.; Reese, K.P.; Connelly, J.W. 1996. An investigation on fire effects within xeric sage grouse brood habitat. Journal of Range Management. 49: 194–198.

Frankham, R. 2005. Genetics and extinction. Biological Conservation. 126: 131–140.

Freese, M.T.; Petersen, S.L.; Miller, R.F.; [et al.]. 2016. Spatial analysis of greater sage-grouse habitat use in relation to landscape level habitat structure. Journal of Ecosystem and Ecography. 6: 3.

Frye, G.G.; Connelly, J.W.; Musil, D.D.; [et al.]. 2013. Phytochemistry predicts habitat selection by an avian herbivore at multiple spatial scales. Ecology. 94: 308–314.

Garton, E.O.; Wells, A.G.; Baumgardt, J.A.; [et al.]. 2015. Greater sage-grouse population dynamics and probability of persistence. In: Knick, S.T.; Connelly, J.W., eds. Greater sage-grouse: ecology and conservation of a landscape species and its habitats. Studies in Avian Biology 38. Berkeley, CA: University of California Press Press for Cooper Ornithological Society: 292–381.

Gelbard, J.L.; Belnap, J. 2003. Roads as conduits for exotic plant invasions in a semiarid landscape. Conservation Biology. 17: 420–432.

Gelbard, J.L.; Harrison, S. 2003. Roadless habitats as refuges for native grasslands: interactions with soil, aspect, and grazing. Ecological Applications. 13: 404–415.

Gibson, D.; Blomberg, E.J.; Atamian, M.T.; [et al.]. 2016a. Nesting habitat selection influences nest and early offspring survival in greater sage-grouse. The Condor. 118(4): 689–702.

Gibson, D.; Blomberg, E.J.; Sedinger, J.S. 2016b. Evaluating vegetation effects on animal demographics: the role of plant phenology and sampling bias. Ecology and Evolution. 6(11): 3621–3631.

Gilbert, M.M.; Chalfoun, A.D. 2011. Energy development affects populations of sagebrush songbirds in Wyoming. Journal of Wildlife Management. 75(4): 816–824.

Gonzales, A.G.; Hoshi, J., eds. 2015a. Cascades and Modoc Plateau Province. California state wildlife action plan, 2015 update: a conservation legacy for Californians. vol. 1 Sacramento, CA: California Department of Fish and Wildlife with assistance from Ascent Environmental Inc. 51 p. Chap. 5.2. https://www.wildlife.ca.gov/SWAP/Final (2 Oct 2017).

Gregg, M.A.; Barnett, J.K.; Crawford, J.A. 2008. Temporal variation in diet and nutrition of pre-incubating sage grouse. Rangeland Ecology and Management 61: 535–542.

Gruber, N.W. 2012. Population dynamics and movements of translocated and resident greater sage-grouse on Anthro Mountain, Utah. Logan, UT: Utah State University. M.S. thesis.

Hagen, C.A. 1999. Sage grouse habitat use and seasonal movements in a naturally fragmented landscape, northwestern Colorado. Winnipeg, MB: University of Manitoba, Natural Resources Institute. 136 p. M.S. thesis.

Hagen, C.A.; Connelly, J.W.; Shroeder, M.A. 2007. A meta-analysis of greater sage-grouse *Centrocercus urophasianus* nesting and brood-rearing habitats. Wildlife Biology. 13: 42–50.

Hansen, M.J.; Clevenger, A.P. 2005. The influence of disturbance and habitat on the presence of non-native plant species along transport corridors. Biological Conservation.125: 249–259.

Harju, S.M.; Dzialak, M.R.; Taylor, R.C.; [et al.]. 2010. Thresholds and time lags in effects of energy development on greater sage-grouse populations. Journal of Wildlife Management. 74: 437–448.

Harju, S.M.; Olson, C.V.; Hess, J.E.; [et al.]. 2018. Common raven movement and space use: influence of anthropogenic subsidies within greater sage-grouse nesting habitat. Ecosphere. 9: e02348. 16 p.

Herman-Brunson, K.M.; Jensen, K.C.; Kaczor, N.W.; [et al.]. 2007. Nesting ecology of greater sage-grouse *Centrocercus urophasianus* at the eastern edge of their historic distribution. Wildlife Biology. 15: 395–404.

Hess, J.E.; Beck, J.L. 2012. Disturbance factors influencing greater sage-grouse lek abandonment in north-central Wyoming. Journal of Wildlife Management. 76: 1625–1634.

Holloran, M.J.; Anderson, S.H. 2005. Spatial distribution of greater sage-grouse nests in relatively contiguous sagebrush habitats. The Condor. 107: 742–752

Horney, M. 2010. Conservation strategy for sage grouse (*Centrocercus urophasianus*) and sagebrush ecosystems within the Devil's Garden / Clear Lake Population Management Unit. Clear Lake Sage Grouse Working Group. Unpublished Report. 109.

Kaczor, N.W.; Jensen, K.C.; Klaver, R.W.; [et al.]. 2011. Nesting success and resource selection of greater sage-grouse. In: Sandercock, B.K.; Martin, K.; Segelbacher, G., eds. Ecology, conservation, and management of grouse. Studies in Avian Biology 39. Berkeley, CA: University of California Press for Cooper Ornithological Society: 107–118.

Kaiser, R.C. 2006. Recruitment by greater sage-grouse in association with natural gas development in western Wyoming. Laramie, WY: University of Wyoming. 102 p. M.S. thesis.

Kirol, C.P.; Beck, J.L.; Dinkins, J.B.; [et al.]. 2012. Greater sage-grouse nesting and brood-rearing microhabitat selection in xeric big sagebrush. The Condor. 114: 75–89.

Klebenow, D.A.; Gray, G.M. 1968. Food habits of juvenile sage grouse. Journal of Range Management. 21: 80–83.

Knick, S.T.; Dobkin, D.S.; Rotenberry, J.T.; [et al.]. 2003. Teetering on the edge or too late? Conservation and research issues for avifauna of sagebrush habitats. The Condor. 105: 611–634.

Knick, S.T.; Hanser, S.E.; Preston, K.L. 2013. Modeling ecological minimum requirements for distribution of greater sage-grouse leks: implications for population connectivity across their western range, USA. Ecology and Evolution. 3: 1539–1551.

Larrucea, E.S.; Brussard, P.F. 2008. Habitat selection and current distribution of the pygmy rabbit in Nevada and California, USA. Journal of Mammalogy. 89: 691–699.

LeBeau, C.W.; Beck, J.L.; Johnson, G.D.; [et al.]. 2014. Short-term impacts of wind energy development on greater sage-grouse fitness. Journal of Wildlife Management. 78: 522–530.

LeBeau, C.W.; Johnson, G.D.; Holloran, M.J.; [et al.]. 2017. Greater sage-grouse habitat selection, survival, and wind energy infrastructure. Journal of Wildlife Management. 81(4): 690–711.

Leonard, K.M.; Reese, K.P.; Connelly, J.W. 2000. Distribution, movements and habitats of sage grouse *Centrocercus urophasianus* on the Upper Snake River Plain of Idaho: changes from the 1950s to the 1990s. Wildlife Biology. 6(4): 265–270.

Loft, E.R. 1998. Economic contribution of deer, pronghorn antelope, and sage grouse hunting to northeastern California and implications to the overall "value" of wildlife. California Wildlife Conservation Bulletin. 11. 42 p.

Longhurst, W.M.; Leopold, A.S.; Dasmann, R.F. 1952. A survey of California deer herds—their ranges and management problems. Sacramento, CA: California Department of Fish and Game Bureau of Game Conservation. 136 p.

Lyon, A.G.; Anderson, S.H. 2003. Potential gas development impacts on sage grouse nest initiation and movement. Wildlife Society Bulletin. 31: 486–491.

Matthew, J.; Holloran, M.J.; Fedy, B.C.; [et al.]. 2015. Winter habitat use of greater sage-grouse relative to activity levels at natural gas well pads. Journal of Wildlife Management. 79: 630–640.

McClain, A., 2012. Ranch level economic impacts of western juniper (*Juniperus occidentalis*) encroachment on sagebrush steppe ecosystems in Owyhee County, Idaho. Moscow, ID: University of Idaho, Moscow. M.S. thesis.

Mezquida, E.T.; Slater, S.J.; Benkman, C.W. 2006. Sage-grouse and indirect interactions: potential implications of coyote control on sage-grouse populations. The Condor. 108(4):747–759.

Miller, R.F.; Bates, J.D.; Svejcar, T.J.; [et al.]. 2005. Biology, ecology, and management of western juniper. Technical Bulletin 152. Corvallis, OR: Oregon State University, Agricultural Experiment Station. https://juniper.oregonstate.edu/bibliography/documents/phpQ65pOk_tb152.pdf (2 Feb. 2013).

Miller, R.F.; Chambers, J.C.; Pyke, D.A.; [et al.]. 2013. A review of fire effects on vegetation and soils in the Great Basin region: response and ecological site characteristics. Gen. Tech. Rep. RMRS-GTR-308. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, https://www.fs.fed.us/rm/pubs/rmrs_gtr308.html (3 Nov. 2015).

Miller, R.F.; Tausch, R.J. 2001. The role of fire in pinyon and juniper woodlands: a descriptive analysis. In: Galley, K.E.M.; Wilson, T.P., eds. Proceedings of the invasive species workshop: the role of fire in the control and spread of invasive species. Tallahassee, FL: Tall Timbers Research Station: 15–30.

Miller, R.F.; Tausch, R.J.; McArthur, E.D.; [et al.]. 2008. Age structure and expansion of piñon-juniper woodlands: a regional perspective in the Intermountain West. RMRS-RP-69. Fort Collins, CO: United States Department of Agriculture, Forest Service. 15 p.

Monroe, A.P.; Aldridge, C.L.; Assal, T.J.; [et al.]. 2017. Patterns in greater sage-grouse population dynamics correspond with public grazing records at broad scales. Ecological Applications. 27(4): 1096–1107.

Mortensen, D.A.; Rauschert, E.S.J.; Nord, A.N.; [et al.]. 2009. Forest roads facilitate the spread of invasive plants. Invasive Plant Science and Management 2 (3): 191–199.

Moynahan, B.J.; Lindberg, M.S.; Rotella, J.J.; [et al.]. 2007. Factors affecting nest survival of greater sage-grouse in northcentral Montana. Journal of Wildlife Management. 71: 1773–1783.

Musil, D.D. 2011. Use of dwarf sagebrush by nesting greater sage-grouse. In Sandercock, B.; Martin, K.; Segelbacher, G., eds. Ecology, conservation, and management of grouse. Studies in Avian Biology 39. Berkeley, CA: University of California Press for Cooper Ornithological Society: 119–136.

Nielsen, R.M.; McDonald, L.L.; Mitchell, J.; [et al.]. 2015. Analysis of greater sage-grouse lek data: trends in peak male counts 1965–2015. Cheyenne, WY: Western EcoSystems Technology, Inc. 43 p.

Nelle, P.J.; Reese, K.P.; Connelly, J.W. 2000. Long-term effects of fire on sage grouse habitat. Journal of Range Management. 53: 586–591.

Noss, R.F.; LaRoe, E.T., III; Scott, J.M. 1995. Endangered ecosystems of the United States: a preliminary assessment of loss and degradation. Biological Report 28. Washington, DC: U.S. Department of the Interior, U.S. Fish and Wildlife Service, National Biological Service. 60 p.

Olson, R.A.; Whitson, T.D. 2002. Restoring structure in late-successional sagebrush communities by thinning with tebuthiuron. Restoration Ecology. 10: 146–155.

Oyler-McCance, S.J.; Taylor, S.E.; Quinn, T.W. 2005. A multilocus population genetic survey of the greater sage-grouse across their range. Molecular Ecology. 14: 1293–1310.

Pedersen, E.K.; Connelly, J.W.; Hendrickson, J.R.; [et al.]. 2003. Effect of sheep grazing and fire on sage grouse populations in southeastern Idaho. Ecological Modelling. 165: 23–47

Peterson, J.G. 1970. The food habits and summer distribution of juvenile sage grouse in central Montana. Journal of Wildlife Management. 34: 147–155.

Popham, G.P.; Gutiérrez, R.J. 2003. Greater sage-grouse *Centrocercus urophasianus* nesting success and habitat use in northeastern California. Wildlife Biology. 9: 327–334.

Pruett, C.L.; Patten, M.A.; Wolfe, D.H. 2009. Avoidance behavior by prairie grouse: implications for development of wind energy. Conservation Biology. 23(5): 1253–1259.

Pyle, W.H.; Crawford, J.A. 1996. Availability of foods of sage grouse chicks following prescribed fire in sagebrush-bitterbrush. Journal of Range Management. 49: 320–324.

Reese, K.P.; Connelly, J.W. 1997. Translocations of sage grouse *Centrocercus urophasianus* in North America. Wildlife Biology. 3: 235–241

Rhodes, E.C.; Bates, J.D.; Sharp, R.N.; [et al.]. 2010. Fire effects on cover and dietary resources of sage grouse habitat. Journal of Wildlife Management. 74: 755–764.

Row, J.R.; Doherty, K.E.; Cross, T.B.; [et al.]. 2018. Quantifying functional connectivity: the role of breeding habitat, abundance, and landscape features on range-wide gene flow in sage-grouse. Evolutionary Applications. 11: 1305–1321.

Rowland, M.M.; Wisdom, M.J.; Suring, L.H.; [et al.]. 2006. Greater sage-grouse as an umbrella species for sagebrush-associated vertebrates. Biological Conservation. 129: 323–335.

Sawyer, H.; Kauffman, M.J.; Nielson, R.M. 2009. Influence of well pad activity on winter habitat selection patterns of mule deer. Journal of Wildlife Management. 73: 1052–1061.

Schrag, A.; Konrad, S.; Miller, S.; [et al.]. 2011. Climate-change impacts on sagebrush habitat and West Nile virus transmission risk and conservation implications for greater sage-grouse. GeoJournal. 76: 561–575.

Sandford, C.P.; Kohl, M.T.; Messmer, T.A.; [et al.]. 2017. Greater sage-grouse resource selection drives reproductive fitness under a conifer removal strategy. Rangeland Ecology and Management. 70(1): 59–67.

Schroeder, M.A.; Young, J.R.; Braun, C.E. 1999. Greater sage-grouse (*Centrocercus urophasianus*). In: Rodewald, P.G., ed. The birds of North America. Ithaca, NY: Cornell University Laboratory of Ornithology. https://birdsna.org/Species-Account/bna/species/saggro (29 Sep. 2017).

Severson, J.P.; Hagen, C.A.; Tack, J.D.; [et al.]. 2017. Better living through conifer removal: a demographic analysis of sage-grouse vital rates. PLoS ONE. 12(3): e0174347.

Smith, R. 2013. Conserving Montana's sagebrush highway: long distance migration in sage grouse. Missoula, MT: University of Montana. 47 p. M.S. thesis.

Smith, J.T.; Tack, J.D.; Berkeley, L.I.; [et al.]. 2018. Effects of rotational grazing management on nesting greater sage-grouse. Journal of Wildlife Management. 82: 103–112.

Smith, J.T.; Tack, J.D.; Doherty, K.E.; [et al.]. 2018. Phenology largely explains taller grass at successful nests in greater sage-grouse. Ecology and Evolution. 8(1): 356–364.

Stevens, B.S.; Connelly, J.W.; Reese, K.P. 2012a. Multi-scale assessment of greater sage-grouse fence collision as a function of site and broad scale factors. Journal of Wildlife Management. 76: 1370–1380.

Stevens, B.S.; Naugle, D.E.; Dennis, B.; [et al.]. 2013. Mapping sage-grouse fence-collision risk: spatially explicit models for targeting conservation implementation. Wildlife Society Bulletin. 37: 409–415.

Stevens, B.S.; Reese, K.P.; Connelly, J.W.; [et al.]. 2012b. Greater sage-grouse and fences: does marking reduce collisions? Wildlife Society Bulletin. 36: 297–303.

Stiver, S.J.; Rinkes, E.T.; Naugle, D.E.; [et al.], eds. 2015. Sage-grouse habitat assessment framework: a multiscale assessment tool. Tech. Ref. 6710-1. Denver, CO; U.S. Department of the Interior, Bureau of Land Management and Western Association of Fish and Wildlife Agencies. 132 p.

Stonehouse, K.F.; Shipley, L.A.; Lowe, J.; [et al.]. 2015. Habitat selection and use by sympatric, translocated greater sage-grouse and Columbian sharp-tailed grouse. Journal of Wildlife Management. 79: 1308–1326.

Thompson, T.R.; Apa, A.D.; Reese, K.P.; [et al.]. 2015. Captive rearing sage-grouse for augmentation of surrogate wild broods: Evidence for success. Journal of Wildlife Management. 79(6): 998–1013.

Trueblood, R.W. 1954. The effect of grass reseeding in sagebrush lands on sage grouse populations. Logan, UT: Utah State Agricultural College. 78 p. M.S. thesis.

U.S. Department of the Interior [USDOI]. 2015. Historic conservation campaign protects greater sage-grouse. [Press release.] Denver, CO. https://www.doi.gov/pressreleases/historic-conservation-campaign-protects-greater-sage-grouse (29 Sep. 2017).

U.S. Geological Survey [USGS]. 2006. West Nile virus in greater sage-grouse. Wildlife Health Bulletin 06-08. Madison, WI: National Wildlife Health Center. 3 p.

Van Lanen, N.J.; Green, A.W.; Gorman, T.R.; [et al.]. 2017. Evaluating efficacy of fence markers in reducing greater sage-grouse collisions with fencing. Biological Conservation. 213: 70–83.

Vavra, M. 2006. Livestock grazing and wildlife: developing compatibilities. Rangeland Ecology and Management. 58: 128–134.

Wakkinen, W.L.; Reese, K.P.; Connelly, J.W. 1992. Sage grouse nest locations in relation to leks. Journal of Wildlife Management. 56: 381–383.

Walker, B.L.; Naugle, D.E.; Doherty, K.E. 2007. Greater sage-grouse population response to energy development and habitat loss. Journal of Wildlife Management. 71: 2644–2654.

Wallestad, R. 1975. Life history and habitat requirements of sage grouse in central Montana. Helena, MT: Montana Fish and Game Department. 65 p.

Wallestad, R.; Peterson, J.G.; Eng, R.L. 1975. Foods of adult sage grouse in central Montana. Journal of Wildlife Management. 39: 628–630.

Wisdom, M.J.; Meinke, C.W.; Knick, S.T.; [et al.]. 2011. Factors associated with extirpation of sage-grouse. In: Knick, S.T.; Connelly, J.W., eds. Greater sage-grouse: ecology and conservation of a landscape species and its habitats. Studies in Avian Biology 38. Berkeley, CA: University of California Press for Cooper Ornithological Society: 451–472.

Wyoming Wildlife Federation. 2017. Our comments to WGF on captive rearing of sage grouse. https://wyomingwildlife.org/our-comments-to-wgf-on-captive-rearing-of-sage-grouse/ (5 Dec. 2017).

Zimmerling, J.R.; Pomeroy, A.C.; d'Entremont, M.V.; [et al.]. 2013. Canadian estimate of bird mortality due to collisions and direct habitat loss associated with wind turbine developments. Avian Conservation and Ecology. 8(2): 10.

Section 5. Society

Chapter 5.1. An Introduction to Social, Economic, and Ecological Factors in Natural Resource Management of Northeastern California Public Lands

David Flores[1]

Socioecological Resilience

The adaptive management of forests and rangelands is directed toward achieving an ecologically sustainable landscape that contributes to social and economic sustainability. This "socioecological resilience" is a significant focus of the U.S. Department of Agriculture, Forest Service 2012 Planning Rule directing how national forests are to revise their forest plans, as introduced in Chapter 1.1 (Dumroese, this synthesis, *The Northeastern California Plateaus Bioregion Science Synthesis: Background, Rationale, and Scope*). This process is complicated not only by ecological disturbances discussed throughout this *Northeastern California Plateaus Bioregion Science Synthesis*, but also by stressors of demographic, economic, and social change at a variety of scales. Stressors relevant to Lassen and Modoc National Forests include anticipated forces of:

- **demographic trends**—human population shifts in the area may produce a reduction in overall population count but with increased racially and ethnically diverse members; and
- **ecosystem services**—increases in population-related "demand for ecological services within the region and from areas farther removed" are expected (Long et al. 2014, p. vi), yet within the context of an anticipated reduction in forest ecosystem services in many areas of the Western United States.

Meanwhile, measures of adaptation may be learned and practiced through:

- **community engagement**—proactive, creative, collaborative community-based partnerships are encouraged to embrace science education and work across institutional boundaries (Vose et al. 2012), despite the resource deficits and social fragmentation that many local rural communities face; and
- **tribes and cultures**—respect of multiple cultures, especially of local tribal values, may encourage a philosophy of sustaining forest health, cultivating public goods, investing in future generations, and participating in the natural environment in ways that may ultimately contribute to the well-being of the local socioecological system.

As noted in Chapter 1.1 (Dumroese, this synthesis, *The Northeastern California Plateaus Bioregion Science Synthesis: Background, Rationale, and Scope*), recent science syntheses, namely, the *Science Synthesis to Support Socioecological Resilience in the Sierra Nevada and Southern Cascade Range* (Long et al. 2014; hereafter Sierra Nevada Science Synthesis); *Synthesis of Science to Inform Land Management Within the Northwest Forest Plan Area* (Spies et al. 2018; hereafter Northwest Forest Plan Science Synthesis); and *Effects of Climatic Variability and Change on Forest Ecosystems: A Comprehensive Science Synthesis for the U.S. Forest Sector* (Vose et al. 2012) review the general approach and the substantive concerns of managing the impact of change on Forest Service-managed lands with integrated ecological, economic, social, and cultural perspectives and strategies. The Sierra Nevada Science Synthesis is in part modeled on other documents and, in addition to social/economic/cultural human dimensions, includes natural science sections on forest ecology, genetics of forest trees, fire, soils, water resources and aquatic ecosystems, terrestrial wildlife, and air quality. The Northwest Forest Plan Science Synthesis reviews socioeconomic and cultural issues, as well as natural and physical aspects of climate and forest science, local species census, biodiversity projections, and aquatic conservation. The synthesis on climatic variability discusses ecological effects, but attends more heavily to suggested social and institutional responses required to navigate the impacts of change upon socioecological systems.

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Both the Sierra Nevada and Northwest Forest Plan science syntheses already addressed aspects of human dimensions with the Lassen and Modoc National Forests (hereafter the Lassen, the Modoc, or the Lassen-Modoc) The Sierra Nevada Science Synthesis devotes Section 9 to Social/Economic/Cultural Components (see Long et al. 2014). The Northwest Forest Plan Science Synthesis (Spies et al. 2018) similarly addresses human dimensions of forest management in Chapters 8 through 11. A science synthesis of climate change impacts on forest ecosystems (Vose et al. 2012) dedicates Chapter 3 to socioeconomic contexts. It also reviews in Chapter 4 some management models in use for the adaptation and mitigation of climate change specifically, but which may be extrapolated to decision making for the sustainability of socioecological systems and management of ecosystem services (see Vose et al. 2012). This *Northeastern California Plateaus Bioregion Science Synthesis* makes note of these science syntheses where relevant to social-economic concerns, but also responds to concerns particular to the Lassen and the Modoc, insofar as peer-reviewed literature exists. Note that the Restoration sections of Chapter 2.1 (Moser, this synthesis, *Understanding and Managing the Dry Conifer Forests of Northeastern California*) and Chapter 3.2 (Dumroese, this synthesis, *Rangeland in Northeastern California*) have additional discussion on the decisionmaking process for dry pine forestland and sagebrush rangelands, respectively.

Tribes, community members, partners, and stakeholders who are invested in the Lassen and the Modoc cited gaps in both the Sierra Nevada and Northwest Forest Plan science syntheses. This science synthesis has distilled the many social, economic, and cultural questions they proposed and responds to them in this society section that includes four chapters:

- Chapter 5.2: *Demographic Trends in Northeastern California*
- Chapter 5.3: *Ecosystem Services and Public Land Management*
- Chapter 5.4: *Community Engagement in the Decisionmaking Process for Public Land Management in Northeastern California*
- Chapter 5.5: *Integrating Tribes and Culture Into Public Land Management.*

The general limitation of these chapters is the scarcity of social science literature specific to the Lassen, the Modoc, and the surrounding areas. The chapters in this section draw on social science topics of concern to local community members and Forest Service staff. Where possible, we include literature that relates directly to Lassen and Modoc Counties, as well as to the Northern California region.

References

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.

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.

Vose, J.M.; Peterson, D.L.; Patel-Weynand, T., eds. 2012. Effects of climatic variability and change on forest ecosystems: A comprehensive science synthesis for the U.S. Forest Sector. Gen. Tech. Rep. PNW-GTR-870. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 265 p.

Chapter 5.2. Demographic Trends in Northeastern California

David Flores and Gregory Russell[1]

The anticipated changes in human populations, natural ecosystems, and the global climate are expected to impact public lands management. Land use is expected to shift due to altered productivity of local ecosystems, to overall increased human population, and to the changing market interests and cultural values. In rural areas, these demographic trends may result in the expansion of the wildland urban interface into previously undeveloped areas for amenity communities, and conversely, to departure from communities dependent on natural resource-supported economies. A recent science synthesis on climate emphasizes the importance of human responses to future opportunities and constraints:

> The ability of communities with resource-based economies to adapt to climate change is linked to their direct exposure to these changes, as well as to the social and institutional structures present in each environment. Human communities that have diverse economies and are resilient to change today will also be better prepared for future climatic stresses, especially if they implement adaptation strategies soon (Vose et al. 2012, p. vi).

Whether about climate impacts or other sources of change, strategies of adaptation are common themes in recent science syntheses and in the experience of economic transition in the area including the Lassen and Modoc National Forests (hereafter the Lassen, the Modoc, or the Lassen-Modoc), in which service and government sector employment has expanded. Local communities near the Lassen and the Modoc have requested information regarding projected local and regional demographic trends and how State and Federal prison populations influence demographic representations in the Lassen-Modoc area. Given existing scientific peer review literature, the following section approaches this request from a more general level. The guiding questions are:

- What are the current population trends across the Lassen-Modoc, including population trends in rural communities?
- What is the impact of prisons on rural communities?

The literature in this chapter is beyond the field of natural resources and predominantly draws from the field of political science. The section begins with an overview of current demographics, followed by projected population change in rural communities, the impact of prisons on rural communities, and ends with Federal considerations of prison populations.

Overview of Current Demographics

We attempt to orient the Lassen-Modoc stakeholders to their demographic context by responding to the following requests:

- What are the current population trends across the Lassen-Modoc, including regional population trends?
- What are the impacts on rural areas by large metropolitan cities, such as Reno, NV?

Population Shifts in Number and in Composition

The areas of Lassen and Modoc Counties cover 4,720 and 4,203 square miles (12,225 and 10,886 km^2) respectively, with an average population density of approximately two people per square mile in Modoc County, and seven people per square mile in Lassen County. Like many rural counties throughout the United States, Lassen and Modoc Counties have experienced gradual population declines since 2010 (fig. 5.2.1).

In contrast to the decreasing population in Lassen and Modoc Counties, the other counties included in the Sierra Nevada Science Synthesis have experienced population growth and settlement due to an influx of seasonal and year-round residents who are drawn to the area by its unique features and amenities (Loeffler and Steinicke 2007). Population projections for 2050 in the 12-county area of the Sierra Nevada region show an anticipated increase of 48.5 percent (State of California, Department of Finance 2012). In contrast, the total populations of Lassen and Modoc Counties are projected to decline by 19.9% and 10.7%, respectively.

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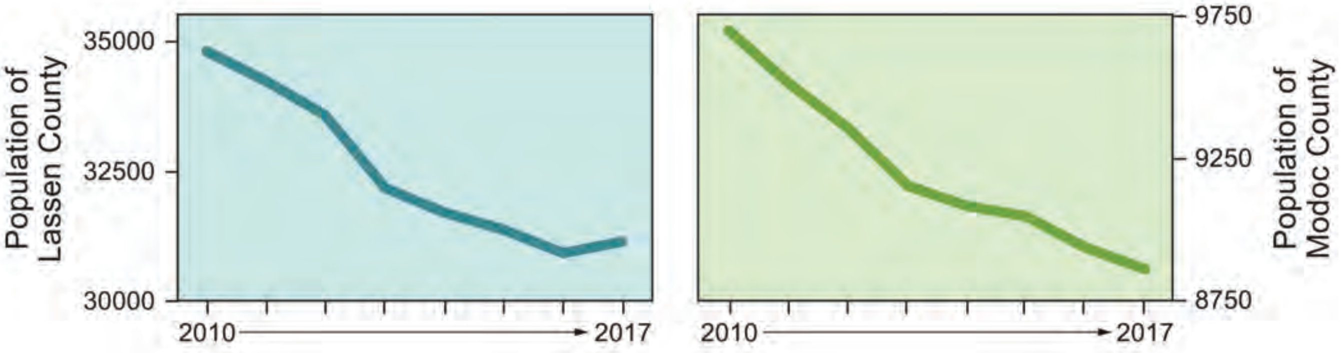

Figure 5.2.1—Population estimates for 2010 to 2017 for Lassen and Modoc Counties, CA, show a steady, gradual decline. U.S. Census Bureau data as of 1 July 2018 (https://data.census.gov).

While counties in the Sierra Nevada Science Synthesis area are expected to experience significant increases in diversity of racial and ethnic composition between 2010 and 2050, Lassen County is expected to experience population declines across groups identified by census data as White, Asian, Black, Native Pacific Islander, and Native American groups (State of California, Department of Finance 2012). Meanwhile, the population of people who identify as Hispanic or multiracial is expected to gradually increase. Modoc County is expected to experience a gradual decline in the population of residents who identify as White and maintain a relatively steady population of people who identify as Asian, Black, Native Pacific Islander, and Native American. Finally, the number of people who identify as Hispanic or multiracial in Modoc County is expected to gradually increase (State of California, Department of Finance 2012).

The Sierra Nevada Science Synthesis Chapter 9.1*, Broader Context for Social, Economic, and Cultural Components* (Winter et al. 2014, pp. 501–541) offers several research suggestions on how to engage and communicate with diverse communities in forest planning. More recently, Chapter 10 of Charnley et al. (2018) found in Spies et al. (2018) addresses environmental justice, low-income, and minority populations, and forest management in the Northwest Forest Plan Area.

Impacts of Metropolitan Areas on Rural Areas

Urbanization includes not only urban ecosystems (Bolund and Hunhammar 1999), but the local cultural benefits they provide (Sander et al. 2010). Urban ecosystems are also "often of poorer quality than their rural equivalents" (Bolund and Hunhammar 1999, p. 299). Urbanization can shift demographic populations (Kahn 2002), impact surrounding native ecosystems (Heckmann et al. 2008), contribute to land-use change, pollution, and watershed stresses that impact rural landscapes.

According to Bolund and Hunhammar (1999), urban ecosystems include "street trees, lawns/parks, urban forests, cultivated land, wetlands, lakes/sea, and streams" (p. 294). Ecosystem services for urban areas are often generated in rural areas surrounding the city. Yet, the increase of local produced urban ecosystems could increase efficiency as well as "contribute to public health and increase the quality-of-life of urban citizens" (p. 294). For example, the planting of urban trees could improve the quality of air, and soft ground cover and vegetation could help reduce the level of noise pollution. Additionally, urban forests or urban tree cover may "enhance social, economic, and environmental conditions in urban environments" (Sander et al. 2010, p. 1655) specific to home ownership and dwelling.

Moreover, the development of urban areas can influence demographic trends (Kahn 2002). According to Kahn (2002), examples of demographic trends include: the aging of the baby boomer community, an increase in educational attainment, and a rise in minority populations. These demographic population changes can result in new median voters, impacting decisions surrounding myriad issues (e.g., public policy, health, social security, environmentalism, education, housing, economy, urbanization, and so forth).

Lastly, urbanization can have "ecologically significant and lasting effects on native ecosystems" (Heckmann et al. 2008). They found that as urbanization sprawls to include surrounding rural forested areas, rural populations are affected. For example, the Lake Tahoe Basin in the Central Sierra Nevada region has witnessed significant urban development. While urbanization of the area has increased tourism, recreation, and residential populations, it has also resulted in “reduced ecological integrity and vulnerability of remnant forests in this urbanizing landscape” (p. 2454).

If metropolitan areas grow into megapolitan subregions as anticipated, the Sacramento area south of the Lassen-Modoc is anticipated to increase by more than 40 percent between 2000 and 2030, expanding eastward to the Nevada State line, that is, to Reno and at a growth rate greater than that of Reno. Such urban systems may stress watersheds with both source-water demands from rural areas and storm-water outputs to them. Overall, it is likely that as urbanizing regions attempt to adjust to change, they may “place added stress on rural and wildland ecosystems that are connected to cities due to greater resource exploitation” (Grimm et al. 2008, p. 270). It is also important to note that more than 50 years of environmental protection efforts in the Lake Tahoe Basin, however, have retained “thousands of parcels of remnant native forest located throughout the urbanizing landscape” (Heckman et al. 2008, p. 2453), perhaps laying both ecological buffers and institutional precedents against urban impacts from Reno upon Lassen-Modoc ecosystem services.

Population Change in Rural Communities

In addition to the demographic trends requested, the Lassen-Modoc community members have asked:

- How do population increases and decreases affect rural communities in general?
- How might these demographic trends impact policy and decisionmaking?

Rural Population Change

In rural forested areas in California, Ferranto et al. (2011, 2012) found landownership was quickly moving away from larger properties owned by the few, such as agriculture land owned by ranchers and farmers, toward smaller parcels owned by the many, such as vacation or second homes owned by “in-migrants” (Ferranto et al. 2011, p. 184) or urban populations moving to rural areas. As new landowners are moving to rural areas, “changes in social values and demographic characteristics” (Ferranto et al. 2011, p. 184) have surfaced. For example, “new landowners often have less experience with vegetation management than traditional foresters and ranchers, and a greater focus on recreational and residential qualities” (Ferranto et al. 2012, p. 132). By contrast, others have found that rural forest living “contributes to environmental stewardship” and that rural population size and forest cover are positively correlated (Clement et al. 2015).

Research has shown that local jurisdictions that have restrictive land-use regulations are critical in slowing the rate of relative density gains in population (Kim et al. 2013). These regulations affect incoming rural populations, which are far from heterogeneous, and middle-aged urban citizens who are looking to gentrify while Fitchen (1995) details the immigration of the urban poor to rural areas. Such demographic discrepancies could be a consequence of small area populations being comprised of incomers who have experienced different types of migration (urbanization, lateral migration, or counter-urbanization) (Mitchell 2004).

As Lichter (2012) notes, “Ethnoracial change is central to virtually every aspect of rural America over the foreseeable future: agro-food systems, community life, labor force change, economic development, schools and schooling, demographic change, intergroup relations, and politics” (p. 3). With the racial makeup of rural communities changing, socioeconomic trends are changing as well. Considerations such as changes in family structure exacerbating economic inequality along racial lines, single motherhood decreasing intergenerational economic mobility, and gender inequality increasing from women bearing the financial brunt of raising children can be given new scrutiny as racial demographics in rural areas shift (McLanahan and Percheski 2008).

Because Lassen and Modoc Counties are expected to see a slight increase in Hispanic or multiracial populations, it is helpful to examine potential reasons for this demographic trend and how this trend may influence decision making. One possible explanation as to why Hispanic or multiracial populations may be gradually increasing in these counties may include land use in these areas. For example, agricultural activity is high in both Lassen and Modoc Counties (Lassen County Crop and Livestock Report 2015; Modoc County, California Local Hazard Mitigation Plan Update 2016). Moreover, the increase in Hispanic populations “may be explained by evidence that half of farm laborers and supervisors in the U.S. are Hispanic (ERS 2012)” (Charnley et al. 2018, p. 812).

Impact of Demographic Trends on Policies in Rural Areas

Specific to Lassen and Modoc Counties, the slow increase in Hispanic or multiracial populations may influence decision making surrounding forested land, including an increased support for environmental regulation. For example, within California, Kahn (2001) argued that Hispanics have been exposed to considerably more air pollution than other racial groups, specifically whites or blacks. In California from 1970 to 1990, as demographic trends shifted toward an increase in minorities, there was more public support for environmental regulation, particularly among black and Hispanic communities who were "consistently pro-environment" (Kahn 2002, p. 54).

Because baby-boomers tended to migrate toward urban areas, leading to a decline in rural employment, they are less likely to have direct ties with the land (Marcin 1993). Estimates of land-use change under different *policy* scenarios in the United States from 2001 through 2051 have indicated that urban areas will increase with population increase; cropland and rangeland will decrease; pasture will shift; and surprisingly, forested areas would increase overall. Throughout the United States "developed land area of the U.S. increased by 14.2 million hectares [35 million acres] between 1982 and 2003. Along with a projected U.S. population increase to more than 360 million individuals by 2030 is an expected continuation of expanding rural land development" (White et al. 2009, p. 37). With the acquisition of greater wealth, smaller families, and diverse families without children, more second homes are being purchased in rural lands, causing increased conflicts in land use and commodity production (Marcin 1993). The South Central and Great Plains regions of the United States experienced the greatest numbers of hectares of newly developed land per additional housing unit while the Pacific Coast and Rocky Mountains experienced the least (White et al. 2009).

However, as Brown et al. (2005) note, "(b)y 2000, the area of low-density, exurban development beyond the urban fringe occupied nearly 15 times the area of higher density urbanized development" (p. 1851). Consistent with research on landscape preferences, recent rural migration patterns show that most migrants are drawn to areas with a combination of forests and open lands, water area, and sparse croplands (McGranahan 2008). Chi and Marcouiller (2013) argue that the attractiveness of forests and wetlands to migrants is contingent upon whether these areas can be accessed through managed recreational areas. In addition, "Federal agricultural policies, such as crop price support programs, likely affect the total area of cropland in production and therefore the area of forests. Local policies regarding land uses affect the rate at which forest land is converted to developed uses" (Vose et al. 2012, p. 105).

In California, and throughout the Western United States, the restructuring of the timber industry in the 1990s provided opportunities for the growth of real estate investment trusts (REITs) on private timberland, which therefore laid a structure for the expansion of the wildland urban interface (Vose et al. 2012, p. 100, 109). Thus, public forest ownership predominates over private forest ownership in the Western United States (Vose et al. 2012, p. 98). Adapting to these changes requires an adaptive and flexible approach to land management practices. For example, Schaich and Plieninger (2013) offer the following recommendations:

> In public forests, close-to-nature management approaches could be complemented with binding goals aimed at promoting old-growth forest attributes and allowing site-specific variation in management practices. To conserve diversity in small-scale private forests, financial incentives and remuneration schemes for the provisioning of forest biodiversity and ecosystem services could be developed. Moreover, consulting and the provision of information on sustainable forest and conservation management could be intensified (p. 148).

Along with rural population change throughout the United States, several communities have turned to the expansion of prison developments as an economic driver, and specifically in Lassen and Modoc Counties, the developments of major State and Federal prisons have led to mixed social and economic impacts.

Impacts of Prisons on Rural Communities

Although crime rates have fallen sharply during the last quarter century (Gramlich 2017), 2.2 million people are currently being housed in the Nation's prisons and jails (The Sentencing Project 2017), representing a greater proportion of United States citizens in prison than any other country in the world (Walmsley 2013). The causes of mass incarceration in the United States vary by State, but statistical analyses have shown that the perceived need for more prisons and the ability to pay for prison development are the primary variables that drive or suppress prison populations (Spelman 2009). Despite experts' warning of the rise and perpetuation of mass incarceration, most States have not reduced their prison populations (Austin 2016). Austin (2016) explains: "Fueled by systemic changes in penal codes, sentencing practices, and Federal funding all designed to increase the use of imprisonment, most states have been unable to reverse the massive increase that has transpired over the past four decades" (p. 84). Meanwhile, in California, prison expansion in rural communities has increased and is perceived as one method of boosting rural employment and economies.

Economic and Social Impacts of Prisons on Rural Communities

Due to the presence of State and Federal prison facilities in the region, the Lassen-Modoc community members have asked for an overview of the effects of prison systems. We are able to address the following question generally, where we see issues relevant to the region:

- What are the economic and social impacts of prisons on local rural communities?

In 2002 Tracy Huling, a national expert on prisons and host communities, stated that the United States has more prisoners than farmers. Although it is necessary to limit Huling's observation to farmers who operate small family farms, recent data from the U.S. Department of Justice (Glaze and Kaeble 2014) and the U.S. Department of Agriculture (2014) show that more Americans are incarcerated (2,227,500) than are working as principal farmers (2,109,303), when "principal farmer" is defined as "the person primarily responsible for the day-to-day operation of the farm". Despite the fact that the incarceration rate appears to have leveled off throughout the United States (Garland et al. 2014), the widespread implementation of prisons in rural communities across the Nation has implications beyond punishment that influence social and economic trends (Eason 2016).

With the growth of the information and service sectors in the latter half of the 20th century, rural communities that once depended on agricultural and manufacturing industries find themselves economically stranded and struggling to maintain financial solvency (King et al. 2004). For several decades now, rural communities across the United States have struggled to stabilize their economies in the face of "farm crises, factory closings, corporate downsizing, shift to service sector employment and the substitution of major regional and national chains for local, main-street businesses," triggering deep and lasting change in rural communities (Huling 2002, p. 1). Government programs to alleviate poverty have primarily focused on urban communities, while rural economies were stimulated with farm subsidies or one-time infusions of capital, which did little to promote sustainable economic growth (King et al. 2004).

Many of these communities have embraced prison development as a means of economic stimulus to counter escalating poverty and unemployment rates, and to reverse population decline (Bonds 2006; Cherry and Kunce 2001). Because the prison system simultaneously faces deregulation and State divestment, prison development occurred as a combination of both public management and private, for-profit industry (Bonds 2006). Prior to the recession of 2008, "the influx of public and private funds related to the punishment industry, the growing demand for prison space and the seeming persistence of crime" led to the impression that prison development was "a 'recession-proof' strategy for rural community development and renewal" (Bonds 2006, p. 174).

The effectiveness of prisons as economic stimulators, however, became contentious among researchers (Burayidi and Coulibaly 2009), and an increasing amount of evidence illustrates the inadequacies of prison development at producing local economic growth (Huling 2002). For instance, Glasmeier and Farrigan (2007) found that, based on a diversity measure of industry sector earnings and employment, prisons have very little impact on the rural economy and that prison development is ineffective at stimulating economic growth. Four years later, another study by Glasmeier and Farrigan (2007) did observe "a limited economic effect on rural places in general, but may have a positive impact on poverty rates in persistently poor rural counties, as measured by diminishing transfer payments and increasing state and local government earnings in places with relatively good economic health" (p. 274).

Likewise, a study looking at the economic outcomes of prisons in rural counties in New York State (King et al. 2004) concluded that the counties that hosted prisons experienced no significant differences in economic gains or losses when compared to other rural counties that did not serve as host communities for prisons. In fact, host communities see very little economic benefits with the introduction of a prison and the surging privatization of prisons impedes employment growth (Genter et al. 2013).

Deller et al. (2001) note that the major appeals of rural communities to businesses include access to open space, natural amenities, as well as small-town values, all of which can be disrupted with the introduction of a prison. Hooks et al. (2010) observe that prisons tend to crowd out other alternative industries in rural areas, often resulting in economies dominated by a single industry. Once a prison becomes established in a community, it can have a deleterious effect on encouraging other industries to move to that community because of the economic dominance imposed by the prison on the local economy (Hooks 2010). In other words, the dilution of a prison-host community's socioeconomic profile acts as a disincentive in attracting other businesses (Burayidi and Coulibaly 2009). Thus, most researchers concur that siting a prison in a rural community does little to affect property values and crime rates generally stay static (King et al. 2004). Interestingly, along with individuals who have been victimized through direct criminal activity, the young and the well-educated express the most concern about the potential impacts that a prison might have on their communities (Maxim and Plecas 1983).

It is also recognized (see Huling 2002) that inmates recompense financially their host communities by making available more State and Federal funds, whose allocation is apportioned in accordance with population counts. In fact, Lotke and Wagner (2004) estimate that a prison with 1,000 beds can generate up to $100,000 annually in new revenues for the host community, even if acquiring much of this revenue comes at the expense of funding other public institutions, such as community colleges (Crookston and Hooks 2012). Furthermore, for every 100 inmates, 35 jobs are created in rural communities, with an average of 275 jobs created per prison (Beale 1996). Unfortunately, these jobs are not as beneficial to rural communities as they may appear. In fact, most public prison jobs do not go to local residents. Instead, 80 percent of jobs on average go to people living outside the host community (Gilmore 2007). Additionally, job competition is extreme in depressed rural areas, with rural residents competing for available positions in a wider-than-normal job market (Gilmore 2007).

Many rural prison sites are chosen not on the basis of whether or not the local workforce is capable of meeting the requirements of prison work, but rather because rural locations are often easier for out-of-town workers to reach from great distances (King et al. 2004). Although the King et al. (2004) study claims a static crime rate with the introduction of a prison to a rural community, other studies (Clear et al. 2003) have found a more complex relationship concerning crime rates and the primarily urban areas where the many inmates originate: "Our analysis revealed that increasing admissions to prison in one year have a negligible effect on crime at low levels and a negative effect on crime the following year when the rate is relatively low, but after a certain concentration of residents is removed from the community through incarceration, the effect of additional admissions is to increase, not decrease, crime" (Clear et al. 2003, p. 55).

Rural employment opportunities can be constrained further with the introduction of a prison labor force. Carlson (1992) observes that the prisons industry has the benefit of accessing captive workers who are accessible for community projects. "Work projects performed by prisoners for local government, churches, hospitals, libraries, and many other kinds of organizations are very common in prisons located in rural communities and small towns, and prison officials tout them as good community relations" (Huling 2002, p. 4). However, once local organizations become reliant upon prison labor, some from the existing community labor force, especially manual laborers, may become displaced, deepening local poverty rates (Gilmore 2007).

Ultimately, siting prisons in rural locations is both a blessing and a curse for the host community. Although most of the jobs offered by these prisons do not benefit the local labor force, some do.

Federal Considerations of Prison Populations

The demographic changes in Lassen County cited previously in *Population Shifts in Number and in Composition* are derived from statistics reported by the U.S. Census Bureau and the State of California. However, these population estimates are skewed because the county's major prison populations are counted in the Census Bureau data. Those statistics include the populations of two major California State prisons (California Correctional Center and High Desert State Prison) and one major Federal penitentiary (Federal Correctional Institution-Herlong). The Census Bureau counts prisoners as residents of the counties where they are incarcerated, even though most inmates do not have ties to those communities and almost always return to their home neighborhoods upon release (Gottschalk 2008). In Lassen County, the estimated population declines may be even more significant than reported by the Census Bureau. Therefore, the level of analysis provided in this science synthesis can be compared to statistics from local county reports that remove the prison population from its demographic data.

Lassen-Modoc stakeholders requested an accounting of how correctional facilities affect Federal data, funding, and decision making. They asked:

- How are prison populations represented in U.S. Census data, and how do prison populations impact funding?

Impact of Prison Populations on U.S. Census Data

Census data is used by Federal and State agencies to draw and redraw congressional lines at the State and Federal levels. Constituent to this data is the inclusion of the 2 million prisoners incarcerated in the United States according to the location of their incarceration, rather than their place of residence or sentencing (Kelly 2012). Additional State and Federal funds become available to communities that host these mostly temporary residents (Bonds 2006). Securing these fiscal incentives creates competition among similar communities, which employ various recruiting techniques to entice public and private developers, such as donating land, making public infrastructure accessible, as well as offering property and tax breaks (Huling 2002). Both State and Federal expenditures have been spent increasingly on incentivizing prisons into rural communities. In 1982, State prison expenditures totaled $6 billion and local expenditures equaled $3 billion; by 1999 total State expenditures had risen to $34.7 billion for prisons, and local expenditures reached $15.1 billion (King et al. 2004).

The U.S. Census Bureau has studied the impacts of including inmates in local population counts. Their 2006 study concluded that counting inmates according to their place of origin instead of their place of incarceration would increase operational costs for both the Census Bureau and the agencies that operate correctional agencies, decrease statistical accuracy, and impede efforts to count people living in different types of group quarters. An increase in operational costs of $250 million would occur because prisons do not have exhaustive records of address for all inmates and, as such, would require the Census Bureau to contract with prison officials to query each inmate in order to verify each address (The Sentencing Project 2017).

Prisons also offer local communities a political advantage because legislative and congressional lines are drawn according to inmate-inclusive population counts (Burayidi and Coulibaly 2009). Lotke and Wagner (2004) sum up the ramifications:

> Overall, in the States, counting urban residents as rural residents dilutes urban voting strength and increases the weight of a vote in the rural districts. In the rural prison districts, the real residents benefit because their own issues can receive individual attention from their representative on a scale unavailable elsewhere. In contrast, urban legislators are responsible not only to their "official" district but also those community members miscounted in the prison diaspora. One can only imagine the political negotiations of reapportionment, and how a plum like a prison can count (p. 599).

This "prison gerrymandering" inflates political representation of rural prison communities, which tend to mostly be white, at the expense of urban and minority communities, from which most prisoners hail (Skocpol 2017). As Kelly (2012) explains, "By drawing these phantom populations into districts that lean heavily toward the majority party, legislators can free up eligible voters from those districts to be distributed among neighboring marginal ones, thereby increasing that party's likelihood of winning additional seats in the state legislature" (abstract).

Skocpol (2017) outlines three obvious harms caused by prison gerrymandering. First, the collective voices of communities, both urban and rural, are manipulated by those who draw legislative lines. This practice has a corrosive effect on the foundations of democratic representation. This is especially concerning because, as noted above, the political clout of minority populations becomes further compromised. Second, prison gerrymandering elicits a dehumanization of inmates, a civil death upon inmates, which goes beyond most of the penological rationales adhered to by most contemporary jurisprudence doctrines. "[I]t is troubling to pretend that human beings who cannot vote, whose freedoms are heavily circumscribed, and who have little meaningful stake in community debates are equal 'constituents' of representatives who have no incentive to heed their views" (p. 1488). And third, the representational distortions caused by prison gerrymandering can inform political policy. Politicians who represent rural districts have strong incentive to support policies that maintain or increase incarceration rates because their political power depends, to some degree, on the continuation of inmates inhabiting their districts. This creates a positive feedback loop, in that mass incarceration occurs in districts where representatives are motivated to support policies that support mass incarceration.

Despite repeated calls by many organizations to have the Census Bureau count inmates according to where they are sentenced and not where they are housed (see Lotke and Wagner 2004), it appears that the Census Bureau is currently unwilling to change course on this matter. However, in 2010, the Census Bureau created and made available to States a granular data set that accounts for residents of *group quarters*, which encompasses prisons and gives individual States the ability to exclude prisoners from their redistricting data, should they choose to do so (Skocpol 2017). At this stage, the legal ramifications of prison gerrymandering continue to be debated in the courts and legislatures.

Indirect Costs of Prisons

In addition to the economic, political, and ethical issues discussed above, prison populations have related social consequences. In the United States, black and Hispanic individuals make up about 30 percent of the total population, which is significantly lower than the 56 percent of the total inmate population in the Nation that these two racial groups occupy (Vogel and Porter 2016). In fact, the incarceration rate for black males is nearly 7 times the rate for white males (Vogel and Porter 2016).

In 2009, the direct cost of incarcerating an individual was more than $20,000 (Spelman 2009), but many economists regard the social costs of incarceration, e.g., legitimate income that could have been earned by prisoners as well as reduced financial earnings for an inmate's family, to be twice the amount of the direct cost of incarceration (Donohue 2007; Kleykamp et al. 2008). Some economic benefits occur, however, in that incarceration preserves potential victims' money and possessions that would have otherwise been lost to crime (Western 2006).

Spelman (2009) contends that we have overshot the mark and are now spending money to incarcerate criminals who would not have committed a financial or corporeal crime to begin with. The implications of mass incarceration as a means of economic development has serious implications for democratic rights and processes. Pfaff (2015) suggests that any decisions to change the trend of national mass incarceration can come from the States, which can support alternatives to incarceration for non-violent criminals. Pfaff also notes the important role of the Federal government:

> …[T]he fiscal story suggests that (I) a state's willingness to incarcerate is tied to its financial ability (or flexibility) to punish, and (II) federal funding of penal practices is not a major contributor to that fiscal ability, though not necessarily an irrelevant one either. But the federal government does provide enough money that it could encourage and assist states in developing new and innovative ways to deal with offenders, and it could attempt to help rectify the glaring moral hazard problem that runs through the criminal justice system (p. 1600).

If Federal spending programs related to punishment and incarceration were to change the way expenditures are allocated to the States—e.g., by mandating an increase in rehabilitation instead of incarceration—it could retrench the booming prison population throughout the United States (Spelman 2009). Many rural economies that depend on prisons as main economic drivers would, however, be significantly influenced by such a change. At the State level, California has already begun to roll back its trend of mass incarceration. In 2011, the California legislature passed Assembly Bill 109, a.k.a. California Public Safety Realignment Act, which mandated reductions in the State's prison population of "non-serious, non-violent, non-sexual" criminals. Despite concerns that releasing such criminals into society would lead to an increase in crime rates, "Within just 15 months of its passage, Realignment reduced the size of the total prison population by 27,527 inmates, prison crowding declined from 181 percent to 150 percent of design capacity, approximately $453 million was saved, and there was no adverse effect on the overall safety of Californians" (Sundt et al. 2016, p. 315). Krisberg (2016) contends that the trends in California of decriminalizing non-violent crimes as well as, correspondingly, of reducing prison population counts will continue and that at present, California is pursuing cautious and gradual policy measures aimed at reducing population counts for even serious crimes.

References

Austin, J. 2016. Regulating California's prison population: The use of sticks and carrots. The ANNALS of the American Academy of Political and Social Sciences. 664(1): 84–107.

Beale, C.L. 1996. Rural prisons: an update. Rural Development Perspectives. 11(2): 25–27.

Bolund, P.; Hunhammar, S. 1999. Ecosystem services in urban areas. Ecological Economics. 29(2): 293–301.

Bonds, A. 2006. Profit from punishment? The politics of prisons, poverty, and neoliberal restructuring in the rural American Northwest. Antipode. 38(1): 174–177.

Brown, D.G.; Johnson, K.M.; Loveland, T.R.; [et al.]. 2005. Rural land-use trends in the conterminous United States, 1950–2000. Ecological Applications. 15(6): 1851–1863.

Burayidi, M.A.; Coulibaly, M. 2009. Image busters: how prison location distorts the profile of rural host communities and what can be done about it. Economic Development Quarterly. 23(2): 141–149.

Carlson, K.A. 1992. Doing good and looking bad: a case study of prison/community relations. Crime and Delinquency. 38(1): 56–69.

Charnley, S.; Jaworski, D.; Huber-Stearns, H.; [et al.]. 2018. Environmental justice, low income and minority populations, and forest management in the Northwest Forest Plan area. In: Spies, T.A.; Stine, P.A.; Gravenmier, R.; [et al.], tech. coords. Synthesis of science to inform land management within the Northwest Forest Plan area. Peer review draft of Gen. Tech. Rep. PNW-GTR-970. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 809–849 Chap. 10.

Cherry, T.L.; Kunce, M. 2001. Do policymakers locate prisons for development? Growth and Change. 32(4): 533–547.

Chi, G.; Marcouiller, D.W. 2013. In-Migration to remote rural regions: the relative impacts of natural amenities and land developability. Landscape and Urban Planning. 117: 22–31.

Clear, T.R.; Rose, D.R.; Waring, E.; [et al.]. 2003. Coercive mobility and crime: a preliminary examination of concentrated incarceration and social disorganization. Justice Quarterly. 20(1): 33–64.

Clement, M.T.; Ergas, C.; Greiner, P.T. 2015. The environmental consequences of rural and urban population change: an exploratory spatial panel study of forest cover in the Southern United States, 2001–2006. Rural Sociology. 80(1): 108–136.

Crookston, A.; Hooks, G. 2012. Community colleges, budget cuts, and jobs: the impact of community colleges on employment growth in rural U.S. counties, 1976–2004. Sociology of Education. 85(4): 350–372.

Deller, S.C.; Tsai, T.H.; Marcouiller, D.W.; [et al.]. 2001. The role of amenities and quality of life in rural economic growth. American Journal of Agricultural Economics. 83(2): 352–365.

Donohue, J.J., III. 2007. Economic models of crime and punishment. Social Research. 74(2): 379–412.

Eason, J.M. 2016. Reclaiming the prison boom: considering prison proliferation in the era of mass imprisonment. Sociology Compass. 10(4): 261–271.

Ferranto, S.; Huntsinger, L.; Getz, C.; [et al.]. 2011. Forest and rangeland owners value land for natural amenities and as financial investment. California Agriculture. 65(4): 184–191.

Ferranto, S.; Huntsinger, L.; Stewart, W.; [et al.]. 2012. Consider the source: the impact of media and authority in outreach to private forest and rangeland owners. Journal of Environmental Management. 97: 131–140.

Fitchen, J. 1995. "The single-parent family," child poverty, and welfare reform. Human Organization. 54(4): 355–362.

Garland, B.; Hogan, N.; Wodahl, E.; [et al.]. 2014. Decarceration and its possible effects on inmates, staff, and communities. Punishment & Society-International Journal of Penology. 16(4): 448–473.

Genter, S.; Hooks, G.; Mosher, C. 2013. Prisons, jobs and privatization: the impact of prisons on employment growth in rural U.S. counties, 1997–2004. Social Science Research. 42(3): 596–610.

Gilmore, R.W. 2007. Golden gulag: prisons, surplus, crisis, and opposition in globalizing California. Berkley, CA: University of California Press. 412 p.

Glasmeier, A.K.; Farrigan, T. 2007. The economic impacts of the prison development boom on persistently poor rural places. International Regional Science Review. 30(3): 274–299.

Glaze, L.E.; Kaeble, D. 2014. Correctional populations in the United States, 2013. NCJ 248479. Washington, DC: U.S. Department of Justice Bulletin. https://www.bjs.gov/content/pub/pdf/cpus13.pdf (8 Sep. 2017).

Gottschalk, M. 2008. Hiding in plain sight: American politics and carceral state. Annual Review of Political Science. 11: 235–260.

Gramlich, J. 2017. Five facts about crime in the U.S. Washington, DC: Pew Research Center. https://www.pewresearch.org/fact-tank/2017/02/21/5-facts-about-crime-in-the-u-s/ (8 Sep. 2017).

Grimm, N.B.; Foster, D.; Groffman, P.; [et al.]. 2008. The changing landscape: ecosystem responses to urbanization and pollution across climatic and societal gradients. Frontiers in Ecology and the Environment. 6(5): 264–272.

Heckmann, K.E.; Manley, P.N.; Schlesinger, M.D. 2008. Ecological integrity of remnant montane forests along an urban gradient in the Sierra Nevada. Forest Ecology and Management. 255(7): 2453–2466.

Hooks, G.; Mosher, C.; Genter, S.; [et al.]. 2010. Revisiting the impact of prison building and job growth: education, incarceration, and county-level employment, 1976–2004. Social Science Quarterly. 91(1): 228–244.

Huling, T. 2002. Building a prison economy in rural America. In: Mauer, M.; Chesney-Lind, M., eds. Invisible punishment: the collateral consequences of mass imprisonment. New York: The New Press: 1–10.

Kahn, M.E. 2001. The beneficiaries of Clean Air Act regulation. Regulation Magazine. 24(1): 34–39.

Kahn, M.E. 2002. Demographic change and the demand for environmental regulation. Journal of Policy Analysis and Management. 21(1): 45–62.

Kelly, J.P. 2012. The strategic use of prisons in partisan gerrymandering. Legislative Studies Quarterly. 37(1): 117–134.

Kim, J.H.; Deal, B.; Chakraborty, A. 2013. Parsing density changes: an outcome-oriented growth management policy analysis. Journal of Housing and the Built Environment. 28(3): 529–546.

King, R.S.; Mauer, M.; Huling, T. 2004. An analysis of the economics of prison siting in rural communities. Criminology Public Policy. 3(3): 453–480.

Kleykamp, M.; Rosenfeld, J.; Scotti, R. 2008. Wasting money, wasting lives. New York: Drug Policy Alliance. http://www.drugpolicy.org/sites/default/files/WMWL_Final_2012.pdf (8 Sep. 2017).

Krisberg, B. 2016. How do you eat an elephant? Reducing mass incarceration in California one small bite at a time. Annals of the American Academy of Political and Social Science. 664(1): 136–154.

Lassen County Crop and Livestock Report. 2015. Lassen, CA: California Department of Food & Agriculture and The Honorable Board of Supervisors of Lassen County. https://www.lassencounty.org/sites/default/files/images/2015%20Crop%20Report.pdf (7 Sep. 2017).

Lichter, D.T. 2012. Immigration and the new racial diversity in rural America. Rural Sociology. 77(1): 3–35.

Loeffler, R.; Steinicke, E. 2007. Amenity migration in the U.S. Sierra Nevada. The Geographical Review. 97(1): 67–88.

Lotke, E.; Wagner, P. 2004. Prisoners of the census: electoral and financial consequences of counting prisoners where they go, not where they come from. Pace Law Review. 24(2): 587–607.

Marcin, T.C. 1993. Demographic change: implications for forest management. Journal of Forestry. 91(11): 39–45.

Maxim, P.; Plecas, D. 1983. Prisons and their perceived impact on the local community: a case study. Social Indicators Research. 13(1): 39–58.

McGranahan, D.A. 2008. Landscape influence on recent rural migration in the U.S. Landscape and Urban Planning. 85(34): 228–240.

McLanahan, S.; Percheski, C. 2008. Family structure and the reproduction of inequalities. Annual Review of Sociology. 34: 257–276.

Mitchell, C.J.A. 2004. Making sense of counterurbanization. Journal of Rural Studies. 20(1):15–34.

Modoc County, California local hazard mitigation plan update 2016. Public review draft. Modoc, CA: Modoc County. 438 p. https://www.cityofalturas.us/Modoc%20County%20LHMP%203.4.16%20Draft.pdf (25 Aug. 2017).

Pfaff, J.F. 2015. Federal sentencing guidelines in the states: some thoughts on federal grants and state imprisonment. Hastings Law Journal. 66: 1567–1600.

Sander, H.; Polasky, S.; Haight, R.G. 2010. The value of urban tree cover: a hedonic property price model in Ramsey and Dakota Counties, Minnesota, USA. Ecological Economics. 69(8): 1646–1656.

Schaich, H.; Plieninger, T. 2013. Land ownership drives stand structure and carbon storage of deciduous temperate forests. Forest Ecology and Management. 305: 146–157.

Skocpol, M. 2017. The emerging constitutional law of prison gerrymandering. Stanford Law Review. 69: 1473–1540.

Spelman, W. 2009. Crime, cash, and limited options: explaining the prison boom. Criminology & Public Policy. 8(1): 29–77.

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.

State of California, Department of Finance, 2012. E-1 state/county population estimates with annual percentage change – January 1, 2011 and 2012. Sacramento, CA.

The Sentencing Project 2017. Fact sheet: trends in U.S. corrections. Washington, DC: The Sentencing Project. https://sentencingproject.org/wp-content/uploads/2016/01/Trends-in-US-Corrections.pdf (8 Sep. 2017).

Sundt, J.; Salisbury, E.J.; Harmon, M.G. 2016. Is downsizing prisons dangerous? The effect of California's Realignment Act on public safety. Criminology & Public Policy. 15(2): 315–341.

U.S. Census Bureau. 2006. Tabulating prisoners at their "permanent home of record" address. U.S. Census Bureau Report. Washington, DC. https://felonvoting.procon.org/sourcefiles/tabulating_prisoners.pdf (8 Sep. 2017).

U.S. Department of Agriculture. 2014. Farm demographics: U.S. farmers by gender, age, race, ethnicity, and more. Data Series ACH12-3. https://www.agcensus.usda.gov/Publications/2012/Online_Resources/Highlights/Farm_Demographics/Highlights_Farm_Demographics.pdf (8 Sep. 2017).

Vogel, M.; Porter, L.C. 2016. Toward a demographic understanding of incarceration disparities: race, ethnicity, and age structure. Journal of Quantitative Criminology 32(4): 515–530.

Vose, J.M.; Peterson, D.L.; Patel-Weynand, T., eds. 2012. Effects of climatic variability and change on forest ecosystems: a comprehensive science synthesis for the U.S. Forest Sector. Gen. Tech. Rep. PNW-GTR-870. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 265 p.

Walmsley, R. 2013. World prison population list. 10th ed. London, UK: International Centre for Prison Studies. https://www.apcca.org/uploads/10th_Edition_2013.pdf (8 Sep. 2017).

Western, B. 2006. Punishment and inequality in America. New York: Russell Sage Foundation. 264 p.

White, E.M.; Morzillo, A.T.; Alig, R.J. 2009. Past and projected rural land conversion in the U.S. at state, regional, and national levels. Landscape and Urban Planning. 89(1-2): 37–48.

Winter, P.L.; Long, J.W.; Lake, F.K.; [et al.]. 2014. Wet meadows. In: Long, J.W.; Quinn- Davidson, L.N.; Skinner, C.N., eds. Broader context for social, economic, and cultural components. Gen. Tech. Rep. PSW-GTR-247. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 501–541. Chap. 9.1.

Chapter 5.3. Ecosystem Services and Public Land Management

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The concept of ecosystem services connects the activity of environmental and natural resource management to desired outcomes of human economic utility, social well-being, and cultural health. A desire for mutual sustainability between human interests and ecological cycles is inherent in the ecosystem services approach (Patterson 2014). The complex network of interactions between natural processes and human impacts is referred to as a socioecological system.

Ecosystem services refer to the material and nonmaterial benefits derived from natural ecosystem processes, as well as the human regulation and support services required for the production and sustainability of those benefits (Vose et al. 2012).

Social science literature details a mixed history of natural resource economies and their impacts on local communities. Cycles of boom and bust are evident, especially through the industrial resource extraction practices of Old West economies (Frickel and Freudenburg 1996). While service-based sectors rose in the New West, so too did ecological rationales and environmental mandates surrounding the use of natural resources. The *Science Synthesis to Support Socioecological Resilience in the Sierra Nevada and Southern Cascade Range* (hereafter, Sierra Nevada Science Synthesis; Long et al. 2014; see Chapter 1.1, Dumroese, this synthesis, *The Northeastern California Plateaus Bioregion Science Synthesis: Background, Rationale, and Scope*) discusses this social-economic transition as one moving from industrial extraction to amenity value consumption. The changes in land use, economic activity, and social relations that accompany this shift from production to the service sector now require public lands managers to balance a "triple bottom line" of ecological, economic, and social concerns (Winter et al. 2014, p. 497). Negotiating the tensions among these three often divergent perspectives, and doing so with a broad scope and strategic foresight, is at the heart of ecosystem services management in the U.S. Department of Agriculture, Forest Service.

More thoroughly reviewed in the Sierra Nevada Science Synthesis, the concept of ecosystem services has been used in Forest Service management to:

- communicate the value of forests;
- collaborate with stakeholders to define stewardship objectives;
- determine the tradeoffs between diverse goals and stakeholders;
- provide options for forest restoration; and
- support emergent markets for ecosystem services that economically benefit the forest and the community (Patterson 2014).

Although not necessarily step-wise, each type of action informs others: Culture- and value-laden social explorations inform community negotiations, and these negotiated discoveries provide frameworks for building economic mechanisms. Each type of management process is subject to revision and adjustment. The term "ecosystem services" therefore expresses a circulating or iterative approach to human and environmental interdependence in the provision of natural resource benefits.

Forest Service managers were authorized in the 1970s to shift from a relatively narrow, progressive-era commercial imperative of only timber and watershed management, toward sustainable models of land management that also included wildlife, recreation, and grazing. Thus, by the late 1990s the national policy of ecosystem management for sustainability had superseded the previous economic goal of timber industry stability, observable in increased environmental litigation and loss of timber revenue (Nelson 2006). As an alternative to a declining timber industry, the ecosystem management model prioritized environmental protection (Charnley et al. 2018a). These strategies were seen as capable of delivering ecological supports for forests and economic benefits to communities (Moseley and Toth 2004). It is suggested that seeking a viable combination of economic development and ecosystem health may also be able to address inequalities that exist in many rural communities. Rural communities

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paradoxically often depend upon conservation programs and define the feasibility of such ecosystem service initiatives on the local level (Ninan and Inoue 2013; Turner et al. 2003).

General interests of the communities near the Lassen and Modoc National Forests (hereafter the Lassen, the Modoc, or the Lassen-Modoc) regarding ecosystem services are based on economic questions such as:

- What are the economic benefits of biomass, timber harvest, and recreation?
- What are the supply and demand processes for water, timber, range, recreation, and multiple use?

Since the 1990s forest managers have been charged with incorporating the economic benefits of ecosystem services with the realities of demographic trends, toward qualities of social well-being, and with values of cultural knowledge. Ecosystem services thus require a multifaceted approach—one not solely based on economic metrics—because ecosystem service products and processes under public land management in the United States very often do not take place in markets.

Variable and Local: Relevance for the National Forests

In response to observed and expected changes in climate, an "all-lands" approach to socioecological resilience has been called for. This approach surpasses the traditional boundaries between local/regional, rural/urban, and public/private domains. It is advocated because ecosystem services are increasingly vulnerable to biophysical disruptions (e.g., climate and fire) and subject to social responses (e.g., cultural and institutional) that do not abide by those political boundaries (Vose et al. 2012). A recent science synthesis of climate impacts on forest ecosystems (Vose et al. 2012) more thoroughly discusses this in the spheres of rural, urban, and wildland urban interface settings and notes that these impacts pose variable challenges to ecosystem services:

> Future changes in forest ecosystems will occur on both public and private lands and *will challenge our ability to provide ecosystem services desired by society,* especially as human populations continue to grow and demands for ecosystem services increase. *Climate change effects in forests are likely to cause losses of ecosystem services in some areas* (e.g., timber production, water supply, recreational skiing), *but they may improve and expand ecosystem services in others* (e.g., increased growth of high-elevation trees, longer duration of trail access in high-snow regions) (Vose et al. 2012, p. v, emphasis added).

While climate models have become more sophisticated, what is less understood are the capacities of communities to respond to these environmental changes (Rocca et al. 2014). Despite the climate-related pressures on forest ecosystem services, proponents of socioecological resilience also remind us that social choices and actions will shape the provision of scarce or shifting resources (Vose et al. 2012). Thus, managing interdependent, resilient socioecological systems call for collaborative participation in adaptive governance as these systems change. This portion of *Ecosystem Services* is informed by this management philosophy and attempts to respond to the following community-generated question:

- What economic and social benefits of ecosystem services do the Lassen and Modoc provide for local and regional communities?

While scientific peer review literature on ecosystem services directly related to the Lassen-Modoc is limited, a general review of the ecosystem services framework suggests that decisionmakers think about the above question: (1) at relevant scales, which are the local and regional levels defined previously in Chapter 5.2 (Flores and Russell, this synthesis, *Demographic Trends in Northeastern California*); (2) with practical means of institutional implementation, considered here by facilitation between forest management and community engagement, addressed in Chapter 5.4 (Flores and Stone, this synthesis, *Community Engagement in the Decisionmaking Process for Public Land Management in Northeastern California*); and (3) through the alignment of economic and ecological interests, noted in the *Forest Management Practices* subsection of this chapter.

Why are local-level scales of collaboration emphasized? One reason is the quality of measurement. Although large-scale economic valuations of forest ecosystem services reveal net benefits of ecosystem conservation, in general, economic analyses of ecosystem services are still developing and often found lacking (Kenter et al. 2015; Ninan and Inoue 2013). This is an indication of the relatively early stage of these economic models and of the complexity involved in quantifying variable, interdependent, and often intangible ecological system benefits. Furthermore, the constraints of ecosystem services valuation are not simply informational: Attempts to determine monetary value may potentially reveal that our current political systems and economic markets are, in many ways, incompatible with the ecological health and social well-being communities may desire. Patterson (2014) reminds us that market-based tools are not entirely adequate, because “the vast majority of ecosystem services are not and will never be marketable” (p. 555).

A second reason is that local scale also prioritizes community capacity and trust. A primary socioeconomic challenge for ecosystem services managers is to mediate the distinctions between market values and social values in an innovative manner with stakeholders. As Winter et al. (2014) encourage, “Decisionmakers who have a thorough knowledge of local social and economic conditions will also be better positioned to make decisions that draw on the existing capacity in a community, and help build local capacities that need to be developed by directing resources accordingly” (p. 615). Or, as Turner and Daily (2008) caution:

> Stakeholder perceptions, property rights and institutional arrangements are thus important components of any scheme to capture benefits on a practical and lasting basis. Failure to recognize and accommodate these components invites a lack of trust, accountability and legitimacy (p. 28).

The Sierra Nevada Science Synthesis notes that the shift in the scale and quality of approach requires a corresponding shift in evaluation criteria: There is a move away from fixed quantitative targets when studying landscape-level implications. Instead, resilience-based approaches typically favor plans “to reduce vulnerability and strengthen capacity to respond and adapt,” that is, to flex with change over time, across geographic areas, and with the involvement of local community members (Long et al. 2014, p. 94).

A recent study of forest adaptation frameworks notes that some conceptions of governance over climate adaptation (and therefore how public forest ecosystem services are supported by the Forest Service) are misleading. Uniform or automatic Federal and State policies—which have been historically prominent in the forestry sector—may offer guidance and administration. Yet related to the above comments about flexibility, local politics, local resources, and community-level capacity are the drivers when community choice and commitment are vital to adaptation plans (Wellstead et al. 2014). The public meeting and comment process has shaped this science synthesis and is intended to frame the land management plans for the Lassen and the Modoc.

Social Benefits of Ecosystem Services

Forest ecosystem services that provide physical sustenance by serving as catchments for drinking water, filters for air quality, timber for fuel and wood products, and so on will be discussed later regarding economic benefits and regarding community engagement. Here, we discuss the social relevance of forest ecosystem services, directed to the questions:

- What social benefits of ecosystem services do forests provide for local and regional communities?
- What are the different social and nonmarket cultural values that local residents, visitors, and different user groups attach to forest land?

Relationships and Realities: Social Values Inform Human Well-Being

Shared social values inform relations between the Forest Service and local communities, and values also grow out of the decision processes they make together for human well-being in the socioecological system. The public hearings among Lassen-Modoc stakeholders illustrate the “deliberation through the public sphere, public debate, and consultation [which some argue] are needed to articulate shared and social values” (Kenter et al. 2015, p. 91).

Social: Interpersonal relationships and activities on small scales between individuals as well as networks on larger group, organizational, community, or society levels.

Decision making in natural resource conservation is shaped by social values. Individuals have (1) underlying value orientations that shape their *perceptions* of the world, including how they understand the natural environment, usually in human-centered or eco-centered ways. People also have (2) values that they assign to the *things* within their perceived worlds, including ecosystem service benefits, which may be valuable in themselves, for use, or for nonuse. Social values are emotionally, historically, culturally, and politically driven. And while they are influenced by the scientific, rational, or economic concerns about an ecosystem, social values may be distinct from those realms (Ives and Kendal 2014).

Social values: Commonly held principles of meaning or importance that exist beyond a singular person, i.e., through a deeper cultural tradition, within particular social context, and on a scale beyond individual values.

Accordingly, the social sciences often study the subjective, qualitative aspects of underlying and assigned values and investigate the value relationships that connect particular people with certain places. Theory suggests that certain social values influence a person's or a society's disposition toward particular attitudes and behaviors—about wildlife, land use, clearfell timber harvesting, etc. Generally, stronger use-values are associated with accepting higher or more intense timber production, while stronger ecological and cultural values are associated with adopting the nonmarket functions of forests (Ives and Kendal 2014).

Social benefits: The enhancements of human relationships and choices to improve human life or well being with interpersonal, material, or health outcomes.

Mentioned casually above, the term *well-being* is often used in social scientific literature to express these human social benefits, but well-being is also referenced by the terms "welfare" or "quality of life" (Balmford and Bond 2005; Bieling et al. 2014; Charnley et al. 2018a; Gasparatos et al. 2011; Krekel et al. 2016). Social benefits are the intangibles, or externalities, for which conventional economic measures can only provide a proxy (Bawa 2017, p. 2). Some of the social benefits of engaging in forest ecosystem services include: a sense of present and future security of one's community and place, through balancing production and conservation; strengthened social relationships and personal identities, through negotiating the management of ecosystem services; and the social agility that comes through practicing collaboration and adaptation over time.

Coupling and Change: Values Connect Ecosystem Services With Social Benefits

Social values are associated variably with different forest ecosystems services. Before considering the variety of potential *Economic Benefits of Ecosystem Services* in the next subsection, we return to the above idea that social values influence a person's or a group's disposition toward certain attitudes and behaviors—and *choices*—regarding their community's forest ecosystem service interests. Presented below are examples of how social values are attached to land conservation and ownership, alternative fuel, fire management, and recreation to produce social benefits.

Research about ecological values are relevant to understanding values of the Lassen-Modoc communities, including consistent findings that:

(i) values change over time,

(ii) values differ between groups of people,

(iii) multiple values can be assigned to the same places,

(iv) multiple pathways exist between values, attitudes, and behaviors towards ecosystems, and

(v) values influence people's judgment of management decisions (Ives and Kendal 2014, p. 70).

For example, land is increasingly valued for the preservation of local open spaces, as opposed to cultivation, production, or restoration. "The personal values many people may associate with local open space lands," whether for recreation, aesthetic, or environmental characteristics, arise in the immediate contexts of where community members live and work, that is, "in people's daily lives" (quoting Kline 2006, p. 646; with sentiment by Bawa 2017). Yet degraded spaces are becoming more prevalent, and the availability of intact landscapes is decreasing. This tension between increasing social value for open space and declining ecosystem health is expected to accelerate the social values that demand conservation (Hjerpe et al. 2015).

This applies also to forestlands, where values have changed over time to valuing timber production and *material* services less and valuing ecological, spiritual, aesthetic, cultural, and recreational *experiences* more (Barrio and Loureiro 2010; Bieling et al. 2014; Ives and Kendal 2014). These occur in both personal and community realms. In California, some forest residents have expressed personal "wildlife values" of the surrounding forests, with statements about the biological diversity of a "beautiful valley … full of grizzly bear and moose and wolves and the few remaining caribou" (Bixler 2014, p. 164). Forest residents also describe community values of working together for wildlife conservation (Bixler 2014). Others have expressed community values surrounding land "preservation" and "protecting the environment," and scenic values of "[living] near natural beauty" (Ferranto et al. 2011, p. 191). But they have also expressed land conservation as a personal value, specifically for the benefit of their posterity or for individual financial benefit, including the provision of conservation trusts or easements.

The trend in some areas of California has been toward lifestyle values centered on amenity and investment (Ferranto et al. 2011). This value shift has occurred with a parallel shift in forest land ownership, wherein private land has shifted as relatively fewer ranchers and farmers divest large parcels of land into drastically smaller parcels of land to relatively more landowners. With these changes in land ownership come "changes in social values and demographic characteristics" tied to the land (Ferranto et al. 2011, p. 184). For example, "new landowners often have less experience with vegetation management than traditional foresters and ranchers, and a greater focus on recreational and residential qualities" (Ferranto et al. 2012, p. 132).

Another coupling of social values with ecosystem services is the development of bioenergy. In forest communities like the Lassen-Modoc, bioenergy projects for rural development and well-being are linked to social values about the role and funding for local education systems to teach applicable skills for bioenergy production. Such projects are also related to values about equity and justice, which shape whether subsidizing opportunities for smallholders is appropriate and illuminate whether bioenergy production places disproportionate stress on women or other subpopulations. Values regarding efficiency determine if local-scale benefits of bioenergy development outweigh land uses like food production or potential health effects. Finally, differing values about investment in the public good are connected to questions about how private leases for bioenergy development on federally managed land are balanced by democratic processes and industry regulation (Gaspartos et al. 2011). It is important to note shifts in values vary between and within communities. We would expect groups within the Lassen-Modoc to hold different social values that create both overlapping and contradictory interests in their local forest ecosystem services. The ecosystems that provide benefits to human society are changing rapidly and potentially in a restrictive direction. Therefore, we may expect the social values connected to ecosystem services to be impacted in similar ways—swiftly, variably, perhaps with great difficulty, but not necessarily negatively. These impacts to values may catalyze people to engage in social action about their environments in new ways, toward what the community determines is for their own well-being, if not also for the good of their forest ecology (Balmford and Bond 2005; Goldman 2010).

An example of social values changing toward creative solutions is the shift during the past 2 decades from valuing fire suppression less to valuing fire management more. Moritz and colleagues (2014) call this "learning to coexist with wildfire," (p. 58) a phrase that recognizes the coupling of natural and human systems. Such awareness has developed in rural forest communities like those of the Lassen-Modoc area more rapidly than in the wildland urban interface. If we extrapolate from Kline (2006), this may be because rural members living and working closer to their ecosystem in their everyday lives, having a more attuned awareness of their environment compared to their exurban neighbors. Materially and ecologically, prescribed fire has been shown to reduce the financial cost of suppression and to improve forest health in the Western United States. But in terms of social benefits, managed fire contributes to well-being by reducing risk to the lives of firefighters and by increasing evacuation success of residents (Moritz et al. 2014). Public health and safety are also affected by wildfire intensification and degraded air quality. Reducing the intensity of wildland fire through management also would allay smoke-related effects such as respiratory and pulmonary disease, compromised visibility, and increased risks on roadways and in air traffic (Stavros et al. 2014).

Eco-tourism and recreation represent social value/ecosystem service combinations with more mixed outcomes for local communities (fig. 5.3.1). Nature-based tourism has been a growth industry with associated economic benefits (Balmford and Bond 2005). It has also been shown, however, to "detrimentally [affect] the social and cultural fabric of local communities," especially when it is a consumer activity where not only cultural

Figure 5.3.1—Eco-tourism and nature-based recreation can have positive economic and social effects such as generating tourism revenue for local communities and creating greater awareness, understanding, and support for environmental conservation. Nonetheless, eco-tourism can also have negative effects such as producing single-industry economies and changing the social and cultural identity of local communities (photo by Dawn M. Davis, used with permission).

practices become marketable commodities, but where people of the community also become products (Stem et al. 2003, p. 388). If economic benefits are distributed unevenly, the social networks of communities and families become vulnerable to fracture. Commodification may also dramatically shift traditional values about natural environments, especially among tribal cultures or in isolated communities. Outdoor recreation, however, is largely defined by user-group aesthetic or athletic preferences and tends toward social engagement that is more experiential than consumptive. Recreation is directly related to a general sense of well-being and improved health and illustrates how "forestlands could also act as a common ground among neighboring communities that helps to strengthen societal bonds" (Bawa 2017, p. 1). Hunt and colleagues note that these value distinctions between tourism operators and independent recreationists create the potential for conflict (Hunt et al. 2009).

Finally, social well-being is also expressed in the successes or sufferings that are deeply intertwined with our economic livelihood. Employment opportunities provided in the ecosystem services sector have social value and use-value, and "both contribute to welfare and utility" (Hjerpe et al. 2015, p. 47). The next section more directly discusses economic benefits of ecosystem services that are of interest to the Lassen-Modoc.

Economic Benefits of Ecosystem Services

The previous section gave examples of how ecosystem services are connected with social benefits through social values and lifestyles, but it cannot ignore the economic and commodity values that are basic to the realities of human livelihood. Economic benefits are particularly important to the communities around the Lassen and the Modoc, which are situated in counties recognized as having high economic distress and as having endured concentrated declines in forest products employment (Winter et al. 2014). Lassen and Modoc Counties were classified in 2015 as nonrecreation dependent, nonmanufacturing dependent, nonretirement destination, low-employment, nonmetropolitan counties (Charnley et al. 2018b). Thus, determining if and how these forest communities align their economic necessities with their social and ecological interests will crucially shape their ecosystem services opportunities and land management plans (Turner and Daily 2008).

This section considers the following question in terms of opportunities in forest products, forest services, and forest-derived energy markets:

- What economic benefits of ecosystem services do forests provide for local and regional communities?

We have added to limited research on the economic benefits of ecosystem services on the Lassen-Modoc with more general benefits forests provide for local and regional communities. As mentioned in the beginning of this chapter and more thoroughly reviewed in the Sierra Nevada Science Synthesis (see Chapter 9.4 in Long et al. 2014), ecosystem services emerged in the 1990s as the multiple-use management paradigm that replaced a one-dimensional timber industry in the Northwest. Ecosystem services as a business sector contributes to economic benefits in a number of ways, including jobs in postdisturbance restoration, recreation, tourism, and infrastructural services; as well as in grazing, biomass removal, nontimber forest products (NTFPs), and, of course, continued timber production (see Chapter 9.5 in Long et al. 2014).

Residents of forested regions often take pride in the surrounding landscape and demonstrate pride through personal and collective action, including economic innovation based on their forest resources. "The rules and norms that these communities formed for their forests tend to correspond to their interest in and knowledge about the products they use" (Poteete and Welch 2004, p. 309). For example, collaborative forest landscape restoration is one project the Lassen has participated in and has knowledge about (Winter et al. 2014). Now, however, the Lassen-Modoc community has expressed interest in moving beyond restoration projects and wishes to examine the possibilities for diversifying their market opportunities in forest products, forest services, and forest energy.

Forest Products: Timber and Nontimber Forest Products

The forest products industry in California is predominantly concentrated in the Northern Sierra Nevada region, including Lassen, Modoc, Plumas, and Sierra Counties. A variety of factors such as changes in regulation, policy, economics, timber supply, wildland fire, and technology have significantly decreased forest industry production. The decrease in timber production has sparked increases in transportation costs, fuel use, and carbon emissions related to timber harvesting; and parallels decreases in the price of timber and local employment (Morgan et al. 2004).

Distinct from the more heavily forested areas of coastal Northern California, the Modoc Plateau in the northeastern-most ecoregion of the State is approximately 50 to 60 percent forest area, primarily of juniper softwoods (see Chapter 2.1, Moser, this synthesis, *Understanding and Managing the Dry Conifer Forests of Northeastern California*). It contains the comparatively lower carbon densities that characterize adjacent areas of Nevada to the east and the Great Basin as a whole, as opposed to other ecoregions of Northern California (Hicke et al. 2007). Drought is becoming the hallmark of climatic temperature rise in the Southwest United States and will lead to a greater likelihood of compromised forest productivity, increased tree mortality, reduced biodiversity, and the resulting capacity for more intense, larger-extent fires (Rocca et al. 2014; Vose et al. 2012; Wellstead et al. 2014). The Lassen-Modoc area is expected to experience this warming, drying scenario (Vose et al. 2012). Timber markets and property values may be affected negatively in private-ownership economies. These private economic losses are expected to have parallel declines in public goods such as aesthetic values, water quality, and recreational space in the national forests (Hicke et al. 2007; Vose et al. 2012).

A review of literature by Pramova and colleagues (2012) suggests ways forests can support adaptation to changing ecosystem conditions. First is the obvious provision of goods to local communities, with timber and nontimber forest products that add security to domestic food consumption and diversity to commercial sales (fig. 5.3.2). Second, by contributing to soil improvement and windbreaks, trees help regulate or stabilize the microclimates of agricultural fields for better production. Third, forested watersheds regulate water and protect soils. Each of these contributes to a combined ecological and economic longevity.

Significant changes have taken place for the type, quantity, and availability of Federal timber sale opportunities. In the 1990s, the Forest Service shifted to an "ecosystem management" and wildfire risk reduction paradigm. Harvest of larger, older trees declined, and new policies and programs such as the Northwest Forest Plan emphasized science-based management to restore ecosystems and protect biodiversity (Spies and Duncan 2009). In addition, with growing concerns about wildfire, several policies including the National Fire Plan and Healthy Forests Restoration Act followed in the early 2000s to remove hazardous fuels and restore forest health to reduce wildfire risk (Steelman and Burke 2007). These earlier shifts have led to an array of challenges and realities for forest-based businesses throughout the United States that are adjacent to national forests, which vary across different forest regions (Davis et al. 2018). Accompanying these shifts are impacts to forest-based businesses adjacent to public lands. In a study of timber-purchasing businesses active in six Forest Service regions where ecosystem restoration and wildfire risk reduction policies have prevailed, Davis et al. (2018) found that most businesses purchase small-diameter timber (8 inches diameter breast height or less) and had sought business assistance, most commonly from accountants and lending institutions. Secondly, those with the greatest dependence on Federal timber—76 percent or more of their supply from Federal sources—were less likely to have sought assistance of any kind. Thus, they suggest that more attention is needed to the timing, quantity, and types of supply that Federal lands offer and how this affects business success, particularly to understand how design of timber sales, service contracts, and stewardship contracts and sales may better serve businesses and allow them to produce community economic outcomes (Davis et al. 2018).

For example, one such study by Daniels et al. (2018) conducted an economic analysis of two contracts on the

Figure 5.3.2—Forest management can provide local communities with timber products that add economic diversity to rural communities. However, increased drought and wildfire, as well as shifts in the timber market industry create significant challenges to the potential economic benefits of timber production (photo by Ken Sandusky, Forest Service).

Mount Hood National Forest in Western Oregon. The study examines economic contributions from stewardship contracts and how they compare against county revenue-sharing systems such as Secure Rural Schools funding. The findings from this case study show that: (1) commercial thinning, service work, and retained receipts projects all contributed to local economic activity; (2) expenditures accounted for $4 million in output and generated 36 jobs; and (3) benefits were distributed across a wider variety of economic sectors than timber harvesting alone. Therefore, county-level expenditures on commercial thinning, service work, and retained receipts projects greatly exceeded what could be expected from payments to counties' revenue sharing (Daniels et al. 2018).

Timber resources in Lassen and Modoc Counties previously have been evaluated across all ownerships, together with Shasta, Siskiyou, and Trinity Counties as the North Interior Resource Area of California (Waddell and Bassett 1997). The Lassen-Modoc community has requested assessment of local timber production at a level of detail beyond the peer-reviewed literature, previously synthesized by Long et al. (2014). With access to primary data sources particular to their bioregion, the local national forests are better suited to determine the current capacity and future forecasts of the economic benefits delivered by industrial forest products in the Lassen-Modoc.

Nontimber Forest Products (NTFPs)

NTFPs such as "foods, medicinal plants and fungi, floral greens and horticultural stocks, fiber and dye plants, lichens, and oils, resins, and other chemical extracts from plants, lichens, and fungi…[as well as] poles, posts, Christmas trees, and firewood" (Winter et al. 2014, pp. 649–650) are collected and used for "energy sources, food items, medicinal products, materials for household equipment, construction materials, as well as equipment and materials for agricultural activities" (Illukpitiya and Yanagida 2010, p. 1952). They are relatively more abundant in the Sierra Nevada and Modoc Plateau compared to drier regions but less abundant compared to wetter regions of California. Most NTFPs in Northern California are harvested for personal or subsistence use, with some sold commercially or for supplementary domestic income.

As mentioned in Chapter 5.2 (Flores and Russell, this synthesis, *Demographic Trends in Northeastern*

California), forest communities may anticipate land use/land cover shifts between agriculture and forestlands and between developed and undeveloped areas. NTFP harvesters in rural areas may experience this agriculture/forest trade-off as restricted access to forest products and depletion from over-harvest as population density increases. The literature indicates, however, that where communities are actively involved in management decisions, forests and forest products are well-conserved (Illukpitiya and Yanagida 2010). Watson (2017) importantly suggests that in a post-timber production market, public forests offer an arena in which forest managers and forest resource users may cooperate in new ways. Harvesters may contribute to the sustainable management of NTFPs on national forest lands by sharing their ecological knowledge and management practices and by participating in NTFP research efforts (Long et al. 2014).

Communities have also been able to reap from the forests economic benefits that exist outside of formal markets. Sharing and reciprocal exchange often characterizes subsistence communities that use wild forest resources. This alternative or informal economic distribution system has been shown to reinforce the social benefits of familial, friendship, clan, and tribal relationships that create community identity (Dick 1996). Other nonmaterial motivations—"maintaining cultural practices, sharing knowledge, building community, engaging in spiritual practices, connecting with nature, supporting stewardship, having fun and recreating, and developing alternative food and health systems"—in turn illustrate social positions from which gatherers advocate for economic rights to wild, sustainable products (Poe et al. 2013, p. 416). Wilsey and Nelson (2008) capture the notion that economic activity is essentially cultural, or "embedded in and guided by underlying social institutions," (p. 815) and reinforce the idea that NTFP harvesting is an example not only of economic livelihood, but of a socio-cultural lifestyle for forest communities.

Forest Services: Recreation, Wildlife and Wild Horse Viewing, Carbon Storage, and Energy

Recreation

The economic value of recreational activities such as hiking, camping, and biking in the national forests has been estimated through a willingness-to-pay, travel cost approach to be $90 per person per trip for site access (Binder et al. 2017)[2]. After experiencing declines in timber production and having explored forest restoration services, forest communities are poised to pursue or maintain the economic benefits of outdoor recreation and eco-tourism. Winter et al. (2014) remind us that the economic benefits in this sector vary a great deal, adding some jobs that pay well but many other jobs that are less-beneficial low-wage jobs and/or restricted to seasonal opportunities.

Often recreation-related management decisions can account for attributes preferred by recreationists and the quality of those attributes, including: topography, ecosystem type, and state of fire recovery; size, age, and species of trees; clearings and wildlife viewing areas (fig. 5.3.3) "Altering attributes of the forest site, either directly or indirectly, can influence the attractiveness and value of a site for recreation" (Binder et al. 2017, p. 19). Management has tradeoffs, however: "[W]hat may be an ecosystem service at one level of provision may be a disservice at a different level" (p. 21). Or, one recreationist's trail is another recreationist's trial.

While many intense, landscape-damaging wildfire scenarios negatively impact user value, the number of recreation visits surges in some postfire sites. This is due to initial booms in wildflower cover and wildlife range, and generally to the thinning of the woods that occurs with lower-intensity fires, which leave larger trees intact (Bawa 2017). Therefore, related to the next section on fire management, a "reduction of hazardous fuels and forest restoration activities are thus likely to be viewed by recreational users as increasing quality at a given site" (Bawa 2017 p. 10).

Wildlife and Wild Horse Viewing

The Wild Free-Roaming Horses and Burros Act of 1971 charged the U.S. Department of the Interior, Bureau of Land Management (BLM) and the Forest Service with protecting and managing wild horses (*Equus caballus*) and burros (*E. asinus*) on public lands. As of March 1, 2016, more than 67,000 wild horses and burros are roaming Western public rangelands, which is currently above the appropriate management level of 26,715 set by the BLM. While herds consistently double in size every 4 years, coupled with the dramatic decrease in adoptions, the current program is becoming increasingly challenging

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[2] For a discussion of the annual economic benefits to the State of California from outdoor recreation on federally managed lands in the Sierra Nevada region, see Chapter 9.1 (Winter et al. 2014, pp. 513–527) in Sierra Nevada Science Synthesis (Long et al. 2014).

Figure 5.3.3—While recreational activities on national forests such as snowmobiling, hang gliding, and fishing can add economic value to surrounding communities, management decisions have tradeoffs, sometimes favoring one aspect of recreation while having negative consequences for another type of recreation, or type of land use (photos by Modoc Outdoor Recreation and Tourism, used with permission).

to maintain—both ecologically and socially. Therefore, updating current management practices is complex, and requires agreement from multiple stakeholders on the ecological, social, and economic costs (White 2016). Consequently, stakeholders are increasingly polarized about how wild horses and burros are or should be managed. While the ecological and animal health and welfare implications of unmanaged wild horse and burro populations are somewhat understood, publicly acceptable strategies to maintain healthy populations, healthy and functioning rangelands, and multiple uses that sustain wildlife and local communities remain unresolved (Scasta et al. 2018).

Wildlife viewing, including wild horses, is a way to observe the wildness, power, resilience, and freedom in the American West. Landscape-scale and experimental investigations have shown that free-roaming horses and burros induce numerous alterations to native-ecosystem components and processes through influences on soil, water, plants, and other aspects of biodiversity. The management of free-roaming horses has been complicated by "socio-ecological mismatches" (Beever et al. 2019). Such mismatches arise from an inability to reconcile conflicting processes and functions in a social-ecological system. These conflicts often reflect differences in the spatio-temporal scales at which diverse components operate. Introduced species, or in the case of wild horses, reintroduction after 10 to 15 millennia, can have important effects on the component species and processes of native ecosystems. Effective management of such species can be complicated by technical and social challenges (Beever et al. 2018). Beever et al. (2019) identify three socioecological mismatches of wild horse and burro management for both ecological and social sustainability: (1) social systems and cultures may adapt to a new species' arrival at a different rate than ecosystems; (2) ecological impacts can arise at one spatial scale while social impacts occur at another; and (3) the effects of introduced species can spread widely, whereas management actions are constrained by organizational and/or political boundaries.

The way that the current wild horse and burro program is being managed is considered ecologically and financially unsustainable (Danvir 2018; GAO 2008; Garrott 2018). How the BLM and the Forest Service manage wild horses and burros if often being challenged through litigation and public pressure and ultimately Congress may have to supply a decision on sustainable social-ecological management in the best interest of the American public and natural resources (Hendrickson 2018). Options that can contribute to achieving the intent of the Wild Free-Roaming Horses and Burros Act of 1971 are: (1) maintain the status quo; (2) gather and place excess horses and burros in holding facilities for the remainder of their lives; (3) increase adoptions; (4) increase fertility control; (5) remove livestock from wild horse management areas; and (6) fully implement the Wild Free Roaming Horses and Burro Act (Hendrickson 2018). Use of each tool in the BLM and Forest Service toolbox has pros and cons, and while scholars recommend the full implementation of the Wild Free Roaming Horses and Burro Act, it is currently up to each unique management area on which option or options to choose (Hendrickson 2018).

Carbon Storage

Global initiatives for forest carbon sequestration incentivize forest conservation and guide land use, and in so doing, create a market for forest carbon (Mahanty et al. 2012). Participation in carbon sequestration projects requires community knowledge around contributions by public and private property owners, and negotiations regarding the distribution of potential benefits. The 2012 science synthesis on climate change more thoroughly reviews issues surrounding carbon storage (see Chapter 4 in Vose et al. 2012). Nelson and Matzek (2016) are mindful about the tradeoffs of cap-and-trade mechanisms like the regional California Air Resources Board program:

> While it is widely recognized that these mechanisms will impact carbon and energy markets in the U.S., these programs also have the potential to impact land use patterns, rural livelihoods, wildlife habitat, and ecosystem service provision across the country (Nelson and Matzek 2016, p. 2).

The rate of carbon sequestration in the United States increased by one-third from the 1990s to 2000s, due to increased forest area and restoration efforts (Powers et al. 2013). The Western United States is expected, however, to experience disturbances of warming climate such as lower forest productivity, increased fire, and beetle kill—a combination of changes that unfortunately reduces carbon storage in forest ecosystems and is generally seen as "jeopardizing the current U.S. forest sink" (Vose et al. 2012, p. 45). Faced with complex decisions between competing carbon strategies, forest managers may have to negotiate between sequestration by afforestation, by managing decomposition, or by fuels and fire reduction, as appropriate to their ecosystems (Powers et al. 2013). In the Lassen-Modoc area, carbon density has been classified largely as soil organic carbon, as opposed to live forest biomass or dead wood and forest-floor carbon (Oswalt et al. 2014). Relatedly, a study of sequestration strategies in postfire Plumas and Lassen Counties suggests that in fire-prone ecosystems, "the green canopy forest and no salvage logging treatments store the most carbon 10 years after a wildfire" (Powers et al. 2013, p. 276). Further, Powers and colleagues advise ecosystem services managers who combine carbon storage and fire hazard reduction to "maximize carbon storage and carbon sequestration rates in large-diameter live trees and other recalcitrant pools, including soils and dead trees, if possible" and to "minimize carbon storage in saplings, understory, and surface fuels" (p. 276).

Other issues of land use and local capital influence potential carbon markets. Relevant to the Lassen-Modoc is a finding by a recent economic model of the carbon market in the Central Valley of California, namely, that carbon farming has performed poorly against some agricultural commodities. In the case of orchard agriculture, for example, the price of "a carbon offset would have to increase nearly a hundredfold to make reforestation compete economically" (Nelson and Matzek 2016, p. 1). A study of California's Improved Forest Management projects showed that making carbon offsets viable required a level of cost and complexity, capital, knowledge, and technology that excluded marginal landowners. Carbon sequestration projects, therefore, may possibly involve more heavily State-managed initiatives, as opposed to balanced collaboration with small community landowners (Kelly and Schmitz 2016).

Energy

We have noted that economic benefits of forest ecosystem services exist on a spectrum between large-scale timber harvesting and domestic nontimber forest product, and between the negotiated goals of fire management and carbon stocking. Forest-dependent communities are being encouraged to respond to ecosystem service opportunities conditioned by climate change policy. "Under the 2011 California Renewable Energy Resources Act (SB X 1-2), electrical utilities are required to obtain 33 percent of the electricity they sell to retail customers in California from renewable sources by 2020" (Winter et al. 2014, p. 645). This section considers issues of bioenergy, solar and wind, and geothermal energy that may be informative for Lassen-Modoc stakeholders.

Bioenergy Fuel. Forest residues differ from timber in that it is the "trees and woody plants—including limbs, tops, needles, leaves, and other woody parts—that grow in forests, woodlands, or rangelands" (Charnley and Long 2014, p. 641). The removal of forest residues in forested land can also reduce wildfire hazards and associated costs of fire disturbance (fig. 5.3.4). The cost of removal and transport of forest residues to facilities can be significant, if not prohibitive.

In areas where cultivating hardwoods to sequester carbon is less viable, developing bioenergy from other forest and wood waste is an alternative (Becker et al. 2009). The margin for growth is large because forest residues, as a renewable source capable of providing a consistent baseload of power, contribute only about 2 percent of California's electricity (Winter et al. 2014, p. 645). As of the publication of the Sierra Nevada Science Synthesis in 2014, California had more biomass power plants than any other State, and the plants existed in various states of operability, generating various types of fuel and/or energy. At the time, Lassen County featured one idled facility, and Modoc County had no biomass power plants.

The development of forest residue production in these counties was constrained in several ways. First, the supply of material was limited ecologically by the prominence of shrub cover over forest cover in many areas. Aboveground live forest residues are dramatically less prevalent in the Lassen and the Modoc, relative to adjacent areas of the Sierra Nevada and Coastal California (Vose et al. 2012). Second, a market disadvantage was already in place, with the preexistence of active plants in the region, concentrated near Redding. Finally, other economic issues related to "lack of industry infrastructure, harvest and transport costs … and market trends" were noted (Winter et al. 2014, pp. 642–643). (See also Charnley and Long 2014.)

The prevailing market trend regarding renewables was that wind and solar were the sources of electricity favored by "the largest electrical utilities in California" (Winter et al. 2014, p. 645). Research is needed to assess whether the September 2012 passage of California's Senate Bill 1122 was successful in removing some market barriers and stimulating the State's bioenergy markets through small, networked biomass projects. More than urbanization and

Figure 5.3.4—Converting noncommercial forest residue into products with economic value would further support traditional harvesting of trees for forest products, as well as bolster efforts to thin younger, overstocked stands to reduce fire risk and intensity (photo by Ken Sandusky, Forest Service).

population growth, policies that govern the direction of the bioenergy sector may ultimately be more impactful for rural land use decisions:

> Stronger shifts in comparative returns to forestry and agriculture would probably result from policy changes, especially those designed to encourage bioenergy production. The degree to which a bioenergy sector favors agricultural feedstocks, such as corn, or cellulosic feedstocks from forests will determine the comparative position of forest and crop returns to land use, and therefore land use allocations. The allocation among feedstock sources depends on energy policies at both federal and state levels, which could differentially affect rural land uses (Vose et al. 2012, p. 106).

Solar and Wind Energy

The 1992 Land and Resource Management Plan for the Lassen indicated that the forest had medium suitability for solar energy, but expanding solar use was cost prohibitive at the time. Similarly, while several exposed ridge areas of the Lassen were classified as providing excellent wind resources usable for wind energy, development faced prohibitive costs of access, infrastructure, and electrical transmission (USDA Forest Service 1992). At that time (1992), demand for solar or wind energy was not significant.

Geothermal Energy

The County of Modoc website indicated that Modoc County has great geothermal potential "right under [its] feet" (Moeller 2017). Modoc County has 119 authorized geothermal energy zones on public land managed by the BLM (Thermal Zones Modoc 2016). Additionally, Surprise Valley in Modoc County is identified as a key geothermal resource site in California (Geothermal Energy Association 2016). Similarly, Lassen County has 32 authorized geothermal energy zones on BLM-managed public land (Thermal Zones Lassen 2016).

According to the Geothermal Technologies Office 2016 Annual Report (2017), which is a product of the U.S. Department of Energy's Office of Energy Efficiency and Renewable Energy, geothermal energy can provide several social and economic benefits. For example, geothermal energy "supplies clean, renewable power around the clock, emits little or no greenhouse gases, and takes a very small environmental footprint to develop" (p. 3). Additionally, the development of "available geothermal can create temporary and permanent jobs and revenue streams in California" (Geothermal Technologies Office 2017, p. 2). Furthermore, the high temperatures produced by geothermal energy may dissolve "rare-earth minerals and other high-value, critical or strategic materials" into "fluids associated with geothermal energy extraction" (p. 11). Subsequently, these materials, which include rare minerals and lithium, are crucial to the production and use of "many clean energy technologies, including solar panels, wind turbines, electric vehicles, and energy-efficient lighting" (p. 11). For additional benefits of geothermal energy, specific to California, see Geothermal Energy Association's (2016) presentation on "Geothermal energy potential: State of California."

While the benefits from geothermal energy are significant, its production can be "challenging and costly, with resource confirmation relying on the drilling of multimillion-dollar wells with varying success rates" (Phillips et al. 2013, p. 1). Therefore, for Lassen and Modoc Counties to pursue the social and economic benefits provided by geothermal energy, forest managers can develop an "understanding of temperature, permeability, and fluid signatures that indicate geothermal favorability" (Phillips et al. 2013, p. 1). Furthermore, research to enhance innovative exploration technologies can continue to be conducted to improve the identification and development of geothermal systems. See Phillips et al. (2013) for more information.

Forest Management Practices

This section focuses on general forest management guidelines and some examples from California forest communities in order to respond to the question:

- How can *forest management* practices improve decisionmaking processes for ecosystem service benefits?

As noted previously in this section, "place matters, the planning or decisionmaking process matters, and original, specific, and local solutions may be best" (Vose et al. 2012, p. 196) because social values (described above) play a critical role in why communities derive meaning from their local ecosystem services and how members invest in them. Assessing social values prepares forest managers and publics to communicate their way through potential conflicts and congruencies, and to participate in democratic, multifaceted decision making. Methods of assessing community values and attitudes include surveys and opinion polls, and more interpretive practices such as public-participation GIS mapping, workshops, focus groups, storytelling, interviews, nature journaling, and a number of psychometric measures of well-being (Bieling et al. 2014; Ives and Kendal 2014; Kenter et al. 2015).

Economic Valuations

A recent review of literature (Binder et al. 2017) indicates that the economic valuation of forest ecosystem services is under-utilized in forest management. The review focuses on nonurban forests, ecological production functions, and economic benefits functions; but does not include social, cultural, or spiritual ecosystem services that we have already identified as "difficult to quantify and whose value is often thought to be antithetical to consideration in monetary terms" (p. 1). In short, the community's social preferences and economic values are either translated into practical negotiations of time and monetary investments or they are measured by observed behaviors.

> Economic valuation of an ecosystem's goods and services represents an attempt to estimate changes in people's economic well-being—as measured by their own preferences—due to incremental (marginal) changes in the ecosystem's components. When ecosystem goods are traded in markets (e.g., timber), the market price (e.g., U.S. dollars/cubic meter) is a measure of the benefit people get from a unit of the good. Since most ecosystem services are not traded in markets, and therefore do not have observable prices, economists estimate the value of changes in ecosystem services by leveraging the information conveyed by individuals' observable decisions (Binder et al. 2017, p. 3).

While economic valuation methods do have limitations, they are able to provide stakeholders with information to make management decisions. Binder et al. (2017) summarize guidance for economic valuation assessments in Forest Service decisions with four suggestions that may shape the forest management plan, and note that this guidance could be tailored to the needs of the specific forest and its community. These involve cataloging assets, projecting possible change, communicating these valuations with stakeholders, and monitoring performance, described below.

How could the Forest Service proceed given the current state of the literature on the assessment and economic valuation of ecosystem services?

1. Estimate the economic benefits of a given forest and associated management policy, using available methods for services related to timber, carbon, water, amenities, recreation, and wildlife. This practice is important to identifying and describing the range of benefits provided by the forest. It also provides a baseline for evaluating changes in management.
2. Estimate the change in economic benefits associated with a change in management, regulations, or incentives, or a natural disturbance. This practice is important to evaluating and prioritizing different policies, evaluating potential tradeoffs in management decisions, and assessing the damages caused by natural disturbances.
3. Enhance communication with stakeholders about the economic benefits and costs of potential changes in forest management. This practice is important because communities' preferences for different ecosystem services may be affected by estimates of economic performance.
4. Monitor the performance of agency programs. This practice is important to tracking whether the actual economic benefits and costs of agency programs are consistent with projections.

(Adapted from Binder et al. 2017)

Vose et al. (2012) describe the socioeconomic vulnerability assessment (SEVA) process. SEVAs first require a review of Census Bureau data and other secondary resources relevant to the local area. SEVAs then could:

> (1) "briefly discuss the social history of the forest and its human geography, including both communities of place and communities of interest,
>
> (2) link current and expected biophysical changes to community-relevant outcomes,
>
> (3) determine stakeholders' perceptions of values at risk (e.g., resources, livelihoods, cultures or places threatened by climate change), and
>
> (4) prioritize threats to vulnerable communities and identify those that the landowner or land manager, singly or with their partners, can best address." (p. 117).

Like other assessment approaches, the SEVA is considered an iterative, monitored, and flexible process subject to revision and adaptation in collaborative forest management.

Fire Management

Changes in the climate regimes are affecting the ecological health and economic costs of forest ecosystem services throughout the United States differentially by region. Increased seasonal temperatures and higher drought indices are projected for the Pacific Southwest Region generally and California specifically, including impacts upon the Lassen and Modoc reviewed elsewhere in this science synthesis. Current and anticipated increases in wildland fire frequency, intensity, and impacts on forest systems are expected to have complex effects on carbon sequestration, vegetation extent, biodiversity, and other ecosystem services. Prescribed and natural fire management, fuel treatments, and forest restoration have been shown to be effective management tools for containing or reducing wildland fire, especially at the local scale; and their use has been encouraged for the health of the ecosystem (Hurteau et al. 2014; Rocca et al. 2014; Vose et al. 2012).

In addition to its ecosystem benefits, a more robust fire management program may also provide economic benefits. Opportunities for more regularized, less seasonally driven employment for the community-based forest and fire labor force could support the local economy more reliably. Employment through fire management and fire hazard reduction represents a reversal of the fire suppression practices that were intended to create jobs in the 1930s. The National Fire Plan in 2000 discusses fire management for the dual tasks of "reducing fire hazard on public lands and providing economic benefit to rural communities and workers" (Moseley and Toth 2004, p. 702). An early evaluation in the Northwest Region found that the National Fire Plan created economic benefits for some rural communities, but not necessarily for the more isolated rural communities, which the Lassen-Modoc area may resemble. Through the local benefit criterion, the National Fire Plan granted more contracts to firms located closer to national forests. Finally, the plan also delivered more contracts to historically underutilized businesses, which in the Northwest as in the Lassen-Modoc are characterized by poor rural counties and tribal lands. Minority-owned 8(a) businesses, typically located in urban areas and not proximate to national forests in the Pacific Northwest, received comparatively fewer contracts (Moseley and Toth 2004).

Fire management, especially in the form of wildland fire use, also "provides economic benefits in terms of avoidance of costs—for environmental damage remediation, fuel treatment projects, comparatively more expensive suppression tactics" (Dale 2006, p. 279). Fire management also adds savings in terms of material security to private homeowners and safer evacuation passage for forest residents (Moritz et al. 2014). Additionally, long-underestimated health effects of wildfire exposure have been re-assessed as monetized damage (Richardson et al. 2012). Doing so enables communities to consider smoke-related health impacts and defensive health strategies as economic costs that could otherwise be mitigated with increased fire management and fire reduction. Estimated spending for fuels management per Forest Service region are by far the highest in the Pacific Southwest Region (see table 3 in Lee et al. 2015, p. 265).

Models of Ecosystem Services Management in California

The Inyo National Forest in Eastern California contains dry forest ecosystems similar to some areas of the Lassen-Modoc bioregion. One outcome of science-management collaboration was the Climate Project Screening Tool, the purpose of which is to assess whether climate change would affect natural resources and therefore impact current-year management of ecosystem services (Vose et al. 2012). Another science-management collaboration tool is the Strategic Framework for Science in Support of Management that was produced for the Southern Sierra Nevada, CA. It addresses parallel concerns of the Lassen-Modoc, including wildland fire and potential wildland urban interface encroachment.

Public and private stakeholders in the Mokelumne River Basin, known as a water source for the San Francisco Bay area and for its whitewater rafting and kayaking, gathered in 2012 to make an economic case to broker increased fuel treatment in the basin (Elliot et al. 2016; also see Buckley et al. 2014). As previously discussed, economic values are difficult to assign to resources such as wildlife, recreation, and cultural sites. While these were left out of the analysis, the avoided costs of sediment erosion were included in a number of fire and hydrology risk-assessment models (Elliot et al. 2016). The resulting analysis estimated that "the economic benefits of the fuels treatments were 2 to 3 times more than the costs of treatments" (p. 884) and helped land and water managers, including the Forest Service and BLM, plan fuel treatment strategies with stakeholders to protect water utility and other infrastructure resources. This recent fuel treatments project illustrates the mutual benefits of collaborative management:

> Offsite stakeholders are not commonly engaged in forest management planning processes. This methodology helped to bring together watershed managers and water users who had not realized they too were stakeholders in forest fuel management (Elliot et al. 2016, p. 885).

This chapter reviews some supply-side issues of forest management in which "the formal methods of professional economic analysis reflect a vision of comprehensive administrative rationality" and in which public land managers "operate in a world of large uncertainties with respect to scientific facts, future social and economic trends, likely political pressures, and many other factors" (Nelson 2006, p. 550). In addition, the concept of ecosystem services as a socioecological paradigm that attempts to balance these factors was introduced. The management of ecosystem services is a negotiation of meaningful social values and practical economic interests among culturally rooted forest stakeholders in a changing world. In times of increased climate disruption and hazard events, the measure of successful forest ecosystem services management is marked by "efforts that benefit and promote goals of ecosystem sustainability," and as has been repeated, is shaped by local contours. Successful implementation occurs

> When projects are developed and deployed for specific places with concrete treatments and prescriptions, explicit objectives, and for definitive time periods. Successful implementation also implies that monitoring and adaptive management schedules are integrated in out-year efforts, and are secured with funds and capacity needed for completion (Vose et al. 2012, p. 125).

Because economic use-values do not necessarily coincide with social well-being, Chapter 5.4 (Flores and Stone, this synthesis, *Community Engagement in the Decisionmaking Process for Public Land Management in Northeastern California*) addresses how local community members advocate for their demand-side concerns, and how forest managers negotiate competing demands with educational and outreach practices.

References

Balmford, A.; Bond, W. 2005. Trends in the state of nature and their implications for human well-being. Ecology Letters. 8(11): 1218–1234.

Barrio, M.; Loureiro, M.L. 2010. A meta-analysis of contingent valuation forest studies. Ecological Economics. 69(5): 1023–1030.

Bawa, R.S. 2017. Effects of wildfire on the value of recreation in western North America. Journal of Sustainable Forestry. 36(1): 1–17.

Becker, D.R.; Skog, K.; Hellman, A.; [et al.]. 2009. An outlook for sustainable forest bioenergy production in the lake states. Energy Policy. 37: 5687–5693.

Beever, E.A.; Huntsinger, L.; Petersen, S.L. 2018. Conservation challenges emerging from free-roaming horse management: a vexing social-ecological mismatch. Biological Conservation. 226: 321–328.

Beever, E.A.; Simberloff, D.; Crowley, S.L.; [et al.]. 2019. Social-ecological mismatches create conservation challenges in introduced species management. Frontiers in Ecology and the Environment. 17(2): 117–125.

Bieling, C.; Plieninger, T.; Pirker, H.; [et al.]. 2014. Linkages between landscapes and human well-being: an empirical exploration with short interviews. Ecological Economics. 105: 19–30.

Binder, S.; Haight, R.G.; Polasky, S.; [et al.]. 2017. Assessment and valuation of forest ecosystem services: state of the science review. Gen. Tech. Rep. NRS-170. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station. 47 p.

Bixler, R.P. 2014. From community forest management to polycentric governance: assessing evidence from the bottom up. Society & Natural Resources. 27(2): 155–169.

Buckley, M.; Beck, N.; Bowden, P.; [et al.]. 2014. Mokelumne watershed avoided cost analysis: why Sierra fuel treatments make economic sense. A report prepared for the Sierra Nevada Conservancy, The Nature Conservancy, and U.S. Department of Agriculture, Forest Service. Auburn, CA: Sierra Nevada Conservancy. https://www.sierranevada.ca.gov/our-work/mokelumne-watershed-analysis (6 Sep. 2017).

Charnley, S.; Jaworski, D.; Huber-Stearns, H.; [et al.]. 2018a. Environmental justice, low income and minority populations, and forest management in the Northwest Forest Plan area. In: Spies, T.A.; Stine, P.A.; Gravenmier, R.; [et al.], tech coords. Synthesis of science to inform land management within the Northwest Forest Plan area. Peer review draft of Gen. Tech. Rep. PNW-GTR-966. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 809–849. Chap. 10.

Charnley, S.; Kline, J.D.; White, E.M.; [et al.]. 2018b. Socioeconomic well-being and forest management in Northwest Forest Plan area communities. In: Spies, T.A.; Stine, P.A.; Gravenmier, R.; [et al.], tech coords. 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: 625–715. Chap. 8.

Charnley, S.; Long, J.W. 2014. Biomass utilization. In: 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: 641–649.

Dale, L. 2006. Wildfire policy and fire use on public lands in the United States. Society & Natural Resources. 19(3): 275–284.

Daniels, J.M.; Nielsen-Pincus, M.; Paruszkiewicz, M.; [et al.]. 2018. The economic contribution of stewardship contracting: two case studies from the Mount Hood National Forest. Journal of Forestry. 116(3): 245–256.

Danvir, R.E. 2018. Multiple-use management of western U.S. rangelands: wild horses, wildlife, and livestock. Human-Wildlife Interactions. 12: 5–17.

Davis, E.J.; Abrams, J.; White, E.M.; [et al.]. 2018. Current challenges and realities for forest-based businesses adjacent to public lands in the United States. The Journal of Rural and Community Development. 13(1): 125–142.

Dick, R.E. 1996. Subsistence economics: freedom from the marketplace. Society & Natural Resources. 9(1): 19–29.

Elliot, W.J.; Miller, M.E.; Enstice, N. 2016. Targeting forest management through fire and erosion modelling. International Journal of Wildland Fire. 25(8): 876–887.

Ferranto, S.; Huntsinger, L.; Getz, C.; [et al.]. 2011. Forest and rangeland owners value land for natural amenities and as financial investment. California Agriculture. 65(4): 184–191.

Ferranto, S.; Huntsinger, L.; Stewart, W.; [et al.]. 2012. Consider the source: the impact of media and authority in outreach to private forest and rangeland owners. Journal of Environmental Management. 97: 131–140.

Frickel, S.; Freudenburg, W.R. 1996. Mining the past: historical context and the changing implications of natural resource extraction. Social Problems. 43(4): 444–466.

Garrott, R.A. 2018. Wild horse demography: implications for sustainable management within economic constraints. Human-Wildlife Interactions. 12: 46–57.

Gasparatos, A.; Stromberg, P.; Takeuchi, K. 2011. Biofuels, ecosystem services and human well-being: Putting biofuels in the ecosystem services narrative. Agriculture Ecosystems & Environment. 142(3-4): 111–128.

Geothermal Energy Association 2016. Geothermal energy potential: State of California. Washington, DC: Geothermal Energy Association. https://geothermal.org/PDFs/Final%20California.pdf. (26 Oct. 2017).

Geothermal Technologies Office. 2017. 2016 Annual report Geothermal Technologies Office. DOE/EERE-1558. Washington, DC: U.S. Department of Energy, Office of Energy Efficiency and Renewable Energy. https://energy.gov/sites/prod/files/2017/03/f34/GTO%202016%20Annual%20Report_1.pdf. (26 Oct. 2017).

Goldman, R.L. 2010. Ecosystem services: how people benefit from nature. Environment. 52(5): 15–23.

Government Accountability Office [GAO]. 2008. Effective long-term options needed to manage unadoptable wild horses. Bureau of Land Management, Report to the Chairman, Committee on Natural Resources, House of Representatives. GAO-09-77. Washington, DC. 88 p.

Hendrickson, D. 2018. Managing healthy wild horses and burros on healthy rangelands: tools and the toolbox. Human-Wildlife Interactions 12(1): 143–147.

Hicke, J.A.; Jenkins, J.C.; Ojima, D.S.; [et al.]. 2007. Spatial patterns of forest characteristics in the western United States derived from inventories. Ecological Applications. 17(8): 2387–2402.

Hjerpe, E.; Hussain, A.; Phillips, S. 2015. Valuing type and scope of ecosystem conservation: a meta-analysis. Journal of Forest Economics. 21(1): 32–50.

Hunt, L.M.; Lemelin, R.H.; Saunders, K.C. 2009. Managing forest road access on public lands: a conceptual model of conflict. Society & Natural Resources. 22(2): 128–142.

Hurteau, M.D.; Bradford, J.B.; Fulé, P.Z.; [et al.]. 2014. Climate change, fire management, and ecological services in the Southwestern U.S. Forest Ecology and Management. 327: 280–289.

Illukpitiya, P.; Yanagida, J.F. 2010. Farming vs forests: trade-off between agriculture and the extraction of non-timber forest products. Ecological Economics. 69(10): 1952–1963.

Ives, C.D.; Kendall, D. 2014. The role of social values in the management of ecological systems. Journal of Environmental Management. 144: 67–72.

Kelly, E.C.; Schmitz, M.B. 2016. Forest offsets and the California compliance market: bringing an abstract ecosystem good to market. Geoforum. 75: 99–109.

Kenter, J. O.; O'Brien, L.; Hockley, N.; [et al.]. 2015. What are shared and social values of ecosystems? Ecological Economics. 111: 86–99.

Kline, J. 2006. Public demand for preserving local open space. Society & Natural Resources. 19(7): 645–659.

Krekel, C.; Kolbe, J.; Wuestemann, H. 2016. The greener, the happier? The effect of urban land use on residential well-being. Ecological Economics. 121: 117–127.

Lee, C.; Schlemme, C.; Murray, J.; [et al.]. 2015. The cost of climate change: ecosystem services and wildland fires. Ecological Economics. 116: 261–269.

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.

Mahanty, S.; Milne, S.; Dressler, W.; [et al.]. 2012. The social life of forest carbon: property and politics in the production of a new commodity. Human Ecology. 40(5): 661–664.

Moeller, J. 2017. Geothermal. Modoc, CA: County of Modoc website. https://www.co.modoc.ca.us/departments/geothermal. [Page last modified: 17 Feb. 2017.] (25 August 2017).

Morgan, T.A.; Keegan, C.E.; Dillon, T.; [et al.]. 2004. California's forest products industry: a descriptive analysis. Gen. Tech. Rep. PNW-GTR-615. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 55 p.

Moritz, M.A.; Batllori, E.; Bradstock, R.A.; [et al.]. 2014. Learning to coexist with wildfire. Nature. 515(7525): 58–66.

Moseley, C.; Toth, N.A. 2004. Fire hazard reduction and economic opportunity: How are the benefits of the National Fire Plan distributed? Society & Natural Resources. 17(8): 701–716.

Nelson, R.H. 2006. Valuing nature: Economic analysis and public land management, 1975–2000. American Journal of Economics and Sociology. 65(3): 525–557.

Nelson, E.; Matzek, V. 2016. Carbon credits compete poorly with agricultural commodities in an optimized model of land use in Northern California. Climate Change Economics. 7(4): 1650009, 24 p.

Ninan, K.N.; Inoue, M. 2013. Valuing forest ecosystem services: what we know and what we don't. Ecological Economics. 93: 137–149.

Oswalt, S.N.; Smith, W.B.; Miles, P.D.; [et al.]. 2014. Forest resources of the United States, 2012: a technical document supporting the Forest Service 2015 update of the RPA Assessment. Gen. Tech. Rep. WO-91. Washington, DC: U.S. Department of Agriculture, Forest Service, Washington Office. 218 p.

Patterson, T. 2014. Ecosystem services. In: 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: 543–568.

Phillips, B.R.; Ziagos, J.; Thorsteinsson, H.; [et al.]. 2013. A roadmap for strategic development of geothermal exploration technologies. In: Proceedings, 38th Workshop on Geothermal Reservoir Engineering. Proceedings SGP-TR-198. Stanford, CA: Stanford University. https://www1.eere.energy.gov/geothermal/pdfs/exploration_technical_roadmap2013.pdf. (26 Oct. 2017).

Poe, M. R.; McLain, R.J.; Emery, M.; [et al.]. 2013. Urban forest justice and the rights to wild foods, medicines, and materials in the city. Human Ecology. 41(3): 409–422.

Poteete, A.R.; Welch, D. 2004. Institutional development in the face of complexity: developing rules for managing forest resources. Human Ecology. 32(3): 279–311.

Powers, E.M.; Marshall, J.D.; Zhang, J.W.; [et al.]. 2013. Postfire management regimes affect carbon sequestration and storage in a Sierra Nevada mixed conifer forest. Forest Ecology and Management. 291: 268–277.

Pramova, E.; Locatelli, B.; Djoudi, H.; [et al.]. 2012. Forests and trees for social adaptation to climate variability and change. Wiley Interdisciplinary Reviews—Climate Change. 3(6): 581–596.

Richardson, L.A.; Champ, P.A.; Loomis, J.B. 2012. The hidden cost of wildfires: economic valuation of health effects of wildfire smoke exposure in Southern California. Journal of Forest Economics. 18(1): 14–35.

Rocca, M.E.; Miniat, C.F.; Mitchell, R.J. 2014. Introduction to the regional assessments: Climate change, wildfire, and forest ecosystem services in the USA. Forest Ecology and Management. 327: 265–268.

Scasta, J.D.; Hennig, J.D.; Beck, J.L. 2018. Framing contemporary U.S. wild horse and burro management processes in a dynamic ecological, sociological, and political environment. Human-Wildlife Interactions 12(1): 31–45.

Spies, T.A.; Duncan, S.L., eds. 2009. Old growth in a new world: a Pacific Northwest icon reexamined. Washington, DC: Island Press. 360 p.

Stavros, E.; Natasha, D.M.; Larkin, N. 2014. The climate-wildfire-air quality system: interactions and feedbacks across spatial and temporal scales. Wiley Interdisciplinary Reviews—Climate Change. 5(6): 719–733.

Steelman, T.A.; Burke, C.A. 2007. Is wildfire policy in the United States sustainable? Journal of Forestry 105(2): 67–72.

Stem, C.J.; Lassoie, J.P.; Lee, D.R.; [et al.]. 2003. Community participation in ecotourism benefits: the link to conservation practices and perspectives. Society & Natural Resources. 16(5): 387–413.

Thermal Zones Lassen 2016. Geothermal Power Leasing in Lassen County, California. Chattanooga TN: Thermal Zones. https://thermalzones.com/usa/california/lassen-ca035. (26 Oct. 2017).

Thermal Zones Modoc 2016. Geothermal Power Leasing in Modoc County, California. Chattanooga TN: Thermal Zones. https://thermalzones.com/usa/california/modoc-ca049. (26 Oct. 2017).

Turner, R.K.; Paavola, J.; Cooper, P.; [et al.]. 2003. Valuing nature: lessons learned and future research directions. Ecological Economics. 46(3): 493–510.

Turner, R.K.; Daily, G.C. 2008. The ecosystem services framework and natural capital conservation. Environmental and Resource Economics. 39(1): 25–35.

U.S. Department of Agriculture, Forest Service [USDA Forest Service]. 1992. Lassen National Forest land and resource management plan, as amended. Gen. Tech. Rep. FS-CA-920336. Susanville, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 142 p.

Vose, J.M.; Peterson, D.L.; Patel-Weynand, T., eds. 2012. Effects of climatic variability and change on forest ecosystems: a comprehensive science synthesis for the U.S. Forest Sector. Gen. Tech. Rep. PNW-GTR-870. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 265 p.

Waddell, K.L.; Bassett, P.M. 1997. Timber resource statistics for the north interior resource area of California. Resour. Bull. PNW-RB-222. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 49 p.

Watson, K. 2017. Alternative economies of the forest: honey production and public land management in Northwest Florida. Society & Natural Resources. 30(3): 331–346.

Wellstead, A.; Rayner, J.; Howlett, M. 2014. Beyond the black box: forest sector vulnerability assessments and adaptation to climate change in North America. Environmental Science & Policy. 35: 109–116.

White, C. 2016. Issues and answers for managing wild horses and burros. Baltimore, MD: John Hopkins University. M.S. thesis.

Wilsey, D.S.; Nelson, K.C. 2008. Conceptualizing multiple nontimber forest product harvest and harvesting motivations among balsam bough pickers in Northern Minnesota. Society and Natural Resources. 21(9): 812–827.

Winter, P.L.; Long, J.W.; Lake, F.K.; [et al.]. 2014. Broader context for social, economic, and cultural components. In: 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: 501–542.

Chapter 5.4. Community Engagement in the Decisionmaking Process for Public Land Management in Northeastern California

David Flores and Leah Stone[1]

Revisions to forest plans, as directed by the U.S. Department of Agriculture, Forest Service, 2012 Planning Rule, has appreciable focus directed toward management of National Forest System lands so that they are ecologically sustainable and contribute to social and economic sustainability (see Chapter 1.1, Dumroese, this synthesis, *The Northeastern California Plateaus Bioregion Science Synthesis: Background, Rationale, and Scope*; and Chapter 5.1, Flores, this synthesis, *An Introduction to Social, Economic, and Ecological Factors in Natural Resource Management of Northeastern California Public Lands*). This social component requires involving the community throughout the forest plan revision process to determine opportunities for continued community engagement in the management of their local national forests, as is the case for the Lassen and Modoc National Forests. Scientific peer-reviewed literature on community engagement specific to the area around the Lassen and Modoc National Forests is limited. Thus, this chapter takes a more general approach to the literature by first exploring community engagement and surveying how local decisions are made across different aspects of ecosystem services, including water, timber, biomass, recreation, and other uses. Second, we explore the social, cultural, and economic nonmarket values local residents and visitors attribute to forests. Third, we explore how land management agencies address the inter-relatedness of landscapes, people, and management actions, including how habitat improvement and forest restoration projects have been conducted through local partnerships. Finally, we explore examples of community collaborations and best practices, especially those surrounding fire management. Where possible, we include scientific literature that directly relates to the Lassen and Modoc National Forests. This chapter is driven by the following question asked by stakeholders in the area surrounding the Lassen and Modoc National Forests (hereafter the Lassen, the Modoc, or the Lassen-Modoc):

- How can the Forest Service work to seek better local engagement, coordination, and involvement of local communities?

Community Engagement in Ecosystem Service Decisions

The Shift to Community Collaboration in Forest Management

"Forest management in the United States derives from the interaction of the two dominant institutional structures of private and public ownership," (Vose et al. 2012, p. 104) but it is influenced by policy and community involvement. Local governments are guaranteed advisory roles in coordinating management plans via the National Forest Management Act of 1976 and the National Environmental Policy Act of 1970. This includes identifying "areas where additional research is needed" during the planning process (Hart 1995, p. 153). Local governments have a unique opportunity to influence Federal land use decisions through public-participation provisions in national forest planning, as the public review period of this science synthesis illustrates. Lassen-Modoc community members and local Forest Service staff asked:

- How do forest managers include local communities in decisionmaking processes for an all-lands management approach to benefits such as water, timber, biomass, recreation, and multiple use?

Decisions surrounding ecosystem services are shaped by land management agencies and organizations (e.g., Federal, State, and local level agencies and nongovernment organizations) and communities using national forests (e.g., local communities, landowners, tourists). Due to the dynamic process of land management and land use, approaches surrounding ecosystem service decision making can examine and include the perspectives of surrounding forest participants.

In the United States, decision making regarding natural resources and ecosystem services is a multilevel governing process including Federal authorities (e.g., Congress, Forest Service), State authorities (e.g., California), and local agencies (e.g., the Lassen-Modoc region).

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In natural resource management policy and practice, a shift in decisionmaking authority from centralization to decentralization has evolved (Ambus and Hoberg 2011; Berkes 2010; Bixler 2014; Larson and Soto 2008). Where centralized processes grant the upper management of a governing authority full power, also described as top-down decision making, decentralized processes redistribute decisionmaking power from upper management to lower and local management within the same governing body, also described as bottom-up decision making.

The reorganization of decision making in the 1990s was a shift toward community-based natural resource management (Bixler 2014). Building on this effort, multilevel and polycentric governance models—systems with "multiple centers of authority" (Bixler 2014, p. 158)—were developed. Additionally, processes surrounding devolution—power distributed throughout local communities including local residents, agencies, groups, organization, and landowners—emerged (Berkes 2010).

Local Involvement Across Ecosystem Services

According to Poteete and Welch (2004), the procedure and practice of decision making regarding local ecosystem services requires a balance between the complexity of the land resources and the characteristics of the participants that benefit from and use those resources (fig. 5.4.1). The complexity of land resources includes the resources themselves (e.g., water, timber, wildlife, and so forth) and the various benefits received from those resources (e.g., water supply, wood products, food, recreation, amenities, income, etc.). Participants of forest land include: landowners; Federal, State, and local government agencies; tribal communities; forest community residents; organizations; management consultants; recreation users; natural resources; wildlife; and more.

Figure 5.4.1—Successfully engaging the local community in the decisionmaking process regarding local ecosystem services is challenging because of the complexity of land resource issues and the diversity of social values brought by a variety of stakeholders (photo by Ken Sandusky, Forest Service).

While data specific to the Lassen and Modoc Counties is limited, information from surrounding areas and similar regions can provide comparative context. Located in Northern California in the Sierra Nevada mountain range, Lassen and Modoc Counties and surrounding mountain areas have historically provided the predominant concentration of California's forest products industry (Charnley and Long 2014). During the last several decades, California's forest products industry has significantly declined in both supply and demand due to a variety of factors (e.g., regulation, policy, economics, wildland fire, and technology) (Morgan et al. 2004, 2012). Decreases in forest industry production have led to a substantial decline in forest industry employment, consequently, affecting the residents and employees within the Lassen-Modoc region. As reviewed in Chapter 5.2 (Flores and Russell, this synthesis, *Demographic Trends in Northeastern California*), the Lassen-Modoc region is now classified as a nonmetropolitan area with a low-income population and a low-minority population, including multiple tribal communities (Charnley et al. 2018).

The exploration of successful collaborations between Federal agencies and tribes may offer insight into various approaches to local decision making. For example, Taylor and Cheng (2012) define community-based forestry as a "strategy that promotes democratic practices, strengthens local livelihoods, and sustains forest ecosystems for the benefit of all community members" (p. 110) that can be enhanced when agencies approach community engagement without previously defined criteria or *a priori* measurements of success (e.g., how the land was to be managed). Community-based forestry can be successful when forest community residents are afforded the opportunity to make decisions concerning ecosystem services, because local residents are "[embedded] in their communities and make their own best possible judgments about how to remain true to their underlying objectives" (p. 120). Considering these community-based forestry principles, collaborative decision making between residents of the Lassen-Modoc region and the Forest Service could involve an investigation into the local demands and desires for various ecosystem services such as timber, water, biomass, recreation, and other uses.

Timber and Restoration: Timing and Communication

Charnley and Long (2014) acknowledge there are "fewer jobs associated with timber production alone, and there is a greater proportion of jobs in forest restoration" (p. 635). One aspect of forest restoration involves the rehabilitation of land after a wildfire. For example, following a wildfire, the amount of quality timber available for harvest is greatly reduced, while damaged and fire scorched timber is abundant. To help restore and rehabilitate the fire-affected land, various postfire activities, such as salvage logging, are implemented (fig. 5.4.2). These restoration and rehabilitation activities provide both benefits and hindrances to the fire-affected area (see Charnley and Long 2014; McCool et al. 2006; Ryan and Hamin 2008, 2009). For example, salvage logging benefits the local fire-affected community by boosting the local economy through an increase in jobs, timber materials, and funds for restoration. In contrast, salvage logging can damage the ecology of the forest but could be approached in a manner that is appropriate to, and less harmful of, the ecology.

Because of the declines in forest products and subsequent increases in postfire restoration, decisions surrounding wildland fire restoration and rehabilitation could be made prior to wildfire occurrence, with both the Forest Service and local community members (Charnley and Long 2014). Additionally, these decisions could benefit the local community (e.g., funding from salvage logging could be invested in restoration and prevention programs for the community). Lastly, wildland fire restoration decisions often utilize ongoing communication from the Forest Service to the community, as well as from the community to the Forest Service, about land restoration processes and outcomes.

Water: Local Agencies, Shared Jurisdictions, Realistic Modeling, and Education

This subsection focuses on the decisionmaking processes surrounding groundwater, watershed, and water supply. California's groundwater resources are managed by the California Department of Water Resources (CA-DWR 2019) and governed by the Sustainable Groundwater

Figure 5.4.2—Salvage logging, such as this operation following the 2017 Parker Fire on the Modoc National Forest, can provide economic value to local communities. Scenarios balancing economic value with other ecosystem services, such as habitat for wildlife, could be discussed among stakeholders before disturbance events in order to provide more timely and effective postdisturbance activities (photo by John Cichoski, Forest Service).

Management Act (SGMA). According to the SGMA, local agencies are required to form groundwater sustainability agencies (GSAs) to act as decisionmakers and plan-developers regarding their local groundwater basins (fig. 5.4.3). The DWR Groundwater Basin Boundary Assessment Tool indicates Modoc County has two medium-priority basins: the Klamath River Valley, whose sub-basin is Tule Lake, and Big Valley (CA-DWR n.d.). Lassen County shares the responsibility to manage Big Valley Basin with Modoc County, but has no other large- or medium-priority basins in its jurisdiction. Modoc and Lassen Counties oversee decisions regarding their respective and shared basins.


Figure 5.4.3—Local communities, engaged in the decisionmaking process concerning water management, can have a profound impact on the quality and quantity of water resources, such as Ash Creek on the Modoc National Forest (photo by Ken Sandusky, Forest Service).

Watershed decisions in Lassen County center around the Feather River watershed, which serves as the main tributary of the Sacramento River and is “an important source of water for California” (Charnley and Long 2014, p. 637). The Feather River flows through Lassen Volcanic National Park and the Lassen National Forest. A subset of watershed decisions included the examination of the supply-side amount of water allotted and available to a community, plus the demand-side amount of water used by that community—all in relation to how to protect the water supply, the quality of the water, and the various ways to educate people about their water supply.

Water management decisions conducted at the local level that include collaboration across multiple stakeholders have seen success, as in the institutional framework provided by California’s GSA requirement. The following studies also show how input from invested local communities can help local and Federal land management agencies determine information, education, and processes that enrich decision making.

A recent study of the Truckee River watershed in California and Nevada evaluated how restoration and land-protection scenarios impacted water quality and quantity, and illustrates the value of community input for resource forecasting (Podolak et al. 2017). Podolak and colleagues used the Natural Capital Project’s Resource Investment Optimization System model (Vogl et al. 2013) to design “four future land use scenarios, with activities targeted to the best locations for water quality and supply improvement” (Podolak et al. 2017, p. 125). They “collaborated with stakeholders who were interested in, benefit from, or that have specific regulatory requirements that could be met by improving water quality and supply in the watershed” (p. 125). They found that the input provided from stakeholders was critical to developing real-life scenario models. “By engaging stakeholders so thoroughly, we were not only able to complete a much more realistic set of models, but to also create information that has a much higher likelihood of getting used” (p. 134).

Another study that informs ecosystem service decisions asked how the distance between “forests to faucets” influenced peoples’ willingness to pay to protect their water supply (Adhikari et al. 2017, p. 2). Adhikari and colleagues found when the distance between forests and faucets was minimal, as in Santa Fe, NM, local residents were willing to pay a monthly service charge to protect their “city’s water supply from catastrophic wildfire” (p. 2). Meanwhile, people living farther from their water supply in Albuquerque, NM, were also willing to pay, but public educational programs were needed to increase their understanding of the effectiveness of watershed restoration (p. 24).

Biomass: Incentive Dilemmas and Landowner Variability

As discussed in Chapter 5.3 (Flores and Haire, this synthesis, *Ecosystem Services and Public Land Management*), the use of biomass for energy production can be costly. Therefore, to encourage biomass removal and reduce fire hazards, the Forest Service has examined several strategies including collaborating with private forested landowners to encourage pro-social conservation efforts. For example, Alpizar et al. (2017) studied how the Forest Service tried cash incentive rebates for landowners to privately conserve their land and contribute to the public good. The Forest Service ran into issues surrounding how best to incentivize landowners, and who to include or exclude from the rebate. The authors found "targeting of a new environmental rebate to those who have shown little pro-social or pro-environmental inclinations could negatively affect the motivation of those who did choose to contribute" (p. 200). While incentive programs to encourage private landowners to conserve their own land may work, the Forest Service can examine whether it is better to reward landowners already conserving the land or to incentivize nonconserving landowners to start conserving.

Additionally, the size of the property owned by landowners can influence forested land conservation. Landowners with properties greater than 494 acres (200 ha) received advice from the Forest Service and other organizations about how to manage their land, while those with properties less than 494 acres (200 ha) did not receive land management advice. Land ownership was quickly shifting from larger properties owned by a few people, toward smaller parcels owned by many people. Therefore, "as ownerships become increasingly fragmented, outreach focus and methods will need to shift to more effectively target the owners of smaller properties" (Ferranto et al. 2012, p. 1082).

Decisions surrounding biomass removal and production involves understanding not only how biomass can provide various benefits to a community (e.g., monetary, energy, production, jobs, funding), but also how biomass removal can reduce wildfire hazards. Because the demographics of private forested landowners is changing, land managers can draw upon the study by Alpizar et al. (2017) on incentivizing pro-social conservation efforts, as well as the demand of Ferranto et al. (2012) that education concerning the management of private forest land include owners of smaller properties when designing strategies to be utilized by the local forested communities.

Recreation and Agriculture: Appropriateness of Policy Direction and Sector Growth

According to Charnley et al. (2018) in the *Synthesis of Science to Inform Land Management within the Northwest Forest Plan Area* (Spies et al. 2018), Modoc and Lassen Counties are classified as nonrecreation dependent and nonretirement destinations. While the Lassen-Modoc region may not be considered a recreation destination, "cross-country skiing was frequently listed" as a recreational activity by Modoc County visitors (Winter et al. 2014, p. 515). Additionally, as mentioned earlier in Chapter 5.3 (Flores and Haire, this synthesis, *Ecosystem Services and Public Land Management*), research into Modoc's geothermal potential is underway, leading to potentially new recreational or other uses of the land (Moeller 2017).

Stem et al. (2003) examined how communities can participate in eco-tourism to "provide local economic benefits while also maintaining ecological integrity through low-impact, nonconsumptive use of local resources" (p. 388). One way is through establishing protected areas to encourage conservation of the land. Additionally, tourism-related park fees can be charged to visitors, allowing them to explore the protected area while also providing financial support. Stem et al. (2003) stressed the importance of "integrating environmental awareness raising and knowledge generation into eco-tourism activities" (p. 410) to increase conservation. The "education could not be limited to employees or the local communities. It could extend to the ecotourists themselves, with an emphasis on the ecological, cultural, and social history of the region they are visiting" (p. 410). Although nature-based tourism has been a growth industry with associated economic benefits and is considered one of the fastest growing industries, it is likely to be limited in "more remote areas with less charismatic species or scenery" (Balmford and Bond 2005, p. 1225). While the Lassen and Modoc areas may not offer exotic or threatened environments equal to the desires of ecotourists, the incorporation of environmental awareness across the local community to include the region's historical values may provide some insight.

The Lassen-Modoc region is, however, an area suited for ranching and agriculture. According to the local Hazard Mitigation Plan Update, the Modoc County Agricultural Commissioner's Office, and the U.S. Department of Agriculture, "agricultural production in Modoc County is the most significant contributor to the local economy" (Modoc County 2016).

Agricultural decisions about the use of forested land are greatly affected by "population-driven urbanization, the comparative returns to agriculture and forestry, and policies that influence the expression of the first two factors" (Vose et al. 2012, p. 106). Modoc County also cautions that agricultural and wildfire hazards could also be taken into consideration, because both are highly likely to occur and the damage caused by either would be catastrophic to an extensive area of the land (Modoc County 2016). State- and Federal-level land use policies that govern the direction of the bioenergy sector may be quite impactful to agriculture production in the county context. For example, Vose et al. (2012) note:

> “[t]he degree to which a bioenergy sector favors agricultural feedstocks, such as corn, or cellulosic feedstocks from forests will determine the comparative position of forest and crop returns to land use, and therefore land use allocations. The allocation among feedstock sources depends on energy policies at both federal and state levels, which could differentially affect rural land uses.” (p. 106).

Community Collaborations and Fire Management

The Lassen-Modoc area stakeholders posed a number of questions regarding community collaborations, especially around fire management:

- What are some examples of formal collaborations through structured agreements with local communities (e.g., Memoranda of Understanding)?
- What are some best practices for land management agencies to engage local communities in collaborative decision making and management actions?
- How does engaging the local public about fire (e.g., through Fire Safe Councils, partnerships, an all-hands approach, and using fire as a management tool) influence community engagement and decision making?

Land management agencies are using various adaptive strategies to encourage and engage local community collaboration surrounding fire management and the use of fire as a management tool (fig. 5.4.4). For example, case studies of “successful adaptation efforts in the United States” (Vose et al. 2012, p. ix), which focus on collaborative partnerships across science and management within national forests and national parks, are available for other land management organizations. These case studies include examples of internal cooperation across national and local agencies plus external collaboration across agencies and stakeholders, with a goal of demonstrating how adaptation strategies have previously been implemented. Development of one such adaptive approach using a science-management partnership as described by Vose et al. (2012) is exemplified by the case study of Littell et al. (2011) that demonstrates how management of the Olympic National Forest is accomplished in collaboration with the Olympic National Park, the University of Washington Climate Impacts Group, tribal groups, and private landowners. Additionally, the Strategic Framework for Science in Support of Management in the Southern Sierra Nevada, CA (Nydick and Sydoriak 2011) shows how Federal resource managers, local agencies, university scientists, and stakeholders collaborated to develop knowledge and decisionmaking tools regarding climate change and adaptation. (See Chapter 4 of Vose et al. 2012 for more examples.)

Figure 5.4.4—In the Western United States, fire management is an area where input from local stakeholders can guide the use of fire as a silvicultural tool to promote resilient forests. Successful engagement of stakeholders by land managers may require using a suite of adaptive strategies tailored to address the social and cultural aspects of a diverse stakeholder group (photo by Ken Sandusky, Forest Service).

In the Southwest, land management agencies have explored collaborative initiatives specific to using fire-reduction practices to protect wildland urban interface areas and using fire to reduce large-scale beetle epidemics (Vose et al. 2012). Regarding fire reduction, forest thinning and assertive use of fuel treatments can be implemented to change forest conditions to limit the intensity of a wildfire and reduce the possibility of crown fires (Vose et al. 2012). Fuel treatments (e.g., reducing forest fuels such as surface, ladder/small trees, and canopies) are completed prior to a wildland fire and are aimed at slowing down fire progression and creating a “defensible space” (Keller 2011, p. 12) around residential areas. An example of successful implementation of fuel treatments is demonstrated by the 2011 Wallow Fire, which started in the White Mountains in Arizona and spread to Western New Mexico. Previously implemented fuel treatments stopped the Wallow Fire from continuing as a "crown fire" and dropped the fire to the ground level, allowing firefighters to eventually stop the fire. See Keller (2011) for more information on fuel treatments and the 2011 Wallow Fire.

For these adaptive strategies to be implemented and produce benefits, "[s]ignificant financial resources and collaboration across different agencies and landowners will be necessary" (Vose et al. 2012, p. 221). For additional examples of formal collaboration in general and regarding fire management specifically, see Charnley et al. (2014).

Community Collaboration Strategies Regarding Wildland Fire

To understand community collaboration strategies regarding wildland fire, it is helpful to describe why fire management strategies are important to California communities. In California, communities located in or adjacent to forest and rangelands are highly focused on fire management strategies. This fire management emphasis is prevalent due to the devastating effects fire has on California's ecosystem, landscape, wildlife, landowners, agencies, local communities, and recreationists. A study concerning forest management in California observed that "fire hazard is clearly a topic for which all types of landowners support cooperative management, and many landowners are already implementing management to reduce fire risk on their land" (Ferranto et al. 2013, p. 1098). For example, landowners were most willing to cooperate with land neighbors and least willing to cooperate with Federal agencies. While landowners were willing to cooperate with local government agencies, their desire to cooperate decreased at the State level and again at the Federal level. Ferranto et al. (2013) claimed that themes discussed by Bergman and Bliss (2004) may be relevant to California, suggesting that cooperative land management efforts surrounding wildfire "may be more successful if implemented at a local level with local partners" (Ferranto et al. 2013, p. 1097).

Various management and decisionmaking strategies focused on cooperative engagement to reduce fire risk are being implemented by local communities and agencies (fig. 5.4.5). Charnley et al. (2014) identified different processes and approaches to the collaborative management of wildland fire, such as the formation of community councils and networks, which address the risks of wildfire and coordinate prescribed fire burning. These community-based organizations were comprised of multiple entities—Federal, State, and local agencies; tribes, academic institutions, landowners, and local residents—and included Fire Safe Councils, prescribed fire councils, and Fire Learning Networks. These councils and networks, along with other community-based groups, facilitated education outreach programs, participatory action research, cooperative forest landscape restoration projects, and community wildfire protection plans. The intention was to help engage the communities in fire-prone areas with practices of wildland fire hazard reduction and to introduce fire as a management tool.

Collaborative programs between land management agencies and local communities surrounding fire management and the use of fire as a management tool have been beneficial. For example, collaborative efforts to reduce wildland fire have returned several co-benefits including the protection of property, increased forest resilience to periodic wildfire, and reduction of wildfire intensity, crown fires, tree mortality, and suppression difficulty (Vose et al. 2012). Additionally, while more research is needed to understand the benefits from collaborative programs between Federal forest managers and tribal communities, these processes "may yield important social and ecological benefits, including landscape heterogeneity" (p. 182). Furthermore, collaboration with tribal communities and "reintroducing traditional Native American burning practices" may provide forest managers "opportunities to learn about these fire effects and incorporate them into forest management practices and applied restoration efforts" (p. 182).

Such collaborations between Federal forest managers and tribal communities have attempted to "incorporate tribal traditional ecological knowledge in research and forest management and to respect tribal needs and traditions regarding access and caretaking" (Lake and Long 2014, p. 180). These collaborative strategies include consulting with and forming partnerships across "forest managers and tribal governments, communities, individuals (where appropriate), and organizations (e.g., the California Indian Basketweavers Association)" (p. 180). Forest managers asked questions and listened to stories surrounding tribal needs specific to "habitats, specific plants, or other valued resources" (p. 180), as well as the "season, frequency, or intensity" (p. 181) related to traditional burning strategies of tribal cultural values.

Figure 5.4.5—Public engagement that fosters cooperation can lead to different processes and approaches in how that cooperation proceeds administratively as well as by identifying on-the-ground approaches to reduce fire risk (photo by Ken Sandusky, Forest Service).

Examples of forest management and tribal community collaboration include partnerships between the Modoc and the Cultural Advocates for Native Youth "to restore native tobacco plants at burn piles" (Lake and Long 2014, p. 181), plus partnerships between the Lassen and Maidu Tribes to restore beargrass to the region (Charnley et al. 2008). Lastly, the Klamath and Six Rivers National Forests and Karuk Tribe integrated ways "to allow for specific cultural management activities, including reintroduction of fire onto the landscape" through the ceremonial burning of "the mountain above Katimiin, a historical village site" (Lake and Long 2014, p. 181).

Integrated Systems of Landscape, Community, and Management

Decisions and actions regarding landscapes, communities, and land management are interrelated, with processes of one or more entities potentially impacting the others. Judgments surrounding land projects are sometimes decided upon and conducted solely by land management agencies, while other projects are undertaken in partnership with local communities. We discuss integrated socioecological systems in this section by examining successful agency-community partnerships involving habitat improvement and forest restoration. We keep in mind the following questions asked by the Lassen–Modoc stakeholders:

- How do land management agencies address the interrelatedness of landscapes, people, and management actions?
- What are the benefits of restoration and habitat improvement projects that have been conducted through local partnerships?

Shared Stewardship

In 2018, the Forest Service published a report on an initiative for shared stewardship and decision making with States, partners, and tribes to "identify landscape scale priorities for targeted treatments in areas with the highest payoffs" (USDA Forest Service 2018). Through shared stewardship, the Forest Service envisions multiple stakeholders coming together across landscapes to co-manage risk, target investments using the latest science tools, focus on outcomes at the appropriate scale, and develop new approaches to deal with the wildland fire environment. The Forest Service initiative argues that across broad landscapes, shared stewardship between the Forest Service, States, and other stakeholders using science-based approaches are needed to assess risk, evaluate tradeoffs, manage insect epidemics, restore watersheds, and conserve species at risk. While different States and stakeholders have different mandates, the impacts of fire, insect outbreaks, and other disturbances do not have land ownership boundaries. Thus, by using a shared stewardship approach, the Forest Service designed a strategy to co-manage large landscapes by bringing their partners and stakeholders together to maximize access to existing science tools, set goals, priorities, tradeoffs, and make decisions on where to invest resources.

Shared stewardship is also referred to in the social science literature as co-management. For forest management, shared stewardship represents a shift from centralized governance toward local community involvement. Co-management is primarily concerned with user participation in decision making and with linking communities and government managers. In a collaborative management context, local knowledge and experience have equal status with experts and expert knowledge (Cardinal and Day 1999). Adaptive management is a concept that is similar to co-management, but places a greater emphasis on learning-by-doing in a scientific way to deal with uncertainty. An adaptive management approach encourages learning throughout structured experimentation and management flexibility (Hilborn and Walters 1992). Although co-management and adaptive management may be more effective in meeting biological and socioeconomic goals than other types of management, such as centralized governance, current research suggests that co-management can lead to local perceptions of inequality (Ward et al. 2018). Thus, the shift from co-management and adaptive management to shared stewardship implies multiple ownership and knowledge that leads to joint responsibility for land management (Laronde 2016). In other words, shared stewardship redistributes power dynamics from centralized top-down resource management across all stakeholders. The benefit is that when individuals and groups take different levels of responsibility for different landscape types, shared stewardship draws on dynamic social networks that hold reserve social and economic capital, as well as expertise across groups that could be better utilized (Svendsen and Campbell 2008).

Shared stewardship is an effort to bridge co-management and adaptive management, recognizing that ecological systems are dynamic and nonlinear. Shared stewardship in the 21st century takes place within an interdisciplinary effort, and challenges established assumptions of scientific certainty, stability paradigms in both ecological and social sciences, and the reliance upon expert solutions. The vision of shared stewardship is continuous learning in order to adapt to rapid changes and complexities that consider humans and ecosystems as an inseparably linked social-ecological system (Armitage et al. 2007).

Polycentric Governance

Bixler (2014) describes how landscapes, communities, and land management are integrated by stating: “all actors are somehow already involved, albeit in different ways” (p. 164) in the management of the forest. To understand the concept of actor involvement, “a shift in thinking about forest users, stakeholders, and other actors” (p. 164) can take place. This shift requires forest management agencies and researchers to move from a mindset that forest agencies can “‘involve’ them [local communities] or give them a voice in forest management decisions” to a mindset of “polycentric governance” (p. 164) or a realization that actors are already embedded and interrelated in an ecological system. This notion of embeddedness involves recognizing how the role of one actor can impact the roles of other actors.

Community collaboration around fire risk management is an example of working through the inter-relatedness within socioecological systems and how multiple actors can influence fire incidents. Or as Adhikari et al. (2017) point out, “high-severity wildfires present significant risk exposure to interconnected natural and human systems” (p. 3). For example, landowners who practice pro-environmental behaviors and remove biomass and fire hazard timber help reduce the start and spread of a fire; while landowners who do not practice fuel removal may aid fire events (Charnley and Long 2014). Outreach efforts concerning wildland fire management from local agencies and organizations help reduce wildland fire, but only when all land actors, not just landowners of larger properties, are informed (Ferranto et al. 2012). Tribal communities assist in fire management efforts due to their knowledge and utilization of the forested land achieved from centuries of experience (Lake and Long 2014), yet past interactions between government agencies and tribal communities have resulted in a distrust of government from tribes (Norgaard 2007). Furthermore, landscape features (e.g., climate, presence of biomass, topography, etc.) greatly contribute to wildland fire activity (Collins and Skinner 2014), including watershed and groundwater impacts on the level of moisture in the landscape. Wildfire and water quality are therefore mutually interactive (Adhikari et al. 2017, Podolak et al. 2017). The dynamic processes of wildland fire also include the knowledge and actions of recreation users, fire councils, firefighters, search and rescue teams, and others who contribute to the start, spread, and extinction of a wildland fire.

Government land management agencies have conducted a variety of restoration and habitat improvement projects in partnership with local communities. From multiple entity councils (e.g., fire safe and prescribed fire councils) to educational outreach (e.g., fire learning, forest ecosystems); community participation action research and plans (e.g., community wildfire protection plan) to restoration projects (e.g., Collaborative Forest Landscape Restoration Projects, Wyden Authority Projects, Stewardship Contracting), these forest management collaborative efforts have seen both positive and negative results (Charnley et al. 2014). Partnerships formed across agencies and local communities that include a focus on habitat improvement and forest restoration can provide interrelated co-benefits to the Lassen-Modoc region. These co-benefits include accountability and ownership of forest management across multiple actors and "opportunities to redress underrepresentation in resource management" (p. 663). Community collaboration projects and conflict management are discussed further in the next section.

Conflict as a Tool for Creative Resolution

Interrelated socioecological systems often produce conflict across multiple participatory actors. When conflict arises, land managers can address these problems. Below are two scenarios regarding socioecological conflict, tension regarding forest road access and disagreements over subsistence resources, and how land managers addressed these tenuous situations using creative resolution.

Resolving Conflict Concerning Forest Road Access

In forest management, conflict surrounding "roads to access natural resources" (Hunt et al. 2009, p. 128) is prevalent (fig. 5.4.6). While a plethora of actors participate in these natural resource road systems, some key influencers include users with: industrial interests such as commercialization, construction, and development; forest management and operation interests; residential interests; tourism and recreation interests; socially and culturally significant interests such as gathering and hunting; and environmental interests. Additionally, animals, forest ecology, vehicles, and so forth participate in this system. Therefore, resource managers can balance their decisions concerning roads and road use carefully, as their judgments can have "varying effects on people and ecological systems" (Hunt et al. 2009, p. 128).

Conflict concerning forest road access is high among "tourism operators and recreationists" (Hunt et al. 2009, p. 129). These conflicts are mainly centered on five themes: goal interference, social values, process inequity, distributive inequity, and context. Conflict surrounding goal interference evolve when the "behaviors of some individuals interfere with the desired outcomes of others" (p. 133). Conflict over social values develop "when individuals have different views over acceptable uses of lands" (p. 133). Conflict concerning process inequity emerge when the decisionmaking process surrounding forest management planning and road management is

Figure 5.4.6—The use of forest roads can be a contentious topic. Engaging and involving the community in decision making can help land managers better understand the social, managerial, and environmental contexts necessary to resolve conflict (photo by Ken Sandusky, Forest Service).

perceived as imbalanced. Conflict regarding distributive inequity arise when "the distribution of economic benefits from past decisions" (p. 135) are perceived as unfair. Lastly, a lack of understanding of the various "social, physical, and managerial contexts" (p. 136) of participating actors also contributes to the conflict.

To understand further the "reasons for conflict over forest road access" (p. 137), Hunt et al. (2009) designed a conceptual model to illustrate this interrelated conflict system, as seen in figure 5.4.7. In this figure, the "relationships among the key themes" (p. 138) is apparent. As Hunt et al. (2009) state:

> Goal interference will affect perceptions of distributive inequity from management decisions and conflict in instances when people disrespect management decisions (e.g., using motorized vehicles in prohibited areas). Social values differences result in varying views on rights of using publicly owned resources and needs to protect remoteness as a value. These social values may affect perceived fairness of decisionmaking processes (process inequity) and the outcomes from these processes (distributive inequity). When people disrespect management decisions, social values may directly lead to conflict. Perceived process inequity will result from differing contexts and social values (e.g., expectations for a public decisionmaking process). These perceptions will affect the ways that individuals evaluate the fairness of outcomes from the process (i.e., distributive inequity). Finally, perceptions of unfair decisions may lead to conflict between remote tourism and road-based recreation interests (p. 138).

Using this conflict conceptual model as a guide, resource managers may be able to identify the "potential causes of conflict for their case" (p. 140) and adjust their management and decision making toward mitigating the sources of conflict. Managers can apply this conflict concept in practice in several ways. For example, managers can reexamine past decision making that was perceived as unfair and introduce ways to ensure decisions are made using a consensus-based approach (e.g., making sure to include local stakeholders, residents, and tribes in the process, and facilitate opportunities for these citizens to provide feedback). Once a consensus-based approach is initiated, this process could be continued for future decision making. Additionally, education tools can be implemented to train citizen decisionmakers. Lastly, managers can identify the reason for the conflict in order to ultimately minimize the conflict.

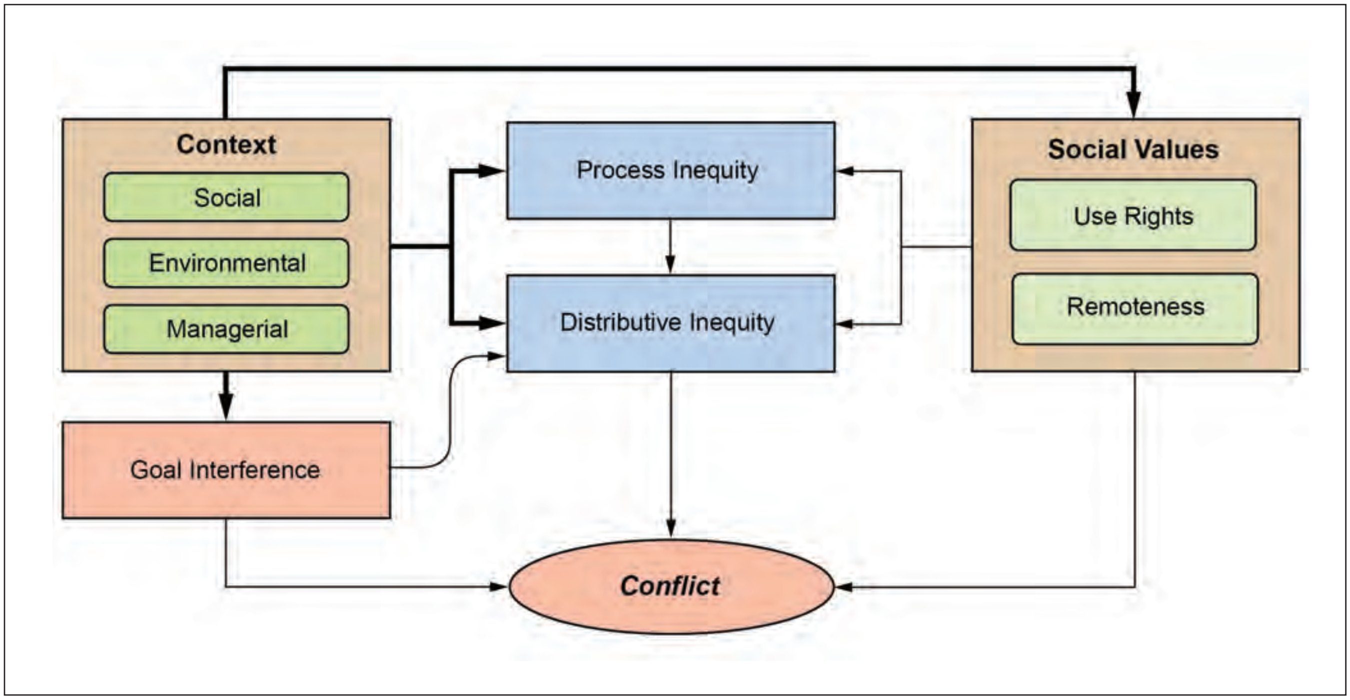

Figure 5.4.7.–Conceptualizing sources of conflict for forest road access management. Bold lines indicate relationships that strongly influence conflict (from Hunt et al. 2009, p. 138).

Resolving Conflict Surrounding Subsistence Resources and Nontimber Forest Products

Within forest management, conflict surrounding subsistence resources, also referred to as "a way of life" (Dick 1996, p. 19) including hunting and gathering, and nontimber forest products (NTFPs) is prevalent. Conflict concerning subsistence resources stems from a "difference in perspective" (p. 19) across groups who use the land for subsistence, and the different types of uses that user groups want to make of limited resources. Some have attributed conflict surrounding NTFPs on the "power differentials between public forest managers and forest-dependent communities" (Watson 2017, p. 333); specifically, how the Federal government manages forest resources. Summaries of two case studies further understanding on how to creatively resolve conflict within forest management: mitigating conflict surrounding subsistence resources (Dick 1996) and NTFPs (Watson 2017).

Dick (1996) claimed that "[w]ithin a subsistence community, the use of wild, renewable natural resources is characterized by sharing within a community identity" (p. 26). Subsistence sharing leads to the establishment of interrelated and interdependent relationships across the community, thereby developing strong partnerships. When forest managers are interacting with rural areas such as tribal communities that practice subsistence values, managers can be aware that placing an economic value on subsistence practices is not possible. Instead, managers can learn about the social and cultural significance and "integrated whole" (p. 26) of the subsistence process. When forest managers learn more about subsistence practices, they will be able to establish stronger relationships and mitigate conflict with tribal and rural forested communities.

For example, forest management practices often examine the subsistence harvest of a natural resource and believe the practice begins and ends at the harvest. Instead, the harvest "consists of pre-harvest activities, harvest, processing, distribution and exchange, preparation, and barter" (Dick 1996, p. 26). "It is the complex system of customs, norms, mores, and values guiding these activities that deserves the focus, because it is fundamentally different from, and often opposed to, the relationships emphasized by the flow of commodities through the marketplace" (p. 26).

To help forest managers understand subsistence resources, Dick (1996, p. 27) proposes six basic tenets that forest managers can be aware of when managing subsistence resources:

1. Subsistence practices may supplement cash income for some people, while "for others it is cash income that supplements the subsistence culture or lifestyle."
2. Subsistence activities can help people during "times of decline or collapse of market economies," therefore, these "subsistence resources could be protected."
3. Subsistence activities can help people during "tough times (e.g., illness in the family or old age)," therefore "subsistence resources could be protected so they can serve to mitigate the adverse impacts of disabilities."
4. Decisionmaking surrounding the management and allocation of resources could take into account the values of subsistence resources (these values may be tangible and/or intangible).
5. To properly analyze subsistence processes, a development of "new methodologies and expanded databases for comparisons between subsistence values and market values for the affected resources" are needed.
6. Lastly, "to address the meaning of subsistence resources in the context of community sharing," forest managers can conduct qualitative analysis.

Watson's (2017) case study of tupelo honey production, an NTFP, in rural northwest Florida, illustrates another example of creative conflict resolution. In Florida's honey industry, conflict surrounding tupelo honey production is high across beekeepers, honey harvesters and producers, private landowners, commercial suburban developers, and public land managers.

Watson identified several contributing factors that assisted in the conflict across forest managers and forest-dependent stakeholders. These factors were: invasive pests and disease, land use change and development, and safe and productive placement for beehives. While these factors have limited tupelo honey production, they have also "increased the need for beekeepers to access and use public lands" (p. 337). Yet, beekeepers have faced difficulty when trying to access public lands, from negotiating permits and being granted access to public sites to combating conservation or restoration efforts already prescribed in the local forest management plan. Thus, it is important for forest managers to understand the intricacies of balancing the management of “national forests under multiple-use goals for a diversity of disparate stakeholders” (p. 341). Moreover, it is helpful to recognize how “[t]he current structure of public land management effectively limits the ability of some stakeholders to participate in the goals and outcomes of forest management” (p. 341).

Therefore, forest managers can work to recognize the importance of NTFPs and the knowledge that local forest communities have regarding forest resources. Forest managers can find ways to empower and cooperate with local forest communities to foster “new possibilities in resource management” (p. 342). Additionally, conflict can be reduced when communication across forest managers and local users is increased, as well as when “long-term goals of public land managers and the needs of local resource users” (p. 342) are in congruence.

Conclusion

This chapter explored how the Forest Service can work to seek better local engagement, coordination, and involvement of local communities. For example, decisions about timber production included a focus on forest restoration and wildland fire rehabilitation. Decisions about forest restoration were successful when ongoing communication from the Forest Service to the community and from the community to the Forest Service occurred. Decisions about water were beneficial when made in collaboration with stakeholders of local watersheds. Decisions about biomass, including the removal of biomass to reduce wildland fire hazards, utilized input from local forested landowners. The study by Alpizar et al. (2017) on incentivizing pro-social conservation efforts and the request by Ferranto et al. (2012) to educate all forested landowners may provide helpful information for design of biomass removal and land conservation used by local forest communities. Lastly, decisions concerning recreation and other land use included understanding how policy, population and urbanization, and forestry and agriculture influence land use. The incorporation of environmental awareness with knowledge of the region’s historical and emerging values may provide some insight into how Lassen and Modoc communities determine their forest ecosystem service interests.

Additionally, this section explored how different user groups form and attach social, cultural, and economic values to their forested lands. These values often related to the communities’ interest in and knowledge of their forests. In California, demographic trends have shifted, resulting in an influx of new forested landowners, but with a decrease in the size of their properties. Long-time forested landowners are moving away or dividing their land, allowing new residents to move to rural areas. These shifts have led to values less centered on vegetation and land management, and instead toward values focused on amenities, natural beauty, and land conservation.

Furthermore, this chapter examined how government land management agencies, in partnership with local communities, have conducted a variety of restoration and habitat improvement projects. Forest management collaborative efforts have proven beneficial when the input from stakeholders and local communities was integrated into project plans. Part of these collaborative efforts included the management of wildland fire and discovering ways to help engage communities in fire-prone areas surrounding wildland fire hazard reduction and how to use fire as a management tool. These collaborative efforts have returned several co-benefits to the forested communities, including ways to protect their properties, increase their forest resilience to periodic wildfire, and reduce the overall intensity of wildfire in their areas.

This chapter may serve as a tool to increase understanding of how the Forest Service can engage with and sustain the livelihoods of residents in the Lassen-Modoc area by working with the community, and, in return, how the local community can participate with the Forest Service. These collaborative systems could include input from stakeholders and local communities that are invested in their land and in the ecosystem services provided by the land. Community collaboration can help local and Federal land management agencies create information, education, and processes that may be more likely to resonate with local uses and, therefore, be more successfully enacted.

References

Adhikari, D.; Thacher, J.A.; Chermak, J.M.; [et al.]. 2017. Linking forest to faucets in a distant municipal area: public support for forest restoration and water security in Albuquerque, New Mexico. Water Economics and Policy. 3(1): 34.

Alpizar, F.; Norden, A.; Pfaff, A.; [et al.]. 2017. Unintended effects of targeting an environmental rebate. Environmental & Resource Economics. 67(1): 181–202.

Ambus, L.; Hoberg, G. 2011. The evolution of devolution: a critical analysis of the community forest agreement in British Columbia. Society & Natural Resources. 24: 933–950.

Armitage, D.; Berkes, F.; Doubleday, N. 2007. Adaptive co-management: collaboration, learning, and multilevel governance. Vancouver, BC: UBC Press. 360 p.

Balmford, A.; Bond, W. 2005. Trends in the state of nature and their implications for human well-being. Ecology Letters. 8(11): 1218–1234.

Berkes, F. 2010. Devolution of environment and resources governance: trends and futures. Environmental Conservation. 37(4): 489–500.

Bixler, R.P. 2014. From community forest management to polycentric governance: assessing evidence from the bottom up. Society & Natural Resources. 27(2): 155–169.

California Department of Water Resources [CA-DWR]. 2019. Actions for local agencies to follow when deciding to become, form, or modify a groundwater sustainability agency (GSA). California Department of Water Resources. https://water.ca.gov/-/media/DWR-Website/Web-Pages/Programs/Groundwater-Management/Sustainable-Groundwater-Management/Groundwater-Sustainability-Agencies/Files/GSA-Formation-Process-FINAL-2019-05-10_ay_19.pdf (25 Feb. 2020).

California Department of Water Resources [CA-DWR]. n.d. Groundwater Basin Boundary Assessment Tool. California: California Department of Water Resources. https://gis.water.ca.gov/app/bbat/ (26 Aug. 2017).

Cardinal, D.; Day, J.C. 1998. Embracing value and uncertainty in environmental management and planning: a heuristic model. Environments. 25 (2&3): 110–125.

Charnley, S.; Fischer, A.P.; Jones, E.T. 2008. Traditional and local ecological knowledge about forest biodiversity in the Pacific Northwest. Gen. Tech. Rep. PNW-GTR-751. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 52 p.

Charnley, S.; Kline, J.D.; White, E.M.; [et al.]. 2018. Socioeconomic well-being and forest management in Northwest Forest Plan area communities. In: Spies, T.A.; Stine, P.A.; Gravenmier, R.; [et al.], tech. coords. 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: 625–715.

Charnley, S.; Long, J.W. 2014. Managing forest products for community benefit. In: Long, J.W.; Quinn-Davidson, L.N.; Skinner, C.N., eds. 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: 629–662.

Charnley, S.; Long, J.W.; Lake, F.K. 2014. Collaboration in national forest management. In: Long, J.W.; Quinn-Davidson, L.N.; Skinner, C.N., eds. 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: 663–704.

Collins, B.; Skinner, C. 2014. Fire and fuels. In: Long, J.W.; Quinn-Davidson, L.N.; Skinner, C.N., eds. 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: 143–172.

Dick, R.E. 1996. Subsistence economics: freedom from the marketplace. Society & Natural Resources. 9(1): 19–29.

Ferranto, S.; Huntsinger, L.; Getz, C.; [et al.]. 2013. Management without borders? A survey of landowner practices and attitudes toward cross-boundary cooperation. Society & Natural Resources. 26(9): 1082–1100.

Ferranto, S.; Huntsinger, L.; Stewart, W.; [et al.]. 2012. Consider the source: the impact of media and authority in outreach to private forest and rangeland owners. Journal of Environmental Management. 97: 131–140.

Hart, J.W. 1995. National forest planning: an opportunity for local governments to influence federal land use. Public Land and Resources Law Review. 16: 138–161.

Hilborn, R.; Walters, C., eds. 1992. Quantitative fisheries stock assessment: choice, dynamics and uncertainty. New York: Springer U.S. 570 p.

Hunt, L.M.; Lemelin, R.H.; Saunders, K.C. 2009. Managing forest road access on public lands: a conceptual model of conflict. Society & Natural Resources 22(2): 128–142.

Keller, P., tech. ed. 2011. How fuel treatments saved homes from the 2011 Wallow fire. Tucson, AZ: Wildland Fire Lessons Learned Center. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5318765.pdf (29 Sep. 2017).

Lake, F.R.; Long, J.W. 2014. Fire and tribal cultural resources. In: Long, J.W.; Quinn-Davidson, L.N.; Skinner, C.N., eds. 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: 173–186.

Laronde, M. 2016. Co-management of lands and resources in n'Daki Menan. In: Mawhiney, A-M., ed. Rebirth: political, economic and social development in First Nations. Toronto: Dundurn Press. 300 p.

Larson, A.; Soto, F. 2008. Decentralization of natural resource governance regimes. Annual Review of Environment and Resources. 33: 213–239.

Littell, J.S.; Peterson, D.L.; Millar, C.I.; [et al.]. 2011. U.S. national forests adapt to climate change through science-management partnerships. Climatic Change. 110: 269–296.

McCool, S.F.; Burchfield, J.A.; Williams, D.R.; [et al.]. 2006. An event-based approach for examining the effects of wildland fire decisions on communities. Environmental Management. 37(4): 437–450.

Modoc County. 2016. California Local Hazard Mitigation Plan Update. Public Review Draft. Modoc, CA: Modoc County. 438 p. http://www.cityofalturas.us/Modoc%20County%20LHMP%203.4.16%20Draft.pdf (25 Aug. 2017).

Moeller, J. Page 2017. Geothermal. Modoc, CA: County of Modoc website. http://www.co.modoc.ca.us/departments/geothermal (25 Aug. 2017).

Morgan, T.A.; Brandt, J.P.; Songster, K.E.; [et al.]. 2012. California's forest products industry and timber harvest, 2006. Gen. Tech. Rep. PNW-GTR-866. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 48 p.

Morgan, T.A.; Keegan, C.E., III; Dillon, T.; [et al.]. 2004. California's forest products industry: a descriptive analysis. Gen. Tech. Rep. PNW-GTR-615. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 55 p.

Norgaard, K.M. 2007. The politics of invasive weed management: gender, race, and risk perception in rural California. Rural Sociology. 72(3): 450–477.

Nydick, K.; Sydoriak, C. 2011. The strategic framework for science in support of management in the Southern Sierra Nevada, California. Park Science. 28: 41–43.

Podolak, K.; Lowe, E.; Wolny, S.; [et al.]. 2017. Informing watershed planning and policy in the Truckee River Basin through stakeholder engagement, scenario development, and impact evaluation. Environmental Science & Policy. 69: 124–135.

Poteete, A.R.; Welch, D. 2004. Institutional development in the face of complexity: developing rules for managing forest resources. Human Ecology. 32(3): 279–311.

Ryan, R.L.; Hamin, E. 2008. Wildfires, communities, and agencies: stakeholders' perceptions of postfire forest restoration and rehabilitation. Journal of Forestry. 106(7): 370–379.

Ryan, R.L.; Hamin, E. 2009. Wildland-urban interface communities' response to post-fire salvage logging. Western Journal of Applied Forestry. 24(1): 36–41.

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.

Stem, C.J.; Lassoie, J.P.; Lee, D.R.; [et al.]. 2003. Community participation in ecotourism benefits: the link to conservation practices and perspectives. Society & Natural Resources. 16(5): 387–413.

Svendsen, E.; Campbell, L.K. 2008. Urban ecological stewardship: understanding the structure, function and network of community-based urban land management. Cities and Environment. 1(1): 1–32.

Taylor, P.L.; Cheng, A.S. 2012. Environmental governance as embedded process: managing change in two community-based forestry organizations. Human Organization. 71(1): 110–122.

U.S. Department of Agriculture, Forest Service [USDA Forest Service]. 2018. Toward shared stewardship across landscapes: an outcome-based investment strategy. FS-1118. Washington, DC. 24 p.

Vogl, A.; Tallis, H.; Douglass, J.; [et al.]. 2013. Resource investment optimization system (RIOS): Introduction and theoretical documentation. Data series v1.1.0. Stanford, CA: Stanford University, The Natural Capital Project. https://data.naturalcapitalproject.org/rios_releases/RIOSGuide_Combined_07May2015.pdf (10 Sep. 2017).

Vose, J.M.; Peterson, D.L.; Patel-Weynand, T., eds. 2012. Effects of climatic variability and change on forest ecosystems: a comprehensive science synthesis for the U.S. Forest Sector. Gen. Tech. Rep. PNW-GTR-870. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 265 p.

Ward, C.; Stringer, L.C.; Holmes, G. 2018. Protected area co-management and perceived livelihood impacts. Journal of Environmental Management. 228: 1–12.

Watson, K. 2017. Alternative economies of the forest: honey production and public land management in Northwest Florida. Society & Natural Resources. 30(3): 331–346.

Winter, P.L.; Long, J.W.; Lake, F.K.; [et al.]. 2014. Broader context for social, economic, and cultural components. In: Long, J.W.; Quinn-Davidson, L.N.; Skinner, C.N., eds. 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: 501–542.

Chapter 5.5. Integrating Tribes and Culture Into Public Land Management

David Flores and Gregory Russell[1]

Many of the cultural traditions practiced by Native Americans were channeled from or associated with their experiences with the natural world. These traditions, in turn, served to inform land management practices that effectively maintained a sustainable ecological balance among people and land for thousands of years. Today, many Native Americans find it difficult to continue the ecological and cultural, or "ecocultural" practices of their ancestors (Long et al. 2018). Here we explore some of the factors that give rise to these challenges. This chapter provides a general overview of traditional land management practices employed by Native American Tribes throughout the American West.

The consequences of Native American land management practices being excluded from decision making are explored, but so too are the ways in which land management agencies have started to become more accepting of these practices. Many of the authors and researchers presented in this chapter contend that factors such as sense of place and traditional ecological knowledge could play a larger role in the decisionmaking process for land management agencies, not only because of the ecological benefits that come with restoring traditional practices but also to provide an avenue for the preservation of important aspects of cultural heritage.

These diverse characteristics and identities present opportunities for multifaceted, and therefore flexible, collaborative decision processes appropriate for managing periods of environmental, institutional, and economic transition. In addition to the economic attachments to reservation and traditional lands, Native Americans have maintained long-established cultural and spiritual connections to the natural environment that precede western economic measures of well-being. For example, the use of prescribed burning as a land management tool plays a prominent and key role in the practices of many tribes.

Guiding Questions

- How do land management agencies include tribal traditional ecological knowledge practices in forest planning and decision making?
- How do land management agencies include spiritual components (i.e., sense of place for tribes) in land management planning?

The United States judicial system differs in the way it litigates cases that deal with Native American rights: In some cases, tribal nations are regarded as sovereign entities with all the rights afforded to other nation states, while in other cases, tribal nations are subject to the same Federal and State controls that oversee other governmental bodies (Stidham and Carp 1995). These differences significantly impact the ways in which Native American Tribes develop and implement land management practices. While courts have granted Native American Tribes sovereignty over some ancestral or reservation lands, these rights are often not enough to empower tribes with land access benefits, as power over these lands ultimately remains under the purview of government and private industry (Wyatt et al. 2015).

A significant difference in land management perspectives exists between governmental agencies and tribal entities. At the crux of this difference lie the ways in which agencies and tribes choose the economic benefits of land versus its cultural uses. Representatives from government agencies tended to emphasize the economic benefits of land, while tribal leaders are, for the most part, dismissive of economic benefits (Wyatt et al. 2015). This rift is further exacerbated when considering the problems that arise from managing the cultural aspects of ecosystems through socioeconomic policies (Pleasant et al. 2014), as many Native Americans regard the right to maintain a cultural connection to the land just as important as any other land-right (Curran and M'Gonigle 1999). Indeed, Burger and Gochfeld (2010) have shown that it is more common for Native Americans to engage in spiritual activities, such as communing with nature and praying or meditating in a natural setting, than it is for Caucasians (fig. 5.5.1).

Also significant is the difference in metaphysical explanations of the natural world between westernized scientists and Native Americans who draw from traditional knowledge (Ermine 2007); as a result, it is often difficult

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Figure 5.5.1—Native Americans often view the natural world differently than do government agencies, particularly in terms of economic benefits, cultural connections, and spirituality (photo by John Cichoski, Forest Service).

for one group to fully appreciate the other's worldview (Christianson 2015). Obviously, the western scientific tradition, and the worldviews that arise from it, possess great import to modern civilization, but this is not the only means for assessing the world and experience. The conclusions reached by indigenous science are distinct from those found in western science, and both are legitimate in their own right; for thousands upon thousands of years, indigenous people have experienced the environment in a deep and emotional way (Dongoske et al. 2015). This perspective has left many tribal managers with a desire to pursue a fresh start by initiating their own land management institutions (Diver 2016).

As Huntsinger and Diekmann (2010) note:

> Since the mid-1990s, the Yurok Tribe has run its own forestry program, although the timber harvest plans they prepare can be done in accordance with federal rules and approved by the BIA and other agencies. Today, the Yurok Tribe uses Douglas-fir harvest as an income source, but some land is dedicated to production of basketry materials, and redwoods are left uncut. Yurok forestry now includes burning for beargrass production, clearing brush around the homes of elders, reducing fuels, creating fuel breaks, making posts and poles for traditional structures, watershed restoration, and selling timber. Spiritual leaders play a significant, if somewhat informal, role in forest management in terms of the BIA-mandated institutional structure (p. 368).

Researchers are beginning to realize the importance of including a variety of Native American perspectives when developing ecological policy that is intended to protect humans and the environment (Greenberg and Crossney 2006). One of the ways in which tribes and State agencies have successfully collaborated after environmental contamination is by conducting a Natural Resource Damage Assessment. Serving as the basis for many management practices, it concerns "determining status and trends of biological, physical, or chemical/radiological conditions, conducting environmental impact assessments, performing remedial actions could remediation fail, managing ecosystems and wildlife, and assessing the efficacy of remediation, restoration, and long-term stewardship" (Burger 2008, p. 6).

To effect holistic understanding of the environment and its stressors, environmental analyses would only be enhanced with the inclusion of Native American perspectives and insights (Dongoske et al. 2015). A policy of sustainable equity could guide management decisions by linking these decisions with social justice and a rights-based interpretation of equity (Huntsinger and Diekmann 2010). Even though including Native American perspectives in land management decisions can be time-consuming and intensive, it can also be rewarding for communities and State agencies (Christianson 2015).

Part of this inclusion entails considering aspects of ecosystem valuation typically not considered. Social, sacred, and cultural aspects of ecosystems have historically been overlooked in land management decision making but are crucial to Native Americans (Burger and Gochfeld 2010). Native American knowledge structures can often be characterized by the deep intergenerational origins that involve family members and are typically conveyed through storytelling or demonstration, embedding within the knowledge structure a cultural situation or context that represents ontological understanding (Grenier 1998).

Instead of framing land management policy entirely around the consumptive properties of nature, environmental values can integrate detailed knowledge of regional ecological conditions with a conservation ethos, especially when considering Native American cultural traditions of places and landscapes (Winthrop 2014). As stated by Peppler (2017), a cultural model of knowledge formation helps "describe the tacit understandings people have about the world around them, and provide insight on how people perceive, remember, and describe natural features and how they understand, utilize, and manage natural resources and their surroundings" (p. 325).

The following subsections are prefaced with specific questions generated from the public and the staffs of the Lassen and Modoc National Forests (hereafter, Lassen, Modoc, or Lassen-Modoc) as part of the process for these two forests to revise their forest plans. While the literature addresses general topics related to these questions, due to the very limited literature on these topics, each specific question cannot be addressed solely using peer-reviewed scientific literature. Therefore, each specific question could be answered more pointedly in the forest plan using the literature in this section as a general guide. Where possible, specific studies to the region are cited below.

Tribal Value of Place

- How do local tribes define "sense of place" across the Lassen-Modoc region?
- What are the social and cultural values that local tribes attach to the Lassen-Modoc region?
- What are the cultural, social, economic, and spiritual uses of tribes in the region?

Place identity concerns the personal relationships human beings form with the physical environment that manifest into experiences of significance and meaning (Gunderson and Watson 2007). It is typical for experiences of this nature to conjure knowledge forms that are localized, place-based, and recognized through the patterns that come with continually inhabiting a place (Lauer and Matera 2016). Knowledge of this sort permeates a situational perceptiveness about a place by producing an intuitive wisdom that is key for recognizing certain types of environmental phenomena, which may not be available through other sources of weather and climate data (Peppler 2017).

Belief systems that arise from this sort of knowledge paradigm have been described as sacred and holistic, engendering reciprocity between humans and the nonhuman world to the point where human beings are inseparable from our surroundings (Deloria 2006). Researchers, such as Cruikshank (2012), conceptualize indigenous forms of holistic knowledge in ways that regard animals and features of landscape as possessing characteristics that western minds typically ascribe only to humans, e.g., having points of view, exhibiting agency, and engaging in reciprocal communication. From this perspective, a sense of place is derived from the traditional meanings that are attributed to certain areas, which can vary in scale—from specific spots in a landscape to an entire crest of a mountain (Gunderson and Watson 2007).

Native American input, including traditional knowledge and place identity information, can play an important role in land management decisions. Just after World War II, the ancestral lands of the Winnemem Wintu, a Native American tribe from Northern California, were submerged after the construction of the Shasta Dam, severely altering their tribal identity (Garrett 2010). According to Garrett, “The tribe’s remaining traditional cultural properties are under continual threat of loss and/or destruction, leaving the tribe’s ability to practice traditional ceremonies crippled by legal battles and fights...” (p. 346). In addition, (Dallman et al. 2013) argue that water policy in the United States has favored urban and agricultural development over Native Americans’ needs, and for the Winnemem Wintu specifically, they argue that culturally hegemonic meanings of natural resources and landscapes have privileged the water needs of modern development and have denied the importance of indigenous emotional connections to sacred places by limiting access to and protection of ancestral territories. Ninety percent of Winnemem ancestral lands along the McCloud River were flooded in 1945 when the Shasta Dam was completed for the Federal Central Valley Project. In 2000, the U.S. Bureau of Reclamation began investigating a proposal to raise Shasta Dam to increase surface water storage capacity for agricultural production. Dallman et al. (2013) argue that this proposal would destroy remaining Winnemem sacred spaces that offer deep emotional connections crucial to maintaining their cultural identity and ancestral memories.

Other tribes from the West have similar experiences. As Glowacka et al. (2009) document, the Hopi Tribe of Northern Arizona regarded the spraying of artificial snow made from treated sewage on Humphrey’s Peak to be a contamination of a sacred place. Humphrey’s Peak is the highest of a group of mountains called the San Francisco Peaks, which is the home of the katsinam, the Hopi’s ancestral deities. The area serves as a place to conduct ceremonies, participate in prayer, and gather the ceremonial objects that are incorporated into their religious practices.

Tribal members from the Klamath and Modoc Tribes of Southern Oregon and Northern California are concerned about the protection and preservation of rock cairns and prayer seats, which are connected to the traditional practice of vision quests. These ancient and sacred sites not only spiritually link living people with their ancestors but also are crucial for the long-term cultural survival of the tribes. As such, the Klamath and Modoc Tribes are committed to protecting these sites from the persistent threats of development, timber harvesting, and vandalism (Haynal 2000).

The preceding examples show how the concerns and perspectives of many Native American Tribes reflect their close attachments to place as well as the historical awareness that anchors and gives meaning to these attachments (Norgaard 2007). The challenges that come with the destruction of sacred places can be understood as challenges related to cultural self-determination, which is a persistent struggle faced by tribes throughout North America (Kingston 2015).

In working with tribes, it is necessary for land management agencies, to recognize “that Indigenous communities have different values, concerns and knowledge bases than non-Aboriginal communities” when it comes to certain land management decisions (Christianson 2015, p. 197). Part of this recognition involves transcending individual monetary valuations so that shared social values encompass social goods and cultural importance (Kenter et al. 2015).

For many Native Americans, the value of human well-being is closely associated with experiencing the natural environment (Bieling et al. 2014). However, it is not just the well-being of people experiencing the natural world today but also those who will be experiencing it in the future.

Interactions in Tribal Land Use and Government Land Management

- What are the traditional land uses of local tribes across the Lassen-Modoc?
- How have tribal land use practices changed?
- What are the interactions between tribal land use (i.e. gathering, hunting, spiritual) and land management planning?

Land management agencies, such as the U.S. Department of the Interior, National Park Service, and the U.S. Department of Agriculture, Forest Service, are wrestling with questions about how and under what conditions Native American perspectives could be incorporated into land management decisions (King 2007). Beatty and Leighton (2012) identify two coinciding trends that have ushered in an increased awareness of and receptiveness to Native American stewardship of forests and other public lands:

> The first is the growing trend within reservations across the United States toward self-determination, leading to forests and other resources managed not by the Bureau of Indian Affairs, but by the tribes themselves, in accordance with their values and objectives. The second is a growing recognition amongst the academic and management communities

> that management of all lands can benefit from seeking out Native perspectives, especially those founded in traditional knowledge systems (p. 565).

Nevertheless, decision making continues to proceed from what Hibbard et al. (2008) describe as rational, top-down approaches, which have marginalized indigenous communities by dismissing their cultural traditions as irrational while simultaneously imposing external values, policies, and actions upon native communities and landscapes. This approach toward land management arises from the traditions of western European philosophy, specifically the assumption that human beings are capable of removing themselves from and controlling the natural world (Pierotti and Wildcat 2000). Indeed, much of what accounts for biodiversity protection comes in the form of policy that prohibits humans from participating in consumptive and nonconsumptive activities through State-established protected areas (Hayes 2006). Well-intentioned laws, such as the National Environmental Policy Act of 1970, become inadvertent records of exclusion because of their exclusive reliance on scientific materialism to evaluate environmental impacts (Dongoske et al. 2015).

Native American ways of understanding the environment, also known broadly as Traditional Ecological Knowledge (TEK), offer an alternative approach. Based on detailed observations of nature connected with specific places, TEK considers all elements (humans, animals, plants, landforms) of a physical space to be constituent parts of a general community (Pierotti and Wildcat 2000). Dongoske et al. (2015) note “many Native American Tribes perceive the environment through an animistic ontological lens that embodies a sense of stewardship, manifest through a spiritual, umbilical connectedness to the natural world” (p. 36). In other words, TEK regards the success of conservation efforts depends on extensive community participation and control over land management decisions that ensue, in part, from networks of localized knowledge (Hayes 2006).

A recent survey (Beatty and Leighton 2012) of forest resource managers and decisionmakers from Native American Tribes showed that tribal managers do have an interest in collaborating with managers from government agencies, especially regarding the integration of TEK with western perspectives. Legislation, such as the 1992 amendments to the National Historic Preservation Act, encourage partnerships among tribal members and government researchers by authorizing tribes greater sovereignty over ancestral lands (Stanfill 1999). However, feelings of mistrust and resentment within tribal communities still linger (Flood and McAvoy 2007). Flood and McAvoy note that it is important for Forest Service managers to treat tribal members with mutual respect before attempting collaboration and that interpersonal dialogs possess great potential in establishing trust among all stakeholders.

Integrating and applying TEK with western science mandates “enduring commitments to knowledge-sharing that extend beyond the usual boundaries of professional training and cultural orientation such that learning can proceed, legacy myths might be corrected, and the forests and the people will benefit” (Mason et al. 2012, p. 187). McOliver et al. (2015) advocate for knowledge-sharing not just between Forest Service managers and tribal members but also among tribes. They argue that when tribes participate in knowledge-sharing, it encourages native communities to sanction their own research projects and establish communal networks of shared knowledge.

The Native American heritage of communal interdependency helped explain why most outside economic programs that were historically imposed upon Native communities usually ended up failing. Thus, it is essential that economic development either be directed by tribal members themselves or in full partnership with outside groups (Kingston 2015). The central challenge, then, becomes how to “grow indigenous economies in ways that increase independence of native communities and overcome the dependence created by colonialism” (Harris et al. 2011, p. 287). One way is to legitimate subsistence economies as viable economic models.

The sharing that occurs in subsistence economies establishes and maintains cohesive bonds among tribal members, whereas market-based commodity exchanges transpire between independent parties whose interactions are based solely on exchanging one item for another (Dick 1996). Dick (1996) goes on to assert that when communal subsistence practices become disrupted through the imposition of market-based forces, the cohesive bonds that tie tribal members to one another may fracture or dissipate entirely.

Just as important as communal bonds are to maintaining the integrity of a subsistence economy, so too are the ecological bonds that tie people to the land (fig. 5.5.2). The protraction of subsistence economies is contingent

Figure 5.5.2—Collecting, sharing, or bartering nontraditional forest products by all Americans helps preserve communal bonds, many of which are also central to Native American communities and reinforces the connections of indigenous cultures to the land (photo by Ken Sandusky, Forest Service).

upon the responsible use and extraction of environmental resources so that the biological integrity of an ecosystem becomes capable of sustaining itself in perpetuity (Burger 2008). Such was the case for many tribes throughout the West who utilized a harvesting technique to collect the inner bark of various pine species, which served as an important nutrient source for many native people (Ostlund et al. 2005). The inner bark was harvested in a sustainable manner, in that trees were not killed during the process, and, consequently, the overall ecological impact on the forest was negligible (Ostlund et al. 2005).

Before European contact, the Klamath and Modoc Tribes of Southern Oregon and Northeastern California took an active role in the management of various plant communities. Deur (2009) identifies multiple and complex plant management strategies utilized by the Klamath and Modoc people that are consistent with modern definitions of plant cultivation. Some of these practices included "the management of black huckleberry (*Vaccinium membranaceum*) yields in subalpine environments, the management of marsh-edge environments for yellow pond lily (*Nuphar polysepalum*), the tending of "epos" or yampah (*Perideridia* species) digging sites, and the selective harvest of tree cambium, sap, and wood—especially from pines (*Pinus* species) and junipers (*Juniperus* species)" (Deur 2009, p. 296). Practices of this nature suggest a widespread application of plant management that functioned to geographically coalesce preferred species.

The Pit River Tribe (Ajumawi band) of Northeastern California also participated in sustainable harvesting practices. During the course of generations, families returned to the same wild mushroom plots to help them clearly distinguish between edible and toxic varieties (Buckskin and Benson 2005). Fungi play a significant role in maintaining the ecological health and sustainability of forests (Trappe et al. 2009), and some of the harvesting practices used by the Pit River Tribe (e.g., not disturbing the mycelium, giving small mushrooms the chance to grow larger, leaving old mushrooms to spread spores) are already regarded as beneficial in modern management contexts, suggesting that a basis for common ground already exists between land managers and tribal members (Anderson and Lake 2013).

Ultimately, collaboration between State and Federal agencies "can not merely encourage Native Americans to respond to agency inquiries, but also ensure that agency procedures for planning and decision making are responsive and sensitive to the special needs and concerns of Native Americans" (Stanfill 1999, p. 70). The most important question in the land management debate may not concern the ownership of public lands or even who does what, but instead asks how land management agencies can create policy that translates into success for both the agency and tribes (King 2007).

Tribal Use of Fire Management

- How have changes in climate, invasive species, and wildlife habitat impacted tribal land-use culturally, socially, spiritually, and economically?

For many generations, Native Americans dealt with the impacts brought upon their communities and lands by Western settlement and expansion. The land use practices that accompanied settlement, such as mining, cattle grazing, and timber extraction, had far-reaching consequences on many landscapes throughout the West (Fry and Stephens 2006). For example, according to Sneider (2012), relationships with the Paiute Nation became key to Western movement into the area as their lands stood directly in the path of settlers and miners moving toward California through the Sierra Nevada. Sneider (2012) argues that Paiutes were subject to various methods of removal and attempts at assimilating or civilizing the Indian, then became wards of the State through the Indian Appropriation Act of 1871. Today, native tribes can grapple not only with the environmental repercussions of Western settlement but can also deal with the effects levied upon their lands by global climate change.

Christianson (2015) predicts that Native Americans may experience greater overall impacts from climate change than the general population. In fact, many indigenous agricultural practices are being adjusted at the local level, as climate change impacts the ability to observe environmental indicators that had been reliable until recently (Peppler 2017).

Impacts "such as increased frequency and intensity of wildfires, higher temperatures, extreme changes to ecosystem processes, forest conversion and habitat degradation are threatening tribal access to... the quantity and quality of resources tribes depend upon to perpetuate their cultures and livelihoods" (Voggesser et al. 2013, p. 615). The ecological balance that has sustained North American temperate and boreal coniferous forests through indigenous burning practices becomes compromised when invasive species grow and flourish in areas they have not previously (Christianson 2015). Traditional burning practices rely on predictable environmental cues (Huffman 2013) that are increasingly being disrupted because of climate change and species invasion (Voggesser et al. 2013).

The significance of ecological and economic damage caused by invasive species is widely recognized and brings to the fore political issues regarding which species could be managed and which populations are impacted by management decisions. For example, in response to a spotted knapweed (*Centaurea stoebe*) invasion in a rural region of Northern California, members of the Karuk Tribe, as well as those outside of the tribal community, agreed that the invasive species was undesirable; however, tribal members were mostly apprehensive of potential herbicide use (Norgaard 2007).

Significant changes to species composition in forests could deprive tribal communities of culturally important resources and negatively impact historical means of subsistence (Voggesser et al. 2013). Voggesser et al. (2013) offer a solution that is grounded in collaboration:

> "To address these challenges, robust federal-tribal relationships are needed, particularly when changes affect treaty rights, tribal lands, and resources held in trust. Collaboration, knowledge-sharing, and joint action by tribes and nontribal stakeholders can lead to more effective and sustainable planning efforts around climate change and invasive species" (p. 622).

Collaboration of this nature requires involving tribes in meticulous and conscientious decisionmaking processes that are open to a variety of knowledge forms, such as TEK (Berkes 2009). Incorporating definitions of health as defined by local tribes into the Federal and State regulations that prescribe land management policy, such as the National Environmental Policy Act of 1970 and the National Forest Management Act of 1970, would give credence to traditional subsistence practices (McOliver et al. 2015). Prescribed burning practices that are based on TEK have the potential to lessen the destruction of forests caused by severe wildfires, which, because of climate change, are increasing in frequency (Stan et al. 2014). Maintaining the productivity of land-based activities in the face of climate change will likely remain a challenge for land managers, which is why incorporating indigenous burning practices that mitigate the severity of wildfires could benefit both policy and land (Hertel 2017). Obviously, building strong relations with indigenous communities is a process that takes time (Christianson 2015), but it is a process that benefits all parties involved.

References

Anderson, M.K.; Lake, F.K. 2013. California Indian ethnomycology and associated forest management. Journal of Ethnobiology. 33(1): 33–85.

Beatty, C.; Leighton, A. 2012. Intertribal Timber Council survey of tribal research needs. Forestry Chronicle. 88(5): 565–570.

Berkes, F. 2009. Indigenous ways of knowing and the study of environmental change. Journal of the Royal Society of New Zealand. 39(4): 151–156.

Bieling, C.; Plieninger, T.; Pirker, H.; [et al.]. 2014. Linkages between landscapes and human well-being: an empirical exploration with short interviews. Ecological Economics. 105: 19–30.

Buckskin, F.; Benson, A. 2005. The contemporary use of psychoactive mushrooms in Northern California. Journal of California and Great Basin Anthropology. 25(1): 87–92.

Burger, J. 2008. Environmental management: Integrating ecological evaluation, remediation, restoration, natural resource damage assessment and long-term stewardship on contaminated lands. Science of the Total Environment. 400(1-3): 6–19.

Burger, J.; Gochfeld, M. 2010. Gender differences in resource use and evaluation of attributes of places of resource use by Native Americans and Caucasians from Western Idaho: relevance to risk evaluations. Journal of Toxicology and Environmental Health A. 73(24): 1655–1664.

Christianson, A. 2015. Social science research on Indigenous wildfire management in the 21st century and future research needs. International Journal of Wildland Fire. 24(2): 190-200.

Cruikshank, J. 2012. Are glaciers 'good to think with'? Recognising indigenous environmental knowledge. Anthropological Forum. 22(3): 239–250.

Curran, D.; M'Gonigle, M. 1999. Aboriginal forestry: community management as opportunity, and imperative. Osgoode Hall Law Journal. 37(4): 711–774.

Dallman, S.; Ngo, M.; Laris, P.; [et al.]. 2013. Political ecology of emotion and sacred space: the Winnemam Wintu struggles with California water policy. Emotion, Space, and Society. 6: 33–43.

Deloria, V. 2006. The world we used to live in: remembering the powers of the medicine men. Golden, CO: Fulcrum. 272 p.

Deur, D. 2009. A caretaker responsibility: revisiting Klamath and Modoc traditions of plant community management. Journal of Ethnobiology. 29(2): 296–322.

Dick, R.E. 1996. Subsistence economics: freedom from the marketplace. Society & Natural Resources. 9(1): 19–29.

Diver, S. 2016. Co-management as a catalyst: pathways to post-colonial forestry in the Klamath Basin, California. Human Ecology. 44(5): 533–546.

Dongoske, K.E.; Pasqual, T.; King, T.F. 2015. The National Environmental Policy Act (NEPA) and the silencing of Native American worldviews. Environmental Practice. 17(1): 36–45.

Ermine, W. 2007. The ethical space of engagement. Indigenous Law Review. 6(1): 193–203.

Flood, J.P.; McAvoy, L.H. 2007. Voices of my ancestors, their bones talk to me: How to balance U.S. Forest Service rules and regulations with traditional values and culture of American Indians. Human Ecology Review. 14(1): 76–89.

Fry, D.L.; Stephens, S.L. 2006. Influence of humans and climate on the fire history of a ponderosa pine-mixed conifer forest in the southeastern Klamath Mountains, California. Forest Ecology and Management. 223(1-3): 428–438.

Garrett, B.L. 2010. Drowned memories: the submerged places of the Winnemem Wintu. Archaeologies: Journal of the World Archaeological Congress. 6(2): 346–371.

Glowacka, M.; Washburn, D.; Richland, J. 2009. Nuvatukya'ovi, San Francisco peaks: balancing western economies with Native American spiritualities. Current Anthropology. 50(4): 545–559.

Greenberg, M.; Crossney, K. 2006. The changing face of public concern about pollution in the United States: a case study of New Jersey. The Environmentalist. 26(4): 255–268.

Grenier, L. 1998. Working with indigenous knowledge: a guide for researchers. Ottawa: International Development Research Centre. 100 p.

Gunderson, K.; Watson, A. 2007. Understanding place meanings on the Bitterroot National Forest, Montana. Society & Natural Resources 20(8): 705–721.

Harris, L.; Sachs, S.M.; Morris, B., eds. 2011. Re-creating the circle: the renewal of American Indian self-determination. Albuquerque, NM: University of New Mexico Press. 528 p.

Hayes, T.M. 2006. Parks, people, and forest protection: an institutional assessment of the effectiveness of protected areas. World Development. 34(12): 2064–2075.

Haynal, P.M. 2000. The influence of sacred rock cairns and prayer seats on modern Klamath and Modoc religion and world view. Journal of California and Great Basin Anthropology. 22(2): 170–185.

Hertel, T.W. 2017. Land use in the 21st century: contributing to the global public good. Review of Development Economics. 21(2): 213–236.

Hibbard, M.; Lane, M.B.; Rasmussen, K. 2008. The split personality of planning indigenous peoples and planning for land and resource management. Journal of Planning Literature. 23(2): 136–151.

Huffman, M.R. 2013. The many elements of traditional fire knowledge: synthesis, classification, and aids to cross-cultural problem solving in fire-dependent systems around the world. Ecology and Society. 18(4): 3.

Huntsinger, L.; Diekmann, L. 2010. The virtual reservation: land distribution, natural resource access, and equity on the Yurok Forest. Natural Resources Journal. 50(2): 341–369.

Kenter, J.O.; O'Brien, L.; Hockley, N.; [et al.]. 2015. What are shared and social values of ecosystems? Ecological Economics. 111: 86–99.

King, M.A. 2007. Co-management or contracting? Agreements between Native American Tribes and the U.S. National Park Service pursuant to the 1994 Tribal Self-Governance Act. Harvard Environmental Law Review. 31(2): 475–530.

Kingston, L. 2015. The destruction of identity: cultural genocide and indigenous peoples. Journal of Human Rights. 14(1): 63–83.

Lauer, M.; Matera, J. 2016. Who detects ecological change after catastrophic events? Indigenous knowledge, social networks, and situated practices. Human Ecology. 44(1): 33–46.

Long, J.; Lake, F.K.; Lynn, K.; [et al.]. 2018. Tribal ecocultural resources and engagement. In: Spies, T.A.; Stine, P.A.; Gravenmier, R.A.; [et al.], tech coords. 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: 851–917.

Mason, L.; White, G.; Morishima, G.; [et al.]. 2012. Listening and learning from traditional knowledge and western science: a dialogue on contemporary challenges of forest health and wildfire. Journal of Forestry. 110(4): 187–193.

McOliver, C.A.; Camper, A.K.; Doyle, J.T.; [et al.]. 2015. Community-based research as a mechanism to reduce environmental health disparities in American Indian and Alaska Native communities. International Journal of Environmental Research and Public Health. 12(4): 4076–4100.

Norgaard, K.M. 2007. The politics of invasive weed management: gender, race, and risk perception in rural California. Rural Sociology. 72(3): 450–477.

Ostlund, L.; Keane, B.; Arno, S.; Anderson, R. 2005. Culturally scarred trees in the Bob Marshall Wilderness, Montana, USA—Interpreting Native American historical forest use in a wilderness area. Natural Areas Journal. 25(4): 315–325.

Peppler, R.A. 2017. It's not balancing out like it could be: perceptions of local climate variability in native Oklahoma. Weather Climate and Society. 9(2): 317–329.

Pierotti, R.; Wildcat, D. 2000. Traditional ecological knowledge: the third alternative (commentary). Ecological Applications. 10(5): 1333–1340.

Pleasant, M.M.; Gray, S.; Lepczyk, C.; [et al.]. 2014. Managing cultural ecosystem services. Ecosystem Services. 8: 141–147.

Stan, A.B.; Fulé, P.Z.; Ireland, K.B.; [et al.]. 2014. Modern fire regime resembles historical fire regime in a ponderosa pine forest on Native American lands. International Journal of Wildland Fire. 23(5): 686–697.

Stanfill, A.L. 1999. Native American participation in federal programs under the National Historic Preservation Act. Plains Anthropologist. 44(170): 65–70.

Stephens, S.L.; Martin, R.E.; Clinton, N.E. 2007. Prehistoric fire area and emissions from California's forests, woodlands, shrublands, and grasslands. Forest Ecology and Management. 251(3): 205–216.

Stidham, R.; Carp, R.A. 1995. Indian rights and law before the federal district courts. Social Science Journal. 32(1): 87–100.

Trappe, J.M.; Molina, R.; Luoma, D.L. 2009. Diversity, ecology, and conservation of truffle fungi in forests of the Pacific Northwest. Gen. Tech. Rep. PNW-GTR-772. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 194 p.

Voggesser, G.; Lynn, K.; Daigle, J.; [et al.]. 2013. Cultural impacts to tribes from climate change influences on forests. In: Maldonado, J.K.; Colombi, B.; Pandya, R., eds. Climate Change and indigenous peoples in the United States. New York: Springer Cham: 107–18.

Winthrop, R.H. 2014. The strange case of cultural services: limits of the ecosystem services paradigm. Ecological Economics. 108: 208–214.

Wyatt, S.; Kessels, M.; Learhoven, F.V. 2015. Indigenous peoples' expectations for forestry in New Brunswick: Are rights enough? Society & Natural Resources. 28: 625–640.

Section 6. Responding to Disturbances

Chapter 6.1. Ecological Disturbance in the Context of a Changing Climate: Implications for Land Management in Northeastern California

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Introduction

Ecosystems of the Lassen and Modoc National Forests depend on disturbance as part of the natural process. However, climate change, a source of disturbance itself, has also been changing patterns of other ecological disturbances, including the frequencies and intensities of fire, pests, and pathogens. This chapter presents a review and synthesis of peer-reviewed literature focused on natural disturbance processes and how those processes relate to and interact with a changing climate. The chapter begins with an overview of different ways climate change can impact the ecosystems of these two national forests in relation to current versus past evidence of climate change, fire frequencies and intensities, changes in patterns of snowfall and snow melt, tree and plant distributions, and insects and pests. Impacts on animal species, particularly species of concern are discussed, along with effects on aquatic systems. The chapter concludes with a discussion of impacts of climate change on seeding projects.

As discussed in Chapter 1.1 (Dumroese, this synthesis, *The Northeastern California Plateaus Bioregion Science Synthesis: Background, Rationale, and Scope*), two other science syntheses have relevance for the Lassen and Modoc National Forests (hereafter the Lassen and the Modoc), namely the *Science Synthesis to Support Socioecological Resilience in the Sierra Nevada and Southern Cascade Range* (hereafter, Sierra Nevada Science Synthesis) (Long et al. 2014a) and *Synthesis of Science to Inform Land Management within the Northwest Forest Plan Area* (hereafter, Northwest Forest Plan Science Synthesis) (Spies et al. 2018). Both syntheses have extensive discussions pertaining to changes in climate (table 6.1.1). Thus, for this chapter, the discussion is primarily on how changes in climate affect other disturbances, such as fire, pests, and restoration for the Lassen and Modoc ecosystems not covered extensively by the other two syntheses.

Overview

Climate change on the Lassen and Modoc is anticipated to impact a number of resources on the forests and their associated plants and animals. First, the discussion focuses on recent studies and science syntheses that include climate models that present potential climate alternative futures for the Lassen and Modoc. Second, these models are put into an historical context, examining the evidence showing how the climate in the study area has changed during the last several thousand years.

Cayan et al. (2008) modeled a set of future climate alternatives for California based on International Panel on Climate Change (IPCC) emissions scenarios. For Northern California, all models show rising temperatures with precipitation expected to remain steady or decrease slightly and continue to follow a Mediterranean pattern, with most precipitation falling in the winter months. Given warmer temperatures, less precipitation is expected to fall as snow. However, Allen and Luptowitz (2017), using a set of newer models, show that precipitation in California may, in fact, increase under climate change because of higher ocean temperatures and shifting precipitation patterns.

In this chapter, discussion focuses mainly on climate change impacts to the parts of the Lassen and Modoc considered to be part of the Great Basin. As stated earlier, other areas of these Forests have already been addressed in the Sierra Nevada Science Synthesis and the Northwest Forest Plan Science Synthesis. The Sierra Nevada Science Synthesis notes that the local climate has already changed during the last 80 years, with warmer temperatures and more frequent drought. With warmer temperature comes less precipitation as snow, and spring thaws that occur earlier, which, in turn, extends the fire season. Many studies also suggest changes in vegetation across the Sierra

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Table 6.1.1—Summary of climate change topics contained within the Sierra Nevada Science Synthesis and the Northwest Forest Plan Science Synthesis.

Sierra Nevada Science Synthesis	
Chapter	Topic
1.4	Forest management issues and research needs
3.1	Seed transfer implications
6.1	Potential impacts on stream flow and temperature; changes in precipitation from snow to rain
7.1	Potential impacts on marten (*Martes caurina*) and fisher (*Martes pennant*)
7.2	Potential impacts on California spotted owl (*Strix occidentalis occidentalis*)
8.1	Interactions with air pollution
9.3	Public perceptions of climate change
9.4	Rural economy resilience
Northwest Forest Plan Science Synthesis	
2	Potential impacts on vegetation, modeling effects, potential management responses, and research needs
3	Impacts on old-growth forests
4	Anticipated effects on northern spotted owl (*Strix occidentalis caurina*) habitat
5	Anticipated effects on terrestrial nesting habitats and marine food resources critical to the marbled murrelet (*Brachyramphus marmoratus*)
6	Effects on old-growth forest biodiversity with discussion on amphibian communities and connectivity for carnivores
7	Potential impacts on aquatic systems with specific attention to Salmonids
8	Socioeconomic well-being
9	Changing public values, for example, as it pertains to recreation
10	Environmental justice issues
11	Changes to tribal ecocultural resources and engagement
12	Climate change uncertainties and research needs

Nevada, with an increase in oaks and other broadleaved trees. Subsequently, in Chapter 1.4 of the Sierra Nevada Science Synthesis, Jardine and Long (2014) present a set of management issues and research needs, all of which are relevant to the Lassen and Modoc:

- **"Recognize and address scale mismatches.**
- **Consider long-term (more than 50 years) risks** in addition to short-term (less than 10 years) expected outcomes.
- **Set adaptable objectives and revisit them,** because there may be a lack of clear solutions, certain options may prove unrealistic, and new opportunities may become apparent as conditions change.
- **Rely more on process-based indicators than static indicators of structure and composition, while recognizing that restoration of structure and process must be integrated.**
- **Integrate valuation tools, decisionmaking tools, modeling, monitoring, and, where appropriate, research** to evaluate responses and better account for the risks and tradeoffs involved in management strategies.
- **Consider the integrated nature of socioecological systems;** approaches that address only one dimension of a problem are less likely to succeed in the long run than strategies that consider ecological, social, economic, and cultural components.
- **Use participatory and collaborative approaches to facilitate adaptive responses and social learning."**

Evidence for Past Climate Change on the Lassen and Modoc

The climate in the Great Basin has changed dramatically since the Last Glacial Maximum and the beginning of the

Holocene approximately 20,000 years ago. Data from lake sediment cores, tree ring data, and packrat and woodrat (*Neotoma* species) middens have been used to develop an understanding of past climate. Each type of data has its strengths and limitations, but together they paint a clear picture that the climate has not remained stable in the Great Basin during the last 20,000 years; periods of drought, warming, and cooling have all been common. These data show changes spanning thousands of years, which is very different from the rapid changes in climate that are being observed today.

Lake sediment contains minerals, pollen, plant debris, and diatoms that can be dated using radioisotopes. These data can help indicate how hot or dry an area was in a given period of time. Benson et al. (2002) found that the Great Basin was cooler in the early Holocene (11,600 to 8,000 years ago), followed by a warmer, and particularly dry, middle Holocene (8,000 to 3,000 years ago). Using pollen cores collected from lakes across the Great Basin, Mensing et al. (2008, 2013) found evidence for an extended drought from 2,800 to 1,850 years ago and a number of drought events each lasting 50 to 100 years since then. Despite this, the forest composition has remained relatively stable for the past 4,300 years near Paterson Lake in Northern California and the vegetation surrounding Lily Lake on the California-Oregon boundary has not changed for more than 10,000 years (Minckley et al. 2007). Comparing sediment cores from lakes across the Great Basin, Wahl et al. (2015) note that the Western and Eastern Great Basin have experienced different climate histories, in part due to the influence of the Pacific Ocean, particularly during the last 2,000 years (fig. 6.1.1).

Tree ring data from long-lived species provide another line of evidence for changing climates in the Great Basin. Salzer et al. (2014) documented changes in temperature during the last 5,000 years based on tree ring data from

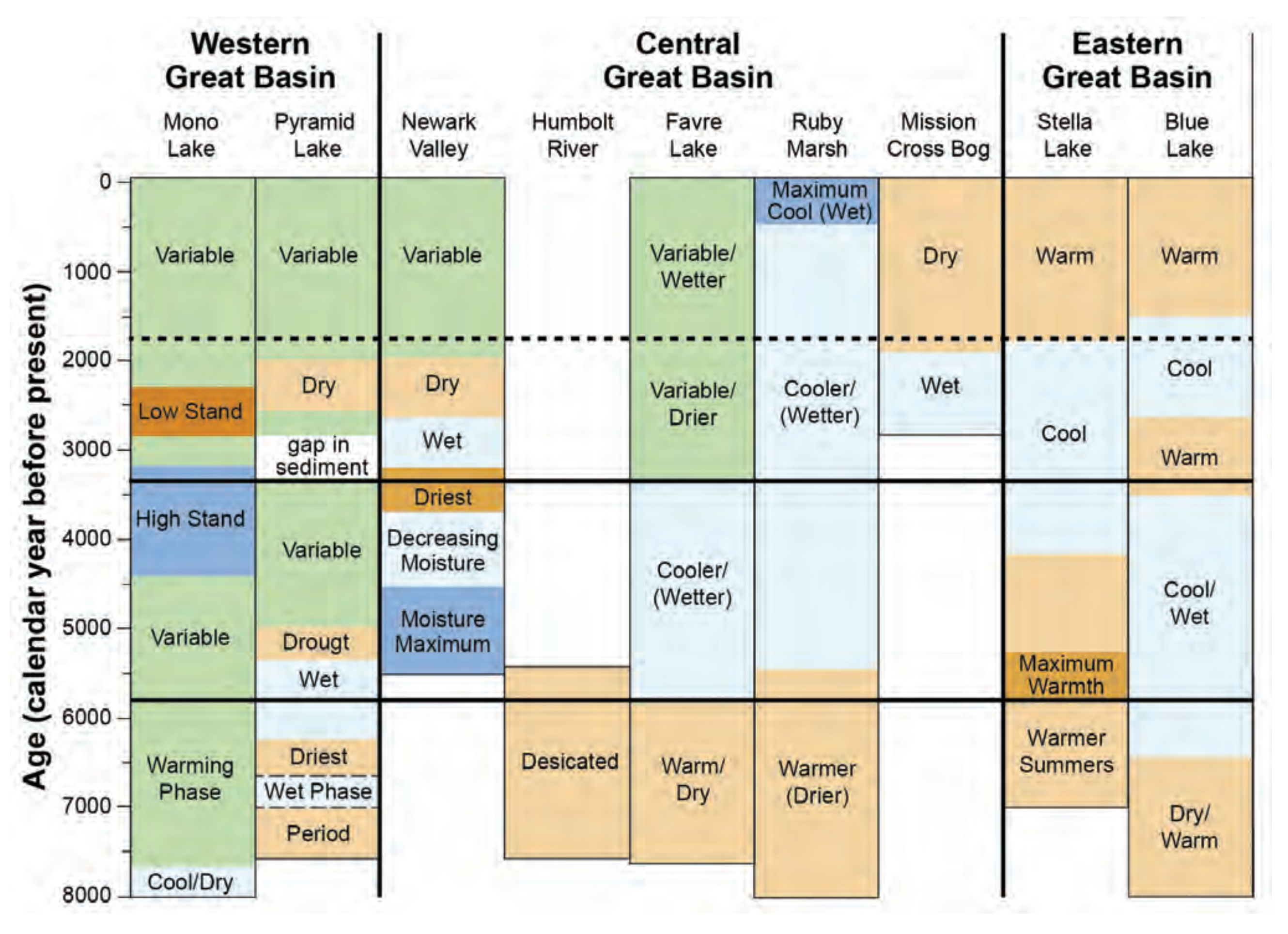

Figure 6.1.1—Sediment core data reveals that climate across the Great Basin has varied spatially and temporally during the past 8,000 years (modified from figure 10 in Wahl et al. 2015).

Great Basin bristlecone pine (*Pinus longaeva*) trees in Nevada. Ring data show that the tree line reached a maximum elevation approximately 5,000 years ago, with the current tree line established in the 1300s. Tree ring data show that since 1900, temperatures have dramatically increased, and are hotter than at any time since 850 CE.

Rodents, such as packrats and woodrats, often make large garbage piles, or middens. These locations can be occupied by the rodents for tens of thousands of years, and the middens become records of the plant community in an area during the period of occupation. Becklin et al. (2014) point out that atmospheric carbon dioxide concentrations were lowest (180 ppm) during the Last Glacial Maximum (21,000 years ago) and are now above 400 ppm. Concurrent with this increase in carbon dioxide concentration, packrat middens show, as evidenced through carbon isotopes of plant material, that the plant community has changed dramatically. In the Reno, NV, area, woodrat middens at different elevation sites showed two very different tree responses to a changing climate: Utah juniper trees (*Juniperus osteosperma*) tended to persist in the same locations and were therefore able to tolerate a range of climates, while singleleaf piñon (*Pinus monophylla*) has changed its distribution over time, disappearing in locations with unsuitable climate (Nowak et al. 1994).

Animal populations also changed throughout the Holocene. Caves where owls have roosted for years are an excellent source of information about these communities. Owl pellets contain small-mammal bones, which can often be identified to species and dated to get a picture of the fauna in a given area at a given time. Grayson (2000) reviewed mammal diversity during the Middle Holocene (8,000 to 5,000 years ago), a period warmer and drier than the time before and after it. Mammal diversity decreased during this period, with kangaroo rats (*Dipodomys* species) becoming much more common as conditions became more arid. Genetic data suggest that kangaroo rats have remained within their current geographic distribution since the Last Glacial Maximum despite changes in climate across the Great Basin, suggesting that they adjusted their niche to survive in whatever the local climate was (Jezkova et al. 2011).

Human settlements were found throughout the Great Basin during the Holocene and have received much research attention. Excavations from a human settlement in Long Valley, NV, as well as other records, showed that the end of the Pleistocene was in a warming trend associated with abundant moisture, and several very large lakes occurred in the Great Basin. This was followed by a cooling trend and then a drying trend until 4,000 years ago, when moisture levels increased once again (Huckleberry et al. 2001). The Younger Dryas period (12,900 to 11,600 years ago) was characterized by cooler temperatures and more available water (Goebel et al. 2011). Many human settlements existed in the Great Basin at that time, though some settlements appear to be more temporary, suggesting a more mobile lifestyle. Vegetation was dominated by sagebrush and grasses. Greater sage-grouse (*Centrocercus urophasianus*) was a common item in the human diet, as were grasshoppers (*Caelifera*), pronghorn (*Antilocapra americana*), mule deer (*Odocoileus hemionus*), and bighorn sheep (*Ovis canadensis*) (Goebel et al. 2011). Jones and Schwitalla (2008) examined the Medieval Climate Anomaly that led to three strong periods of drought in California between 1000 and 1375 CE and found evidence from across California for widespread movement in human populations during that time and that diets changed and health problems increased.

Together these various sources of data about historical climate provide a consistent picture of climate transitions across the Great Basin since the Last Glacial Maxima. They also put the current changes in climate being observed into a historical perspective. The historical climate of the Lassen and Modoc was also reviewed by Merriam et al. (2013) as part of the Forest Revision Planning process. Reviewing weather station and PRISM data, they discuss the historic range of variation for several important climate variables. Their analysis showed that temperatures have increased by 1.7 to 2 °F (0.9 to 1.1 °C) and precipitation has remained steady since 1895.

Projected Climate Change Effects on the Lassen and Modoc

Projecting forward, one recent study shows some interesting insight. Wintertime temperature and precipitation, using oxygen isotopes found in cave stalagmites in the Great Basin, reflect levels of arctic ice for the last 160,000 years. If levels of arctic ice decline as predicted as a result of a changing climate, then more warming and drying trends in the Great Basin can also be anticipated (Lachniet et al. 2017).

Snowpack Accumulation and Melt

Climate change is expected to have a dramatic impact on snowpack depth and subsequent melting in the Lassen and Modoc: Warmer temperatures are expected to cause more precipitation to fall as rain rather than snow resulting in reduced snowpack, and warmer temperatures occurring earlier in spring are expected to melt that limited snowpack faster (fig. 6.1.2). Indeed, this trend is being observed in weather station data (reviewed in Merriam et al. 2013). This may have implications for the plants associated with the Lassen and Modoc. A number of studies, reviewed below, have looked at how reduced snowpack influences growth and ecophysiological traits (traits of adaptation of an organism's physiology to environmental conditions) of woody sagebrush steppe species.

Various studies have manipulated how much moisture is available for plants by changing snow depth, temperature, and rain fall capture. These manipulation studies have been useful for understanding the response of ecophysiological traits to changes in water availability. There were no differences in stem water relations and photosynthetic gas exchange in big sagebrush (*Artemisia tridentata*) and bitterbrush (*Purshia tridentata*) among snow depth treatments within a year (Loik et al. 2015) but much difference associated with year-to-year precipitation (Loik et al. 2015). These results may be different at higher elevations. In a long-term heating experiment on an alpine meadow in Colorado, Perfors et al. (2003) found that big sagebrush had increased growth rates in heated plots associated with an earlier snow melt date but Roy et al. (2004) note an increase in herbivory of big sagebrush associated with the earlier snow melt date. In a study in Oregon, Bates et al. (2006) manipulated patterns of seasonal precipitation in a natural sagebrush ecosystem. They changed when plants received the most water—winter, spring, or fixed (receiving the same as the average rainfall in the last 50 years)—and then measured the response in the plant community. Big sagebrush did not show changes in abundance across precipitation treatments, but when most of the rain fell in the spring, plants had higher reproductive success. Further, Gillespie and Loik (2004) looked at pulses of precipitation by simulating a summer rainstorm (predicted to be more common under future climate) and suggested that seedlings of big sagebrush, because of an ability to maintain higher photosynthetic rates through better use of pulse events of precipitation, may cope with climate change better than seedlings of bitterbrush. Enhanced photosynthesis would allow the sagebrush to allocate more carbohydrates to root production and thereby increase water and nutrient acquisition. Together, these results suggest changes in precipitation patterns under climate change will influence the fitness of sagebrush plants.

A strong relationship exists among habitat type, climate change, and snowfall. Bradford et al. (2014), with ecohydrological modeling, found that warming temperatures changed how much precipitation fell as snow in a sagebrush steppe and adjoining lodgepole pine (*Pinus contorta*) ecosystems in Colorado, and as a result,

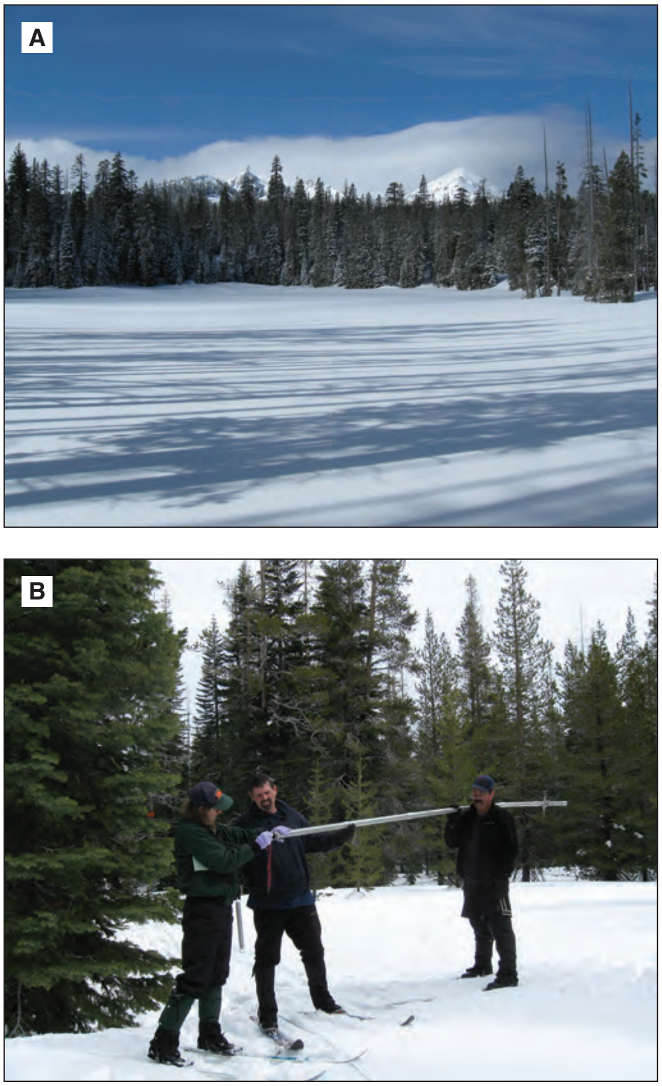

Figure 6.1.2—The potential changes to snowpack depth and its seasonal longevity (A), monitored on the Lassen and Modoc (B), expected to occur because of changing climate may have profound effects on water availability and plant distribution (photo A by Debbie Mayer, Forest Service; photo B by Forest Service).

the availability of that water to plants changed as well. There were differences among the ecosystems with climate change scenario, impacting drainage and transpiration. Under the IPCC's A2 "business as usual" future climate scenario, the ecohydrology of the lodgepole pine system more strongly resembled that of the sagebrush steppe, suggesting potential vegetation transitions. Kormos et al. (2017), looking at juniper encroachment just east of the Idaho and Oregon border, examined the relationship between the amount of snow accumulating in sagebrush habitat versus juniper habitat. Their models showed more snow accumulation in the juniper woodlands compared to open sagebrush habitat but that the additional snow melted earlier. Thus, juniper encroachment into sagebrush habitat changes the water balance and overall ecohydrology of the system, and reduces foraging and nesting resources available for greater sage-grouse (Connelly et al. 2011; Miller et al. 2011, 2017). Similar results were found by Roundy et al. (2014), with more water available for plant growth when trees were removed from sagebrush habitat.

Insect and Disease

Bark Beetles

Although damage in 2016 from drought and bark beetles on the Lassen and Modoc was extensive, given the lower amount of forested land area, these patterns of tree mortality were, in general, less when compared to Southern California. The U.S. Department of Agriculture, Forest Service, published a bark beetle forecast map for California (https://usfs.maps.arcgis.com/apps/MapJournal/index.html?appid=7b78c5c7a67748808ce298efefceaa46) that, for 2017, predicted tree mortality was not expected to increase on the Lassen and Modoc. The forecast also shows declines in the southern part of the State, where much more extensive damage has occurred.

Modeled projections by Bentz et al. (2010) for the Western United States suggested little change in regards to the probability of a bark beetle outbreak on the Lassen and Modoc, even under warming temperatures for the next century. The interaction of climate, insects, and tree mortality is, however, complex. Creeden et al. (2014) note that for several forests suffering bark beetle outbreaks, the climate and weather differed, which affected the reproduction and winter survival of the beetles and the drought stress of host trees. Thus, Anderegg et al. (2015) argue for a multifaceted approach to account for different responses by the insects and trees to climate drivers.

Forest management and treatments (i.e., thinning and controlled burning) can affect bark beetle populations. On the west slope of the Warner Mountains of the Modoc, Egan et al. (2010) found less tree mortality caused by fir engraver beetles (*Scolytus ventralis*) and mountain pine beetles (*Dendroctonus ponderosae*) in pre-commercially thinned plots compared to nonthinned plots. On the Klamath National Forest, while results were not significant, a trend for less mortality in thinned plots was also observed (Fettig et al. 2010). Multiple studies have, however, observed an increase in insect-related tree mortality in plots that had been burned, either with or without thinning (Fettig et al. 2010; Fettig and McKelvey 2010). For example, Fettig and McKelvey (2010) saw a dramatic increase in beetle mortality in burned plots on the Lassen, where trees in burned plots had higher mortality due to beetle attack immediately after burning. Adjacent, unburned plots had higher levels of attack 3 to 5 years after treatment, likely because beetles moved off the burned areas to the neighboring, unburned, plots. These results led Fettig and McKelvey (2010, p. 37) to note, however, that "unburned areas (split plots) were adjacent to burned areas (split plots) that likely served as an important source of beetles once the abundance of fire-injured trees that were highly susceptible to bark beetle attack declined."

While shifts in climate are expected to allow pine beetles to move uphill, and inhabit novel territory, studies in Great Basin of high-elevation bristlecone pine have shown that this species is not often chosen for oviposit by mountain pine beetles, but when eggs are laid, the resulting larvae do not develop, suggesting this pine may have resistance to the insect (Eidson et al. 2017, 2018).

Aroga Moths

Aroga moth (*Aroga websteri*), also known as the sagebrush defoliating moth, can kill sagebrush plants after a single season of defoliation. They occur in outbreaks (irruptions) throughout the Great Basin, and how the frequency of these irruptions will change under climate change is uncertain. In Utah, Bolshakova and Evans (2014) found that sagebrush growing on north-facing slopes with lower incident solar radiation made good habitat for the moths, conditions not expected to be altered under climate change. The suitability of habitat for the moths might change, however, as temperatures associated with different slopes, aspects, and elevation change. Indeed, further study showed that years with high precipitation in June and July—corresponding to late-stage larval development—

was associated with population irruptions (Bolshakova and Evans 2016).

Nursery Pathogens

Nurseries can, because they are monocultures of plants, provide ideal environments for pathogens to develop. Diseases can be spread from nurseries to natural areas. In the United States, diseases can be introduced from outside our borders (Liebhold et al. 2012), or be common, widespread, indigenous diseases found naturally in our Nation's forests and spread by windborne spores (e.g., fusiform rust in the South and Diplodia blight in the Midwest), or be ubiquitous, common nursery diseases not found in natural areas. For some diseases, infected seedlings subsequently have reduced survival and growth when outplanted on restoration sites in natural areas (Stanosz and Carlson 1996; Palmer et al. 1988; Powers et al. 1981). For other diseases, infected nursery stock that otherwise meets quality standards for outplanting survives and grows well in the forest and the disease organisms rapidly disappear (Dumroese et al. 1993, 2000; Smith 1967).

Globally, diseases caused by *Phytophthora* species are garnering more attention because of their potential to harm natural areas. For example, a *Phytophthora* introduced to Australia through nursery stock threatens one of the world's most biologically diverse areas by driving some species to extinction (Shearer et al. 2007). In California, the most well-known *Phytophthora* is *P. ramorum*, which causes sudden oak death. Since its detection 2 decades ago, the amount of worldwide research into this *Phytophthora* species has grown exponentially, and in the United States, escalated in 2004 after nurseries shipped stock contaminated with sudden oak death nationwide (Stokstad 2004).

Phytophthora species, commonly found in nurseries (Jung et al. 2016) including those in the West (Dumroese and James 2005), are commonly called "water molds" because they thrive in moist conditions and their spores are unusual in that they can swim through films of water on leaves or through water-filled pores in the soil. Thus, in nurseries that grow seedlings in containers (container nursery), use excessive irrigation and place containers on the ground, or nurseries that grow plants in soil (bareroot nursery) and use excessive irrigation or have poorly drained soils, *Phytophthora* easily moves from plant to plant. Unfortunately, infected but symptomless plants escape detection and can spread the disease (Migliorini et al. 2015; Simamora et al. 2017). Indeed, *Phytophthora* species have been found in native plant nurseries and on restoration sites in California (Rooney-Latham et al. 2015). Once introduced into natural areas, these pathogens are most likely impossible to eradicate (Hillman et al. 2016). In nurseries, Phytophthora diseases, like all other root diseases, can be effectively managed by using integrated pest management techniques with an emphasis on sanitation (Dumroese 2012). The Phytophthoras in Native Habitats Work Group (www.calphytos.org) provides extensive best-management practices for nurseries and land managers to reduce threats from introductions of potentially devastating *Phytophthora* species. Fortunately, the Lassen and the Modoc are in a low risk area for sudden oak death (Koch and Smith 2012).

Tree and Shrub Range Changes

Niche modeling has provided a great deal of information about the distribution of species now, and their predicted distributions into the future. Schlaepfer et al. (2012a) developed an ecohydrological niche model (which included information on soil water availability to plants) for big sagebrush from across the Western United States, including Northeastern California, which the authors felt explained more about the biology of the species than a climate niche model alone, particularly under climate change. Given two future climate-change scenarios, Schlaepfer et al. (2012b) used species distribution models based on climate and hydrology to predict areas that will be suitable for sagebrush into the future, and found that the amount of area in Northeastern California that is appropriate for sagebrush is forecasted to decrease (fig. 6.1.3), though much will still be consistent with sagebrush habitat (see Chapter 4.3, Dumroese, this synthesis, *Sagebrush Rangelands and Greater Sage-grouse in Northeastern California,* for more discussion about sagebrush rangelands and greater sage-grouse). Still and Richardson (2015) modeled the niche of Wyoming big sagebrush (*A. tridentata wyomingensis*) and, for Northeastern California, noted that the area continues to be appropriate for Wyoming big sagebrush through anticipated climate change to 2050 and suggested that restoration efforts should focus on areas that are predicted to be appropriate for sagebrush into the future (fig. 6.1.4).

The ability for seeds to move across the landscape and germinate will be important as ranges shift under climate change. Schlaepfer et al. (2014) modeled germination in

Figure 6.1.3—Models of climate and species distribution suggest that the area suitable for mountain big sagebrush, such as these growing in Eastern Lassen County, will decrease in Northeastern California (photo by Dawn M. Davis, used with permission).

big sagebrush and concluded that Northeastern California had consistently good conditions for germination to take place. Based on that study, Schlaepfer et al. (2015) modeled regeneration at leading- and trailing-edge populations and found one-third lower probabilities of recruitment at trailing-edge sites under current climate conditions, whereas leading-edge populations are predicted to have higher recruitment. Sagebrush seeds have been found in the local seed bank in Northeastern Nevada (Barga and Leger 2018), but work in Northeastern Montana at the range edge of big sagebrush habitat showed little germination of sagebrush seedlings from the seed bank (Martyn et al. 2016). These results of Martyn et al. (2016) support other, earlier findings of the short lifespan of sagebrush seeds in the seed bank (Meyer 1994; Young and Evans 1989). In Northeastern Nevada, shrub cover was the factor most associated with the size and content of the total (not just sagebrush) local seed bank (Barga and Leger 2018).

Empirical tests of the effects of warming on plant growth and species interactions have also contributed to our understanding of plant responses to climate change. Kopp and Cleland (2015) used artificial warming chambers on the Inyo National Forest to look at species interactions with sagebrush and a native plant, fiveleaf clover (*Trifolium andersonii*), which appears to be doing poorly under sagebrush encroachment. They show this clover had earlier flowering with warmer temperatures, but that the presence of sagebrush changed the phenological response to warming as well as the number of flowers produced, suggesting the importance of including species' interactions as climate changes and species' range shifts in response to climate. Another, less-investigated aspect includes the diversity of fungal species living inside leaf tissue (endophtypes). A study in Gambel oak (*Quercus gambelii*) found that endophytic communities varied based on solar radiation and, hence, climate, suggesting that climate change could impact these community interactions as well (Koide et al. 2017). Sagebrush plants are also subject to drought stress and drought-related mortality. Karban and Pezzola (2017), on sites near Truckee, CA, examined effects of drought on sagebrush during the 2010–2015 drought. By 2016, most of the observed plants had some branch mortality, and 14 percent, growing in a more competitive environment, had died. Branches that flowered more often suffered mortality the following year, and plants with more branches were able to keep a larger proportion of them alive through the drought.

The association between sagebrush and climate is complex, though precipitation has often been found to be an important driver. Several different lines of evidence show that the distribution and range of sagebrush are closely tied to climate. Climatic envelope modeling of the entire range of sagebrush (considering seven species and subspecies of *Artemisia*) showed that a range of predicted climate change impacts on sagebrush ecosystems on the Lassen and Modoc is possible, from low to very high (Bradley 2010). Dalgleish et al. (2011) used historical data from the U.S. Sheep Experiment Station in Idaho to examine the demographic rates and climate change impacts of two grasses—bluebunch wheatgrass (*Pseudoroegneria spicata*) and needle-and-thread grass (*Hesperostipa comata*)—as well as three-tipped sagebrush (*Artemisia tripartita*). They found that precipitation could have a potential impact on plant demography, particularly February and March snow, total annual precipitation as well as summer temperature. Annual growth rings in mountain big sagebrush (*Artemisia tridentata* ssp. *vaseyana*) in Colorado reveal this species is water limited and a decrease in mountain sagebrush growth with increasing temperatures under climate change is predicted (Poore et al. 2009). Similar results were found by (Apodaca et al. 2017) in Nevada, where the size of annual growth rings in big sagebrush was positively influenced by total annual precipitation, and negatively influenced by the mean maximum temperature during the growing season, suggesting slowing growth in a drier, warmer climate.

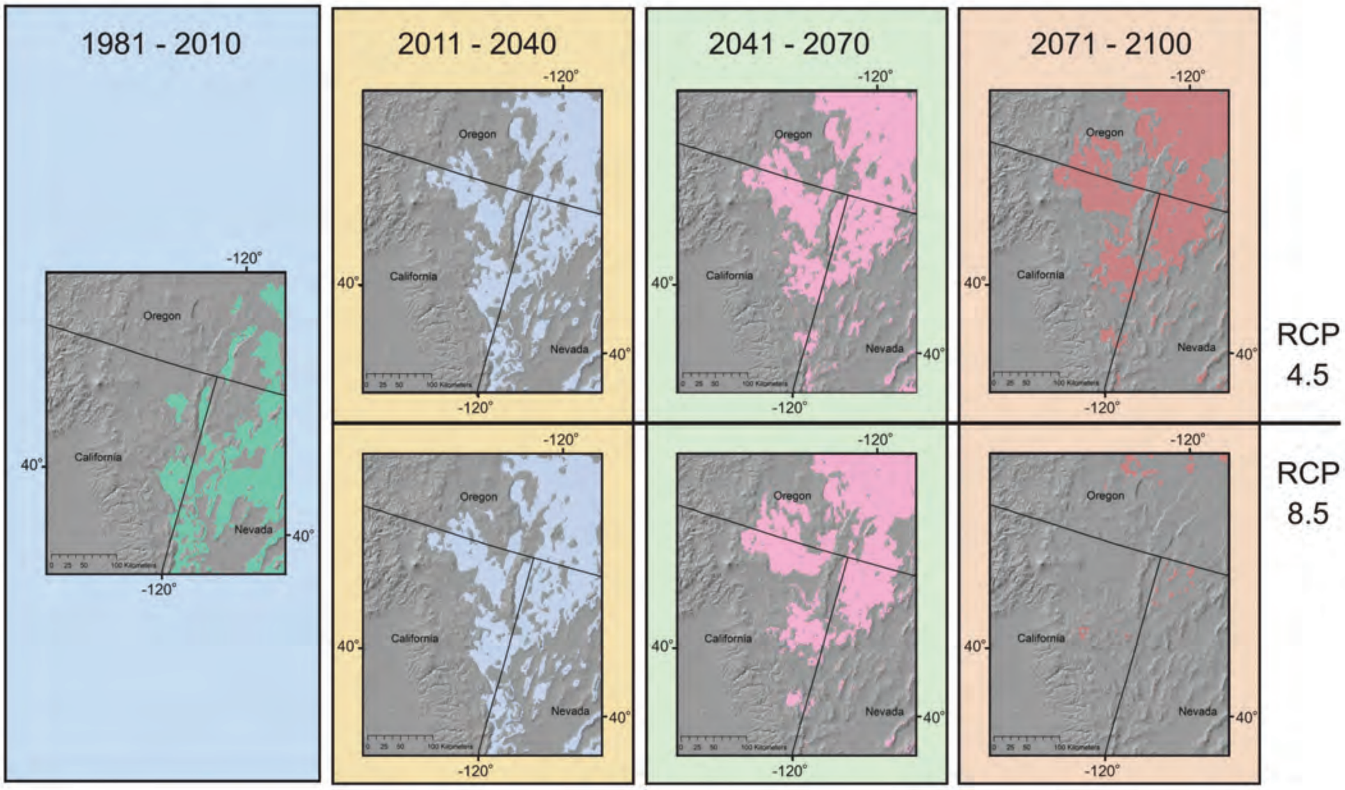

Figure 6.1.4—The climate niche for Wyoming big sagebrush under different climate projections in Northeastern California. Each panel represents a specific combination of Representative Concentration Pathway (RCP) and years. RCPs are projections of the trajectory of greenhouse gas concentrations and are expressed in terms of radiative forcing (the difference between incoming solar insolation and re-radiation of energy back into space). An RCP 8.5 scenario represents no change in the current rate of greenhouse gas emissions whereas the RCP 4.5 scenario reflects reductions in emissions from current rates. For each RCP, three 30-year increments are presented (adapted from Still and Richardson 2015).

Fire and Climate Change

The Sierra Nevada Science Synthesis (Long et al. 2014a) dedicates an entire section to fire in the Sierra Nevada, which is considered a fire-adapted system where historically fire played an important role in shaping and maintaining the ecosystem. Under predicted climate change, warmer, drier conditions are expected to lengthen the fire season and increase fire severity across the West. Not all models, however, provide the same fire predictions. While the models of Brown et al. (2004) and Stavros et al. (2014) both suggest an increase in fire activity because the fire season will start earlier and end later, they disagree on the frequency of very large wildland fires (VLWFs). Stavros et al. (2014), modeling across the West, predict more frequent VLWFs, whereas Brown et al. (2004), looking specifically at Northeastern California, predict a decline in VLWFs because of an anticipated decline in the number of extremely dry days having conditions conducive to large fires.

On the Lassen and Modoc, sagebrush habitats are also impacted by fire, but are not considered historically fire-adapted systems (Knapp 1996). The fire ecology of the region was described in detail by Riegel et al. (2006). The focus of this chapter is on the interaction between sagebrush and disturbances such as fire, invasive species, and climate change.

Fire Processes and Disturbance

In sagebrush habitats, fire cycles have changed since the introduction of cheatgrass (*Bromus tectorum*; Knapp 1996; Miller et al. 2011). Parks et al. (2015) looked at departure from the "expected" amount of fire to determine where fire deficits were occurring. The Great Basin landscape type in Northern California had a surplus of fire, that is, more acres burned than were predicted to burn. The authors showed an association between cheatgrass distribution in the Lassen and Modoc and increased fire frequency. In a review of fire effects on soils in sagebrush steppe, Sankey et al. (2012) found that, before fire, soils under shrubs had more nutrients, and this did not change after fire. However, more intense fires caused greater soil temperatures that made soils more susceptible to erosion due to water. While herbaceous plants recovered from fire faster than shrubs, they remain easier to burn, which can result in more frequent fires. Patterns of precipitation can be used to model cheatgrass fire occurrence (Poore et al. 2009). The authors found that fire was more likely to occur in a dry year that followed several wetter years, which allowed for an accumulation of fine fuels from the cheatgrass.

Taylor et al. (2014) reviewed 18 studies to determine if cheatgrass was recruited after fire, and if the likelihood of that recruitment was associated with climate variables. Indeed, warmer and drier sites had the highest probability of a positive response by cheatgrass to fire. In Northeastern California, Coates et al. (2016) noted that increased precipitation may, in fact, result in less sagebrush, as fine-fuel densities (i.e., cheatgrass) increase with more precipitation, resulting in larger burned areas. Indeed, Wade and Loik (2017) found that cheatgrass is able to take advantage of a pulse of spring precipitation, as measured through carbon assimilation, compared to other native plant species.

Fire also shapes the distribution of species. Dodson and Root (2015) looked at postfire recovery after the Eyerly fire in Northern Oregon. They found an association with climatic moisture deficit and vegetation, with more native, perennial species in wetter locations. The authors state, "The strong pattern in the present observational study suggests that increasing temperatures and decreasing moisture availability may be key drivers of vegetation changes following wildfire in the future, but caution is warranted in extrapolating these results" (p. 672). They explain that the study area was unique, and different sites could show different results. Yang et al. (2015) modeled the distribution of quaking aspen (*Populus tremuloides*) in Northeastern Nevada and examined how the range of quaking aspen is expected change under climate change. Their models show that the land area occupied by quaking aspen is predicted to be reduced under climate change, however, when fire is included in the model, the area occupied by quaking aspen is larger. Finally, for five California tree species, Hood et al. (2010) created models for tree mortality based on extent of crown injury in response to fire: smaller sugar pine (*Pinus lambertiana*) trees were more often killed, whereas larger ponderosa (*Pinus ponderosa*) and Jeffrey (*Pinus jeffreyi*) pines were more likely to succumb.

Managed Fire, Fuels Reduction

In some sagebrush communities, livestock grazing is an available approach to fuels management to reduce the risk or intensity of wildfire. According to Davies et al. (2009) livestock grazing started in sagebrush communities in the mid to late 1800s and is not considered a historical disturbance (see Chapter 3.1, Warren, this synthesis, *Perceptions and History of Rangeland;* Chapter 3.2, Dumroese, this synthesis, *Rangeland in Northeastern California;* and Chapter 3.3, Padgett, this synthesis, *Weeds, Wheels, Fire, and Juniper: Threats to Sagebrush Steppe*). Davies et al. (2009) compared grazing and fire disturbances on the vegetation at the Northern Great Basin Experimental Range, 56 km west of Burns, OR. Cheatgrass invaded the most in nongrazed and burned plots, while sagebrush densities decreased dramatically with burning, regardless of grazing treatment. Grazing disturbed the native plant community less than fire, the historical disturbance regime. In fact, this could be due to fuel loading resulting in more loss of native plants after fire. Davies et al. (2016) examined the relationship between winter grazing by cattle and fire intensity in Southeastern Oregon. They found it was an effective fuels reduction treatment for sagebrush, with a reduction in the amount of litter in the grazed plots, and a subsequent lower burning temperature. Diamond et al. (2009) also found an association with timing of grazing and a change in fire behavior, with lower flame lengths associated with the grazed plots.

Hurteau et al. (2014) reviewed the literature on fire in the Southwestern United States (Arizona, New Mexico, Nevada, and California). They discuss fire frequencies, suppression, climate and fire associations, vegetation, and climate change. Hurteau et al. (2014) also discuss management implications of projected increased fire frequencies under climate change, and the research that has gone into the idea to "restore fire as an ecologically beneficial process" (p. 286). They review the research behind several management practices, including forest thinning and prescribed burning, which they conclude "is the most effective means of reducing high-severity fire risk" (p. 286). Safford and Van de Water (2014) determined the Fire Return Interval Departure across California and found an average of 72 years for the Lassen and 50 years for the Modoc, noting that fewer fires are now occurring in these two national forests compared to pre-European settlement.

Postfire recovery in sagebrush habitats can require many years, or recovery to prior conditions may be impossible due to transitioning to an altered state (Knapp 1996). Moreover, different types of sagebrush habitats have different fire-return intervals, depending on sagebrush species and fire-return intervals (Miller et al. 2011). Ellsworth et al. (2016), working in Southcentral Oregon 17 years after a wildfire, found cheatgrass very common in all post-burning plots, but not in unburned controls. Shrub cover decreased dramatically after burns. However, early stages of recruitment were observed and young sagebrush plants were colonizing the plots. They felt there was "strong postfire resiliency" in their study system, and suggested fire suppression in sagebrush steppe habitats may not be "universally appropriate." Haubensak et al. (2009) looked at salt deserts in Northern Nevada and found that 5 years after wildfire the local species, bud sagebrush (*Artemisia spinescens,* also referred to as *Picrothamnus desertorum*), was not found in the burned sites, potentially due to grazing, but that nonnative species abundance was much greater. Several guides for managing and restoring sagebrush ecosystems are listed in Chapter 3.2 (Dumroese, this synthesis, *Rangeland in Northeastern California*).

The relationship between plant species composition and fire has impacts on other species as well. MontBlanc et al. (2007) examined the association between piñon-juniper/shrublands and fire and ant species diversity in Central Nevada. They found more ants on plots that were burned than unburned, though species richness stayed the same. Elevation was the major driver of species compositional changes in ants at this site. "Our study results may indicate that burns conducted in a patchy, heterogeneous fashion can provide a variety of habitat conditions and facilitate the persistence of diversity in ant species" (p. 485).

Social Impacts of Smoke

The social impacts of smoke in California have been reviewed extensively in the Sierra Nevada Science Synthesis (Long et al. 2014a) as well as the Northwest Forest Plan Science Synthesis (Spies et al. 2018). Section 8 of the Sierra Nevada Science Synthesis addresses air quality in California and highlights research on social impacts of smoke from wildfires and prescribed fires, particularly in the Southern Sierra (fig. 6.1.5). They contrast wildfires with prescribed fires and point out that wildfires generally burn when fuels are drier, and thus produce more particulate matter, which are lofted higher into the atmosphere. Controlled fires often occur in moister, cooler conditions, and their timing can be regulated to minimize impacts on local air quality, though with cooler air, smoke tends to be retained for longer periods, and controlled burns have caused high pollution days in the Tahoe Basin.

Figure 6.1.5—Smoke from wild and prescribed fires, such as this rising from Halls Flat on the Lassen, is an important issue in forest management in the Western United States. Both the Sierra Nevada Science Synthesis and the Northwest Forest Plan Science Synthesis have extensive discussions about the social aspects of smoke (photo by Deborah Mayer, Forest Service).

Chapter 10 of the Northwest Forest Plan Science Synthesis discusses the environmental and social justice implications of the impacts of wildfire smoke in their study area. They cite studies from the Southeastern United States that have concluded that smoke impacts all communities equally, regardless of socioeconomic status, however, they caution that more studies are needed in their study area.

Implications of Climate Change for Postfire Restoration

Chapter 4.3, *Post-Wildfire Management,* in the Sierra Nevada Science Synthesis (Long et al. 2014b) examines different management actions after fire. Burned Area Emergency Response is the first restoration action to occur after wildfire. This initial response is designed to stabilize the soils, and "protect life, property, water quality and ecosystems" (Long et al. 2014b, p. 188). Salvage logging, replanting, and other long-term restoration practices are also considered in the Sierra Nevada Science Synthesis. The chapter discusses these different approaches in depth and concludes that balancing economic and ecological costs and benefits in the short- and long-term must be considered. Issues include impacts to native plant and shrub species. For example, Knapp and Ritchie (2016) showed that increasing the intensity of salvage logging subsequently reduced the number of native shrub species on the site and the percent cover of them.

Postfire restoration is also addressed in the Northwest Forest Plan Science Synthesis (Spies et al. 2018). Chapter 3 discusses old-growth forests, and the interaction with fire and succession. It discusses the ecological consequences of salvage logging, highlighting concerns about altered ecosystem conditions and impacts with the removal of the dead biomass.

The implications of climate change in seeding projects that often occur as part of postfire restoration are discussed below.

Climate Change Impacts on Threatened Ecosystems and Species

In each section below, research regarding the response of species within the study area to a changing climate is synthesized, with a focus on Great Basin ecosystems.

Invertebrates

The diversity of butterflies across the Great Basin has been studied in detail. Fleishman et al. (2001) looked at the potential impacts of climate change on the diversity of butterfly species across the Great Basin. They noted that while vegetation zones in general are expected to shift upslope with climate change, individual host plant species (plants consumed by larvae) may not. Moreover, butterflies also require nectar sources, specific oviposition environments, etc. and hence the distribution of host plants may not match the distribution of the butterfly species. Few butterfly species are predicted to be lost from the Great Basin as a whole. Fleishman et al. (2001) note that in the Middle Holocene, temperatures in the basin were several degrees warmer than today, so the authors predicted that there may be some ability to withstand the current warming trends, as species that could not withstand such climatic changes have already been extirpated. Fleishman and Mac Nally (2003) examined butterfly diversity in two mountain ranges in Central Nevada and compared data collected 6 years apart. The prediction is for a change in temperature in the Great Basin of 1.1 to 1.6 °F (2 to 3 °C), and a 10- percent decrease in summer precipitation, and a 15- to 40-percent increase in precipitation in other seasons. With 6 years of sampling, they found very similar results and concluded there may be a time lag in response to a changing climate. They speculate that fauna that live in "the Great Basin are 'tough-tested'—species with low tolerance for environmental variability probably were extirpated long ago" (p. 400).

Butterfly diversity is indeed tied to vegetation, and management activities have the potential to impact that diversity. McIver and Macke (2014) examine the effects of fire and fuels treatments on butterfly diversity and found that any treatments that resulted in the removal of trees (e.g., junipers) tended to increase the amount of soil moisture available. This increased the amount of herbaceous food plants available for butterfly larvae, and hence the number of butterflies. Because of differences in butterfly abundances between years and the time lags involved, they suggest that any butterfly monitoring program needs to be intensive and long-term in order to be informative.

Two studies suggest that the local invertebrate populations in the Lassen and Modoc are unique compared to other parts of the Great Basin. Miller et al. (2014) examined the phylogenetic relationship between populations of the northern scorpion (*Paruroctonus boreus*) in the Intermountain West and found the Lassen population to be genetically different from those in other parts of the range. Based on their modeling, they conclude that suitable habitat for these scorpions in Northeastern California has persisted from the Last Interglacial Maximum through the Last Glacial Maximum to the present day. Schultheis et al. (2012) examined the current and Last Glacial Maximum distribution of stoneflies (*Doroneuria baumanni*) across the Great Basin. Using molecular genetic approaches, they showed that the stoneflies in Northeastern California and Nevada were genetically distinct from other stoneflies. They conclude that changes in historic climate have shaped current patterns of genetic diversity.

Birds

A number of bird species found within the study area have been studied with regards to responses to climate change. Siegel et al. (2014) used the Climate Change Vulnerability Index to look at predicted climate change impacts in a number of bird species in the Sierra Nevada. Northern goshawks (*Accipiter gentilis*) were listed as "moderately vulnerable" under two different climate scenarios. Spotted owls (*Strix occidentalis*) and black-backed woodpeckers (*Picoides arcticus*) were "presumed stable" (see Chapter 4.2, Hanberry and Dumroese, this synthesis, *Biodiversity and Representative Species in Dry Pine Forests* for more discussion about black-backed woodpeckers).

The northern goshawk has a circumpolar distribution, so global studies are relevant to a discussion of climate change impacts on the species. De Volo et al. (2013) studied the population genetics of northern goshawks from Southeastern Alaska to the Appalachians (including samples from the Lassen and Modoc) and found four genetic variants (haplotypes) in the Sierra-Cascade region and evidence that this population was isolated from other populations during the most recent period of glaciation, with a glacial maximum approximately 21,000 years ago, and ending 11,000 years ago.

Two studies have shown interesting changes in nest and body size in northern goshawks. Møller and Nielsen (2015) examined northern goshawks in Denmark and found a strong positive relationship between nest size and temperature, with larger nests occurring where temperatures are warmer. Tornberg et al. (2014) studied bird size from 200 museum specimens in Finland and observed a decrease in body size from 1962 to 2008, possibly relating to changes in prey type and availability. They commented that this was consistent with what is expected under climate change. This seems to contrast with the larger nest sizes observed in Denmark, as larger nests were correlated with larger bodies.

Northern spotted owls (*Strix occidentalis caurina*) have been the focus of a great deal of research (empirical and modeling) in regards to their response to predicted climate change; however, studies have yielded varying results. Glenn et al. (2011) looked at weather patterns and the northern spotted owl from Oregon and Washington and concluded that climate change, in the form of hotter, drier summers along with wetter winters, could have a negative impact on their populations. Climate-based niche modeling focusing on Oregon, Washington, and Northern California showed changes in the probability of occurrence of owls under different climate change scenarios, but the model predictions were quite variable (Carroll 2010). Focusing on the Tahoe and Eldorado National Forests, Jones et al. (2016) modeled California spotted owl (*S. o. occidentalis*) distributions and found that rising summer temperatures resulted in a decline in predicted owl occurrence. Fledgling counts of California spotted owls on the Lassen were associated with climate: warmer early nesting temperatures and less precipitation resulted in higher fledgling counts (Cade et al. 2017). Unexpectedly, these authors did not find an association with previous-year precipitation and fledgling number.

Cicero and Koo (2012) looked at divergence in the sage sparrow (*Amphispiza belli*) species complex (including samples from the Lassen and Modoc), by examining variation in several morphological traits as well as sequence variation at an mtDNA marker, and climate niche modeling. They found strong evidence for divergence between the taxonomic groups (at the time of publication, there were three subspecies). Further, they showed that during the last 120,000 years, the distribution of the climate niche for sage sparrows has changed dramatically across the West. While they did not hypothesize about the birds' distribution in the future under climate change, a strong association exists between climate and local distribution of these taxa.

All sage-grouse are considered sagebrush obligates, relying on intact sagebrush habitats during all life stages (Schroeder et al. 1999) and that habitat is under threat from a number of factors (Knick and Connelly 2011). Among climate-related variables, precipitation and the availability of water are the key factors for sage-grouse survival and reproduction in arid and semiarid regions. Donnelly et al. (2016) found that greater sage-grouse lek distribution was structured by the proximity to mesic (wetlands, etc.) resources with an observed average distance of 3.3 miles (5.3 km). Gibson et al. (2017) looked at nest site selection of greater sage-grouse in Nevada and found an association between weather and drought patterns and nest site selection. They found that drought reduced fitness, but that females were able to select the most productive nest sites among those available, thus mitigating, but not eliminating the impacts of drought on fitness. Blomberg et al. (2014) studying greater sage-grouse in Nevada, found an association with climate and probability of post-fledgling survival, with a positive correlation with precipitation (fig. 6.1.6). An earlier study in Nevada by Nisbet et al. (1983) found positive associations between lek locations and water sources (less than 1.25 miles [2 km]) and precipitation (more than 10 inches [25 cm]).

Other approaches have been used to look at the relationship between climate, habitat, and sage-grouse success. Homer et al. (2015), focusing on Southwestern Wyoming, used remote sensing data to project trends in sagebrush habitat to 2050 based on changes in precipitation using IPCC climate models. Their models "predicted losses of -12 percent of greater sage-grouse nesting habitat and -4 percent of summer habitat from 2006 to 2050 due to climate alone" (p. 143). They considered these losses to be substantial.

Because sage-grouse rely almost exclusively on sagebrush habitats, encroachment by trees into sagebrush communities presents a major threat to population persistence, as it can severely reduce the amount and quality of available habitat (Miller et al. 2011, 2017; see *Juniper Woodlands* in Chapter 2.1 [Moser, this synthesis, *Understanding and Managing the Dry, Conifer Forests of Northeastern California*] and *Impacts of Conifer Encroachment* in Chapter 4.3 [Dumroese, this synthesis, *Sagebrush Rangelands and Greater Sage-grouse in Northeastern California*] for more information). Falkowski et al. (2017) mapped where woody plants, mostly conifers and mesquite (*Prosopis* species), were found within the established range of greater sage-grouse. In Northeastern California, about half of the range had conifers at levels that could represent habitat loss for the sage-grouse. The authors discuss efforts to remove conifers from areas where there is encroachment to increase greater sage-grouse habitat. Pennington et al. (2016) conducted a literature review of the relationship between greater sage-grouse and forb species. They found a number of studies documenting the greater sage-grouse use of forbs for food and nesting habitat. However, they also found a lack of information on the relationship between climate and forbs, suggesting that their ability to understand how forbs important to sage-grouse will respond to climate change is limited, and they identified this as an information gap.

Figure 6.1.6—Ongoing research on greater sage-grouse informs land managers about critical habitat needs of this species and how predicted increases in drought brought by climate change may affect the resilience of this species on the landscape (photo by Dawn M. Davis, used with permission).

Other stressors impact greater sage-grouse populations and habitat including disease and human development. Walker et al. (2007) worked in Montana and Wyoming and presented the first data documenting the West Nile virus infection rate for wild greater sage-grouse. They also found the first documented occurrence of West Nile virus antibodies in greater sage-grouse, suggesting the birds had survived infection. Taylor et al. (2013), also working in Wyoming and Montana, studied energy development and West Nile virus occurrence in relation to the number of males on greater sage-grouse leks. They found that in non-disease-outbreak years, their models predicted drilling alone reduced the number of leks by 61 percent. In the absence of energy development, in an outbreak year, they predicted a reduction in the number of leks by 55 percent. Indeed, coal bed methane extraction results in the construction of ponds, which increases the amount of available mosquito habitat (Zou et al. 2006).

Reptiles

Jezkova et al. (2016) looked at range shifts in the desert horned lizard (*Phrynosoma platyrhinos*) species complex, and concluded the species expanded rapidly into the Great Basin after glaciation. Moreover, their results suggest that the niche of the species itself has shifted as the climate continued to change after the Last Glacial Maximum, with the species moving into warmer and drier climates (fig. 6.1.7).

Figure 6.1.7—After the last glaciation, the desert horned lizard moved rapidly into the Great Basin and its habitat niche has continued to shift with changes in climate since then (Jezkova et al. 2016; photo by Tony Kurz, used with permission).

Mammals

Carnivores

The black bear (*Ursus americanus*) population range in Western Nevada, just south and east of the Lassen and Modoc, has rebounded from its low observed around 1940. The genetic consequences of that recent range expansion were studied by Malaney et al. (2018), who concluded that the levels of connectivity between populations were insufficient to avoid a genetic bottleneck and to maintain a single genetic metapopulation. The authors argue that, given these results, it is important to maintain the connectivity between populations as temperatures rise with climate change, and more fragmentation occurs with spreading urbanization (Malaney et al. 2018).

Mesocarnivores

Barton and Wisely (2012) examined the population genetics of skunks and found that California striped skunk (*Mephitis mephitis*) populations are genetically distinct from all other striped skunk populations. They posited this was the result of post-and interglacial migration and isolation. This, along with the studies described earlier about scorpions (Miller et al. 2014) and stoneflies (Schultheis et al. 2012) suggest this isolation was not limited to skunks.

Rodents and Lagomorphs

Studies focused on rodents have found associations between the number of prey and the numbers of predators—results that link predator and prey abundance with specific management approaches. Holbrook et al. (2016) studied Piute ground squirrels (*Urocitellus mollis*) and American badgers (*Taxidea taxus*) in Southwestern Idaho and examined the relationship between climate and disturbance on the occupancy of these mammals. Specifically, they looked at the cheatgrass/fire cycle in relation to mammal distribution. Badger occupancy was directly related to squirrel (prey) occupancy. Increasing abundance of cheatgrass was associated with decreasing abundance of ground squirrels. Moreover, increasing frequency of fire was also associated with a decreased abundance of ground squirrels. If the cycle of cheatgrass/fire continues or increases, the data here suggest there could be a negative impact on both prey and predator species. They also showed that successful reseeding projects after fire have a strong positive effect on ground squirrel abundance, compared to untreated plots.

An important question in rodent and lagomorph (rabbits, hares, and pikas) diversity and abundance is the question of how ranges are shifting under climate change. Some, but not all, rodent species show evidence for shifting ranges. Rowe et al. (2015) looked at the ranges of 34 species of small mammals (rodents, shrews, and lagomorphs) in three regions of California (Lassen National Park, Yosemite National Park, and Sequoia Kings Canyon National Park) and found widespread evidence for range shifts in many of these species. However, the direction and patterns of these range shifts was not consistent across the different regions of the State. Local temperature was the best predictor of range shifts. Larrucea and Brussard (2008) looked at the presence of pygmy rabbit (*Brachylagus idahoensis*) at sites last surveyed in the 1950s, including sites in Modoc and Lassen Counties in California. They found rabbits at only 36 percent of the historic sites, with evidence for a shift uphill of about 500 feet (150 m). Of the historic sites, 16 percent had been burned, and the authors concluded that wildfires are "probably the greatest threat to pygmy rabbit habitat today." Morelli et al. (2012) used historical data to look at changes in the distribution of Belding's ground squirrel (*Urocitellus beldingi*), including Northeastern California. They found that 42 percent of historical sites are now unoccupied. Species distribution models show that under climate change, much of the current range of this squirrel will be extirpated. This is particularly alarming as this species is an important food source for raptors and other predators. Rodents have been shown to respond to climate change through tracking changing habitat. Through occupancy modeling, the piñon mouse (*Peromyscus truei*) was shown to be associated with its namesake, piñon pine (*Pinus monophylla*), which are expanding into sagebrush habitat. Thus, the range of the piñon mouse is also hypothesized to expand (Massey et al. 2017).

Other evidence has suggested, however, that some species are expanding their niche, and remaining in the same locations. Hornsby and Matocq (2012) focused on the bushy-tailed woodrat (*Neotoma cinerea*) and found that the woodrats in the study area are part of the intermountain clade that includes woodrats from Nevada, Utah, and Idaho. "The genetic pattern of recent demographic expansion is supported by the clade-specific ecological niche model, which emphasizes the notion that distinct evolutionary lineages within species may have different niche associations, and thus, unique responses to past and future climatic shifts" (p. 300). Jezkova et al. (2011) examined niche shifts in the chisel-toothed kangaroo rat (*Dipodomys microps*) and found that, "On the other hand, species that persisted in place throughout the climatic fluctuation of the late Pleistocene (such as *D. microps*) might respond differently to future climate changes as they might be capable of tolerating conditions beyond their current limits through either 'niche drifting' or 'niche evolution'" (p. 3500). This idea of a more flexible niche is supported by the findings of Terry et al. (2017) who showed, using carbon and nitrogen isotopes collected from bones in cave deposits as well as modern populations, that the diet of kangaroo rats was (and is) much more diverse than previously thought, and the species was (and is) not a dietary specialist.

Smith et al. (1995) showed that in woodrats, body size is closely tied to fecal pellet size. They then analyzed samples collected from paleo-middens in the Great Basin dating from 20,000 years ago to present. Body size fluctuated with temperature, with larger body sizes at times when the temperature was cooler. These results suggest that woodrat body size may continue to decrease with increasing temperatures under climate change.

Pika

Pikas (*Ochotona princeps*) are lagomorphs, small montane mammals related to rabbits. They occur throughout mountains of Western North America (Smith and Weston 1990), including the Sierra Nevada and adjacent Great Basin ranges. They have received a great deal of research attention for their potential vulnerability to warming climates (Beever et al. 2011). The species has come to serve as a model for climate change in mountain environments (fig. 6.1.8). They are best known for inhabiting high-elevation talus slopes, although they extend to lower elevations when habitat is available (Beever et al. 2008; Jeffress et al. 2017; Manning and Hagar 2011; Millar et al. 2013; Ray et al. 2016), and have a low tolerance for heat stress (Smith and Weston 1990). With climate change, a question is whether or not pika populations will move uphill, and what will happen when there is no more room for them to move uphill. This question has merit because studies have shown that the pattern of uphill movement of pikas in the Great Basin portion of the species range is not new, and has been occurring for thousands of years. Grayson (2005) reviewed evidence for the paleohistory dynamics of pikas in the Great Basin. Grayson argued that current pattern of local extinction at low and warm margins of the range follows a trend discernible throughout the past thousands of years. The Middle Holocene (7,500 to 4,000 years ago) in the Great Basin was warm and dry, and pika populations moved uphill about 1,430 feet (435 m) on average. Calkins et al. (2012) modeled future distribution of pikas across the Western United States and showed that with each degree of increased temperature, the distribution of pikas decreased. At about a 13-°F (7-°C) increase in temperature, pikas were nearly gone. Models by Mathewson et al. (2017) with a climate change component also predicted a reduced distribution of pika in the near future.

Figure 6.1.8—Perceived to be a species vulnerable to changes in climate, the pika, and its distribution in response to past changes in climate, is helping scientists predict where pikas may thrive in the future (photo by Ken Hickman, used with permission).

Multiple repeat-survey studies have compared historic (20th century) observational records of pika occupancy in the Great Basin to current occupancy. Most of these sites were at low and ecological margins of the species range. For such sites, these studies are in general agreement that pikas have shifted their distribution uphill and that warmer temperatures as well as cold winters are associated with pika extirpation. Beever et al. (2011) found that the minimum elevation of pika at 10 sites in the Great Basin Ecoregion has moved uphill at least 475 feet to a mean elevation of 1,194 feet (145 m to 364 m) between 1999 and 2008. Studies comparing historical records (generally 1890s to 2000s) have found a trend for pika populations to be extirpated from lower-elevation sites (Beever et al. 2016; Stewart et al. 2015; Wilkening et al. 2011). However, Beever et al. (2010) found an association between temperature and extirpation of pika populations in the Great Basin including both chronic heat stress and acute cold stress.

Although temperature is a significant driver of pika distribution, precipitation may also be an important variable. In general, studies have shown that pika are less likely to be found in drier places (Henry et al. 2012; Jeffress et al. 2013). Beever et al. (2013) found an association with precipitation (positive) and temperature (negative) and occurrence of pika in the Great Basin (including the Lassen and Modoc). In Colorado and Montana, Bhattacharyya and Ray (2015) found evidence that several direct and/or indirect effects of climate change, such as warming summers, loss of preferred winter forage, reduced snow cover, and changing cache composition may adversely affect pika.

Despite this evidence, when more typical upland habitat in the Great Basin (as well as elsewhere) has been surveyed, pika populations occur widespread across the region as well as in previously unexpected locations (e.g., Jeffress et al. 2017; Millar and Westfall 2010; Millar et al. 2018). Beever et al. (2008) published the first records of pikas in Hays Canyon Mountain Range (just east of the Lassen and Modoc), and many other marginal locations have been similarly reported. In the Hayes Canyon Range, pikas were using cheatgrass as a food resource, interesting because cheatgrass does not normally grow at elevations where pikas live. Millar et al. (2013) and Millar and Westfall (2010) documented newly observed low-elevation and montane pika sites in the Great Basin including two low-elevation, dry-climate populations of pikas in the Madeline Uplands on MacDonald and Observation Peaks of Lassen County (Millar et al. 2013). This area is much drier and warmer than might be expected to support pikas, although pikas were documented in the region in the early 20th century by Howell (1924). Pikas have been found living in lava-talus slopes and caves within the Lava Beds National Monument of Northeastern California, a site on the edge of their range (Ray et al. 2016). The microenvironments of these slopes and caves are relatively cooler and the authors hypothesize these conditions are more similar to those at the center of the pikas' range, and that perhaps local adaptation has occurred in response to the warmer summer temperatures of the area.

In general, pikas appear able to persist under what might appear to be marginal conditions due to the unique microclimate processes generated within taluses (Millar et al. 2014; Rodhouse et al. 2017). In summer, convection circulates cool air in the talus interior to the bases where pikas live, with the result that mean temperatures are lower than external ambient air and diurnal fluctuations greatly attenuated. In winter, the opposite happens, where snow cover adds thermal insulation to the open talus matrix, keeping these habitats stably warmer than external conditions.

However, as pika distributions are moving, often uphill, in response to a warming climate, so are other rodent species, and with those species come new disease threats that pikas have not been exposed to previously. Foley et al. (2017) have shown that in high-elevation sites in Colorado and Montana, the diversity of flea species found on pikas includes species that specialize on pikas as well as species that specialize on other rodent species, presumably from contact with those other rodent species.

Genetic approaches have also been used to determine patterns of movement among pika populations. Castillo et al. (2016) looked at gene flow and connectivity among sites and determined that gene flow was likely, and consistently, influenced by climate-related variables (especially the relative potential for heat stress) across multiple landscapes. Across the entire range of the pika, genetic variation is highly structured with five primary groups; the Lassen and Modoc pika are genetically similar to those from all of California, parts of Nevada, Western Utah, and Oregon (Galbreath et al. 2010).

Finally, given the increasing frequency and intensity of fire, it appears pika have considerable resiliency to this disturbance. Varner et al. (2015) looked at the response of a pika population to a wildfire on the north face of Mt. Hood in Oregon and found that pika were returning to even the most severely burned patches 2 years after burning. In the Madeleine Plains of Lassen County, pika occupancy was greatest in taluses where fires had burned, renewing forb and graminoid vegetation around the taluses that pikas use for foraging and caching (Millar et al. 2013).

Risks of Climate Change to Riparian, Meadows, and Other Aquatic Systems

Vernal pools have received a great deal of research attention, as they harbor a large number of native plant and animal species (see Chapter 4.2, Padgett, this synthesis, *Aquatic Ecosystems, Vernal Pools, and Other Unique Wetlands* for more information about vernal pools as well as redband trout [*Oncorhynchus mykiss* ssp.]). Grazing is an issue for some of these native plant species.

Merriam et al. (2016) looked at fenced and unfenced vernal pools in Northeastern California and found that while livestock grazing late in the season increased the occurrence of the threatened grass species, slender Orcutt grass (*Orcuttia tenuis*), variation in annual precipitation patterns had a greater effect on *O. tenuis.* Looking at the broader vernal pool plant community, Bovee et al. (2018) found that excluding cattle for up to 20 years resulted in fewer annual vernal pool specialists, but promoted perennial vernal pool species. They concluded that “livestock grazing may contribute more to plant community structure than inter-annual or between-pool variation in precipitation in montane vernal pools” (p. 17).

Climate change is expected to impact vernal pools through changes in temperature and hydrology. Gosejohan et al. (2017) looked at vernal pools on the Modoc Plateau, and found a strong association between-pool hydrology (maximum depth and inundation period) and plant community organization. They point out that understanding the relationship between-pool hydrology and plant species is the first step in understanding how climate change will impact plant distribution in vernal pool habitats. Pyke (2005) studied the relationship between climate change and brachiopods (small aquatic crustaceans) found in vernal pools in the Central Valley of California. While this is a very different ecosystem than found in the Lassen and Modoc, his findings are still relevant. He examined effects of climate change on different scales of vernal pools and found that changes in precipitation (inundation) and temperature (evaporation) will impact individual pools. Across a local landscape, the availability of reproductive habitat and the movement of predators such as dragonflies will influence branchiopod distributions (fig. 6.1.9).

Amphibians in the Western United States face increasing threats, including introduced species and climate change. For example, in their review, Ryan et al. (2014) discuss studies showing the return of amphibian populations after fish removals in montane ponds, and point out that this is the one management approach that can be done in the short term by local managers. The impacts of a changing climate are, however, complex. O’Regan et al. (2014) show that for the Great Basin spadefoot toad (*Spea intermontana*), the anticipated faster drying of temporary ponds under climate change accompanied by higher temperatures, which reduced the amount of time for metamorphosis, resulted in faster growth and no reduction in toad size.

As the Great Basin becomes drier and temperatures increase, native fish habitat is a concern. Warren et al. (2014) examined limits to upstream migration of native and nonnative trout species. They found more limits on upstream migration in northern species of trout in the Great Basin, compared to southern ones. “Our assessment of potential upstream shifts in the bioclimatic envelope (with both moving and static upper limits) supports the overall conclusion that even slight shifts in stream conditions could substantially reduce available habitat for trout in the northwestern Great Basin.” Howard and Nobel (2018) looked at hydrology and phytoplankton in Butte Lake in

Figure 6.1.9—Aquatic habitats support a variety of wildlife, including this four-spotted skimmer (*Libellula quadrimaculata*). Dragonflies can also affect the populations and distributions of other fauna inhabiting vernal pools (photo by Tony Kurz, used with permission).

Lassen Volcanic National Park and found a dramatic shift in diatom communities during a drought year, and suggested that this community shift could influence fish communities as well. Schultz et al. (2017) looked at stream temperatures associated with both drought and fire at a location directly north of the Lassen and Modoc and noted that many streams listed as perennial lacked water during drought years. In addition, they found temperatures increased 7.2 °F (4 °C) immediately after a fire. They conclude that, "Combined drought and postfire conditions appeared to greatly restrict thermally-suitable habitat for Lahontan cutthroat trout (*Oncorhynchus clarkii henshawi*)" (p. 60).

Climate Change Impacts on Seeding Projects

Similar to conifer trees, different species of rangeland forbs have different germination strategies and populations within a species have been shown to have variation in germination requirements, often associated with variability in the climate where the seeds originate (Barga et al. 2017). Thus, national forest restoration projects, for which native plants are the first choice (Johnson et al. 2010)—whether they be trees, shrubs, grasses, or forbs—require managers to decide what sources of seeds to use. Several guidelines exist. For example, Buck et al. (1970) describe a set of tree seed zones for California that are currently used by the Forest Service to guide their reforestation projects. Seeds are transferred within a 500-foot (152-m) elevation band within a given seed zone. However, these seed zones are geographically based, and as such, their use under a changing climate is being questioned. Moreover, the seed zones focus only on trees, and not other native plant species. To address this gap, Johnson et al. (2010) suggest using seeds from similar ecosystems and to sample a seeds from parents from different locations within that ecosystem. This will result in a set of genetically diverse, locally adapted seeds for seeding projects. Bower et al. (2014) further refined these recommendations by proposing seed zones for native plants based on temperature and precipitation variables, paired with Level III Ecoregions (Omernik 1987). These zones are designed to be discrete units to maximize their usefulness for managers while providing flexibility for moving seeds across the landscape and ensuring the resulting plants are adapted to their new location. Another approach has been released recently; the Seedlot Selection Tool (https://seedlotselectiontool.org/sst/) uses climate models to determine the best seed source for a restoration project, based on climate-matching between the seed source and the planting site. The user can define several variables in the tool, including which climate variables to include in the model. The user can also select a climate change scenario and determine which seed sources have the best climate-matching in future climates as well as current ones.

References

Allen, R.J.; Luptowitz, R. 2017. El Nino-like teleconnection increases California precipitation in response to warming. Nature Communications. 8: 16055.

Anderegg, W.R.L.; Hicke, J.A.; Fisher, R.A.; [et al.]. 2015. Tree mortality from drought, insects, and their interactions in a changing climate. New Phytologist. 208: 674–683.

Apodaca, L.F.; Devitt, D.A.; Fenstermaker, L.F. 2017. Assessing growth response to climate in a Great Basin big sagebrush (*Artemisia tridentata*) plant community. Dendrochronologia. 45: 52–61.

Barga, S.; Dilts, T.E.; Leger, E.A. 2017. Climate variability affects the germination strategies exhibited by arid land plants. Oecologia. 185: 437–452.

Barga, S.; Leger, E.A. 2018. Shrub cover and fire history predict seed bank composition in Great Basin shrublands. Journal of Arid Environments. 154 40–50.

Barton, H.D.; Wisely, S.M. 2012. Phylogeography of striped skunks (*Mephitis mephitis*) in North America: Pleistocene dispersal and contemporary population structure. Journal of Mammalogy. 93: 38–51.

Bates, J.D.; Svejcar, T.; Miller, R.F.; [et al.]. 2006. The effects of precipitation timing on sagebrush steppe vegetation. Journal of Arid Environments. 64: 670–697.

Becklin, K.M.; Medeiros, J.S.; Sale, K.R.; [et al.]. 2014. Evolutionary history underlies plant physiological responses to global change since the last glacial maximum. Ecology Letters. 17: 691–699.

Beever, E.A.; Dobrowski, S.Z.; Long, J.; [et al.]. 2013. Understanding relationships among abundance, extirpation, and climate at ecoregional scales. Ecology. 94: 1563–1571.

Beever, E.A.; Perrine, J.D.; Rickman, T.; [et al.]. 2016. Pika (*Ochotona princeps*) losses from two isolated regions reflect temperature and water balance, but reflect habitat area in a mainland region. Journal of Mammalogy. 97: 1495–1511.

Beever, E.A.; Ray, C.; Mote, P.W.; [et al.]. 2010. Testing alternative models of climate-mediated extirpations. Ecological Applications. 20: 164–178.

Beever, E.A.; Ray, C.; Wilkening, J.L.; [et al.]. 2011. Contemporary climate change alters the pace and drivers of extinction. Global Change Biology. 17: 2054–2070.

Beever, E.A.; Wilkening, J.L.; McIvor, D.E.; [et al.]. 2008. American pikas (*Ochotona princeps*) in northwestern Nevada: a newly discovered population at a low-elevation site. Western North American Naturalist. 68: 8–14.

Benson, L.; Kashgarian, M.; Rye, R.; [et al.]. 2002. Holocene multidecadal and multicentennial droughts affecting Northern California and Nevada. Quaternary Science Reviews. 21: 659–682.

Bentz, B.J.; Régnière, J.; Fettig, C.J.; [et al.]. 2010. Climate change and bark beetles of the western United States and Canada: direct and indirect effects. Bioscience. 60: 602–613.

Bhattacharyya, S.; Ray, C. 2015. Of plants and pikas: evidence for a climate-mediated decline in forage and cache quality. Plant Ecology & Diversity. 8: 781–794.

Blomberg, E.J.; Sedinger, J.S.; Gibson, D.; [et al.]. 2014. Carryover effects and climatic conditions influence the postfledging survival of greater sage-grouse. Ecology and Evolution. 4: 4488–4499.

Bolshakova, V.L.J.; Evans, E.W. 2014. Spatial and temporal dynamics of *Aroga* moth (Lepidoptera: Gelechiidae) populations and damage to sagebrush in shrub steppe across varying elevation. Environmental Entomology. 43: 1475–1484.

Bolshakova, V.L.J.; Evans, E.W. 2016. Phenology of the sagebrush defoliating moth *Aroga websteri* (Lepidoptera: Gelechiidae), with application to population irruptions. Annals of the Entomological Society of America. 109: 424–431.

Bovee, K.M.; Merriam, K.E.; Gosejohan, M.C. 2018. Livestock grazing affects vernal pool specialists more than habitat generalists in montane vernal pools. Applied Vegetation Science. 21: 12–20.

Bower, A.D.; St Clair, B.; Erickson, V. 2014. Generalized provisional seed zones for native plants. Ecological Applications. 24: 913–919.

Bradford, J.B.; Schlaepfer, D.R.; Lauenroth, W.K. 2014. Ecohydrology of adjacent sagebrush and lodgepole pine ecosystems: the consequences of climate change and disturbance. Ecosystems. 17: 590–605.

Bradley, B.A. 2010. Assessing ecosystem threats from global and regional change: hierarchical modeling of risk to sagebrush ecosystems from climate change, land use and invasive species in Nevada, USA. Ecography. 33: 198–208.

Brown, T.J.; Hall, B.L.; Westerling, A.L. 2004. The impact of twenty-first century climate change on wildland fire danger in the western United States: an applications perspective. Climatic Change. 62: 365–388.

Buck, J.M.; Adams, R.S.; Cone, J.; [et al.]. 1970. California tree seed zones. San Francisco: U.S. Department of Agriculture, Forest Service, California Region. 5 p.

Cade, B.S.; Noon, B.R.; Scherer, R.D.; [et al.]. 2017. Logistic quantile regression provides improved estimates for bounded avian counts: a case study of California spotted owl fledgling production. The Auk. 134: 783–801.

Calkins, M.T.; Beever, E.A.; Boykin, K.G.; [et al.]. 2012. Not-so-splendid isolation: modeling climate-mediated range collapse of a montane mammal *Ochotona princeps* across numerous ecoregions. Ecography. 35: 780–791.

Carroll, C., 2010. Role of climatic niche models in focal-species-based conservation planning: assessing potential effects of climate change on northern spotted owl in the Pacific Northwest, USA. Biological Conservation. 143: 1432–1437.

Castillo, J.A.; Epps, C.W.; Jeffress, M.R.; [et al.]. 2016. Replicated landscape genetic and network analyses reveal wide variation in functional connectivity for American pikas. Ecological Applications. 26: 1660–1676.

Cayan, D.R.; Maurer, E.P.; Dettinger, M.D.; [et al.]. 2008. Climate change scenarios for the California region. Climatic Change. 87: S21–S42.

Cicero, C.; Koo, M.S. 2012. The role of niche divergence and phenotypic adaptation in promoting lineage diversification in the Sage Sparrow (*Artemisiospiza belli,* Aves: Emberizidae). Biological Journal of the Linnean Society. 107: 332–354.

Coates, P.S.; Ricca, M.A.; Prochazka, B.G.; [et al.]. 2016. Wildfire, climate, and invasive grass interactions negatively impact an indicator species by reshaping sagebrush ecosystems. Proceedings of the National Academy of Sciences of the United States of America. 113: 12745–12750.

Connelly, J.W.; Knick, S.T.; Braun, C.E.; [et al.]. 2011. Conservation of greater sage-grouse: a synthesis of current trends and future management. In: Knick, S.T.; Connelly, J.W., eds. Greater sage-grouse: ecology and conservation of a landscape species and its habitats. Berkeley, CA: University of California Press: 549–563.

Creeden, E.P.; Hicke, J.A.; Buotte, P.C. 2014. Climate, weather, and recent mountain pine beetle outbreaks in the western United States. Forest Ecology and Management. 312: 239–251.

Dalgleish, H.J.; Koons, D.N.; Hooten, M.B.; [et al.]. 2011. Climate influences the demography of three dominant sagebrush steppe plants. Ecology. 92: 75–85.

Davies, K.W.; Boyd, C.S.; Bates, J.D.; [et al.]. 2016. Winter grazing can reduce wildfire size, intensity and behaviour in a shrub-grassland. International Journal of Wildland Fire. 25: 191–199.

Davies, K.W.; Svejcar, T.J.; Bates, J.D. 2009. Interaction of historical and nonhistorical disturbances maintains native plant communities. Ecological Applications. 19: 1536–1545.

de Volo, S.B.; Reynolds, R.T.; Sonsthagen, S.A.; [et al.]. 2013. Phylogeography, postglacial gene flow, and population history of North American northern goshawks (*Accipiter gentilis*). The Auk. 130: 342–354.

Diamond, J.M.; Call, C.A.; Devoe, N. 2009. Effects of targeted cattle grazing on fire behavior of cheatgrass-dominated rangeland in the northern Great Basin, USA. International Journal of Wildland Fire. 18: 944–950.

Dodson, E.K.; Root, H.T. 2015. Native and exotic plant cover vary inversely along a climate gradient 11 years following stand-replacing wildfire in a dry coniferous forest, Oregon, USA. Global Change Biology. 21: 666–675.

Donnelly, J.P.; Naugle, D.E.; Hagen, C.A.; [et al.]. 2016. Public lands and private waters: scarce mesic resources structure land tenure and sage-grouse distributions. Ecosphere. 7(1): e01208.

Dumroese, R.K. 2012. Integrated nursery pest management. In: Cram, M.M.; Frank, M.S.; Mallams, K.M., tech. coords. Forest nursery pests. Agriculture Handbook 680 rev. 2012. Washington, DC: U.S. Department of Agriculture, Forest Service: 5–12.

Dumroese, R.K.; James, R.L. 2005. Root diseases in bareroot and container nurseries of the Pacific Northwest: epidemiology, management, and effects on outplanting performance. New Forests. 30: 185-202.

Dumroese, R.K.; James, R.L.; Wenny, D.L. 1993. Fusarium root infection of container-grown Douglas-fir: effect on survival and growth of outplanted seedlings and persistence of the pathogen. New Forests. 7: 143–149.

Dumroese, R.K.; James, R.L.; Wenny, D.L. 2000. An assessment of *Cylindrocarpon* on container western white pine seedlings after outplanting. Western Journal of Applied Forestry. 15: 5–7.

Egan, J.M.; Jacobi, W.R.; Negron, J.F.; [et al.]. 2010. Forest thinning and subsequent bark beetle-caused mortality in northeastern California. Forest Ecology and Management. 260: 1832–1842.

Eidson, E.L.; Mock, K.E.; Bentz, B.J. 2017. Mountain pine beetle host selection behavior confirms high resistance in Great Basin bristlecone pine. Forest Ecology and Management. 402: 12–20.

Eidson, E.L.; Mock, K.E.; Bentz, B.J. 2018. Low offspring survival in mountain pine beetle infesting the resistant Great Basin bristlecone pine supports the preference-performance hypothesis. PLOS One. 13(5): e0196732.

Ellsworth, L.M.; Wrobleski, D.W.; Kauffman, J.B.; [et al.]. 2016. Ecosystem resilience is evident 17 years after fire in Wyoming big sagebrush ecosystems. Ecosphere. 7(12): e01618.

Falkowski, M.J.; Evans, J.S.; Naugle, D.E.; [et al.]. 2017. Mapping tree canopy cover in support of proactive prairie grouse conservation in western North America. Rangeland Ecology & Management. 70: 15–24.

Fettig, C.; Borys, R.; Dabney, C. 2010. Effects of fire and fire surrogate treatments on bark beetle-caused tree mortality in the Southern Cascades, California. Forest Science. 56: 60–73.

Fettig, C.J.; McKelvey, S.R. 2010. Bark beetle responses to stand structure and prescribed fire at Blacks Mountain Experimental Forest, California, USA: 5-year data. Fire Ecology. 6: 26–42.

Fleishman, E.; Austin, G.T.; Murphy, D.D. 2001. Biogeography of Great Basin butterflies: revisiting patterns, paradigms, and climate change scenarios. Biological Journal of the Linnean Society. 74: 501–515.

Fleishman, E.; MacNally, R. 2003. Distinguishing between signal and noise in faunal responses to environmental change. Global Ecology and Biogeography. 12: 395–402.

Foley, P.; Roth, T.; Foley, J.; [et al.]. 2017. Rodent-pika parasite spillover in western North America. Journal of Medical Entomology. 54: 1251–1257.

Galbreath, K.E.; Hafner, D.J.; Zamudio, K.R.; [et al.]. 2010. Isolation and introgression in the Intermountain West: contrasting gene genealogies reveal the complex biogeographic history of the American pika (*Ochotona princeps*). Journal of Biogeography. 37: 344–362.

Gibson, D.; Blomberg, E.J.; Atamian, M.T.; [et al.]. 2017. Weather, habitat composition, and female behavior interact to modify offspring survival in greater sage-grouse. Ecological Applications. 27: 168–181.

Gillespie, I.G.; Loik, M.E. 2004. Pulse events in Great Basin Desert shrublands: physiological responses of *Artemisia tridentata* and *Purshia tridentata* seedlings to increased summer precipitation. Journal of Arid Environments. 59: 41–57.

Glenn, E.M., Anthony, R.G., Forsman, E.D., [et al.]. 2011. Local weather, regional climate, and annual survival of the northern spotted owl. The Condor. 113: 159–176.

Goebel, T.; Hockett, B.; Adams, K.D.; [et al.]. 2011. Climate, environment, and humans in North America's Great Basin during the Younger Dryas, 12,900–11,600 calendar years ago. Quaternary International. 242: 479–501.

Gosejohan, M.C.; Weisberg, P.J.; Merriam, K.E. 2017. Hydrologic influences on plant community structure in vernal pools of Northeastern California. Wetlands. 37: 257–268.

Grayson, D.K. 2000. Mammalian responses to Middle Holocene climatic change in the Great Basin of the western United States. Journal of Biogeography. 27: 181–192.

Grayson, D.K. 2005. A brief history of Great Basin pikas. Journal of Biogeography. 32: 2103–2111.

Haubensak, K.; D'Antonio, C.; Wixon, D. 2009. Effects of fire and environmental variables on plant structure and composition in grazed salt desert shrublands of the Great Basin (USA). Journal of Arid Environments. 73: 643–650.

Henry, P.; Henry, A.; Russello, M.A. 2012. Variation in habitat characteristics of American pikas along an elevation gradient at their northern range margin. Northwest Science. 86: 346–350.

Hillman, J.; Swiecki, T.; Bernhardt, E.; [et al.]. 2016. 31 flavors to 50 shades of grey: battling Phytophthoras in native habitats managed by the Santa Clara valley water district. In: Proceedings of the sudden oak death sixth science symposium. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 57.

Holbrook, J.D.; Arkle, R.S.; Rachlow, J.L.; [et al.]. 2016. Occupancy and abundance of predator and prey: implications of the fire-cheatgrass cycle in sagebrush ecosystems. Ecosphere. 7(6): e01307.

Homer, C.G.; Xian, G.; Aldridge, C.L.; [et al.]. 2015. Forecasting sagebrush ecosystem components and greater sage-grouse habitat for 2050: learning from past climate patterns and Landsat imagery to predict the future. Ecological Indicators. 55: 131–145.

Hood, S.M.; Smith, S.L.; Cluck, D.R. 2010. Predicting mortality for five California conifers following wildfire. Forest Ecology and Management. 260: 750–762.

Hornsby, A.D.; Matocq, M.D. 2012. Differential regional response of the bushy-tailed woodrat (*Neotoma cinerea*) to late Quaternary climate change. Journal of Biogeography. 39: 289–305.

Howard, K.; Noble, P. 2018. Hydrological perturbations drive rapid shifts in phytoplankton biodiversity and population dynamics in Butte Lake (Lassen Volcanic National Park, California). Lake and Reservoir Management. 34: 21–41.

Howell, A.H. 1924. Revision of the American pikas. North American Fauna. 47: 1–57.

Huckleberry, G.; Beck, C.; Jones, G.T.; [et al.]. 2001. Terminal Pleistocene/early Holocene environmental change at the sunshine locality, north-central Nevada, USA. Quaternary Research. 55: 303–312.

Hurteau, M.D.; Bradford, J.B.; Fulé, P.Z.; [et al.]. 2014. Climate change, fire management, and ecological services in the southwestern U.S. Forest Ecology and Management. 327: 280–289.

Jardine, A.; Long, J.W. 2014. Synopsis of climate change. In: Long, J.W.; Quinn-Davidson, L.; Skinner, C.N., eds. 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: 71–81.

Jeffress, M.R.; Rodhouse, T.J.; Ray, C.; [et al.]. 2013. The idiosyncrasies of place: geographic variation in the climate-distribution relationships of the American pika. Ecological Applications. 23: 864–878.

Jeffress, M.R.; Van Gunst, K.J.; Millar, C.I. 2017. A surprising discovery of American pika sites in the northwestern Great Basin. Western North American Naturalist. 77: 252–268.

Jezkova, T.; Jaeger, J.R.; Olah-Hemmings, V.; [et al.]. 2016. Range and niche shifts in response to past climate change in the desert horned lizard *Phrynosoma platyrhinos*. Ecography. 39: 437–448.

Jezkova, T.; Oláh-Hemmings, V.; Riddle, B.R. 2011. Niche shifting in response to warming climate after the last glacial maximum: inference from genetic data and niche assessments in the chisel-toothed kangaroo rat (*Dipodomys microps*). Global Change Biology. 17: 3486–3502.

Johnson, R.; Stritch, L.; Olwell, P.; [et al.]. 2010. What are the best seed sources for ecosystem restoration on BLM and USFS lands? Native Plants Journal. 11: 117–131.

Jones, G.M.; Gutiérrez, R.J.; Tempel, D.J.; [et al.]. 2016. Using dynamic occupancy models to inform climate change adaptation strategies for California spotted owls. Journal of Applied Ecology. 53: 895–905.

Jones, T.L.; Schwitalla, A. 2008. Archaeological perspectives on the effects of medieval drought in prehistoric California. Quaternary International. 188: 41–58.

Jung, T.; Orlikowski, L.; Henricot, B.; [et al.]. 2016. Widespread *Phytophthora* infestations in European nurseries put forest, semi-natural and horticultural ecosystems at high risk of Phytophthora diseases. Forest Pathology. 46: 134–163.

Karban, R.; Pezzola, E. 2017. Effects of a multi-year drought on a drought-adapted shrub, *Artemisia tridentata.* Plant Ecology. 218: 547–554.

Knapp, E.E.; Ritchie, M.W. 2016. Response of understory vegetation to salvage logging following a high-severity wildfire. Ecosphere. 7: e01550.

Knapp, P.A. 1996. Cheatgrass (*Bromus tectorum* L) dominance in the Great Basin Desert: history, persistence, and influences to human activities. Global Environmental Change. 6: 37–52.

Knick, S.T.; Connelly, J.W., eds. 2011. Greater sage-grouse: ecology and conservation of a landscape species and its habitats. Studies in Avian Biology 38. Berkeley, CA: University of California Press for Cooper Ornithological Society. 664 p.

Koch, F.H.; Smith, W.D. 2012. A revised sudden oak death risk map to facilitate national surveys. In: Potter, K.M.; Conkling, B.L., eds. Forest health monitoring 2009 national technical report. Gen. Tech. Rep. SRS-167. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station: 109–136.

Koide, R.T.; Ricks, K.D.; Davis, E.R. 2017. Climate and dispersal influence the structure of leaf fungal endophyte communities of *Quercus gambelii* in the eastern Great Basin, USA. Fungal Ecology. 30: 19–28.

Kopp, C.W.; Cleland, E.E. 2015. A range-expanding shrub species alters plant phenological response to experimental warming. PLoS ONE. 10(9): e0139029.

Kormos, P.R.; Marks, D.; Pierson, F.B.; [et al.]. 2017. Ecosystem water availability in juniper versus sagebrush snow-dominated rangelands. Rangeland Ecology and Management. 70: 116–128.

Lachniet, M.; Asmerom, Y.; Polyak, V.; [et al.]. 2017. Arctic cryosphere and Milankovitch forcing of Great Basin paleoclimate. Scientific Reports 7: 12955.

Larrucea, E.S.; Brussard, P.F. 2008. Shift in location of pygmy rabbit (*Brachylagus idahoensis*) habitat in response to changing environments. Journal of Arid Environments. 72: 1636–1643.

Liebhold, A.M.; Brockerhoff, E.G.; Garrett, L.J.; [et al.]. 2012. Live plant imports: the major pathway for forest insect and pathogen invasions of the U.S. Frontiers in Ecology and the Environment. 10: 135–143.

Loik, M.E.; Griffith, A.B.; Alpert, H.; [et al.]. 2015. Impact of intra- versus inter-annual snow depth variation on water relations and photosynthesis for two Great Basin Desert shrubs. Oecologia. 178: 403–414.

Long, J.W.; Quinn-Davidson, L.N.; Skinner, C.N., eds. 2014a. 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.

Long, J.W.; Skinner, C.; Charnley, S.; [et al.]. 2014b. Post-wildfire management. In: Long, J.W.; Quinn-Davidson, L.; Skinner, C.N., eds. 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: 187–220.

Malaney, J.L.; Lackey, C.W.; Beckmann, J.P.; [et al.]. 2018. Natural rewilding of the Great Basin: Genetic consequences of recolonization by black bears (*Ursus americanus)*. Diversity and Distributions. 24: 168–178.

Manning, T.; Hagar, J.C. 2011. Use of nonalpine anthropogenic habitats by American pikas (*Ochotona princeps*) in western Oregon. Western North American Naturalist. 71: 106–112.

Martyn, T.E.; Bradford, J.B.; Schlaepfer, D.R.; [et al.]. 2016. Seed bank and big sagebrush plant community composition in a range margin for big sagebrush. Ecosphere. 7: e01453.

Massey, A.L.; Rickart, E.A.; Rowe, R.J. 2017. Habitat use of the piñon mouse (*Peromyscus truei*) in the Toiyabe Range, central Nevada. Western North American Naturalist. 77: 464-–77.

Mathewson, P.D.; Moyer-Horner, L.; Beever, E.A.; [et al.]. 2017. Mechanistic variables can enhance predictive models of endotherm distributions: the American pika under current, past, and future climates. Global Change Biology. 23: 1048–1064.

McIver, J.; Macke, E. 2014. Short-term butterfly response to sagebrush steppe restoration treatments. Rangeland Ecology and Management. 67: 539–552.

Mensing, S.; Smith, J.; Norman, K.B.; [et al.]. 2008. Extended drought in the Great Basin of western North America in the last two millennia reconstructed from pollen records. Quaternary International. 188: 79–89.

Mensing, S.A.; Sharpe, S.E.; Tunno, I.; [et al.]. 2013. The late Holocene dry period: multiproxy evidence for an extended drought between 2800 and 1850 cal yr BP across the central Great Basin, USA. Quaternary Science Reviews. 78: 266–282.

Merriam, K.; Safford, H.; Sawyer, S. 2013. A summary of current trends and probable future trends in climate and climate-driven processes in the Sierra Cascade Province, including the Lassen, Modoc, and Plumas National Forests. U.S. Department of Agriculture, Forest Service. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprd3820062.pdf (9 Dec. 2019)

Merriam, K.E.; Gosejohan, M.C.; Weisberg, P.J.; [et al.]. 2016. Livestock use has mixed effects on slender orcutt grass in northeastern California vernal pools. Rangeland Ecology and Management. 69: 185–194.

Meyer, S.E. 1994. Germination and establishment ecology of big sagebrush: implications for community restoration. In: Monsen, S.B.; Kitchen, S.G., eds. Ecology and management of annual rangelands. Gen. Tech. Rep. GTR-INT-313. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 244–251.

Migliorini, D.; Ghelardini, L.; Tondini, E.; [et al.]. 2015. The potential of symptomless potted plants for carrying invasive soilborne plant pathogens. Diversity and Distributions. 21: 1218–1229.

Millar, C.I.; Westfall, R.D. 2010. Distribution and climatic relationships of the American pika (*Ochotona princeps*) in the Sierra Nevada and Western Great Basin, USA: periglacial landforms as refugia in warming climates. Arctic, Antarctic, and Alpine Research. 42: 76–88.

Millar, C.I.; Delany, D.L.; Hersey, K.A,; [et al.]. 2018. Distribution, climatic relationships, and status of American pikas (*Ochotona princeps*) in the Great Basin, USA, Arctic, Antarctic, and Alpine Research. 50: e1436296.

Millar, C.I.; Westfall, R.D.; Delany, D.L. 2013. New records of marginal locations for American pika (*Ochotona princeps*) in the western Great Basin. Western North American Naturalist. 73: 457–476.

Millar, C.I.; Westfall, R.D.; Delany, D.L. 2014. Thermal regimes and snowpack relations of periglacial talus slopes, Sierra Nevada, California, USA. Arctic, Antarctic, and Alpine Research. 46: 483–504.

Miller, A.L.; Makowsky, R.A.; Formanowicz, D.R.; [et al.]. 2014. Cryptic genetic diversity and complex phylogeography of the boreal North American scorpion, *Paruroctonus boreus* (Vaejovidae). Molecular Phylogenetics and Evolution. 71: 298–307.

Miller, R.F.; Knick, S.T.; Pyke, D.A.; [et al.]. 2011. Characteristics of sagebrush habitats and limitations to long-term conservation. In: Knick, S.T.; Connelly, J.W., eds. Greater sage-grouse: ecology and conservation of a landscape species and its habitats. Berkeley, CA: University of California Press: 145–184.

Miller, R.F.; Naugle, D.E., Maestas, J.D.; [et al.]. 2017. Special issue: targeted woodland removal to recover at-risk grouse and their sagebrush-steppe and prairie ecosystems. Rangeland Ecology and Management. 70: 1–8.

Minckley, T.A.; Whitlock, C.; Bartlein, P.J. 2007. Vegetation, fire, and climate history of the northwestern Great Basin during the last 14,000 years. Quaternary Science Reviews. 26: 2167–2184.

Møller, A.P.; Nielsen, J.T. 2015. Large increase in nest size linked to climate change: an indicator of life history, senescence and condition. Oecologia. 179: 913–921.

MontBlanc, E.M.; Chambers, J.C.; Brussard, P.F. 2007. Variation in ant populations with elevation, tree cover, and fire in a pinyon-juniper-dominated watershed. Western North American Naturalist. 67: 469–491.

Morelli, T.L.; Smith, A.B.; Kastely, C.R.; [et al.]. 2012. Anthropogenic refugia ameliorate the severe climate-related decline of a montane mammal along its trailing edge. Proceedings of the Royal Society B-Biological Sciences. 279: 4279–4286.

Nisbet, R.A.; Berwick, S.H.; Reed, K.L. 1983. A spatial model of sage grouse habitat quality. In: Lauenroth, W.K.; Skogerboe, G.V.; Flug, M., eds. Analysis of ecological systems state-of-the-art in ecological modelling. Developments in Environmental Modelling, vol. 5. Copenhagen, Denmark: International Society for Ecological Modelling: 267–276.

Nowak, C.L.; Nowak, R.S.; Tausch, R.J; [et al.]. 1994. Tree and shrub dynamics in northwestern Great-Basin woodland and shrub steppe during the late-Pleistocene and Holocene. American Journal of Botany. 81: 265–277.

O'Regan, S.M.; Palen, W.J.; Anderson, S.C. 2014. Climate warming mediates negative impacts of rapid pond drying for three amphibian species. Ecology. 95: 845–855.

Omernik, J.M. 1987. Ecoregions of the conterminous United States. Annals of the Association of American Geographers. 77: 118–125.

Palmer, M.A.; McRoberts, R.E.; Nicholls, T.H. 1988. Sources of inoculum of *Sphaeropsis sapinea* in forest tree nurseries. Phytopathology. 19: 353–371.

Parks, S.A.; Miller, C.; Parisien, M.A.; [et al.]. 2015. Wildland fire deficit and surplus in the western United States, 1984–2012. Ecosphere. 6: 1–13.

Pennington, V.E.; Schlaepfer, D.R.; Beck, J.L.; [et al.]. 2016. Sagebrush, greater sage-grouse, and the occurrence and importance of forbs. Western North American Naturalist. 76: 298–312.

Perfors, T.; Harte, J.; Alter, S.E. 2003. Enhanced growth of sagebrush (*Artemisia tridentata*) in response to manipulated ecosystem warming. Global Change Biology. 9: 736–742.

Poore, R.E.; Lamanna, C.A.; Ebersole, J.J.; [et al.]. 2009. Controls on radial growth of mountain big sagebrush and implications for climate change. Western North American Naturalist. 69: 556–562.

Powers, H.R.J.; Schmidt, R.A.; Snow, G.A. 1981. Current status and management of Fusiform rust on southern pines. Annual Review of Phytopathology. 19: 353–371.

Pyke, C.R. 2005. Assessing climate change impacts on vernal pool ecosystems and endemic branchiopods. Ecosystems. 8: 95–105.

Raumann, C.G.; Cablk, M.E. 2008. Change in the forested and developed landscape of the Lake Tahoe basin, California and Nevada, USA, 1940–2002. Forest Ecology and Management. 255: 3424–3439.

Ray, C.; Beever, E.A.; Rodhouse, T.J. 2016. Distribution of a climate-sensitive species at an interior range margin. Ecosphere. 7: e01379.

Riegel, G.M.; Miller, R.F.; Skinner, C.N.; [et al.]. 2006. Northeastern plateaus bioregion. In: Sugihara, N.; van Wagtendonk, J.; Fites-Kaufmann, J.; [et al.], eds. Fire in California's ecosystems. Berkeley: University of California Press: 225–263.

Rodhouse, T.J.; Hovland, M.; Jeffress, M.R. 2017. Variation in subsurface thermal characteristics of microrefuges used by range core and peripheral populations of the American pika (*Ochotona princeps*). Ecology and Evolution. 7: 1514–1526.

Rooney-Latham, S.; Blomquist, C.L.; Swiecki, T.; [et al.]. 2015. First detection in the U.S.: new plant pathogen, *Phytophthora tentaculata,* in native plant nurseries and restoration sites in California. Native Plants Journal. 16: 23–27.

Roundy, B.A.; Young, K.; Cline, N.; [et al.]. 2014. Piñon-juniper reduction increases soil water availability of the resource growth pool. Rangeland Ecology and Management. 67: 495–505.

Rowe, K.C.; Rowe, K.M.C.; Tingley, M.W.; [et al.]. 2015. Spatially heterogeneous impact of climate change on small mammals of montane California. Proceedings of the Royal Society B-Biological Sciences. 282(1799): 20141857.

Roy, B.A.; Gusewell, S.; Harte, J. 2004. Response of plant pathogens and herbivores to a warming experiment. Ecology. 85: 2570–2581.

Ryan, M.E.; Palen, W.J.; Adams, M.J.; [et al.]. 2014. Amphibians in the climate vice: loss and restoration of resilience of montane wetland ecosystems in the western U.S. Frontiers in Ecology and the Environment. 12: 232–240.

Safford, H.D.; Van de Water, K.M. 2014. Using fire return interval departure (FRID) analysis to map spatial and temporal changes in fire frequency on national forest lands in California. Res. Pap. PSW-RP-266. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 59 p.

Salzer, M.W.; Bunn, A.G.; Graham, N.E.; [et al.]. 2014. Five millennia of paleotemperature from tree-rings in the Great Basin, USA. Climate Dynamics. 42: 1517–1526.

Sankey, J.B.; Germino, M.J.; Sankey, T.T.; [et al.]. 2012. Fire effects on the spatial patterning of soil properties in sagebrush steppe, USA: a meta-analysis. International Journal of Wildland Fire. 21: 545–556.

Schlaepfer, D.R.; Lauenroth, W.K.; Bradford, J.B. 2012a. Ecohydrological niche of sagebrush ecosystems. Ecohydrology. 5: 453–466.

Schlaepfer, D.R.; Lauenroth, W.K.; Bradford, J.B. 2012b. Effects of ecohydrological variables on current and future ranges, local suitability patterns, and model accuracy in big sagebrush. Ecography. 35: 374–384.

Schlaepfer, D.R.; Lauenroth, W.K.; Bradford, J.B. 2014. Modeling regeneration responses of big sagebrush (*Artemisia tridentata*) to abiotic conditions. Ecological Modelling. 286: 66–77.

Schlaepfer, D.R.; Taylor, K.A.; Pennington, V.E.; [et al.]. 2015. Simulated big sagebrush regeneration supports predicted changes at the trailing and leading edges of distribution shifts. Ecosphere. 6(1): 1–31.

Schroeder, M.A.; Young, J.R.; Braun, C.E. 1999. Greater sage-grouse (*Centrocercus urophasianus*), vers. 2.0. In: Poole, A.F.; Gill, F.B., eds. The birds of North America online. Ithaca, NY: Cornell Lab of Ornithology.

Schultheis, A.S.; Booth, J.Y.; Perlmutter, L.R.; [et al.]. 2012. Phylogeography and species biogeography of montane Great Basin stoneflies. Molecular Ecology. 21: 3325–3340.

Schultz, L.D.; Heck, M.P.; Hockman-Wert, D.; [et al.]. 2017. Spatial and temporal variability in the effects of wildfire and drought on thermal habitat for a desert trout. Journal of Arid Environments. 145: 60–68.

Shearer, B.L.; Crane, C.E.; Barrett, S.; [et al.]. 2007. *Phytophthora cinnamomi* invasion, a major threatening process to conservation of flora diversity in the southwest botanical province of Western Australia. Australian Journal of Botany. 55: 225–238.

Siegel, R.B.; Pyle, P.; Thorne, J.H.; [et al.]. 2014. Vulnerability of birds to climate change in California's Sierra Nevada. Avian Conservation and Ecology. 9(1): 7.

Simamora, A.; Paap, T.; Howard, K.; [et al.]. 2017. *Phytophthora* contamination in a nursery and its potential dispersal into the natural environment. Plant Disease. 102(1): 132-139.

Smith, A.T.; Weston, M.L. 1990. Mammalian species, *Ochotona princeps*. American Society of Mammologists. 352: 1–8.

Smith, F.A.; Betancourt, J.L.; Brown, J.H. 1995. Evolution of body-size in the woodrat over the past 25,000 years of climate-change. Science. 270: 2012–2014.

Smith, R.S.J., 1967. Decline of *Fusarium oxysporum* in the roots of *Pinus lambertiana* seedlings transplanted into forest soils. Phytopathology. 57: 1265.

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. 370 p. Vol. 1.

Stanosz, G.R.; Carlson, J.C. 1996. Association of mortality of recently planted seedlings and established saplings in red pine plantations with Sphaeropsis collar rot. Plant Disease. 80: 750–753.

Stavros, E.N.; Abatzoglou, J.T.; McKenzie, D.; [et al.]. 2014. Regional projections of the likelihood of very large wildland fires under a changing climate in the contiguous Western United States. Climatic Change. 126: 455–468.

Stewart, J.A.E.; Perrine, J.D.; Nichols, L.B.; [et al.]. 2015. Revisiting the past to foretell the future: summer temperature and habitat area predict pika extirpations in California. Journal of Biogeography. 42: 880–890.

Still, S.M.; Richardson, B.A. 2015. Projections of contemporary and future climate niche for Wyoming big sagebrush (*Artemisia tridentata* subsp. *wyomingensis*): a guide for restoration. Natural Areas Journal. 35: 30–43.

Stokstad, E. 2004. Nurseries may have shipped sudden oak death pathogen nationwide. Science. 303: 1959.

Taylor, K.; Brummer, T.; Rew, L.J.; [et al.]. 2014. *Bromus tectorum* response to fire varies with climate conditions. Ecosystems. 17: 960–973.

Taylor, R.L.; Tack, J.D.; Naugle, D.E.; [et al.]. 2013. Combined effects of energy development and disease on greater sage-grouse. PLoS ONE. 8(8): e71256.

Terry, R.C.; Guerre, M.E.; Taylor, D.S. 2017. How specialized is a diet specialist? Niche flexibility and local persistence through time of the chisel-toothed kangaroo rat. Functional Ecology. 31: 1921–1932.

Tornberg, R.; Liuska, L.; Rytkönen, S.; [et al.]. 2014. Diet shift induced rapid evolution of size and function in a predatory bird. Oecologia. 176: 781–788.

Varner, J.; Lambert, M.S.; Horns, J.J.; [et al.]. 2015. Too hot to trot? Evaluating the effects of wildfire on patterns of occupancy and abundance for a climate-sensitive habitat specialist. International Journal of Wildland Fire. 24: 921–932.

Wade, C.E.; Loik, M.E. 2017. A spring rainfall pulse causes greater in situ photosynthetic upregulation for *Bromus tectorum* compared to co-occurring native herbaceous species. Environmental and Experimental Botany. 143: 51–58.

Wahl, D.; Starratt, S.; Anderson, L.; [et al.]. 2015. Holocene environmental changes inferred from biological and sedimentological proxies in a high elevation Great Basin lake in the northern Ruby Mountains, Nevada, USA. Quaternary International. 387: 87–98.

Walker, B.L.; Naugle, D.E.; Doherty, K.E.; [et al.]. 2007. West Nile virus and greater sage-grouse: estimating infection rate in a wild bird population. Avian Diseases. 51: 691–696.

Warren, D.R.; Dunham, J.B.; Hockman-Wert, D. 2014. Geographic variability in elevation and topographic constraints on the distribution of native and nonnative trout in the Great Basin. Transactions of the American Fisheries Society. 143: 205–218.

Wilkening, J.L.; Ray, C.; Beever, E.A.; [et al.]. 2011. Modeling contemporary range retraction in Great Basin pikas (*Ochotona princeps*) using data on microclimate and microhabitat. Quaternary International. 235: 77–88.

Yang, J.; Weisberg, P.J.; Shinneman, D.J.; [et al.]. 2015. Fire modulates climate change response of simulated aspen distribution across topoclimatic gradients in a semi-arid montane landscape. Landscape Ecology. 30: 1055–1073.

Young, J.A.; Evans, R.A. 1989. Dispersal and germination of big sagebrush (*Artemisia tridentata*) seeds. Weed Science. 37: 201–206.

Zou, L.; Miller, S.N.; Schmidtmann, E.T. 2006. Mosquito larval habitat mapping using remote sensing and GIS: implications of coalbed methane development and West Nile virus. Journal of Medical Entomology. 43: 1034–1041.

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